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Authors:	Assumpció Anton, Marlia M. Hanafiah, Stefanie Hellweg, Mark A.J. Huijbregts, Ronnie Juraske, Llorenç Milà i Canals, Montse Núñez, Stephan Pfister, Catherine Raptis, Dominik Saner, Franziska Stössel, Rosalie van Zelm, Francesca Verones ***(alphabetical order)***
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List of abbreviations

A:	Area
AC:	Lateral cone area of a wetland
AET:	Actual evapotranpiration
AGWW:	Agricultural groundwater withdrawal
AI:	Aridity index
AoP :	Area of Production
aq:	Aquatic
ARWW:	Agricultural river water withdrawal
ASG :	Average Spring Groundwater
AWW:	Agricultural water withdrawal
CF:	Characterization Factor
CON:	Wate consumption
D:	Diameter of the area of concern for GW-GW wetlands
DALY:	Disability Adjusted Life Years lost
Dom:	Domestic sector
EF:	Effect Factor
Elec:	Elecricity production
ET:	Evapotranspiration
ET_0:	Reference evapotranpiration
FAO:	Food and Agriculture Organization
FEI:	Freshwater ecosystem impact
FF:	Fate Factor
GHG:	Greenhouse gases
GW:	Groundwater
GW-GW:	Groundwater-fed wetlands impacted by groundwater consumption
GW-SW:	Groundwater-fed wetlands impacted by surface water consumption
GWI:	Groundwater inflow into the area
GWR_I:	Groundwater recharge from excess irrigation
GWR_S:	Groundwater recharge from river seepage
GWW:	Groundwater withdrawal
h:	Water depth
I:	Amount of water infiltration into groundwater-fed wetlands (chapter 1+2)
I:	River inflow into the area (chapter 3)
IC:	Infiltraction capacity
IIASA:	International Institute for Applied Systems Analysis
Im:	Impact
IPCC :	Intergovernmental Panel on Climate Change
Irr:	Irrigation water use
ISO:	International Standards Organization
ISR:	Infiltration into the wetland Santa Rosa (chapter 3)
IW:	Irrigation Withdrawal
k_f:	Saturated hydraulic conductivity
LCA:	Life Cycle Assessment
LCIA:	Life Cycle Impact Assessment
LCI:	Life Cycle Inventory
Liv:	Livestock

m: Thickness of an aquifer
Man: Mabufacturing sector
NHI : National Hydrological Instrumentation
NPP: Net Primary Productivity
OS: Other sectors's water use (domestic, manufacturing, livestock, electricity prod.)
P: Precipitation
PAF : Potentially Affected Species
PDF : Potentially Disappeared Species
PET: Potential evapotranspiration
PNOF : Potentially not occurring fraction of plant species
PNV: Potential natural vegetation
Q: Surface water flow (river, into/out of the wetland)
r: Radius of a wetland
RF: Return flow
RWW: River water withdrawal
S: Species number
SW: surface water
SW-SW: Surface water-fed wetlands impacted by surface water consumption
T: Transmissivity
terr: Terrestrial
UNEP : United Nations Environment Programme
USDA : United States Department of Agriculture
V: Wetland volume
WC: Water consumption
WDF : Water Deprivation Factor
WIT: Water withdrawal
WSI: Water stress index
x: Net water consumption
z: Groundwater level change
α: Angle between vertical axis in wetland and slope of the embankment
τ: Residence time

Definitions

Green water: the precipitation on land that does not run off or recharge the groundwater but is stored in the soil or temporally stays on top of the soil or vegetation. Green water also refers in the deliverable to rainwater harvested and then reused in production systems.

Wetland: “areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres” (according to Ramsar convention).

Water consumption: water that is incorporated into a product, evaporated or diverted to another watershed and is not returning to the watershed or aquifer of origin.

Water withdrawal: The abstraction of water from surface water or groundwater.

Recommended assessment framework, method and characterisation factors for water resource use impacts

Executive Summary

The goal of Task 1.2 is to develop operational and scientifically sound methods for the assessment of groundwater and surface water use. The methods address impacts to human health (in terms of DALY/m^3) and impacts to ecosystems (in terms of PDF*yr*m^3/m^3 as well as absolute species loss per m^3). As impacts depend on the location of water consumption, the characterization factors of all methods were hence developed in a site-dependent manner. Fig.A gives an overview of the LCIA methods for water consumption developed within this project, with their main authors and reference to the according chapters of this deliverable as well as the matching of the relevant LCI flows.

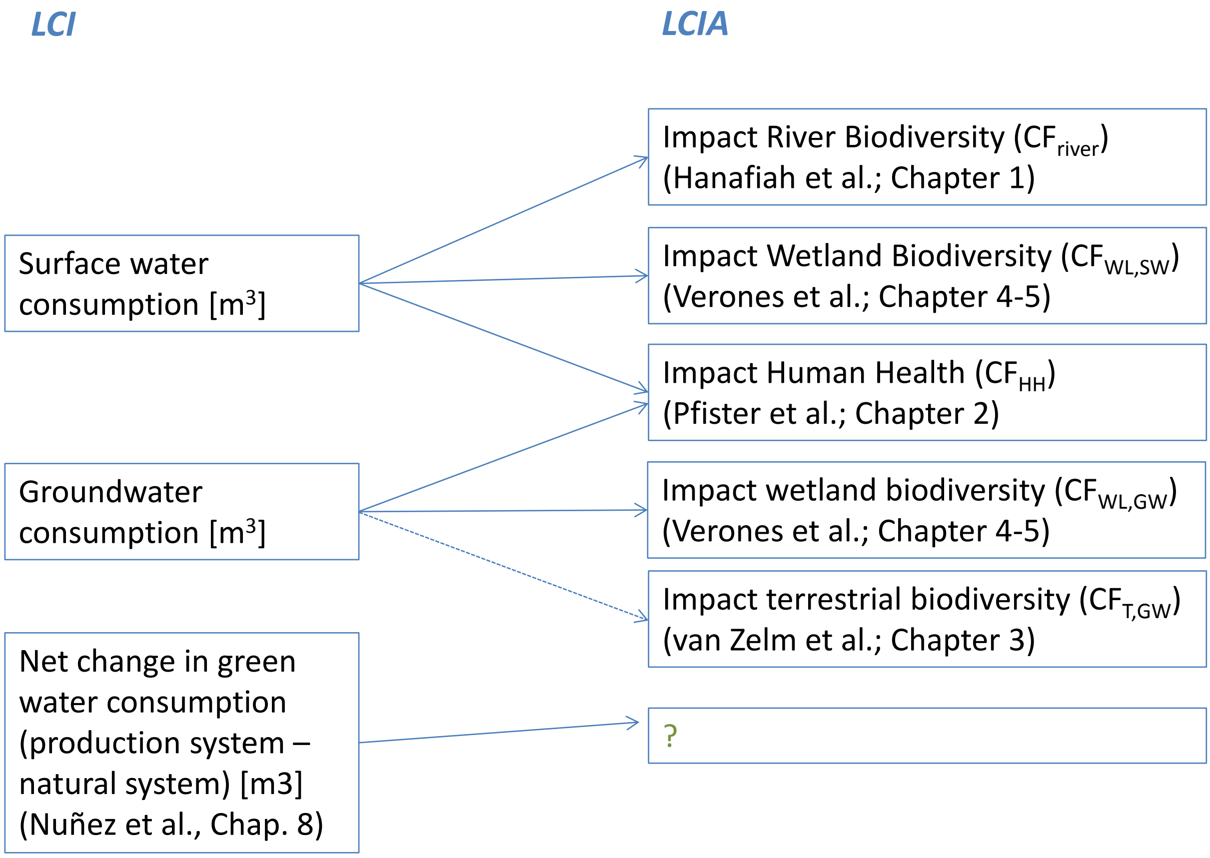

Fig.A: Overview of LCIA methods proposed by Task 1.2 of LC-IMPACT, matched to the respective inventory flows.

The methods are complementary and shall be used in conjunction. For example, if water is consumed from a river, the method of Hanafiah et al. (Chapter 1) quantifies the impact on freshwater species extinction in the river system (Characterization factors denoted as CF_{river}), while the method of Verones et al. (Chapters 4 and 5) assesses the biodiversity impact in wetlands of international importance ($CF_{WL,SW}$ and $CF_{WL,GW}$) downstream of the place of use. Additionally, the impact to human health (Chapter 2) should be assessed with the method of Pfister et al. (CF_{HH}). Similarly, for groundwater consumption the volume of water consumed needs to be multiplied with CF_{HH} according to the Pfister et al. method to assess impacts on human health as well as with $CF_{WL,GW}$ to assess the biodiversity impact on groundwater-fed wetlands. The method by van Zelm et al. (Chapter 3) additionally quantifies the impact of groundwater use on terrestrial biodiversity, but so far characterization factors are not available on a global scale. In the following, a summary of each method will be given one-by-one.

In Chapter 1, a new method for assessing water consumption on freshwater fish species extinction is presented. Characterization factors were derived for water consumption and global warming based on freshwater fish species loss. Calculation of characterization factors for potential freshwater fish losses from water consumption were estimated using a generic species-river discharge curve for 214 global river basins. Characterization factors were also derived for potential freshwater fish species losses per unit of greenhouse gas emission. Based on five global climate scenarios, characterization factors for 63 greenhouse gas emissions were calculated. Depending on the river considered, characterization factors for water consumption can differ up to 3 orders of magnitude. Characterization factors for greenhouse gas emissions can vary up to 5 orders of magnitude, depending on the atmospheric residence time and radiative forcing efficiency of greenhouse gas emissions. An emission of 1 tonne of CO_2 is expected to cause the same impact on potential fish species disappearance as the water consumption of 10-1000 m^3, depending on the river basin considered. The results make it possible to compare the impact of water consumption with greenhouse gas emissions. The method for assessing water consumption on freshwater fish species extinction is the first that explicitly addresses effects of water consumption on aquatic ecosystems. It has the strengths that it can be used for all rivers located at latitudes lower than 42^o and that its unit is compatible with endpoint units of other impact categories contributing to ecosystem damage. Short comings are that the method is currently not able to assess impacts for river systems that are located north of latitude 42^o and also it is only valid for rivers close to natural state.

For the Life Cycle Impact Assessment of human health impacts due to water consumption, the method of Pfister et al. (2009) was used as a starting point (Chapter 2). Characterization factors were directly taken from the original method, but enhanced with an uncertainty analysis. Parameter uncertainties were estimated based on analysis of original data sources and, where applicable, on quantitative assessment of the model uncertainty of the data provided by third parties. Spatial variability was also considered. The uncertainties were propagated within the cause-effect impact model by applying the stochastic procedure of Latin Hypercube with the software @Risk. The average k-values (dispersion factor denoting the 95% confidence interval if the median is divided (lower bound) and multiplied (upper bound) by k) was equal to k=2.76 on the midpoint level and 18.1 on the endpoint level. The aggregation from watershed to country level resulted in an average uncertainty of k=19.2 for the endpoint characterization factors and represents the variability of watershed factors within the country. The results show high spatial diversity of k-values and make it difficult to derive generic estimates of uncertainty, especially regarding the endpoint characterization factor. While the k-values of the midpoint assessment are generally rather robust, with 95% confidence intervals of one order of magnitude in most cases, the endpoint factors for DALYs feature extreme uncertainty, in some cases stretching over 4 orders of magnitude. Aggregation on country level adds to the uncertainty. Uncertainty related to the scope of the cause-effect chain was not addressed. It is concluded that spatial aggregation of impact factors should be carefully considered, as the uncertainties are considerably larger on country level than on higher level of spatial detail (i.e. watershed level). Furthermore, uncertainties on the endpoint level can be very large and should be revealed in any analysis assessing health impacts from water consumption. One key strength of the method presented is that it is available on various spatial resolutions, thus allowing to reduce spatial uncertainty if the according inventory data is available. One shortcoming of this methodology is, however, that only the impact pathway of health effects of malnutrition as a consequence of lack of water for agriculture is represented. This is not the only cause-effect chain existing and therefore the method runs at risk of not accounting for the full extent of health damage.

Next, a method to address the effects of groundwater extraction on the species richness of terrestrial vegetation in life cycle impact assessment was developed (Chapter 3). Characterization factors were derived for the Netherlands. They indicate the change in potential plant species richness due to a change in groundwater extraction, and consist of a fate and an effect factor. The fate factor was defined as the change in drawdown due to a change in groundwater extraction and expresses the amount of time required for groundwater replenishment. It was obtained with a hydrological model describing the groundwater regime. The effect factor was obtained from a groundwater level response curve of potential plant species richness, which was constructed based on the soil moisture requirements of 625 plant species. Depending on the initial groundwater level, effect factors range up to 9.2% loss of species per 10 cm of groundwater level decrease. Resulting characterization factors are up to 4.6 $m^2 \cdot yr/m^3$. With the proposed approach effects of groundwater extraction on the ecosystem can be included in LCA and aggregated with effects related to other stressors. The method is showcasing how impacts to terrestrial ecosystems may be quantified in a particular location. The advantage of this method is that it builds on very good and detailed data and models, with regard to groundwater modeling as well as plant data. Drawbacks are that this method is, so far, not applicable to other regions, particularly not to arid regions. .

