Universidade de Aveiro Departamento de Ambiente e Ordenamento
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Dissertação apresentada à Universidade de Aveiro para cumprimento dos requisitos necessários à obtenção do grau de Doutor em Ciências e Engenharia do Ambiente, realizada sob a orientação científica do Doutor Luís Manuel Guerreiro Alves Arroja, Professor Associado com Agregação do Departamento de Ambiente e Ordenamento da Universidade de Aveiro e co-orientação do Doutor Bradley George Ridoutt, Investigador Principal do Commonwealth Scientific and Industrial Research Organisation (CSIRO), e da Doutora Ana Cláudia Relvas Vieira Dias, Equiparada a Investigadora Auxiliar do Departamento de Ambiente e Ordenamento da Universidade de Aveiro.

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Para vocês … que ‘caminham’ sempre ao meu lado ….

Para ti, amor meu…

o júri / the jury

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palavras-chave

Algas, macrófitas e macroinvertebrados; Avaliação de Ciclo de Vida (ACV); *Eucalyptus globulus*; erosão de solo; fluxos de água verde; modelação de impactes *endpoint*; pegada de água; uso de água doce

resumo

As perturbações no equilíbrio dos ecossistemas devido ao aumento do uso de água doce, do aumento da sua poluição, e da erosão do solo pela água, são temas emergentes e de elevada significância na avaliação da sustentabilidade.

Na presente tese é efetuada uma revisão de literatura dos métodos desenvolvidos para contabilizar e avaliar os impactes do uso de água doce, numa perspetiva de ciclo de vida. O método da pegada de água desenvolvido pela *Water Footprint Network*, e os vários métodos desenvolvidos com base na metodologia de Avaliação de Ciclo de Vida (ACV) foram analisados. Os impactes decorrentes do uso de água para a produção do vinho verde branco produzido em Portugal, foram contabilizados e avaliados, por aplicação de alguns destes métodos de ACV.

A relevância da água verde na regulação dos serviços do ecossistema tem sido 'esquecida' devido à ausência de um método capaz de avaliar os impactes ambientais decorrentes de alterações dos fluxos de água verde. Para superar esta lacuna, na presente tese é apresentado um método de Avaliação de impacte do Ciclo de (AICV) *midpoint* que permite uma avaliação espacialmente diferenciada dos impactes decorrentes das alterações dos fluxos de água verde que retornam à atmosfera em resultado de atividades de uso de solo. Este método permite também uma avaliação da redução de produção de água azul devido a reduções no escoamento superficial. A aplicabilidade deste método é demonstrada num caso de estudo de povoamentos de *Eucalyptus globulus* localizados em Portugal, os quais dependem fortemente da precipitação local.

A erosão do solo pela água afeta os ecossistemas aquáticos, nomeadamente quando os sólidos suspensos (SS) atingem os rios. Na presente tese foi desenvolvida uma abordagem para estabelecer inventários de SS espacialmente diferenciados, e um método de AICV *endpoint* que permite obter fatores de caracterização específicos para avaliar os impactes ambientais diretos dos SS em algas, macrófitas e macroinvertebrados. A aplicabilidade da abordagem de inventário e do método *endpoint* foi demonstrada num caso de estudo de povoamentos de *E. globulus* localizados em Portugal.

Tanto os impactes associados aos fluxos de água verde como os impactes relativos aos SS variam significativamente em função da localização do sistema de uso de solo em análise, pelo que se conclui que a inclusão da variabilidade espacial deve ser considerada em métodos de ACV.

keywords

Algae, macrophytes and macroinvertebrates; *Eucalyptus. globulus*; fate and effect modelling; green water flows; freshwater use; Life Cycle Assessment (LCA); topsoil erosion; water footprint

abstract

Perturbation of natural ecosystems, namely by increasing freshwater use and its degradative use, as well as topsoil erosion by water of land-use production systems, have been emerging as topics of high environmental concern.

Freshwater use has become a focus of attention in the last few years for all stakeholders involved in the production of goods, mainly agro-industrial and forest-based products, which are freshwater-intensive consumers, requiring large inputs of green and blue water.

This thesis presents a global review on the available Water Footprint Assessment and Life Cycle Assessment (LCA)-based methods for measuring and assessing the environmental relevance of freshwater resources use, based on a life cycle perspective. Using some of the available midpoint LCA-based methods, the freshwater use-related impacts of a Portuguese wine (white 'vinho verde') were assessed.

However, the relevance of environmental green water has been neglected because of the absence of a comprehensive impact assessment method associated with green water flows. To overcome this constraint, this thesis helps to improve and enhance the LCA-based methods by providing a midpoint and spatially explicit Life Cycle Impact Assessment (LCIA) method for assessing impacts on terrestrial green water flow and addressing reductions in surface blue water production caused by reductions in surface runoff due to land-use production systems. The applicability of the proposed method is illustrated by a case study on *Eucalyptus globulus* conducted in Portugal, as the growth of short rotation forestry is largely dependent on local precipitation.

Topsoil erosion by water has been characterised as one of the most upsetting problems for rivers. Because of this, this thesis also focuses on the ecosystem impacts caused by suspended solids (SS) from topsoil erosion that reach freshwater systems. A framework to conduct a spatially distributed SS delivery to freshwater streams and a fate and effect LCIA method to derive site-specific characterisation factors (CFs) for endpoint damage on aquatic ecosystem diversity, namely on algae, macrophyte, and macroinvertebrates organisms, were developed. The applicability of this framework, combined with the derived site-specific CFs, is shown by conducting a case study on *E. globulus* stands located in Portugal as an example of a land use based system.

A spatially explicit LCA assessment was shown to be necessary, since the impacts associated with both green water flows and SS vary greatly as a function of spatial location.

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List of publications

This thesis is based on the following papers:

1. Quinteiro P, Dias AC, Ridoutt BG, Arroja L (2015) Measuring and assessing freshwater-use related impacts from a life cycle perspective: a global review. J Clean Prod. *Submitted*

2. Quinteiro P, Dias AC, Pina L, Neto B, Ridoutt BG, Arroja L (2014) Addressing the freshwater use of a Portuguese wine ('vinho verde') using different LCA methods. J Clean Prod 68:46-55.

3. Quinteiro P, Dias AC, Silva M, Ridoutt BG, Arroja L (2015) A contribution to the environmental impact assessment of green water flows. J Clean Prod 93:318-329.

4. Quinteiro P, Dias AC, Ridoutt BG, Arroja L (2014) A framework for modelling the transport and deposition of eroded particles towards water systems in a Life Cycle Inventory. Int J Life Cycle Assess 19(6):1200-1213.

5. Quinteiro P, Dias AC, Araújo A, Pestana JLT, Ridoutt BG, Arroja L (2015) Suspended solids in freshwater systems: characterisation model describing potential impacts on aquatic biota. Int J Life Cycle Assess 20(9):1232-1242.

6. Quinteiro P, Van de Broek M, Dias AC, Ridoutt BG, Govers G, Arroja L (2015) Life cycle impacts of suspended solids on aquatic biota from forest systems. Int J Life Cycle Assess. *Submitted*

Abbreviations

A	Average eroded soil
BIER	Basin internal evaporation recycling ratio
C	Crop management parameter
CF	Characterisation factor
CF_{TGWI}	Characterisation factor for the assessment of TGWI
CF_{RBWP}	Characterisation factor for the assessment of RBWP
C_{SS}	Concentration of SS
DEM	Digital elevation model
EF	Effect factor
EP	Eutrophication
ESM	Ecological Scarcity Model
EWR	Environmental water requirement
ET	Evapotranspiration
ET_{act}	Total green ET of actual land-use production system
$ET_{act,eff}$	Effective ET of actual land-use production system
$ET_{EWR,\ eff}$	Effective threshold ET level of the land-use production system that guarantees the EWR
ET_{PNV}	Total green ET of the PNV
$ET_{PNV,eff}$	Effective ET of the alternative reference land use
ET_{pot}	ET_{PNV} under non-water limited conditions
FAO	United Nations Food and Agriculture Organization
FD	Freshwater depletion
FE	Freshwater aquatic ecotoxicity
FF	Fate factors
FEI	Freshwater ecosystem impacts
FU	Functional unit
GIS	Geographic information systems
GWSI	Green water scarcity index
HCx	Hazardous concentrations affecting x-th percentile of species
K	Soil erodibility parameter

k_{till}	Tillage transport coefficient
$K_{c,\ ini}$	Crop coefficient for the initial stage of growing
$K_{c,\ mid}$	Crop coefficient for the mid-season stage of growing
ktc	Transport capacity coefficient
ktc_{min}	Transport capacity coefficient for non-erodible land surfaces
ktc_{max}	Transport capacity coefficient for arable land surfaces
$K_{c,\ end}$	Crop coefficient for the end stage of growing
LS_{2D}	Two-dimensional slope-length parameter
LAI	Leaf area index
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LC_{50}	Median lethal concentrations
MAI	Mean annual wood volume growth increment
MAR	Long-term mean annual runoff
ME	Marine aquatic ecotoxicity
MF	Modified Fournier index
NS	Nash-Sutcliffe model efficiency statistic
NGW_{eff}	Effective net green water flow
NOECs	No-observed-effect concentrations
NUTS	Nomenclature of territorial units for statistics
PAF	Potentially affected fraction
PESERA	Pan European Soil Erosion Risk Assessment
PDF	Potentially disappeared fraction
PNV	Potential natural vegetation
P	Precipitation
P	Phosphorous
P	Support-practice parameter to reduce runoff and soil erosion
Pr	Effective precipitation
$Q_{s,t}$	Rate of net downslope soil transport per tillage translocation
R	Rainfall-runoff erosivity parameter
RBWP	Reductions in surface blue water production

rt-ET_{act}	Total mean annual ET of *E. globulus* stand for each production region
rt-$ET_{act, eff}$	Total mean annual effective ET of *E. globulus* stand for each production region
rt-ET_{PNV}	Total mean annual ET of PNV for each production region
RUSLE	Revised Universal Soil Loss Equation
s-ET_{act}	Mean annual ET of each *E. globulus* stand
s-$ET_{act, eff}$	Mean annual effective ET for each *E. globulus* stand
s-$ET_{PNV, eff}$	Mean annual effective ET of PNV for each *E. globulus* stand
Sg	Local slope gradient
SS	Suspended solids
SSD	Species sensitivity distribution
SSRL	Soil static reserve life
SRTM-DEM	Shuttle radar topography mission – digital elevation model
Tc	Runoff transport capacity
TGWI	Impacts on terrestrial green water flows
USLE	Universal Soil Loss Equation
WaTEM/SEDEM	Water and Tillage Erosion Model – Sediment Delivery Model
WFA	Water Footprint Assessment
WFN	Water Footprint Network
WFSA	Water Footprint Sustainability Assessment
WRPC	Water resources per capita
WSI	Water scarcity index
WSI_{nd}	Water scarcity indicator
WU	Freshwater use
WUPR	Water use per resource
x_{EWR}	EWR expressed as a fraction of long-term mean annual actual runoff in a river
3-PG model	Physiological Principles Predicting Growth model

CHAPTER 1

Chapter 1: Introduction

1. General introduction

'Water is the driving force of all nature' Leonardo Da Vinci (1452-1519) claimed.

Observed human-induced climate changes have been changing precipitation patterns or melting snow in many regions, affecting the quantity and quality of freshwater resources (IPCC 2014). In many regions, human activities have been increasing freshwater stress (due to freshwater appropriation and quality degradation), leading to competition for its use and resulting in a loss of freshwater functionality for downstream users (Bogardi et al. 2012; Falkenmark and Rockström 2004; Peters and Meybeck 2000). In this context, several methods have been developed to quantify the consumptive use of freshwater and its degradative use from a life cycle perspective, ensuring freshwater resources management and fair freshwater supply to human livelihoods and ecosystems. Consumptive use refers to freshwater that evaporates, is transpired by plants, embodied into products, consumed by humans or livestock, or discharged into different catchments or seas (Bayart et al. 2010; Hoekstra et al. 2011). Degradative freshwater use refers to a negative change in freshwater quality during its use.

Two parallel developments have been emerging:

- Water Footprint Assessment (WFA), which presents the Water Footprint as a comprehensive spatiotemporal indicator of freshwater appropriation and freshwater management through the supply chain (Hoekstra et al. 2011; Hoekstra and Chapagain 2008).
- Consideration of the freshwater-use related impacts within the Life Cycle Assessment (LCA) methodology (Kounina et al. 2013). LCA models the cause-effect chain from freshwater use to impacts on humans and ecosystems, with characterisation factors (CFs) along that pathway at the midpoint or endpoint level (Berger et al. 2012; Boulay et al. 2015; Kounina et al. 2013; Milà i Canals et al. 2010).

Three freshwater components have been differentiated (Hoekstra et al. 2011; Kounina et al. 2013):

- blue water – includes fresh surface and groundwater; i.e. the freshwater in lakes, rivers and aquifers;
- green water – includes rainfall on land that does not run off or recharge the groundwater, but is stored in the soil or temporarily stays on the top of the soil or vegetation;

- grey water – denotes degradative freshwater use, being defined by the WFN 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.

Although the WFA and LCA-based methods agree on considering both consumptive and degradative freshwater use, the WFA method focuses more on the management and sustainability of freshwater resources through the supply chain, mainly using freshwater use indicators in the inventory phase, while LCA focuses on the potential environmental impact assessment. The impact assessment phase in the WFA method, which focuses on environmental sustainability, economic efficiency, and social equity of freshwater use and allocation, is still in an earlier stage of development.

Over the last 5 years, consideration of freshwater use in LCA has progressed rapidly, resulting in a complex set of methods for addressing different freshwater types and sources, pathways and characterisation models at midpoint and endpoint levels, and with different spatial and temporal scales (Kounina et al. 2013; Tendall et al. 2014). Studies have been carried out on freshwater abstraction and human appropriation (e.g. Boulay et al. 2011; Milà i Canals et al. 2009; Núñez et al. 2012; Pfister et al. 2009; Quinteiro et al. 2014; Ridoutt et al. 2010), the potential impacts of blue water use on ecosystems (e.g. Hanafiah et al. 2011; Pfister et al. 2009; Tendall et al. 2014; Van Zelm et al. 2011; Verones et al. 2013) and on degradative freshwater use related to the discharge of eutrophying, acidifying and ecotoxic compounds into freshwater systems (e.g. Azevedo et al. 2013; Goedkoop et al. 2013; Helmes et al. 2012; Knuuttila 2004; Struijs et al. 2011).

Despite the crucial relevance of green water for the long-term sustenance of terrestrial ecosystem services, a LCA-based method focusing on the land use impacts on the green water flow has been, so far, disregarding within LCA. Green water flow refers to the portions of green water used by soil and vegetation that is evaporated or transpired (Quinteiro et al. 2015).

Furthermore, the current climate change trends in combination with other freshwater stressors such as population growth and land transformation and occupation could lead to irreversible disturbances of terrestrial and aquatic ecosystems and affect human health (Kundzewicz et al. 2008; McDonald et al. 2011; Woodward et al. 2010). Land transformation (also called land use changes) refers to the process of changing the flora, fauna, and soil from its potential natural state to an altered state, for example, when a grassland is ploughed to establish arable fields (Koellner et al. 2013; Milà i Canals et al. 2007; Weidema and Lindeijer 2001). Land occupation refers to the maintenance of the flora, fauna and soil hindering the regrowth of grasslands or other potential natural vegetation (Weidema and Lindeijer 2001), leading to a substantial decrease of the available land. The increase in frequency and intensity of seasonal heavy precipitation events (IPCC 2014) in combination with land transformation and occupation involved in agriculture and forestry (Grimm et al. 2002; Pimentel et al. 1995), have been increasing the susceptibility of land use production systems to soil erosion by water.

Topsoil erosion is a natural, inevitable and complex process that varies depending on local soil characteristics, ground slope, vegetation cover and climatic conditions. Water is one of the major causes of soil erosion (Jones et al. 2012a), and the average soil erosion by water in Europe is about 2.8 $t.ha^{-1}.yr^{-1}$ (Jones et al. 2012a,b), affecting 60 % of the total European land area excluding Russia. The loss of topsoil can lead to a loss of productive capability of land, and this effect in the context of ecosystem services has begun to be considered recently in LCA (Núñez et al. 2013; Saad et al. 2011). However, displaced soil is itself a source of potential environmental harm, especially when it reaches water systems, which has been, so far, mostly neglected within LCA methodology.

The displaced soil reaches freshwater systems as suspended solids (SS). These SS contribute to the sustainability of the aquatic biodiversity due to SS-associated nutrient transport. However, high concentrations of SS, particularly clays and silts, can be significant stressors to the ecological integrity of the aquatic ecosystems, degrading the water quality and directly affecting the aquatic biota, namely primary producers, such as algae and macrophytes, macroinvertebrates and fish (Angermeier et al. 2004; Bilotta and Brazier 2008; Collins et al. 2011; Jones et al. 2012c; Kasai et al. 2005).

1.1. Objectives of the study

The main objectives of this thesis are the improvement and enhancement of the LCA methodology regarding freshwater use and its quality degradation impacts through the development of Life Cycle Impact Assessments (LCIA) methods, in particular to assess the potential environmental impacts related to changes in green water flows, at midpoint level, and the SS impacts from land use systems on aquatic environments, at endpoint level. Regarding green water flows, a method is proposed for incorporating green water consumption in the impact category "water scarcity". Concerning SS impacts, a method is provided for assessing the environmental impacts arising from SS due to topsoil erosion by water on macroinvertebrates, algae and macrophyte organisms, which was until now never modelled in the LCA methodology.

The impacts of freshwater use and SS that reach freshwater streams vary greatly as a function of location; therefore, spatial differentiation was considered in the operational methods developed for impact assessment, both in the Life Cycle Inventory (LCI) and LCIA phases. The applicability of these methods was tested by employing case studies on *Eucalyptus globulus* stands, located in Portugal.

E. globulus is one of the dominant species in the Portuguese forest, accounting for approximately 26 % of the total forest area in Portugal, covering approximately 812 thousand hectares (ICNF 2013). Although the risk of erosion is lower for forested areas than in agricultural areas (FAO 2013), almost one-third of the Portuguese territory, including high density *E. globulus* forested areas, is at high risk of erosion by water (Grimm et al. 2002), such as after forest fires in Portugal.

Spatially explicit LCA-based methods to address the impact of green water flows and SS on aquatic biota is of paramount importance, due to: (1) the spatial heterogeneity of land cover, climatic conditions and soil moisture; and (2) the spatial heterogeneity of soil erodibility, topography, rainfall erosivity and the magnitude of the SS effects for different macroinvertebrates, algae and macrophyte organisms at the catchment scale.

1.2. General overview of the thesis

The present thesis contains modified versions of published or submitted peer-reviewed papers from Science Citation Index (SCI) journals.

The paper modifications concern the harmonization of (1) literature references, because the papers were published or submitted to different journals, using different reference styles; (2) the units of measurement according to both Directive 2009/3/CE of the European Parliament and of the Council of 11 March 2009 and Portuguese Decree-law 128/2010 of 3 December 2010, and (3) document formatting to make the text easier to read.

The thesis is structured in four chapters including the present one as Chapter 1. In this chapter, the framework of the work presented in this thesis was developed and its objectives were presented.

Chapter 2 discusses three papers related to freshwater use and its impacts. The first one is a comprehensive review of the cause-effect chains covered by both WFA methodology and LCA-based methods, as well as the most recent achievements in these fields. Special attention was focused on the potential environmental impacts of green water flow and on the refinement of the impact characterisation factors, considering a finer spatial and temporal resolutions. The second paper applied several LCA-based methods to quantify freshwater use in the production of a Portuguese white ‘vinho verde,’ considering both viticulture and wine production stages, disaggregated into foreground and background sub-systems. This work relies on primary data provided by Aveleda S.A. The third paper proposes a species and spatially explicit LCIA method to assess the potential environmental impacts on terrestrial green water flow (TGWI) and addresses reductions in surface blue water production (RBWP) caused by reductions in surface runoff to land-use production systems. Both TGWI and RBWP are analysed, taking into account the interactions between green freshwater use and atmosphere, and between green freshwater use and soil interfaces. Furthermore, the applicability of the method proposed is illustrated by applying a case study on *E. globulus* stands in Portugal. This work relies on primary data provided by the Portuguese Forest and Paper Research Institute (RAIZ).

Chapter 3 encompasses three papers, associated with the potential environmental impacts of SS caused by topsoil erosion of land-use production systems on aquatic invertebrate, algae and macrophyte organisms. The first paper is a framework for modelling the spatial distribution of eroded soil and the transport of SS through the landscape towards the surface freshwater streams in an LCI. In order to understand how topsoil erosion issues have been considered in LCA, an overview of existing methods addressing topsoil erosion was conducted. To support a spatially explicit LCI, modelling of topsoil erosion and the SS transport method is proposed, suggesting the most adequate topsoil erosion model that fits the LCI purpose of the WaTEM/SEDEM model. The second paper presents the LCIA method developed to derive spatially differentiated endpoint characterisation factors (CFs), based on a cause and effect model, addressing the direct effects of SS on the potential loss of aquatic invertebrate, algae and macrophyte organisms. The applicability of the proposed method is shown by deriving endpoint CFs for 22 different European river sections. The third paper illustrates the applicability of the LCI framework presented in the first paper of this chapter to conduct a spatially distributed SS delivery to freshwater streams using the

WaTEM/SEDEM model combined with the method presented in the second paper of this chapter to derive spatially differentiated CFs for endpoint damage on aquatic ecosystem diversity. Once again, a case study on *E. globulus* stands in Portugal was selected to show the relevance of SS delivery to freshwater streams, providing a more comprehensive assessment of the SS impact from land-use production systems on aquatic environments. This work relies on primary data provided by RAIZ.

Finally, Chapter 4 presents the overall conclusions and some suggestions for future work.

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CHAPTER 2

Chapter 2: Freshwater use and its potential environmental impacts due to the production of agricultural and forest products

Introduction

This chapter addresses the freshwater use-related impacts. Freshwater use refers to consumptive freshwater use (freshwater subtracted that is evaporated, transpired by plants, embodied into the products, consumed by humans or livestock or discharged into different catchments or sea).

A comprehensive overview of the cause-effect chains covered by both WFA and LCA-based methods, and the most recent achievements, with a special focus on the potential environmental impacts of green water flow and on the refinement of the impact characterisation factors, considering a finer spatial and temporal resolutions, is provided in Chapter 2.1.

In accordance with the different LCA-based methods presented in the review, a case study was conducted to assess the impacts of freshwater use associated with the production of white wine (Chapter 2.2). The study considered both viticulture and wine production stages, using primary data provided by Aveleda SA, a well-known Portuguese winemaker.

Chapter 2.3 presents the newly developed method to assess the potential impacts of green water flow. For the purpose of conducting a realistic case study on *Eucalyptus globulus* stands located in Portugal, a collaboration with RAIZ has been established. Valuable information related to the 3-PG model operation, supporting by the models, and primary data related to the soil and stands were provided by RAIZ.

2.1. Measuring and assessing freshwater-use related impacts from a life cycle perspective: a global review

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Abstract

The critical role of freshwater and its related impacts on food production, ecosystem maintenance and resource depletion is globally recognised. Urban and industrial water-use supplies are already competing with agriculture in many parts of the world.

The actual freshwater scarcity in some regions, the increasing deterioration of freshwater quality, and the global forecast of freshwater shortages, have led to the development of life cycle-based methods for addressing the freshwater use and the related impact. In this way, robust methods have been developed to provide tools for freshwater resources management and fair freshwater supply to human livelihoods and ecosystems.

This work presents a global review on the available methods for measuring and addressing the freshwater-use related impacts from a life cycle perspective. The current state of development of both Water Footprint Assessment and Life Cycle Assessment-based methods is presented and discussed. Despite the significant methodological improvements that have been developed for addressing the freshwater-use related impacts, there are still some issues that remain partially unsolved, and need further research. These issues are presented and discussed in this study, namely: (1) accounting and assessing the potential environmental impacts of green water flows; (2) temporal and spatial variation to establish explicit characterisation factors (CFs), considering the local environmental uniqueness; (3) adequate connection between inventory flows and spatio-temporal explicit CFs. Some suggestions for further research to ameliorate these methodological issues are also presented.

Keywords: freshwater use, green water flows, Life Cycle Impact Assessment, spatial variation, temporal variation, water footprint

1. Introduction

Local and global perturbations in freshwater resources, such as the increase of freshwater scarcity and stress, and changes of soil moisture due to land use changes, clearly represent a major environmental and societal worldwide concern (ISO 2014; Pfister et al. 2011a). Freshwater scarcity refers to the pressure on the resource from a quantity perspective, being calculated as a ratio of annual freshwater withdrawal to hydrological availability in a specific catchment (UN/WWAP 2003; WMO 1997). Freshwater stress considers several aspects related to freshwater, such as: freshwater scarcity, freshwater quality degradation, environmental flows and accessibility of freshwater (Vorosmarty et al. 2005). The growing demand of freshwater from human activities (e.g. agriculture, industry, household usage) on the one hand, and current climate change trends (temperature rise, changes of rainfall patterns) on the other hand, have led to the depletion of freshwater resources and declining of freshwater quality in many regions (Bogardi et al. 2012; Falkenmark and Rockström 2004; Peters and Meybeck 2000). Temperature rise, increase of rainfall intensity, longer dry spells, increased frequency of floods and droughts have also been disturbing the distribution, availability and quality of freshwater (Collins et al. 2011; IPCC 2007; Jiménez et al. 2014; Lehner et al. 2006; Milly et al. 2005).

Freshwater use considers both blue water (surface and groundwater, i.e. water in lakes, dams, rivers and aquifers) and green water (precipitation on land that does not run off or recharge the groundwater but is stored in the soil, or temporarily stays on the top of the soil or vegetation (Hoekstra et al. 2011)). Consumptive freshwater use refers to freshwater abstraction that is evaporated, transpired by plants, embodied into the product, consumed by humans or livestock use, or discharged into different catchments or sea (Bayart et al. 2010; Hoekstra et al. 2011).

Human activities are currently appropriating more than 61 % of the proposed planetary boundary for consumptive freshwater use of around 2,800 $km^3.yr^{-1}$ of available surface and groundwater (blue water) as calculated by Gerten et al. (2013). The planetary boundary for consumptive freshwater use refers to the determined value of water that can be appropriated within a safe operating space for humanity with respect to the functioning of the Earth's system, i.e. without reaching dangerous thresholds that can lead to irreversible damages on water-related ecosystem services (Alcamo et al. 2007; Rockström et al. 2009; Shen et al. 2008; Steffen et al. 2015). According to Ridoutt and Pfister (2010a) the majority of global freshwater withdrawals currently take place in basins already experiencing high water scarcity, disturbing local and global ecosystems to a critical level.

Regarding green water, the global volume of green water consumed for crop production has been estimated as 6,684 $km^3.yr^{-1}$ (Hoekstra and Mekonnen 2012). An increase of 2,500 $km^3.yr^{-1}$ of green water flow (green water used by soil and vegetation that is evaporated or transpired) in developing countries is deemed necessary to halve hunger by 2030 (Rockström et al. 2007), but a planetary boundary for green water has not yet been set (Gerten et al. 2013).

Freshwater stress and its implications for present and future human welfare and the natural environment awakened the need to develop methods to measure and assess the potential environmental impacts of both consumptive and degradative freshwater use, ensuring the sustainable allocation of freshwater resources among competing users. Degradative freshwater use refers to a negative change in freshwater quality during its use (ISO 2014). In this sense, Water Footprint (WF) has become a focus of attention in the last 15 years in academia, organisations, governments and all stakeholders involved in the production of goods, mainly agro-forest based products, which are freshwater-intensive consumers, requiring large inputs of green and blue water (Chapagain and Hoekstra 2008; Page et al. 2011; Pfister et al. 2009).

The Water Footprint Network (WFN) has been developing a method for Water Footprint Assessment (WFA) that focuses on the management and sustainability of freshwater resources through the supply chain (Hoekstra et al. 2011). In turn, Life Cycle Assessment (LCA) models consider the cause-effect chain from freshwater use and the impact on humans and ecosystems, with characterisation factors (CFs) along that pathway at midpoint or endpoint levels (Berger et al. 2012; Boulay et al. 2015a; Kounina et al. 2013; Milà i Canals et al. 2010). The midpoint level or problem-oriented approach translates potential environmental impacts of freshwater use in categories such as freshwater scarcity and freshwater eutrophication, prior to the endpoint level, at which CFs can be derived to reflect the relative importance of consumptive and degradative freshwater use (Bare et al 2000, Goedkoop et al. 2013; Jolliet et al. 2004). The endpoint level damage-oriented approach translates environmental impacts in the cause-effect chain to a given area of protection, i.e. human health, ecosystem quality or resources (Bare et al. 2000, Goedkoop et al. 2013).

Although WFA and LCA-based methods agree on considering the freshwater consumptive use and addressing water degradation (Boulay et al. 2013), there are conceptual differences (Berger and Finkbeiner 2013; Boulay et al. 2013; Jefferies et al. 2012; Quinteiro et al. 2014). One of the main differences is related to green water use accounting, as WFA method (Hoekstra et al. 2011) disclaims the consideration of the net green water use concept, which is supported by the LCA methodology (Núñez et al. 2013, 2012; Quinteiro et al. 2014; Ridoutt et al. 2010). The net green water use compares the green water use from an actual land-use production system to a potential natural reference land use. Green water use refers to precipitation on land that does not runoff or recharges the groundwater but is stored in the soil or temporarily stays on the top of the soil or vegetation, as well as rainwater incorporated into harvested crops or wood (Hoekstra et al. 2011).

Another difference is related to the degradative freshwater use. While WFA method use the critical dilution volume approach, disregarding the residence time of pollutants in the environment (Pfister and Ridoutt 2014), LCA-based methods address the environmental mechanisms of the pollutants emitted to freshwater systems and soil, and their damages to terrestrial and aquatic ecosystems (Goedkoop et al. 2013). The critical dilution volume approach characterises each emission in terms of the volume of freshwater required to dilute it, so that acceptable water quality standards of the receiving water body are met (Hoekstra et al. 2011). These water quality standards depend on the pollutant and receiving freshwater body under study. The standards can be available at national or state level, or defined by legislation (e.g. European water framework directive) (Hoekstra et al. 2011).

Within LCA, there is a set of different methods addressing different, complementary or sometimes the same impact pathway, based on different modelling approaches and assumptions (Boulay et al. 2014, 2011a; Milà i Canals et al. 2009; Motoshita et al. 2010; Núñez et al. 2013; Peters et al. 2010; Pfister et al. 2009). Also, with the purpose of communicating the freshwater use impacts as a single score, some stand-alone environmental methods based on LCA have emerged (Bayart et al. 2014; Ridoutt and Pfister 2013).

The need to develop a consensual method to assess, compare and disclose the environmental performance of products and organisations regarding consumptive and degradative freshwater use, led to the development of the international standard ISO 14046 (ISO 2014), and to the WULCA group founded under the auspices of the Life Cycle Initiative of the United Nations Environment Programme (UNEP)/Society of Environmental Toxicology and Chemistry (SETAC) (WULCA 2014). The WULCA group is currently working on developing consensual-based indicators covering human health and ecosystem quality, and building consensus on a generic water scarcity indicator to be used in LCA (Boulay et al. 2015b; WULCA 2015).

Two previous reviews, evaluating a set of LCA-based methods (Berger and Finkbeiner 2010) and both LCA-based methods and WFA method (Kounina et al. 2013) have been published. Berger and Finkbeiner (2010) evaluated the scope, input data and requirements of the available methods to enable accounting and impact assessment of freshwater use, whereas Kounina et al. (2013) conducted a deep analysis, studying the differences and similarities in modelling choices using a comprehensive set of criteria based on the International Reference Life Cycle Data System (ILCD) (JRC-IES 2010). However, the WF field is still an emerging field that has been evolving rapidly, requiring a regular update to include new developments (Kounina et al. 2013).

The main objective of this work is to provide a comprehensive global review on the available methodologies and methods for measuring and addressing the freshwater-use related impacts from a life cycle perspective. The current state of development of both WFA and LCA-based methods is presented and discussed. Also the issues that remain partially unsolved are analysed, namely: (1) accounting and assessing the potential environmental impacts of green water flows; (2) temporal and spatial variation to establish explicit CFs, considering the unique nature of the local environmental; and (3) adequate connection between inventory flows and spatio-temporal explicit CFs.

2. Towards comprehensive research for addressing freshwater use

2.1. State of development

The WFA and LCA-based methods are focused on addressing the sustainability and preservation of freshwater use, differing, however, in the path followed to achieve this purpose (Boulay et al. 2013).

Several midpoint methods including the one proposed by the WFN, as well as endpoint methods, have been developed to address the vulnerability of the local freshwater systems, and the potential environmental impacts of freshwater use (consumptive use and withdrawal) and degradative use. Fig. 2.1 illustrates those methods and the respective cause-effect chain of freshwater use.

Freshwater enters a river basin via two natural ways: rainfall or by groundwater flow through infiltration of rainwater (The British geographer 2015). Rainfall can be distinguished in green water, surface water (blue water) and renewable groundwater (Milà i Canals 2009). Groundwater flows can be differentiated into two types of freshwater (Milà i Canals 2009): renewable groundwater, in which aquifers are recharged by rainfall, and fossil groundwater, where no surface recharge is applied or when the replenishment period needed is hundreds or even thousands of years.

The use of blue freshwater resources can lead to lowering of the surface and groundwater courses, reducing the availability of water for users with damages to aquatic ecosystems, resources, and at mid-/long term to human health (Fig. 2.1). Furthermore, lowered water tables can lead to groundwater salinisation by saltwater intrusion in coastal regions (Aeschbach-Hertig and Gleeson 2012). Given the current state of knowledge, this cause-effect chain needs to be a matter of research in LCA.

In Appendix 1, the main characteristics of these midpoint and endpoint impact assessment methods related to freshwater use are presented, including a short description of each one. In Pfister et al. (2009), Boulay et al. (2011a,b), and Motoshita et al. (2014) methods, CFs are also derived at midpoint level, as shown in Appendix 1.

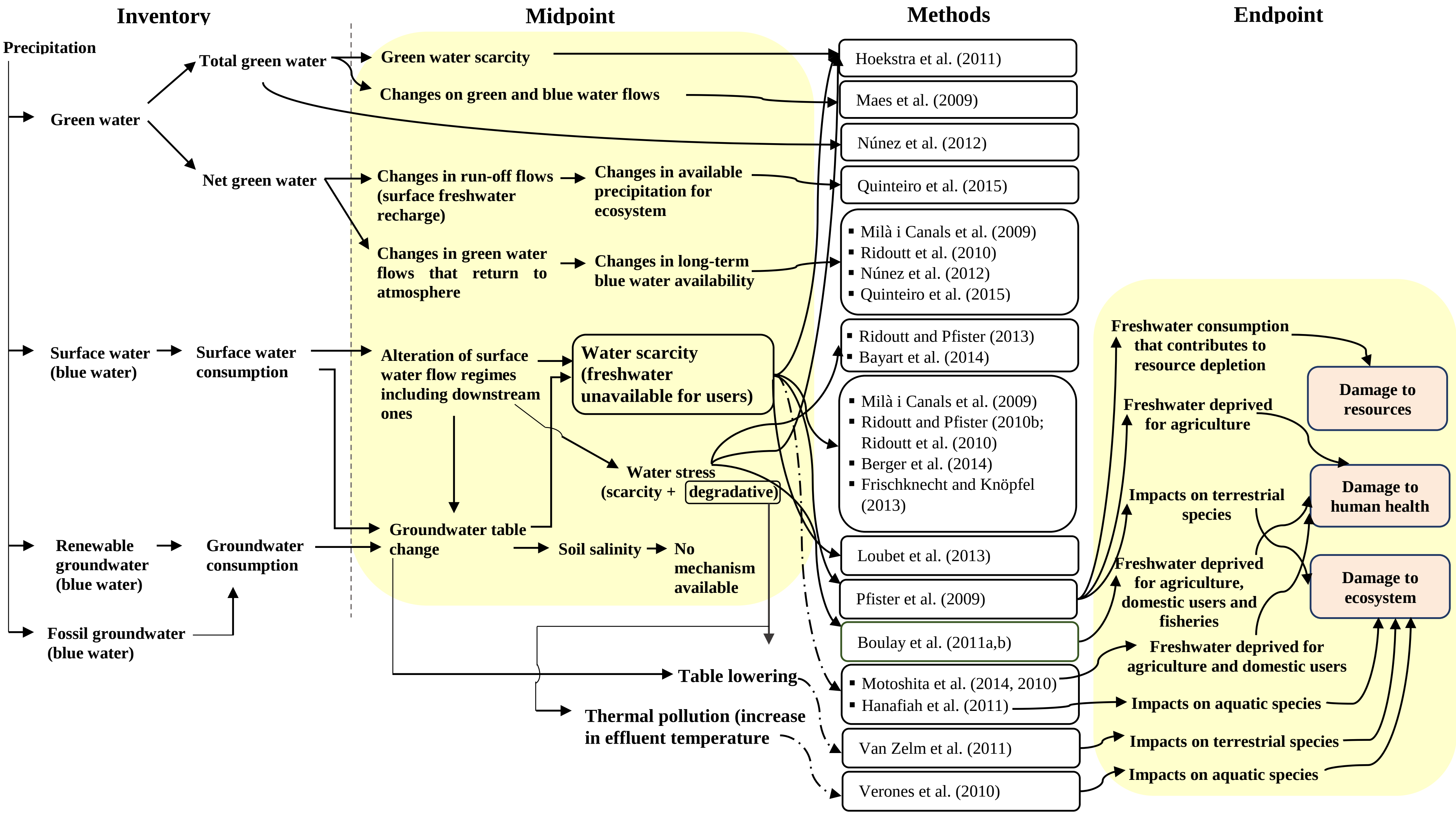

Fig. 2.1. The cause-effect chain of freshwater use and the related midpoint and endpoint methods. Dashed arrows indicate the identified method compromises characterisation factors only at endpoint level, with the exception of Motoshita et al. (2014) method that model midpoint CFs.

2.1.1. WFA method

The WF concept, which was first proposed by Hoekstra and Hung (2002) and described in greater detail by Chapagain and Hoekstra (2004), Chapagain et al. (2006) and Hoekstra et al. (2011) appeared as a natural follow-up to carbon footprint, supporting and guiding stakeholders' decision-makers. For many stakeholders, the WF concept has been a simple, transparent and intuitive means of raising the awareness of consumptive and degradative freshwater use impacts associated with the production and consumption of products and services.

The WF accounts for three components: blue water, green water and grey water (Hoekstra et al. 2011). Blue water refers to fresh surface and groundwater, i.e. the freshwater in lakes, rivers and aquifers (Hoekstra et al. 2011). Green water refers to the precipitation on land that does not run off or recharges the groundwater but is stored in the soil or temporarily stays on top of the soil or vegetation (Hoekstra et al. 2011). Grey water expresses water pollution in terms of a volume polluted, i.e. the volume of freshwater that is required to assimilate the load of pollutants based on natural background concentrations and existing ambient water quality standards (Hoekstra et al. 2011).

Several studies accounting the blue, green and grey water of a wide range of crop products have been published (Aldaya and Hoekstra 2010; Chapagain and Hoekstra 2007; Mekonnen and Hoekstra 2012, 2011a, 2010; Rodriguez et al. 2015; Yoo et al. 2013). Often the WF accounting results have been published in terms of the volume of freshwater required to make a specific product, aggregating green, blue and grey water, without a proper sustainability assessment and disregarding the pressure and potential impacts on freshwater resources and ecosystems (Wichelns 2010).

Regarding grey water, most studies consider nitrogen emissions for agriculture the most critical pollutant for industry effluents, without distinguishing pollution grades and loss of freshwater functionality for downstream users (Mekonnen and Hoekstra 2011b; Pfister and Ridoutt 2014). Several industries discharge their treated effluents to sea, and therefore, the grey WF is accounted as zero because no freshwater is used for assimilation of the pollutants, giving the wrong idea that no potential environmental impacts occur as a result of the discharge of effluent to sea.

Although WFA method gives some guidelines to conduct a spatially and temporarily explicit Water Footprint Sustainability Assessment (WFSA) of a product, considering both environmental, social and economic components, the operationalisation of this phase is still in an early stage of development. Regarding the environmental sustainability, three indicators have been developed: the blue and green water scarcity, and the water pollution level (Hoekstra et al. 2011). Blue and green water scarcity indicators are defined as the ratio of the total blue or green WF in the catchment to the blue or green freshwater availability, respectively, while water pollution level is defined as the ratio of the total grey water and the actual runoff of the catchment under analysis. For instance, Jefferies et al. (2012) addressed the local environmental sustainability by comparing the blue WFs of the tea and margarine with freshwater scarcity in the different regions where the blue WF was accounted. Rep (2011) assessed the WF sustainability of paper using the WBCSD Global Water Tool (WBCSD 2011) for the social and economic component, and the water pollution level and blue water scarcity indicator (Hoekstra et al. 2011) for the environmental component. Spatial and temporal specific water scarcity indicators are critical in addressing the sustainable and fair use of freshwater for agricultural, domestic and industrial users (Antón et al. 2014, Hoekstra et al. 2012, Pfister and Bayer 2014, Reap et al. 2008). While the latter users maintain almost the same volume of blue water consumption during the year, the same does not occur to the former user, where the blue WF depends on the crops' seasonality, water-holding soil characteristics, rainfalls levels and irrigation requirements (Mekonnen and Hoekstra 2011a,b,c, Pfister and Bayer 2014). In the studies conducted by Jefferies et al. (2012) and Rep (2011), the environmental sustainability of green WF was not assessed, as no green water scarcity maps of the catchment under study were available. Although Núñez et al. (2012) calculated green water scarcity indexes for specific energy crops grown in Spain, the applied equation does not allow for deriving green water scarcity in catchments, as recommended by the WFSA method (Hoekstra et al. 2011). Green water scarcity is still largely unexplored due to the lack of data on ecosystem environmental green water requirements (green water to maintain natural vegetation) (Hoekstra et al. 2011). However, Núñez et al. (2012) calculated green water scarcity indexes based on the ratio of the green water consumption by a crop to the effective rainfall in that land-use production system. It still required further research on green freshwater availability, and environmental green water requirements to establish an adequate and accurate method for making sustainability evaluations of green water use. The evaluation of grey WFSA of a river basin is also complex, as the water pollution level in catchments and periods where grey WF occur, should be addressed. Further research is needed to build water pollution level maps for major pollutant categories for industries and agricultural activities.

In conclusion, the WFA method needs more consistent development when interpreting the WF results, in particular the: (1) aggregation of green, blue and grey components; (2) accounting of green water and its link with land use; (3) accounting of grey water, and; (4) assessment of the WFSA.

2.1.2. LCA-based methods

Some of the available LCA-based methods (Fig. 2.1) are complementary regarding the impact pathways, while others assume the same impact pathways but consider the types of freshwater used for different resources, forms of freshwater use, characterisation models at different spatial resolution, or different modelling choices (Berger and Finkbeiner 2010; Boulay et al. 2014; Kounina et al. 2013) (further details in Appendix 1). In some of these methods, the consumptive and degradative freshwater use impacts are modelled at both midpoint and endpoint level (Boulay et al. 2011b; Motoshita et al. 2014; Pfister et al. 2009).

Midpoint methods focus on freshwater scarcity based on a withdrawal-to-availability ratio (Frischknecht and Knöpfel 2013; Loubet et al. 2013; Milà i Canals et al. 2009; Núñez et al. 2015; Pfister and Bayer 2014; Pfister et al. 2009; Ridoutt et al. 2010), or on a consumption-to-availability ratio (Berger et al. 2014; Boulay et al. 2011b; Hoekstra et al. 2012, 2011). A detailed description of the midpoint assessment methods was conducted by Kounina et al. (2013), Berger et al. (2014) and Loubet et al. (2013).

The plurality of midpoint impact assessment methods that have been developed can lead to the report of contradictory results (e.g. Jeswani and Azapagic (2011) and Quinteiro et al. (2014)). Many of the methods have reported the impact assessment results in freshwater equivalent, making it look as though the results are in the same units (e.g. Frischknecht et al. 2009a,b; Herath et al. 2013; Milà i Canals 2010, 2009; Ridoutt and Pfister 2010b). However, they are not, as the point of equivalence is different (e.g. Frischknecht et al. 2009a,b; Herath et al. 2013, Milà i Canals 2010; Ridoutt and Pfister 2010b). This is a major source of misunderstanding, mainly for non-LCA experts who are being presented with freshwater-use related impacts in freshwater equivalent, and are not aware that there are different types of freshwater equivalent. To overcome this report constraint, Ridoutt and

Pfister (2013) defined what the freshwater equivalence is, i.e. equivalent freshwater use at the global average water.

Seeking to provide a meaningful single stand-alone indicator of assessing the freshwater-use related impacts by non-LCA experts, Ridoutt and Pfister (2013) proposed a weighted indicator, integrating both consumptive and degradative water use, while Bayart et al. (2014) proposed the Water Impact Index, a single indicator integrating scarcity and quality issues. In these two single stand-alone methods, the blue water scarcity is addressed based on the spatially explicit Water Scarcity Index (WSI) of the Pfister et al. (2009) method. The degradative freshwater use is assessed at endpoint level using the ReCiPe Life Cycle Impact Assessment method in the Ridoutt and Pfister (2013) method, and based on quality indexes that express quality of withdrawal and released freshwater from and to the catchment under analysis in the Bayart et al. (2014) method. These methods aim for ease of applicability by non-LCA experts, enabling a stand-alone reporting of freshwater use impacts suitable for a general public audience as a means of raising awareness of a sustainable use of freshwater resources.

At an endpoint level, beyond the method of Pfister et al. (2009) that for the first time evaluated the effects of freshwater use on terrestrial ecosystem quality (Fig. 2.1), damages to ecosystem quality are covered by:

- Hanafiah et al. (2011) method, with derived endpoint CFs at river basin level, based on fish species-river discharge relationship (Xenopoulos et al. 2005) and marginal change in water discharge at the river basin mouth.
- Van Zelm et al. (2011) method, describes the impacts on terrestrial vegetation due to groundwater withdrawal, using regression analysis and statistical models to derive CFs for the Netherlands (with no spatial differentiation). This method addresses the effects on the renewable shallow water table (<3.5 m depth), meaning that the water table is well connected with soil, thus depending on green water for replenishing. Mechanisms that consider the interconnection between soil moisture and groundwater are not yet available in literature. In this sense, and although this model takes into account the soil moisture, it remains unclear how green water demand and vegetation may affect renewable groundwater.
- Verones et al. (2010) method, deals with the impact of cooling freshwater discharges on aquatic ecosystems. Verones et al. (2010) highlighted that in addition to the

conventional impacts from chemicals and nutrients that reach freshwater systems, thermal emissions are a further stressor for aquatic ecosystems. These authors derived CFs for thermal pollution in freshwater aquatic environments, based on species' sensitivity distribution following a normal temperature-response function for 36 aquatic species (including fish, molluscs, meduzosa, crustacean, and annelida from temperate regions), and Qual2kw model. CFs for cooling water from a nuclear power plant in Switzerland to two rivers were derived.