A new method for assessing water consumption on wetland ecosystems was developed within this project (Chapters 4 and 5). Wetlands harbour particularly high biodiversity (e.g. bird biodiversity) and are being degraded at rapid pace globally. They therefore deserve special attention within life cycle impact assessment. Characterization factors were derived for water consumption in a site-specific manner, on a global scale. Surface water abstraction and groundwater abstraction was modeled separately. Fate factors relate the change in wetland area as a consequence of consuming 1 m^3 of water from the ground- or surface water. Effect factors quantify the number of species globally lost per unit of area lost in a wetland, taking into account the species-area relationship as well as the global occurrence of species. Thus, the characterization factor accounts for the number of species lost per m^3 of water use. Both fate and effect factors show a high spatial dependence and vary over several orders of magnitude between different wetlands. Factors are provided for waterbirds, non-residential birds, water-dependent mammals, reptiles and amphibians. The characterization factors are mostly largest for waterbirds (2E-04 species-eq$\cdot$yr/m^3 for surface water consumption and 1.6 E-04 species-eq$\cdot$yr/m^3 for groundwater consumption), which is related to the large species richness. For bird species 76% of the land surface are covered for surface water consumption from surface water-fed wetlands and 37% of land surface for groundwater consumption from groundwater-fed wetlands. Reptiles, amphians and mammals are not present in all wetlands and thus the CF can be zero.

In addition to the mandatory parts of the deliverable, two studies in which newly developed impact assessment methods were applied in case studies are provided (Chapters 6 and 7.

A modified version of the methodological approach outlined above is applied in Chapter 6. Regionalized characterization factors for a groundwater-fed wetland at the arid coast of Peru are developed for ground- and surface water withdrawal and consumption. Several agricultural scenarios for 2020 were developed in a workshop with local stakeholders and used for calculating total biodiversity impacts. Irrigation with surface water leads in this specific region to benefits for the groundwater-fed wetland, due to additional groundwater recharge from surplus irrigation water. However, irrigation with groundwater leads to ecological damages on the wetland.

In another case study, the impact of increased salinization of a coastal wetland in Spain is quantified (Chapter 7). The fate factor was calculated from seasonal water balances of the wetland Albufera de Adra (Almería, Spain). The effect factor was obtained from the fitted curve of potentially affected fraction of species due to salinity, constructed from original salinity concentration-response curves for species native to the wetland. An assessment of water consumption of the greenhouse crops in the river basin was conducted as a case study for the wetland.

Finally, the impact of green water consumption has been addressed by a method to quantify regionalized green water consumption (Chapter 8). The approach presented consists in quantifying, in the LCI phase of an LCA study, the net change in the evapo(transpi)ration of the production system compared to the natural reference situation. The potential natural vegetation (PNV) was the natural reference situation selected. We provided regionalised PNV evapotranspiration values at different spatial aggregation levels in order to assist in the applicability of the method for different spatial information detail levels available in the LCI: from the most detailed grid-cell level (10 arc-minutes resolution) to the sizable biomes and continents, and a global default value. Two methods were combined to estimate PNV green water consumption: an empirical equation, based on precipitation and potential evapotranspiration data; and an empirical model, based on remote sensing data sets. The method is designed to quantify direct soil-water uptake by (especially rain-fed) agriculture and forestry systems and in cases where rainwater is harvested and then reused in the production system. The major drawback of the work is that we focused on quantifying PNV evapotranspiration on dry lands, when green water consumption can have negative environmental effects in all climates (e.g., floods). The other major drawback is that we addressed green water consumption in the LCI phase. This means that we did not develop a proper LCIA scheme, although we provided some tracks that can be followed to define the impact category indicator. A temporal solution until a most sophisticated approach is developed is assessing the change in green water consumption with the CFs developed in Chapters 1-7.

Task 1.2 is of a very innovative nature, as impacts are assessed which have not or only very coarsely been addressed within LCIA so far. Because some of these methodological developments were starting from scratch and due to the heterogeneous nature of impacts assessed, it was not always possible to harmonize the endpoint indicators used. For example, conventional indicators, such as potentially damaged fraction of species for ecosystem impacts, are provided by all methods presented in this deliverable; additionally, these indicators were also modified to address biodiversity loss in a more complete manner by some methods, e.g. taking into consideration the rarity of species. This may make the application of LCA more complex, on the one hand, but also increase robustness and reliability, on the other hand.

The methods proposed have several advantages over the ILCD recommended method for water use:

- they distinguish between surface and groundwater use
- they are able to model impact pathways up to the endpoints human health and ecosystem health
- they have a more adequate spatial resolution, based on environmental mechanisms and not political boundaries.

1. Characterization factors for water consumption and greenhouse gas emissions based on freshwater fish species extinction

Marlia M. Hanafiah,§,¥, Marguerite A. Xenopoulos≠, Stephan Pfister#, Rob S.E.W. Leuven§, Mark A.J. Huijbregts§*

§Department of Environmental Science, Institute for Water and Wetland Research, Radboud University Nijmegen, Nijmegen, The Netherlands

¥Department of Environmental Science, National University of Malaysia, 43600 UKM Bangi, Selangor, Malaysia

≠Department of Biology, Trent University, Peterborough, Ontario, K9J 7B8, Canada

#ETH Zurich, Institute of Environmental Engineering, 8093 Zurich, Switzerland

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1.1. Introduction

Life cycle assessment (LCA) is a technique used to assess the environmental impacts associated with a product, process or service (ISO 2006). This paper focuses on life cycle impact assessment (LCIA), the phase where inventory data are assessed in terms of environmental impacts. Impact categories in LCIA can be associated with areas of protection (AoPs), such as natural resources, ecosystem quality and human health (Udo de Haes et al. 2002). The relationship between inventory data and the magnitude of impacts on the AoPs in LCIA are expressed in terms of characterization factors (Pennington et al. 2004).

Global freshwater biodiversity is one of the AoPs which has experienced large adverse effects (Dudgeon et al. 2006). Although freshwater fish species losses due to anthropogenic impacts have been addressed in earlier studies (Reist et al. 2006, Wrona et al. 2006, Buisson et al. 2008), less attention has been paid to assessing these impacts in an LCA perspective (Koehler 2008). At present, freshwater-related studies using LCA techniques have mostly focused on toxicological effects (Pennington et al. 2004, 2006, Van de Meent and Huijbregts 2005, Larsen and Hauschild 2007). The environmental impacts of water consumption on terrestrial ecosystems has only recently been conducted by Pfister et al. (2009). Impacts of water consumption and greenhouse gas emissions in relation to freshwater biodiversity have so far not been addressed in LCA context.

Global warming and increases in water consumption can significantly affect freshwater ecosystems (Vorosmarty et al. 2000, Xenopoulos et al. 2005). For example, reduced river discharge (the volume of water flowing through a river per unit time) due to water consumption and greenhouse gas emissions could lead to freshwater fish species losses (Xenopoulos and Lodge 2006). In lotic freshwater ecosystems, river discharge can be used as a surrogate of habitat space to generate species-discharge relationships similar to terrestrial species-area curves (Xenopoulos and Lodge 2006, Oberdorff et al. 1995, Poff et al. 2001). Because climate warming and water consumption is expected to reduce river discharge in many parts of the world (Postel 2000), these species-discharge relationships have been used to forecast species diversity losses associated with reductions in freshwater. In addition, river discharge reduction can, for instance, lead to a higher concentration of nutrients and pollutants in freshwater thus compounding the negative effects of water quantity reductions alone on biodiversity (Xenopoulos and Lodge 2006). Changes in temperature and precipitation associated with global warming can also adversely affect water availability. It is expected that river discharge reduction due to global warming can negatively influence the distribution and occurrence of many fish species (Figure 1.1) (Buisson et al. 2008, Mohseni et al. 2003, Chu et al. 2005).

The aim of this paper is to derive characterization factors related to freshwater ecosystem damage for water consumption and greenhouse gas emissions. The present study focuses on the occurrence of freshwater native fish species in global rivers. In order to put our results into LCA perspective, we also calculate normalization factors for water consumption and global warming as input for overall normalization factors that represent biodiversity impacts in freshwater. Normalization factors provide information about the relative importance of each impact category considered, such as impacts on freshwater biodiversity.

1.2. Methods

1.1.1 Framework

Figure 1.1 gives an overview of the cause-effect chain regarding the disappearance of freshwater fish species caused by greenhouse gas emissions and water consumption. In this study, water consumption refers to water used for human activities, (e.g. communal, agricultural and industrial) that is not returned to the river. The influence of reduced flow rates on fish species numbers can be quantified with the global species-discharge model, an index of habitat space, feeding and reproductive opportunities. This model was developed on the basis of information on native fish species and river discharges in various river basins (Xenopoulos et al. 2005). This model assumes a positive correlation between the number of freshwater fish species and average river discharges at the mouth of river basins.

$$R = 4.2 \cdot Q_{mouth,i}^{0.4} \quad (1.1)$$

where R is the freshwater fish species richness and Q_{mouth} is the annual average river discharge at the river mouth of basin i ($m^3.s^{-1}$).

The species-discharge relationship can be used as a basis to calculate characterization factors for water consumption that specify freshwater fish species extinction per unit of reduced river discharge for river basins in different regions of the world (Xenopoulos et al 2005). This has been done in a river basin-specific way. Using the data provided in Xenopoulos et al. 2005, information of the average river discharge for 326 river basins was considered. These 326 rivers include well-known river basins in the world, representing a wide geographical distribution of rivers around the various continents. However, we excluded 83 river basins which are located at latitudes higher than 42^o, because these river basins were recently (in geological time) glaciated, i.e. covered by ice. As such, these rivers have not had enough time to evolve to their maximum species richness potential. It follows that the species-discharge relationship for these river basins is weak as they have much fewer species per unit discharge than the rivers below 42^o. This indicates that most of the world's river basins located in the high latitudes including Northern Europe, Northern America and Canada were not taken into account. In addition, due to data limitations in the river volume and length calculations, 29 river basins were also excluded. Thus, a total of 214 river basins were used in our final models.

The species-discharge relationship can also be used to derive characterization factors that quantify the potential extinction of freshwater fish species per unit of greenhouse gas emission. The endpoint modelling for global warming further includes the influence of greenhouse gas emissions on global mean temperature and subsequent effects on river water discharge (see Figure 1.1). The calculation of the characterization factors for water consumption and global warming is explained below.

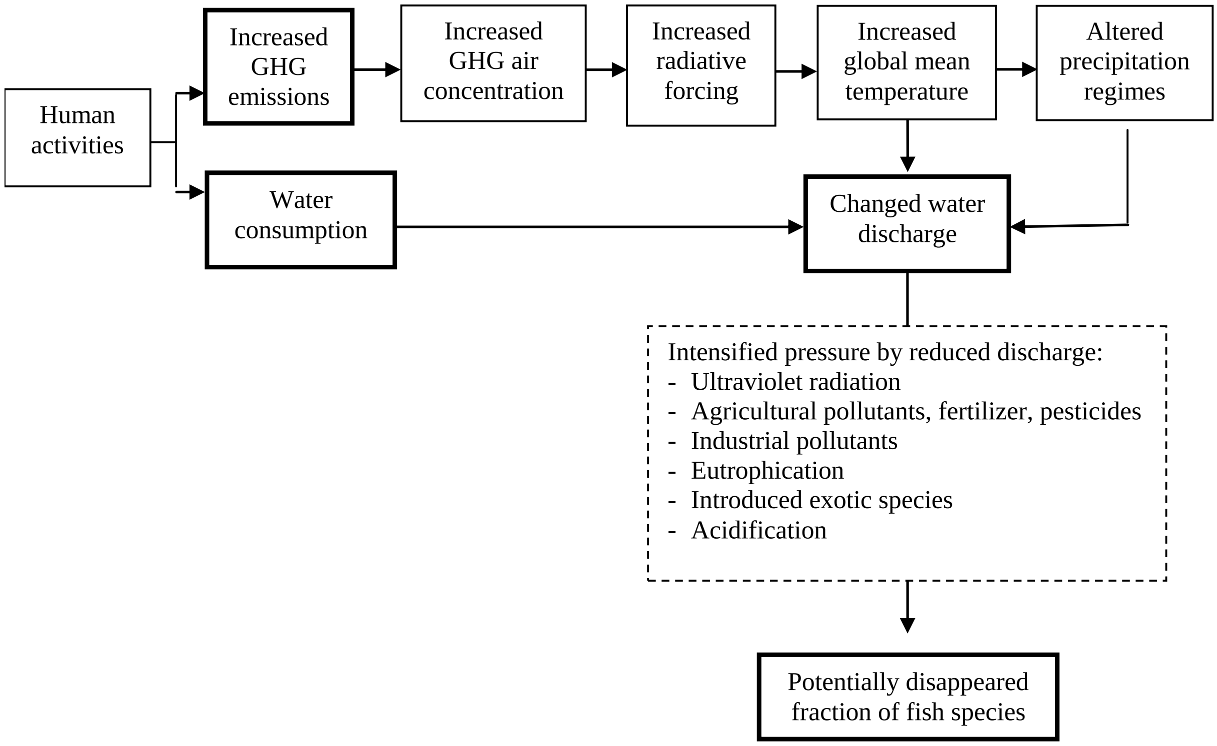

Figure 1.1. Cause-effect chain for impact of greenhouse gas emissions and water consumption on freshwater fish species (Xenopoulos et al. 2005, Xenopoulos and Lodge 2006).

1.1.2 Water Consumption

Characterization factors for water consumption reflect the impact of water use due to human activities on freshwater fish species richness, expressed in units of PDF·m^3·yr·m^{-3}. The river basin-specific characterization factors for water consumption (CF$_{wc,i}$) were calculated by:

$$CF_{wc,i} = FF_i \cdot EF_i = \underbrace{\frac{dQ_{mouth,i}}{dW_i}}_{fate} \cdot \underbrace{\left(\frac{dPDF_i}{dQ_{mouth,i}} \cdot V_i \right)}_{effect} \tag{1.2}$$

where FF_i is the fate factor of river basin i, EF_i is the effect factor of river basin i (PDF·m^3·yr·m^{-3}), $dQ_{mouth,i}$ is the marginal change in water discharge at the river mouth in basin i (m^3·yr^{-1}), dW_i is the marginal change in water consumption by human activities in river basin i (m^3·yr^{-1}), $dPDF_i$ is the marginal change in the potentially disappeared fraction of the freshwater fish species due to the marginal river discharge change $dQ_{mouth,i}$ and V_i is the volume of river basin i (m^3). The $dQ_{mouth,i}/dW_i$ was assumed to be equal to one, indicating that a change in water consumption (m^3·yr^{-1}) is fully reflected in a change in water discharge at the mouth for that river basin (m^3·yr^{-1}).

The effect factor for each river basin was calculated by:

$$\frac{dPDF_i}{dQ_{mouth,i}} = \frac{dR_i}{R_i \cdot dQ_{mouth,i}} = \frac{4.2 \cdot 0.4 \cdot Q_{mouth,i}^{0.4-1}}{4.2 \cdot Q_{mouth,i}^{0.4}} = \frac{0.4}{Q_{mouth,i}} \tag{1.3}$$

where $dPDF_i$ is the marginal change in the potentially disappeared fraction of the freshwater fish species for river basin i, d$Q_{mouth,i}$ is the marginal discharge change at the river mouth in basin i (m^3·yr^{-1}) and dR_i is the marginal change of the freshwater fish species richness in river basin i. River basin-specific discharges at the river mouth $Q_{mouth,i}$ were derived from the WaterGap model (Alcamo et al. 2003b).

The river volumes (m^3) for all river basins were calculated by:

$$V_i = \frac{Q_{mouth,i}}{2} \cdot \tau_i \qquad (1.4)$$

where V_i is the water volume in river basin *i* (m^3), $Q_{mouth,i}$ is the discharge at the river mouth in basin *i*, and τ_i is the average residence time of water in river basin *i* (s). Assuming a linear increase of river flow over the distance, we estimated that the average river discharge was half of the discharge at the river mouth. Derivation of the river volume was based on data from various sources (Xenopoulos et al. 2005, Alcamo et al. 2003b, Hugueny 1989, Fekete et al. 2000, Döll et al. 2003, Earthtrends 2007). Further details of the derivation of the river volume can be found in the Annex, Section 9.1 (estimation of river volumes).

1.1.3 Greenhouse Gas Emissions

Characterization factors for greenhouse gas emissions quantify the fraction of freshwater fish species that potentially disappear due to a change in emission of greenhouse gases. The characterization factors for 63 greenhouse gas emissions (in PDF·m^3·yr·kg^{-1}) were calculated by:

$$CF_{ghg,x} = FF_x \cdot EF = \underbrace{\frac{dTEMP}{dGHG_x}}_{fate} \cdot \underbrace{\left(\sum_i \frac{dQ_{mouth,i}}{dTEMP} \cdot \frac{dPDF_i}{dQ_{mouth,i}} \cdot V_i \right)}_{effect} \qquad (1.5)$$

Where FF_x is the fate factor for greenhouse gas emission *x* (°C·yr·kg^{-1}), *EF* is the effect factor (PDF·m^3·°C^{-1}), $dGHG_x$ is the change in greenhouse gas emission *x* (kg·$year^{-1}$), *dTEMP* is the change in global mean temperature (°C), $dQ_{mouth,i}$ is the change in water discharge at the river mouth in basin *i* (m^3·yr^{-1}), $dPDF_i$ is the marginal change in the potentially disappeared fraction of freshwater fish species in river basin *i* and V_i is the volume of river basin *i* (m^3).

Temperature factors were taken from De Schryver et al. (2009) and consist of three calculation steps. The first step resembles the change in air concentration of greenhouse gases due to a change in emission and reflects the atmosphere life time of a greenhouse gas. The second step represents the change in radiative forcing due to a concentration change. The third step reflects the change in global mean temperature due to the change in radiative forcing. The climate sensitivity and heat absorption rate by the oceans determine the relation of global mean temperature change and radiative forcing change (Randall et al. 2007). A time horizon of 100-year was applied in the present study. The indirect cooling effect of ozone depleting substances was not included in the greenhouse gas calculations due to the high uncertainties involved (see De Schryver et al. 2009).

Freshwater effect factors related to climate change require river basin-specific information on the change in PDF due to a change in global mean temperature. The effect factor was derived by:

$$EF = \sum_i \frac{dQ_{mouth,i}}{dTEMP} \cdot \frac{dPDF_i}{dQ_{mouth,i}} \cdot V_i \approx \sum_i \frac{\Delta Q_{mouth,i}}{\Delta TEMP} \cdot \frac{0.4}{Q_{mouth,i}} \cdot V_i \qquad (1.6)$$

where $dQ_{mouth,i}$ is the change in the water discharge at the river mouth in basin *i* (m^3·yr^{-1}) and *dTEMP* is the change in global mean temperature (°C). It is not possible to derive $dQ_{mouth,i}/dTEMP$ analytically, thus, data from IPCC (2001) and Millennium Ecosystem Assessment (2005) as described in Xenopoulos et al. (2005) and Sala et al. (2005) were used for the derivation of $\Delta Q_{mouth,i}/\Delta TEMP$ for five global climate scenarios in the year 2100. For every scenario, we divided the modelled change in river discharge from the WaterGap model (Alcamo et al. 2003b) by the predicted temperature change for the year 2100. Further information on the five global climate scenarios can be found in the Annex (Section 9.1).

River discharge is predicted to increase in some areas of the world due to increased precipitation (Rosenzweig et al. 2007). Without human accidental or intentional fish introductions, it is unlikely that increasing river discharge will have a positive effect on fish species richness, particularly at the current time scale as related to local scale and isolated river basins (Xenopoulos et al. 2005). Therefore, river basins with increased discharge were excluded in the calculation of the effect factor for global warming.

1.1.4 Normalization

Normalization factors provide information about the relative importance of each impact category and were expressed as the potentially disappeared fraction of species over a certain river volume per capita. Normalization factors for water consumption refer to the year 1995 (Alcamo et al. 2003a, 2003b, Shiklomanov 1999), while normalization factors for global warming were based on greenhouse gas emissions in year 2000 (Sleeswijk et al. 2009). The population numbers were taken from the U.S. Census Bureau (2010). Due to lack of data, we were only able to derive the normalization factors for water consumption and global warming for 112 river basins and 21 greenhouse gas emissions, respectively.

1.3. Results

1.1.5 Water Consumption

River basin-specific characterization factors for water consumption differs 3 orders of magnitude (Figure 1.2). Most of the river basins (57%) have characterization factors for water consumption between 10^{-4} – 10^{-3} PDF·m^3·yr·m^{-3}. The characterization factors for the largest river basins in the world, such as the Nile, the Amazon and the Yangtze Rivers are between 10^{-3} – 10^{-2} PDF·m^3·yr·m^{-3}. Characterization factors for all 214 river basins can be found in the Annex (Section 9.1).

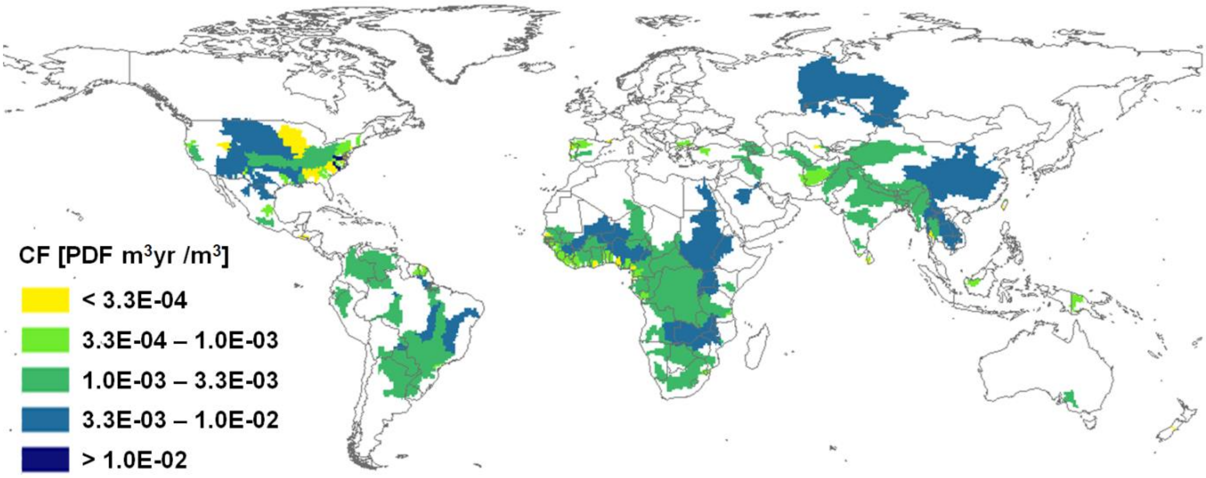

Figure 1. 2. Characterization factors for water consumption (PDF·m^3·yr·m^{-3}).

1.1.6 Greenhouse Gas Emissions

Characterization factors for CO_2, CH_4, N_2O, CFC-11, SF6 and HFC-125 emissions are shown in Figure 1.3 (ranges from $8.5 \cdot 10^{-5}$ to 2.1 PDF·m^3·yr·kg^{-1}). The largest characterization factor is found for SF6 (around 4 orders of magnitude larger than CO_2). The differences between the greenhouse gases are determined by the differences in atmospheric residence time and radiative forcing efficiency. The rivers with the largest contribution to the characterization factors for global warming are the Amazon, Madeira, Orinoco, Purus and Brahmaputra. These rivers explain together 65% of the freshwater ecosystem impact per unit of greenhouse gas emission. The river basin-specific effect factors and the characterization factors of 63 greenhouse gases are listed in the Annex (Section 9.1).

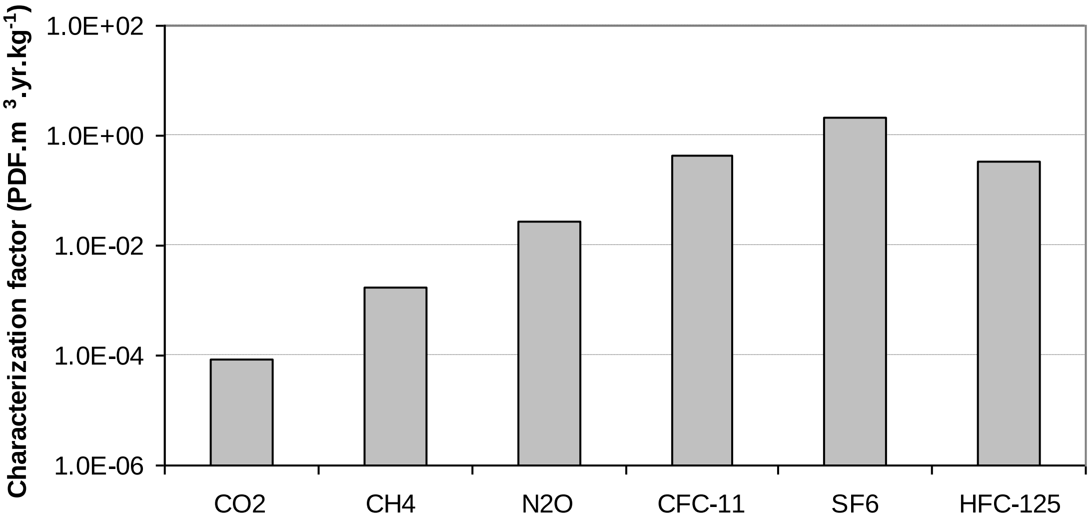

Figure 1.3. Characterization factors of six greenhouse gas emissions (PDF·m^3·yr·kg^{-1}) from a 100-year time horizon.

1.1.7 Normalization

The normalization factors per capita for water consumption and global warming are approximately equal (respectively 0.54 and 0.57 PDF·m^3/capita). For water consumption, the highest normalization factor is found for the Ganges River, which constitutes 22% impact of the river basins considered (Figure1. 4A). The normalization factor based on emissions in year 2000 shows that CO_2 contributes most to global warming, with 70% of the total greenhouse gas emissions included (Figure 1.4B). Normalization factors for river basin-specific water consumption and greenhouse gas emissions are given in the Annex (Section 9.1).