Damages to resources (Fig. 2.1) are covered by Pfister et al. (2009) method, which evaluates the amount of water withdrawn above freshwater availability. Considering the energy required for seawater desalination (backup technology) to compensate for freshwater depletion and the fraction of freshwater consumption that contributes to depletion, Pfister et al. (2009) derived endpoint CFs for damage on resources at sub-watershed and country levels.

Damages to human health (Fig. 2.1) are covered by several authors, namely:

- Pfister et al. (2009) method, describes the damages of water deprivation for agriculture use leading to malnutrition, deriving endpoint CFs at sub-watershed and country levels. For this purpose, these authors considered the water scarcity index (WSI) developed for midpoint level, the fraction of agricultural freshwater use, the annual number of malnourished people per freshwater quantity deprived, and the damage caused by malnutrition.
- Boulay et al. (2011a,b) method, considers the impacts from freshwater deprivation for agriculture, aquaculture, and for domestic uses, considering the loss of functionality for each freshwater use. Boulay et al. (2011b) proposed endpoint CFs at watershed and country level, taking into account local freshwater stress, the extent to which users will be affected by a change in freshwater availability and the adaptation capacity, and the importance of human health impacts caused by a freshwater deficit for a user. Adaptation capacity defines whether the change in freshwater availability creates deficit or compensation scenarios to overcome freshwater shortage.
- Motoshita et al. (2014, 2010) methods, describe the damage from shortages in food production resulting from agricultural water scarcity, and the damage from infectious diseases arising from domestic freshwater consumption, respectively. Motoshita et al. (2010) proposed CFs, at country level, based on non-linear multiple relationships

between freshwater scarcity, accessibility to safe freshwater and damage to human health caused by infectious diseases. Motoshita et al. (2014) proposed CFs, also at country level, based on relationships between agricultural freshwater scarcity, crop productivity and undernourishment damage caused by changes in food production.

Endpoint characterisation is more complex and relatively more uncertain than midpoint characterisation, as the endpoint modelling results are based on fate and effect modelling founded in subjective judgments. However, endpoint results have the benefit of being concise and easy to understand and communicate to non-LCA experts. For instance, endpoint characterisation allows the comparison concerning: (1) the malnutrition damage due to freshwater deprivation for agriculture use (Pfister et al. 2009); (2) the human health impacts from freshwater deprivation for agriculture, aquaculture and domestic uses (Boulay et al. 2011b), and; (3) the relative importance of malnutrition damage from food deprivation due to irrigation freshwater demand for agricultural production (Motoshita et al. 2014), with other impact categories of LCA methods (e.g. ReCiPe, Impact 2002+, LC-Impact) and on the area of protection human health using the concept of disability adjusted life years (DALY). However, this unit may not be easily understood by non-LCA experts, hindering the communication of products with freshwater-use and the related impacts for the general public.

In some endpoint methods, socio-economic compensation mechanisms are considered to derive CFs from damage to resources (Pfister et al. 2009), and damage to human health (Boulay et al. 2011b; Motoshita et al. 2014, 2010). Compensation accounts for the capacity of human users to adapt to a freshwater deficit through the use of backup technology (Boulay et al. 2011b). Endpoint results for human health assume critical relevance in developed countries, where no socio-economic compensation mechanisms are available.

3. Methodological challenges

Spatially explicit stress indicators normalised with a reference flow of the world weighted value are required for assessing midpoint impacts from freshwater use (Boulay et al. 2015b).

Despite the need for acquiring better data, i.e. improve Life Cycle Inventory (LCI) databases and developing models to better describe the impact pathways of freshwater and its degradative use, at both midpoint and endpoint level, it is more appropriate at the impact assessment development to focus on the establishment of consensual indicators and guidelines for consistent freshwater use LCIA (Boulay et al. 2015b).

Regarding blue water, the WULCA group have been building consensus around the development of blue water scarcity CFs for LCA-based freshwater-use related impacts. This group developed a water use midpoint indicator that represents the relative available water remaining per area in a watershed (called AWaRe), after the demands of humans and aquatic ecosystems have been met (WULCA 2015). Currently, this indicator considers the total freshwater available for runoff, but is expected to make a distinction between the available surface water and groundwater in the future.

Despite the significant progresses that have been achieved for addressing the freshwater-use related impacts, there are still some issues that remain partially unsolved, requiring particular attention, such as the accounting and assessing of the potential environmental impacts of green water flows, the establishment of spatially and temporarily explicit CFs to consider the local environmental uniqueness, and its link with inventory flows.

3.1. Green water flows

3.1.1. Background

Terrestrial ecosystems primarily consume freshwater from soil moisture and groundwater. The interactions between land use (encompassing soil characteristics and dynamic soil interaction processes), crop characteristics and related growing patterns and rainfall have been hindering the establishment of a robust mechanism of green water consumption quantification, and the development of an operational method to evaluate the potential environmental impacts of green water flows. Any perturbation of the local evapotranspiration (ET) is considered to have an environmental impact, which differs depending on whether the perturbation is an increase or decrease of green water flows, and also on the location. Changes on green water flows due to land use can lead to changes in available rainfall both on a regional and continental scale (considering the bridges of atmospheric moisture transport) (Berger et al. 2014; Keys et al. 2012; Launiainen et al. 2014; Quinteiro et al. 2015) and changes in long-term blue water availability (Maes et al. 2009; Milà i Canals et al. 2009; Núñez et al. 2012; Quinteiro et al. 2015; Ridoutt et al. 2010). For instance, the conversion of natural land (e.g. forests, grassland, shrubland) into agriculture fields or other human land modifications lead to changes on green water flows recycling into the atmosphere. When the new land cover leads to an increase of ET flows recycled to the atmosphere, the surface runoff is reduced due to the higher freshwater requirements of the new land cover (Calder 2003; Núñez et al. 2013; Rockström and Gordon 2001; Scanlon et al. 2007), affecting blue water production, which can lead to a long-term reduction of blue water availability. The increase in surface runoff may disturb stream flows, suspended solids and nutrient transport towards freshwater streams, leading to freshwater quality degradation (Collins et al. 2011; Jones et al. 2012), and rise of the groundwater table, which in turn can lead to the increase of soil salinity (Annenberg Foundation 2014). This interaction between soil, green water flows and atmosphere can also disturb the rainfall patterns at both regional and global scales (Pielke et al. 2006; Van Dijk and Keenan 2007). Moreover, the environmental mechanism that allows the understanding of the relevance of green water availability and its seasonal variation for both replenishing renewable groundwater and vegetation growing, especially when the water table is hallow and well connected with soil moisture, is not yet available in the literature.

Regarding green water, only midpoint methods, following WFA and LCA-based methods are currently available.

Concerning the WFA method, the non-consideration of the net green water concept when accounting green water use by a crop remains a major conceptual controversy, as land use changes influence the changes on green water flows and green freshwater availability (Núñez et al. 2013; Pfister and Ridoutt 2014; Quinteiro et al. 2015; Ridoutt et al. 2010).

Concerning the LCA, some efforts have been made to address the impacts from green water flows. Changes in long-term blue water availability due to changes on regional green water flows have been evaluated by Milà i Canals et al. (2009), Núñez et al. (2012), and Ridoutt et al. (2010). Maes et al. (2009) developed an impact assessment method to assess the effect of land use changes and green water flows on freshwater-related ecosystem services. Quinteiro et al. (2015) developed a method to assess both impacts on terrestrial green water flows and reductions in surface blue water production caused by reductions in surface runoff due to changes in the land-use production system. According to the latter authors, when assessing the potential environmental impacts of a land-use production system resulting from changes in green water flows, the interactions between vegetation and freshwater should be considered at both green water use and soil, and green water use and atmosphere interfaces. The interface between green water and soil refers to how a change in green water use affects the regional long-term availability of surface blue water, whereas green water and atmosphere interface refers to how land use affects the evaporation and transpiration that is recycled into the atmosphere, and then, the rainfall that returns to the regional terrestrial ecosystem.

All these methods follow different approaches to achieve and calculate the impacts from green water flows into LCA. Milà i Canals et al. (2009) consider two different interventions in the LCI: 1) the green water flow of the actual land cover and 2) land use effects on infiltration and runoff, by considering a set of percentage values of rainfall “lost”, developed by Zhang et al. (1999) for different reference land uses.

On the other hand, Núñez et al. (2012) also present two alternative interventions: 1) the green water flow of the actual land cover and 2) the net green water flow, while Ridoutt et al. (2010) and Quinteiro et al. (2015) account for net green water flow. Maes et al. (2009) do not provide information on how to conduct the LCI.

After the initial controversy associated with the green water concept, it is now well accepted by the LCA community that the inventory of green water use shall consider the land use changes by applying the net concept. This leads to other controversy issues, namely the meaning of potential natural vegetation (PNV), and the influences of climate changes and anthropogenic activities on natural vegetation distribution (Chiarucci et al. 2010; Farris et al. 2010; Loidi et al. 2010; Somodi et al. 2012; Tuxen 1956). For instance, the phytosociological models identified evergreen forest (*Quercus*) as PNV of the Iberian Peninsula (Carrión and Fernández 2009, Farris et al. 2010), while palaeoecological studies show that pine and mixed pine-oak forests have adapted better to soil and climate change, and should also be considered as PNV of the Iberian Peninsula (Carrión et al. 2009; Loidi et al. 2010).

Regarding the impact characterisation model, none of the methods developed by Milà i Canals et al. (2009), Ridoutt et al. (2010) and Núñez et al. (2012) provide specific CFs to evaluate the green water availability and specific impacts from green water flow of actual land cover. Milà i Canals et al. (2009) recommend the use of country and basin explicit CFs to assess the land use effects on infiltration and runoff (Appendix 1). Ridoutt et al. (2010) and Núñez et al. (2012) used the WSI originally developed by Pfister et al. (2009) for blue water to assess the potential impacts of net green water flow, which can lead to inconsistent green water impact results. Based on ET of PNV and ET of actual land-use production systems, the Maes et al. (2009) method allows regional CFs to be derived for the land use impact on the terrestrial green water flows and on the aquatic blue water flows. However, no indications or recommendations on how to calculate these values are provided. Finally, Quinteiro et al. (2015) developed a method to derive regional and species-specific CFs for both green water use and soil interface, and green water use and atmosphere interface. The derived CFs depend mainly on solar radiation, rainfall levels, soil moisture, root zone water holding capacity and canopy conductance (approach based on the leaf area index and stomatal conductance) of the actual land cover.

3.1.2. Calculation of ET of PNV and actual land cover

Terrestrial ET has been identified as a significant source of rainfall for land-use production systems (Ellison et al. 2012; Trenberth 1999; Van der Ent et al. 2010), and it is inherently difficult to measure and predict. Due to these reasons, green water flows deserve more emphasis than received so far. To understand how land use affects the volume of freshwater that is evaporated or transpired and recycled to the atmosphere, the accounting of ET of PNV (pristine conditions) and actual land-use production systems should be carried out.

There are limited methods based on Penman-Monteith (Landsberg and Waring 1997; Monteith 1965) and Priestley-Taylor (Priestley and Taylor 1972) equations calibrated and validated to calculate potential ET of different species on a regional and local scale. Penman-Monteith equation requires climatic data (e.g. temperature, rainfall, solar radiation, vapour pressure deficit) and non-climatic data (e.g. canopy conductance, leaf area index), while Priestley-Taylor equations require less input data, which depend mainly on solar radiation and air temperature. These equations are not parameterised to be applicable to PNV (Quinteiro et al. 2015), therefore, in estimating the ET of natural forested ecosystem, grasslands and shrublands, the Zhang et al. (2001) approach was used by Quinteiro et al. (2015) and Ridoutt et al. (2010), while for natural vegetation in Mediterranean dry lands, the Piñol et al. (1991) approach was used by Núñez et al. (2013, 2012).

Actual ET can be estimated based on hydrological water balance equations embedded in crop and forest growth models, from field measurements at meteorological stations and more recently from remote sensing. ET of PNV can be estimated using Zhang et al. (2001) and Komatsu et al. (2012) methods, and derived from recorded satellite data for ecoregions

(Bailey 1998). Ecoregions are large units of land containing a geographically distinct assemblage of natural communities and species that can be taken as a proxy of pristine conditions (Olson et al. 2001).

The ET of crops can be calculated using CROPWAT model (FAO 2013), while ET of forest catchments shall be calculated using specific models, such as 3-PG model (Physiological Principles Predicting Growth) (Landsberg and Waring 1997), CABALA model (Battaglia et al. 2004) or Soilflux model (Sinclair Knight Merz 2005). The CROPWAT model (FAO 2013) calculates ET of crops, based on an assumed crop height of 0.12 m, a fixed surface resistance (describes the resistance of vapour flow through stomata openings, total leaf area and soil surface) of 70 $s.m^{-1}$ and an albedo of 0.23 (Allen et al. 1998). Forested catchments have different characteristics from crops. Tree height is significantly higher than crop height, leading to rainwater sustained in the foliage of the trees that is released back to atmosphere (interception). Interception by canopies assumes a significant contribution to the total freshwater amount evaporated or transpired by a tree (Launiainen et al. 2014), as shown by measurements carried out at the boreal forest station of Helsinki University, in which 42 % of the total evaporation of Finnish boreal forests is from rainfall interception in its canopy (Rep 2011).

Concerning field measurements, actual ET is commonly measured using micro-meteorological techniques, i.e. total green water flows are measured with eddy covariance technique (Baldocchi et al. 2001; Sellers et al. 1995; Verstraeten et al. 2008). This is an accurate technique to measure ET from trees and crops. However, the regional scale applications of field-based measurement techniques are expensive and time consuming, which have hampered the establishment of a spatially explicit ET database.

Regarding remote sensing, continuous satellite observations, supplemented with routine meteorological data, provide a record of actual ET over large areas and ET of PNV from ecoregions. The constraint comes when remote sensing records are applied to lower scales. For instance, ET datasets from Moderate Resolution Imaging Spectroradiometer (MODIS) are available at 100 ha resolution (Mu et al. 2011, 2007), which is a higher resolution than the dimension of many of the land-use production systems. The use of satellite ET recorded data can give an indication of regional ET. However, satellite data are quite inadequate for accounting ET of small land-use production systems, unless modifications can be performed to use satellite-based data at a moderate resolution scale (30 to 120 m), to derive growing seasonal ET at a 30 m resolution (Bhattarai et al. 2014).

At the LCI level, continued efforts to operationalise the available methods to calculate the ET of forest trees – considering their specificities and rainfall canopy interception – should be taken. Further research to achieve a consensus on the most adequate methods to calculate ET of PNV is also desirable. Furthermore, at the LCIA level, further efforts should be taken to derive spatially and species-specific CFs, understanding in what way green water flows lead to disturbances in terrestrial and aquatic ecosystems (Quinteiro et al. 2015).

3.2. Spatial variation

Spatial differentiation has been acknowledged as a relevant issue in WF studies because of the local/regional environment uniqueness of freshwater use and its degradative use (Antón et al. 2014; Herath et al. 2013; Pfister et al. 2009). Unlike global impacts, such as global warming, that do not need spatial detail in the application of the CFs as the damages are spreading on a global level, freshwater use has very local and specific impacts. Therefore, the consideration of spatial details in the LCA-based methods is of paramount benefit. Spatial resolution assumes critical relevance specially for crop production, as freshwater consumption is not uniformly distributed, and varies depending on the properties of the crop, soil conditions, different agricultural practices (crop field or protected – tunnel or greenhouse, the rainwater-harvesting systems), crop irrigation efficiency, and freshwater availability (Pfister and Bayer 2014).

Regional, single CFs based on mean, annual blue freshwater availability and freshwater scarcity – which induces freshwater deprivation for downstream human users and ecosystems – have been used in midpoint assessment methods (Boulay et al. 2011b; Frischknecht and Knöpfel 2013; Loubet et al. 2013; Milà i Canals et al. 2009). In addition, Quinteiro et al. (2015) provide single-site and species-specific CFs for green water flows.

Pfister et al. (2009) developed regionalised WSI (for more than 11,000 sub-watersheds for the 1961-1990 time series). The WSI is not based on consumption-to-availability but on the use-to-availability ratio. This WSI includes the degradative use indirectly, such as thermal and chemical pollution, which may overestimate scarcity, mainly in industrialised countries with large non-consumptive freshwater use and adequate wastewater treatment (Pfister et al.

2011b). Spatially explicit values of total groundwater and surface freshwater stocks at finer resolutions are hard to obtain.

Furthermore, when a single freshwater scarcity CF is determined for large sub-watersheds, especially when the sub-watersheds have a non-uniformly freshwater availability and demand, uncertainty in the freshwater-use related impacts is introduced. Therefore, a deeper analysis at sub-watershed level integrating downstream cascade effects as illustrated by Loubet et al. (2013), would increase the awareness on how a local freshwater consumption could lead to potential freshwater deprivation on downstream users and ecosystems.

3.3. Temporal variation

Temporal aspects related to WFA and LCA-based methods are also relevant because depending on the growing season and climate, crop can shift between irrigation and non-irrigation practises within a year and from year to year, leading to higher or lower freshwater scarcity. The non-consideration of the interannual seasonality of rainfall and crop-growing seasons, can mask the high variability of freshwater use and its availability in some regions, giving misleading guides to the choice of the most adequate land-use production system for a region.

Freshwater scarcity is known to be a seasonal problem in many parts of the world. The non-consideration of intra-annual rainfall can lead to a biased estimation of freshwater availability, which is of critical relevance for semi-arid or arid regions (in which the potential ET is significantly larger than the rainfall levels) and for monsoonal areas (Boulay et al. 2015b). To provide monthly freshwater scarcity CFs, instead of a single CF for the whole year, some temporal and spatial methods have been developed (Hoekstra et al. 2012; Núñez et al. 2015; Pfister and Bayer 2014).

In the derived WSIs by Pfister et al. (2009), a fixed variation factor reflecting monthly and annual temporal variability of water availability was applied, masking the impact of blue water use during the growing seasons of crops. To overcome this constraint, Pfister and Bayer (2014) developed monthly WSIs, capturing the interannual rainfall variability in the 11,000 worldwide sub-watersheds analysed. However, these WSIs neither account for the perturbation of seasonal runoff patterns by river flow regulation with dams and reservoirs, nor do they include inter sub-watersheds transfers of freshwater, and thus overestimate the local freshwater stress of net sub-watershed importers and underestimate the local freshwater stress of the net exporters. Also, monthly WSI calculations should take into account the construction of local freshwater storage infrastructures for the purpose of supplying livestock, agriculture, and domestic users. Especially in arid and semi-areas, the capture of rainwater during the wet season for use at a later time is a common practice (UNEP 1997).

To operationalise the environmental WFSA, Hoekstra et al. (2012) developed monthly blue water scarcity indicators for 405 river basins (based on consumptive use-to-availability ratios), taking into account the intra-annual rainfall variability, the monthly natural runoff and blue WF by river basins for the 1996-2005 time series. According to these authors, the blue freshwater availability is assessed taking into account the Environmental Water Requirements (EWR) needed to sustain the functioning of ecosystem services. Alteration to the natural flow regimes of rivers and streams can be a risk to the maintenance of the freshwater dependent ecosystems (Smakhtin et al. 2004).

Estimations of EWR are difficult due to lack of data between river flows and multiple components of river ecology (Smakhtin and Eriyagama 2008). Smakhtin et al. (2004) estimated a spatial variability of EWR of 20 to 50 % of annual natural flows that need to be allocated to freshwater dependent ecosystems ensuring no ecological damage. These estimations were strongly criticised by ecologists who argue that this range is too low to maintain a fair condition of freshwater systems (Arthington et al. 2006). Richter et al. (2011) developed a presumptive standard of 80-90 % of mean annual natural discharge, which can be regarded as a precautionary estimate of environmental flow requirements to ensure a moderate to high level of protections of freshwater ecosystems.

Global hydrological models, such as WaterGAP (Alcamo et al. 2003), GLDAS-2 (Aqueduct) (Gassert et al. 2013) LPJml (Bondeau et al. 2007) estimate EWR under pristine conditions, which despite being taken as a proxy of the current state could bring some uncertainty to the actual EWR. Currently, it is the best way to establish the essential freshwater needs to sustain aquatic ecosystem and human livelihoods. Furthermore, the increase in the frequency and intensity of storms, floods, and droughts, and an amplification of global warming effects may lead to changes in freshwater-resource availability, alterations on global freshwater cycle (Alcamo et al. 2007; Döll 2009; Huntington 2006), and to an increase of regional freshwater scarcity (IPCC 2007), affecting the fucntioning of ecosystem services. In this sense, the frequency and intensity of storms, floods, droughts, and global warming should be considered to estimate the EWR.

Concerning the LCA-based methods, Boulay et al. (2011b) developed stress single CFs, differentiating the blue freshwater into surface and groundwater and accounting for temporal variations of surface blue water availability, by using the statistical low-flow Q90 (represents the flow that is exceeded in 9 of 10 months) instead of total freshwater availability (Döll et al. 2009). Although it seems relevant to consider surface freshwater and groundwater in the development of the stress indicator, it is also crucial to differentiate between surface water and groundwater in the inventory, since freshwater availability may not affect the same users (Boulay et al. 2014). For instance, a decrease of groundwater availability may affect industrial and domestic users, but normally it would not directly affect agriculture users who mainly utilise surface freshwater. Moreover, a better accounting of surface freshwater availability that factors in the seasonal variability of rainfall and also the influence of peak flows in regional freshwater quality (Pfister and Bayer 2014, Van Beek et al. 2011), could improve the accuracy of these CFs.

With the first steps to consider a finer spatial resolution of WSIs, encompassing also the short and long-term temporal variability of freshwater due to climate change effects, Núñez et al. (2015) calculated WSIs on a yearly basis for 117 sub-watersheds of Spain. Further work would contribute to the development of more robust and accurate regional and temporal freshwater scarcity CFs. This includes river flow regulation, improving of the estimations of EWR, climate change scenarios and a deeper understanding of drivers that affect atmospheric evaporation recycling, and atmospheric water vapour transport, linked to global climate models.

The recently derived AWaRe indicator for water scarcity footprint also considers the EWR (called above as demands of humans and aquatic ecosystems) (WULCA 2015). These CFs are normalised with the reference flow of the consumption weighted world average. They have been calculated at the sub-watershed level and monthly time-step, with the possibility of aggregating them to an annual resolution, when necessary. Although total freshwater availability is considered, including the renewable resources, no explicit differentiation between renewable groundwater and surface water is performed, arising in issues related to the time horizon required to recharge groundwater table and thus, to sustain aquatic ecosystem and human livelihoods. Furthermore, the use of backup technologies (e.g. desalination plants and freshwater importation) to compensate for freshwater resource depletion should also be considered when estimating the human freshwater demand.

The CFs for damage to human health, ecosystem quality and resources related to freshwater, have been provided by Pfister et al. (2009), Boulay et al. (2011b), Motoshita et al. (2014, 2010), Hanafiah et al. (2011), Van Zelm et al. (2011), and Verones et al. (2010) at country, watershed, and sub-watershed level, with an average time-step. Similarly to midpoint models, endpoint models are going towards the inclusion of spatial and temporal differentiation, as well as a better understanding of fate and effect factors within different regions. For instance, malnutrition from a decrease in food production caused by freshwater scarcity in a country may spread to other countries through food trade (Motoshita et al. 2014). In this case, the monthly variability of food trade can be relevant to understand the intra-annual trade-induced effects in importer countries, and to study a possible adaptation of agricultural production to climates that drive cropping patterns to less freshwater intensive crops and higher crop yields. Also, depending on the considered time horizon, the expected population growth – mainly in less developed countries – economic growth and climate changes can lead to changes in freshwater withdrawal for agriculture, industrial and domestic purposes, changing the human health, ecosystem and resources dynamics.

3.4. Adequate connection between inventory and impact assessment phase

From a broad perspective, global datasets of freshwater use are satisfactory. However, there are several details that require harmonisation, consensus and parametrisation to be included in inventory databases, as supported by Kounina et al. (2013), such as: (1) the differentiation of both freshwater types (surface or groundwater) and freshwater sources (river, lake, dam and reservoir for surface water, renewable, fossil, shallow, deep of groundwater and freshwater stored as soil moisture); (2) the differentiation between consumptive and non-consumptive use; (3) the quality of input and output of freshwater flows from background and foreground processes.

Potential environmental impacts related to consumptive and degradative freshwater use require adequate spatial information in order to ensure a spatial consistency between inventory and the impact assessment phase (Reap et al. 2008). The establishment of spatially explicit databases to perform the inventory remains a major scientific challenge in LCA (Huijbregts 2013). Also, both midpoint and endpoint CFs are strongly influenced by the spatial freshwater variability. The establishment of a consistent link between CFs with a finer spatial resolution and the related spatial inventory is crucial to better understand the environmental relevance of the impact results. However, despite the uncertainty of the freshwater-use related impacts caused by different spatial and temporal variations between inventory and impact assessment phase, when spatial inventory has no temporal resolution, the related impact assessment phases can be performed by averaging the monthly CFs values (Boulay et al. 2015b; Pfister and Bayer 2014) or more generic ones (Pfister et al. 2009).

Although several studies have considered geographical information (e.g. Antón et al. 2014, Boulay et al. 2015a, Milà i Canals et al. 2010, Núñez et al. 2015, Pfister and Bayer 2014), there is not currently any software available that integrates geographical systems (GIS) and temporal information databases on both inventory and impact assessment phases into LCA studies. Furthermore, on the contrary to what happens with foreground processes, the plurality of the related background processes make it quite difficult to map the geographical location of all background processes encompassed in an LCA study (Antón et al. 2014; Reap et al. 2008). The difficulty in identifying the exact location of background process and characterising the local environmental uniqueness can hinder the elaboration of an accurate spatial differentiated impact assessment, as more generic CFs can be applied. Therefore, additional efforts, should be undertaken to include the exact location of the supply chain processes in the LCI databases, ensuring the consistency of data from foreground and background processes.

4. Summary

The methodological developments currently achieved for measuring and assessing the freshwater-use related impacts were presented and discussed. The available methods differ significantly according to the type of freshwater considered (green water, surface water, renewable groundwater, fossil groundwater), the cause-effect chain covered, the consideration of freshwater scarcity, stress and/or degradative freshwater use, as shown by Fig. 2.1.

The main methodological challenges are:

- **Accounting and assessing of the potential environmental impacts of green water flows.**

  Despite the difficulty to measure and calculate the ET of small and medium-size land-use production systems, further research on models to operationalise these calculations is required.

For forested ecosystems, specific models should be used that account for the rainfall interception, which contributes to a significant share of the total ET of actual land. However, these models need to be adapted and calibrated to the different forest trees and spatial soil conditions. Remote sensing could be an option, when the satellite resolution is adequate to the dimension of land-use production systems.

The definition and calculation of ET of PNV is also an issue that brings uncertainty to the freshwater-use related impacts. Further research is required to achieve a consensus on the most adequate methods to calculate ET of PNV.

▪ **Spatial variation.**

The development of spatially explicit methods capable of addressing the local and/or regional environmental uniqueness of freshwater use and its degradative use should be maintained as a research priority. A deeper research at sub-watershed level, and the integration of the downstream cascade effects, would increase the awareness on how a local freshwater consumption could lead to potential freshwater deprivation on downstream users and ecosystems.

There is still a lack of spatially explicit values of total groundwater and surface freshwater stocks. In addition, a deeper understanding of the fate of ET is needed, mainly related to green water flows due to the atmospheric moisture transport.

▪ **Temporal variation.**

A deeper analysis of the rainfall seasonality and intensity is required to establish better models predicting renewable groundwater recharge, surface freshwater availability, and giving indications to choose the most adequate land-use production systems attending to local or regional rainfall patterns. In addition to these aspects, a deeper understanding on the influence of regional very high runoff flows during episodic storms in freshwater quality could improve the accuracy of temporal and spatially explicit CFs.

- **Adequate connection between inventory flows and spatio-temporal explicitly characterisation factors.**

  To ensure an accurate and comprehensive spatially explicit assessment of freshwater-use related impacts, adequate spatial and temporal information between the inventory and the impact assessment phase, should be considered.

  Additional efforts should also be undertaken to include the exact location and temporal information about the supply chain processes in the LCI databases, ensuring the consistency of data from foreground and background processes.

Further research on these pertinent problems can lead to the attainment of more comprehensive and robust spatially and temporally explicit methods for assessing the freshwater-use related impacts, considering the environmental uniqueness. This would be beneficial to support decision-making and freshwater management activities.

Appendix 1

Table 2.A.1. Main characteristics of midpoint and endpoint impact assessment methods related to freshwater use.

Approach	Method: Midpoint	Method: Endpoint	Accounting/ impact assessment	Intervention	Characterisation model	Notes
WFA	Hoekstra et al. (2011)	n.a.	▪ Water Footprint (WF) accounting Chapagain and Tickner (2012), (units – m^3 eq.) ▪ WF Sustainability Assessment (WFSA) (environmental, social and economic criteria), (units – m^3 eq.)	▪ Blue water ▪ Green water ▪ Grey water (freshwater pollution)	▪ WFSA: - green water scarcity indicator (dimensionless) - blue water scarcity indicator (Hoekstra et al. 2012) (dimensionless) - water pollution level (Liu et al. 2012) (dimensionless)	▪ WF accounting: blue water is not distinguished into other sub-components (e.g. surface, fossil and renewable groundwater) (Sala et al. 2013) ▪ Intake freshwater quality not considered and released water quality indirectly considered by grey water ▪ WFSA (environmental pillar): - blue WF is weighted by river basin-specific freshwater scarcity indicator available at monthly time-step (Hoekstra et al. 2012) - no proper methods and required data are available to proceed to green water scarcity evaluations (Jefferies et al. 2012; Pfister and Ridoutt 2014) - water pollution levels available for more than 1,000 rivers (Liu et al. 2012) ▪ WFSA (social and economic sustainability pillars): no proper methods available ▪ Compensation mechanisms not considered
LCA-based	Maes et al. (2009)	n.a.	▪ Land use impacts on terrestrial green water flow (TWI) (units – m^3 eq.) ▪ Land use impact on aquatic blue water flow (AWI) (units – m^3 eq.)	Green water	Regional and CFs for TWI and AWI	▪ Basin atmospheric evaporation recycling (BIER) – the share of water evaporated or transpired through plants to the atmosphere that returns to the same basin – not considered ▪ Degradative freshwater use not considered ▪ Compensation mechanisms not considered
LCA-based	Núñez et al. (2012)	n.a.	▪ Blue water deprivation (units – m^3 eq.) ▪ Green water deprivation (units – m^3 eq.)	▪ Blue water ▪ Green water flow ▪ Net green water flow	▪ Blue water deprivation: Water Scarcity Index (WSI) from Pfister et al. (2009) ▪ Green water deprivation: - Green Water Scarcity Index (GWSI) (Hoekstra et al. 2011) - net green water flow assessed taking into account the WSI for blue water from Pfister et al. (2009)	▪ Degradative freshwater use not considered – intake and release water quality are not taken into account ▪ Green water deprivation: Piñol et al. (1991) equation to calculate natural reference land use; equation valid only for dry lands ▪ Compensation mechanisms not considered

Table 2.A.1. Main characteristics of midpoint and endpoint impact assessment methods related to freshwater use. (cont. I)

Approach	Method		Accounting/ impact assessment	Intervention	Characterisation model	Notes
	Midpoint	Endpoint				
LCA-based	Quinteiro et al. (2015)	n.a.	▪ Impacts on terrestrial green water (TGWI) (units – m^3 eq.) ▪ Reductions in surface blue water production (RBWP) caused by reductions in surface runoff (units – m^3 eq.)	Net green water flow	Regional and species-specific CFs for TGWI and RBWP	▪ BIER is considered ▪ Degradative freshwater use not considered – intake and release water quality are not taken into account ▪ CFs for *Eucalyptus globulus*; but model is applicable to other land-use production systems ▪ Compensation mechanisms not considered
LCA-based	Milà i Canals et al. (2009)	n.a.	▪ Freshwater ecosystem impact (FEI) (units – m^3 eq.) ▪ Freshwater depletion impact (FD) (units – kg Sb-eq.kg^{-1})	▪ Green water flow ▪ Blue water ▪ Land use effects on infiltration	▪ CFs not provided for green water flow impacts ▪ CF for FEI (surface water): - Water scarcity indicator at river basin level (Smakhtin et al. 2004) - Total renewable water resources per capita (WRPC) at country level (Falkenmark 1986) - Water use per resource (WUPR) at country level (Raskin et al. 1997) ▪ CF for FEI (land-use effects): percentages for rainfall "lost" derived from Zhang et al. (1999) ▪ CF for FD: abiotic depletion potential of aquifer (Guinée and Heijungs 1995; Guinée et al. 2002)	▪ Degradative freshwater use not considered – intake and release freshwater quality are not taken into account ▪ Inventory distinguishes between freshwater consumptive use (evaporative use) and non-consumptive freshwater use (non-evaporative use) ▪ Impacts of green water, surface and aquifer evaporative uses are disregarded ▪ FEI performed at least at river basin stress level; CFs less spatially explicit than the one developed by Pfister et al. (2009) ▪ CFs for FD not available due to data constraints ▪ Compensation mechanisms not considered
LCA-based	Ridoutt et al. (2010), Ridoutt and Pfister (2010b)	n.a.	Stress-weighted WF (units – m^3 eq.)	▪ Blue water ▪ Net green water flow ▪ Grey water	For all interventions, WSI developed by Pfister et al. (2009) at 0.5º grid cell, watershed and country level	▪ Impact of green water after it became blue water (impact of land on blue water resources) is evaluated ▪ Stress-weighted WF can be normalised dividing it by global average WSI of 0.602 (Page et al. 2011) ▪ Intake freshwater quality not considered and released freshwater quality indirectly considered by grey water ▪ Compensation mechanisms not considered
LCA-based	Frischknecht and Knöpfel (2013)	n.a.	Scarcity-weighted consumptive use (units – eco-points)	Blue water	CFs (eco-factors) are calculated based on actual emission flow and political targets, and available at basin and country level	▪ Degradative freshwater use not considered – intake and release water quality are not taken into account ▪ Scarcity-weighted freshwater consumptive use can be normalised using the Swiss level of freshwater withdrawal (2.6 $km^3.yr^{-1}$) (FAO 2011) ▪ Compensation mechanisms not considered

Table 2.A.1. Main characteristics of midpoint and endpoint impact assessment methods related to freshwater use. (cont. II)

Approach	Method Midpoint	Method Endpoint	Accounting/ impact assessment	Intervention	Characterisation model	Notes
LCA-based	Loubet et al. (2013)	n.a.	Water deprivation integrating downstream cascade effects (units – m^3 eq.)	Blue water	CFs based on freshwater scarcity at sub-river basin considering weighting parameters for downstream sub-river basin: area (U.S.G.S 2012); river volume (Hanafiah et al. 2011); and number of inhabitants (CIESIN/CIAT 2005)	▪ Degradative freshwater use not considered – intake and release freshwater quality are not taken into account ▪ Compensation mechanisms not considered
LCA-based	Berger et al. (2014)	n.a.	Risk to freshwater depletion (RFD) (units – m^3 eq.)	Blue water	CFs for RFD are based on consumption-to-availability ratios, considering surface water stocks and aquifers, at basin scale	▪ Degradative freshwater use not considered – intake and release freshwater quality are not taken into account ▪ BIER is considered ▪ The method does not intend to 'predict' potential impacts on freshwater resources but to address the vulnerability of basins to freshwater use ▪ Compensation mechanisms not considered
Stand-alone LCA-based indicator: Bayart et al. (2014)	n.a.	n.a.	Water impact index (units – m^3 eq.)	Blue water	▪ WSI from Pfister et al. (2009) ▪ Quality indexes of intake and released freshwater are considered	▪ Quality indexes calculated to the most penalising pollutant (Directive 2000/60/EC 2000) ▪ Compensation mechanisms not considered
Stand-alone LCA-based indicator: Ridoutt and Pfister (2013)	n.a.	n.a.	Consumptive freshwater use (CWU) and degradative freshwater use (DWU) expressed as a single stand-alone WF result (units – m^3 eq.)	▪ Blue water ▪ Emissions affecting water quality	▪ CWU uses WSI from Pfister et al. (2009) weighted by the global average WSI of 0.602 (Page et al. 2011) ▪ DWU based on ReCiPe method (Goedkoop et al. 2013) with some adjustments (Pfister et al. 2011c)	▪ DWU include freshwater eutrophication, freshwater ecotoxicity and impacts related to human health (environmental mechanisms at the endpoint level) ▪ Compensation mechanisms not considered

Table 2.A.1. Main characteristics of midpoint and endpoint impact assessment methods related to freshwater use. (cont. III)

Approach	Method: Midpoint	Method: Endpoint	Accounting/ impact assessment	Intervention	Characterisation model	Notes
LCA-based	Pfister et al. (2009)		▪ Midpoint: water deprivation (units – m^3 eq.) ▪ Endpoint: damages to human health, ecosystem and resources (units – DALY, PDF.m^2.yr^{-1}, MJ, respectively)	Blue water	▪ Midpoint: WSI based on withdrawal-to availability ratios, developed at 0.5º grid cell, watershed and country level ▪ Endpoint: characterisation factors (CFs) developed at 0.5º grid cell, watershed and country level ▪ CFs for human health evaluate to agriculture freshwater use (not domestic freshwater and fisheries), considering scarcity and economic development levels. ▪ CFs for ecosystem consider freshwater scarcity and freshwater ecological value (through net primary production) ▪ CFs for resources evaluate freshwater use portion that contributes to depletion	▪ Degradative freshwater use not considered – intake and release freshwater quality are not taken into account ▪ Damage to resources: compensation mechanisms partially considered due to desalination technology ▪ Damage to human health and ecosystem: compensation mechanisms not considered
LCA-based	Boulay et al. (2011a, 2011b)		▪ Midpoint: water stress (units – m^3 eq.) ▪ Endpoint: impacts on human health caused by malnutrition and disease from water deprivation (units – DALY)	Blue water	▪ Midpoint: water stress indicator, at watershed and country level ▪ Endpoint: - CFs take into account the level of competition among users, addressing quality and seasonal variations of freshwater availability - CFs at watershed and country level	▪ Midpoint: - Intake and released freshwater quality considered through 137 quality parameters - Compensation mechanisms not considered ▪ Endpoint: - Damages to human health caused by malnutrition and diseases from freshwater deprivation for domestic, agriculture and fishery users - Gross national income classification (World Bank 2008) as adaptation capacity (mechanism compensation) of freshwater deficit for other users (Boulay et al. 2011a)
LCA-based	Motoshita et al. (2014)	Motoshita et al. (2010), Motoshita et al. (2014)	▪ Midpoint: agricultural production – crop production loss by irrigation (units – m^3 eq.) ▪ Endpoint: impacts on human health caused by undernourishment related to agricultural water scarcity (units – DALY)	Blue water	▪ Midpoint: CFs consider irrigated crop production vulnerability, physical vulnerability of freshwater resources and social compensation capacity ▪ Endpoint: CFs consider relationships between the supply shortage of a commodity, the human development index, and changes in the undernourished population rate caused by changes in average daily dietary supply ▪ CFs at country level	▪ Degradative freshwater use not taken into account ▪ Compensation mechanisms considered at both midpoint and endpoint level

Table 2.A.1. Main characteristics of midpoint and endpoint impact assessment methods related to freshwater use. (cont. IV)

Approach	Potential impact Midpoint	Potential impact Endpoint	Impact category	Intervention	Characterisation model	Notes
LCA-based	n.a.	Motoshita et al. (2010)	Damage to human health caused by infectious diseases from domestic freshwater use (units – DALY)	Blue water	▪ CFs developed based on non-linear multiple regression analysis (modelling relationships between freshwater scarcity, accessibility to safe water and damage to health caused by infectious diseases) ▪ CFs at country level	▪ Degradative freshwater use not taken into account ▪ Compensation mechanisms partially taken into account considering house connection rate to freshwater supply and sanitation
LCA-based	n.a.	Hanafiah et al. (2011)	Damage to fish species richness from freshwater consumption (units – PDF.m^3.yr^{-1})	Blue water	▪ CFs derived based on generic species-river discharge curve for 214 global river basins (Xenopoulos et al. 2005). They express the change in potentially disappeared faction of freshwater (PDF) fish species due to a change in river mouth discharge	▪ Degradative freshwater use not taken into account ▪ This method allows for comparing the impact of freshwater use with greenhouse gas emissions ▪ Compensation mechanisms not considered
LCA-based	n.a.	Van Zelm et al. (2011)	Impacts on the species richness of terrestrial vegetation caused by groundwater extraction (units – PNOF.m^2.yr^{-1})	Blue water	▪ CFs for groundwater (only available for Netherlands): - express the change in potentially not occurring fraction of plant species (PNOF) due to a change in extraction of groundwater. - Changes in groundwater levels were addressed by MODFLOW model (Facchi et al. 2004; Gedeon et al. 2007; McDonald and Harbaugh 1984) - the occurrence of plant species was predicted by the statistical MOVE model (Bakkenes et al. 2002)	▪ Degradative use not taken into account ▪ Only effects on renewable shallow (<3.58 m depth) groundwater are considered ▪ Compensation mechanisms not considered
LCA-based	n.a.	Verones et al. (2010)	Damage to ecosystem: thermal pollution in freshwater aquatic biota (units – PDF.m^3.day)	Blue water (cooling water discharges)	▪ CFs (available for nuclear power plant in Switzerland): - express the impact of cooling water discharges on aquatic ecosystems express the change in potentially disappeared faction (PDF) of aquatic species due to a change in river temperature - to estimate the river temperature profiles, the Qual2Kw model (Pelletier et al. 2006) was used - to address the direct temperature-induces mortality in species, a species sensitivity distribution following a normal temperature-response function was established	▪ The model compares the thermal emissions with ReCiPe method for the three areas of protection: human health, ecosystem and resources ▪ Compensation mechanisms not considered

n.a. not applicable

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2.2. Addressing the freshwater use of a Portuguese wine ('vinho verde') using different LCA-based methods

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Addressing the freshwater use of a Portuguese wine ('vinho verde') using different LCA methods

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Abstract

The increasing scarcity of freshwater in many parts of the world triggered a growing concern about freshwater use and its quality degradation. Currently, a number of methods to assess the potential environmental harm in ecosystems services derived from freshwater use are available under the framework of Life Cycle Assessment (LCA). In this study, the assessment of the quantitative freshwater use impact of a Portuguese wine (white 'vinho verde') was undertaken using the methods suggested by: Pfister et al. (2009), Frischknecht et al. (2009a,b), Ridoutt et al. (2010), and Milà i Canals et al. (2009). These methods differ significantly concerning the type of freshwater, freshwater scarcity level, and characterisation factors considered. The quantitative freshwater use of white 'vinho verde' considering both viticulture and wine production stages (disaggregated into foreground and background sub-systems) is analysed at the inventory and impact assessment levels. Moreover, the freshwater footprint profile i.e. the compilation of quantitative and degradative environmental impacts related to freshwater use is also evaluated. The inventory results of freshwater use obtained by the Milà i Canals et al. (2009) method differ significantly from the ones obtained by other used methods due to the consideration of land use effects. At the impact assessment level, a large variability for the freshwater use impact was obtained, mainly due to different characterisation factors considered by each method. Besides, the background sub- systems arise as the major hotspots for all methods other than that proposed by Milà i Canals et al. (2009) and for all degradative impact categories other than eutrophication.

Keywords: Life Cycle Assessment; freshwater use; water scarcity; white wine

1. Introduction

Human activity is a major contributor to the consumption and pollution of freshwater. According to the United Nations (UN 2011), the world population in 2011 was approximately 7 billion and, by 2050, is estimated to grow to roughly 9.3 billion. This predicted global population will lead to unsustainable use of natural resources, namely freshwater. Indeed, freshwater has already become a scarce and overexploited natural resource in many parts of the world, which has serious consequences for water related ecosystem services (UNEP 2007; WWAP 2009). Therefore, the quantification and characterisation of freshwater use (throughout this whole study, freshwater use respects to the consumptive freshwater use as defined in Table 2.1) is crucial to identify potential opportunities to reduce consumption and minimise freshwater scarcity problems. Over the last few years, several methods have been developed specifically for this purpose. For instance, the water footprint concept was first proposed by Hoekstra and Hung (2002) and described in greater detail by Chapagain and Hoekstra (2004), Chapagain et al. (2006), and Hoekstra et al. (2011). This method quantifies both direct and indirect volumetric freshwater use and pollution along supply chains, and has been used to calculate the water footprint of a wide range of agro-industrial products. For instance, the average water footprint has been estimated to be 40 L per slice of bread, 74 L per 250 mL of beer, 2,497 L per kg of rice, 3,178 L per kg of hard cheese (WFN 2010) and 2-13 L per A4 sheet of printing and writing paper (Van Oel and Hoekstra 2010).

More recently, different scientific methods have been developed to assess the freshwater use related impacts as an integral part of the Life Cycle Assessment (LCA) methodology (e.g. Bayart et al. 2010; Boulay et al. 2011a,b; Frischknecht et al. 2009a; Milà i Canals et al. 2009; Pfister et al. 2009; Ridoutt et al. 2010). These LCA-based methods have also been applied to agro-industrial products such as pasta sauce and peanut (Ridoutt et al. 2009), broccoli (Milà i Canals et al. 2010), asparagus and tomato (Frischknecht et al. 2006), among others. Due to the relatively recent development of these methods, the LCA community has been recommending their application in case studies in order to understand the individual significance of each one (Kounina et al. 2013). So far, the quantitative life cycle impacts related to freshwater use in the production of Portuguese white 'vinho verde' have never been analysed. However, Neto et al. (2013) had evaluated the degradative impact related to freshwater use associated with this wine by assessing the eutrophication, freshwater aquatic ecotoxicity, marine aquatic ecotoxicity and acidification impacts. The wine sector assumes a relevant role in Portugal, as the country is among the 12 leading countries for wine worldwide exportation. With an annual production of approximately 50 million litters (OIV 2010) in 2010-2011, the white 'vinho verde' is the white wine with the largest production share in Portugal. Moreover, 30 % of this wine production were exported (CVRVV 2011; IVV 2010). The white 'vinho verde' is produced the northwest of Portugal at the Demarcated Region of 'Vinho Verde', occupying an area of 34 thousand hectares.

The goals of this study are:

1) to assess the quantitative freshwater use associated with the production of a Portuguese white 'vinho verde' using the freshwater use LCA-based methods developed by Pfister et al. (2009) (from now on referred to as the Pfister method), Frischknecht et al. (2009a,b) (Ecological Scarcity Model (ESM) method, from now on referred to as the ESM method), Ridoutt et al. (2010) and Ridoutt and Pfister (2013) (from now on referred to as the Ridoutt method), and Milà i Canals et al. (2009) (from now on referred to as the Milà i Canals method);

2) to address strengths and constraints of each method;

3) to carry out the freshwater footprint profile, i.e. the compilation of quantitative and degradative environmental impacts related to freshwater use, throughout the white 'vinho verde' life cycle (from cradle to gate);

4) to identify the production stages and sub-systems that mostly contribute to the freshwater footprint profile.