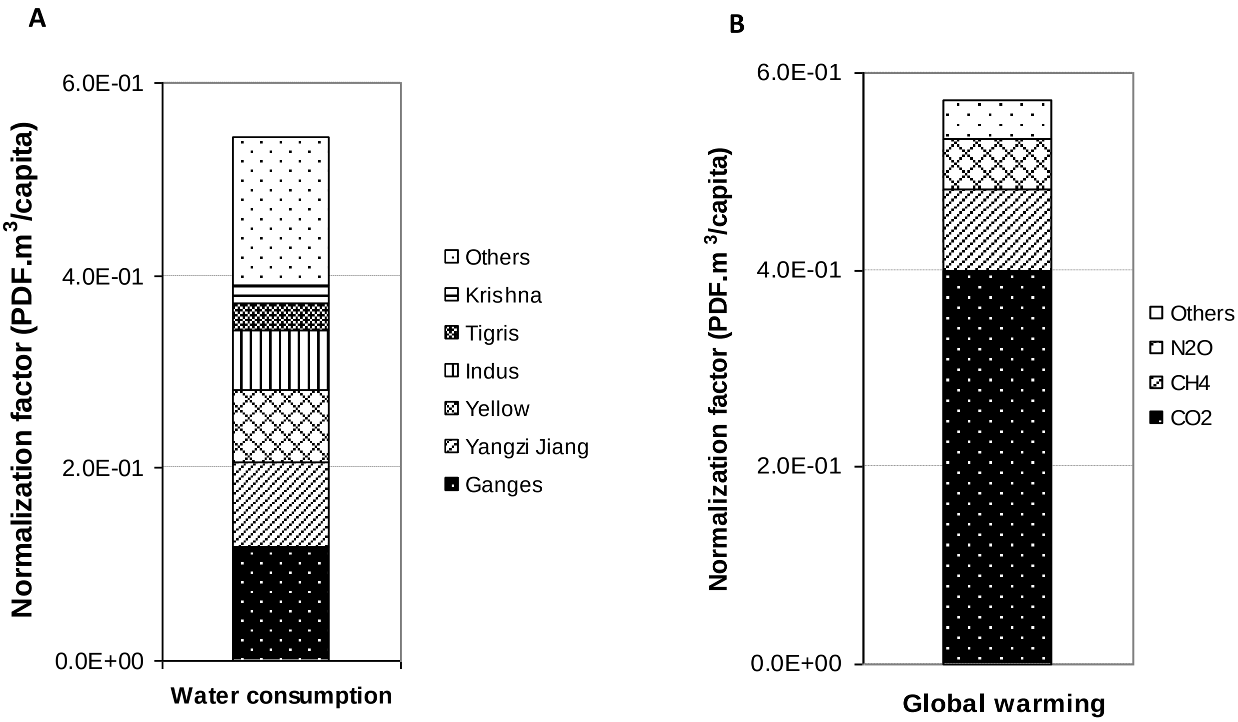

Figure 1.4. River basin-specific normalization factors (PDF·m^3/capita) for water consumption in year 1995 (4A) and normalization factors for global warming based on emissions in year 2000 (4B).

1.4. Discussion

We were able to derive characterization factors for water consumption and global warming based on information of potential freshwater fish species disappearance for 214 river basins worldwide. Below we discuss the uncertainties related to our calculations and provide the implications of our study.

1.2.1 Fate factors

The estimation of river volumes, based on the average river discharge and the average water residence time in river, affects both the fate factors for water consumption and greenhouse gas emissions. We assumed as a first approximation that the average river discharge was half of the discharge at the river mouth and that the average travel time was half of the total length of river. Furthermore, integration of data from multiple data sources in the water volume calculation of the rivers will lower the degree of data consistency. A complete data for worldwide river characteristics is however, not available. Therefore, we had to combine heterogeneous data sources for deriving river volumes (see Annex, Section 9.1).

Second, an uncertainty specifically related to the calculation of fate factors for global warming, is the arbitrary selection of a 100-year time horizon. For a number of greenhouse gases, particularly with a relative long lifetime in the atmosphere such as SF6, the results are sensitive to the choice of time horizon (De Schryver et al. 2009, Levasseur et al. 2010). For instance, the characterization factor of SF6 will increase with about 2 orders of magnitude if an infinite time horizon is chosen instead.

Finally, we excluded in our global warming calculations the indirect influence of ozone depleting chemicals, such as cholorofluorocarbons and halons, on radiative forcing. The indirect effects of ozone depleting chemicals can result in net negative radiative forcing and therefore negative fate factors (De Schryver et al. 2009, Brakkee et al. 2008).

1.2.2 Effect factors

A number of uncertainties are also related to the effect factor calculations of water consumption and global warming. First, due to recent geological glaciation, we had to exclude river basins in the effect factor calculations that are located at the latitude higher than 42^o. Applying the current species-discharge curve would lead to overestimation of effect factors for water consumption and global warming in these rivers, as the rivers above 42^o have much fewer species per unit discharge. In order to consider river basins above 42^o, a specific species-discharge curve need to be built for these river basins. For global warming we conducted a sensitivity analysis by including other river basins (> 42^o) as well in the calculation of the characterization factors. As shown in the Annex (Section 9.1), including all river basins (297 river basins in total) in the calculation of the characterization factors for global warming increases the effect factor by 1.5%. This uncertainty is considered low compared to the uncertainties in the calculation from emission to global mean temperature increase (see De Schryver et al. 2009).

Second, we used a global fish species-discharge model as opposed to basin-specific fish species-discharge curves which may be more accurate (Xenopoulos et al. 2005). However, global data sets of fish species are often not available to build watershed-specific species-discharge models.

Third, the modification of the flow regime at a range of spatial scales that affects fish species may also affect the associations between aquatic macroinvertebrates and their habitat (Bunn and Arthington 2002, Dewson et al. 2007, Poff and Zimmerman 2010). However, other aquatic freshwater taxonomic groups could not be included in this study because of insufficient data on the global scale. This implies that our characterization factors do not fully represent all the lotic aquatic ecosystems.

Fourth, the influence from building dams and abstractions was not considered in the study (see Xenopoulos et al. 2005). The absence of dams allowed us to model more accurate species-discharge curves without any human influences, as dams are known to reduce the average downstream river discharge (Rosenberg et al. 2000, Magilligan and Nislow 2005). In future research, the species-discharge curve as employed in this paper, could also be used to provide river-specific characterization factors for the construction of dams to produce hydropower.

Fifth, we estimated the river basin specific dQ/dTEMP for global warming based on five future scenarios. Uncertainty in the calculation of dQ/dTEMP is associated with the future scenario chosen. Future climate change projection is difficult and uncertain to define because changes in the future economic growth, technology and policy-making processes concerning human actions are unknown (Trenberth et al. 2000). In the present study, the dQ/dTEMP can be a factor of 2 higher or lower, depending on the scenario chosen. This uncertainty can particularly influence the relative importance of impacts of greenhouse gas emissions compared to other stressors.

Finally, we compared our effect factors for global warming with effect factors reported in a previous study on direct temperature effects towards aquatic organisms (Verones et al. 2010). Our volume-weighted effect factor for the impact of climate change on fish species is typically $7 \cdot 10^{-3}$ and ranges between $3 \cdot 10^{-3}$ and $2 \cdot 10^{-2}$ $PDF \cdot {}^oC^{-1}$. This implies that an increase in global mean temperature of 1^oC would typically result in 0.7% (0.3-2%) fish species loss. Verones et al. (2010) calculated effect factors for freshwater ecosystems due to direct water temperature increase of cooling water discharge in the river Rhine. They found that the effect factor is significantly higher in summer than in winter time (5 orders of magnitude), with a yearly average effect factor of around 1% species loss per ^oC increase and a highest monthly effect factor of 4% species loss per ^oC increase. The results from Verones et al. (2010) imply that including direct temperature effects on freshwater species occurrence could significantly increase the characterization factors for greenhouse gas emissions. The river basin specific information, required to calculate the effect factors according to Verones et al. (2010) in a meaningful way, is, however, currently not available. For generalization, river-specific data for the ambient water temperature over the seasons, key river characteristics for heat exchanges and information on species pools, based on the susceptibility of species in different climatic zones, should be gathered.

1.2.3 Implications

We developed regionalized characterization factors for water consumption and generic characterization factors for global warming related to freshwater ecosystem impacts on the global scale. Regionalized inventory data of water consumption is required to apply the new characterization factors in practice. With this information, comparison between the new characterization factors of water consumption and greenhouse gas emissions with other stressors for freshwater biodiversity are now possible.

Acknowledgements. We thank An de Schryver for her useful suggestions in the preparation of normalisation data for water consumption. Marlia Hanafiah was financed by the Ministry of Higher Education Malaysia and the National University of Malaysia.

Annex (Section 9.1): Information on the river volume estimation, derivation of $dQ_{mouth,i}/dTEMP$, summary of the five global climate scenarios (Table 9.1.1), influence of including river basins located above 42^o, normalization factors for water consumption and global warming, river characteristics data – below 42^o (Table 9.1.2), river characteristic data – above 42^o (Table 9.1.3), characterization factors and normalization factors for water consumption (Table 9.1.4) and characterization factors and normalization factors for global warming (Table 9.1.5).

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2. **Surface water use – human health impacts**

Stephan Pfister[,§,¥], Stefanie Hellweg[§]*

[§]ETH Zurich, Institute of Environmental Engineering, 8093 Zurich, Switzerland
[¥]Bren School, UC Santa Barbara, Santa Barbara, Ca. 93106-5131; USA

2.1. Introduction

2.1.1. Characterization Factors

Characterization factors for impact on human health are based on a cause-effect chain as depicted in Equation 2.1 and Figure 2.1 (both adopted from Pfister et al. 2009): the first step includes a fate and exposure factor, the water deprivation factor (WDF), accounting for water deprivation per water consumption ("fate" of the lack of water) and water use share of the analyzed human activity (exposure), i.e. agriculture. In the second step the effect factor relates malnutrition to the water deprivation and in the third step, the damage factor provides DALY from malnutrition per case of undernourished person.

[illegible] (2.1)

where $CF_{malnutrition,i}$ [DALY/$m^3_{consumed}$] is the expected specific damage per unit of water consumed (as specified in the LCI-phase). The water deprivation factor WDF_i [$m^3_{deprived}$/$m^3_{consumed}$] consists of the physical water stress index WSI_i (Figure X) and the fraction of agricultural water use $WU_{\%,agriculture,i}$ which was calculated for each watershed i. The effect factor EF_i quantifies the annual number of malnourished people per water quantity deprived [capita•yr/$m^3_{deprived}$]. It incorporates the per-capita water requirements $WR_{malnutrition}$ to prevent malnutrition [m^3/(yr•capita)] and the human development factor $HDF_{malnutrition,i}$ (Figure 2.1) which relates the human development index (HDI) to malnutrition vulnerability. The damage factor $DF_{malnutrition}$ denotes the damage caused by malnutrition [DALY/(yr•capita)]. $WR_{malnutrition}$ and $DF_{malnutrition}$ are independent of location.

Due to the intrinsic spatial dependence of water impacts caused by natural and socioeconomic factors, the uncertainties are large, but difficult to quantify.

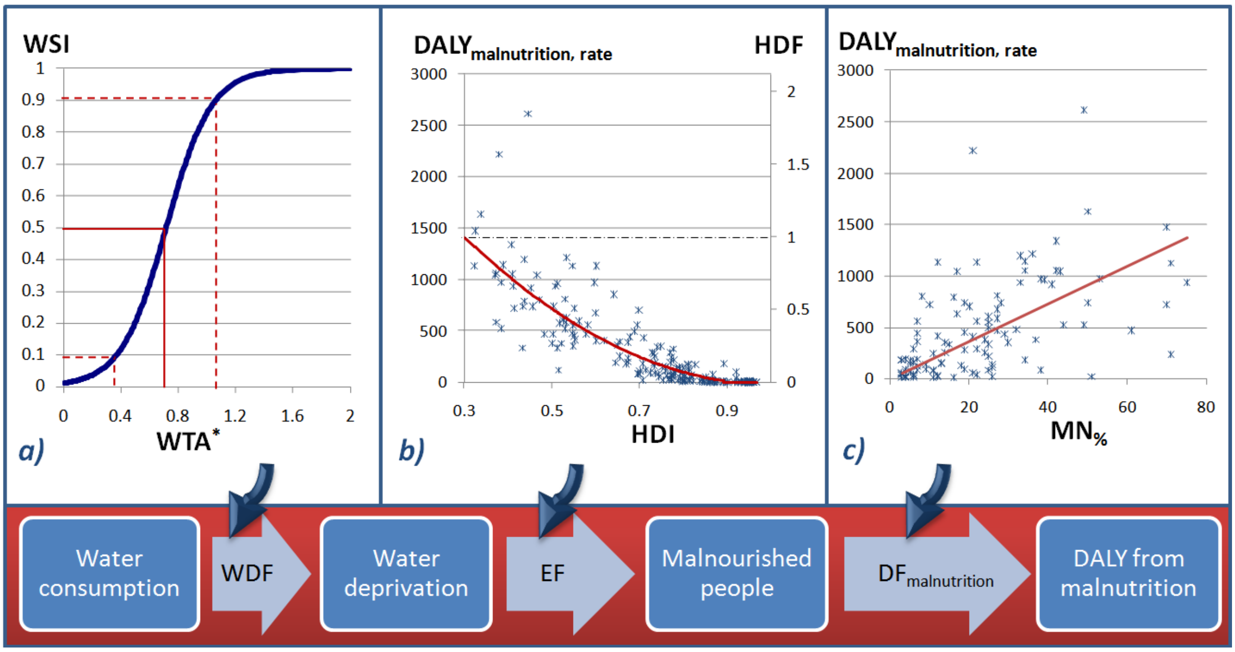

Figure 2.1. Cause effect chain for human health impacts caused by water consumption (adopted from Pfister et al. 2009).