This study relies on primary data provided by the largest producer of 'vinho verde' in Portugal, Aveleda S.A. In 2008-2009, this company produced nearly 25 % of the Portuguese white 'vinho verde' and exported worldwide around 50 % of its total production. The company is a medium-sized enterprise as defined in European Union law (European Union, 2003) with 170 employees and a turnover of almost 30 million euros in 2010.

2. Freshwater use LCA-based methods

This section presents the principles of each LCA-based method applied in the current study and a comparative analysis of these methods. The terminology and definitions used are presented in Table 2.1.

Table 2.1. Terminology and definitions used in the methods applied in this study.

Terminology		Definition
Type of water	Blue freshwater	Blue water includes surface and groundwater, i.e. the freshwater in lakes, rivers and aquifers (Hoekstra et al. 2011)
	Green freshwater	Green water is the precipitation on land that does not run off or recharge the groundwater but is stored in the soil or temporarily stays on the top of the soil or vegetation (Hoekstra et al. 2011)
Type of use	Consumptive use/evaporative use	Freshwater abstracted that evaporates, is transpirated by plants, embodied into the product, consumed by humans or livestock or discharge into different watershed/sub-watershed or seas (Bayart et al. 2010; Hoekstra et al. 2011)
	Non-evaporative use	The freshwater that returns to the original watershed and is available for another use purposes (Milà i Canals et al. 2009)
	Water abstracted	All the volume of blue freshwater abstraction from surface or groundwater, including the volume of this freshwater that is returned to the same catchment area where it was abstracted during the same period of time (Hoekstra et al. 2011)

2.1. Pfister method (Pfister et al. 2009)

The Pfister method assesses the environmental impact associated with the use of blue freshwater at midpoint and endpoint level. This method proposes water scarcity indexes (WSIs) estimated at the sub-watershed level as midpoint characterisation factors. The WSI is a modified withdrawal-to-availability ratio that accounts monthly and annual variability in precipitation influence. It is calculated using the WaterGAP2 global hydrology and global freshwater models (Alcamo et al. 2003; Marker et al. 2003). At the endpoint level, the impact assessment is carried out according to the Eco-indicator-99 method (Goedkoop and Spriensma 2001), which assesses the environmental damages on human health, ecosystem quality and resources depletion.

2.2. ESM method (Frischknecht et al. 2009a,b)

The ESM method was first described by Muller-Wenk (1978), but has been refined by Ahbe et al. (1990), Braunschweig (1982), Brand et al. (1998) and Frischknecht et al. (2009b). This method is based on the distance-to-target principle (i.e. comparison of existing emission flow and target flow) and provides eco-factors (characterisation factors) for multiple environmental impacts.

To assess the impact of blue freshwater use, the ESM method provides eco-factors at country level, which are calculated based on withdrawal-to-availability ratios (Frischknecht et al. 2009a,b). Unlike the Pfister method, it does not account for monthly and annual variability in precipitation and, consequently, may overestimate or underestimate freshwater scarcity. In this study, we used the eco-factor available for Portugal, 0.260 eco-points.L^{-1} of blue freshwater (Frischknecht et al. 2009a).

2.3. Ridoutt method (Ridoutt et al. 2010; Ridoutt and Pfister 2013)

The Ridoutt method assesses the environmental impact of the consumptive use of green and blue freshwater. Ridoutt et al. (2010) suggested that the use of green freshwater in land use activities, per se, does not necessarily leads to a reduction in surface water and groundwater or contribute to freshwater scarcity. As such, this method assesses the impact of green water immediately after it becomes blue freshwater (i.e. the impact of land use on blue freshwater resources). Evapotranspiration of natural vegetation after land occupation is first estimated and then compared to evapotranspiration of the crop under study. Ridoutt et al. (2010) used the method described by Zhang et al. (2001) to perform this calculation. The impact assessment is performed using midpoint characterisation factors that are the same as the regional WSI developed by Pfister et al. (2009) divided by the global average WSI as defined by Ridoutt and Pfister (2013).

2.4. Milà i Canals method (Milà i Canals et al. 2009)

The Milà i Canals method evaluates the influence of the blue and green freshwater use, as well as land use, on freshwater scarcity at the watershed level. At the inventory level, sources of freshwater are classified into 5 groups: soil moisture (green freshwater), river/ lake (blue surface freshwater), aquifer (blue ground freshwater), land use (changes in evapotranspiration and/or runoff from green freshwater in relation to a reference land use type), and fossil water (non-renewable blue ground freshwater). The use of river/lake, aquifer and fossil freshwater sources is divided into consumptive use (evaporative freshwater use) and non-consumptive use (non-evaporative freshwater use). The evaporative green freshwater component is only considered at the inventory phase since this method does not provide any Life Cycle Impact Assessment method to evaluate the impact of this component (Milà i Canals et al. 2009). Therefore, at impact assessment level, only consumptive use of blue freshwater and land use effects are assessed.

Two impact pathways are considered in the Milà i Canals method: the freshwater ecosystem impacts (FEI) of surface freshwater and the freshwater depletion (FD) of ground freshwater bodies. To assess FEI on a freshwater ecosystem at the watershed level, Milà i Canals et al. (2009) used different freshwater indexes, depending such as:

- the water scarcity indicator (WSI_{nd}), focused on freshwater resources available for human use and defined by the Eq.1:

$$WSI_{nd} = \frac{\text{freshwater abstracted}}{\text{freshwater resources} - \text{environmental freshwater requirements}} \qquad \text{(Eq.1)}$$

- the water use per resource (WUPR) indicator, defined by the percentage of water resources that are being withdrawn from natural water systems and their availability (Eq.2).

$$\text{WUPR (\%)} = \frac{\text{freshwater abstracted}}{\text{freshwater resources}} \times 100 \qquad \text{(Eq.2)}$$

It should be noted that both characterisation factors do not take into account the monthly and annual variability in precipitation. In this study, we used a WUPR characterisation factor of 0.164 as suggested by Milà i Canals (2009).

For the FD index, Milà i Canals et al. (2009) adapted the method originally proposed by Guinée et al. (2002) to calculate the Abiotic Depletion Potential (ADP) of aquifer and fossil freshwater.

2.5. Comparative analysis of methods

The strengths and constraints of the LCA-based methods applied in this study are summarised in Table 2.2. Specifically, the various methods differ significantly concerning the type of freshwater assessed, freshwater scarcity level and characterisation factors. With regard to the type of freshwater considered, the Pfister and ESM methods only assess the blue freshwater impact. Besides blue freshwater, the Ridoutt method also evaluates the land use effects. The Milà i Canals method assesses the impact of blue and land use effects, whereas the impact of evaporative green freshwater use is disregarded because, according to Milà i Canals et al. (2009), there is currently no LCA-based method to evaluate the impact of this component.

The Ridoutt method suggests to estimate land use effects on freshwater availability as the difference between the evapotranspiration from the crop under analysis and the evapotranspiration of a reference land use type. The question of what consider as reference land use remains (Chiarucci et al. 2010; Hickler et al. 2012; Loidi et al. 2010). As referred in Section 3.3.1, the Ridoutt method considers herbaceous vegetation as reference land use whereas the Milà i Canals method considers forest land use as reference land use. Further research is needed to give scientific consistence to this procedure. The WSI (used in Pfister and Ridoutt methods) is applied at the sub-watershed level, whereas the WSI_{nd} and WUPR characterisation factors of the Milà i Canals method are applied at the watershed level.

These characterisation factors are calculated based on withdrawal-to-availability ratios rather than on freshwater use-to-availability ratios. Moreover, while WSI takes into account the precipitation variability, whereas WSI_{nd} and WUPR and eco-factor do not consider this aspect.

The freshwater eco-factors of the ESM method are calculated taking into account the total freshwater abstracted (current flow), withdrawal-to-availability ratios, and the critical freshwater flow of available resources as explained by Frischknecht et al. (2009).

Table 2.2. Strengths and constraints of the quantitative freshwater use in LCA-based method.

Method	Strengths	Constraints
Pfister	+ The regional blue freshwater scarcity level is assessed, especially at the sub-watershed level + The WSI characterisation factor considers the monthly and annually variability on precipitation + Easy to be used by stakeholders	- The WSI characterisation factor is calculated based on withdrawal-to-availability ratios rather than of consumption-to-availability ratios of freshwater at the sub-watershed level
Ridoutt	+ Evaluates the green (impact of land on blue freshwater resources) and blue freshwater impact + It assesses the freshwater scarcity level at the sub-watershed level + The WSI characterisation factor considers the monthly and annually variability on precipitation + It is a single indicator approach in a similar way to the carbon footprint. It is possible the comparison between the impacts of freshwater use of the same product produced and consumed in different countries + Easy to be used by the stakeholders	- The WSI characterisation factor is calculated based on withdrawal-to-availability ratios rather than of consumption-to-availability ratios of freshwater at the sub-watershed level
ESM	+ The freshwater scarcity level is assessed at country and watershed level, depending on available data	- It does not specify the different effects of freshwater abstractions from different sources of freshwater (groundwater, surface water), including temporal aspects - The eco-factors are calculated based on the abstracted freshwater and on environmental political target values
Milà i Canals	+ Evaluates the green, blue freshwater, and land use effects on freshwater resources + It specifies the different sources of freshwater (soil moisture, river, aquifer, fossil water) + The inventory distinguishes between the consumptive use of freshwater (evaporative freshwater use) and the non-consumptive use of freshwater (non-evaporative freshwater use)	- The environmental impact assessment is performed at the watershed freshwater scarcity level, i.e. less specific freshwater scarcity level than the one considered by the Pfister and Ridoutt methods - The WSI_{nd} and WUPR are based on the abstracted freshwater - It does not provide regional characterisation factor for FD

3. Scope, inventory and impact assessment

This section presents the functional unit (FU), describes the system boundary and explains how inventory data, impact assessment and water footprint profile were obtained.

3.1. Functional unit

In the current study, the FU is defined as the volume of one bottle of white 'vinho verde' with a total liquid volume of 0.75 L.

3.2. Description of the system

The system boundary, schematically presented in Fig. 2.2, includes two stages: the viticulture processes at the vineyard level and the wine production processes at the winemaking plant.

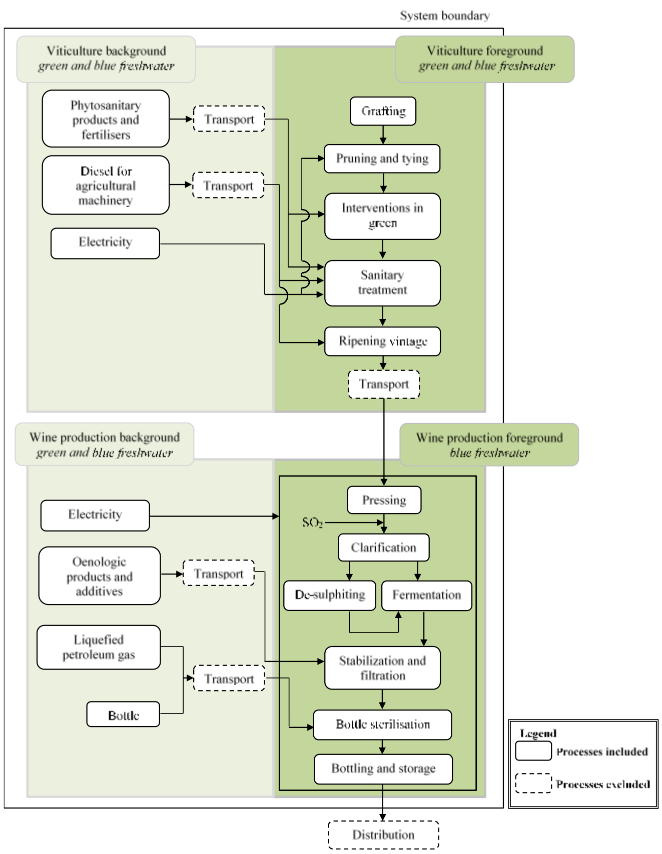

Fig. 2.2. System boundary of the white 'vinho verde'.

The viticulture foreground sub-system comprises multiple stages. The first one is grafting, in which tissues from a plant are inserted into tissues of another plant. Pruning is a manual operation in which is used electric pruning shears to cut a portion of the vine branches, as well as a simultaneous operation called typing that consists of folding the branch and ties it to a cord. Interventions in green correspond to a number of manual and mechanical operations carried out during the vegetation stage. The removal of sprouts is a manual operation that burst out of the pretended places in the vine. Topping and defoliation are mechanical operations that consist on cutting of the ends of vegetation, and leaves and young branches unnecessary to enable ripening clusters, respectively. During the vegetative growth, the fertilisers (ammonia nitrate, ammonia sulphate and urea ammonia nitrate) are manually applied in the vineyard. Another important operation is the mechanical sanitary treatment, whose objective is to protect vines from diseases or pests such as mildewor red spider mite. The electricity consumption during the sanitary treatment is related to the pumping of the freshwater that is used to prepare the aqueous solution of phytosanitary products and to wash tanks at the end of this operation. When the grapes are ripe, the vintage is done mainly by mechanical means, and grapes are transported to the winemaking plant.

The wine production foreground sub-system involves three main phases: vinification (includes pressing and clarification, de-sulphiting and alcoholic fermentation), conservation and preparation of lots (comprises stabilization and filtration), and bottling and storage (includes bottle sterilisation). During vinification, grapes are pressed to extract the must (unfermented liquid obtained from grapes). After pressing, the must is sent to vats where sulphur dioxide, which acts as an antioxidant agent, is added. The next operation, clarification, consists in the removal of lower granulometric particles from the must.

With amounts of sulphite higher than 1000 ppm, the must cannot undergo alcoholic fermentation. To avoid this, the clarified must is subjected to a de-sulphiting process, consisting in the removal of sulphur dioxide from the unfermented must at temperatures around 110-120 ºC. After this, the de-sulphited must follows to the alcoholic fermentation process, where the must is transformed into wine due to the action of the yeasts.

The stabilisation process consists in the addition of oenologic substances to make the wine stable and improve its sensory characteristics. The removal of lees resulting from the stabilization process is carried out by filtration. Meanwhile, the bottles are sterilised. After this process the wine is bottled and stored, being ready for sale.

The production of ancillary materials was considered in the viticulture and wine production background sub-systems. However, all the transportation operations as well as the production of cork stoppers, labs and caps have been excluded from the system boundaries.

3.3. Inventory of freshwater use

Data on freshwater use from viticulture and wine production foreground sub-systems were collected based on direct measurements (primary data) from Aveleda, S.A. Input and output data related to the degradative quality of the used freshwater were collected by Neto et al. (2013). All data were considered to be representative of the entire Portuguese sector of white 'vinho verde'.

Sections 3.3.1 and 3.3.2 present the details about how blue and green freshwater components and land use effects on freshwater were quantified.

For both viticulture and wine production background sub-systems (Fig. 2.2), data of freshwater use were taken from the GaBi professional database (PE International 2012).

3.3.1. Viticulture foreground sub-system

Blue freshwater component

In Portugal, the average precipitation during the 2008–2009 winery campaign was 641 $L.m^{-2}.yr^{-1}$ (IM 2009), whereas the average annual precipitation at the Demarcated Region of 'vinho verde' was approximately 1,500 $L.m^{-2}.yr^{-1}$ (CVRVV 2011), which is considerably higher than the national average. Even the dry months (July and August) had an average monthly precipitation just below 30 $L.m^{-2}$ (CVRVV 2011), which was a sufficient amount to maintain vineyards productivity. Since no irrigation systems were present, the blue freshwater component of the viticulture foreground sub-system only included the amount of blue freshwater used for spraying during phytosanitary treatments. It was assumed that this amount of freshwater completely evaporates into the atmosphere.

Green freshwater component and land use effects on freshwater availability

The total green freshwater of the viticulture foreground sub-system was obtained by determining the total amount of green freshwater evapotranspiration that occurred throughout the entire growing period of the grapes, plus the green freshwater fraction incorporated into the harvested grapes.

The green freshwater evapotranspiration was calculated using the CROPWAT 8.0 model and the irrigation schedule as ‘no irrigation (rain-fed)’ (FAO 2009). This option included a soil water balance, which kept track of soil moisture content over time using a daily time step. Input data for the model included information on climate factors, red sandy loam soil properties and vine characteristics.

The climate data was obtained from the New_LocClim model (FAO 2005). Using the nearest neighbour meteorological stations interpolation method, this model provides estimations of average climate conditions at locations for which no observations are available (applicable condition to the present study).

The CROPWAT requires data about the crop characteristics, such as rooting depth, crop height and length of grape development stages, which were provided by the wine company. Crop coefficients during the growing period of the vine ($K_{c,\ ini}$ for the initial stage, $K_{c,\ mid}$ for the mid-season stage, and $K_{c,\ end}$ for the end of the late season stage) and a critical depletion value (p) were obtained from Allen et al. (1998) for grapes in general given that process-specific data were not readily available. The crop coefficient is the ratio of crop evapotranspiration to reference (grass) evapotranspiration. Critical depletion is the fraction of the total available soil freshwater that can be depleted from the root zone before freshwater scarcity occurs.

The green freshwater fraction incorporated into the harvested grapes was estimated based on a moisture content of 75 % in the grapes. This information was provided by the wine company.

The Milà i Canals method requires the calculation of the land use effects on freshwater availability, defined as the change in rainwater availability for land infiltration and runoff in relation to forest (reference land use). In the absence of specific data for vines, both data from ‘lost’ percentage of precipitation of arable non-irrigated land in Europe (73 %) and ‘lost’ percentage of precipitation of forested land (67 %) (reference land use) were used, as suggested by Milà i Canals et al. (2009).

The Ridoutt method also requires the calculation of the impact of land use on blue freshwater resources (green freshwater that is accessible only through land occupation) in relation to herbaceous vegetation (reference land use). The green freshwater evapotranspirated from this vegetation (ET, in mm per year) was calculated using Eq. 3 (Zhang et al. 2001), as suggested in the Ridoutt method.

$$ET = \left(\frac{1 + 0.5\ \frac{1{,}100}{P}}{1 + 0.5\ \frac{1{,}100}{P} + \frac{P}{1{,}100}} \right) P \qquad \text{(Eq.3)}$$

where, P is the average precipitation during the 2008-2009 winery campaign in the Demarcated Region of 'vinho verde' (1,500 $L.m^{-2}.yr^{-1}$).

3.3.2. Wine production foreground sub-system

Blue freshwater

The wine production foreground sub-system only considers the evaporation component of blue freshwater, which occurs during de-sulphiting and bottle sterilisation, since there is no incorporation of freshwater into the wine.

3.4. Impact assessment and freshwater footprint profile

The impact assessment of blue freshwater and land use effects on freshwater availability of each production process was assessed by multiplying the volume of blue freshwater and land use effects collected at the inventory level by the characterisation factors of each applied LCA-based method, as described in Section 2.

A freshwater footprint profile (compilation of quantitative use and degradative environmental impacts related to freshwater) was performed. This range of impacts related with freshwater allows the identification of the main hotspots, being thereby the first step for identifying and planning actions to decrease freshwater use related impacts and to improve freshwater use efficiency.

The degradative component was obtained from Neto et al. (2013) that carried out a conventional LCA for the Portuguese white 'vinho verde' under analysis. The degradative impact assessment was carried out according to the CML 2001 methodology considering the same system boundary and FU of the current study. Eutrophication (EP), freshwater aquatic ecotoxicity (FE) and marine aquatic ecotoxicity (ME) were the considered impact categories related to freshwater. The freshwater use (WU) impact was calculated accordingly to the four freshwater use LCA-based methods analysed in this study.

4. Results and discussion

In this section, the inventory and impact assessment results are presented and discussed in Section 4.1 and 4.2, respectively. The freshwater footprint profile is presented in Section 4.3.

4.1. Results of freshwater use inventory

Table 2.3 presents the freshwater use results at the inventory level for all the LCA-based methods considered in this study, expressed in L per FU, disaggregated per stage and per background and foreground sub-systems.

The total blue freshwater use calculated at the inventory level is 4.6 $L.FU^{-1}$ in the four methods. The wine production stage accounts for 75 % of this freshwater, being the majority consumed by the background sub-system. This contribution is mainly due to production of the energy carriers (electricity and liquefied petroleum gas). In the viticulture stage, which is responsible for 25 % of the total blue freshwater use, the background sub-system also presents the largest contribution (79 %) due to the production of the energy carriers (diesel and electricity). Although not being a hotspot, the foreground sub-system of the wine production stage is already implementing measures to reduce blue freshwater use by preventing water losses during bottle washing, as well as during de-sulphiting and bottle sterilisation processes.

Viticulture foreground sub-system has a minor contribution (5 %) to the total blue freshwater used since the crop field is not irrigated. This minor contribution is only due to the phytosanitaries treatments of the vineyard.

In addition to blue freshwater, the Ridoutt method also considers the land use effects, in which the vine consumes a smaller proportion of the precipitation (20 %) than the herbaceous vegetation (50 %). Therefore, a negative difference of 35.0 L of green freshwater evapotranspirated was determined.

According to the Milà i Canals method, which besides blue freshwater, also accounts for green freshwater and land use effects, the total freshwater use is 511.3 $L.FU^{-1}$. About 78 % of this use is green freshwater, 21 % is due to land use effects, and only 1 % is blue freshwater. For an average annual precipitation of 1,500 $L.m^{-2}.yr^{-1}$ in the Demarcated Region of 'vinho verde' (CVRVV 2011), the land use effects resulting from the land occupation by the viticulture was estimated to be 90 $L.m^{-2}.yr^{-1}$. This means that the vine land uses a larger fraction of green freshwater than the forest ecosystem that it replaced (i.e. for every m^2 of land occupied over 1 year for viticulture, 90 L less freshwater is available compared to the reference situation).

The viticulture foreground sub-system arises as a hotspot as it uses more than 95 % of the total freshwater mainly due to green freshwater use associated with the grape growing.

The results of freshwater use inventory obtained under this study were compared with results of other wine studies. Mekonnen and Hoekstra (2011) estimated that the global average of green freshwater use in the viticulture stage is 607 $L.FU^{-1}$, which is significantly higher than the 396.9 $L.FU^{-1}$ calculated in the present study. However, it should be noted that Mekonnen and Hoekstra (2011) do not provide information about the type of wine studied neither the defined system boundary nor the edaphoclimatic conditions of the agricultural regions, which may considerably affect the green freshwater use.

Ene et al. (2013) reported a green and blue freshwater use in the viticulture stage of a Romanian wine of 1,133 $L.FU^{-1}$ and 40 $L.FU^{-1}$, respectively. Concerning the wine production, a blue freshwater use of 2 $L.FU^{-1}$ has been reported. The green freshwater use in the viticulture stage of the Romanian wine is significantly higher than that calculated in the present study, which can be explained by the different edaphoclimatic conditions in the two regions. For instance, in the Romanian region where viticulture takes place, soil has a clay loam texture, i.e. higher clay content soil in the Demarcated Region of ‘vinho verde’ (Ailincai et al. 2011), which results in a higher water retention capacity (Calik et al. 2004). Also, the effective precipitation is higher in the Romanian study (396 $L.m^{-2}.yr^{-1}$) than that in the current study (289 $L.m^{-2}.yr^{-1}$). With regard to the wine production stage is unclear whether Ene et al. (2013) considers only the wine production foreground sub-system or the full wine production stage (foreground plus background sub-systems).

Herath et al. (2013) found that the green freshwater use by the viticulture foreground sub-system in two regions of New Zealande – Marlborough and Gisborne – was 611 $L.FU^{-1}$ and 601 $L.FU^{-1}$, respectively, whereas the blue freshwater use by the viticulture foreground sub-system in these regions was 70.6 $L.FU^{-1}$ and 3.6 $L.FU^{-1}$. The viticulture foreground sub-system was determined to be mainly responsible for the total freshwater use in Marlborough and Gisborne regions with 82 % and 90 %, respectively. In these regions, the green and blue freshwater use in the viticulture foreground sub-system is higher than the calculated in the current study. This higher green freshwater use may be due to different vine characteristics and edaphoclimatic conditions. The Marlborough vineyards are irrigated, which does not occur in the Portuguese vineyards under study. Despite the Gisborne vineyards are not irrigated, the higher blue freshwater use can be explained by the use of different agrichemicals and different profile of application of these products requiring higher freshwater use, as well as by the use of different energy sources.

Concerning to the wine production stage, Herath et al. (2013) reported a blue freshwater use of 17.2 and 19.2 $L.FU^{-1}$ for Marlborough and Gisborne, respectively. These values are significantly higher than those obtained in the current study, mainly due to higher freshwater use in the New Zealander wine production background sub-system.

The land use effect on freshwater use is not addressed by any of the above mentioned studies on wine. According to the Milà i Canals method, the ET from arable non-irrigated land is higher than the ET from forested land (reference land use). However, other studies show that forests have higher ET than agricultural and grassed land use (Farley et al. 2005; Jackson et al. 2005; Nosetto et al. 2011; Zhang et al. 2001). This highlights the need of further research on this subject.

Table 2.3. Inventory results of freshwater use expressed as L of freshwater per FU.

Stage	Sub-system	Pfister and ESM methods	Ridoutt method			Milà i Canals method			
		Blue freshwater	Blue freshwater	Land use	Total	Green freshwater	Blue freshwater	Land use	Total
Viticulture	Foreground	0.2	0.2	(-35.0)*	0.2	385.4	0.2	109.1	494.8
	Background	0.9	0.9	n/a	0.9	11.5	0.9	n/a	12.4
	Energy carriers	0.8	0.8	--	0.8	11.4	0.8	--	12.2
	Phytosanitary products and fertilisers	0.1	0.1	--	0.1	0.04	0.1	--	0.04
	Total (background + foreground)	1.1	1.1	(-35.0)*	1.1	396.9	1.1	109.1	507.0
Wine production	Foreground	0.1	0.1	n/ap	0.1	0.00	0.1	n/ap	0.1
	De-sulphiting	0.07	0.07	--	0.07	0.00	0.07	--	0.07
	Bottle sterilisation	0.06	0.06	--	0.06	0.00	0.06	--	0.06
	Background	3.4	3.4	n/ap	3.4	0.8	3.4	n/ap	4.1
	Energy carriers	2.1	2.1	--	2.1	0.4	2.1	--	2.4
	Oenologic products and additives	0.09	0.09	--	0.09	0.008	0.09	--	0.09
	Bottle	1.2	1.2	--	1.2	0.39	1.2	--	1.6
	Total (background + foreground)	3.5	3.5	n/ap	3.5	0.8	3.5	0.0	4.3
Total		4.6	4.6	(-35.0)*	4.6	397.7	4.6	109.1	511.3

*as land use effects are less than zero, they are disregarded as recommended by Ridoutt et al. (2010) method

n/a: no available data

n/ap: not applicable

4.2. Results of freshwater use impact

The results obtained for the freshwater use impacts from the application of the different LCA-based methods are presented in Table 2.4. The results obtained with the Pfister, ESM and Ridoutt methods are different although they rely in the same inventory results.

Based on the Pfister method, the impact associated with blue freshwater use was 1.0 Leq, with the viticulture being responsible for 56 % of the total impact. With the ESM method, blue freshwater use was 1.20 eco-points. In this case, the wine production stage was the most relevant, being responsible for 76 % of the total freshwater use impact.

The relative contributions to the freshwater use impact obtained with the Ridoutt method are similar to those obtained with the Pfister method, although the total freshwater use impact is higher. Although the Ridoutt method considers the land use effects at the inventory level, they were considered to have no impact on freshwater resource availability, as recommended by Ridoutt et al. (2010), because they were smaller than zero.

Following the Milà i Canals method, the viticulture foreground sub-system is responsible for more than 95 % of the total freshwater use impact due to the contribution of the land use effects on freshwater impact. The FEI calculated with this method was the highest compared with the other methods (18.7 Leq), although the total blue freshwater use impact is the smallest one (0.8 Leq). This method also considers the land use effects, which was responsible for nearly 100 % of the total freshwater use impact.

The blue freshwater impacts of wine production stage calculated by the Milà i Canals method are in agreement with those obtained by Herath et al. (2013). The wine production background sub-system is the main contributor to the total freshwater use impact of Gisborne wine (0.45 Leq) with 57 %, which is also in line with our study, whereas in the wine produced in Marlborough, the viticulture foreground sub-system is the main contributor to the total freshwater use impact (1.17 Leq) with 67 %. This can be explained because Marlborough vineyards are full irrigated while Gisborne vineyards are non-irrigated.

Fig. 2.3 shows the relative contributions to the freshwater use impact of each unit process/system considered in the white 'vinho verde' life cycle. Regarding the Pfister and Ridoutt methods, the energy carriers from viticulture background sub-system are the unit processes with the largest relative contribution to the overall freshwater use impact (40 %), while the energy carriers from the wine production background sub-system appears as the second largest contributors (38 %). Production of phytosanitaries and synthetic fertilisers accounts for 7 % of total freshwater use impact.

Concerning the ESM method, the energy carriers of the wine production background sub-system are responsible for more than 40 % of the total freshwater use impact. Bottle production appears as the second major contributor to the freshwater use impact with 26 %, whereas the energy carriers of viticulture background sub-system contributes with 17 % to the total freshwater use impact.

As the background sub-systems arise as the major hotspot for all methods other than Milà i Canals, it is important to widely include the freshwater use in LCA databases to facilitate the Life Cycle Inventory, avoid duplication in data compilation and allow a reliable and representative freshwater impact assessment of products.

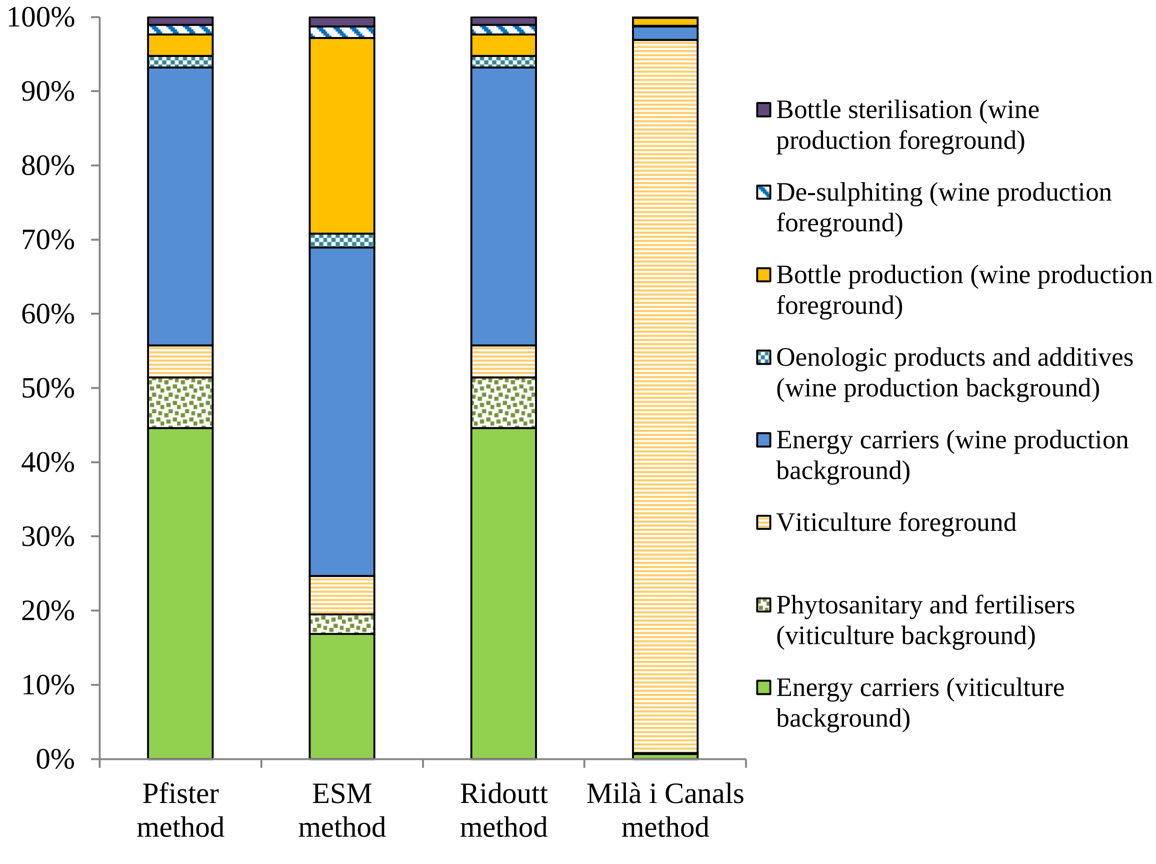

Fig. 2.3. Relative contributions to freshwater use impact of each unit process and sub-system considered in the white 'vinho verde' life cycle.

Table 2.4. Impact assessment of freshwater use expressed per FU.

Stage	Sub-system	Pfister method (L eq)		ESM method (eco-points)		Ridoutt method (L eq)		Milà i Canals method (L eq)		
		Blue water	Total	Blue water	Total	Blue water	Total	Blue water	Land use effects	Total
Viticulture	Foreground	0.04	0.04 (4 %)	0.06	0.06 (5 %)	0.07	0.07 (4 %)	0.04	17.9	17.9 (96 %)
	Background	0.5	0.5 (52 %)	0.2	0.2 (19 %)	0.8	0.8 (51 %)	0.2	n/a	0.2 (1 %)
	Total	0.5	0.5 (56 %)	0.3	0.3 (24 %)	0.9	0.9 (55 %)	0.2	17.9	20.1 (97 %)
Wine Production	Foreground	0.02	0.02 (2 %)	0.03	0.03 (3 %)	0.02	0.02 (3 %)	0.02	n/ap	0.02 (0 %)
	Background	0.4	0.4 (42 %)	0.9	0.9 (73 %)	0.7	0.7 (42 %)	0.6	n/ap	0.6 (3 %)
	Total	0.5	0.5 (44 %)	0.9	0.9 (76 %)	0.7	0.7 (45 %)	0.6	n/ap	0.6 (3 %)
Total		1.0	1.0	1.2	1.2	1.6	1.6	0.8	17.9	18.7

n/a: no available data

n/ap: not applicable

4.3. Freshwater footprint profile

The relative importance of freshwater use impacts and the degradative environmental impacts related to freshwater – EP, FE and ME impact categories – is illustrated in Fig. 2.4.

The viticulture stage is the major contributor to the freshwater use impact methods, except for the ESM method, for which the major contributor is the wine production stage.

As presented in Section 4.2, the viticulture stage is also the main hotspot in the degradative impacts contributing with 92, 72 and 68 % to EP, FE and ME, respectively. The viticulture foreground sub-system is particularly important to EP, with a contribution higher than 80% due to emission of nitrate resulting from the use of fertilisers, whereas the viticulture background is the sub-system that most contributes to FE and ME with 58 and 60 % of the total impacts, respectively. Barium and polyaromatic hydrocarbons emitted during the production of diesel, as well as formaldehyde and zinc emitted during the production of phytosanitary products, explain this high percentage in the FE impact. The impacts related with ME are mainly caused by transition metals (e.g. vanadium and nickel) used in the production of phytosanitary products and electricity. The wine production stage has contributions to the total impacts varying from 8 % in EP to 32 % in ME.

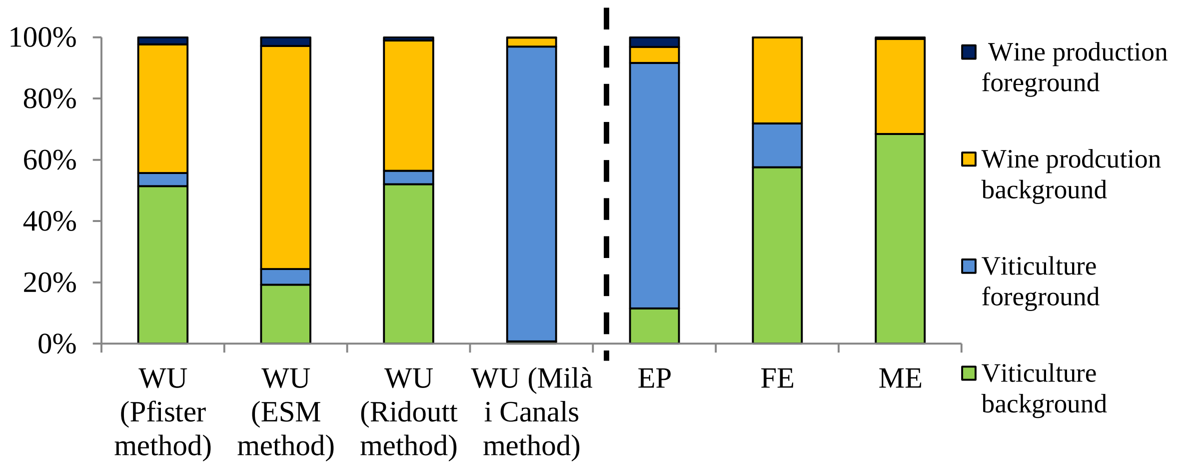

Fig. 2.4. Freshwater footprint profile of the Portuguese white 'vinho verde'. WU acronym: freshwater use.

4.4. The suitability of the LCA-based methods as a sustainability instrument

Despite the different results obtained, all of the LCA-based methods offer relevant insights into the environmental impacts associated with freshwater use and help to identify hotspots in a product life cycle. The various methods could even be used comparatively in a research-oriented LCA study. However, the ESM should be applied with caution. Although the water use results are comparable to other LCA methods at the Life Cycle Inventory stage, the same does not occur at the Life Cycle Impact Assessment stage due to the use of eco-factors as characterisation factors (in eco-points units).

The Milà i Canals method should also be applied with caution as the higher evapotranspiration values for non-irrigated arable land compared to forest land would not appear to be reasonable in many cases and, as found in our case study of wine production, the land use aspect can dominate the inventory results when using this method.

One of the motivations for freshwater use assessment in the wine industry is to answer to consumer and stakeholder demand for greater transparency and reporting of environmental impact. For this purpose, the Ridoutt method is preferred as the results are reported in units which are likely to be most understandable for a non-technical audience (i.e. relative to an equivalent volume of consumptive freshwater use at the global average WSI). By this approach, indicator results can be smaller or larger than the inventory results depending on whether water is consumed in locations which have lesser or greater water scarcity compared to the global average. Other methods, such as the Pfister method, which involve multiplying the consumptive water use inventory result by a local WSI ranging from 0.01 to 1 lead to indicator results which are smaller than the inventory results and this could be considered misleading. The only exception is where freshwater is used in a location of maximum water scarcity index (i.e. WSI=1) in which case the indicator result is the same as the inventory results. In effect, the reference substance for such methods is a volume of water consumed at a location of highest possible water scarcity.

5. Conclusions

This study contributes to the on-going debate about the methods to be used to assess the impact derived from freshwater use under the LCA framework, by applying several available methods to the case study of a Portuguese wine (white ‘vinho verde’). The methods differ significantly concerning the type of freshwater, freshwater scarcity level and characterisation factors considered. Therefore, they lead to different impacts for the same FU: 1.0, 1.6 and 18.7 Leq, following the Pfister, Ridoutt and Milà i Canals methods, respectively, and 1.2 eco-points with the ESM method.

Another objective of this study was to identify the production stages and sub-systems that mostly contribute to the freshwater footprint profile. These hotspots also differed depending on the freshwater use method and impact category. The Pfister and Ridoutt methods, as well as the FE and ME impact categories, identified the viticulture background sub-system as the major hotspot. On the other hand, the ESM method identified the wine production background sub-system as the major hotspot, whereas viticulture foreground sub-system appeared as the major hotspot using the Milà i Canals method and for EP impact category. The relevant contribution of these background sub-systems indicates that it is important to widely include the freshwater use into LCA databases to facilitate the Life Cycle Inventory, avoid duplication in data compilation and allow a reliable and representative freshwater footprint profile of products. Moreover, further research is needed to identify and plan actions to reduce both quantitative and degradative freshwater use impacts for the identified hotspots.

Given the large variability obtained in this case study for the freshwater use impact, we conclude that is important to define internationally how to consider the freshwater use types at inventory and impact assessment level in LCA-based methods. In particular further research is needed to develop characterisation factors based on consumption-to-availability ratios and to agree on how to consider land use effects on freshwater availability.

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2.3. A contribution to the environmental impact assessment of green water flows

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A contribution to the environmental impact assessment of green water flows

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Abstract

In recent years, numerous efforts have been made to include water-related issues in Life Cycle Assessment (LCA) methodology. This study provides an overview of existing methods that address green water use in LCA. In this overview, we analyse the main features of existing LCA-based methods used to examine changes in long-term blue water availability caused by variations in green water flows, particularly with respect to inventory, the characterisation model and characterisation factors.

Moreover, we propose a method of assessing impacts on terrestrial green water flows (TGWI) and addressing reductions in surface blue water production (RBWP) caused by reductions in surface runoff due to land-use production systems. Both TGWI and RBWP are analysed, taking into account the green water use/atmosphere and green water use/soil interfaces.

In this proposed method, the Life Cycle Inventory (LCI) phase considers the net green water flow that leaves the land-use production system, allowing the study of two alternative reference land uses: 1) quasi-natural forest and 2) grasslands/shrublands. In the Life Cycle Impact Assessment (LCIA) phase, regional- and species-specific characterisation factors (CFs) for the amount of green water evaporated or transpired are also proposed.

To illustrate the applicability of the proposed method, we employed a case study on *Eucalyptus globulus* stands (first rotation), located in Portugal. The results show that

different impacts on terrestrial green water flows and on surface blue water production are obtained depending on the alternative reference land use. Moreover, the case study shows that the method developed can be a useful tool assisting in improved national *E. globulus* forest planning.

Keywords: characterisation factors (CFs), *Eucalyptus globulus*, green water flow, Life Cycle Assessment (LCA)

1. Introduction

Anthropogenic activities associated with land-use changes and current climate change trends have been increasing pressure on freshwater natural resources. Numerous efforts have been undertaken to include water-related issues in Life Cycle Assessment (LCA) methodology. Over the last five years, consideration of water use in LCA has progressed rapidly, resulting in a complex set of methods for addressing different water types and sources, pathways and characterisation models at midpoint and endpoint levels, and with different spatial and temporal scales (Kounina et al. 2013; Tendall et al. 2013).

Studies have been carried out related to water abstraction and human appropriation (e.g. Boulay et al. 2011; Milà i Canals et al. 2009; Núñez et al. 2012; Pfister et al. 2009; Ridoutt et al. 2010), the potential impacts of blue water use on ecosystems (e.g. Hanafiah et al. 2011; Pfister et al. 2009; Tendall et al. 2014; Van Zelm et al. 2011; Verones et al. 2013) and water pollution, related to the discharge of eutrophying, acidifying and ecotoxic compounds into freshwater systems (e.g. Azevedo et al. 2013; Goedkoop et al. 2013; Helmes et al. 2012; Seppala et al. 2004; Struijs et al. 2011). However, less attention has been paid to green water use and green water flows. Green water use refers to 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. It also refers to the rainwater incorporated into harvested crops or wood (Hoekstra et al. 2011). Green water flow refers to the portions of green water used by soil and vegetation that is evaporated or transpired.

Both green water use/soil and green water use/atmosphere interfaces should be considered in assessing the potential environmental effects resulting from changes to green water flows due to land use. The methods developed for assessing green water flows have been more concerned with the interface between green water use/soil, i.e. how a change in green water affects the regional long-term availability of surface blue water (Fig. 2.5) (Milà i Canals et al. 2009; Núñez et al. 2012; Ridoutt et al. 2010). The water use/atmosphere interface assumes particular relevance because terrestrial evapotranspiration (ET) has been identified as a significant source of precipitation for land-use production systems (Ellison et al. 2012; Trenberth 1999; Van der Ent et al. 2010). Recently, Berger et al. (2014) have taken the first steps in considering the water use/atmosphere interface in LCA, i.e. how land use affects the ET that is recycled to the atmosphere and then the precipitation that returns to the regional terrestrial ecosystem (Fig. 2.5). These authors examine the atmospheric evaporation recycling within watersheds and analyse their vulnerability to water depletion.

Human-induced vegetation can significantly change the volume of water that is evaporated or transpired into the atmosphere in comparison to potential natural vegetation (PNV) (Ridoutt et al. 2010; Rost et al. 2008; Scanlon et al. 2007). Precipitation depends on the evaporation from oceans and recycled moisture from terrestrial surfaces. Van der Ent et al. (2010) have demonstrated that 40 % of global terrestrial precipitation on average originates from terrestrial surfaces and 60 % comes from oceans. Moreover, these authors suggest that 57 % of all terrestrial evaporation returns as precipitation to land surfaces. The external forcing and climatic parameters, such as solar radiation, aerosols and greenhouse gases, affect sea surface temperature and therefore influence ocean evaporation. On the other hand, terrestrial evaporation is strongly influenced by climatic parameters (e.g. precipitation, air temperature, daily solar radiation and relative humidity), as well as by nonclimatic parameters for soil and vegetation (e.g. soil-root zone water holding capacity, canopy conductance and leaf area index). The growth and resilience of vegetation is largely precipitation-dependent and the recycled moisture contributes to regulation of the hydrological cycle (Foley et al. 2003; Jung et al. 2010) as well as to regulation of biomass/food production (Falkenmark and Rockström 2004; Rockström et al. 1999). For instance, deforestation reduces the surface roughness and leaf area, which in turn limits the green water flows recycled into the atmosphere, thereby contributing to a decrease in precipitation levels (Pielke et al. 2006; Van Dijk and Keenan 2007).

In this study, an overview of methods addressing ET is conducted in order to understand how green water flows have been and should be considered in LCA. Furthermore, we propose a method for assessing impacts on terrestrial green water flows (TGWI) and addressing reductions in surface blue water (river) production (RBWP) caused by reductions in surface runoff due to a land-use production system. This method encompasses the Life Cycle Inventory (LCI), which accounts for green water flows recycling into the atmosphere due to specific types of land use, and proposes regional- and species-specific characterisation factors (CFs) in the Life Cycle Impact Assessment (LCIA) phase.

The applicability of the proposed method is illustrated by using *Eucalyptus globulus* stands located in Portugal. *E. globulus* is one of the dominant species in the Portuguese forest and it covers approximately 812 thousand hectares, accounting for approximately 26 % of the total forest area in Portugal (ICNF 2013). Most of the *E. globulus* wood is consumed in the pulp and paper industry.

1.1. Overview of LCA-based methods addressing green water

Fig. 2.5 illustrates the impacts on ecosystems and the effects on natural resources caused by changes in green water flows at two interfaces: 1) green water use/soil and 2) green water use/atmosphere (as mentioned in the Introduction, Section 1).

At the interface between green water use and atmosphere, changes in ET have an influence on terrestrial ecosystem quality both on a regional (watershed or sub-watershed) and continental scale (considering the linkage of geographically separate regions by bridges of atmospheric moisture transport, linking upwind evaporation sources with downwind precipitation sinks) (Berger et al. 2014; Keys et al. 2012; Launiainen et al. 2014). An increase in ET recycled to the atmosphere can, in fact, increase downwind precipitation levels. For instance, the water resources of China depend almost entirely on terrestrial ET from the Eurasian continent (Van der Ent et al. 2010). In addition, negative trends in precipitation levels may already be detectable in the Río de Plata watershed in South America due to deforestation of the Amazon region (Van der Ent et al. 2010).

At the interface between green water use and soil, the increase in green water flows to the atmosphere, due to direct green water use caused by agriculture and forest practices, leads to reduced regional surface runoff, affecting surface blue water production. This can reduce the regional long-term blue water availability for ecosystem services. A decrease in the green water flows can result in more water being available for runoff, which may disturb stream flows and cause the groundwater table to rise. For instance, fast stream flows can lead to the transport of soil particles and nutrients towards water streams, leading to potential increases in water turbidity and the degradation of quality of water needed for human use and the sustainability of fauna and flora (Collins et al. 2011; Jones et al. 2012). In regions with high precipitation levels and when the actual land cover uses relatively low quantities of green water, water may recharge the groundwater by saturating large soil pores and cracks very quickly downward to the water table (Annenberg Foundation 2014). In this situation, the increased infiltration can raise the groundwater table towards the surface, promoting the evaporation of water directly from the soil and therefore increasing salt concentrations (Annenberg Foundation 2014). Furthermore, rainwater capture/harvesting may also induce changes in groundwater recharge. However, this should be analysed specifically in the context of each region and its soil properties. For instance, in arid and semi-arid areas, where precipitation levels are low during the dry season, the capture of rainwater during the wet season for use at a later time – especially for livestock, agriculture and domestic use – is common practice (UNEP 1997). This could suggest that rainwater capture is the main driver for low rates of groundwater recharge. However, in these areas, runoff and low soil infiltration rates occur mainly due to soil sealing, which seriously hampers groundwater recharge (Abu-Awwad and Shatanawi 1997; Ben-Asher and Berliner 1994; Yaseef et al. 2009).

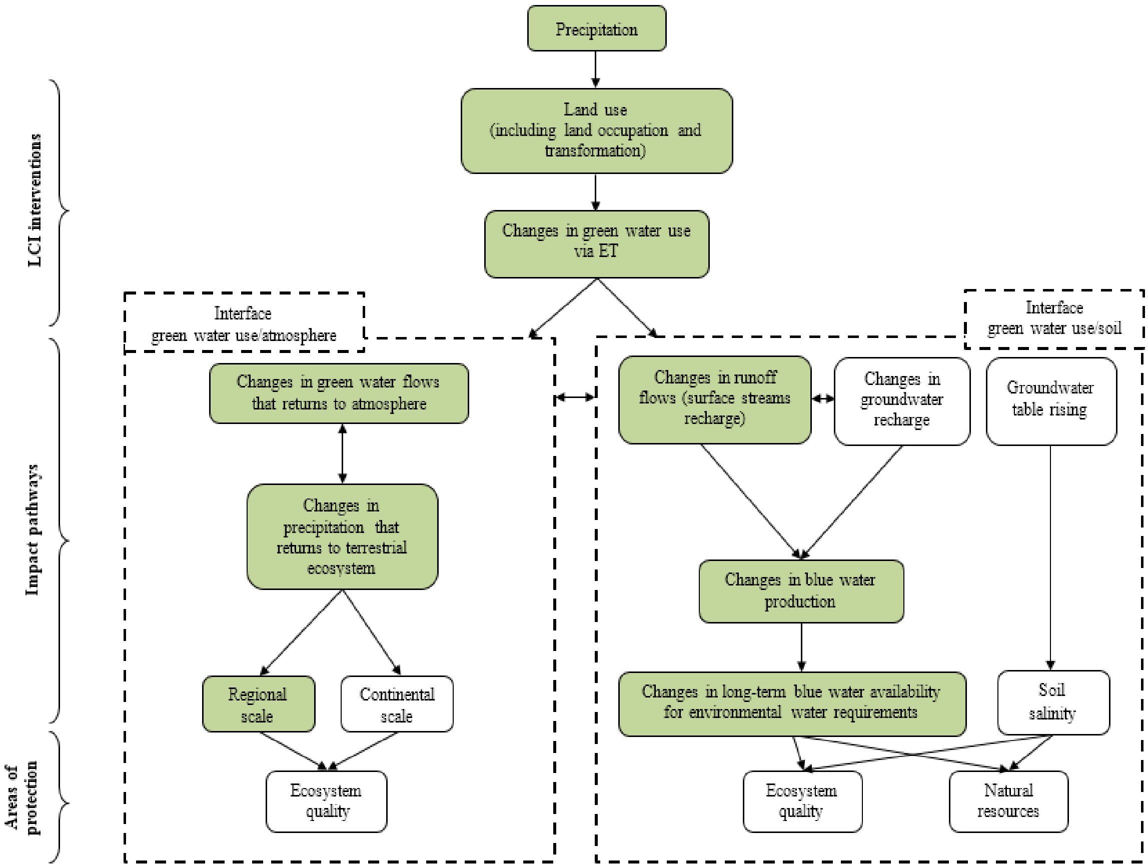

Fig. 2.5. Potential environmental impact pathways related to changes in green water flows. The green boxes are related to the pathway focused on this study.

Table 2.5 presents the main characteristics of the existing methods used to assess the potential environmental impacts derived from green water flows in LCA at the interface between green water use and soil. All methods (Milà i Canals et al. 2009; Núñez et al. 2012; Ridoutt et al. 2010) focus on changes in long-term blue water availability due to an increased use of green water, but using different approaches. Milà i Canals et al. (2009) consider two different interventions: 1) green water flow and 2) land use effects on infiltration and runoff. Núñez et al. (2012) also present two alternative interventions: 1) green water flow and 2) net green water flow. Ridoutt et al. (2010) account only for net green water flow.

Net green water flow is defined as the difference between the green water flow of crops or forests and the green water flow of a reference land use, e.g. PNV, providing a more meaningful result for the effects of net green water flow recycling to the atmosphere. The concept of PNV was developed by Tuxen (1956) as a hypothetical natural stage of vegetation in order to show the biotic potential in nature. The comparison between actual vegetation and PNV has been integrated into Ridoutt et al.'s (2010) and Núñez et al.'s (2012) LCA methods addressing green water because PNV is the potential natural land state, which does not cause additional pressure to the ecosystem due to anthropogenic disturbances.

According to Ridoutt et al. (2010) and Núñez et al. (2013, 2012), the ET of reference land use can be calculated following the Pinol et al. (1991) and Zhang et al. (2001) approaches, respectively. In the methods proposed by Milà i Canals et al. (2009) and Ridoutt et al. (2010), the green water flow of crop fields is calculated using the CROPWAT model, following the United Nations Food and Agriculture (FAO) approach (FAO 2009), whereas Núñez et al. (2012) estimates the green water flow using Allen et al.'s (1998) equation. It should be noted that the CROPWAT model was developed based on an approach suggested by Allen et al. (1998). To account for the land-use effects on infiltration in relation to a reference land use, Milà i Canals et al. (2009) suggest the use of a set of percentage values of precipitation "lost" developed by Zhang et al. (1999) for different reference land uses.

With regard to the level of damage, all methods analyse the regional effects of green water flow, i.e. all are focused on understanding how an increased use of green water flow reduces runoff and groundwater recharge, and therefore affects the long-term availability of blue water at the midpoint level. In particular, Núñez et al. (2012) look into the regional effects of green water flow on dry lands.

The name of the impact category related to green water flow differs depending on the method: water use; blue water availability due to land use; green water deprivation. For the water use impact category, Núñez et al. (2012) employ the green water index (GWSI) defined by the Water Footprint Network (WFN) (Hoekstra et al. 2011), whereas Milà i Canals et al. (2009) do not provide CFs.

For blue water availability due to land use, Milà i Canals et al. (2009) recommend the use of the water scarcity indicator (WSI_{nd}) developed by Smakhtin et al. (2004) at the watershed level. However, when this indicator is not available, two other options are suggested: 1) the use of an indicator based on water resources per capita (WRPC) proposed by Falkenmark (1986); 2) the use of an indicator of water use per resource (WUPR) developed by Raskin et al. (1997). The characterisation model suggested by Ridoutt et al. (2010) uses the spatial water scarcity indexes (WSIs) calculated by Pfister et al. (2009) for blue water availability due to land use at the sub-watershed level. For the green water deprivation impact category, Núñez et al. (2012) also use Pfister et al.'s (2009) WSIs. The use of these WSIs to assess the environmental impacts of green water flow is not entirely accurate because the CF is based on blue water, rather than directly evaluating the soil moisture reserves.

Table 2.5. Main characteristics of the existing methods to assess the potential environmental impacts derived from green water flows in LCA at the interface between green water use and soil.

Method	Milà i Canals et al. (2009)	Ridoutt et al. (2010)	Núñez et al. (2012)
Level of damage	Regional	Regional	Regional (on dry lands)
Impact pathway	Changes on regional green water flows via ET – changes in long-term blue water availability	Changes on regional green water flows via ET – changes in long-term blue water availability	Changes on regional green water flows ET – changes in long-term blue water availability
Assessment level	Midpoint	Midpoint	Midpoint
Impact category	▪ Water use ▪ Blue water availability due to land use	Blue water availability due to land use	▪ Water use ▪ Green water deprivation
Intervention	▪ Green water flow ▪ Land-use effects on infiltration	Net green water flow	▪ Green water flow ▪ Net green water flow
Inventory	▪ CROPWAT model for green water flow ▪ Percentages for precipitation "lost" derived from Zhang et al. (1999)	CROPWAT model for green water flow, and Zhang et al. (2001) approach for ET of reference land use	▪ Equation developed by Allen et al. (1998) for green water ▪ Pinol et al. (1991) approach for ET of natural reference land use
Characterisation model	▪ Water flow: not provided ▪ Land use: estimation of land-use effects on water cycle (infiltration and runoff), taking into account the CFs for blue water	Net green water flow taking into account the CFs for blue water	▪ Green water scarcity index (GWSI) ▪ Net green water flow taking into account the CFs for blue water
Characterisation factor (CF)	▪ CFs not provided for water use impact category ▪ Spatial CF for blue water availability due to land use: ◆ WSI_{nd} at watershed level – water resources available for human use, developed by Smakhtin et al. (2004) ◆ WRPC at country level – total renewable water resources being used per capita per year, put forward by Falkenmark (1986) ◆ WUPR at country level – percentage of actual renewable water resources being used, developed by Raskin et al. (1997)	Spatial water scarcity index (WSI) at sub-watershed level, developed by Pfister et al. (2009)	▪ 1/Pr, in which Pr is the effective precipitation at local level – for GWSI ▪ Spatial WSI at sub-watershed level, developed by Pfister et al. (2009) – for net green water flow

2. Materials and methods

2.1. Life Cycle Inventory

To quantify the change in ET for a land-use production system compared to the natural reference situation, the net green water concept has been applied (Núñez et al. 2012, 2013; Ridoutt et al., 2010). This means that the green water flow inventory is defined as the difference between the total green water flow (i.e. green ET) of actual crops or forests, ET_{act} (in $m^3_{H_2O}.ha^{-1}.yr^{-1}$), and the total green water flow of the PNV, ET_{PNV} (in $m^3_{H_2O}.ha^{-1}.yr^{-1}$).

A new method for calculating net green water flows at the inventory level is now proposed, taking into account the basin internal evaporation recycling ratio (BIER) (dimensionless) (Berger et al. 2014), i.e. the share of the water evaporated or transpired through plants to the atmosphere that returns to the same watershed. Therefore, the inventory of net green water flow explicitly accounts for the effective net green water flow, NGW_{eff} (in $m^3_{H_2O}.ha^{-1}.yr^{-1}$), which leaves the land-use production system (system boundary) towards downwind sink regions due to atmospheric moisture transport (Keys et al. 2012). It represents the difference between the effective ET of the land-use production system, $ET_{act,eff}$ (in $m^3_{H_2O}.ha^{-1}.yr^{-1}$), and the effective ET of the alternative reference land use, $ET_{PNV,eff}$ (in $m^3_{H_2O}.ha^{-1}.yr^{-1}$), as shown in Eq. 1.

$$NGW_{eff} = ET_{act,eff} - ET_{PNV,eff} \qquad \text{(Eq.1)}$$

The $ET_{act,eff}$ is determined by subtracting the portion of ET_{act} that is recycled within the watershed, as shown in Eq. 2, whereas the $ET_{PNV,eff}$ is determined by subtracting the portion of ET_{PNV} that is recycled within the watershed, as shown in Eq. 3.

$$ET_{act,eff} = ET_{act} - ET_{act} \times BIER \qquad \text{(Eq.2)}$$

$$ET_{PNV,eff} = ET_{PNV} - ET_{PNV} \times BIER \qquad \text{(Eq.3)}$$

Through identification of the remnants of natural or quasi-natural land cover of the past, several PNV maps have been developed for regional, national and continental scales (Bohn et al. 2007; Capelo et al. 2007; Kuchler 1964; Loidi and Báscones 1995; Loidi et al. 2011; Ramankutty and Foley 1999) addressing different quasi-natural land cover, e.g. deciduous, broadleaved, mixed and tropical forests, grasslands, shrublands and semi-deserts.

The method proposed allows the consideration of two alternative reference land uses: 1) quasi-natural forests and 2) grasslands/shrublands. On the local or regional scale, depending on the specific location of a crop or forest, both quasi-natural forests and grasslands/shrublands can co-exist. Therefore, in situations in which two reference vegetations are appropriate, the ET_{PNV} should be calculated for both. The ET_{PNV} depends on climatic parameters (temperature, precipitation and solar radiation) and nonclimatic parameters (soil-root zone water holding capacity and specific characteristics of different natural vegetation species, such as canopy conductance and leaf area index). In the absence of more specific methods to estimate the ET_{PNV}, we propose to estimate it based on models developed by Zhang et al. (2001) and Komatsu et al. (2012) for quasi-natural forests and by Zhang et al. (2001) for grasslands/shrublands.

With regard to quasi-natural forests, the ET_{PNV} is calculated under non-water limited conditions in the system, ET_{pot} (in $m^3_{H_2O}.ha^{-1}.yr^{-1}$), mean annual precipitation, P (in $m^3_{H_2O}.ha^{-1}.yr^{-1}$), and soil-root zone water holding capacity, w (dimensionless) as shown by Eq. 4 (Zhang et al. 2001) and Eq. 5 (Komatsu et al. 2012). The w parameter (dimensionless) is a parameter related to the capability of plants to store water in the root zone for transpiration purposes, which depends on the depth root zones of plants and soil hydraulic properties (Zhang et al. 2001). For forests, this parameter assumes different values according to nine mean annual air temperature classes, T_a (in ºC), as defined by Komatsu et al. (2012).

$$ET_{PNV} = P\left(\frac{1 + w \times \frac{ET_{pot}}{P}}{1+w\frac{ET_{pot}}{P} + \frac{P}{ET_{pot}}}\right) \qquad \text{(Eq.4)}$$

Eq. 4 assumes that ET_{PNV} is controlled by P and ET_{pot}. In turn, for forests, ET_{pot} depends on the physiological aspects of each tree species, such as stomatal closure and leaf fall, due to temperature variation as established by Eq. 5 (Komatsu et al. 2012).

$$ET_{pot} = 0.488 \times T_a^2 + 27.5 \times T_a + 412 \qquad \text{(Eq.5)}$$

Regarding grasslands/shrublands, ET_{PNV} can be calculated directly using Eq. 6 (Zhang et al. 2001), in which 0.5 is the w parameter value and 11,000 $m^3_{H_2O}.yr^{-1}.ha^{-1}$ is ET_{pot}.

$$ET_{PNV} = P\left(\frac{1 + 0.5 \times \frac{11{,}000}{P}}{1 + 0.5 \times \frac{11{,}000}{P} + \frac{P}{11{,}000}}\right) \qquad \text{(Eq.6)}$$

2.2. Life Cycle Impact Assessment (LCIA)

In the LCIA phase, both interfaces, i.e. green water use/atmosphere and green water use/soil, are considered. Thus, we quantify the impacts on terrestrial green water flows (TGWI) and reductions in surface blue water (river) production (RBWP) caused by reductions in surface runoff from a land-use production system. TGWI and RBWP (in $m^3_{H_2O}.ha^{-1}.yr^{-1}$) are calculated according to Eq. 7 and Eq. 8, respectively.

$$TGWI = NGW_{eff} \times CF_{TGWI} \qquad \text{(Eq.7)}$$

$$RBWP = NGW_{eff} \times CF_{RBWP} \qquad \text{(Eq.8)}$$

where CF_{TGWI} is the characterisation factor for the assessment of TGWI and CF_{RBWP} is the characterisation factor for the assessment of RBWP.

Both CF_{TGWI} and CF_{RBWP} are regional- and species-specific, as they depend mainly on solar radiation, precipitation levels, soil moisture, root zone water holding capacity and canopy conductance (determined from the leaf area index and stomatal conductance) of the crop or forest species under study. These CFs, ranging from 0 to 1, give information concerning the functional impairment of the net regional natural green water flow "reserves", i.e. they indicate the deprived portion of $ET_{act,eff}$ that can contribute to the regulation of the hydrological cycle.

CF_{TGWI} is proposed as a function of $ET_{act,eff}$ and $ET_{PNV,eff}$, these being indicators of available water for the interface green water use/atmosphere. CF_{RBWP} is proposed as a function of $ET_{act,eff}$ and $ET_{EWR,eff}$, used as indicators of available water for the interface green water use/soil, i.e. green water available for surface blue water production. $ET_{EWR,eff}$ refers to the effective threshold ET level of the land-use production system that guarantees the environmental water requirement (EWR) (Smakhtin et al. 2004). This is the quantity of water flow required to sustain positive functioning of the blue-water-dependent aquatic ecosystems services, including those required for human livelihoods and well-being (Eq. 9).

$$ET_{EWR,\,eff} = P - x_{EWR} \times \left(P - ET_{PNV,\,eff}\right) \qquad \text{(Eq.9)}$$

where, x_{EWR} (dimensionless) is the EWR expressed as a fraction of long-term mean annual actual runoff in a river, as stated in Maes et al. (2009).

When a crop or forest species evaporates or transpires the same effective volume of water as the reference land use, CFs are set to zero as shown in Eq. 10, which means there are no disturbances to either interface under analysis.

$$CF_{TGWI} = CF_{RBWP} = 0, \text{ when } ET_{act,\,eff} = ET_{PNV,\,eff} \qquad \text{(Eq.10)}$$

When $ET_{act,eff}$ is lower than $ET_{PNV,eff}$, the green water flow that returns to the atmosphere is reduced, which has an impact on the precipitation levels that return to the terrestrial ecosystem. The CF_{TGWI} is determined as shown in Eq. 11. However, the land-use production system does not reduce surface runoff to surface blue water production. Therefore, $CF_{RBWP} = 0$, as shown in Eq. 12.

$$CF_{TGWI} = 1 - \frac{ET_{act,\,eff}}{ET_{PNV,\,eff}}, \text{ when } ET_{act,eff} < ET_{PNV,\,eff} \qquad \text{(Eq.11)}$$

$$CF_{RBWP} = 0, \text{ when } ET_{act,eff} < ET_{PNV,\,eff} \qquad \text{(Eq.12)}$$

When the $ET_{act,eff}$ is higher than $ET_{PNV,eff}$, the crops or forests do not cause disturbances to terrestrial green water flows. Therefore, CF_{TGWI} is zero, as shown in Eq. 13. When $ET_{act,eff}$ is higher than $ET_{PNV,eff}$, reductions in surface runoff occur, leading to reductions in surface blue water production until a critical level of $ET_{EWR,eff}$ is reached. In this case, CF_{RBWP} is calculated according to Eq. 14.

When $ET_{act,\,eff}$ is equal to or higher than $ET_{EWR,\,eff,}$, the critical level is achieved. This means that the reductions in surface blue water production may lead to high impairment of regional long-term blue water availability, so that not all aquatic ecosystems services can be fulfilled. In this situation, CF_{RBWP} is set to 1 (Eq. 15).

$$CF_{TGWI} = 0, \text{ when } ET_{PNV,\,eff} < ET_{act,\,eff} \qquad \text{(Eq.13)}$$

$$CF_{RBWP} = \frac{ET_{act,\,eff}}{ET_{EWR,\,eff}}, \text{ when } ET_{PNV,eff} < ET_{act,\,eff} < ET_{EWR,\,eff} \qquad \text{(Eq.14)}$$

$$CF_{RBWP} = 1, \text{ when } ET_{act,\,eff} \geq ET_{EWR,\,eff} \qquad \text{(Eq.15)}$$

3. Case study: *E. globulus* stands

3.1. Functional unit

In this study, the functional unit (FU) is defined as one hectare of land occupied by *E. globulus* during a complete rotation of 12 years, with a tree density ranging from 833 to 1,904 trees per hectare.

3.2. Description of the system

E. globulus has the capacity to sprout from the stump after harvesting and therefore, in Portugal, it is managed as a coppiced stand, typically in three successive coppice rotations of 12 years each.

The system boundary considers the full *E. globulus* growth during its first 12-year coppice rotation, following a gate-to-gate approach. The nursery stage (first year before the stand installation) was excluded from the system boundary.

As *E. globulus* in Portugal has shallow, horizontal root systems, it has been assumed that *E. globulus* does not use groundwater (blue water). Therefore, only green water is evaporated or transpired throughout *E. globulus* growth.

Specific data for the delimited regions of *E. globulus* stands identified in Fig. 2.6 were used. This delimitation is performed based on the Nomenclature of Territorial Units for Statistics (NUTS), which is a hierarchical system used for dividing up the territory of the European Union (Eurostat 2013). In particular, the NUTS 3 classification was used to delimit the regions of *E. globulus* production. The remaining territory was excluded from this assessment because it corresponded to land-use production systems that are not favourable to the growth of *E. globulus* due to soil characteristics (texture and depth, stoniness, water holding capacity) and precipitation patterns and/or to very small areas of *E. globulus*.

Data from several stands within each region were considered, but the specific characteristics and location of each stand analysed are not presented due to data confidentiality. In each one of the regions, an average of eight *E. globulus* stands were analysed. However, in the Peninsula of Setúbal, only two stands were analysed due to the smaller dimension of this region.

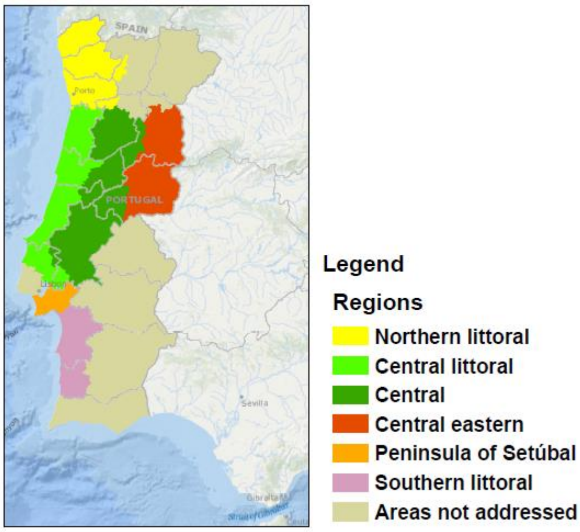

Fig. 2.6. Delimitation of the geographic regions of *E. globulus* stands in Portugal.

3.3. Inventory and impact assessment of effective net green water flow

With regard to inventory, to calculate ET_{act} of *E. globulus*, we used a modified version of the 3-PG (Physiological Principles Predicting Growth) model, originally developed by Landsberg and Waring (1997). The 3-PG is a process-based model of forest growth that simulates canopy development, light capture, photosynthesis and net-carbon assimilation. It also predicts the mean annual wood volume growth increment (MAI, in $m^3_{wood}.ha^{-1}.yr^{-1}$, excluding bark and stump) and water-use efficiency of wood production. A set of parameter values for adapting the 3-PG model to Portuguese *E. globulus* growth conditions were developed by Fontes et al. (2006).

ET_{act} in this model is calculated using the Penman-Monteith equation (Monteith 1965) in the form given in Landsberg and Gower (1997), taking into account climatic conditions (temperature, precipitation and solar radiation) and nonclimatic data (canopy conductance and leaf area index). As input data, the model requires information on the initial conditions of the stands (stem, foliage, tree density), soil (soil moisture, soil-root zone water holding capacity, texture, stoniness) and climate. The data related to the stands and soil were collected from *E. globulus* inventories carried out by the Forest and Paper Research Institute (RAIZ). The long-term mean monthly precipitation, air temperature and daily solar radiation data for each *E. globulus* stand analysed were supplied by the Portuguese Sea and Atmosphere Institute (IPMA 2013). The air temperature is the monthly mean for the so-called normal period (1971-2000). The precipitation and daily solar radiation are also monthly means for the periods 1980-1995 and 1938-1970, respectively.

The PNV map developed for Portugal (Capelo et al. 2007) shows that the delimited regions can be covered naturally by both quasi-natural forest and shrublands. Therefore, to understand the influence of ET rates from these two different natural land covers, both quasi-natural forest and grasslands/shrublands were considered as alternative reference land uses. For grasslands/shrublands, ET_{PNV} was calculated based on monthly long-term average data of precipitation, as explained in Section 2.1. For quasi-natural forest, ET_{PNV} was calculated based on monthly long-term average data for precipitation, ET_{pot} and the w parameter. This parameter assumes a value of 1.3 when T_a is within the temperature class ranging between 10-15 ºC, or a value of 2.0 when T_a is within the temperature class ranging between 15-20 ºC (Komatsu et al. 2012).

The NGW_{eff} was calculated taking into account the following BIER values taken from Berger (2014): 0.03 for all stands in the northern littoral, 0.01 for all stands in the central littoral, 0.02 for all stands in the central and central-eastern regions and the Peninsula of Setúbal, and zero for the southern littoral as in this region local ET makes no contribution to local precipitation.

With regard to the impact assessment, TGWI and RBWP were addressed by multiplying the NGW_{eff} calculated in the inventory phase by the CFs of each area under analysis, as described in Section 2.2. To calculate CF_{RBWP}, $ET_{EWR,\ eff}$ is required, which was calculated considering an average x_{EWR} for Portugal of 0.31, a value provided by Vladimir Smakhtin (personal communication, June 2014).

4. Results and discussion

4.1 Results of inventory and impact assessment

4.1.1. Inventory of net green water flows

Table 2.6 presents the mean annual ET for each *E. globulus* stand (hereinafter referred to as s-ET_{act}) and the total mean annual ET of *E. globulus* for each production region (hereinafter referred to as rt-ET_{act}), calculated using a modified version of the 3-PG model, as explained in Section 3.3. The stands in the northern littoral and central littoral regions have the highest s-ET_{act} and rt-ET_{act} rates. These regions also present the highest long-term mean annual precipitation of 15,326 $m^3_{H_2O}$.ha^{-1}.yr^{-1} and 9,217 $m^3_{H_2O}$.ha^{-1}.yr^{-1}, respectively. *E. globulus* stands in the central and central-eastern regions present the lowest r-ET_{act} rates during the rotation cycle, with a long-term mean annual precipitation ranging from 7,940 $m^3_{H_2O}$.ha^{-1}.yr^{-1} in the central region to 8,011 $m^3_{H_2O}$.ha^{-1}.yr^{-1} in the central-eastern region. For all the regions studied, the mean annual precipitation is higher than the mean annual ET, indicating that the annual precipitation rates seem to be sufficient to fulfil the green water requirements of *E. globulus*.

Table 2.6. Mean annual ET for each *E. globulus* stand, s-ET_{act} , total mean annual ET of *E. globulus* for each production region, rt-ET_{act} , mean annual effective ET for each *E. globulus* stand, s-$ET_{act,\ eff}$, and mean annual effective ET of ET of *E. globulus* for each production region, rt-$ET_{act,\ eff}$.

Regions	s-ET_{act} ($m^3.ha^{-1}.yr^{-1}$)								rt-ET_{act} ($m^3.ha^{-1}.yr^{-1}$)	s-$ET_{act,\ eff}$ ($m^3.ha^{-1}.yr^{-1}$)								rt-$ET_{act,\ eff}$ ($m^3.ha^{-1}.yr^{-1}$)
	Stand 1	Stand 2	Stand 3	Stand 4	Stand 5	Stand 6	Stand 7	Stand 8		Stand 1	Stand 2	Stand 3	Stand 4	Stand 5	Stand 6	Stand 7	Stand 8	
Northern littoral	9,082.2	8,926.8	8,306.0	8,971.3	9,893.3	9,893.3	9,276.4	11,137.9	9,435.9	8,809.7	8,659.0	8,056.8	8,702.1	9,596.5	9,596.5	8,998.1	10,803.8	9,152.8
Central littoral	5,642.8	5,642.8	6,585.0	6,585.0	8,408.8	9,399.6	6,733.8	6,568.9	6,945.8	5,586.4	5,586.4	6,519.2	6,519.2	8,324.7	9,305.6	6,666.5	6,503.2	6,876.4
Central	5,032.2	5,398.0	4,857.0	5,509.1	6,369.7	6,242.2	6,103.5	6,754.2	5,783.2	4,931.6	5,290.1	4,759.9	5,398.9	6,242.3	6,117.3	5,981.5	6,619.1	5,667.6
Central eastern	5,336.2	4,895.2	5,886.6	5,501.3	6,367.6	6,383.9	6,489.4	5,034.8	5,736.9	5,229.5	4,797.3	5,768.9	5,391.3	6,240.3	6,256.2	6,359.7	4,934.1	5,622.2
Peninsula of Setúbal	5,658.0	6,156.0	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	5,907.0	5,544.9	6,032.9	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	5,788.9
Southern littoral	5,274.5	6,305.1	6,501.3	5,247.9	6,182.4	7,041.9	6,168.0	6,204.5	6,115.7	5,274.5	6,305.1	6,501.3	5,247.9	6,182.4	7,041.9	6,168.0	6,204.5	6,115.7

n.a. not applicable

Fig. 2.7 presents how the calculated mean annual ET of *E. globulus* per region (hereinafter referred to as r-ET_{act}) varied throughout the first rotation cycle (12 years). Again, the stands in the northern littoral and central littoral regions have the highest r-ET_{act} rates over the full period. During the early part of the rotation cycle, until canopy closure, which happens at around the age of five years, the trees present a substantially lower r-ET_{act} when compared to the period after canopy closure. After canopy closure, the leaf area index (LAI) of *E. globulus* reaches its maximum, resulting in higher r-ET_{act} values. LAI is defined as the projected or silhouette area of leaves per unit area of ground (Monteith and Unsworth 1990). As trees grow and get older, they suffer self-pruning of shaded branches, as well as a decline in the capacity to produce new leaves (Pereira et al. 1997). This leads to the decline of both leaf area and canopy conductance at the end of the rotation, which explains the modest decrease in r-ET_{act} during the last year of *E. globulus* growth.

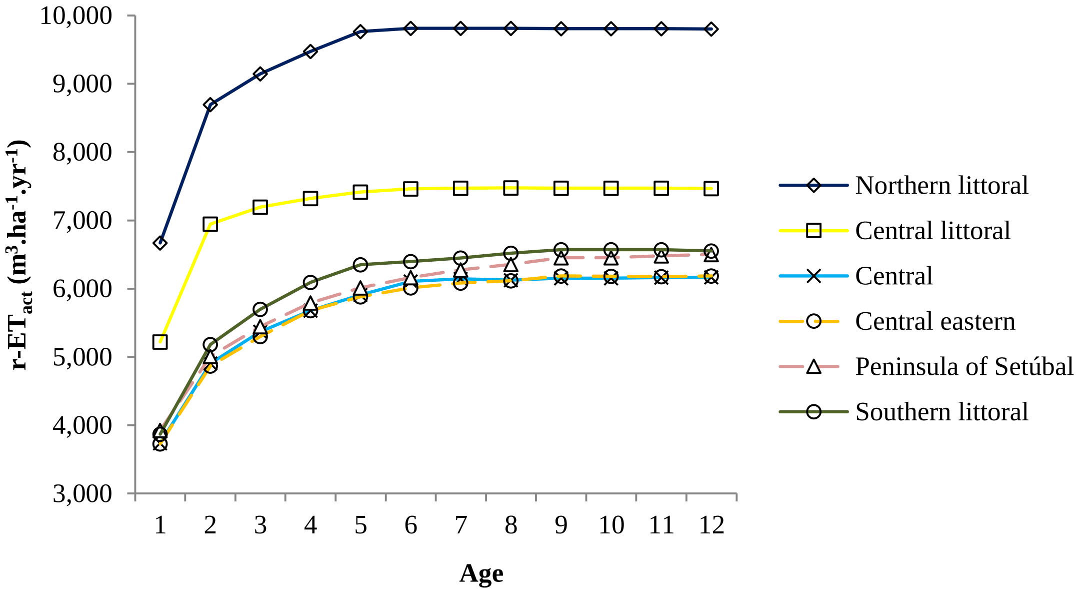

Fig. 2.7. Predicted mean annual ET of *E. globulus* per region, throughout the first rotation (12 years), r-ET_{act}.

The mean annual effective ET for each *E. globulus* stand (hereinafter referred to as s-$ET_{act,\ eff}$) and the total mean annual effective ET of *E. globulus* for each production region (hereinafter referred to as rt-$ET_{act,\ eff}$) are also presented in Table 2.6. The rt-ET_{act} of *E. globulus* represents 61-81 % of the precipitation depending upon the growing region. In turn, the predicted rt-ET_{PNV} of quasi-natural forest systems ranges from 52 % to 84 % of precipitation, whereas the rt-ET_{PNV} of grasslands/shrublands systems ranges from 49 % to 73 %.

Table 2.7 presents the inventory results of the net effective green water flow, NGW_{eff}, for each stand studied within each production region of *E. globulus*, expressed per FU. As all *E. globulus* stands within the same region present different initial conditions (tree density, stem, foliage), soil properties (water-holding capacity, soil texture) and climate characteristics, the calculated NGW_{eff} for each stand also varies. When NGW_{eff} is positive, this means that the stands under analysis consume a higher proportion of the precipitation than the alternative reference land use they have replaced. In this case, the *E. globulus* stands do not cause potential impacts on terrestrial green water flows (TGWI), but contribute to reductions in surface blue water production (RBWP). On the other hand, when NGW_{eff} is negative, this means that the growth of *E. globulus* reduces the green water flow that is recycled to the atmosphere, causing impacts on TGWI, whereas no impacts are expected for RBWP, as explained in Section 2.2.

For grasslands/shrublands, the NGW_{eff} of *E. globulus* stands in the northern littoral and central littoral regions are respectively on average 11 % and 30 % higher than that obtained using quasi-natural forest as an alternative reference land use. For the remaining regions, with the exception of the Peninsula of Setúbal, higher values of NGW_{eff} are obtained when quasi-natural forest is considered as an alternative reference land use. In the Peninsula of Setúbal, one of the studied stands presents a higher NGW_{eff} when quasi-natural forest is considered as reference land use, whereas the other stand presents a much higher NGW_{eff} when grassland/shrublands are considered as reference land use.

Table 2.7. Inventory results of the net effective green water flow, NGW_{eff} for each stand studied within each production region of *E. globulus*, expressed per functional unit (FU).

Regions	NGW_{eff} ($m^3.ha^{-1}.yr^{-1}$) with quasi-natural forest as reference land use								NGW_{eff} ($m^3.ha^{-1}.yr^{-1}$) with grasslands/shrublands as reference land use							
	Stand 1	Stand 2	Stand 3	Stand 4	Stand 5	Stand 6	Stand 7	Stand 8	Stand 1	Stand 2	Stand 3	Stand 4	Stand 5	Stand 6	Stand 7	Stand 8
Northern littoral	1,144.5	1,000.8	638.9	1,017.5	2,143.2	2,143.2	1,479.2	2,734.0	1,201.2	1,222.5	374.7	1,250.4	2,446.2	2,446.2	1,555.2	2,834.5
Central littoral	-353.0	-353.0	480.2	480.2	1,312.5	2,044.3	209.9	250.2	396.8	396.8	773.1	773.1	1,720.5	2,422.4	528.3	599.0
Central	-1,049.8	-990.9	-1,098.1	-773.0	-73.6	-159.8	-295.7	429.3	-309.2	-195.8	-404.3	2.1	755.9	631.0	495.2	774.6
Central eastern	-803.7	-878.5	-712.1	-1,294.2	307.0	-59.6	-33.2	-885.1	-87.3	-519.5	144.1	-395.2	753.9	769.8	792.8	-155.5
Peninsula of Setúbal	-396.2	91.8	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	368.8	856.9	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Southern littoral	-645.5	1,865.0	-642.1	-824.0	-193.6	-101.5	277.3	-261.0	-480.4	550.2	223.2	-209.1	498.6	763.7	465.5	460.7

n.a. not applicable

4.1.2. Impact assessment of net green water flows

In the impact assessment phase, regional- and species-specific CFs were calculated to obtain both TGWI and RBWP, as explained in Section 2.2. To quantify TGWI, CF_{TGWI} ranges from 0.01 to 0.19 when quasi-natural forest is considered as an alternative reference land use and from 0.02 to 0.10 when grasslands/shrublands are considered. To quantify RBWP, CF_{RBWP} ranges from 0.56 to 0.87 when quasi-natural forest is considered as an alternative reference land use and from 0.56 to 0.88 when grasslands/shrublands are considered. Although CF_{RBWP} represents a decrease in surface blue water production, the land-use production systems under analysis are not expected to cause severe impairment to regional long-term blue water required to sustain EWR. The long-term mean annual precipitation levels (ranging from 7,400 to 15,326 $m^3_{H_2O}$ .ha^{-1}.yr^{-1}) are higher than $ET_{EWR,\ eff}$ (ranging from 7,300 to 13,658 $m^3_{H_2O}$ .ha^{-1}.yr^{-1} with quasi-natural forest as reference land use and from 7,025 to 13,628 $m^3_{H_2O}$ .ha^{-1}.yr^{-1} with grasslands/shrublands as reference land use), indicating that they still guarantee the long-term blue water levels required to fulfil EWR within the regions considered.

Regarding the quasi-natural forest as an alternative reference land use, we find that in all *E. globulus* stands of the north littoral region, s-$ET_{act,\ eff}$ values are higher than those of s-$ET_{PNV,\ eff}$, causing no disturbances in the terrestrial green water flows (Fig. 2.8) (i.e. no TGWI occurs). Although in the central littoral region almost all the stands also present s-$ET_{act,\ eff}$ values higher than those of s-$ET_{PNV,\ eff}$, there are two stands for which s-$ET_{PNV,\ eff}$ is higher than s-$ET_{act,\ eff}$, causing the smallest impact on TGWI compared to the other regions under analysis.

However, the larger s-$ET_{act,\ eff}$ of the north littoral and central littoral regions is expected to reduce surface blue water production due to a decrease in surface runoff. The *E. globulus* produced in the northern littoral area have the highest average MAI (approximately 21 m^3_{wood}.ha^{-1}. yr^{-1}.These stands do not cause TGWI and although they present the highest average RBWP (1,072.0 $m^3_{H_2O}$ .ha^{-1}.yr^{-1}) – being in the region with the highest potential to reduce surface blue water production – the critical level of $ET_{EWR,\ eff}$ is not reached, meaning that aquatic ecosystem functions are still fulfilled.

As a strategy to maximise the MAI of *E. globulus*, the national strategy for the Portuguese forest (DGRF 2006), based on precipitation levels and tree productivity, recommends the reordering of the forest occupation, relocating the *E. globulus* stands to the northern littoral and central littoral areas. The results of this study are in agreement with this recommendation for relocation.

Although the Peninsula of Setúbal presents a slightly higher TGWI (26.4 $m^3_{H_2O}.ha^{-1}.yr^{-1}$) than the central littoral regional (21.0 $m^3_{H_2O}.ha^{-1}.yr^{-1}$), mainly due to soil properties and climate conditions, it presents a very low average MAI (approximately 10 $m^3_{wood}.ha^{-1}.yr^{-1}$). This means that this region is not the most appropriate in terms of relocating *E. globulus* for pulp and paper production purposes.

In the central region, $ET_{act,\,eff}$ is higher than $ET_{PNV,\,eff}$ for most *E. globulus* stands studied, resulting in RBWP of 347.9 $m^3_{H_2O}.ha^{-1}.yr^{-1}$. The two stands where $ET_{act,\,eff}$ is lower than $ET_{PNV,\,eff}$ (see Table 2.7) cause an average land-use impact on terrestrial green water flow of 94.6 $m^3_{H_2O}.ha^{-1}.yr^{-1}$.

In comparison with the central-eastern and central regions, the trees growing in southern littoral region are expected to disturb terrestrial green water flows less than the production of blue water; i.e. they present a lower impact at the green water use/atmosphere interface with a TGWI of 43.0 $m^3_{H_2O}.ha^{-1}.yr^{-1}$ and a higher impact at the green water use/soil interface with an RBWP value of 925.5 $m^3_{H_2O}.ha^{-1}.yr^{-1}$.

The error bars in Fig. 2.8 indicate the range of variation obtained for TGWI and RBWP within each region. The greatest variations in TGWI were obtained for the central-eastern and southern littoral regions, ranging from an increase of 148 % to a decrease of 100 % for the central-eastern region and from an increase of 160 % to a decrease of 97 % for the southern littoral region. Regarding RBWP, major variations were observed for the central littoral and southern regions. In the central littoral, RBWP ranged from an increase of 163 % to a decrease of 76 %, whereas in the southern littoral, RBWP ranged from an increase of 76 % to a decrease of 76 %. These fluctuations can be explained by the same reasons discussed above for NGW_{eff}, i.e. within the same delimited region of *E. globulus*, the stands present different tree density, stem, foliage and climate conditions, and the soil presents different water-holding capacity.

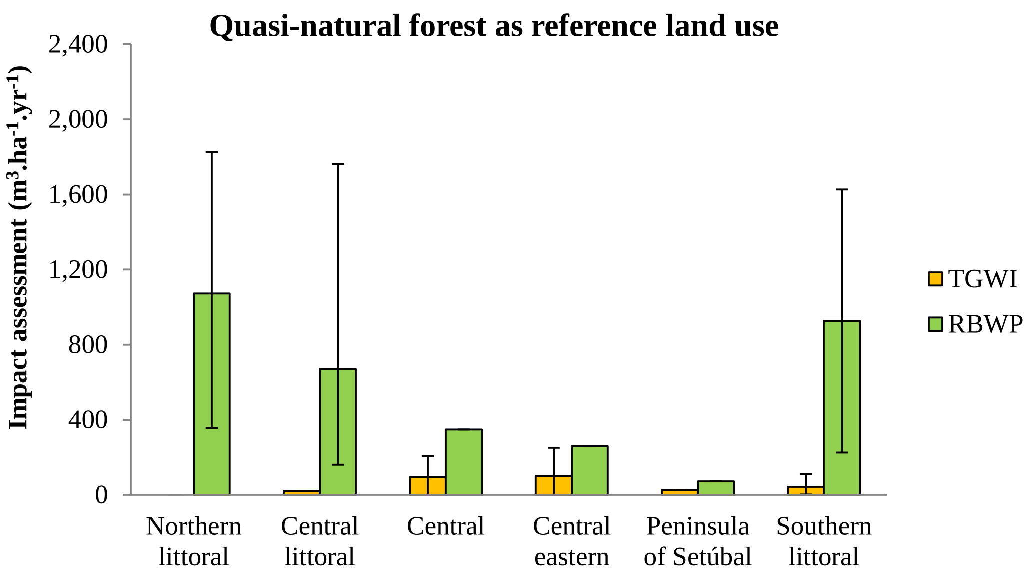

Fig. 2.8. Regional impacts on terrestrial green water flows (TGWI) and reductions of surface blue water production (RBWP) of *E. globulus* for each production region expressed per functional unit (FU), considering the quasi-natural forest as an alternative reference land use. The error bars indicate the variation range obtained for the TGWI and RBWP within each region.

As observed for the quasi-natural forest, when grasslands/shrublands are taken as the reference land use, *E. globulus* stands in the northern littoral do not present TGWI but contribute to a reduction in surface blue water production (RBWP). In addition, employing grasslands/shrublands as the reference land use, the central littoral region and the Peninsula of Setúbal do not present TGWI, but do contribute to the reduction of surface blue water production. For grasslands/shrublands, *E. globulus* produced in the northern littoral region presents an RBWP value of 1,173.9 $m^3_{H_2O}.ha^{-1}.yr^{-1}$; thus, this is the region with the highest potential to reduce surface blue water production (Fig. 2.9). However, the aquatic ecosystem functions also remain fulfilled, as observed for the quasi-natural forest.

The other regions of *E. globulus* stands – central, central-eastern and southern littoral – simultaneously present TGWI and RBWP, with RBWP for all these regions being much higher when compared to TGWI (Fig. 2.9). The error bars in Fig. 2.9 indicate the range of variation obtained for TGWI and RBWP within each region. The greatest variation in TGWI is within the central-eastern region, ranging from an increase of 142 % to a decrease of 93 %. Regarding RBWP, major variations are observed for the northern littoral and central littoral areas. In the northern littoral region, RBWP ranges from an increase of 65 % to a decrease of 82 %, whereas in the central littoral area, RBWP ranges from an increase of 162 % to a decrease of 58 %.

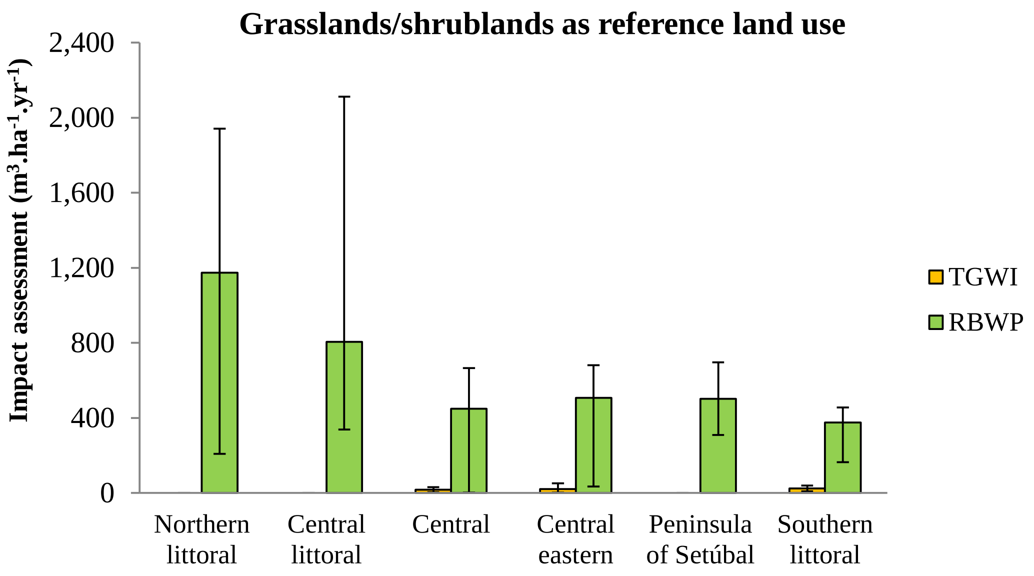

Fig. 2.9. Regional impacts on terrestrial green water flows (TGWI) and reductions of surface blue water production (RBWP) of *E. globulus* for each production region expressed per functional unit (FU), considering the grasslands/shrublands as an alternative reference land use. The error bars indicate the variation range obtained for the TGWI and RBWP within each region.