2.1.2. Concept

In current life cycle impact assessment (LCIA), human health impacts due to lack of water consumption have only been addressed by two methods (Pfister et al. 2009 and Motoshita et al. 2010) and the robustness of the results is low, as they rely on many uncertain input parameters. Both existing methods assess impacts as DALY due to lack of water for human use. However the two existing methods are completely different in terms of modeling approach as well as impact pathway considered.

The method of Pfister et al. (2009) was used as a starting point as it considers the impact of water consumption, while the method of Motoshita et al. (2010) starts from lack of water at the household and is not linked to an inventory flow. As we found no data to relate water consumption to lack of water at the household on a global scale, the latter method was not used.

2.1.3. Spatial scope and resolution

The method was applied to roughly 11'000 watersheds with global coverage. The uncertainty analysis deals especially with the spatial dimension and conversion from watershed to country resolution.

2.2. Model and parameter uncertainties

We estimated model uncertainties based on reassessing original supplemental material from Pfister et al. (2009). Parameter uncertainties were estimated based on analysis of original data sources and, where applicable, on quantitative assessment of the model uncertainty of the data provided by third parties. The uncertainties estimated are presented in Table 2.1 and additional information is provided in the Annex (Section 9.2) of this report.

Table 2.1. Estimation of k-values used to describe the 95% confidence interval according to Slob (1994). We separated uncertainties coming from the model function (column Function) and the input parameters (column Parameters). Further details are provided in the Annex (Section 9.2).

Impact assessment model step	Function	Parameters	k-value / Uncertainty function	Main source
Withdrawal to availability ratio (WTA)				
		Availability	GIS model (See Annex)	Based on Fekete et al. (2004)
		Withdrawals	HDI function (See Annex)	Based on Alcamo et al. (2003)
Water Stress Index (WSI)				
		VF-exponent	Binominal distribution (80/20%, see Annex)	Assumption of data accuracy
	WTA* function		VF (See Annex)	Precipitation distribution analysis
	WSI function		1.7	Assumption considering the logistic function
Damage Assessment				
		Agricultural water use share	HDI function (See Annex)	Based on Alcamo et al. (2003)
		HDI values	1.7-0.55 • HDI	Assumption based on HDI concept
	HDF function		4.83 (See Annex)	From analysis of HDF function
		Water requirements	3.0	Assumption based on correlation analysis
	Damage per case relation		2.0 (See Annex)	Based on continental damage reports (WHO 2007)

2.3. Error propagation

The uncertainties were propagated within the cause-effect impact model by applying the stochastic procedure of Latin Hypercube (approach similar to Monte-Carlo simulation) with the software @Risk. We modeled it for 1000 individual runs. This was reasonable, as Latin Hypercube is more efficient than Monte-Carlo analysis.

The k-value (dispersion factor) of the model results for each watershed was derived according to Slob (1994) as the root of the ratio between the 97.5% and the 2.5% percentile (assuming a log-normal distribution). However, according to Slob (1994), the dispersion factor might also be used to quantify uncertainties and propagate errors if there is no evidence for log-normal distribution, provided that it is a multiplicative model. When the 2.5% result was zero, the k-value was estimated based on the average of the ratio of 97.5% percentile over the median and the k estimation based on the coefficient of variation CV of the results of the 1000 runs:

$$k_P = \exp(\ln(CV^2 + 1)^{0.5})^{1.96} \quad (2.2)$$

In some cases even the median was zero and the k-value was estimated based on the ratio of the 97.5% interval and the mean to have a proxy.

We analyzed the k-value for the water deprivation factor (WDF; midpoint level) and the final characterization factor (CF_{HH}; endpoint level) for each watershed with reported water withdrawal (6693 out of the 11050 watershed available in the model), covering most parts of the world.

2.4. Results

The average k-factor of the midpoint level (k_{WDF}) was found equal to 2.76 and the average k on endpoint was 18.1 (K_{CF}) with a large range (Table 2.2, Figures 2.2 and 2.3).

The aggregation from watershed to country level resulted in an average uncertainty of k=19.2 for the CF_{HH} and represents the variability of watershed factors within the country (for country information see Annex). These aggregation uncertainties are shown in Figures 2.4 and 2.5.

For each country we also calculated the total uncertainty due to watershed CF_{HH} uncertainties and aggregation (Figure 2.6). To achieve this, we propagated the average k-value within the watersheds with the k-value from the aggregation, applying the Latin Hypercube stochastic model with 1000 runs as well as analytical error propagation by first order Taylor series expansion according to MacLeod et al. (2002):

$$k_A = \exp(\ln(k_P)^2 \cdot S_P{}^2)^{0.5} \quad (2.3)$$

The latter was used when stochastic modeling gave no value. This approach seemed reasonable, as the results of the two approaches turned out to be consistent as depicted in Figure 2.7.

Table 2.2. The average k-factor of the midpoint level (k_{WDF}) results 2.76 and the average k on endpoint is 18.1 (K_{CF}).

	k_{WDF}	K_{CF}
Average	2.76	18.14
Min	1.68	2.07
Max	12.20	571.22

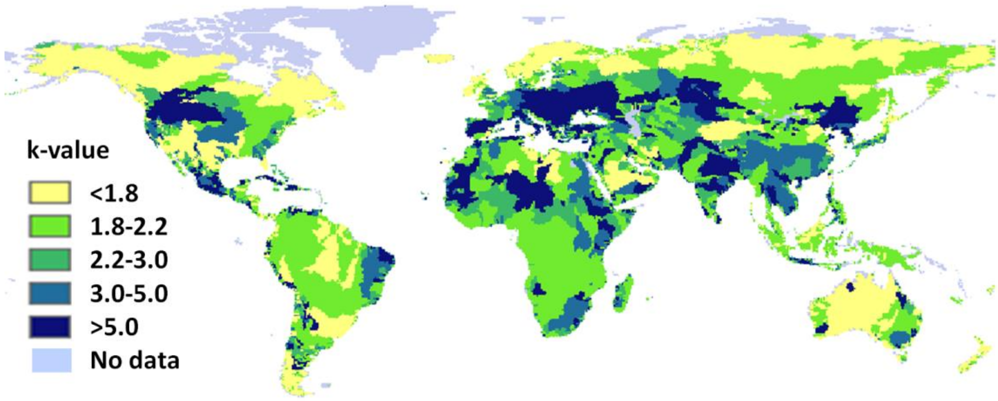

Figure 2.2. k-value of midpoint impact factors (WDI) on watershed level.

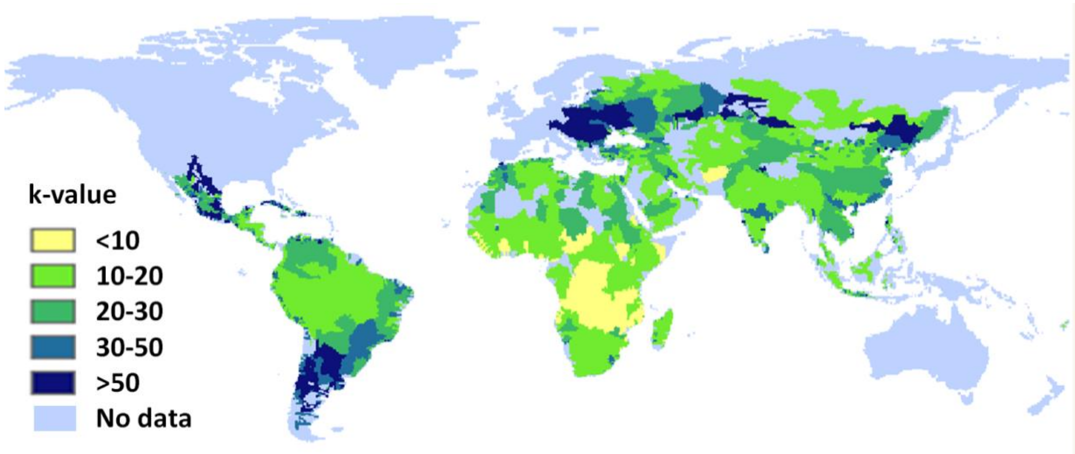

Figure 2.3. k-value of endpoint characterization factors (CF_{HH}) on watershed level.

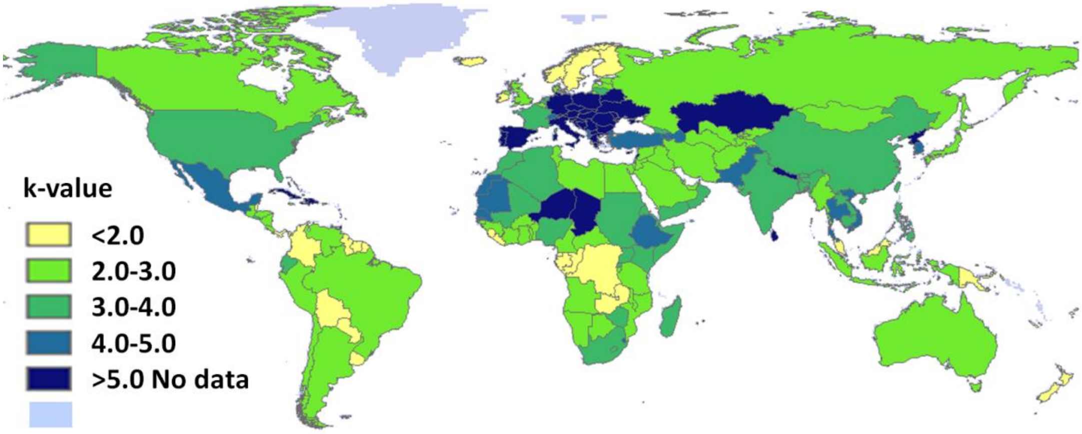

Figure 2.4. k-value caused by the aggregation of watershed to country resolution for WDF (midpoint).

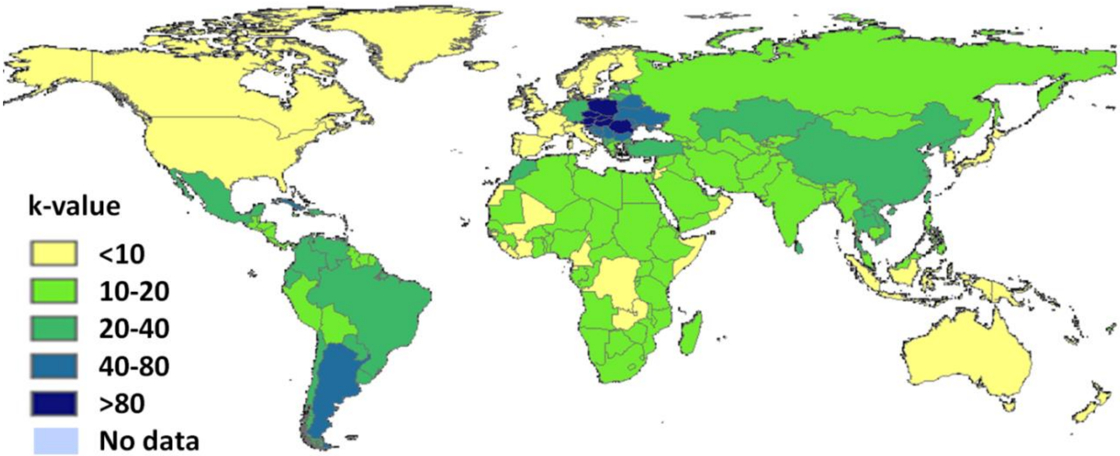

Figure 2.5. k-value caused by the aggregation of watershed to country resolution for CF_{HH} (endpoint).