4.2. Limitations of this study and further research

The use of different methods to estimate ET_{act} and ET_{PNV} can be a source of error in the calculation of NGW_{eff}. The approach recommended is always to use the best available model for the estimation of ET. In the case of *E. globulus*, this was the Landsberg and Gower model. However, this model could not be applied to PNV as it is not parameterized for such land use. Thus, the methods proposed by Zhang et al. (2001) and Komatsu et al. (2012) for quasi-natural forests and by Zhang et al. (2001) for grasslands/shrublands were used instead, as these methods are parameterized for different types of land use. A hypothetical solution for avoiding this source of error would be to apply the Zhang et al. (2001) and Komatsu et al. (2012) models to both actual and reference land use. However, this would result in an obviously rougher estimation of ET for the *E. globulus* forest as a superior model could be used.

Furthermore, the proposed approach has a few constraints concerning data requirements and their availability, the mean lifespan of the reference land use and the EWR expressed as a fraction of long-term mean annual actual runoff in a river to ensure ecosystem functions.

The reliability of the ET_{act} calculation depends on the input parameters. Collecting data on soil characteristics (texture, stoniness, available soil moisture) and the condition of stands and/or crops (stem, foliage, root, tree density) is often challenging due to the lack of appropriate data for the land-use production system under study. In this study, these input parameters were derived from regional- and species-specific measurements. Obtaining these measurements is labour- and time-intensive, but is also the first step towards developing databases, thus facilitating the LCI phase. The collection of information on local climatic parameters, such as precipitation, temperature and the incidence of solar radiation above the crop and/or forest growth, is also challenging. The use of normal climatological data (long-term data) is an option to facilitate the collection of climatic parameter data. By this means, it is possible to ensure that the time series over the 30 years considered represent the predominant value of the required climatic parameter, which characterises the land-use production system under study. However, due to lack of climatological data for the specific areas under study, data on both precipitation and solar radiation were collected for a different time series, which may introduce uncertainty in the calculation of ET_{act}.

With regard to the alternative reference land uses, Eqs. 4 and 6, which are applicable to quasi-natural forest and grasslands/shrublands respectively, do not take into account the mean lifespan of these land covers. For instance, the quasi-natural forest presents a much higher mean lifespan than the temporal scale of 30 years of the long-term climate data. This short timescale can affect the ET values of quasi-natural forest, as ET rates tend to decrease after maturity, which is defined as 80 years or older for coastal forests and 100-120 years or older for high-elevation forests (Parminter 1995; Vertessy et al. 2001). Further research should establish better ways of evaluating the influence of the mean lifespan of vegetation on ET rates.

With regard to impact assessment, when the forest or crop evaporates or transpires more than the alternative reference land use, surface blue water production is reduced due to the decrease in surface runoff, assuming that no impacts on regional long-term blue water availability occur until the critical level of $ET_{EWR,eff}$ is reached. According to Eq. 9, x_{EWR} should be related to annual mean runoff under PNV rather than actual runoff data. However, no data on x_{EWR} as a proportion of the long-term mean annual runoff (% MAR) were found under PNV conditions in the Portuguese regions. To understand how actual x_{EWR} values influence the results of the impact assessment (i.e. surface blue water production, RBWP), we performed a sensitivity analysis. The x_{EWR} values changed by ±15 % for both alternative reference land uses (quasi-natural forest and grasslands/shrublands). For both quasi-natural forest and grasslands/shrublands, this change in x_{EWR} had little influence on the impact assessment results of RBWP, with variations ranging from an increase and decrease of around 1 % to 3 % in the RBWP values presented in Figs. 2.8 and 2.9. The greatest variation in RBWP is obtained for *E. globulus* stands in the northern littoral region regardless of the alternative reference land use. The least variation in RBWP is obtained for *E. globulus* stands in the central-eastern region when quasi-natural forest is considered as the reference land use and for *E. globulus* produced in southern littoral areas when grasslands/shrublands are considered the reference land use.

Furthermore, on-field measurements and a deeper characterisation of the quantity and quality of river water flows required for ecosystems in the regions under study are needed to address specific regional EWR. In addition, EWR simply takes into account river flows, overlooking groundwater stocks. A full characterisation of river flows and groundwater stocks is required to calculate a more accurate critical level of $\mathrm{ET}_{\mathrm{EWR,eff}}$, improving the reliability of the results obtained in the impact assessment phase. Furthermore, a long-term decrease in surface runoff will progressively reduce surface and groundwater blue water production, which, in combination with an increase in blue water use (for instance, for industry purposes), can hamper long-term water availability for aquatic functions (such as power generation). Therefore, a full characterisation of watershed users within the region under analysis will help in understanding whether impacts on long-term blue water availability occur below the critical level of $\mathrm{ET}_{\mathrm{EWR,eff}}$.

For a complete assessment at the green water use/atmosphere interface, in addition to examining TGWI on a regional scale, further research on the impacts of the reduced ET on the continental scale is also required (Fig. 2.5). A deeper understanding of drivers that affect precipitation disturbances and atmospheric moisture transport, connecting upwind evaporation sources with downwind precipitation sinks (Ellison et al. 2012; Keys et al. 2012; Van der Ent et al. 2010), linked to global climate models, is needed.

5. Conclusions

We have proposed a method to assess impacts both on terrestrial green water flows and on reductions in surface blue water production caused by green water deficits due to land use. The model embodies mature and robust principles and requires a modest amount of input data, easily accessible to stakeholders and non-LCA practitioners. On-field measurements of climatic and non-climatic parameters should be executed, to derive more realistic regional-dependent characterisation factors, improving the reliability of the results obtained in the impact assessment phase.

The applicability and functionality of this method is illustrated by using *E. globulus* stands (first rotation) in Portugal as a case study. The results have shown that different impacts on terrestrial green water flows and on surface blue water production are obtained, depending on the alternative reference land use. Furthermore, the case study shows that the methodology developed can be a useful tool for assisting in national *E. globulus* forest planning.

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CHAPTER 3

Chapter 3: Impacts of suspended solids on the aquatic environment

Introduction

This chapter encompasses both a framework to conduct a spatially distributed SS delivery to freshwater streams using the WaTEM/SEDEM model (inventory) (Chapter 3.1), and a newly developed method to derive site-specific CFs for damage to the aquatic biota (impact assessment) (Chapter 3.2). A *Eucalyptus globulus* case study was performed to illustrate the applicability of the framework and the method developed.

In the first paper, the detached and entrainment of eroded topsoil by water was called eroded soil particles. However, in the course of research presented in this chapter, and following the recommendation of experts on soil erosion issues, in the last two papers of this chapter they have used the concept of SS to refer to the detached and entrainment of eroded topsoil transported by runoff towards the drainage network.

To properly understand the direct effects of the SS stressor on the potential loss of aquatic macroinvertebrate, algae and macrophyte, a collaboration with an expert on ecotoxicology at the Department of Biology of the University of Aveiro has been extremely helpful. In the second paper, the derived spatially explicit endpoint CFs have units of PDF.m^3.day.mg^{-1}. The PDF.m^3.day.mg^{-1} represents the fraction of species disappearing from 1 m^3 of earth freshwater volume during 1 day per 1 mg of SS in the water column.

Based on both newly derived CFs and a framework to conduct the spatial differentiated SS inventory, a realistic case study on *E. globulus* stands located in Portugal was conducted (Chapter 3.3). Primary data were provided by RAIZ. In addition, to properly understand the operation of the WaTEM/SEDEM model and to develop a deeper understanding of the topsoil erosion issues, a collaboration with experts of University of Leuven has also been established.

3.1. A framework for modelling the transport and deposition of eroded particles towards water systems in a Life Cycle Inventory

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LAND USE IN LCA

A framework for modelling the transport and deposition of eroded particles towards water systems in a life cycle inventory

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Bradley G. Ridoutt · Luís Arroja**

Abstract

Purpose: Topsoil erosion due to land use has been characterised as one of the most damaging problems from the perspective of soil-resource depletion, changes in soil fertility and net soil productivity and damage to aquatic ecosystems.

On-site environmental damage to topsoil by water erosion has begun to be considered in Life Cycle Assessment (LCA) within the context of ecosystem services. However, a framework for modelling soil erosion by water, addressing off-site deposition in surface water systems, to support Life Cycle Inventory (LCI) modelling is still lacking.

The objectives of this paper are to conduct an overview of existing methods addressing topsoil erosion issues in LCA and to develop a framework to support LCI modelling of topsoil erosion, transport and deposition in surface water systems, to establish a procedure for assessing the environmental damage from topsoil erosion on water ecosystems.

Methods: The main features of existing methods addressing topsoil erosion issues in LCA are analysed, particularly with respect to LCI and Life Cycle Impact Assessment methodologies. An overview of nine topsoil erosion models is performed to estimate topsoil erosion by water, soil particle transport through the landscape and its in-stream deposition. The type of erosion evaluated by each of the models, as well as their applicable spatial scale, level of input data requirements and operational complexity issues are considered. The WATEM-SEDEM model is proposed as the most adequate to perform LCI erosion analysis.

Discussion: The definition of land use type, the area of assessment, spatial location and system boundaries are the main elements discussed. Depending on the defined system boundaries and the inherent routing network of the detached soil particles to the water systems, the solving of the multifunctionality of the system assumes particular relevance.

Simplifications related to the spatial variability of the input data parameters are recommended. Finally, a sensitivity analysis is recommended to evaluate the effects of the transport capacity coefficient in the LCI results.

Conclusions: The published LCA methods focus only on the changes of soil properties due to topsoil erosion by water. This study provides a simplified framework to perform an LCI of topsoil erosion by considering off-site deposition of eroded particles in surface water systems.

The widespread use of the proposed framework would require the development of LCI erosion databases. The issues of topsoil erosion impact on aquatic biodiversity, including the development of characterisation factors are now the subject of on-going research.

Keywords: Life Cycle Inventory, surface water systems, topsoil erosion

1. Introduction

Topsoil provides ecosystem services essential for life. It acts as a water depuration filter and as a substrate for growing food, fibre and biomass; it provides habitats for multiple organisms, contributing to biodiversity and it provides a foundation for construction activities. Topsoil erosion is a natural and complex process that varies around the world depending on climate, land use, soil texture, ground slope, vegetation cover, rainfall patterns, and field-level conservation practices (Montgomer 2007). Awareness of the on-site impacts of accelerated topsoil erosion began as early as the 1920s (e.g. Bennett and Chapline 1928) and since then, topsoil erosion has become a worldwide major environmental issue. This process occurs in three stages: detachment and entrainment of soil particles, their transport and eventual deposition. Topsoil erosion has impacts that are both on-site (the place where the topsoil is detached) and off-site (wherever the eroded soil particles reach the surface water systems). The eroded soil particles may come from several sources: soil erosion (by water, wind and tillage), mining and construction activities (Grimm et al. 2002).

Because of the very slow rate of soil formation, any soil loss of more than 1 $t.ha^{-1}.yr^{-1}$ can be considered as irreversible within a time span of 50-100 years. On average, the soil erosion rate for cropland is of the order of 7.5 $t.ha^{-1}.yr^{-1}$ (Grimm et al. 2002; Wilkinson 2005). Water is one of the major causes of soil erosion (Jones et al. 2012a) and the average soil erosion by water in Europe is about 2.8 $t.ha^{-1}.yr^{-1}$ (Jones et al. 2012a,b) affecting 60% of the total land area excluding Russia. However, losses as high as 20-40 $t.ha^{-1}.yr^{-1}$ have been measured in individual storms in Europe (Jones et al. 2012a).

In the last decades, the environmental damage to topsoil by water erosion has begun to be considered in Life Cycle Assessment (LCA) within the context of ecosystem services (EC JRC 2010; Guinée et al. 2006). Special attention has been given to changes in soil properties at the local level, which may include loss of soil nutrients, salinisation, changes in soil organic matter, reduction of soil depth, increase of stoniness and reduction of water-holding capacity.

However, there are also off-site impacts that affect aquatic biota caused by the eroded soil particles that are transported and deposited into surface water systems and into downstream soil. To date, these off-site impacts have not been considered in LCA, which is a substantial failing considering the scale of the problem.

The existence of Life Cycle Inventory (LCI) data on soil erosion by water is crucial for the consideration of the associated impacts in Lie Cycle Impact Assessment (LCIA). Currently, there is no regionalised and/or local database of topsoil erosion by water. In addition, the spatial variability of soil texture and topographic parameters at a regionalised/local scale hampers the establishment of a conventional LCI. Therefore, to perform an unambiguous LCI of topsoil erosion, considering these spatial variabilities, it is necessary to use models based on geographic information systems (GIS).

Several different models for predicting topsoil erosion by water and the transport of the detached soil particles into the surface water systems, for different scales of catchment areas, have been developed (Kirkby et al. 2004; Merrit et al. 2003; Van Rompaey et al. 2001a). However, these models have not been developed specifically for LCI purposes; they are applied primarily to watershed and river basin management issues, as well as to evaluate changes in water quality resulting from flood events and to assess siltation issues in catchment areas and navigable channels.

In this study, in order to understand how topsoil erosion issues have been considered in LCA, an overview of existing methods addressing topsoil erosion is conducted. Furthermore, to support LCI modelling and to fill the existing gap in current soil erosion research within the context of LCA, a framework for modelling soil erosion by water and deposition in water surface systems is proposed. Firstly, an overview of the range of different models available for estimating the quantity of soil particles detached and entrained by water, their transport through the landscape and ultimate deposition into the surface water systems is performed. Subsequently, the topsoil erosion model that best fits the inventory of an LCA study is suggested. The constraints of the suggested model are underlined and with the goal of operationalising the LCI modelling, some proper simplifications are recommended.

2. Overview of methods addressing topsoil erosion in LCA

Because of the scale of soil erosion, the integration of topsoil erosion issues in the LCA structure assumes significant relevance, contributing to the establishment of environmental strategies for soil protection (e.g. CEC 2006).

Fig. 3.1 illustrates the general cause-effect chain of topsoil erosion and showing the damage to human health, and terrestrial and aquatic ecosystems. The topsoil erosion process causes local changes to soil properties and leads to off-site deposition of eroded soil particles in surface water systems and downstream soil. This last pathway can bring off-site benefits for the downstream soil. When nutrients from the topsoil are transported from one place to another, the nutrients are redistributed downstream, which can lead to the increase of fertilisation of a downstream nutrient-deficient soil.

Table 3.1 presents the methods developed for assessing the impact of topsoil erosion within the LCA context. All the methods: Cowell and Clift (2000), Mattsson et al. (2000), Muys and García Quijano (2002), Garrigues et al. (2012) and Núñez et al. (2010, 2013) focus on topsoil erosion by water. However, Muys and García Quiano (2002) acknowledge that when relevant, both wind and tillage erosion should be considered. For wind erosion, the authors suggest using wind erosion equations following the methodology of Schwab et al. (1993), whereas for tillage erosion, no recommendations are forthcoming. With regard to the level of damage, all methods analyse the local effects of topsoil erosion, i.e. the changes to the soil properties pathway (Fig. 3.1), focusing mainly on soil quality and soil depth indicators.

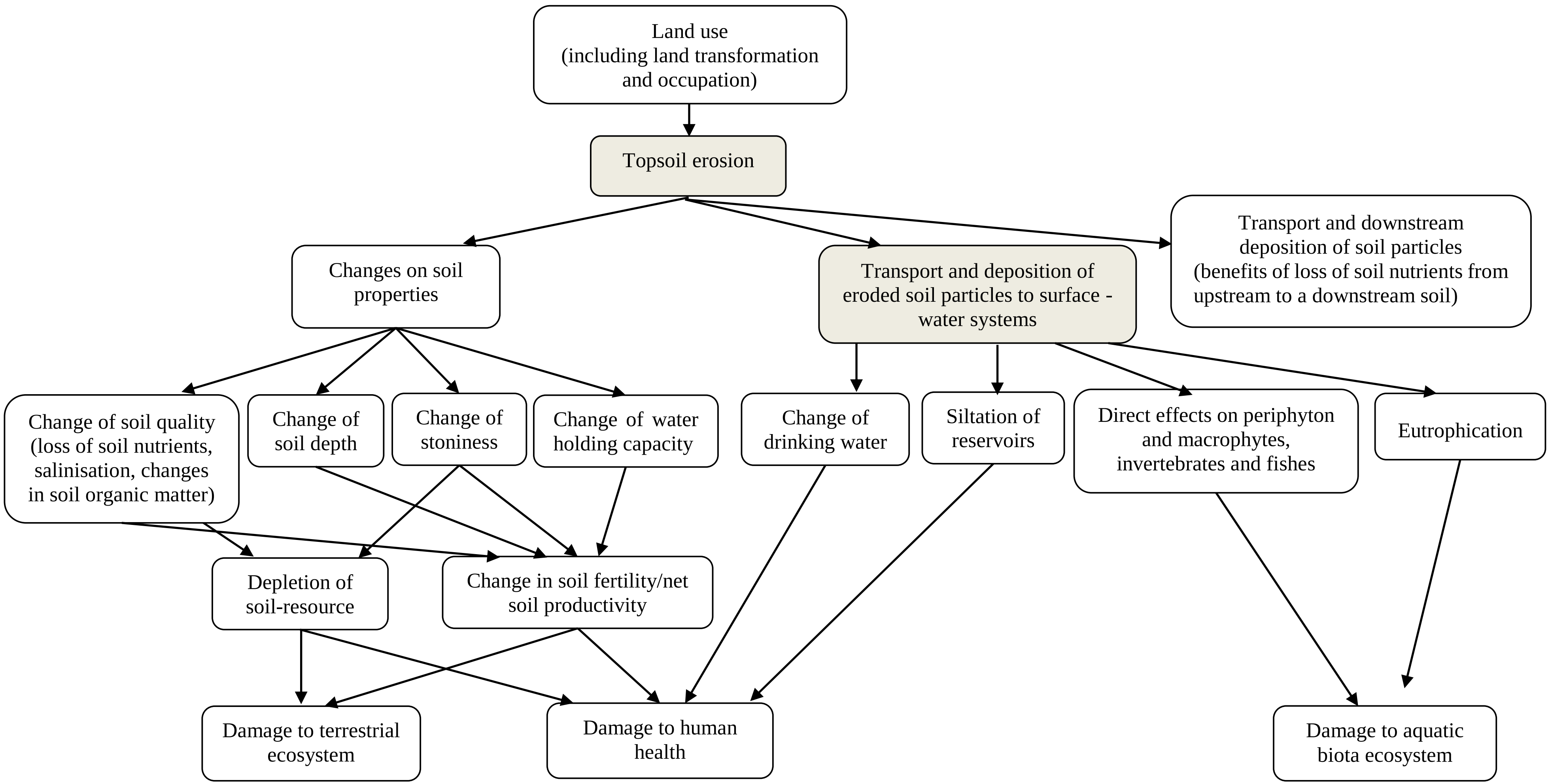

Fig. 3.1. Cause-effect chain of topsoil erosion impacts. The grey boxes are related to the pathway focused on this study.

Table 3.1. Overview of methods of topsoil erosion issues in the LCA structure.

Method	Cowell and Clift (2000)	Mattsson et al. (2000)	Muys and Garcia Quijano (2002)	Garrigues et al. (2012)	Núñez et al. (2010)	Núñez et al. (2013)
Cause of soil erosion	Water	Water	Water	Water	Water	Water
Level of damage	Local	Local	Local	Local	Local	Local
Impact pathway	Changes on soil properties – change of soil quality	Changes on soil properties – change of soil quality	Changes on soil properties – change of soil depth	Changes on soil properties	Changes on soil properties	Changes on soil properties – change of soil quality
Assessment level	Midpoint	Midpoint	Midpoint	Midpoint	Midpoint	Endpoint
Impact category	Depletion of soil resource	Soil fertility	Soil	Soil	Desertification	Damage to terrestrial ecosystem due to depletion of soil resource
Intervention	Loss of soil ($t.ha^{-1}.yr^{-1}$)	Loss of soil ($t.ha^{-1}.yr^{-1}$)	Loss of soil ($t.ha^{-1}.yr^{-1}$)	Loss of soil ($t.ha^{-1}.yr^{-1}$)	Loss of soil ($t.ha^{-1}.yr^{-1}$)	Loss of soil ($t.ha^{-1}.yr^{-1}$)
Erosion model	USLE	Not provided	USLE	RUSLE	USLE	USLE
Category indicator	Soil Static Reserve Life (SSRL) – total global soil reserve (m) in relation to current annual global net loss of topsoil ($t.yr^{-1}$)	Loss of soil	Loss of soil	Loss of soil	Loss of soil	Loss of soil
Characterisation model	SSRL	Unweighted aggregation	Soil depth loss over 100 years compared with total rootable soil depth	Unweighted aggregation	Desertification taking into account the spatial area subjected to desertification, the decimal logarithm of the ecoregion area where desertification occurs and the time of perturbation	Loss of soil as the local available soil reserves normalised with reference soil depth with transformation to emergy units per unit of area and time of land use
Characterisation factor (CF)	Not provided	1 for all interventions (dimensionless)	Not provided	1 for all interventions (dimensionless)	1 or 2 for interventions depending on the ecoregion under analysis (dimensionless)	Spatial CF at resolution 10 × 10 km^2 ($MJ_{se} \times g^{-1}_{soil\ loss}$)

Núñez et al. (2013) applied the model following an endpoint approach, whereas the models proposed by the other authors evaluate the effects of topsoil erosion at the midpoint level. At the midpoint level, these methods proposed different impact categories such as depletion of the soil resource, soil fertility and desertification are proposed depending on the method, whereas at the endpoint level, the area of protection from damage to the terrestrial ecosystem is assessed. Several of these methods are not dedicated exclusively to topsoil erosion damage. In Muys and García Quijano (2002) and Garrigues et al. (2012), the indicator of the loss of soil is a sub-indicator in the soil sub-impact category; however, in Mattsson et al. (2000), the loss of soil is a sub-indicator in the soil fertility sub-impact category. These methods evaluate the topsoil erosion issues as a sub-category of the broader impact category of land use. In addition, at the endpoint level, Núñez et al. (2010) consider soil erosion as one of the four indicators (aridity, aquifer overexploitation, fire risk and erosion) for the desertification impact category.

At the LCI phase, in the Cowell and Clift (2000), Muys and García Quijano (2002) and Núñez et al. (2013, 2010) methods, the amount of topsoil eroded by water has been estimated by the Universal Soil Loss Equation (USLE) (Wischmeier and Smith 1978). Garrigues et al. (2012) use the Revised Universal Soil Loss Equation (RUSLE) (Renard et al. 1997). Both equations quantify the locally eroded soil particles but do not consider the runoff of these particles. However, the RUSLE equation presents some improvements in relation to the USLE, such as new calculation procedures to account for slope length and steepness, additional sub-factors for evaluating the cover and management factors for cropland and rangeland and new conservation practice values for cropland and rangeland (Jones et al. 1996). Further details about the RUSLE are given in Section 4.

With regard to the characterisation model, Cowell and Clift (2000) considered the Soil Static Reserve Life that is a function of the global reserves of agricultural soil (total topsoil in the world, in t) and current annual global net loss of topsoil mass by erosion (in $t.yr^{-1}$). Muys and García (2002) recommend transforming the loss of soil mass into a loss of soil depth (in m) using the bulk density of the soil. Therefore, the loss of soil depth over a period of 100 years is compared with the total rootable soil depth up to 1 m. However, neither method provides a set of operational characterisation factors (CFs). In fact, to derive such site specific CFs, data on the reserve of the topsoil (area and depth of soil) under analysis is required. Both Garrigues et al. (2012) and Mattsson et al. (2000) use the loss of topsoil as an indicator for erosion impact. These authors argue that the estimates of loss of topsoil in the inventory level are informative enough to serve as impact indicators without requiring CFs. The characterisation model suggested by Núñez et al. (2010) takes into account the spatial area subjected to desertification, the decimal logarithm of the ecoregion area where desertification occurs and the time of perturbation. Moreover, this method proposes CFs for eight large natural areas, i.e. ecoregions (Marine, Prairie, Temperate steppe, Temperate desert, Savanna, Mediterranean, Tropical/subtropical steppe and Tropical/subtropical desert) that consider the loss of topsoil and terrain definition/mass movement in each ecoregion soil weighted by the surface area of the ecoregion. Núñez et al. (2013) proposed an alternative characterisation model that considers the ultimate damage to the depletion of the soil resource by assessing the loss of soil as the locally available soil reserves normalised with a reference soil depth of 3 m with transformation to emergy units per unit of area and time of land use. A set of regional CFs were developed at a spatial resolution of 10×10 km^2, taking into account the locally available soil reserves, the reference soil depth and the solar energy factor (the quantity of solar energy that is required to generate a gram of eroded soil particles) (Núñez et al. 2013).

To our knowledge, there are no available LCI/LCIA (Life Cycle Impact Assessment) methods to assess the off-site effects due to topsoil erosion by water, i.e. the transport and deposition of eroded soil particles into surface water systems and the transport of soil particles from one place to another downstream (Fig. 3.1). Therefore, this study focuses on how to estimate topsoil erosion and to understand the transport of soil particles at the inventory level, evaluating the net quantity of soil particles that reaches the surface water systems.

3. Models of topsoil erosion

There are several purposes behind performing LCI modelling of topsoil erosion:

- to describe as accurately as possible the topographic conditions of the land use system under analysis and the types of erosion that can occur (sheet, rill and gully erosion)
- to characterise the soil profile, identifying the fraction of organic matter and the range size of the detached soil particles that reach the surface water systems
- to understand the influence of tillage operations on the quantity of eroded soil

- to quantify the quantity of detached soil particles that are transported and deposited into the surface water systems at sub-watershed level.

The topsoil erosion models that have become used most widely for predicting eroded soil particles and their transportation and deposition into surface water systems are:

- ANSWERS – Areal Nonpoint Source Watershed Environment Response Simulations (Beasley et al. 1980; Dillaha et al. 2001)
- CREAMS – Chemical Runoff and Erosion from Agricultural Management System (Knisel 1980)
- GUEST – Griffith University Erosion System Template (Rose et al. 1997; Yu et al. 1997)
- HSPF – Hydrological Simulation Program Fortran (Johanson et al. 1980)
- TOPOG (Gutteridge & Davey 1991)
- WEPP – Watershed Erosion Prediciton Project (Laflen et al. 1991)
- MIKE-11 (Hanley et al. 1998)
- WaTEM/SEDEM – Water and Tillage Erosion Model – Sediment Delivery Model (Van Oost et al. 2000; Van Rompaey et al. 2001a)
- EUROSEM – European Soil Erosion Model (Borselli and Torri 2010; Morgan et al. 1998).

The goals, characteristics and limitations of each model are identified in Table 3.2. These models differ significantly regarding the type of erosion, their temporal and spatial scales, level of input data requirements, ease of use and in the algorithms used to predict the amount of detached and transported soil particles.

The spatially distributed ANSWERS model was developed to evaluate the effect of agricultural land use practices on water quality of sub-watersheds. This model predicts the topsoil erosion by adopting the physics-based continuity equations used by Foster and Meyer (1972). The transport of soil particles and nutrients is estimated using a form of Yalin’s (1963) equation for bed load transport. The application of this model is limited by the large input data requirements and it only considers sheet and rill erosion, disregarding gully erosion, which can be as extensive as the former two. Sheet erosion occurs by the detachment and entrainment of soil particles, which are then transported downslope by overland runoff (Hairsine and Rose 1992); whereas rill erosion occurs when the field slope increases, such that the overland runoff gains velocity, forming superficial channels (Rose 1993).

The CREAMS model estimates the topsoil erosion and deposition of soil particles applying the USLE (Wischmeier and Smith 1978). The transport of soil particles is simulated in the overland flow using a steady-state continuity equation (Foster's equation) at a plot sub-watershed level (range of 40-400 ha). The unlikely assumption that all areas of the study are uniform in soil topography is a pertinent source of uncertainty and inaccuracy in the predicted topsoil erosion.

The GUEST model predicts erosion, transport and deposition using physics-based equations describing the steady-state soil particle flux (Rose 1993). The model was developed to be applicable at plot sub-watershed level (< 100 ha). Despite being an adequate spatial scale to perform an LCI around agricultural or forest land use, this model assumes uniform soil topography and does not consider tillage operations.

In addition to topsoil erosion, both the HSPF and MIKE-11 models allow the assessment of in-stream water quality parameters (e.g. nitrogen, phosphorus, inorganic pollutants) at the watershed level. The HSPF model is applicable at the watershed level, in which the spatial area is divided into sub-regions with homogeneous hydrologic characteristics, i.e. homogenous edaphoclimatic data. The MIKE-11 model is also applicable at the watershed level and the water system flow is described using physics-based St Venant equations.

Both models require large amounts of input data and overlook the relevance of gully erosion in a similar way to the ANSWERS and GUEST models. Gully erosion describes deep channels of runoff water and soil particles.

The TOPOG model describes water movement through the landscape; over the land surface, into the soil, through the soil and groundwater and back to the atmosphere via evaporation. Topsoil erosion is simulated and the transport of soil particles is simulated following the Engelhund and Hansen (1968) equation. It disregards gully erosion processes and it requires large and detailed input data.

The WEPP model evaluates net soil loss as well as the effects of hydraulic structures and impoundments on runoff flow and soil particle transport. It assesses the effects of tillage operations on the quantity of soil particles deposited into the surface water systems. The erosion and transport components are determined using the same relationships as the ANSWERS model. It tends to over-predict the transport of soil particles for small events and under-predict their transport for large events (Yu 2005). Gully erosion is disregarded at plot sub-watershed level, which does not happen at the watershed level.

The WaTEM/SEDEM model predicts the long-term mean annual topsoil erosion following the RUSLE (Renard et al. 1997). It is applicable at plot sub-watershed and watershed levels and it allows the evaluation of the influence of tillage operations on the quantity of eroded soil. Sheet, rill and ephemeral gully erosion processes are considered.

The EUROSEM model predicts topsoil erosion and simulates soil particle transport to the surface water systems at the plot sub-watershed level. However, it only simulates single intense rainfall-runoff events, i.e. long-term simulation of multiple rainfall-runoff events is a performance constraint. It overlooks the relevance of gully erosion in a similar way to the ANSWERS, GUEST, HSPF, MIKE-11 and TOPOG models.

Table 3.2. Goal, characteristics and limitations of models to predict topsoil erosion and the transport of soil particles to surface-water systems.

Models	Goal and characteristics	Limitations
ANSWERS	▪ Developed to evaluate the effects of agricultural land use practices in the on-stream water quality ▪ Applicable at sub-watershed level (> 400 ha) ▪ A long-term oriented model using a continuous time step (e.g. days) ▪ Considers sheet and rill erosion	▪ Disregards gully erosion ▪ Complex model requiring large input data (e.g. river network, total porosity of soil, infiltration control zone depth, erodibility, steady state infiltration, among others) ▪ Does not take into account tillage operations ▪ Requires calibration and validation
CREAMS	▪ Evaluates the effects of agricultural practices on pollutants in surface rainfall-runoff and topsoil erosion ▪ Applicable at plot sub-watershed level (range of 40-400 ha) ▪ Operates either on an event basis or long-term using a continuous time step (e.g. days) ▪ Includes sheet, rill and ephemeral gully erosion In addition to the prediction of topsoil erosion, also predicts the evapotranspiration of the crop, soil infiltration, among others	▪ Predicts topsoil erosion following the one-dimensional USLE approach ▪ Assumes uniform topography (slope) of soil and land use ▪ Does not take into account tillage operations ▪ Complex model with large input data requirements (e.g. precipitation series, monthly air temperature and solar radiation values, crop type data, among others)
GUEST	▪ Developed to understand temporal fluctuations in soil particle concentration at the transport limit ▪ Applicable at plot sub-watershed level (≤100 ha) ▪ An event-oriented model ▪ Considers sheet and rill erosion	▪ Assumes uniform topography (slope) of soil ▪ Very complex model requiring large input data (e.g. runoff rate, length, width and slope of the soil, percent of sand grains of primary particles, soil particle/water-stable aggregate size distribution, among others) ▪ Does not take into account tillage operations ▪ The general lack of input data means that considerable effort is required to use it to predict topsoil erosion without prior calibration against parameterisation of field measurements

Table 3.2. Goal, characteristics and limitations of models to predict topsoil erosion and the transport of soil particles to surface-water systems. (cont. I)

HSPF	▪ Developed to simulate the hydrology of watersheds and its in-stream water quality ▪ Applicable at watershed level (~1,000 ha) ▪ Operates either on an event basis or long-term using a continuous time step (from 1 min to 1 day, as long as the time step divides equally into 1 day) ▪ Includes sheet, rill and ephemeral gully erosion ▪ Takes into account tillage operations	▪ As inputs requires of precipitation data, estimates of potential evapotranspiration, topography, solar radiation, humidity, among others ▪ Relies greatly on calibration against parameterisation of field measurements
TOPOG	▪ Simulates the soil particles balance through the landscape until the reaching the surface water system ▪ Simulates the transient hydrologic behaviour of watersheds and how this is affected by changing watershed vegetation and by the growth of the vegetation ▪ Applicable at plot sub-watershed and watershed levels (range of 10-1000 ha) ▪ Temporal resolution can be applied at daily time steps or sub-daily time steps ▪ Uses detailed topographic information that can be provided by a digital elevation model of the area of study ▪ Considers sheet and rill erosion	▪ Disregards gully erosion ▪ Does not take into account tillage operations ▪ Requires calibration
WEPP	▪ Developed to estimate net soil loss, the effect of hydraulic structures and impoundments on runoff flow and soil particle transport. ▪ There are two versions: one that is applicable at plot sub-watershed (≤ 100 ha) and a second applicable at watershed level (~1,000 ha) ▪ Operates either on an event basis or long-term using a continuous daily step ▪ Includes sheet, rill and ephemeral gully erosion ▪ Takes into account tillage operations	▪ Large computational and input data requirements (e.g. leaf area index, canopy cover and height, duration of runoff, bulk density of the soil, among others) ▪ Relies on calibration against parameterisation of the field measurements

Table 3.2. Goal, characteristics and limitations of models to predict topsoil erosion and the transport of soil particles to surface-water systems. (cont. II)

MIKE-11	▪ Used to predict topsoil erosion and to model the in-stream water quality ▪ Applicable at watershed level ▪ Operates either on an event or long-term basis ▪ Uses detailed topographic information that can be provided by a digital elevation model of the area of study ▪ Considers sheet and rill erosion	▪ Large input data requirements (e.g. hydrometric and topographic data, moisture content in surface and root zone, among others) ▪ Does not take into account tillage operations ▪ Low quality data of the input parameters ▪ Relies greatly on calibration against parameterisation of the field measurements
WaTEM/SEDEM	▪ Used to simulate eroded soil particles by the RUSLE method considering a two-dimensional landscape structure and the transport and deposition of soil particles into surface water systems ▪ Applicable at plot sub-watershed and watershed levels ▪ A long-term average annual deposition of soil particles in the surface-water systems ▪ Focuses on the spatial resolution of 20 × 20 m ▪ Includes sheet, rill and ephemeral gully erosion ▪ Uses detailed topographic information that can be provided by a digital elevation model of the area of study ▪ Takes into account tillage operations ▪ Requires a modest amount of input data parameters ▪ The parameter-based nature of the model allows an easy analysis of the contribution of individual parameters (sensitivity analysis) to the predicted topsoil erosion and deposition	▪ Low quality data of the input parameters ▪ Requires partial calibration on the soil particles transport capacity
EUROSEM	▪ Predicts topsoil erosion ▪ Applicable at plot sub-watershed level (≤100 ha) ▪ Operates on an event basis considering a temporal resolution of minute-by-minute	▪ Only considers sheet and rill erosion ▪ Large input data requirements (rainfall data, soil mechanical properties, topographical information, micro topographical information, soil properties and vegetation information) ▪ Does not take into account tillage operations ▪ Low quality data of the input parameters ▪ Requires calibration and validation

4. Life Cycle Inventory modelling of topsoil erosion and deposition

4.1. Model suggestion and its overall structure

Because of the local or regional nature of topsoil erosion, the spatial differentiation assumes particular relevance. A model that performs an erosion-deposition inventory at a plot scale and uses site-dependent parameters allows the reduction of the uncertainty of the erosion LCI compared with generic default data on a broader scale. The ANSWERS, HSPF and MIKE-11 models have been developed to be applicable on a broad spatial scale. In addition, these models present operational complexity, require large amounts of input data, evaluate in-stream water quality, which is not an aim of the soil-deposition inventory and require calibration. Although the ANSWERS model considers gully erosion and allows the inclusion of tillage operations, it is only applicable at the watershed level, i.e. the scale of assessment is approximately 1,000 ha, hindering the topsoil analysis of a local land use system. For these reasons, these models are not adequate to perform an LCI of an agricultural or forestry system on a plot scale. The CREAMS and GUEST models also do not fit the erosion-deposition LCI purposes. Despite being applicable at a localised scale, these models consider uniform soil topography. Any model that uses a digital elevation model (DEM) to define the two-dimensional slope length parameter will reduce the underestimation of the erosion-deposition inventory of a non-uniform soil and land use. The shape of a slope affects the average topsoil eroded and the transport along the hill slope to the surface water system. For instance, according to Wischmeier and Smith (1978), the eroded soil from a convex slope can easily be 30 % greater than that from a uniform slope. In addition, these models disregard gully erosion and need calibration.

The EUROSEM model also does not fit the LCI purpose. It operates on an event basis, i.e. by each rainfall-runoff event. This means that to estimate the average eroded soil particles transported to the surface water systems during a sown-harvested crop land use study or forest land use study, numerous simulations should be performed to obtain the cumulative erosion-deposition inventory. In addition, gully erosion is not considered and it requires a large amount of input data, as mentioned in the previous section.

The WEPP model could be adequate for erosion-deposition inventory purposes as it is applicable at a plot sub-watershed level, operates on a long-term basis, considers gully erosion and takes into account the influence of tillage operations in the erosion and transportation of soil particles. However, the large computational and input data requirements and the calibration needed for many of the model parameters (e.g. slope, land cover, soil type) are shortcomings preventing its widespread use in LCI studies.

The WaTEM/SEDEM model has the same characteristics as the WEPP model but requires fewer input data parameters. It is the only model that uses a RUSLE (Renard et al. 1997) in considering the variability of soil texture and topographic parameters (elevation, slope, upslope, profile curvature). This acquires crucial relevance in establishing the routing network transport of soil particles to the water systems. Indeed, the primitive form of this equation, the USLE (Wischmeier and Smith 1978) and its update, the RUSLE, have been well accepted by the LCA community for estimating the annual mass of topsoil loss, as mentioned in Section 2. However, both the primitive equation and RUSLE account only for the detached and entrained stages of the soil particles. The significant variability of soil texture and topographic parameters suggests the need for GIS to account for a less crude and unambiguous inventory of the detached soil particles deposited into the surface water systems. Indeed, performing a geo spatial LCI is a step forward from conventional LCI studies. After considering the main characteristics and constraints of each model regarding performing an LCI of topsoil erosion, presented in Section 2, it is suggested that the use of the WaTEM/SEDEM model is most appropriate.

The WaTEM/SEDEM is a simple topography-driven model (Fig. 3.2) that uses the RUSLE (Eq. 1) to estimate the topsoil eroded by water per land area (Renard et al. 1997):

$$A = R \times K \times LS_{2D} \times C \times P \qquad \text{(Eq.1)}$$

where A is the average eroded soil per year ($kg.m^{-2}.yr^{-1}$), R is the rainfall-runoff erosivity parameter ($MJ.mm.m^{-2}.h^{-1}.yr^{-1}$), K is the soil erodibility parameter ($kg.ha.h.ha^{-1}.MJ^{-1}.mm^{-1}$) that represents the soil resistance to erosion by water and is related to the soil characteristics, namely the structure, texture, organic matter content and permeability, LS_{2D} is the two-dimensional slope-length parameter (dimensionless), C is the crop management parameter (dimensionless) allowing an understanding of the extent to which vegetation cover prevents soil erosion and P is the support-practice parameter to reduce runoff and soil erosion (dimensionless).

The tillage erosion process is also simulated by the WaTEM/SEDEM model following a diffusion-type equation (Eq. 2) developed by Govers et al. (1994):

$$Q_{s,t} = -k_{till}\frac{dh}{dx} \qquad \text{(Eq.2)}$$

where $Q_{s,t}$ is the rate of net downslope soil transport per tillage translocation (kg.m^{-1} per tillage operation), k_{till} is the tillage transport coefficient (kg.m^{-1} per tillage operation), h is the height at a given point of the hill slope (m) and x is the distance in the horizontal direction (m).

Unlike water erosion, topsoil displacement will only occur within the field, i.e. it is not transported directly to the surface water systems. However, these detached soil particles can contribute to the increase in the quantity of soil particles that are transported by overland runoff flow. This means that soil tillage operations have a non-negligible influence on topsoil erosion (Govers et al. 1994) and thus, they should be included in an LCI study.

The transport of the detached soil by overland runoff to the surface water systems is calculated according to Eq. 3. It is assumed that the transport capacity is proportional to the potential gully erosion (Desmet and Govers 1995; Van Oost et al. 2000; Van Rompay et al. 2001a):

$$Tc = ktc \times R \times K \times (LS_{2D} - 4.12 \times Sg^{0.8}) \qquad \text{(Eq.3)}$$

where Tc is the transport capacity (kg.m^{-1}.yr^{-1}), ktc is the transport capacity coefficient (m), R, K and LS_{2D} are the parameters from the RUSLE and Sg is the local slope gradient (dimensionless).

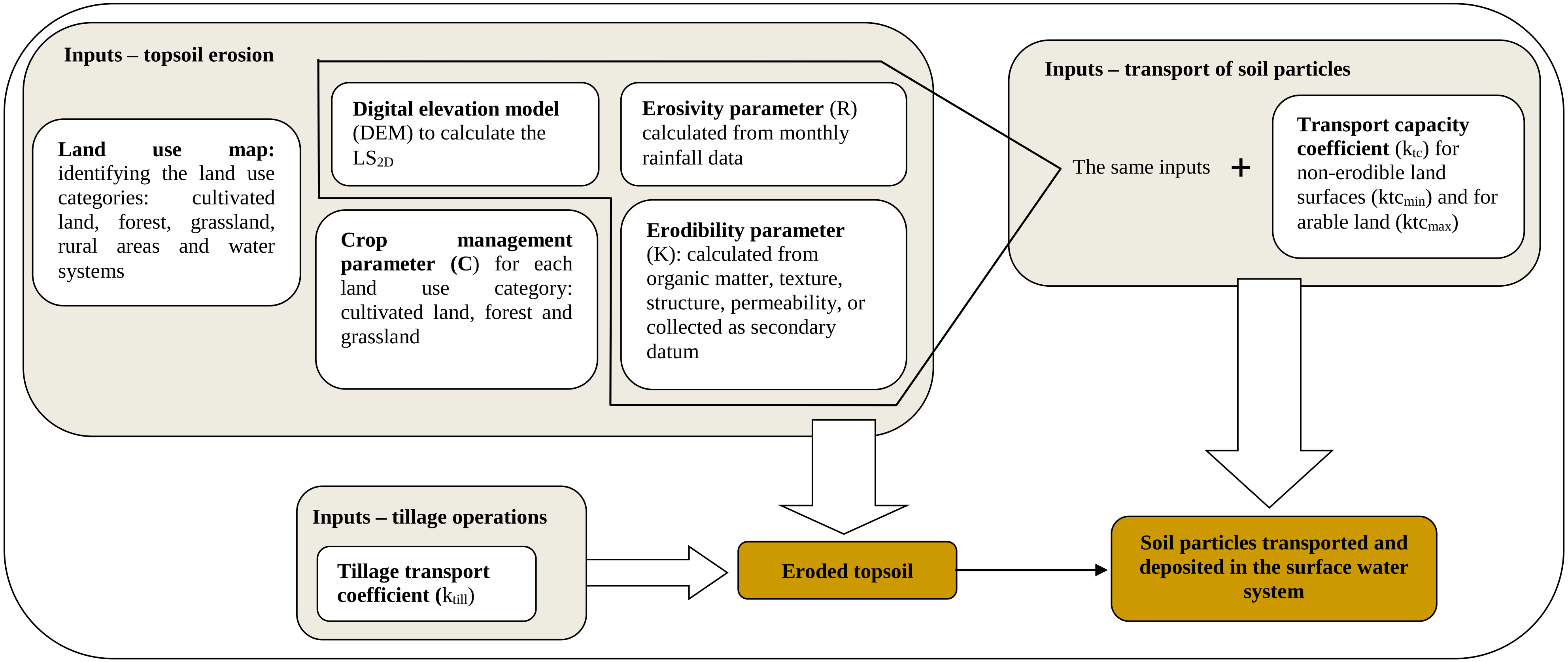

Fig. 3.2. An overview of the WaTEM/SEDEM model structure distinguishing the topsoil eroded ($kg.m^{-2}.yr^{-1}$), the transport of soil particles by overland runoff and their deposition in surface water systems ($kg.m^{-1}.yr^{-1}$).

4.2. Guidelines and simplifications

In this section, the guidelines, methodological simplifications and recommendations for the LCI modelling of topsoil erosion using the WaTEM/SEDEM model are identified and described.

4.2.1. Geographical area and multifunctionality

To estimate the quantity of detached soil particles from a land use system that reach the surface water systems, the geographical land area should be defined. Moreover, the LCI should include the classification of the land use type under analysis.

After the definition of the geographical land area, the next step is to delineate the related DEM, which represents the topographical variability of the soil. In addition to the GIS information, it is also necessary to establish a soil particle routing network, which represents the runoff flow through the landscape towards the surface water systems. For this, a land use map identifying the different land use types (for instance, cultivated land, forest, grassland, rural areas and water system) is required as an input to the WaTEM/SEDEM model (Alatorre et al. 2012). The establishment of the soil particle routing network can raise the problem of incoherence between the pre-defined geographical land area under study and the overall downstream landscape, because the delineated DEM of the land use system under study may not match the land use map. To overcome this constraint, the DEM should be expanded to include all the downstream land area towards to the surface water system. This highlights a system multifunctionality problem, because an overestimated erosion-deposition inventory is performed, i.e. it includes soil particles detached and transported from the land use system under study and downstream land use systems. As a result, in most cases, is impossible to conduct an LCI study through a single run of the WaTEM/SEDEM model. In order to assign the quantity of detached soil particles from the pre-defined land use system under study and its delivery to the surface water systems, a system subdivision should be carried out, as illustrated in Fig. 3.3. By considering the agricultural/forest land area under study and the downstream land area routing to the surface water system, two co-functions are provided by this unique system (AB) (the topsoil erosion process from the system under study and the topsoil erosion process from the downstream landscape). Fig. 3.3 shows that the inventory of the pre-defined land use system (system A) results from the subdivision of the overall system

(AB). In practice, the LCI practitioner must have in mind that the spatially distributed WaTEM/SEDEM model needs to be run at least twice to establish the inventory: the first run accounts for the inventory of the overall system (AB) and the second run accounts for the inventory of system B.