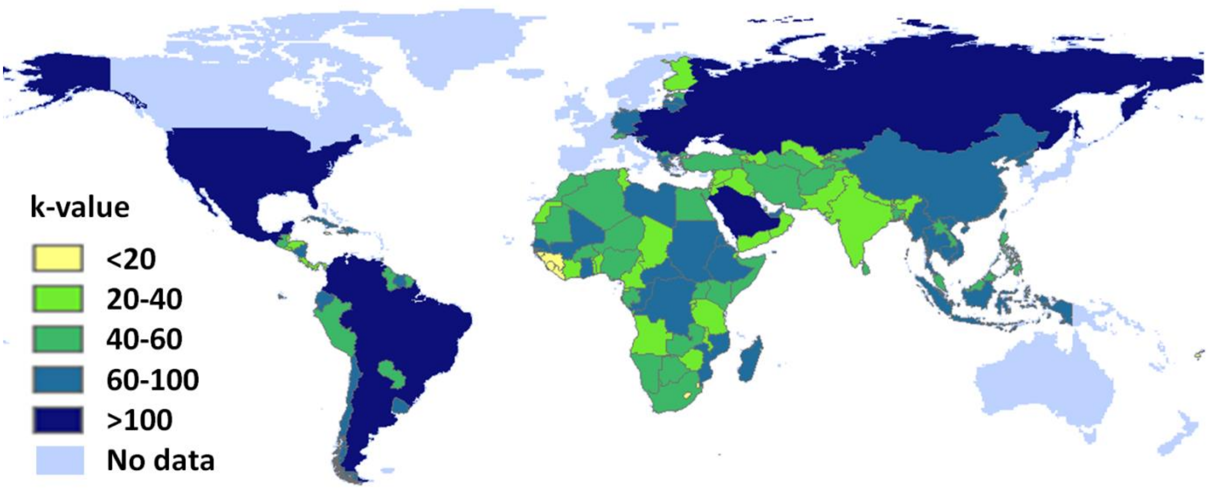

Figure 2.6. Total endpoint impact uncertainty on country resolution combining uncertainties of watershed model and due to aggregation.

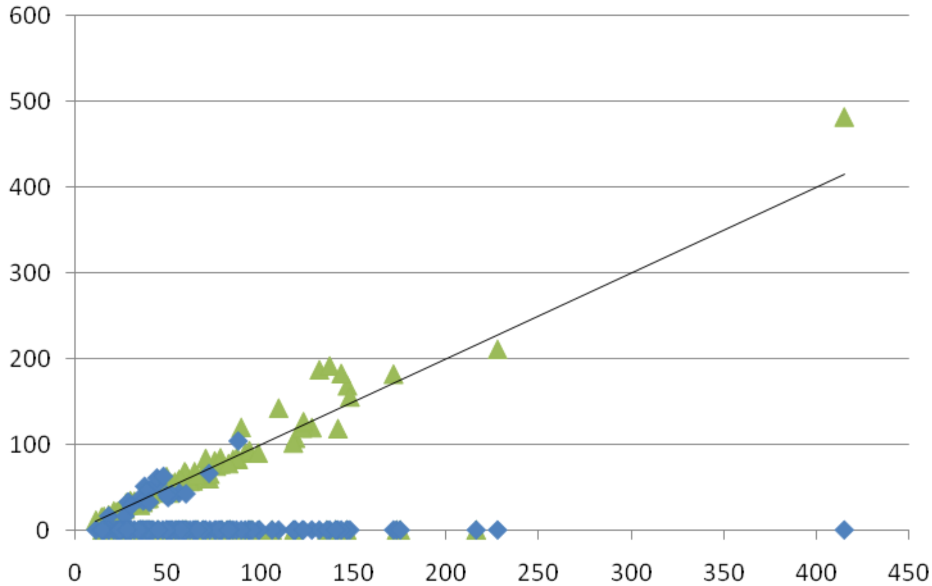

Figure 2.7. Comparison of the total k-values of country CF_{HH} (endpoint), resulting from Taylor series expansion (x-axis) and from stochastic error propagation (Latin Hypercube, y-axis) based on the 95% value to calculate the k-values (blue diamonds) and the 67% interval to include regions with 2.5% percentiles being 0 (green triangles) .

2.5. Discussion

The results show high spatial diversity of k-values and make it difficult to derive generic estimates, especially regarding the endpoint characterization factor (Table 2.2). While the k-values of midpoint assessment (water deprivation) are generally rather robust with 95% confidence intervals of one order of magnitude in most cases, the endpoint factors for DALYs feature extreme uncertainty, in some cases stretching over 4 orders of magnitudes. Aggregation on country level also adds significantly to the uncertainty. While in some countries, like Russia, the model uncertainty dominates the characterization factor, in other cases, such as India, it adds considerably to the country CF_{HH} uncertainty. As a result, it can be concluded that such aggregation of impact factors should be carefully considered, and that a higher level of spatial detail, if possible, as well as uncertainties turn out to be relevant in the overall result.

The application of k-values ideally requires a log-normal distribution, which is not always provided. In general, both WDF (midpoint) and CF_{HH} (endpoint) factors are skew continuous distributions principally matching the shape of a log-normal distribution. Testing the CF_{HH} for best distributions fits results in a log-normal or inverse Gaussian distribution for the most important watersheds. Moreover, figure 2.7 supports the assumption of a multiplicative model, as required for applying dispersion factors (Slob 1994).

Limitations of the analysis include the uncertainty of the model choice and the principal mechanism of the cause-effect chain. Such assessment might be added once other methods addressing the same issue are available. Handling such uncertainties also depends on how choices are dealt with; this discussion is yet to be held in the LC Impact group and a decision about how to deal with these is yet to be taken.

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3. Implementing groundwater extraction in life cycle impact assessment: characterization factors based on plant species richness

Rosalie van Zelm*†, Aafke M. Schipper†, Michiel Rombouts†, Judith Snepvangers‡, and Mark A.J. Huijbregts†

† Department of Environmental Science, Institute for Water and Wetland Research, Radboud University Nijmegen, P.O. Box 9010, 6500 GL, Nijmegen, the Netherlands

‡ Deltares, Soil and Groundwater Systems, P.O. Box 85467, 3508 AL, Utrecht, the Netherlands

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3.1. Introduction

Groundwater accounts for more than 98% of available freshwater resources. Approximately one-fifth of the total amount of water used for drinking purposes, for industrial cooling, for agricultural purposes, or as process water comes from groundwater (Anderson 2007). Excessive groundwater withdrawal results in a lowering of the groundwater level, causing phreatophytic stress for both natural and agricultural vegetation (UNEP 2007). This, in turn, may have a significant impact on the number of terrestrial plant species that could occur within the vegetation communities affected (Hellegers et al. 2001; Elmore et al. 2006; Hancock et al. 2005; Latour and Reiling 1994).

Until recently, an operational method to evaluate the environmental impacts associated with water use was lacking in Life Cycle Assessment (LCA). Therefore most case studies left out water use as an impact category, even if water withdrawal was identified as a large inventory flow (e.g. Humbert et al. 2009; Koroneos et al. 2005). If water use was incorporated in the impact assessment, it was usually addressed by simply taking the inventory data, i.e. the total amount of water used (e.g. Goedkoop et al. 2009; Peters et al. 2010).

Recently, efforts have been made to incorporate water use in LCA, firstly by means of reviewing possibilities and setting up frameworks (Owens 2001; Bayart et al. 2010; Stewart and Weidema 2005; Berger and Finkbeiner 2010). Milà i Canals et al. (Milà i Canals et al. 2009) provide a midpoint approach relating water use to the availability of freshwater resources for further human use after 'reserving' the necessary resource for ecosystems (water stress indicator). Van Ek et al. (2002) investigated various hydrological models and a groundwater level-effect curve to predict the change in nature-value as an effect of desiccation due to groundwater extraction. Specific characterization factors were, however, not provided. Pfister et al. (2009) introduced a method to address effects of freshwater consumption on biodiversity, expressed as the vulnerability of vascular plant species, and calculated impact indicators to be used in life cycle impact assessment (LCIA). They assumed that any water that is used can directly be replaced by precipitation, disregarding dynamic soil interaction processes. Furthermore they used the net primary production which is limited by water availability as an indicator for ecosystem quality, and related this to the potentially disappeared fraction of species (PDF).

The aim of the current study is to develop a method to address the effects of groundwater extraction on the species richness of terrestrial vegetation in an LCIA context. Characterization factors, expressing the change in potentially not occurring fraction of plant species (PNOF) due to a change in extraction of groundwater, are derived with the intention to be incorporated in LCA. We apply a method comparable to the one applied by Van Zelm et al. (2007) for acidification, where forest plant species loss was determined by coupling a fate model with multiple regression equations that predict plant species occurrence. In the context of groundwater extraction, the fate model, applicable for the Netherlands, deals with the lowering of the average groundwater level per unit of groundwater extraction, and includes processes such as precipitation, evapotranspiration, and soil permeability. Plant species richness is linked to the lowering of the groundwater table by means of a response curve based on the occurrence of 625 plant species in relation to various abiotic variables, including soil moisture content, in the Netherlands. To assess the applicability of the characterization factor derived, we determine the contribution of groundwater extraction to the total terrestrial ecosystem damage resulting from tap water production.

3.2. Methods

3.2.1. Characterization factor

The characterization factor for groundwater extraction ($CF_{T,GW}$ in m2•yr/m3) in the Netherlands is defined as the change in the number of plant species due to a change in extraction of groundwater over a certain area. The $CF_{T,GW}$ consists of a fate factor (FF in m3•yr/m3) and an effect factor (EF in 1/m). To account for spatial variation in FF and EF, a spatially explicit grid-based approach was followed whereby FF and EF were multiplied per grid cell and then summed over all grid cells i:

$$CF = \sum_i FF_i \cdot EF_i \tag{3.1}$$

3.2.2. Fate factor

The fate factor, describing the drawdown in relation to the change in groundwater extraction, expresses the time that is needed for groundwater replenishment. The fate factor was determined with the National Hydrological Instrumentation (NHI), which is a national hydrological model for the Netherlands developed by the Dutch Institute for Applied Natural Science Research TNO (Snepvangers et al. 2008). With a resolution of 250x250m, NHI covers 95% of the country, excluding the islands in the north and the southernmost part (see Annex, Section 9.3). Grid-specific partial fate factors (FF_i in years) were calculated as follows

$$FF = \frac{\sum_i A_i \cdot \Delta AG_i}{\Delta q} \tag{3.2}$$

where Ai is the area of grid cell i (m2), ΔAGi is the change in yearly average groundwater level in grid cell i (m), and Δq is the change in extraction rate set at 1% increase of the current extraction rate (m3/year).

For saturated zone calculations, NHI uses the United States Geological Survey's MODFLOW code (McDonald and Harbaugh 1988; Facchi et al. 2004; Gedeon et al. 2007). A schematic representation of the NHI groundwater module is shown in Figure 3.1. The geohydrological structure is defined by an impervious basis underlying four aquifers separated by three semi-pervious layers. The horizontal flow through the aquifers depends on the transmissivity (kD in m2/day) of the corresponding layer and the vertical flow through the semi-pervious layers depends on the vertical resistance (c in days) of the corresponding layer. The NHI describes the groundwater regime in the Netherlands, as surveyed in the year 2000. River interaction is included by a total drainage flux per junction. Anisotropies and sheet pilings are included as well, by indicating place and amount of barriers (Snepvangers et al. 2008). A constant recharge value was used, representing the net recharge from precipitation and evapotranspiration. Groundwater extraction was parameterized with average extraction data for the year 2000 for each of the 872 major groundwater wells in the Netherlands, with extraction depths of up to ca. 300 m. Yearly average groundwater levels were modelled by running MODFLOW to a steady-state. The location of each major well in the Netherlands is shown in the Annex (Section 9.3).

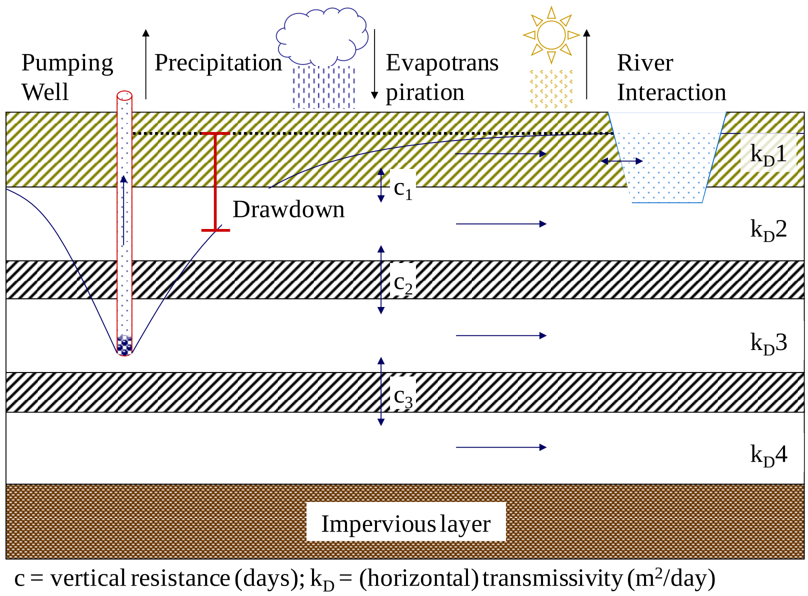

Figure 3.1. *Simplified representation of the NHI saturated zone model.*

3.2.3. Effect factor

The effect factor in grid cell i (1/m) describes the change in potentially not occurring fraction of plant species (PNOF) due to a change in AG:

$$EF_i = \frac{dPNOF_i}{dAG_i} \tag{3.3}$$

The effect factor was determined with groundwater level response functions, following the procedure outlined by Van Zelm et al. (2007). The PNOF was derived from the probability of occurrence of individual plant species (P_s). Statistical model MOVE was applied to predict the occurrence of plant species with a range of environmental parameters as input (Bakkenes et al. 2002). As measurements on abiotic parameters are scarce, MOVE uses Ellenberg indicator values of plant species to assess environmental conditions (Bakkenes et al. 2002). Ellenberg (1979) summarized the ecology of the Central-European vascular plants by assigning to each species indicator values for environmental variables, such as moisture, salt, nitrogen, and acidity. Site conditions in MOVE are determined as the average of the Ellenberg indicator values of all species present at a site. Multiple regression equations are used to express the occurrence probability of individual species as a function of the site-specific average Ellenberg values:

$$\ln\left(\frac{P_s}{1-P_s}\right) = b_0 + (b_1 \cdot n + b_2 \cdot n^2) + (b_3 \cdot f + b_4 \cdot f^2) + (b_5 \cdot r + b_6 \cdot r^2) + (b_7 \cdot s) + (b_8 \cdot tox) + (b_9 \cdot PGR) + (b_{10} \cdot VEG) + (b_{11} \cdot r \cdot n) + (b_{12} \cdot r \cdot f) + (b_{13} \cdot n \cdot f) \tag{3.4}$$

where n, f, r and s, are Ellenberg values describing nitrogen-, moisture-, acid-, and salt-content, tox is the potentially affected fraction of plants due to heavy metals, and PGR and VEG describe the influence of the physical-geographical region, and the vegetation type, respectively. The last three terms in Equation 3.4 describe the interactions between r, n, and f. Finally, b_0 to b_{13} are regression coefficients (De Heer et al. 2000).