The exception occurs when the pre-defined land use system is located in the water system bank. In this case, it is reasonable to assume that all the eroded soil is transported to the water. In this case, the erosion-deposition inventory can be performed by the direct use of the RUSLE.

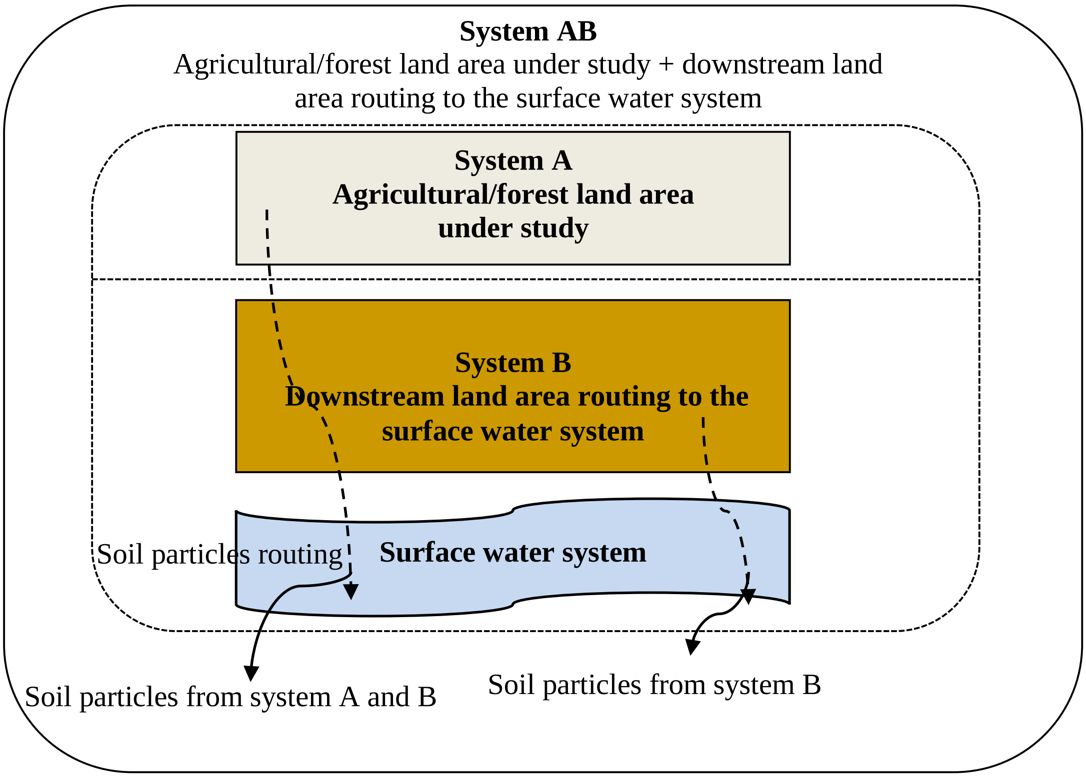

Fig. 3.3. Solving the system multifunctionality by subdivision of system AB. Subdivision aims to obtain the LCI topsoil erosion of system A.

4.2.2. Input data requirements and simplifications

The input data requirements to perform an LCI modelling using the WaTEM/SEDEM model are the soil erodibility parameter, rainfall-runoff erosivity parameter, two-dimensional topographical parameter, cover management parameter, tillage transport coefficient and the transport capacity coefficient of the soil particles.

Soil erodibility parameter (K)

The soil erodibility parameter (K), as mentioned previously, describes the susceptibility to erosion by rainfall and it depends on the texture, structure, organic matter, permeability and the physicochemical interactions of the soil. Generally, the availability of detailed primary soil data and soil erodibility parameters at the local scale are scarce. Therefore, instead of a soil erodibility map with a 20 × 20 m spatial resolution, the LCI practitioner can assume that the overall system (AB) has a single and a long-term average K parameter.

Depending on the soil profile, the average K parameter can be estimated following different approaches. If the silt fraction of soil does not exceed 70 %, the K value (t.ha.h.ha^{-1}.MJ^{-1}.mm^{-1}) can be estimated using the nomograph developed by Wischmeier and Smith (1978) or by applying its algebraic approximation (Eq. 4) (Renard et al. 1997):

$$K = \frac{\left(\frac{2.1\times10^{-4}(12\text{-OM})M^{1.14} + 3.25(s\text{-}2) + 2.5(p\text{-}3)}{100}\right)}{7.59} \qquad \text{(Eq.4)}$$

where OM is the fraction of organic matter (%), M is the particle-size parameter: (percentage silt + fine sand fraction content) × (100-clay fraction) and s and p are the soil structure (dimensionless) and permeability classes (dimensionless), respectively.

If the soil profile has more than 70 % silt, the K parameter can be calculated following Eq. 5 (Wischmeier and Smith 1978):

$$K = 7.594\left\{0.0034 + 0.0405\,\exp\left[-0.5\left(\frac{\log D_g + 1.659}{0.7101}\right)^2\right]\right\} \qquad \text{(Eq.5)}$$

where D_g is the geometric mean weight diameter of the primary soil particles (mm), which can be measured experimentally by analysing soil samples. This relationship is only validated for soil with less than 10 % of rock fragments by weight (fraction higher than 2 mm). When this equation is not valid, it is recommended to collect the input K parameter datum from literature (e.g. Borselli et al. 2009; Gómez et al. 2003; Irvem et al. 2007; Sanchis et al. 2008), national or international research projects (e.g. Van der Knijff et al. 2000) and soil databases attending to the texture of the soil under study. For instance, depending on the soil profile, this secondary datum can be collected from the Land Use and Cover Area frame Survey database (Panagos et al. 2012).

Rainfall-runoff erosivity parameter (R)

The rainfall-runoff erosivity parameter (R) represents the impact of rain on topsoil erosion and it is assessed based on monthly rainfall data that can be collected from meteorological stations within the land area under study and for a long-term period (for instance 20-30 years). When there are no meteorological stations within the land area, the average monthly precipitation data can be taken from the New_LocClim model (FAO 2005), which provides estimations of average climatic conditions at locations for which no observations are available, considering the nearest neighbour interpolation method. In these situations, the R parameter can be calculated using Eqs. 6, 7 and 8 (Renard and Freimund 1994):

$$MF = \frac{\sum_{i=1}^{12} p_i^2}{P} \qquad \text{(Eq.6)}$$

$$R = 0.07397 \times MF^{1.847}\text{, for } MF < 55\text{mm} \qquad \text{(Eq.7)}$$

$$R = 95.77\text{-}6.081 \times MF + 0.4770 \times MF^2\text{, for } MF > 55\text{mm} \qquad \text{(Eq.8)}$$

where MF is the Modified Fournier index (mm) (Arnoldus 1977), p_i is the average monthly precipitation (mm) and P is the average annual precipitation (mm). Attending to the area of assessment, i.e. plot sub-watershed level, it is recommended to assume that the spatial variability of parameter R is negligible. Therefore, the long-term variability of R on a yearly basis is considered uniform for the entire study area.

Two-dimensional topographical parameter (LS_{2D})

The two-dimensional topographical parameter (LS_{2D}) shows the impact of the length and slope of the landscape on soil erosion and it has a large value whenever the length and slope of the landscape are large. In WaTEM/SEDEM, the DEM is used to calculate the LS_{2D} parameter.

Crop management parameter (C)

The crop management parameter (C) reflects the effect of cropping and management practices on erosion rates, i.e. it indicates the extent to which the vegetation cover prevents topsoil erosion. This parameter depends on the size of cover plants, the state of the surface area, plant roots, surface roughness and amount of contained water (Park et al. 2011). There is very limited availability of C parameters for different crops and/or forests at different locations (Núñez et al. 2013). Because of this lack of data, Núñez et al. (2013) developed a methodology to estimate specific C parameters for agricultural and/or forestry systems. However, this approach requires a follow-up field to calculate the percentage of vegetative cover throughout the sown-harvest process. To overcome this time-consuming procedure, we suggest using single values concerning the land use categories included in the defined overall system (AB), as well as in system B: crop, forest and/or grassland. Even though this increases the uncertainty of the inventory results, it is the best way in which to minimise the complexity of calculating C parameters at the plot scale. If the system (AB) encompasses various types of crops and/or forest land use, a weighted average between each culture cover management and the occupied area should be performed. The C parameters can be collected from scientific publications (e.g. Keesstra et al. 2009; Lee and Lee 2006; Yang et al. 2003), books and reports (e.g. Almorox et al. 1994; Pimenta 2003; Van der Knijff et al. 2000).

Transport capacity coefficient of soil particles (ktc)

The transport capacity coefficient of soil particles (ktc) depends on the local land vegetative cover and represents the slope length needed to runoff an equivalent quantity of topsoil particles from a bare surface with a similar slope gradient (Verstraeten 2006). The WaTEM/SEDEM model requires distinction between arable land surfaces, ktc_{max} (cultivated land) and non-erodible land surfaces, ktc_{min} (forests and/or grassland). Moreover, Van Rompaey et al. (2001a) have indicated that a calibration of transport capacity coefficient would be desirable because of the unique and specific routing network of soil particles for each land use system. In fact, land use systems with high rainfall erosivity, also present high runoff flows and soil particle deposition into the surface water system resulting in higher transport capacity (Verstraeten et al. 2007). Because of the unavailability of sufficient data to calibrate ktc_{max} and ktc_{min} separately, Verstraeten (2006) suggested maintaining a fixed ratio of ktc_{max}/ktc_{min} values. For instance, the following typical values between ktc_{min} and ktc_{max} have been used: 1:3.33 in central Belgium and the southwestern part of Slovenia (Keesstra et al. 2009; Verstraeten et al. 2006; Verstraeten 2006) and in Spain (Alatorre at al. 2010); values ranging between 1:3.80 and 1:2.20 for mountainous and non-mountainous areas in Italy (Van Rompaey et al. 2005); 1:3.89 for seven small catchments in South Africa (Van Rompaey et al. 2001b; Verstraeten et al. 2001) and; 1:2.50 for the Czech Republic (Van Rompaey et al. 2003). The LCI practitioner should compare the topographic characteristics (e.g. LS_{2D}) of the land use system under study with the characteristics of the several regions mentioned above. Regions with similar topographic characteristics should be characterised by similar ktc_{max}/ktc_{min} ratios.

Data on annual suspended soil particles input into the water system channels under study, as well as data on rainfall erosivity for the same time series, are required to perform the calibration. For each combination of ktc_{max} and ktc_{min}, a quantity of soil particles delivered into the surface water system is predicted, allowing the comparison of the quantities predicted by the model and those measured. The Nash-Sutcliffe model efficiency statistic (NS) (Nash and Sutcliffe 1970) is used as a measure of likelihood following Eq. 9:

$$NS = 1 - \frac{\sum_{i=1}^{n}(O_i - P_i)^2}{\sum_{i=1}^{n}(O_i - \bar{O})^2} \qquad \text{(Eq.9)}$$

where n is the number of observations, O_i is the measured soil particles in the water system, P_i is the predicted soil particles in the water system and $\bar{O}$ is the average of the measured soil particles. NS can range from $-\infty$ to 1 and represents the proportion of the initial variance accounted for by the model. The closer the value of NS is to 1, the more efficient is the model (Alatorre et al. 2010).

Tillage transport coefficient (k_{till})

The intensity of topsoil erosion is influenced by the tillage transport coefficient (k_{till}). Experimental tillage erosion studies have shown that tillage erosivity is affected by parameters other than slope gradient, such as tillage depth, tillage speed and soil condition (Gerontidis et al. 2001). Van Muysen et al. (2000) presented a dataset of k_{till} calculated for different implements, tillage speed and tillage depth. In addition, k_{till} following the up-and down-slope tillage direction for each implementation can be calculated following Eq. 10 (Van Muysen et al. 2002):

$$k_{till} = 2.026 \times \rho_b \times D^{1.989} \times v^{0.406} \qquad \text{(Eq.10)}$$

where ρ_b is the bulk density of the soil ($kg.m^{-3}$), D is the tillage depth (m) and v is the tillage speed ($m.s^{-1}$). For a set of tillage operations during the sown-harvest cycle, the tillage transport coefficient is the sum of the coefficients for each individual operation.

When the system (AB) encompasses cultures that require different tillage operations, a weighted average tillage transport coefficient should be used.

4.2.3. Complexity of LCI modelling and uncertainty parameters

When defining the scope of the study, special attention should be given to the definition of the geographic system boundary, identifying and reporting the location of the land use system. On the one hand, depending on the spatial location of the land use, different modelling procedures can be followed. In some situations, as described in Section 4.2.1, a system subdivision is required to solve the multifunctional and to predict the erosion-deposition inventory of the pre-defined land use system.

On the other hand, depending on the area of assessment, it can include different land use types and huge variability of soil properties. From an ideal perspective, a map that could show the variability of each input parameter necessary to estimate the quantity of topsoil eroded would be desirable.

However, currently, this is an unrealistic procedure because of the non-availability of the required input data. First, there are no specific or site-generic LCI databases containing soil erodibility and cover management maps. In particular, data related to cover management parameters for specific cultures and locations are quite rare. Secondly, although it is possible to perform experimental measurements during the overall growth of the crop/forest from sowing to harvest, this is impracticable from a time and economic point of view. Therefore, to overcome these constraints and to operationalise the LCI inventory, it seems reasonable to use an average value for each input parameter, i.e. constant values instead of maps.

The overall uncertainty of the erosion-deposition LCI is caused by the empirical inaccuracy of the input data, unrepresentative data, lack of data sets and the inherent uncertainties of the model. Perhaps the biggest problem with the modelling of eroded soil and its transport to the surface water systems, related to the availability and reliability of input data, is that the transport capacity coefficient is a major source of uncertainty and inaccuracy in an LCI performed with the WaTEM/SEDEM model (Van Rompaey et al. 2001a; Verstraeten et al. 2006). To understand the influence of a set of different ktc_{max}/ ktc_{min} ratios on the mass of soil particles detached and deposited a sensitivity analysis is recommended. Moreover, the influence of other input parameters, for instance, K, C and

k_{till}, which are considered constant and/or adapted from literature, should also be evaluated.

5. Conclusions

The existing LCA-based methods addressing soil erosion focus only on the local changes of soil properties. They differ in the way that they quantify erosion at the LCI level and in the LCIA methodology, namely in the assessment level (midpoint/endpoint), impact pathway, category indicator, characterisation model and characterisation factors. None of these methods considers the transport and deposition of the eroded particles towards the surface water systems. This study provides the first step for the inclusion of this issue in the LCA structure. The conducted overview of erosion models shows that there is a range of available models for predicting the quantity of detached soil particles and their transport towards the surface water systems. Following consideration of their operational complexity, input requirements and spatial scales, it is been suggested that the WaTEM/SEDEM model is most appropriate for performing an LCI analysis. Therefore, this paper provides a framework with which to perform an LCI of the topsoil eroded and transported towards the surface water systems. This operational framework has the following key points:

- definition of the spatial land area of assessment (system boundary) stating its geographical location;
- when necessary, a system subdivision should be performed to avoid allocation procedures, depending on the pre-defined land use systems (allocation procedures);
- collecting all the required input data to run the WaTEM/SEDEM model;
- performing a sensitivity analysis to evaluate the influence of the transport capacity coefficient on the LCI results.

This framework seems a reasonable approach to establish realistic estimations of the order of magnitude of topsoil erosion and the amount of soil particles that reach the surface water systems, causing turbidity and in-stream soil particle deposition, which in turn affect aquatic biota.

It should be noted that this framework should be operationalised by performing case studies. Moreover, displaced soil particles are themselves a source of potential environmental harm, especially when they reach water systems. Therefore, a method to evaluate the impact of topsoil erosion on aquatic biodiversity within the LCIA is also required and it is the subject of on-going research.

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3.2. Suspended solids in freshwater systems: characterisation model describing potential impacts on aquatic biota

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LAND USE IN LCA

Suspended solids in freshwater systems: characterisation model describing potential impacts on aquatic biota

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Abstract

Purpose: High concentration of suspended solids (SS) – fine fraction of eroded soil particles – reaching lotic environments and remaining in suspension by turbulence can be a significant stressor affecting the biodiversity of these aquatic systems. However, a method to assess the potential effects caused by SS on freshwater species in the Life Cycle Impact Assessment (LCIA) phase still remains a gap.

This study develops a method to derive endpoint characterisation factors, based on a fate and effect model, addressing the direct potential effects of SS in the potential loss of aquatic invertebrate or algae and macrophyte species.

Methods: Characterisation factors for the assessment of the direct effects of SS in the potential disappearance of macroinvertebrates, algae and macrophytes in 22 different European river sections were derived by combining both fate and effect factors. Fate factors reflect the environmental residence time of SS in river sections per unit of water volume in this same section.

Effect factors were calculated from an empirical relationship between the potentially disappeared fraction (PDF) of aquatic species and the concentration of SS. These factors were determined based on a concentration-response function, on gross soil erosion data, and detrimental concentrations of SS for different taxa in river sections.

Results and Discussion: The product of fate with effect factors constitutes the characterisation factors for both macroinvertebrates, algae and macrophytes. The estimated EFs are higher for macroinvertebrates in almost all river sections under study, showing that the potential effects caused by SS throughout the water column are higher for macroinvertebrates than for algae and macrophytes.

For macroinvertebrates, characterisation factors range between 2.8×10^{-7} and 3.1×10^{-3} PDF.m^3.day.mg^{-1}, whereas for algae and macrophytes, they range between 1.6×10^{-7} and 4.7×10^{-4} PDF.m^3.day.mg^{-1}.

Conclusions: The developed method and the derived characterisation factors enable a consistent assessment and comparison of the potential detrimental effects of SSs on aquatic invertebrates and macrophytes communities at different locations.

Long-term, on-site monitoring of SS levels in the water column should be performed to understand the magnitude of the effects of SS on aquatic biota, and to determine the taxa that are more sensitive to the SS stressor. This monitoring will improve the robustness of the proposed Life Cycle Inventory (LCI) and LCIA method, the reliability of the characterisation factors, as well as the development of characterisation factors for a wider range of rivers.

Keywords: algae and macrophyte communities, macroinvertebrates, erosion, fate and effect modelling, Life Cycle Impact Assessment, water footprint, water quality

1. Introduction

Suspended solids (SS), i.e. suspended fine particulate matter with a diameter lower than 62 µm (Jones et al. 2012a), in freshwaters may arise from several sources: soil erosion (by water, wind, tillage operations, and deforestation), mining, and/or from construction activities (Kefford et al. 2010). Soil erosion caused by water has been characterised as one of the most upsetting problems in water systems (Jones et al. 2012a; Ricker et al. 2008). The European average water soil erosion is about 2.8 $t.ha^{-1}.yr^{-1}$ (Jones et al. 2012a), which, during episodic storms, can easily achieve a rate of 20 $t.ha^{-1}.yr^{-1}$ (Jones et al. 2012b). SS reach freshwater systems, contributing to the sustainability of the aquatic biodiversity due to sediment-associated nutrient transport. However, high concentrations of SS, particularly clays and silts, can be significant stressors to the ecological integrity of the aquatic ecosystems, degrading the water quality and directly affecting the aquatic biota, namely primary producers, such as algae and macrophytes, macroinvertebrates, and also fish (Angermeier et al. 2004; Collins et al. 2011; Jones et al. 2012c; Kasai et al. 2005).

For macroinvertebrates, high concentrations of SS can damage the gills, small appendages, or filter feeding structures of these organisms, leading to lethal and sub-lethal effects. Also, grazing macroinvertebrates can be affected, since SS, particularly clay materials, can be stuck in algae, hampering their feeding behaviour (Bilotta and Brazier 2008).

For algae and macrophytes, the turbidity caused by SS reduces the required light penetration through the water column. This decrease of light availability can reduce photosynthetic rates, leading to adverse effects on these primary producers (Parkhill and Gulliver 2002). Moreover, high concentrations of SS combined with high water flow rates can strongly affect algae communities, damaging, by abrasion, the photosynthetic structures of these organisms, as well as dragging them downstream (Allan and Castillo 2007; Luce et al. 2010).

With respect to fish, the effects of SS may arise by the abrasion and the clogging of fish gills, and indirectly by reducing the abundance of their food (Kefford et al. 2010; Richardson and Jowett 2002).

Although the potential effects of SS on aquatic invertebrates/primary producers have been considered in earlier studies (e.g. Kirk and Gilbert 1990; Levine et al. 2005; Quinn et al. 1992; Soeken-Gittinger et al. 2009), these effects have not yet been considered by the Life Cycle Assessment (LCA) research community. Quinteiro et al. (2014) developed a framework to support a Life Cycle Inventory (LCI) modelling of topsoil erosion, transport, and deposition of eroded solids in surface-water systems, but a method to assess the detrimental potential effects caused by SS on freshwater remains a gap in the Life Cycle Impact Assessment (LCIA) phase.

The purpose of this study is to develop a method to derive endpoint characterisation factors (CFs), based on a fate and effect model, addressing the direct effects of SS in the potential disappearance of aquatic species at different locations. The fate factor (FF) expresses the change in the exposure concentration of SS in a river section due to a change of SS delivered to this same section (Van Zelm et al. 2009). The effect factor (EF) reflects the change in the potentially disappeared fraction (PDF) of aquatic species due to a change in the concentration of SS in a river section (Van Zelm et al. 2009). CFs for macroinvertebrates, algae and macrophytes in a set of 22 river sections, distributed throughout Europe were developed. Fishes are also appropriate indicators of ecological status of lotic ecosystems, and some research to assess the effects of SS on salmonid fish, mainly on salmon and trout, have been conducted (e.g. Alabaster and Lyod 1982; Harrod and Theurer 2002; Newcombe and MacDonald 1991). However, salmon is now extinct in the most European rivers (WWF 2011), and no representative detrimental data on concentration of SS and related effects on other fish species are available. Therefore, fishes were excluded from this study.

2. Methods

The SS delivered to rivers can lead to the loss of biodiversity in these ecosystems, following the impact pathway illustrated in Fig. 3.4.

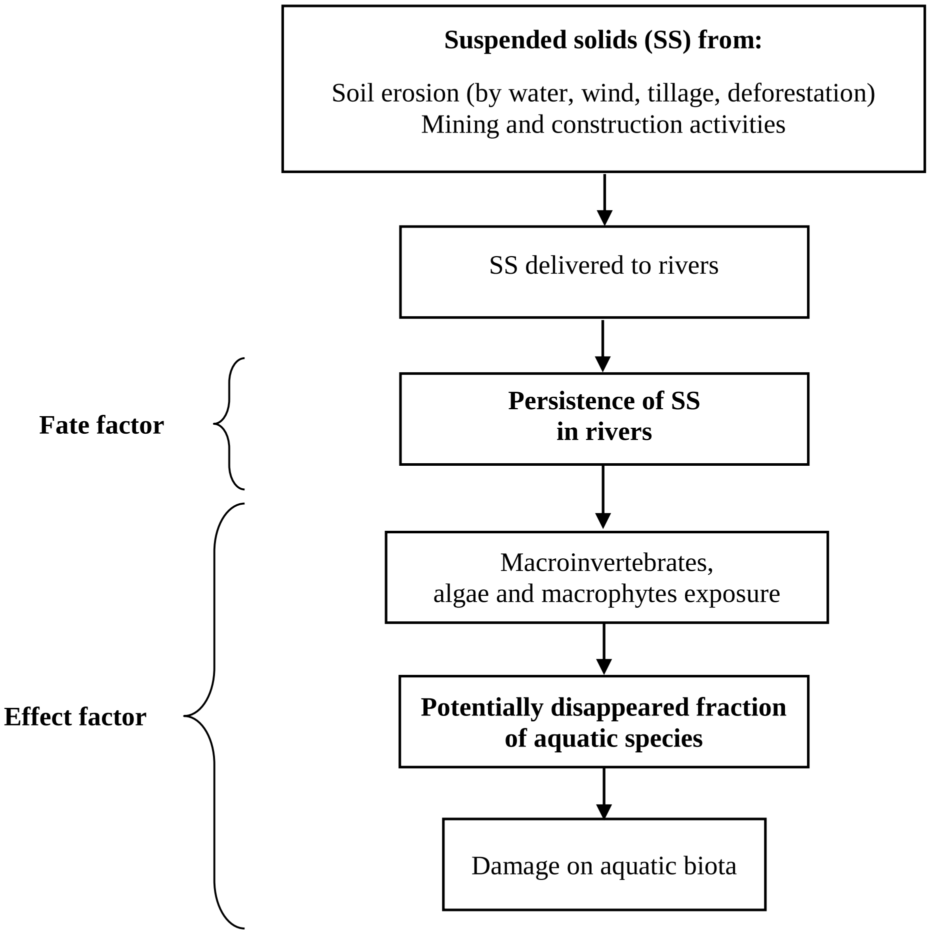

Fig. 3.4. Schematic representation of the impact pathway on ecosystem quality related to suspended solids delivered to rivers.

A fate and effect model is proposed to calculate site-specific CFs, which address the direct potential effects of SS in aquatic biota. This model allows a thorough assessment of local environmental damage, as the CFs take into account the specific river flow, volume of water, and concentration of SS in the water column for the different river sections.

The endpoint characterisation factor in river section *i* (CF_i), expressed in $PDF.m^3.day.mg^{-1}$, expresses the fate of the SS stressor (reflecting the persistence of SS in the aquatic environment) and the potential effects in aquatic species occurring in the average water volume of river section *i*, as shown in Eq. 1.

$$CF_i = FF_i \times EF_i \times V_i \qquad \text{(Eq.1)}$$

where FF_i (in $day.L^{-1}$) is the fate factor for river section *i*, EF_i (in $PDF.L.mg^{-1}$) is the effect factor for river section *i*, and V_i (in m^3) is the average water volume of river section *i*. Combining CF_i with the environmental intervention in the LCI, i.e. the quantity of SS (in mg per functional unit – FU) delivered to river section *i*, the results of the potential effects in aquatic ecosystems return in PDF.m^3.day.FU^{-1} units. The inventory parameter can be estimated following the approach developed by Quinteiro et al. (2014).

The proposed fate and effect model is based on the modelling principles of the ReCiPe , USES-LCA and Impact 2002+ methods (Goedkoop et al. 2012; Huijbregts et al. 2000; Humbert et al. 2012), as explained throughout the following sections.

2.1. Fate factors

The FF*i* reflects the environmental residence time of SS in river section *i* per unit of water volume in that section. The residence time of SS in the water column depends strongly on the flow rate and particle-size distribution along the river. SS loads with a fine particle-size distribution present a long residence time, which can be further increased due to episodes of resuspension caused by turbulence forces in the water, resulting in sub-lethal and lethal responses of aquatic species over a long-term. The residence time of SS in a river section is independent of aquatic species living in that river section, which means that the residence time of SS is equal for all the macroinvertebrate, algae and macrophyte living in the same river section *i*.

The fate factors represent the marginal increase in the concentration of SS in river section *i*, $\Delta C_{SS,i}$ (in mg.L^{-1}), due to a marginal increase of the emission rate of SS in river section *i*, ΔE_i (in mg.day^{-1}), as shown in Eq. 2.

$$FF_i = \frac{\Delta C_{SS,i}}{\Delta E_i} \qquad \text{(Eq.2)}$$

However, data of in-situ concentrations of SS, C_{SS} (in mg.L^{-1}), in river section *i* are not available, hampering the understanding of the annual variation of $\Delta C_{SS,i}$ and its implication for aquatic biota, although some environmental government agencies recognize the relevance of SS as an important pollutant in rivers (Bilotta and Brazier 2008). To overcome this constraint, we propose that $\Delta C_{SS,i}$ can be estimated based on the increased rate of SS throughout river section *i* (ΔLSS_i, in mg.day^{-1}) and the average flow rate of river section *i* (Q_i, in L.day^{-1}), as indicated in Eq. 3.

$$\Delta C_{SS,i} = \frac{\Delta LSS_i}{Q_i} \qquad \text{(Eq.3)}$$

The ΔLSS_i is related to the emission rate of SS to river section *i*, ΔE_i, minus the rate of sediment deposition. Marginal increments of SS delivered to the water column result in marginal changes in the deposited sediment that is found on the bed of the river. However, these fine and unconsolidated sediments in combination with turbulence forces in the water result in an increase of SS due to resuspension. In fact, measures of deposited SS into river beds are also scarce and frequently limited, mainly due to the influence of the resuspension of sediments. The variation in volume and frequency of rainfall and runoff events affect the turbulence forces in the water, and, therefore, the extent of resuspension, which cause fluctuations in the measurements of deposited sediments (Chon et al. 2012; Taylor and Owens 2009; White 2008), as well as the sampling procedure to collect deposited sediments (UNEP/WHO 1996). Therefore, most of the studies on sediment transport to rivers consider that the quantity of SS reaching the outlet of rivers is equal to the delivery of SS to rivers (e.g. Alatorre et al. 2010; de Vente and Poesen 2005; Lane et al. 1997; López-Tarazón et al. 2009; Moatar et al. 2006). Consequently, we have also assumed that the marginal increment of eroded SS delivered to river section *i* remains in suspension in the water column (i.e. the fine particulate fraction, SS). This means that ΔE_i can be equated to ΔLSS_i. Therefore, FF_i is given by Eq. 4.

$$FF_i = \frac{\Delta LSS_i}{\Delta E_i \times Q_i} = \frac{1}{Q_i} \qquad \text{(Eq.4)}$$

2.2. Effect factors

The EF considers the marginal changes in ecological damages due to a marginal change in an environmental stressor (Hanafiah et al. 2011; Verones et al. 2010), according to Eq. 5.

$$EF = \frac{\partial PDF}{\partial \text{environmental stressor}} \qquad \text{(Eq.5)}$$

The PDF is defined as an empirical function that is developed based on the principles of the species sensitivity distribution (SSD) approach (Posthuma et al. 2002). The SSD describe the joint sensitivity distribution for different species to a particular stressor, following a concentration-response function. The SSDs are based on available toxicological data and are commonly used in risk assessment to derive thresholds in terms of the fraction of species potentially affected by a certain chemical concentration (PAF) or a hazardous concentrations affecting x-th percentile of species (HCx) (Larsen and Hauschild 2007; Motulsky and Cristopoulos 2003; Posthuma et al. 2002).

As no data are available to analyse the distribution of median lethal concentrations (LC_{50}) or the no-observed-effect concentrations (NOECs) for different invertebrates, algae and macrophytes taxa exposed to a range of C_{SS}, we propose to use punctual detrimental C_{SS} affecting species of macroinvertebrates, algae and macrophytes.

These detrimental C_{SS} were collected from literature and mainly from field studies.

In this sense, EF_i reflects the change in the PDF of macroinvertebrates, algae and macrophytes in river section i, $PDF_{i,}$, due to a marginal change in $C_{SS,i}$, according to Eq. 6.

$$\mathrm{EF}_i = \frac{\partial \mathrm{PDF}_i}{\partial \mathrm{C}_{\mathrm{SS},i}} \qquad \text{(Eq.6)}$$

The PDF function can be calculated by fitting the commonly used log-logistic distribution (sigmoid function) to the proportion of depleted species, as a function of detrimental C_{SS}, as defined by Eq. 7 (Zajdlik 2006; Zwart 2002).

$$\mathrm{PDF}(\mathrm{C}_{\mathrm{SS}}) = \frac{1}{1 + \mathrm{e}^{-\frac{\log_e \mathrm{C}_{\mathrm{SS}} - \alpha}{\beta}}} \qquad \text{(Eq.7)}$$

where α is the location parameter and also the median of the distribution, and β is the scale parameter estimated from the standard deviation of the log-transformed C_{SS} data.

As referred above, the PDF functions were derived using detrimental $C_{SS,i}$ affecting macroinvertebrates, algae and macrophytes survival, population size or diversity of species, mainly from field studies. Indeed, each point used to derive the PDF function (Eq. 7) is related to a detrimental $C_{SS,i}$, measured in different experiments and for different aquatic species: for macroinvertebrates, detrimental $C_{SS,i}$ values that affect the survival of four juveniles cladocerans species were collected from laboratory experiments performed by Kirk and Gilbert (1990); detrimental $C_{SS,i}$, measured in field experiments, affecting the diversity and survival of several invertebrate taxa were collected from Nuttal and Bielby (1973) and Quinn et al. (1992); *in-situ* measured detrimental $C_{SS,i}$, that affects the population size of benthic invertebrates were collected from Wagener and LaPierre (1985); detrimental $C_{SS,i}$ that affect the survival of mysid shrimps were obtained from laboratory experiments conducted by Nimmo et al. (1982).

For algae and macrophytes, detrimental $C_{SS,i}$, affecting the survival and diversity of macrophytes, phytoplankton, and periphyton communities, were collected from field studies conducted by Lloyd et al. (1987), Quinn et al. (1992), and Van Nieuwenhuyse and LaPierre (1986), respectively.

Fitting the log-logistic function to the collected detrimental $C_{SS,i}$ (Fig. 3.5), the PDF_i for macroinvertebrates was obtained ($\alpha = 4.90$ and $\beta = 0.890$), as shown in Eq. 8.

$$\mathrm{PDF}(C_{SS}) = \frac{1}{1 + e^{-\frac{\log_e C_{SS} - 4.90}{0.890}}} \qquad \text{(Eq.8)}$$

With the purpose of simplification, Eq. 8 was mathematically rearranged, as shown in Eq. 9.

$$\mathrm{PDF}_i = \frac{1}{1 + C_{SS,i}^{-1.12} \times e^{5.51}} \qquad \text{(Eq.9)}$$

Therefore, taking into account Eqs. 6 and 9, the EF_i for macroinvertebrates is given by Eq. 10.

$$\mathrm{EF}_i = \frac{e^{5.51}}{C_{SS,i}^{2.12} \times 0.89 \times \left(\frac{e^{5.51}}{C_{SS,i}^{1.12}} + 1\right)^2} \qquad \text{(Eq.10)}$$

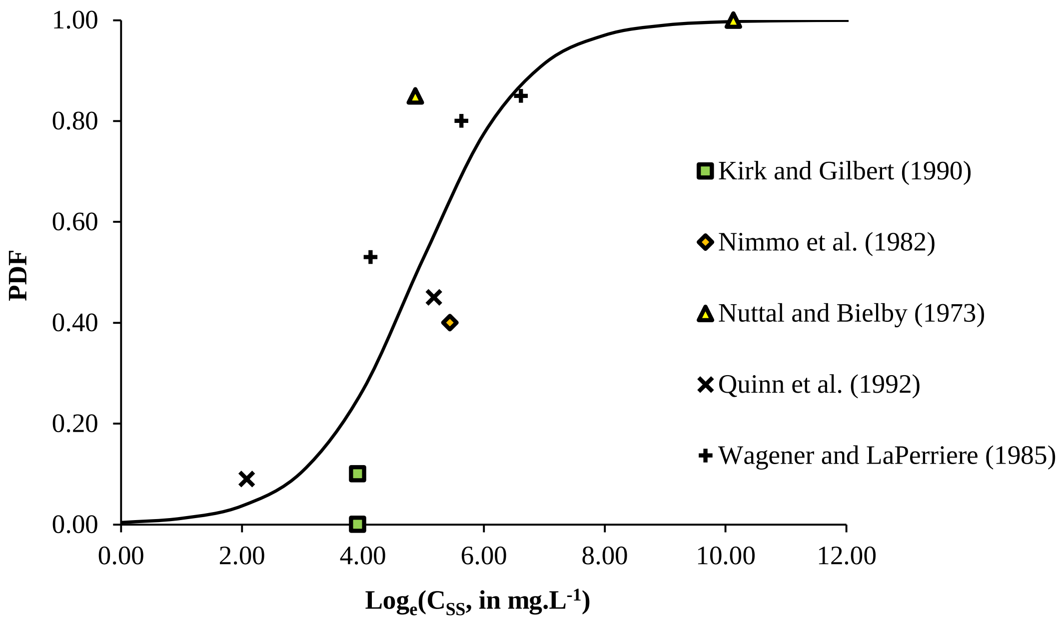

Fig. 3.5. The potential disappeared fraction (PDF) of aquatic macroinvertebrates versus the natural logarithm of the average concentration of suspended solids, C_{SS} (in $mg.L^{-1}$).

For algae and macrophytes, the PDF_i was obtained by a fitted log-logistic function through the collected C_{SS} ($\alpha = 4.73$ and $\beta = 1.50$), as shown in Fig. 3.6 and Eq. 11.

$$\mathrm{PDF}(C_{SS}) = \frac{1}{1 + e^{-\frac{\log_e C_{SS} - 4.73}{1.50}}} \qquad \text{(Eq.11)}$$

Eq. 11 was mathematically rearranged, as shown in Eq. 12, in a similar manner as previously obtained for macroinvertebrates.

$$\mathrm{PDF}_i = \frac{1}{1 + C_{SS,i}^{-0.67} \times e^{3.16}} \qquad \text{(Eq.12)}$$

Therefore, taking into account Eqs. 6 and 12, the EF_i for algae and macrophytes was given by Eq. 13.

$$\mathrm{EF}_i = \frac{e^{3.16}}{C_{SS,i}^{1.67} \times 1.50 \left(\frac{e^{3.16}}{C_{SS,i}^{0.67}} + 1 \right)^2} \qquad \text{(Eq.13)}$$

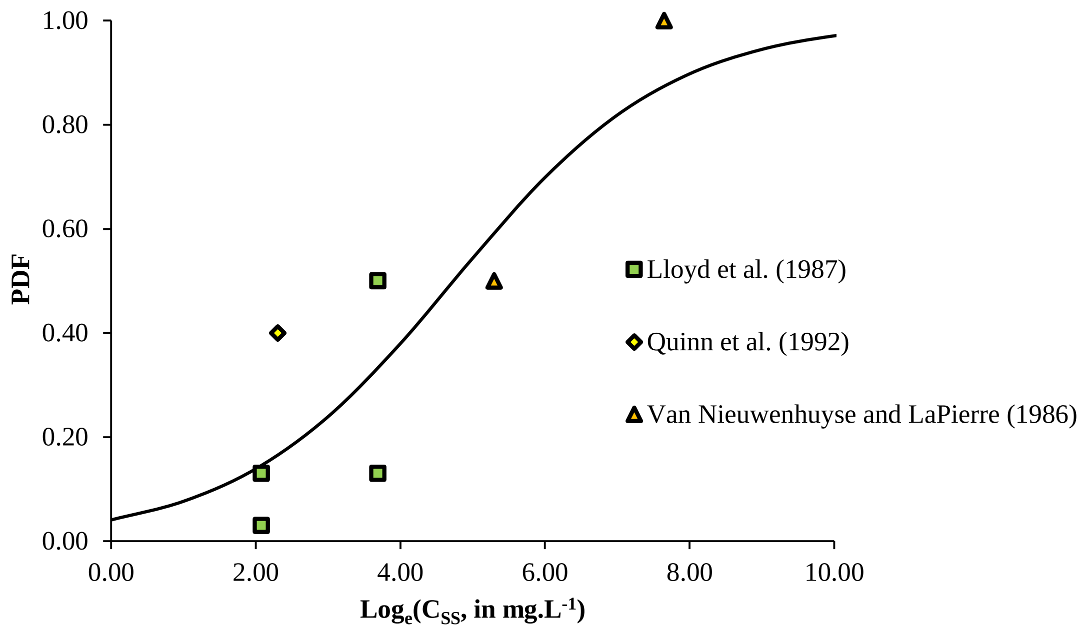

Fig. 3.6. The potential disappeared fraction (PDF) of algae and macrophytes versus the natural logarithm of the average concentration of suspended solids, C_{SS} (in mg.L^{-1}).

The $C_{SS,i}$ can be estimated based on the ratio between the SS delivery to river section *i* and Q_i. The SS delivery to river section *i* is equal to the quantity of SS reaching the outlet of this river section, i.e. sediment yield, as explained in Section 2.1. Moreover, sediment yield corresponds to 20 % of the gross eroded topsoil (Becvar et al. 2010), and their estimations have been provided by modelling-based studies, such as the PESERA – Pan European Soil Erosion Risk Assessment – project (Gobin and Govers, 2003) and G2 erosion model (Panagos et al. 2014, 2012).

The PESERA model estimates mean annual gross soil erosion rates by running water, particularly due to sheet and rill erosion (Vanmaercke et al. 2012), taking into account the effect of topography, climate, and land use.

The G2 model follows the principles of the Universal Soil Loss Equation (USLE) (Wischmeier and Smith 1978), but adopts some modifications, namely regarding the crop management factor, the estimation of rainfall erosivity based on the revised USLE (RUSLE) (Renard et al. 1997), and the consideration of the effect of seasonality in rainfall and vegetation data (Panagos et al. 2014, 2012).

3. Operationalisation: endpoint characterisation factors

The operationalization of the proposed method is illustrated through the calculation of FFs, EFs, and CFs for 22 European river sections. These are the river sections for which there are data available concerning SS delivered to water column and Q_i to allow the estimation of $C_{SS,i}$. It should be noted that these river sections belong to some of the most important European rivers. Table 3.3 presents the characteristics (volume V_i, and Q_i) of each river section. The Q_i data were collected from the River Discharge Database (SAGE 2010). The average flow rate of each river section *i* was measured over periods ranging from 5 to 149 years, depending on the measuring station. The value of V_i was derived from the average water depth, length, and width of each river section *i*. When no specific water depth data was available, an average water depth of 3 m was considered (Goedkoop et al. 2012).

For the estimation of $C_{SS,i}$ needed to develop the EFs, the gross topsoil erosion was taken from Becvar (2006) and was based on the PESERA model and the characteristics of each river section studied, as explained in Section 2.2.

Table 3.3. Average volume of water (V_i), and average flow (Q_i) of each river section *i* (Fekete et al. 2002; Goedkoop et al. 2012; SAGE 2010).

Section	River	Country	V_i (km^3)	Q_i (L.day^{-1})
Drobeta-TurnuSeverin	Danube	Romania	0.55	4.7×10^{11}
Regua	Douro	Portugal	0.09	4.7×10^{10}
Villachica	Douro	Spain	0.11	1.2×10^{10}
Wittenberg	Elbe	Germany	0.03	6.4×10^{10}
Mas d'Agenais	Garonne	France	0.44	5.3×10^{10}
Alcala del Rio	Gualdalquivir	Spain	0.05	3.8×10^{10}
Pulo de Lobo	Guadiana	Portugal	0.05	1.0×10^{10}
Blois	Loire	France	0.15	3.1×10^{10}
Montjean	Loire	France	0.28	7.2×10^{10}
Gozdowice	Odra	Poland	0.06	4.6×10^{10}
Piacenza	Po	Italy	0.09	8.5×10^{10}
Pontelagoscuro	Po	Italy	0.14	1.3×10^{11}
Paris	Seine	France	0.14	2.3×10^{10}
Bewdley	Severn	United Kingdom	3.30	5.6×10^{9}
Almourol	Tagus	Portugal	0.07	3.2×10^{10}
Vila Velha de Rodão	Tagus	Portugal	0.04	2.7×10^{10}
Rome	Tiber	Italy	0.11	2.0×10^{10}
Teddington	Thames	United Kingdom	1.10	7.1×10^{9}
Colwick	Trent	United Kingdom	0.05	1.2×10^{9}
Norham	Tweed	United Kingdom	0.01	7.1×10^{9}
Intschede	Weser	Germany	0.12	2.7×10^{10}
Warsaw	Wisla	Poland	0.03	4.9×10^{10}

4. Results and discussion

The results of the CFs based on FFs and EFs for the studied 22 river sections are presented in Section 4.1. The limitations of the present study and recommendations for further research are presented in Section 4.2.

4.1. Fate, effect, and characterisation factors

The FFs represent the persistence of the SS in the environment and are presented in Table 3.4. These factors are the same for all macroinvertebrates, algae and macrophytes, as the residence time of SS in water column is not related to the biologic traits of these ecosystems, but only to the rate of SS in the river section and also the characteristics of the river section, namely with the average flow rate (Table 3.3), as shown in Eqs. 2 and 3.

The determined FFs range from 2.1×10^{-12} to 8.4×10^{-10} day.L^{-1}. The lowest value was found for the Drobeta–TurnuSeverin section (Danube river), where the persistence of SS is low due to the high average flow rate of this river section, whereas the highest value was found for the Trent–Colwick section (Trent river), where the SS are transported downstream more slowly as this section has the lowest average flow rate.

The effects of SS on aquatic biota are comparable with other impact categories, such as eutrophication that also address potential impact on aquatic biodiversity (Struijs et al. 2013, 2011). Therefore, to compare the spatial explicit FFs due to SS with the also spatial explicit FFs of phosphorous emission to freshwater determined by Helmes et al. (2012), FFs presented in Table 3.4 were converted to days by considering the volume of water of each river section *i*. FFs due to SS show considerable variability, ranging from 0.5 to 574 days. The FFs for freshwater eutrophication, due to phosphorous emissions range from 0.8 to 310 days in non-arid areas (Helmes et al. 2012). The results show that SS can be more persistent in freshwater than phosphorous.

The calculated EFs for macroinvertebrates, algae and macrophytes are presented in Table 3.4. The EFs were derived from empirical functions through the establishment of relationships between C_{SS} and its potential effects on freshwater biota, as explained in Sections 2.2 and 3.2.

At the lowest reported detrimental C_{SS}, 8 mg.L^{-1}, algae and macrophytes are more affected than macroinvertebrates, i.e. at this concentration PDFs of 0.09 and 0.03 were obtained, for algae and macrophytes, and macroinvertebrates, respectively (Nuttal and Bielby 1973, Quinn et al. 1992). For macroinvertebrates, the PDF function increases sharply in the range between 25 (PDF $\geq$ 0.10) and 300 mg.L^{-1} (PDF < 0.70) (Fig. 3.5), meaning that the fraction of affected invertebrates sharply increases within this range. For algae and macrophytes, the fraction of affected organisms sharply increases in the range between 15 (PDF $\geq$ 0.22) and 330 mg.L^{-1} (PDF < 0.70) (Fig. 3.6). The potential severe depletion, i.e. PDF > 0.70, is reached at a lower detrimental C_{SS} for macroinvertebrates than for algae and macrophytes. For instance, for macroinvertebrates, a PDF of 0.80 was found when C_{SS} reaches 518 mg.L^{-1} (Fig. 3.5), whereas for algae and macrophytes, the same PDF was found for a higher C_{SS} value (812 mg.L^{-1}, Fig. 3.6). This gives the indication that macroinvertebrates are more sensitive to high levels of SS (higher than 300 mg.L^{-1}) than algae and macrophytes as SS can clog their feeding structures, decreasing their feeding efficiency, and thus affecting growth and reproduction rates (Henley et al. 2000).

By the differentiation of the PDF functions for macroinvertebrates, the EF ranges from 5.7×10^{-5} PDF.L.mg^{-1} for the Guadiana–Pulo de Lobo section to 5.3×10^{-3} PDF.L.mg^{-1} for the Thames-Teddington section, whereas for algae and macrophytes ranges from 9.4×10^{-5} PDF.L.mg^{-1} for Guadiana–Pulo de Lobo section to 4.1×10^{-3} PDF.L.mg^{-1} for Danube–Dorbeta–TurnuSeverin.