Equation 3.4 was simplified in order to relate species occurrence P_s specifically to the moisture indicator f:

$$\ln\left(\frac{P_s}{1-P_s}\right) = a_s + b_s \cdot f + c_s \cdot f^2 \tag{3.5}$$

where as describes the situation of all environmental variables except f, relevant for plant species s, and b_s and c_s are species specific regression constants related to f.

Within the MOVE model κ-values are provided, which express the probability of occurrence related to the model predictors. When $P_s > κ$ a plant species is assumed to be present, and when $P_s < κ$ a plant species is assumed not to occur (Fielding and Bell 1997). The κ -values were used to predict the occurrence of 625 terrestrial plant species (see Annex (Section 9.3). In order to determine whether a plant species could occur at a specific f (Eq. 5), variability in the other site conditions had to be accounted for. By varying r, n, s, tox, PGR, and VEG, Equation 3.5 was parameterized 500 times for each plant species at each f. If at least one of the realizations yielded Ps > κ, it was assumed that the plant species could occur at that f. The site conditions were varied according to measurement data in the MOVE model, with r values between 4 and 8; n between 3 and 7; s between 0 and 3; and tox between 0 and 0.4. These numbers correspond with pH between 3 and 9, N stock of 2 to 500 kg/ha/yr, chloride concentrations between 3 and 10,000 mg/L, and a potentially affected fraction of plants due to heavy metals between 0 and 0.4 (Ertsen et al. 1998; Bakkenes et al. 2002; Alkemade et al. 1996). The physical-geographical regions (PGR) included were North Sea area, tidal area, closed estuaries, rivers, hills, urban area, sea clay, peat, higher sand grounds north, higher sand grounds south, and dunes. The vegetation types (VEG) included were nutrient-poor grassland (low herbaceous vegetation), pine forest, spruce forest, deciduous forest, and heath. A region-vegetation combination was judged to be likely, and therefore taken into account, when at least 100 records were available in MOVE (Bakkenes et al. 2002). The resulting 27 combinations are provided in the Annex (Section 9.3). Subsequently, a groundwater level-response curve was obtained, based on the potentially not occurring fraction of plant species (PNOF) at each f value:

$$PNOF_f = 1 - POF_f \tag{3.6}$$

with

$$POF_f = \frac{N_f}{N_{\max}} \tag{3.7}$$

where POF_f represents the potentially occurring fraction of plant species at a certain f, Nf is the number of species that can occur at a certain f, taking into account varying r, n, s, tox, PGR, and VEG, and N_{max} is the maximum number of co-occurring species within the range of moisture values. N_{max} is lower than the total number of species (N_{tot}), because interspecific variation in moisture requirements prevents the co-occurrence of all plant species at a single f. We do not consider N_{tot} but rather N_{max} as background situation (zero stress, independent of groundwater level).

To ensure an appropriate connection between the fate factor and the effect factor, the Ellenberg value f was linked to average groundwater level (AG) with the regression found by Schaffers and Sýkora (2000):

$$AG = -2.55 + 0.26 \bullet f \quad (3.8)$$

The derivative at each point of the response curve, showing the PNOF in relation to AG, represents the effect factor at each AG. Average groundwater levels AG_i were calculated with NHI and effect factors could then be allocated to each grid cell i. Groundwater level-response curves were created based on all plant species (n = 625) and for the species that are on the red list in the Netherlands (n = 141; (Dutch ministry of Agriculture and Nature and Food Quality Accessed dd September 25, 2010)). This red list is based on the IUCN criteria. A full species list is provided in the Annex (Section 9.3).

3.2.4. Cultural perspectives

To handle value choices in the modeling procedure in a consistent way, we applied the cultural perspective theory (Hofstetter 1998; Thompson et al. 1990). Three cultural perspectives, i.e. individualist, hierarchist and egalitarian were used. The individualist coincides with the view that mankind has a high adaptive capacity through technological and economic development and that a short time perspective is justified. The egalitarian coincides with the view that nature is fragile, with many factors to damage it, that a long time perspective is justified, and a worst case scenario is needed (the precautionary principle). The hierarchist perspective coincides with the view that impacts can be avoided with proper management, and that the choice on what to include is based on the existence of evidence. Table 3.1 provides an overview of the value choices that can be included within groundwater modeling.

Time perspective can be applied by considering effects within a certain time horizon, emphasizing long term or short-term processes. In general time horizons of 20, 100 and infinite years are applied for the individualist, hierarchist, and egalitarian respectively (Goedkoop et al. 2009). As no delay of over 10 years is expected in the lowering of the groundwater table due to extractions (Snepvangers et al. 2008), time horizons are not included in the perspectives.

An assumption regarding ecosystem damage is the inclusion of species. For the individualist and hierarchist perspectives, all plant species were assumed equally important. For the egalitarian perspective high importance was given to species that are already threatened in their existence and therefore red list species were included only.

The individualist is risk seeking, the hierarchist accepts a high level of risk as long as the decision is made by experts, and the egalitarian perspective is risk adverse (Thompson et al. 1990). Based on these attitudes towards risks, the individualist perspective only includes empirically proven effects. The hierarchist perspective includes scientifically accepted effects, while the egalitarian perspective includes all potential effects that may occur.

Potential positive effects were included for the individualist perspective as they have a positive attitude towards environmental benefits (Hofstetter 1998), and if they are not uncertain for the hierarchist as well.

Table 3.1. Value choices for groundwater extraction for three different perspectives

Value choice	Individualist	Hierarchist	Egalitarian
Time Horizon	-	-	-
Species protection level	All	All	Red list
Likelihood of effects	Proven effects	Likely effects	All known effects
Positive effects	Yes	Yes	No

3.2.5. LCA application

To assess the applicability of the characterization factors for groundwater extraction, we calculated the relative contribution of groundwater extraction compared to other terrestrial ecosystem impact categories for tap water production. Inventory data were taken from the ecoinvent database v2.2 (Ecoinvent Centre 2010) and characterization factors for land use, ecotoxicity, acidification, and climate change were applied according to the individualist, hierarchist, and egalitarian perspectives of the ReCiPe method (Goedkoop et al. 2009).

3.3. Results

The partial fate factors over the Netherlands range from $-1.2\cdot10^{-5}$ to $2.7\cdot10^{-2}$ yr and are shown in Figure 3.2.

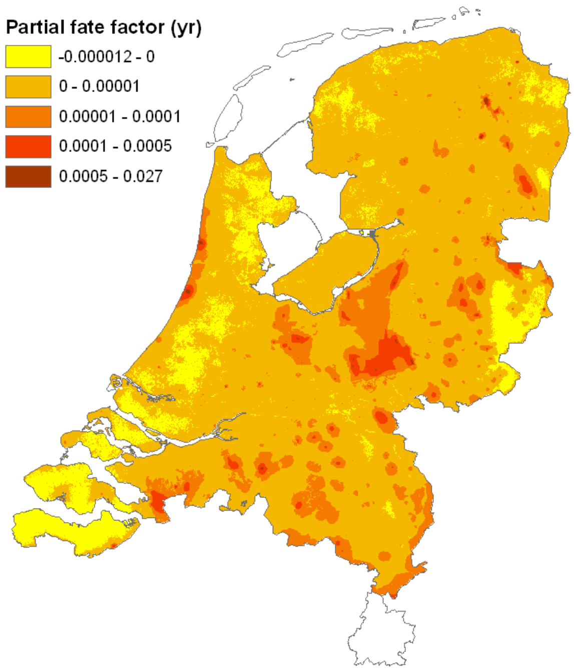

Figure 3.2. Partial fate factors (yr) for the Netherlands.

Figure 3.3a shows the groundwater level response curve, depicting the PNOF at various AGs, for the Netherlands. From AG of -2.30 m up to -1.25 m the PNOF decreases as the groundwater level increases. In the shallower groundwater range the PNOF increases when the groundwater level increases. The groundwater level response curve was divided in four parts and for each an effect factor ($\frac{dPNOF}{dAG}$) was calculated. EFs are 0.24 m^{-1} (-2.30 < AG < -1.98 m), 0.92 m^{-1} (-1.98 < AG < -1.25 m), -0.23 m^{-1} (-1.25 < AG < -0.83 m), and -0.85 m^{-1} (-0.83 < AG < 0 m) respectively. Figure 3.3b shows the groundwater level-response curve for the red list species only. A similar trend is observed and the curve for red list species can be divided in four parts as well. EFs are 0.25 m^{-1} (-2.30 < AG < -1.95 m), 1.18 m^{-1} (-1.95 < AG < -1.21 m), -0.05 m^{-1} (-1.21 < AG < -0.72 m), and -1.01 m^{-1} (-0.72 < AG < 0 m) respectively. For lower groundwater levels, effects on red list species are 4 to 28 % larger. Figure 3.3c shows curves for nutrient poor grassland, pine forest, deciduous forest, and heath separately. It can be seen that the variation in effect factor among vegetation types is relatively small (around a factor of 1.5).

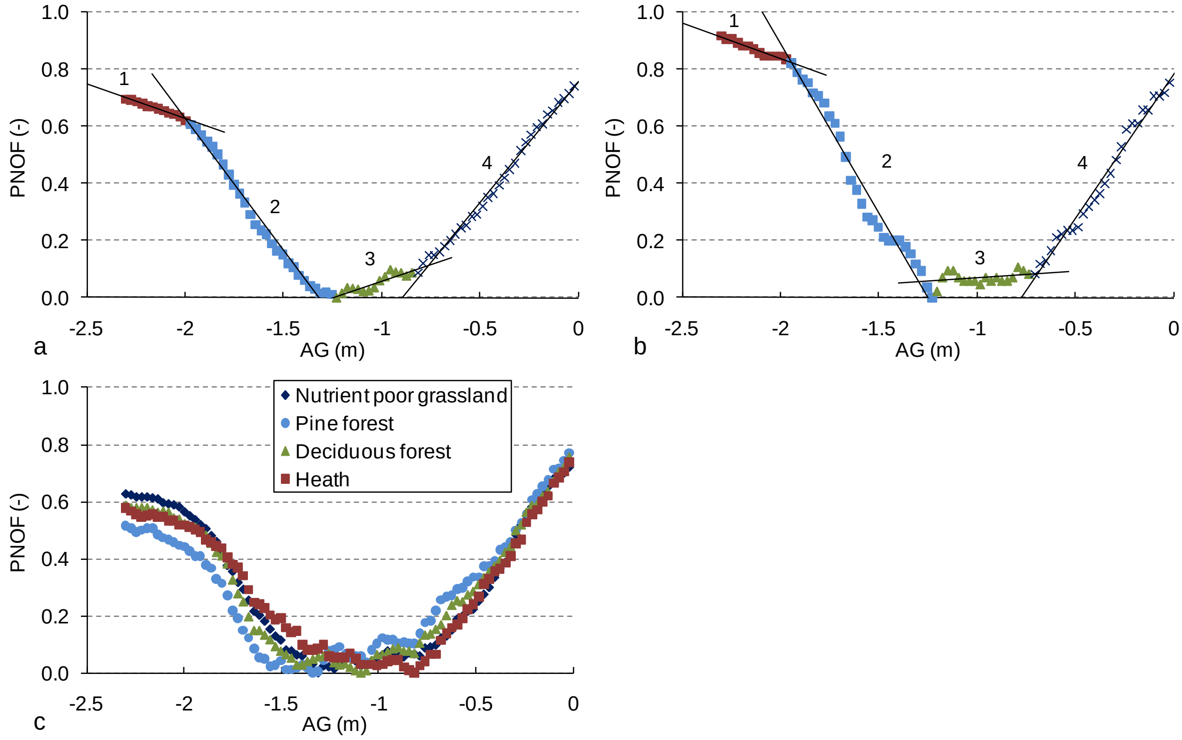

Figure 3.3. Groundwater level-response curves representing the potentially not occurring fraction of plant species (PNOF) as a function of the yearly average groundwater level (AG). (a) shows the overall curve with fitted linear functions that follow (1) PNOF = -0.24*AG + 0.14 with an explained variance R^2= 0.99; (2) PNOF = -0.92*AG - 1.21 with R^2 = 0.98; (3) PNOF = 0.23*AG + 0.29 with R^2= 0.82, (4) PNOF = 0.85*AG + 0.75 with R^2= 0.99. (b) shows the curve for 141 species that are on the red list in the Netherlands with fitted linear functions that follow (1) PNOF = -0.25*AG + 0.34 with an explained variance R^2 = 0.96; (2) PNOF = -1.18*AG - 1.48 with R^2 = 0.97; (3) PNOF = 0.05*AG + 0.11 with R^2 = 0.12, (4) PNOF = 1.01*AG + 0.78 with R^2 = 0.99. (c) shows curves per vegetation type.