With the exception of the Guadiana–Pulo de Lobo section, the EFs for each river section under study are higher for macroinvertebrates than for algae and macrophytes. This shows that the potential effects caused by SS in the water column are higher for macroinvertebrates than for algae and macrophytes. This can be corroborated by the physical and biological mechanisms mentioned in Section 1. For macroinvertebrates, the presence of SS can clog their feeding structures and damage their gills, appendances and digestive systems. For algae and macrophytes, the presence of SS reduce light penetration through the water column hampering their photosynthetic rates and damaging their photosynthetic structures in high water flow rates. Therefore, the damage of respiratory and digestive structures make macroinvertebrates more susceptive to lethal effects due to SS loads than algae and macrophytes. For algae and macrophytes to experience high lethal levels, a very high concentration of SS, capable of blocking entirely the light penetration through the water column, would be required. Moreover, following Eq.1, and as the FFs are several orders of magnitude lower than EFs, the CFs are higher for macroinvertebrates than for algae and macrophytes (Table 3.4).

The CFs for macroinvertebrates range between 2.8×10^{-7} PDF.m^3.day.mg^{-1} (Guadiana–Pulo de Lobo section) and 3.1×10^{-3} PDF.m^3.day.mg^{-1} (Severn–Bewdley section), whereas for algae and macrophytes, they range between 1.6×10^{-7} PDF.m^3.day.mg^{-1} (Alcala del Rio section) and 4.7×10^{-4} PDF.m^3.day.mg^{-1} (Severn–Bewdley section). Therefore, the emissions of SS to the Guadiana–Pulo de Lobo and the Guadalquivir–Alcala del Rio result in the lowest potential effects on both invertebrates, and algae and macrophytes. The highest potential effects were obtained for the Severn–Bewdley section for all organisms under analysis. Despite being the section with the highest volume of water, it presents a high persistence of SS in the water column, which with combination of increments of C_{SS}, implies stronger effects on aquatic biota.

Table 3.4. Fate factors (FFs), effect factors (EFs), and characterisation factors (CFs) for macroinvertebrates, algae and macrophytes.

Section	River	FF (day.L^{-1})	EF (PDF.L.mg^{-1})		CF (PDF.m^{3}.day.mg^{-1})	
			Macroinvertebrates	Algae and macrophytes	Macroinvertebrates	Algae and macrophytes
Drobeta-TurnuSeverin	Danube	2.1×10^{-12}	$4.7x10^{-3}$	$4.1x10^{-3}$	$5.4x10^{-6}$	$4.7x10^{-6}$
Regua	Douro	2.1×10^{-11}	$7.6x10^{-4}$	$2.3x10^{-4}$	$1.4x10^{-6}$	$4.2x10^{-7}$
Villachica	Douro	8.4×10^{-11}	$7.6x10^{-4}$	$2.2x10^{-4}$	$6.6x10^{-6}$	$2.1x10^{-6}$
Wittenberg	Elbe	1.6×10^{-11}	$3.7x10^{-3}$	$4.7x10^{-4}$	$1.8x10^{-6}$	$2.3x10^{-7}$
Mas d'Agenais	Garonne	1.90×10^{-11}	$4.5x10^{-3}$	$3.7x10^{-4}$	$2.1x10^{-5}$	$3.1x10^{-6}$
Alcala del Rio	Guadalquivir	2.7×10^{-11}	5.7×10^{-4}	2.1×10^{-4}	4.3×10^{-7}	$1.6x10^{-7}$
Pulo de Lobo	Guadiana	9.7×10^{-11}	5.7×10^{-5}	9.4×10^{-5}	2.8×10^{-7}	$4.7x10^{-7}$
Blois	Loire	3.2×10^{-11}	4.6×10^{-3}	5.9×10^{-4}	2.3×10^{-5}	$2.9x10^{-6}$
Montjean	Loire	1.4×10^{-11}	4.0×10^{-3}	5.0×10^{-4}	1.5×10^{-5}	$1.9x10^{-6}$
Gozdowice	Odra	2.2×10^{-11}	2.4×10^{-3}	3.6×10^{-4}	2.8×10^{-6}	$4.3x10^{-7}$
Piacenza	Po	1.2×10^{-11}	4.5×10^{-3}	$5.7x10^{-4}$	4.7×10^{-6}	$5.9x10^{-7}$
Pontelagoscuro	Po	7.6×10^{-12}	4.0×10^{-3}	5.0×10^{-4}	4.3×10^{-6}	$5.4x10^{-7}$
Paris	Seine	4.3×10^{-11}	2.6×10^{-3}	3.8×10^{-4}	1.5×10^{-5}	$2.2x10^{-6}$
Bewdley	Severn	1.7×10^{-10}	5.3×10^{-3}	8.2×10^{-4}	3.1×10^{-3}	$4.7x10^{-4}$
Almourol	Tagus	3.1×10^{-11}	6.5×10^{-4}	2.2×10^{-4}	1.4×10^{-6}	$4.6x10^{-7}$
Vila Velha de Rodão	Tagus	3.7×10^{-11}	1.8×10^{-3}	3.1×10^{-4}	2.4×10^{-6}	$4.2x10^{-7}$
Rome	Tiber	5.0×10^{-11}	4.0×10^{-4}	1.8×10^{-4}	2.3×10^{-6}	$1.0x10^{-6}$
Teddington	Thames	2.2×10^{-10}	5.3×10^{-3}	8.2×10^{-4}	8.3×10^{-4}	$1.3x10^{-4}$
Colwick	Trent	8.4×10^{-10}	3.2×10^{-3}	4.3×10^{-4}	1.4×10^{-4}	$1.9x10^{-5}$
Norham	Tweed	1.4×10^{-10}	4.4×10^{-3}	5.5×10^{-4}	7.2×10^{-6}	$9.1x10^{-7}$
Intschede	Weser	3.7×10^{-11}	3.1×10^{-3}	4.2×10^{-4}	1.3×10^{-5}	$1.8x10^{-6}$
Warsaw	Wisla	2.0×10^{-11}	3.4×10^{-3}	1.8×10^{-3}	1.8×10^{-6}	$2.3x10^{-7}$

Comparing SS with other chemical CF with different spatial resolution requires some caution, as site-specific CFs are dependent on site-specific conditions, including aquatic biodiversity. However, a comparison between site-specific CFs for both SS and eutrophication can be performed, in order to understand the ‘toxicity’ of SS. The average CFs calculated by Struijs et al. (2013) under the ReCiPe methodology, for European freshwater eutrophication due to phosphorous emissions from different sources are 4.0×10^{-4} PDF.m^3.day.mg^{-1} for manure, 1.1×10^{-3} PDF.m^3.day.mg^{-1} for fertilizer and 2.1×10^{-2} PDF.m^3.day.mg^{-1} for sewage treatment plants. These CFs are, in average, 2 and 3 orders of magnitude lower than the CFs estimated in this study for SS. However, some river sections present SS CFs for macroinvertebrates, algae and macrophytes in the same order of magnitude of the average CFs for freshwater eutrophication mentioned above. This shows the relevance of considering the SS damages in LCA.

A fate and effect model is proposed to calculate site-specific CFs, which address the direct potential effects of SS on the aquatic biota. This model allows a thorough assessment of local environmental damage, as the CFs take into account the specific river flow, volume of water, and concentration of SS in the water column for the different river sections.

Despite the focus on the damages of SS to macroinvertebrates, algae and macrophytes as a result of water topsoil erosion, the proposed method can be extended and applicable to other sources of SS, as identified in Section 1. However, it should be noted that some changes may be required, namely on the modelling of eroded SS transported to water systems. Moreover, application of similar methods can also be useful for assessing potential effects of SS on fish populations/communities.

4.2. Limitations of the study and further research

Some assumptions were considered in the proposed method and application phase, namely to overcome data constraints.

No data on the emission rate of SS to river sections and on the in-situ C_{SS} are available. Moreover, the absence of inventory data is largely a societal issue, since the knowledge of the SS fate, its consequent delivery to oceans (also affecting the ocean ecosystems), and the measurements of river flow rates, are limited (Koellner et al. 2013). Also, the measurement of unconsolidated sediments is extremely difficult. The establishment of a consistent high-resolution monitoring of SS concentrations at different flows during the hydrological year is being recognised as a priority (Barcelo and Petrovic 2007; Bilotta and Brazier 2008; Reis et al. 2010). In addition, the characterisation of land use and the area of influence of each river section would improve the understanding of the SS in rivers due to anthropogenic activities.

The developed fate model determines FFs by assuming that the marginal increase of the rate of SS in river section *i*, ΔLSS_i, is equal to the marginal increase of the emission rate of sediments to river section *i*, ΔE_i, as explained in Section 2.1. Furthermore, the residence time of SS in the water column can be increased due to the resuspension phenomenon. As no specific data are available to consider any increase in the residence time of SS due to the resuspension phenomenon, the residence time considered for both macroinvertebrate, algae and macrophyte communities may be underestimated.

Macroinvertebrates, algae and macrophytes, unlike fish, cannot swim away during an episode of increased SS loads in water column. However, in the presence of increased SS loads and due to increased flow rates, some macroinvertebrates can migrate downstream by drifting. Despite the residence time of SS is the same for both macroinvertebrates, algae and macrophytes, macroinvertebrates may be subjected to a lower exposure time to SS than macrophytes due to their mobility. However, due to lack of data, the duration of exposure to SS in a river section was not considered in this study. Further research on how to quantify both sediment deposition rate into river beds and exposure time would be pertinent. In addition, the FFs were determined based on measurements of average flow rate of each river section over periods ranging from 5-149 years, as mentioned in Section 3.1. Each station measures Q_i on a monthly basis, showing that flow rates vary along the year. Despite the different monitoring time series of the stations and recognising that would be desirable having long-term data for all stations, we consider that all monitoring stations characterise well the trend of the flow rate of each river section *i*, even the ones that operated during a period time of 5 years.

Different sources of uncertainty may be identified in the proposed effect model. Firstly, the PDF functions were established based on detrimental C_{SS}, instead of plotted effect concentrations derived from controlled acute or chronic experiments, using standardised endpoints such as lethal concentration (LC_{50}) values or no-observed-effect concentrations (NOECs). To overcome this data gap, field monitoring of C_{SS} and laboratory experiments can help the understanding of the potential effects of SS in aquatic biota. Secondly, the PDF functions were mainly derived from data collected for non-European organisms, inhabiting in different environmental characteristics. This fact may hampering the widespread use of the proposed method to assess the potential effects on invertebrate/macrophyte communities in European waters. However, the collected data from United Kingdom (Nuttal et al. 1973) allowed us to considerer European organisms in the established PFD functions. In addition, the main non-European data refer to regions with temperate climate, similar to Europe. Therefore, nearly all the macroinvertebrate species used to derive the PDFs are also found in European rivers. Macroinvertebrates of other continents have the same type of ecological and biological traits as in Europe (Dolédec et al. 1999; Statzner et al. 2001; Tomanová 2007). The same assumptions were applied for algae and macrophytes. However, further research is required to consider the specific characteristics of Mediterranean rivers such as the spatiotemporal rainfall and runoff variability. Mediterranean rivers have large periodic floods, transporting significant amounts of SS, which can contribute to higher potential effects on communities than the established PDF functions considered on the present study. To deeply understand the depletion of aquatic species due to SS, a monitoring network assumes a crucial role, as mentioned above.

As aforementioned, PDF distributions were based on methods used for SSDs, but there is a scarcity of data relating concentrations of SS with effects on aquatic species. This poses a challenge for environmental scientists that needs to be tackled. First, laboratory and field ecotoxicological data are needed to relate environmental relevant concentrations of SSs with relevant endpoints on several aquatic taxa, so as to capture not only acute but also chronic and delayed effects of SS; it also critical to determine the taxa that are especially sensitive to this stressor; finally it is also important to consider the different size of particles in suspension and the interactions with chemicals, since SS can be a source of contaminants for the aquatic environment and at the same time a factor that alters their availability to biota.

5. Conclusions

This study provides a first fate and effect model to calculate endpoint CFs, addressing the direct effects of SS in the potential disappearance of aquatic species, depending on different rivers at a global scale.

The applicability of the developed method was illustrated through the calculation of CFs to address the potential effects on macroinvertebrates, algae and macrophytes, caused by the SS stressor in different European river sections. For macroinvertebrates, the CFs range from

2.8×10^{-7} to 3.1×10^{-3} PDF.m^3.day.mg^{-1}, whereas for algae and macrophytes, they range from 1.6×10^{-7} to 4.7×10^{-4} PDF.m^3.day.mg^{-1}. Based on data relating potential effects of SS with community survival, population size, and diversity of species, macroinvertebrates taxa appear to be more sensitive to SS in the water column than algae and macrophytes. Therefore, the developed method and the calculated CFs enable a consistent assessment and comparison of the potential effects on aquatic species at different locations.

Long-term, on-site monitoring of SS levels in the water column should be performed to understand the magnitude of the effects of SS on aquatic biota and to determine the taxa that are more sensitive to the SS stressor. Thereby, it will be possible to improve the robustness of the method, the reliability of the CFs, as well as to develop CFs for a wider range of rivers.

The method and CFs presented in this study, in combination with a LCI study, enable the understanding and evaluation of the magnitude and significance of the potential environmental impacts of a land use system on aquatic biodiversity (macroinvertebrates, algae and macrophytes). A complete LCA study is the object of on-going research. Furthermore, the proposed method can also be extended and applied to other sources of SS and/or aquatic species, such as fish, allowing the calculation of further CFs.

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3.3. Life cycle impacts of suspended solids on aquatic biota from forest systems

Submitted paper:

Quinteiro P, Van de Broek M, Dias AC, Ridoutt BG, Govers G, Arroja L (2015) Life cycle impacts of suspended solids on aquatic biota from forest systems. Int J Life Cycle Assess.

Abstract

Purpose: Topsoil erosion by water is an important source of suspended solids (SS) in freshwater streams but their impacts have not yet been quantified in Life Cycle Assessment (LCA) studies. This study illustrates the applicability of a framework to conduct a spatially distributed inventory of SS delivery to freshwater streams combined with a method to derive site-specific characterisation factors for endpoint damage on aquatic ecosystem diversity. A case study on *Eucalyptus globulus* stands located in Portugal was selected as an example of a land based system. The main goal was to assess the relevance of SS delivery to freshwater streams, providing a more comprehensive assessment of the SS impact from land use systems on aquatic environments.

Methods: The WaTEM/SEDEM model, which was used to perform the SS inventory, is a raster-based empirical erosion and deposition model. This model allowed to predict the amount of SS from *E. globulus* stands under study and route this amount through the landscape towards the drainage network. Combining the spatially explicit SS inventory with the derived site-specific endpoint characterisation factors of SS delivered to two different river sections, the potential damages of SS on macroinvertebrates, algae and macrophytes were assessed. In addition, this damage was compared with the damage obtained with the commonly used ecosystem impact categories of the ReCiPe method.

Results and discussion: The relevance of the impact from SS delivery to freshwater streams is shown, providing a more comprehensive assessment of the SS impact from land use systems on aquatic environments. The SS impacts ranged from

15.5 to 1234.9 PDF.m^3.yr.ha^{-1}.$revolution^{-1}$ for macroinvertebrates, and from 5.2 to 411.9 PDF.m^3.yr.ha^{-1}.$revolution^{-1}$ for algae and macrophytes.

For some stands, SS potential impacts on macroinvertebrates have the same order of magnitude than freshwater eutrophication, freshwater ecotoxicity, terrestrial ecotoxicity and terrestrial acidification impacts. For algae and macrophytes, most of the stands present SS impacts of the same order of magnitude as terrestrial ecotoxicity, one order of magnitude higher than freshwater eutrophication, and two orders of magnitude lower than freshwater ecotoxicity and terrestrial acidification.

Conclusions: The SS model used enables ecosystem impacts to be assessed more comprehensively, showing that spatial SS impacts from land based systems on aquatic biota can be of the same order of magnitude or higher than those obtained for other ecosystem impact categories. Therefore, SS related impacts on macroinvertebrates, algae and macrophytes should be included in LCA studies of land use systems.

Keywords: *Eucalyptus globulus***,** macroinvertebrates, algae and macrophytes, Life Cycle Impact Assessment, suspended solids, WaTEM/SEDEM model

1. Introduction

Suspended solids (SS) in freshwater streams are predominantly associated with topsoil erosion by water (Jones et al. 2012a; Quinteiro et al. 2015a). SS contribute to the sustainability and biodiversity of aquatic biota due to SS–associated nutrient transport to freshwater streams. However, high concentrations of SS, particularly clay and silt size fractions, can cause major disruptions in the aquatic ecosystems, leading to sub-lethal and lethal effects on macroinvertebrates, primary producers (e.g. algae and macrophytes) and fish communities (Angermeier et al. 2004; Bilotta and Brazier 2008; Collins et al. 2011; Jones et al. 2012b; Quinteiro et al. 2015a). The presence of high concentrations of SS affects macroinvertebrates by clogging feeding structures and damaging gills and digestive structures (Bilotta and Brazier 2008). Also algae and macrophytes can be affected by high concentrations of SS, which reduce the required light penetration through the water column for photosynthesis purposes and also cause damage by abrading the photosynthetic parts of algae (Allan and Castillo 2007; Luce et al. 2010; Parkhill and Gulliver 2002). Fish communities are affected due to the abrasion and the clogging of fish gills (Kefford et al. 2010; Richardson and Jowett 2002). As a result of human-induced climate change, the frequency and intensity of seasonal heavy precipitation events have been increasing in a broad range of worldwide areas (IPCC 2007; Lima et al. 2013), which in combination with high soil erodibility and steep slopes of land use production systems (both agro and forest ecosystems) and anthropogenic activities can result to significant loss of soil (Grimm et al. 2002; FAO 2013; Pimentel et al. 1995).

Life Cycle Assessment (LCA) is a methodology that assesses the environmental impacts of products and organisations (ISO 2006) and is being used to support decision-making, ecolabelling schemes and environmental product declarations (JRC-IES 2012), among other applications. A more comprehensive impact assessment can be achieved by including additional types of environmental impacts beyond the commonly used impact categories in the LCA context (e.g. climate change, eutrophication and acidification). Several efforts have been undertaken to consider the damage of land use and land use changes on terrestrial biodiversity (Geyer et al. 2010a,b; Koellner and Scholz 2007; Koellner et al. 2013, 2012; Michelsen 2008; Schmidt 2008) and ecosystem services (Brandão and Milà i Canals 2013; Beck et al. 2011; Milà i Canals et al. 2012, 2007; Núñez et al. 2012; Reinhard and Zah 2009; Saad et al. 2013, 2011) (e.g. biomass production and freshwater filtration). However, less attention has been paid to potential impacts of SS from topsoil erosion on aquatic biota, mainly because of the complexity of establishing a spatial inventory of eroded SS from upland sources. Recently, Quinteiro et al. (2015a, 2014) developed a framework to construct inventories of both soil erosion and SS delivery to aquatic systems using a spatially distributed soil and SS delivery model (WaTEM/SEDEM) (Van Oost et al. 2000; Van Rompaey et al. 2001; Verstraeten et al. 2002) and proposed a method to derive regional characterisation factors for endpoint damage on aquatic ecosystem diversity (macroinvertebrates, algae and macrophytes).

This study illustrates the applicability of the framework and method proposed by Quinteiro et al. (2015a, 2014) by performing a case study on a land use system of *Eucalyptus globulus* stands, which is often located on steep slopes, being susceptible to soil erosion. A spatially explicit inventory of sources of SS and their potential impact on aquatic biota is of paramount importance, given the spatial heterogeneity of soil erodibility (Panagos et al. 2014), topography and rainfall erosivity (Diodato and Bellocchi 2010) at the catchment scale.

The relevance of SS delivery to freshwater streams was assessed, providing a more comprehensive assessment of the SS impact from land use systems on aquatic environments. Also, to understand the contribution of SS to the overall environmental impacts, the damage resulting from additional SS input to the water column was compared with the damage obtained from the commonly used ecosystem impact categories of the ReCiPe method.

2. Methods

2.1. Scope

This study considers four *E. globulus* stands located in the Central interior region of Portugal. Table 3.5 presents the site-specific soil characteristics on which the SS inventory is strongly dependent and area of the four stands. The *E. globulus* was chosen for assessment because it is one of the most abundant forest species in Portugal, covering 26 % (812,000 ha) of the total forest area (ICNF 2013). Almost one third of the Portuguese territory, including the Central interior region, is at high risk of erosion by water (Grimm et al. 2002). The Central interior region, particularly the lower-middle watershed of *Tagus* river (Electronic supplementary material, Fig. 3.S.1), has a very weakly developed mineral soil layer in unconsolidated material classified as Regosols (European Commision 2005), which, in combination with outbreaks of rain after frequent dry periods, make most of the sloping areas within this region prone to erosion (Grimm et al. 2002; Panagos et al. 2014). *E. globulus* stands are mainly used for pulp and paper production and are managed as a coppiced stand in short rotations of typically 12 years each, during three successive coppice rotations over one revolution (from site preparation to final cutting – 36 years).

The functional unit (FU) was defined as 1 ha of *E. globulus* managed forest over one revolution.

Table 3.5. SS produced and delivered to the *Tagus* river sections during one revolution of *E. globulus* and site-specific soil characteristics (K (soil erodibility) and LS (slope length) parameters) on which the SS inventory is strongly dependent.

	Tagus river section	Area (ha)	SS delivery to *Tagus* river ($t.ha^{-1}.revolution^{-1}$)	Average K parameter ($t.ha.h.ha^{-1}.MJ^{-1}.mm^{-1}$)	Average LS parameter (dimensionless)
Stand 1	Almourol	8.2	4.1	0.029	1.34
Stand 2		29	329.0	0.037	10.27
Stand 3	Vila Velha de Rodão	83	147.6	0.038	16.27
Stand 4		103	131.6	0.040	7.02

2.2. Suspended solids inventory

The Life Cycle Inventory of SS originating from soil erosion in *E. globulus* stands was performed by applying the WaTEM/SEDEM model (Van Oost et al. 2000; Van Rompaey et al. 2001; Verstraeten et al. 2002). This model predicts the spatial distribution of long-term mean annual soil loss by sheet, rill and ephemeral gully erosion (Desmet and Govers 1996a) and SS delivery at the catchment scale. The amount of eroded soil is calculated using the empirical RUSLE (Renard et al. 1997), and routed through the landscape using the flux-decomposition algorithm (Desmet and Govers 1996b) and soil particles are deposited in cells where the transport capacity is exceeded (Desmet and Govers 1995; Van Oost et al. 2000; Van Rompaey et al. 2001).

In a first step, eroded soil transport in the WaTEM/SEDEM model has to be constrained by selecting adequate transport capacity coefficients (ktc). The ktc values represent the slope length needed to produce the amount of soil equal to the transport capacity from a bare surface with the local slope gradient (Van Rompaey et al. 2001). Due to a lack of an extensive dataset of measurements, the most adequate ktc values were selected based on existing literature. In a second step, the sensitivity of the model results to the ktc values was evaluated. Subsequently, model performance using the optimal set input parameters was validated against SS measurements in the main river channel. In a last step, the model was used to calculate the SS delivery from the four *E. globulus* stands towards the main river channel.

2.2.1. WaTEM/SEDEM model calibration

WaTEM/SEDEM (Van Oost et al. 2000; Van Rompaey et al. 2001; Verstraeten et al. 2002) is a raster-based empirical erosion and deposition model. For every grid cell, the amount of eroded soil is calculated based on the RUSLE (Renard et al. 1997) and is subsequently routed through the landscape. When the total amount of soil in a grid cell exceeds the runoff transport capacity, the amount of material leaving the grid cell equals the runoff transport capacity, the remaining part of the material is deposited in that grid cell. The runoff transport capacity (Tc) (Van Rompaey et al. 2001) is proportional to the potential gully erosion as shown in Eq.1.

$$Tc = ktc \times R \times K \times (LS_{2D} - 4.12 \times Sg^{0.8}) \qquad \text{(Eq.1)}$$

Where ktc is the runoff transport capacity coefficient, Sg the local slope and R, K and LS_{2D} are RUSLE parameters. The ktc value is a scaling factor to determine the runoff transport capacity, and this capacity depends on multiple factors such as land use and grid cell size. WaTEM/SEDEM model allows the definition of a ktc value for non-erodible land surfaces (forests and grasslands) as well as one value for erodible land surfaces (crop land) (Van Rompaey et al. 2001; Verstraeten et al. 2002). Long-term measurements of SS load in the catchment under study are scarce, thereby limiting the ability to perform a reliable and accurate calibration of the ktc parameters. To overcome this constraint, a pre-established calibration of the ktc performed by Verstraeten (2006) was used. In this study, ktc values were calibrated for seven large river catchments in Belgium based on a SRTM-DEM, resampled to a resolution of 100 m. This author found optimal ktc values of 8 m for non-erodible land surfaces and 27 m for erodible land surfaces. Although this calibration was performed in another region than the catchment under study, the same input data, i.e. a SRTM-DEM, resampled to a resolution of 100 m and a parcel map derived from CORINE Project land cover (EEA 2012) were used. The use of a pre-established calibration of ktc values results in an uncertainty that cannot be avoided due to the absence of long-term SS data. Therefore, the uncertainty of the set of ktc values on the model results was assessed by performing a sensitivity analysis. The ktc values for non-erodible surfaces were varied from 4 to 12 in increments of 1. The respective ktc values for erodible surfaces were calculated by multiplying each ktc value for non-erodible land surfaces by a factor of 3.38. This factor is the ratio of the optimal ktc values for erodible to non-erodible surfaces as found by Verstraeten (2006). This means that regardless of grid cell size, a grid cell under forest can transport 3.38 times less SS than a grid cell under agriculture Verstraeten (2006). Further details about the model, input data and parameters, land cover and data sources can be found in the Electronic supplementary material (Section S2).

2.2.2. WaTEM/SEDEM model validation

The observed SS load (Horowitz 2003) was calculated based on measurements of in-situ concentrations of SS (C_{SS}) and discharge (Q) from the Almourol gauge station (SNIRH 2015), located at the outlet of the watershed of the *Tagus* river. Monthly measurement data of C_{SS} were available for a period of five years, from January 1985 until December 1989. Total SS load was calculated as shown in Eq. 2.

$$\text{observed SS load}\ (\text{t.month}^{-1}) = C_{SS}\ (\text{kg.m}^{-3}) \times Q\ (\text{m}^3.\text{month}^{-1}) \times 0.001\ (\text{t.kg}^{-1}) \qquad \text{(Eq.2)}$$

The observed annual SS load was calculated by summing the observed monthly SS loads for every year. The observed annual average of SS load for the entire period 1985-1989 (585,278 $t.yr^{-1}$) was 20 % lower than the predicted annual SS load that leaves the entire watershed through the river network (727,692 $t.yr^{-1}$, as shown in Fig. 3.7). The overprediction of the annual SS load can be explained by the model simplifications related to land cover (further details in Section 3.6). Another reason for the overprediction of SS lies in the fact that WaTEM/SEDEM does not model internal river dynamics and assumes that all the SS that reach the river channel are transported to the catchment outlet, i.e. SS delivery to the river corresponds to the SS load. SS that are deposited on the river bed are thus not taken into account by the model. Although the assumed model simplifications, the results show that the model represents the general catchment dynamics and is capable of calculating SS loads in the order of magnitude of the measurements.

2.2.3. Suspended solids delivery from *E. globulus* stands

After the optimal input parameters for the model were selected, the SS delivery from the four *E. globulus* stands towards the *Tagus* river was modelled. This was done using an adapted version of the WaTEM/SEDEM model that was able to track SS from the upland source area to the main river channel (Notebaart et al. 2005). Spatial data layers delimiting the boundaries of the *E. globulus* stands were used to define the source areas of SS.

The SS delivered to the *Tagus* river during one revolution of *E. globulus* depends on site-specific soil characteristics, rainfall erosivity and on what happens to both the up- and downstream pathways of the stands, which is modelled using the flux-decomposition algorithm (Desmet and Govers 1996b) of WaTEM/SEDEM. The average crop management parameter (Pimenta 1998) (C parameter in RUSLE; further details in Electronic supplementary material, Section S2) is constant for all *E. globulus* stand with a value of 0.2, while the average value of soil erodibility (Panagos et al. 2014) (K parameter in RUSLE equation; further details in Electronic supplementary material, Section S2) ranged from 0.028 $t.ha.h.ha^{-1}.MJ^{-1}.mm^{-1}$ in stand 1 to 0.048 $t.ha.h.ha^{-1}.MJ^{-1}.mm^{-1}$ in stand 4, and slope-length (LS) ranged from 1.34 (dimensionless) in stand 1 to 16.27 (dimensionless) in stand 3.

Tillage erosion, i.e. the net downslope movement of soil by tillage operations increasing the exposure of less productive sub-soils, is particularly important in hilly areas used for intensive agriculture and forestry (Govers et al. 1994; Lindstrom et al. 1992; Van Oost et al. 2000). *E. globulus* stands are prone to tillage erosion, mainly due to site preparation and tillage management activities in young stands (Croke 2004; Kosmas et al. 2012). Management activities lead to bare soil surfaces during the first two or three years of the first rotation, making soils under *E. globulus* particularly sensitive to soil erosion during this period. Unlike water erosion, soil displacement by tillage in the WaTEM/SEDEM model will only occur within a field, i.e. SS are not transported towards the drainage network (Van Oost et al. 2000). Therefore, the contribution of tillage erosion to the total SS that reach the drainage network was not taken into account.

During the first part of each rotation cycle the canopy is not completely closed until approximately the age of five years (Quinteiro et al. 2015b). During this period the soil is less protected from raindrop impact and therefore more prone to erosion by water. After canopy closure the soil is more protected from water erosion. This situation is taken into account in the average crop management parameter of *E. globulus* of the RUSLE used by the WaTEM/SEDEM model. The average C value of 0.2 means that topsoil erosion will be reduced to 20 % compared to the amount that would have been eroded under continuous fallow conditions.

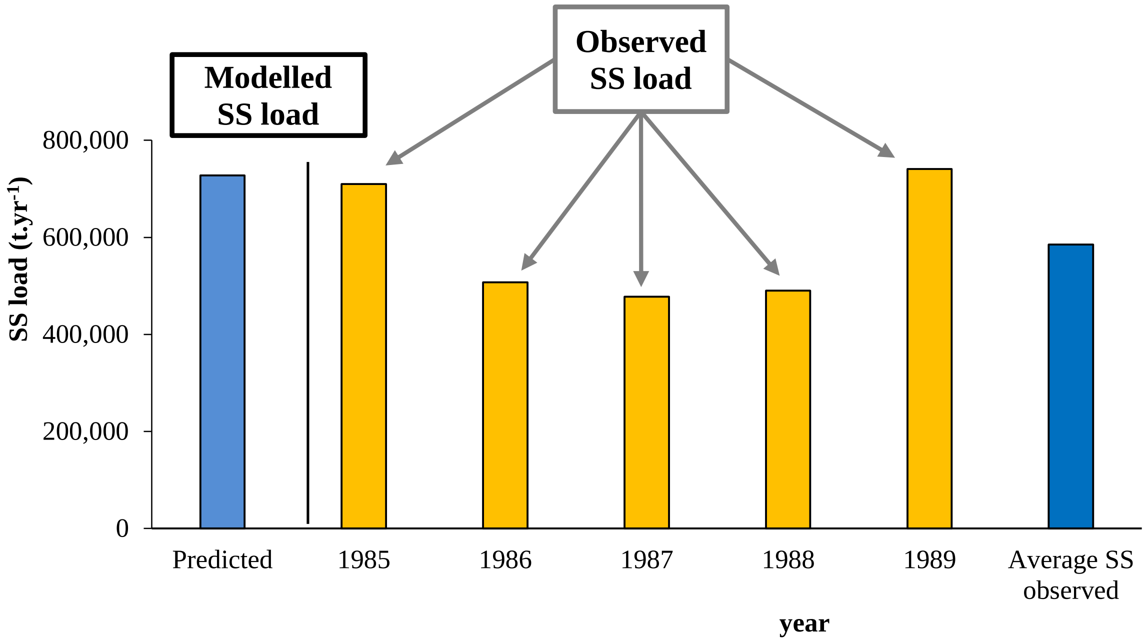

Fig. 3.7. Results of predicted annual SS load, and observed SS load originating from the watershed of the Tagus river during the 5-year period of on-field measuring (1985-1989).

2.3. Forest management inventory

Inventory data for the ecosystem impact categories of the ReCiPe method (Goedkoop et al. 2013) associated with the forest management operations were collected. These are representative of a high intensity management model, characterised by adoption of the best management practices recommended for *E. globulus* stands, as described in Dias and Arroja (2012) and Dias et al. (2007). All forest management operations undertaken during the stages of site preparation, stand establishment, stand tending, felling and infrastructure establishment were taken into account.

The amount of carbon dioxide (CO_2) assimilated during *E. globulus* growth was not taken into account. It was assumed that this amount is equal to the amount of CO_2 that will be released back to the atmosphere due to wood oxidation along the downstream life cycle stages of wood (Dias and Arroja 2012), i.e. biogenic carbon that is emitted during the use and end-of-life stages of paper or other forest-derived products. The production of fuels, lubricants and fertilisers required for management operations were considered (Dias and Arroja 2012; Dias et al. 2007). Transport of workers, machinery, and materials (fuels, lubricants and fertilisers), as well as capital goods, were excluded as they did not contribute significantly to the overall result and the associated distances and means of transportation greatly vary within the country.

For the operations carried out during site preparation (stand tending and infrastructure establishment), the inputs of fuels, lubricants and fertilisers were directly obtained per unit of land area (Dias et al. 2007). For felling, the inputs of fuels and lubricants were first obtained per unit of wood volume under bark and then were expressed per unit of land area by considering the average wood productivity (average annual increment) of each stand (8.5 $m^3.ha^{-1}$ for stands 1 and 2, and 7.4 $m^3.ha^{-1}$ for stands 2 and 3) and the density of wood (550 kg dry matter.m^{-3}). A detailed list of the inputs related to forest management operations can be found in the Electronic supplementary material (Section S3).

2.4. Impact assessment of suspended solids and sensitivity analysis of model input parameters

The quantitative impact assessment of SS on the potential disappearance of macroinvertebrates, algae and macrophytes in the *Tagus* river was performed using the site-specific characterisation factors (CFs) developed by (Quinteiro et al. 2015a). Based on the drainage network (Electronic supplementary material, Fig. 3.S.1), it can be seen that stands 1 and 2 deliver SS to the Almourol river section, while stands 3 and 4 deliver SS to the Vila Velha de Rodão river section. The Almourol river section has a slightly higher average water volume and average flow than the Vila Velha de Rodão section (Quinteiro et al. 2015a).

The CFs (expressed in potentially disappeared fraction (PDF).m^3.day.mg_{SS}^{-1}) for both these river sections were derived based on a spatially explicit environmental fate and effect model (Quinteiro et al. 2015a). For every river section this model takes into account the specific river flow, residence time of SS, volume of water and concentration of SS in the water column and the detrimental concentrations of SS affecting the survival of macroinvertebrates, algae and macrophytes organisms.

For macroinvertebrates, the CF was defined as the change in PDF of aquatic organisms due to a change in the SS load in water column (Quinteiro et al. 2015a). The CF is higher for the *Tagus*–Vila Velha de Rodão river section (2.4×10^{-6} PDF.m^3.day.mg_{SS}^{-1}) than for the

Tagus–Almourol river section (1.4×10^{-6} PDF.m^3.day.mg$_{SS}^{-1}$). In contrast, for algae and macrophytes, the CF for the *Tagus*–Almourol river section (4.6×10^{-7} PDF.m^3.day.mg$_{SS}^{-1}$) is slightly higher than the one developed for the *Tagus*–Vila Velha de Rodão (4.2×10^{-7} PDF.m^3.day.mg$_{SS}^{-1}$).

The spatial SS inventory contains some assumptions and simplifications (further details in the Electronic supplementary material, Section S4) and the calculation of eroded SS strongly depends on the parameter values of the RUSLE. Therefore, a sensitivity analysis was performed in which the values for the crop factor (C), soil erodibility (K) and rainfall erosivity (R) parameters were varied in a range of ± 10 %.

To demonstrate how the SS characterisation model developed by (Quinteiro et al. 2014, 2015a) can help to improve the environmental assessment in forestry, the results were compared with the commonly used endpoint ecosystem impact categories from the ReCiPe method (Goedkoop et al. 2013), namely freshwater eutrophication, freshwater acidification, terrestrial ecotoxicity, terrestrial acidification and climate change. Damage to macroinvertebrates, algae and macrophytes in the model developed by Quinteiro et al. (2015a) is expressed in different units (PDF.m^3.day) than those used to express ecosystem damage estimated in the ReCiPe method (species.yr). Therefore, to ensure comparability, SS CFs in PDF.m^3.day.m$_{SS}^{-1}$ were converted to units of species.yr.mg$_{SS}^{-1}$, considering the total macroinvertebrates, algae and macrophytes species density. Total species density was determined by counting the total number of least concerned, near threatened, vulnerable, endangered and critically endangered macroinvertebrates species (11,777 species) and algae and macrophytes species (20,837 species) in freshwater systems listed by the International Union for Conservation of Nature (IUCN 2015) and the total volume of freshwater present in the earth rivers, streams and lakes (126,700 km^3) (Goedkoop et al. 2013). However, the use of a global average value of macroinvertebrates, algae and macrophytes for addressing the total species density in the *Tagus* river sections creates a source of uncertainty for site-specific SS impact results. This issue is discussed in Section 3.6.

3. Results and discussion

3.1. Suspended solids inventory

As mentioned in the Section 2.2, the transport of eroded soil in the WaTEM/SEDEM model has to be constrained by selecting adequate ktc values. Since long-term measurements of SSC in the catchment under study are scarce and there is limited ability to develop locally calibrated ktc values, a pre-established calibration of the transport capacity coefficients by Verstraeten (2006) was used. The lack of data also justifies the use of the WaTEM/SEDEM model, which has a limited amount of necessary input data but still captures major SS dynamics in large catchments (Schindler and Hilborn 2015). This is a common challenge in LCA, in which there is an increasing trend toward the use of regionalised impact assessment models, but the availability of relevant data to define local parameters is often scarce. The use of a pre-established calibration of ktc values results in uncertainty that cannot be avoided due to absence of long-term monitoring SS data. Therefore, the uncertainty is analysed by means of a sensitivity analysis. The Electronic supplementary material (Section S2) provides the description and the results of the sensitivity analysis performed. WaTEM/SEDEM predictions of SS are indeed sensitive to the ktc values used and vary between -55 % and +23 % of the measured SS load for the parameter value range considered (further details in Electronic supplementary material, Fig. 3.S.7). Differences ranging from -7 to -55 % (considering ktc values lower than the default values) indicate that the predicted SS delivered to river are lower than the SS delivery estimated with the default ktc values. In contrast, differences ranging from +6 to +23 % (considering ktc values higher than the default values) indicate that the predicted SS are higher than the SS load estimated with the default ktc values. Lower ktc values than the default ones result in less reliable model predictions, as the long-term predicted SS load was increasingly lower than the yearly SS observed load. Higher ktc values than the default ones increase the overestimation of the SS load compared to the observed yearly average SS load observed.

Stand 4 was characterised by the highest susceptibility to soil detachment as it has the highest average K parameter (0.041 $t.ha.h.ha^{-1}.MJ^{-1}.mm^{-1}$) and was responsible for the highest total SS load delivered to the *Tagus* river, in particular to the Vila Velha de Rodão river section (13,556.2 $t.revolution^{-1}$). However, this stand also has the highest productive area (103 ha), which means that per FU, the SS are 131.6 $t.ha^{-1}.revolution^{-1}$, being lower than the SS for stands 2 and 3 (Table 3.5). SS delivery per FU of stand 4 is only 12 % lower than that determined for stand 3, which has a slightly lower average K value but a higher average LS_{2D} value than stand 4. Stand 2 has a similar average K value than stands 3 and 4, and a lower average LS_{2D} value than stand 3. Moreover, it also has a lower productive area than both these stands, which results in the highest SS load per FU (329.1 $t.ha^{-1}.revolution^{-1}$). Stand 1 has the lowest SS delivery, in particular to the Almourol river section as this stand has the lowest average K values and productive area and is located in a relatively flat area, therefore having the lowest average LS_{2D} value.

3.2. Suspended solids impact assessment

As shown in Table 3.5, the *E. globulus* stands under study deliver SS to two different sections of the *Tagus* river (Vila Vellha de Rodão and Almourol), for which spatial SS CFs are available (Quinteiro et al. 2015a). By linking the SS inventory with spatial CFs of these river sections, the potential damage on macroinvertebrates, algae and macrophytes due to the increased SS in the water column can be assessed. As can be seen in Fig. 3.8, the resulting SS impacts ranged from 15.5 to 1,234.9 $PDF.m^3.yr.ha^{-1}.revolution^{-1}$ for macroinvertebrates, and from 5.2 to 411.9 $PDF.m^3.yr.ha^{-1}.revolution^{-1}$ for algae and macrophytes. The potential impacts for macroinvertebrates are higher compared to those for algae and macrophytes because high SSC clog the feeding structures and damage the gills and digestive structures of macroinvertebrates, making these organisms more sensitive to SS than algae and macrophytes Quinteiro et al. (2015a). Although the SS CF for macroinvertebrates is higher for the *Tagus*–Vila Velha de Rodão river section, the highest impact on macroinvertebrates biodiversity was observed for the *Tagus*–Almourol river section. This is because of the higher SS production by stand 2, which produces a SS load per FU which is 40 % higher than inventoried for stands 3 and 4 (Tagus–Vila Velha de Rodão river section). This high SS delivery from stand 2 per FU in combination with the higher SS CF for algae and macrophytes in the *Tagus*–Almourol river section results in the highest damage on algae and macrophytes biodiversity. Stand 1 contributes to a significantly less to damage on aquatic species than the other stands under analysis because as it is located in a flat area, it delivers the lowest amount of SS per FU to the *Tagus* river (4.1 $t.ha^{-1}.revolution^{-1}$).

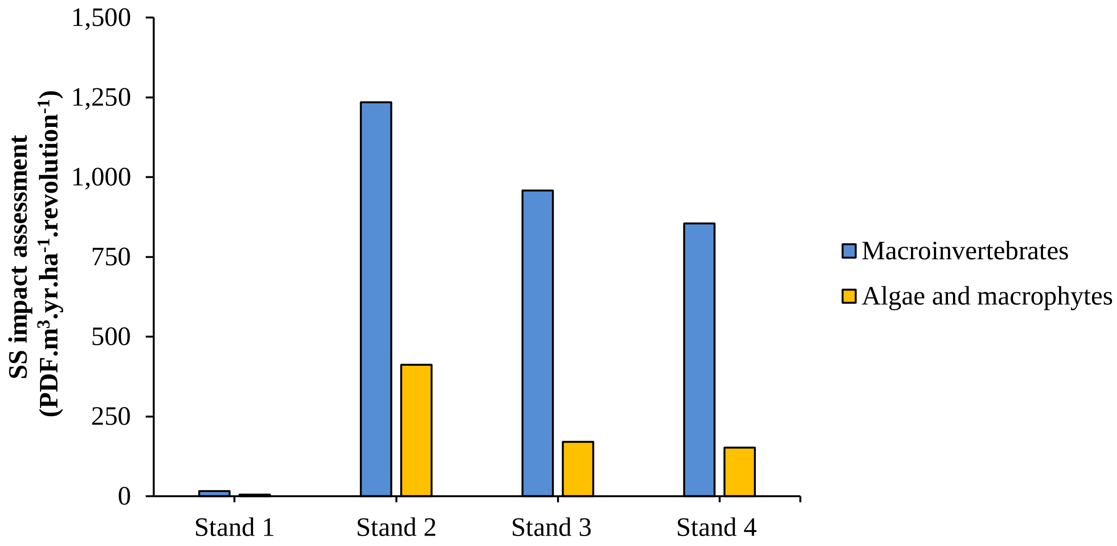

Fig. 3.8. Potential damage on aquatic biota due to the increased SS load in the water column from the *E. globulus* stands studied.

3.3. Sensitivity analysis

A sensitivity analysis was carried out to evaluate the influence of input data of the RUSLE on the potential impacts on macroinvertebrates, algae and macrophytes. Changes of ± 10 % in the default input C, K, and R parameters were considered. Significant changes in the SS potential impacts were obtained when the C parameter was changed by -10 %. Potential impacts increased by 10 % for stand 1, and decreased between 24 (stand 2) and 50 % (stand 4) compared to the potential impact of the reference case (Table 3.6). It should be kept in mind that erosion from the *E. globulus* stands depends on erosion dynamics upslope of these areas, while SS delivery to the river channel depends on the characteristics of the area between the stand and the river. The increase of SS potential impacts can be explained by the fact that when a lower C parameter is chosen for the entire watershed, the runoff that reaches a stand is not saturated with SS, thus more soil can be eroded from that stand, resulting in a higher SS load delivered to river networking. However, the SS potential impacts can also decrease when a lower C parameter is chosen because less erosion can occur. Besides, as the ktc parameter is proportional to the C parameter, runoff can transport smaller amounts of SS from the *E. globulus* stands.

When a higher C parameter is chosen, it is possible that all runoff that reaches the stand is SS-saturated (the transport capacity of SS that can pass through one pixel is reached). When the C parameter is increased by 10 %, minor or even no increase in SS potential impacts for aquatic biota were determined, which indicates that runoff that reaches *E. globulus* stands is SS–saturated. In contrast to the C parameter, changes in the K and R parameters have a significantly lower influence on the SS potential impacts on macroinvertebrates, algae and macrophytes (up to ±11 %) for all stands under analysis (Table 3.6). This indicates that the uncertainties of the K and R factors that were used for the simulations are limited. The C parameter on the other hand did affect model results significantly. Because a C parameter of 0.2 for *E. globulus* was determined in Pimenta (1998), it was chosen to perform the simulation using this value. It should however be kept in mind that model uncertainty arises from the different parameters used.

Table 3.6. Sensitivity analysis results, calculated for the impact assessment phase. The results express the changes on the SS potential impacts on macroinvertebrates, algae and macrophytes, resulting from changing the default input parameters used in the RUSLE.

Stands	C parameter		K parameter		R parameter		C parameter		K parameter		R parameter	
	-10%	+10%	-10%	+10%	-10%	+10%	-10%	+10%	-10%	+10%	-10%	+10%
	SS potential impacts on macroinvertebrates ($PDF.m^3.yr.ha^{-1}.revolution^{-1}$)						SS potential impacts on algae and macrophytes ($PDF.m^3.yr.ha^{-1}.revolution^{-1}$)					
Stand 1	17.1	5.7	13.8	16.7	14.0	17.1	5.7	5.2	4.6	5.6	4.7	5.7
Stand 2	934.8	311.8	1,107.5	1,365.8	1,111.9	1,359.0	311.8	414.3	369.5	420.7	370.9	453.3
Stand 3	637.9	113.6	862.7	1062.2	863.0	1054.7	113.6	169.6	153.6	189.1	153.7	187.8
Stand 4	431.8	76.9	769.5	939.4	769.5	940.5	76.9	152.6	137.0	167.3	137.0	167.5

3.4. Impact assessment of forest management operations in *E. globulus* stands

The impact assessment of *E. globulus* forest management operations was performed using the endpoint characterisation factors recommended by ReCiPe method (Goedkoop et al. 2013). Table 3.7 shows the impact assessment results due to the occupation of 1 ha by *E. globulus* during one revolution. Climate change impact accounted for the majority of the total estimated ecosystem impacts, with 3.0×10^{-5} $species.yr.ha^{-1}.revolution^{-1}$ (98 % of the total ecosystem impacts). The largest contribution to climate change comes from CO_2 (65 %), which is emitted mainly during *E. globulus* forest operations due to fossil fuel combustion (Electronic supplementary material, Table 3.S.1). Dinitrogen monoxide (N_2O) emissions resulting from fertiliser application were responsible for 35 % of predicted climate change impacts. In contrast, freshwater eutrophication was responsible for the lowest impact, representing 0.01 % of the total *E. globulus* forest management environmental impacts, mainly due to phosphorous (P) and phosphate (PO_4^{3-}) emissions associated with application of P-containing fertilisers during stand tending (Electronic supplementary material, Table

S2). The emission of some pollutants to air and soil, such as cadmium (soil and air emissions), chromium and polycyclic aromatic hydrocarbons (air emissions) during tillage operations and carbofuran (soil emissions) and bromine (air, soil and water emissions) emitted during fertilisers production are the main contributors to freshwater and terrestrial ecotoxicity impacts (Table 3.7). Terrestrial acidification impacts come mainly from stand tending operations, in particular ammonia (NH_3) emission due to fertilisation (Electronic supplementary material, Table 3.S.1).

3.5. Comparison of impacts from suspended solids and forest management

SS potential impacts and forest management impacts resulting from the occupation of 1 ha by *E. globulus* during one revolution are presented in Table 3.7. The results show that for macroinvertebrates, the SS impacts range from 1.4×10^{-9} species.yr.ha^{-1}.revolution^{-1} (stand 1) to 1.1×10^{-7} species.yr.ha^{-1}.revolution^{-1} (stand 2) while for algae and macrophytes the SS impacts range from 8.5×10^{-10} species.yr.ha^{-1}.revolution^{-1} (stand 1) to 6.8×10^{-8} species.yr.ha^{-1}.revolution^{-1} (stand 2). For some stands, SS potential impacts on macroinvertebrates have the same order of magnitude than freshwater eutrophication, freshwater ecotoxicity, terrestrial ecotoxicity and terrestrial acidification impacts (Table 3.7). For algae and macrophytes, all stands with exception of stand 1 present SS impacts of the same order of magnitude as terrestrial ecotoxicity, one order of magnitude higher than freshwater eutrophication, and two orders of magnitude lower than freshwater ecotoxicity and terrestrial acidification. For stand 1, the SS potential impacts on algae and macrophytes are at least one order of magnitude lower than all other impacts resulting from *E. globulus* production. For all stands under study, ecosystem climate change presents a significantly higher impact compared to SS impacts on macroinvertebrates, algae and macrophytes.

In some stands, SS potential impacts have a higher relevance than other commonly established impact categories, with the exception of ecosystem climate change. These results highlight that the newly developed framework and method (Quinteiro et al. 2015a, 2014) to determine the spatial SS potential impacts on aquatic biota is a significant contribution to a more comprehensive local environmental assessment of forest and agricultural systems. Extensive deforestation for cropland and pasture and intensive agriculture have dramatically accelerated soil erosion, leading to a gradually thinner soil and a loss of productivity (Montgomery 2007; Pimentel and Burgess 2013). Land areas covered by forest are more resistant to water erosion than croplands because tree canopies have the effect of decreasing the effective rainfall erosive force (FAO 2013). However, it should be noted that for the five endpoint ecosystem impact categories, the non-spatially explicit CFs were derived from the ReCiPe method, which could skew the comparison between theses impact categories and the SS impacts, which uses spatially explicit CFs. In order to improve spatially explicit environmental impact assessments, Struijs et al. (2010) determined spatially differentiated CFs for P emission due to fertiliser applications. These authors provide a CF for the *Tagus* river equivalent to 6.5×10^{-15} species.yr.mg_{SS}^{-1} (following the recommendation of the ReCiPe method to convert units of PDF.m^3.day.m_{SS}^{-1} to species.yr.mg_{SS}^{-1}, considering the freshwater species density of 7.9×10^{-10} species.yr.kg^{-1}), which is lower than the generic CF for P emission due to fertiliser application available in ReCiPe method (2.4×10^{-12} species.yr.mg_{SS}^{-1}). Consequently, a CF for the *Tagus* river of 6.5×10^{-15} species.yr.mg_{SS}^{-1} leads to lower freshwater eutrophication impacts (7.1×10^{-12} species.yr.ha^{-1}.$revolution^{-1}$) compared to the results obtained with the generic CF available in ReCiPe method (2.6×10^{-9} species.yr.ha^{-1}.$revolution^{-1}$). This indicates that ecosystem freshwater species are less vulnerable to P emissions in the *Tagus* river compared to the average results for European continental waters.

Table 3.7. Impact assessment results associated with occupation of 1 ha by *E. globulus* during one revolution (36 years).

Stands	species.yr.ha^{-1}.$revolution^{-1}$						
	SS potential impacts on macroinvertebrates	SS potential impacts on algae and macrophytes	Freshwater eutrophication	Freshwater ecotoxicity	Terrestrial ecotoxicity	Terrestrial acidification	Ecosystem climate change
Stand 1	1.4×10^{-9}	8.5×10^{-10}	3.6×10^{-9}	1.3×10^{-7}	2.3×10^{-8}	5.4×10^{-7}	3.0×10^{-5}
Stand 2	1.1×10^{-7}	6.8×10^{-8}					
Stand 3	8.9×10^{-8}	2.8×10^{-8}					
Stand 4	7.9×10^{-8}	2.5×10^{-8}					

3.6. Implications and recommendations

There are some aspects that should be a priority in further research: (1) improvement of SS modelling, in particular the calibration of the transport capacity coefficient and the identification of SS-trapping sites and forest roads, (2) CFs at an adequate spatial resolution related to life cycle inventories, (3) determination of site-specific freshwater species richness, and (4) application of framework and method developed for other land uses in other regions with different soil and climatic conditions.

Concerning the first item, to conduct a spatial SS inventory, the WaTEM/SEDEM model requires a calibration of the ktc parameters in order to obtain an optimal relation between model parameters and observed erosion dynamics, as discussed in Section 2.2. Because the transport capacity depends on multiple factors such as climate and landscape structure, a new calibration should be performed when applying the model to a new environment. For this purpose, long-term on-site monitoring of SS load at the catchments outlet is necessary. As this data was not present for the watershed under study, predefined transport capacities that were obtained for the same spatial resolution in another region were used. However, the ktc sensitivity analysis (Electronic supplementary material, S2) showed that the pre-established calibrated ktc values give a better approximation of the SS load on a yearly basis than other ktc values. In addition, in the Section 2.2.2, it is shown that despite the use of pre-established calibrated ktc values and model simplifications, this model is capable of calculating SS loads in the order of magnitude of the measurements.

The general overprediction of the SS delivered to the *Tagus* river with the WaTEM/SEDEM model may result from the fact that SS-trapping sites, i.e. vegetation barriers (e.g. hedges and grass strips), ponds, reservoirs and dams (in particular the Cedillo, Fratel and Belver dams), were not properly taken into account. For example, it has been observed that a fraction of the overland flow can infiltrate near the field boundary and SS are likely to be deposited here, due to differences in vegetation and soil surface conditions (Meyer et al. 1995; Slattery and Burt 1997; Takken et al. 1999). Although this mechanism is taken into account in the WaTEM/SEDEM model, the input parameters for soil trapping at field borders were determined for the Belgian loam belt and their applicability was not validated for our study area. Although ponds, reservoirs and dams also act as a SS trap, these infrastructures were not considered due to data constraints. In addition, although forest roads contribute disproportionally to runoff and SS production in forested areas (Chappell et al.

1999; Croke and Hairsine 2006: La Marche and Lettenmaier 2001), these were not considered due to data constraints, which can introduce uncertainty on SS inventory. Further research is needed to implement the abovementioned constraints in an operational model structure for the area under study.

Regarding the second item, the spatial potential environmental impacts of SS on aquatic biota depend on (1) the quantity of SS production, i.e. the amount of soil eroded from the land-use production systems, (2) the properties of the these SS, and on the (3) the characteristics of land-use production systems (e.g. soil, rainfall and geomorphological characteristics). The assessment of these impacts and others, affecting local, regional and continental scales, adequate spatial information is required in order to accurately establish accordance between the inventory and the impact assessment phase (Reap et al. 2008). The quality of LCA studies can be compromised if the LCA practitioner has difficulties to establish a connection between CFs with high spatial resolution and the related spatial inventory or vice versa. It is also not easy to obtain information about background processes (e.g. fertiliser, diesel, petrol and lubricating oil production processes in this case study), hampering the development of endpoint site-specific CFs for these processes. Indeed, finding an optimal spatial resolution to construct life cycle inventories remains a major scientific challenge in LCA (Huijbregts 2013).

Regarding the third item (species richness), it is important to keep in mind that determining this factor for specific river sections is complex. Adequate information on the number of macroinvertebrates, algae and macrophytes species in the river sections under study was not available. The use of a global average value of macroinvertebrates, algae and macrophytes density for site-specific SS impacts (in units of species.yr.ha^{-1}.revolution^{-1}) creates a source of uncertainty, since local species richness can vary depending on the characteristics of the river sections under study. Further improvements in the characterisation of freshwater species richness should be considered in order to increase the robustness of the comparison between impact categories.

Finally, in order to further improve the framework and method developed, the SS potential impacts should also be assessed for other forest and agriculture systems, thereby enhancing the knowledge on the impact of SS on aquatic biota in comparison with other impact categories.

4. Conclusions

The inclusion of site-specific potential impact of SS on macroinvertebrate, algae and macrophyte biodiversity in LCA methodology helps to fill a relevant gap related to the impact of land use on freshwater species. This study shows that SS impacts can have a similar or even higher contribution to the total environmental impact than the commonly established endpoint impact categories of the ReCiPe method (such as freshwater eutrophication, freshwater ecotoxicity, terrestrial ecotoxicity and terrestrial acidification).

Electronic supplementary material

The online version of this article contains supplementary material, which is available to authorised users. Additional information is provided on the location of the *E. globulus* stands studied, input parameters required for the WaTEM/SEDEM model, model validation, sensitivity analysis of the ktc parameters, forest management operations, SS impact assessment, and impact assessment of forest management operations.

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Electronic supplementary material

S1. Watershed location

The lower-middle watershed of the *Tagus* river (Fig. 3.S.1) has a Mediterranean climate with a warm and dry summer, according to the Köppen–Geiger climate classification system. The average annual rainfall (SNIRH 2010) ranges from 701 to 800 mm, being concentrated in autumn and winter. This catchment is located in the transition zone from a semi-arid (to the south) to sub-humid (to the north) Mediterranean climate, with an average annual temperature (SNIRH 2010) ranging from 16 to 17.5 ºC, with the exception of stand 1, in which the average annual temperature is below 16 ºC. The catchment is characterised by a heterogonous relief, vegetation and soil, having an elevation between 19 and 1458 m above sea level, in which the stands have average elevations below 300 m.

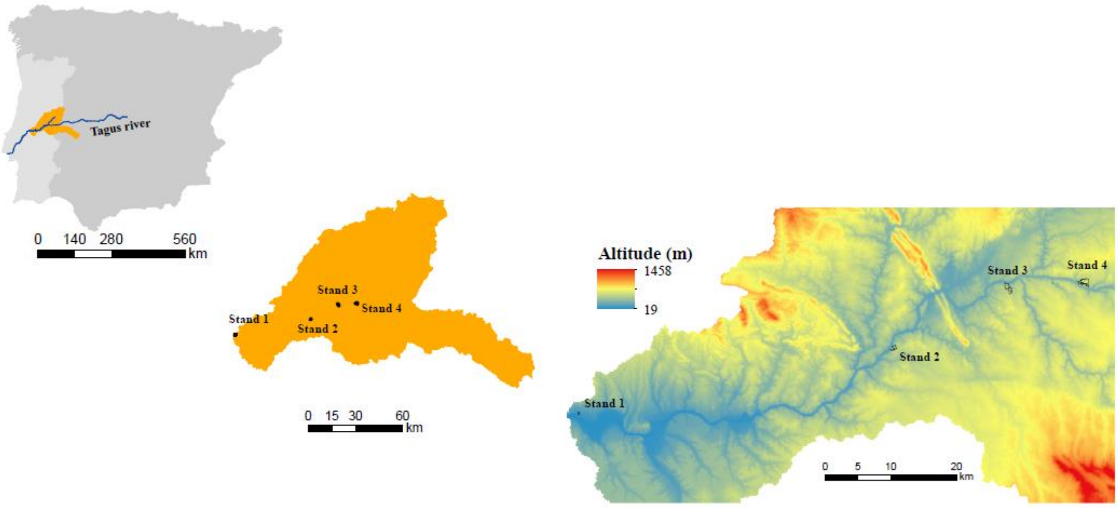

Fig. 3.S.1. Location of the lower-middle watershed of the Tagus river and the four *E. globulus* stands studied.

S2. WaTEM/SEDEM inputs

In the WaTEM/SEDEM model mean annual water erosion is calculated on a grid cell basis using Eq.S1 (Renard et al. 1997) (Revised Universal Soil Loss Equation – RUSLE). The eroded soil particles are routed through the landscape using the 2D flux decomposition algorithm (Desmet and Govers 1996). When the sum of suspended solids (SS) and incoming SS exceeds the local transport capacity the amount of SS leaving the grid cell equals the transport capacity (Eq.S2) (Desmet and Govers 1995; Van Oost et al. 2000; Van Rompaey et al. 2001)).

$$A = R \times K \times LS_{2D} \times C \times P \qquad \text{(Eq.S1)}$$

$$Tc = ktc \times R \times K \times (LS_{2D} - 4.12 \times Sg^{0.8}) \qquad \text{(Eq.S2)}$$

This model was applied to the lower-middle watershed of the *Tagus* river. Input data (raster layers with identical spatial resolution and coverage) consists of spatially distributed data or average values of soil erodibility (K parameter), rainfall erosivity (R parameter), crop management (C parameter), support-practice parameter (P parameter), a digital elevation model (DEM), drainage network, land occupation (parcel map), transport capacity coefficients (ktc) and slope gradient (Sg). For this study a grid cell resolution of 100 m was used, as used by Verstraeten (2006). The DEM is used to calculate the Sg (dimensionless) and the LS_{2D} parameter (dimensionless), as well as to trace the routing of the SS downstream to the river channel network. A functional DEM is not available for the entire watershed of the *Tagus* river. Therefore, we used shuttle radar topography mission (SRTM) elevation data (NASA 2003) to produce a DEM for the entire watershed (SRTM–DEM). The original resolution of the SRTM is 3 arc-seconds (90 m). The grid resolution of the SRTM–DEM was resampled to 100 m (Fig. 3.S.2).

The drainage network map (Fig. 3.S.3) was constructed based on the digital map of river channels for Portugal (SNIRH 2010) and the SRTM–DEM for the Spanish part of the watershed under study. On the drawing of the river channels for the Spanish part, we considered an upstream contributing area of 100 ha to determine whether a pixel was a river or not.

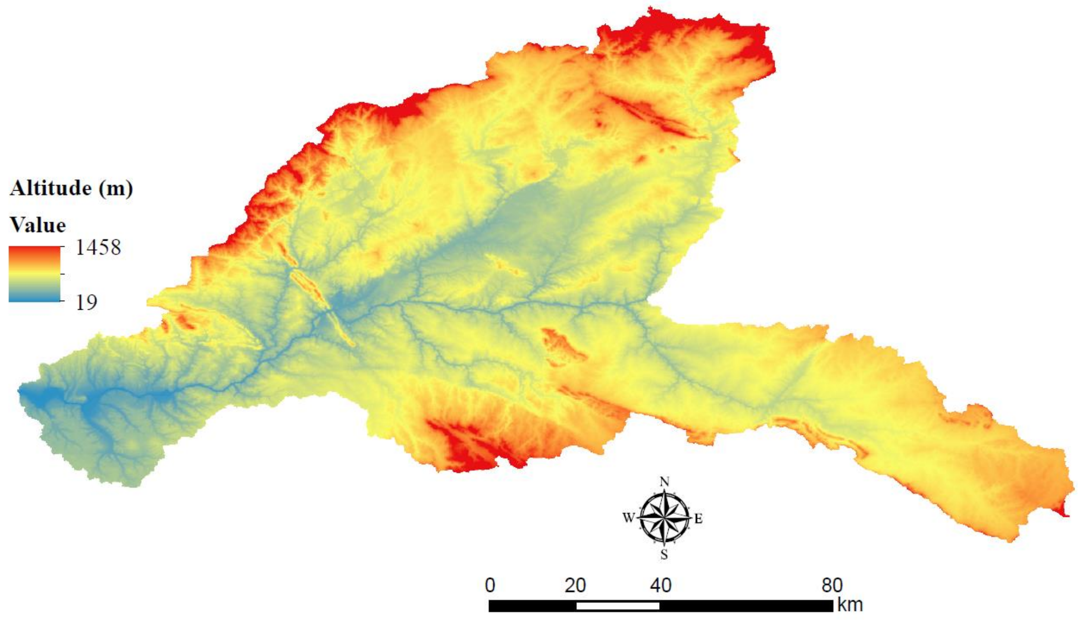

Fig. 3.S.2. Digital elevation model (SRTM–DEM) for the lower-middle watershed of the Tagus river with a resolution of 100 m.

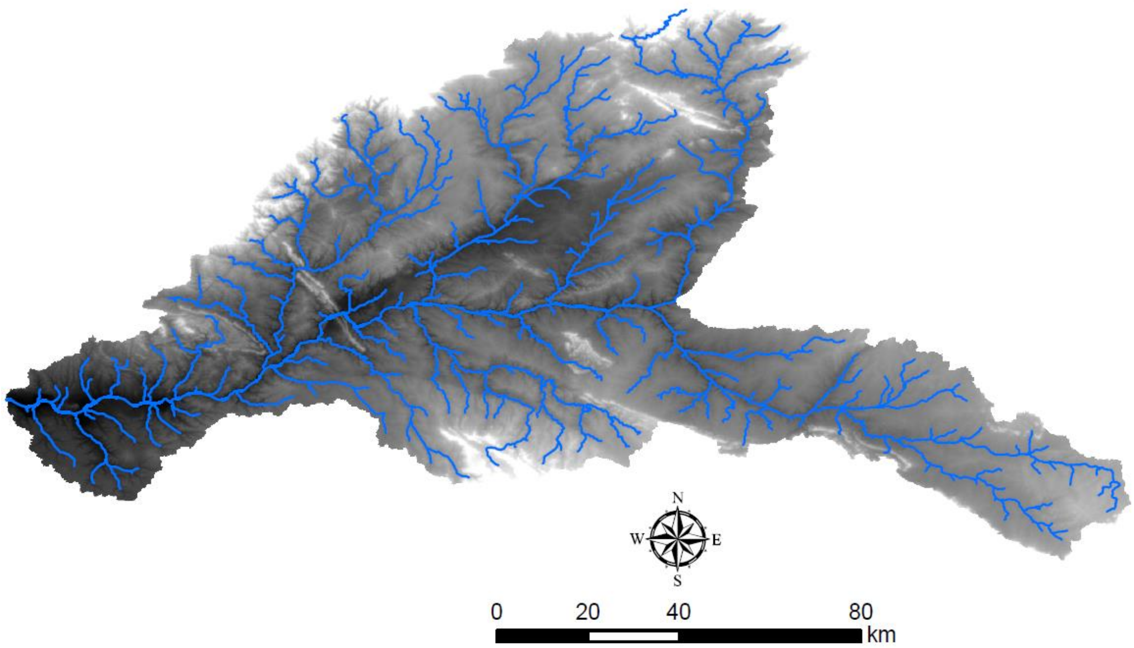

Fig. 3.S.3. Drainage network map of the lower-middle watershed of the Tagus river.

The land cover map (Fig. 3.S.4) was a reclassification of the CORINE 2006 Project land cover data (Caetano et al. 2009; EEA 2012). This map has a resolution of 100 m and uses 44 land use categories. The categories were grouped into five major categories: pasture and grassland (45.5 %), fields under agriculture (33.8 %), forests (17.8 %), infrastructure and built-up areas (2.2 %) and rivers and water bodies (0.7 %).

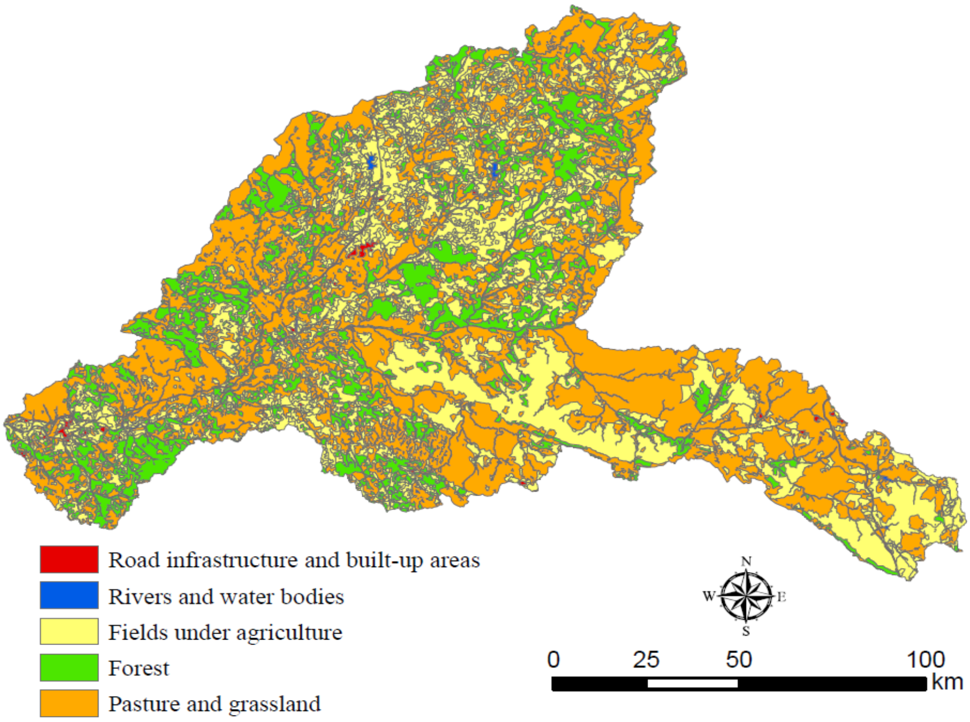

Fig. 3.S.4. Parcel map of the lower-middle watershed of the Tagus river (Caetano et al. 2009; EEA 2012).

The K parameter represents the susceptibility of the soil to erosion by water and is related to soil characteristics, namely structure, texture, organic matter content and permeability. A K parameter map is available from the Land Use/Cover Area frame Survey (LUCAS) (Panagos et al. 2014) topsoil data. The map is available at a spatial resolution of 500 m at European level. The dataset related to the entire watershed under study was extracted and resampled to the resolution of 100 m (Fig. 3.S.5). To adjust for the different units used in LUCAS ($t.ha.h.ha^{-1}.MJ^{-1}.mm^{-1}$) and WaTEM/SEDEM ($m^2.kg.h.MJ^{-1}.m^{-2}.mm^{-1}$), the LUCAS values were multiplied by 1000.

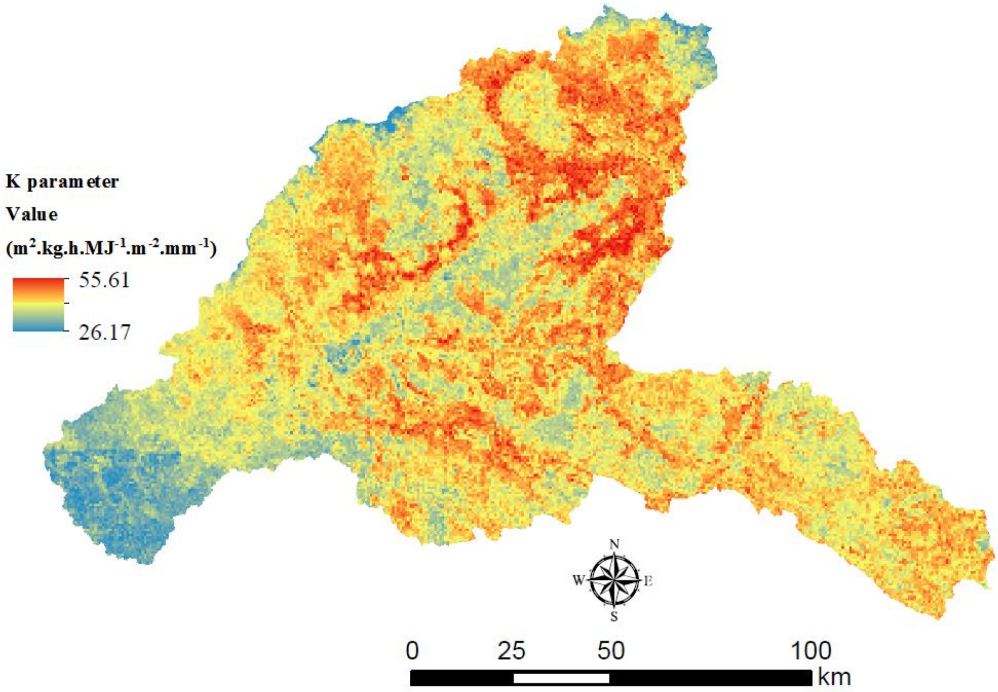

Fig. 3.S.5. Soil erodibility map (K parameter in RUSLE) of the lower-middle watershed of the Tagus river (Panagos et al. 2014).

The C parameter (dimensionless) determines the effectiveness of the land cover and management practices in the prevention of soil erosion. The C parameter map for the entire watershed under study (Fig. 3.S.6) was constructed based on the mean C parameters for different cropping systems under study (Pimenta 1998), and the 44 land use categories of the CORINE land cover map.

The R parameter ($MJ.mm.ha^{-1}.yr^{-1}$) represents the impact of rainfall on topsoil erosion. Based on Brandão et al. (2006), who assessed rainfall erosivity in Portugal, it was assumed that the average annual R parameter is constant throughout the entire watershed and equal to 223 $MJ.mm.ha^{-1}.yr^{-1}$ based on 5-min rainfall data for the period 1965-1995.

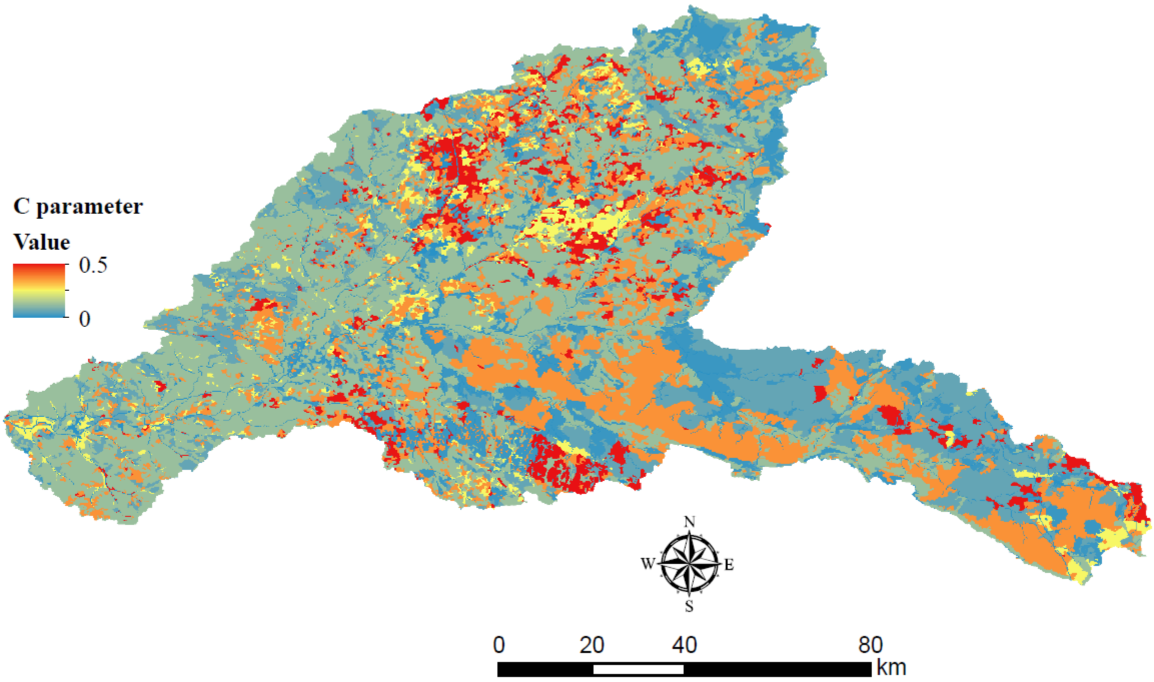

Fig. 3.S.6. Crop management map (C parameter in RUSLE) of the lower-middle watershed of the Tagus river (Pimenta 1998).

The transport capacity (Tc) represents the maximal amount of SS that can pass through one grid cell. To calculate the Tc, the ktc (transport capacity coefficient) values of 8 m for non-erodible land surfaces and 27 m for arable land surfaces (ratio of 1:3.38) developed by Verstraeten (2006) were used.

Finally, the support-practice parameter (P parameter, dimensionless) reflects the positive effects of the management operations in reducing soil erosion. This parameter is set to one in the WaTEM/SEDEM model (Notebaert et al. 2006).

Sensitivity analysis – transport capacity coefficient (ktc)

The pre-established ktc values for non-erodible land surfaces (8 m) and arable land surfaces (27 m) were varied jointly at a fixed ratio of 3.38. For each combination of ktc values the average annual SS load was predicted (Fig. 3.S.7). The modelled long-term average annual SS load for the entire watershed of the *Tagus* river with ktc values of 7 and 24 m was 7 % lower than when predicted with the pre-established calibrated ktc values (default values) from Verstraeten (2006) (727,692 $t.yr^{-1}$), whereas the predicted SS load with ktc values of 9 and 30 m was 6 % higher than when using the default ktc values. The simulations have been compared with the observed average SS load for a 5 years period on the one hand and with each individual year on the other hand. ktc values higher than the default ones result in a higher overprediction of SS delivery to the river (771,275 $t.yr^{-1}$ using ktc values of 9 and 30 m) compared to the observed average SS delivery for the entire period 1985-1989 (582,278 $t.yr^{-1}$). ktc values lower than the default ones result in a lower downstream transport capacity, i.e. the reduction of the maximum amount of SS that can pass through one pixel. The 7 and 24 m ktc values result in a lower long-term average annual SS load (679,338 $t.yr^{-1}$) than the long-term average annual SS load obtained using the default ktc values (727,692 $t.yr^{-1}$) compared to the observed average SS load for the entire period 1985-1989 (582,278 $t.yr^{-1}$). Although 7 and 24 m ktc values give a better approximation of the long-term average annual SS load, they underpredict the SS load on a yearly basis. When comparing the long-term predicted SS load of 679,338 $t.yr^{-1}$ with the observed SS load for each specific year, the model (using 7 and 24 m ktc values) underpredicts the SS load for 1985 and 1989 (709,969 and 741,171 $t.yr^{-1}$, respectively). Based on reason we have chosen to maintain the use of the default ktc values.

Furthermore, using ktc values lower than 7 and 24 m led to more underprediction of the SS load comparatively to yearly predictions.

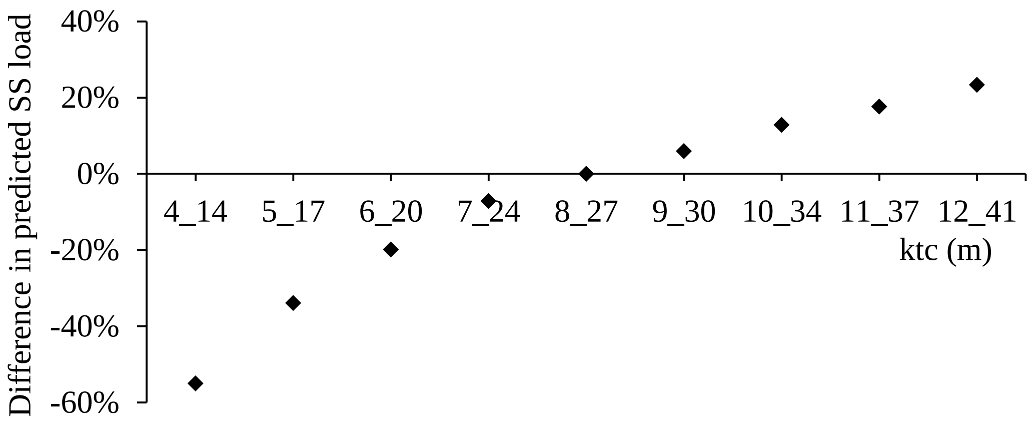

Fig. 3.S.7. Relative difference between the predicted SS load using the pre-established ktc values (8 and 27 m) and the predicted SS load using a different single ktc values set, keeping the ratio between both ktc values constant (1:3.38).

S3. Forest management inventory

Table 3.S.1. Consumption of materials during forest management operations over one *E. globulus* revolution.

Forest management operations			Consumption ($kg.ha^{-1}.revolution^{-1}$)	Source
Tillage, site preparation	**Disking, clearing**			(Carrilho et al. 2001; CAOF 2003; Emporsil & Soporcel 1995; Louro et al. 2000;)
		Diesel	40.3	
		Lubricating oil	2.0	
	Ripping			(Barros and Salinas 1981; Carrilho et al. 2001; CAOF 2003; Emporsil & Soporcel 1995; Louro et al. 2000;)
		Diesel	74.0	
		Lubricating oil	3.7	
	Subsoiling			(Carrilho et al. 2001; CAOF 2003; Emporsil & Soporcel 1995; Louro et al. 2000;)
		Diesel	36.9	
		Lubricating oil	1.8	
Tillage, stand tending	**Disking, cleaning**			(Carrilho et al. 2001; Emporsil & Soporcel 1995; Louro et al. 2000;)
		Diesel	201.3	
		Lubricating oil	10.1	
	Soil loosening			(Carrilho et al. 2001; Emporsil & Soporcel 1995; Louro et al. 2000;)
		Diesel	36.7	
		Lubricating oil	1.8	
Chainsaw, selection of coppice stems				Portuguese forest management company
		Petrol	13.1	
Road and firebreak building/maintenance				(CAOF 2003)
		Diesel	71.0	
		Lubricating oil	3.5	
Fertilising				Portuguese forest management company
	Ammonium sulphate, as N		257.1	
	Phosphate fertiliser, as P_2O_5		104.0	
	Potassium chloride, as K_2O		63.1	
Felling				
	Petrol		79.3	
	Lubricating oil		4.0	

S4. Impact assessment and sensitivity analysis

In order to establish the spatial SS inventories for the four *E. globulus* stands using the RUSLE, the following input parameters were considered: a C parameter of 0.2 for all the stands, and K parameters of 0.028 $t.ha.h.ha^{-1}.MJ^{-1}.mm^{-1}$ for stand 1, 0.037 $t.ha.h.ha^{-1}.MJ^{-1}.mm^{-1}$ for stand 2, 0.038 $t.ha.h.ha^{-1}.MJ^{-1}.mm^{-1}$ for stand 3 and 0.048 $t.ha.h.ha^{-1}.MJ^{-1}.mm^{-1}$ for stand 4. A R parameter value of 223 $MJ.mm.ha^{-1}.yr^{-1}$ for the entire watershed was used as input for the WaTEM/SEDEM model, thus also for each of the *E. globulus* stands under study. No data on the local long-term R parameter for the different stands were available.

The K parameter map of the entire watershed was based on the LUCAS survey (Panagos et al. 2014), as mentioned in Section S2, and has a spatial resolution of 500 m, i.e. an area of 25 ha. It should be kept in mind that stand 1 has an area of 8.2 ha and stand 2 has an area of 29 ha, which is much smaller than the resolution for which the K values have been derived. The K parameter map has been resampled to a spatial resolution of 100 m and soil erodibility values are still related to soil texture, organic matter, coarse fragments, structure and permeability parameters, which have been determined from topsoil samples and regression interpolation for a 25 ha resolution. Soil erodibility can vary within the extent of 25 ha, but to our knowledge no better data are available at a higher resolution. Therefore, we assume that the K values are suitable to establish the SS inventory for all the *E. globulus* stands under analysis.

Table 3.S.2. Impact assessment results, per forest management operation, associated with the occupation of 1 ha by *E. globulus* during one revolution.

E. globulus forest management operations	species.yr.ha^{-1}.revolution^{-1}				
	Freshwater eutrophication	Freshwater ecotoxicity	Terrestrial ecotoxicity	Terrestrial acidification	Climate change
Tillage, disking, clearing, site preparation	5.9×10^{-11}	3.7×10^{-11}	2.4×10^{-10}	6.3×10^{-9}	1.2×10^{-6}
Tillage, ripping, site preparation	1.1×10^{-10}	6.7×10^{-11}	$4.3\times10\text{-}^{10}$	1.2×10^{-8}	2.2×10^{-6}
Tillage, subsoiling, site preparation	5.4×10^{-11}	3.4×10^{-11}	2.2×10^{-10}	5.8×10^{-9}	2.7×10^{-7}
Tillage, disking, cleaning, stand tending	2.9×10^{-10}	1.8×10^{-10}	1.2×10^{-9}	3.1×10^{-8}	5.9×10^{-6}
Tillage, disking, soil loosening, stand tending	5.4×10^{-11}	3.3×10^{-11}	2.2×10^{-10}	5.7×10^{-9}	1.1×10^{-6}
Chainsaw, selection of coppice stems, stand tending	2.5×10^{-11}	1.5×10^{-11}	7.7×10^{-11}	4.2×10^{-10}	4.1×10^{-7}
Road and firebreak building/maintenance	1.0×10^{-10}	6.4×10^{-11}	3.1×10^{-10}	8.5×10^{-9}	2.0×10^{-6}
Fertilising	2.7×10^{-9}	1.3×10^{-7}	2.0×10^{-8}	4.6×10^{-7}	1.4×10^{-5}
Felling	1.2×10^{-10}	7.2×10^{-11}	3.5×10^{-10}	9.9×10^{-9}	2.4×10^{-6}
Total	3.5×10^{-9}	1.3×10^{-7}	2.3×10^{-8}	5.1×10^{-7}	2.9×10^{-5}

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CHAPTER 4

Chapter 4: General conclusions and future perspectives

1. General conclusions

Working towards a life cycle environmental sustainability assessment, this thesis aims to improve and enhance the Life Cycle Assessment (LCA) methodology regarding freshwater use and its quality degradation through the incorporation of new impact pathways, moving towards the inclusion of spatial differentiation: changes in green water flow due to land-use production systems and the direct effects of SS on the potential loss of aquatic macroinvertebrate, algae and macrophytes. This is relevant because evapotranspiration rates as well as cause and effect factors such as persistence of 'pollutants' in the receiving environment and its sensitivity can vary significantly in different geographical areas. Thus, the inclusion of spatial knowledge regarding the Life Cycle Inventory (LCI) and the receiving environment can increase the discriminating power of Life Cycle Impact Assessment (LCIA) as it can provide a better and more reliable prediction of local environmental impacts, helping in decision-making based on LCA.

The main conclusions that can be drawn from this thesis for freshwater-use related impacts are:

- The available methods differ significantly concerning the type of freshwater considered (green water, surface water, renewable groundwater, fossil groundwater), the cause-effect chain covered, the consideration of freshwater scarcity, stress and/or degradative freshwater use, and the spatial and temporal information.

  Despite the improvements achieved for measuring the freshwater-use related impacts and promoting freshwater use efficiency, there are still some unsolved problems needing further research, namely, the: (1) accounting for and assessing the potential environmental impacts of green water flows; (2) temporal and spatial variation to establish explicit characterisation factors (CFs), considering the local environmental uniqueness; (3) adequate connection between inventory flows and spatio-temporal explicit CFs.

  This thesis contributes to the resolution of some of these issues. A method for accounting for and assessing the potential environmental impacts of green water flow was developed and spatially explicit and species-specific CFs were derived for *Eucalyptus globulus* in Portugal, establishing an adequate connection between green water flows and CFs.

- Crop production and derived downstream products such as viticulture and wine production requires the inclusion of freshwater use impact pathways for meaningful LCA studies of processed and non-processed agricultural products. Various available methods emphasise different aspects of green and blue freshwater use. At the midpoint level, most methods are related to blue freshwater scarcity, considering CFs with various levels of spatial differentiation, i.e. country, river basin, watershed and grid cell differentiation, while green water availability and related impacts due to changes on green water flows have been poorly covered. Different spatial differentiation can lead to a large variability in the blue freshwater use impacts when assessing a product with different LCA-based methods, also increasing the uncertainty when comparing the same crop growing under different soil and climate conditions.

  The case study on white 'vinho verde' showed that background systems have a major impact on freshwater use downstream effects, suggesting that additional efforts should be made to provide databases with regional data instead of average data from generic databases.

- The growth of short rotation forestry is largely dependent on local precipitation. When assessing the potential environmental impacts of forest and agriculture resulting from changes in green water flow due to a land-use production system, the interactions between forest/crops and freshwater should be considered at both green water use and soil, and green water use and atmosphere interfaces.

  The difficulty of constructing potential natural vegetation maps based on remnants of natural or near-natural vegetation can increase the uncertainty in the LCI of green water flow. In addition, several areas can be covered by two different natural land covers such as quasi-natural forest and grasslands/shrublands. In this situation, both natural land covers should be considered when conducting a green water flow midpoint impact assessment.

  The large variation of impacts on terrestrial green water flow (TGWI), which ranges from 0 to 24 $m^3_{H_2O}.ha^{-1}.yr^{-1}$, when considering grasslands/shrublands as an alternative reference land use, and reductions in surface blue water production (RBWP), which ranges from 375 to 1174 $m^3_{H_2O}.ha^{-1}.yr^{-1}$, when considering grasslands/shrublands as alternative reference land use, caused by reductions in surface runoff due to *E. globulus* production, indicates the importance of understanding the spatial variability of local environmental conditions and cultivated plants.

Moreover, the newly developed method to derive midpoint CFs and to conduct a LCIA of green water flow supports the sustainable freshwater use of wood products, maximizing the mean annual wood volume growth increment, demystifying the idea that biomass production and forest-based products present a huge consumptive use of green water, leading to a severe depletion of soil moisture, groundwater and surface resources.

The main conclusions concerning the direct effects of SS on the potential loss of aquatic organisms are:

- Topsoil erosion by water is an important source of SS in freshwater streams but a comprehensive LCA of their damage due to a land-use production system has not yet been performed, until now. Displaced SS from soil are themselves a source of potential environmental harm, especially when they reach freshwater systems, leading to potential losses of macroinvertebrates, algae and macrophytes. To evaluate these potential damages, spatially explicit information it is required for quantifying the amount of SS from a specific land-use production system due to erosion by water that is transported through the landscape towards the drainage network, and for quantifying the environmental residence time of SS and the related effects at different locations.

  Although spatial information requires additional effort, especially for LCI data collection, it is critical for carrying out a more realistic environmental assessment, encompassing the local environmental uniqueness, such as soil erodibility, altitude and crop management, and accurately associating SS sources with receiving freshwater environments of variable sensitivity. In this work, a framework to conduct a spatially distributed SS delivery to freshwater streams was developed, using the WaTEM/SEDEM model combined with the developed method to derive site-specific CFs for endpoint damage on aquatic ecosystems diversity.

With a few exceptions, in which a higher CF is derived for algae and macrophytes, macroinvertebrates are more sensitive to SS than algae and macrophytes. This is due to the presence of SS in the water column damaging the respiratory and digestive structures of macroinvertebrates. In the case of algae and macrophytes, the presence of SS reduces light penetration through the water column, hampering their photosynthetic rates and damaging their photosynthetic structures in high water flow rates. This has been shown by the newly developed method that allows quantifying the potential impacts of SS on aquatic biota by deriving endpoint CFs for 22 different European river sections.

- A specific case study on *E. globulus* stands was performed as an example of a land-use production system. This case study showed the relevance of SS delivery to freshwater streams, providing a more comprehensive assessment of the SS impact from land use systems on aquatic environments. The damage resulting from additional SS input to the water column was compared with the damage obtained from the commonly used ecosystem impact categories of the ReCiPe method. The SS impacts ranged from 15.5 to 1234.9 PDF.m^3.yr.ha^{-1}.$revolution^{-1}$ for macroinvertebrates, and from 5.2 to 411.9 PDF.m^3.yr.ha^{-1}.$revolution^{-1}$ for algae and macrophytes. For some *E. globulus* stands, SS potential impacts on macroinvertebrates had the same order of magnitude of freshwater eutrophication, freshwater ecotoxicity, terrestrial ecotoxicity and terrestrial acidification impacts. For algae and macrophytes, most of the stands present SS impacts of the same order of magnitude as terrestrial ecotoxicity, one order of magnitude higher than freshwater eutrophication, and two orders of magnitude lower than freshwater ecotoxicity and terrestrial acidification.

The SS method that was developed enables ecosystem impacts to be assessed more comprehensively, showing that spatial SS impacts from land based systems on aquatic biota can be of the same order of magnitude or higher than the ones obtained for other ecosystem impact categories. Therefore, SS related impacts on macroinvertebrates, algae and macrophytes should be included in LCA studies of land use systems.

2. Future perspectives

The main shortcoming of the developed methods for accounting and assessing the impacts of green water flow, and for assessing the direct effects of SS on the potential loss of aquatic macroinvertebrate, algae and macrophytes are still related to the temporal variation. Average time series of climatic data and average data of other parameters (such as concentration of SS in water column) were used in the absence of better time discretized data. Additional efforts to include intra- and inter-annual time variation for both LCI and LCIA phases should be made.

Additional work, to establish environmental mechanisms that describe the interconnection between green water, surface blue water and recharge of renewable green water will be also required.

Additionally, complementary studies could also be performed:

- assessing the impacts on terrestrial green water flow and addressing reductions in surface blue water production caused by reductions in surface runoff for different land-use production systems. Other forest species and agricultural crops than *E. globulus* stands should be studied based on the proposed method, improving the model operationalisation and evaluating eventual adjustments to consider the specificity for each specie;

- extending the green water flow impact assessment model to the continental scale. A deeper understanding of drivers that affect rainfall disturbances and atmospheric moisture transport, connecting upwind evaporation sources with downwind rainfall sinks, linked to global climate models, is needed;

- implementation of a monitoring network to deeply understand the depletion of aquatic species due to SS, and therefore, to derive endpoint CFs at a global scale (beyond the European level);

- extending the study of the SS potential impacts for other forest and agriculture systems, thereby enhancing the knowledge of the impact of SS on aquatic biota in comparison with other impact categories;

- evaluating the feasibility of applying the developed framework and characterisation method for assessing the potential impacts caused by SS to SS from wastewater treatment processes.