The groundwater level-response curve for all species can be extrapolated from AG = -2.30 m to AG= -3.58 m. Grid cells with AGs of -2.30 to -3.58 m will then be assigned the EF for the AG-range of -2.30 m to -1.98 m. For AG < -3.58 m, the PNOF equals 1, implying that these areas do not contain groundwater-dependent vegetation. Therefore, the EF was set to

0 m^{-1} for an AG < -3.58 m. For the red list species the same extrapolation strategy was applied.

For the calculation of the characterization factor $CF_{T,GW}$, the response curve for all species is included for the individualist and the hierarchist perspective, while the egalitarian perspective takes into account the red list species only. The effects likely to occur in the groundwater level range below -2.3 m where the effect curve is extrapolated are included in the hierarchist and egalitarian perspective, but excluded from the individualist perspective due to the relatively high uncertainty of this part of the response curve. The individualistic and hierarchist perspective include positive effects, while the egalitarian perspective does not include positive effects from a precautionary point of view. Figure 3.4 shows the three $CF_{T,GW}$ s for the Netherlands.

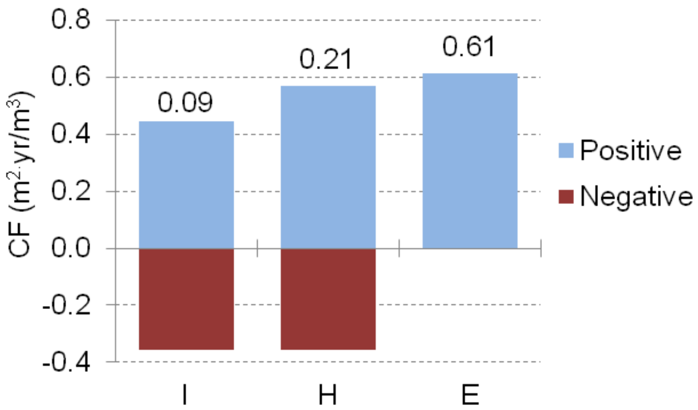

Figure 3.4. Characterization factors for the individualist (I), hierarchist (H), and egalitarian (E) perspectives, consisting of a positive and a negative part.

Application of our calculated $CF_{T,GW}$ shows that groundwater extraction causes 2.2 to 13.2% of the total ecosystem damage resulting from the production of tap water, depending on the perspective taken (Figure 3.5).

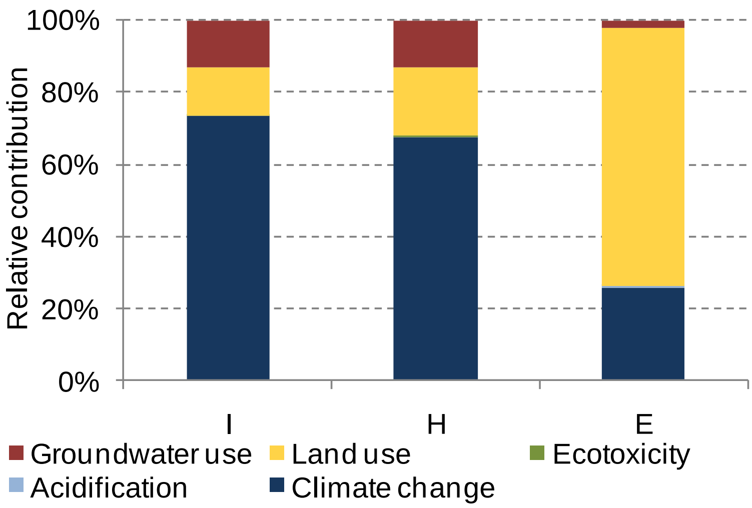

Figure 3.5. The relative contribution of five impact categories to the terrestrial ecosystem damage of tap water production following the individualist (I), hierarchist (H), and egalitarian (E) perspective.

3.4. Discussion

This paper described the development and application of a method that predicts the change in plant species richness, modelled as the potentially not occurring fraction of plant species, per unit of groundwater extraction. The characterization factor derived provides the opportunity to combine the ecological consequences of groundwater extraction with the effects of other types of stressors, such as land use and acidification, in the Life Cycle Assessment of products. Below, we discuss the benefits and limitations of the modelling procedure and provide an interpretation of the results obtained.

3.4.1. Fate factors

To obtain fate factors for groundwater extraction, the MODFLOW model was run to steady-state and yearly average changes in groundwater levels were derived. A steady-state approach seems appropriate for groundwater wells where water is being pumped constantly, thus having a permanent effect on the groundwater level. In the Netherlands, 75% of the extracted groundwater is used for drinking water (Van den Berg et al. 2000), which is extracted with continuously pumping wells (Provincie Noord-Brabant 2003). Therefore, the effects of an intermittently pumping well were not taken into account in our study. More research on the effects of intermittently pumping wells is needed in order to include these wells in LCA studies.

Current European policy aims at a sustainable use of groundwater, which would mean a decrease of groundwater extraction in the future (European Commission policy on water issues. 2010. Available at http://ec.europa.eu/environment/water/index_en.htm 2010). As a reference situation, we used the amount of extraction as it was in the year 2000. To account for possible future decreases in extraction a different reference situation can be assumed for calculating fate and effect factors. When more information is available on future scenario's, these can be included in the three perspectives as well, as future optimistic, baseline, and pessimistic views correspond to the individualist, hierarchist, and egalitarian perspective, respectively (Hofstetter 1998).

Using the ecohydrological DEMNAT model, Van Ek et al. (2002) derived a typical factor for dAG/dq of 0.02 mm lowering of the groundwater level per Mm^3/yr of extracted groundwater, whereas our total factor ($\sum_i FF_i$) was 0.14 mm per Mm^3/yr. Extractions from wells located near the borders with Germany and Belgium cause a drawdown in these countries as well. These effects are not included by the NHI, which causes a small underestimation of the full drawdown over the affected area and thus of the fate factor.

Next to regional variation caused by diverging extraction rates, the fate factor can vary due to variation in hydro-geological parameters: soil permeability, recharge, ground pack around the extraction (e.g. is it mainly clay, sand, or peat) and depth of extraction. For LCA purposes it would be interesting to derive fate factors as a function of these varying parameters to account for location-specific conditions. Our fate model provides the possibility to link grid-specific groundwater table lowering to environmental variables, such as the vertical resistance and transmissivity of the soil layers, and precipitation and evapotranspiration. Further research is required to quantify the influence of variation in hydro-geological model parameters on the fate factor.

3.4.2. Effect factor

To obtain effect factors for groundwater level change, the MOVE model was applied. The DEMNAT model also provides response curves for plant species pools, showing a decline in species diversity for dropping groundwater levels (Runhaar et al. 1997). Runhaar et al. (Runhaar et al. 1997) found a maximum of 13.5% species richness decrease per 10 cm decrease of Average Spring Groundwater level decrease which corresponds well with the maximum of 9.2% species richness decrease per 10 cm groundwater level decrease found in our research.

Laidig et al. (Laidig et al. 2010) showed that it depends on the vegetation type and species included whether there is a positive or negative relationship between species occurrence and groundwater level change, corresponding to the increase in species diversity for higher groundwater levels found in our research.

For the connection between fate and effects, we applied the relationship between Ellenberg moisture value f and average groundwater level as derived from Schaffers and Sýkora (2000). As shown by Ertsen et al. (1998), there is also a good correlation between Average Spring Groundwater level and f. The relationship between ASG and f could have been used as well, but would have required dynamic calculations with the MODFLOW model to derive fate factors related to ASG.

We showed that the effect factors for our full list of terrestrial plant species did not largely differ from the effect factors for the red list species only. The response curves showed similar trends and both curves could be divided into four parts. It was also shown that the effect factors hardly differ between different vegetation types. These findings indicate that the variation in effect factors among vegetation types occurring in a temperate maritime climate is relatively small, suggesting that our generic response curve can be used in other regions with comparable vegetation types. However, it should be stressed that our method predicts responses of species richness irrespective of species composition, as we used one generic groundwater level response curve based on the total plant species pool in the Netherlands. Specific response curves for vegetation types characteristic of, for instance, wet or dry circumstances will facilitate more location-specific assessments of the effects of groundwater extraction on plant species richness. This should be subject to further research.

The groundwater level-response curve showed that the point of departure is relevant in the derivation of the effect factor. For yearly average groundwater levels lower than -1.25 meters, a decrease in species richness is expected if groundwater levels are lowered (maximum 9.2% per 10 cm of groundwater level decrease). In contrast, for groundwater levels higher than -1.25 meters, a lowering of the groundwater level is expected to increase species richness (maximum 8.5% increase per 10 cm of groundwater level decrease). It should, however, be stressed that our work should not be used as an argument to lower groundwater levels in ecosystems where groundwater tables are naturally high. In these cases, a shift towards a different vegetation community with higher species diversity should not be automatically interpreted as beneficial, especially because the increase in species diversity might go on the expense of particular species that rely on high groundwater levels. Natural heterogeneity in landscape characteristics, including natural variability in groundwater levels, is an important driver for maintaining overall species diversity.

3.4.3. Application in LCA studies

Characterization factors were derived for the generic Dutch situation. Effect factors were based on data on the occurrence of plant species, and therefore expressed as potentially not occurring fraction of plant species (PNOF). This, in contradiction to effects caused by for example, toxic compounds, for which data are available on the effect and lethal dose for species (Rosenbaum et al. 2008). On an endpoint level, the PNOF can be considered equal to the potentially affected or potentially disappeared fraction of species.

For LCAs, the Netherlands is a relatively specific spatial context. This brings up the question whether the current research can be applied outside the Netherlands. Provided that the required geohydrological data are available, as is the case for e.g. China (Wang et al. 2007), Canada (Meriano and Eyles 2003), and Italy (Facchi et al. 2004), the U.S. Geological Survey model MODFLOW can be parameterized for every region of the world to calculate fate factors according to the method outlined in this paper. The effect factors apply to temperate maritime climates with similar vegetation types as the Netherlands. The Ellenberg numbers were based on observations of realized niches of plant species in Central Europe. As the ecological behavior of species can be different in other regions, calibration of the Ellenberg values is needed according to regional deviations. This was successfully done for several other European areas, e.g. the Faroe islands (Lawesson et al. 2003), Britain (Hill et al. 2000), Sweden (Diekmann 1995) and Greece (Boethling et al. 2002).

Our research is among the first to include the impacts of groundwater extraction on terrestrial ecosystems in LCA context. For the production of tap water we showed that groundwater extraction contributes to terrestrial ecosystem damage up to 32%. We recommend to further elaborate on the inclusion of groundwater extraction in LCA by developing $CF_{T,GW}$s for regions outside the Netherlands as well.

Acknowledgement - We thank Gerrit Hendriksen, Jarno Verkaik, and Joachim Hunink from Deltares for their help with NHI/MODFLOW modelling and fruitful discussions.

–Annex: Information on all included terrestrial plant species, their κ-values, and physical-geographical region–vegetation types included is in the Annex, Section 9.3.

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4. Quantifying area changes of internationally important wetlands due to water consumption in LCA

Francesca Verones[1], Stephan Pfister[1], Stefanie Hellweg[1]

[1] ETH Zurich, Institute of Environmental Engineering, 8093 Zurich, Switzerland

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4.1. Introduction

Wetlands are complex ecosystems, providing multiple services such as water purification, buffering of water flows, resources for human uses (e.g., food, water and medicines)(Bacon 2012), as well as habitat for many species, of which a considerable part are dependent on or linked to wetlands (Silvius et al. 2000). We define wetlands according to article 1 of the Ramsar Convention as "areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres" (Ramsar Convention 1994). It is estimated that today wetlands cover more than 1'280 million hectares of the world's surface (Millennium Ecosystem Assessment 2005) of which about 10% (134'216'253 ha, Aug 2012, excluding coastal and marine wetlands)(Ramsar Sites Information Service 2012) were designated as wetlands of international importance under the Ramsar Convention (Ramsar wetlands) since 1975. More than 50% of global wetland area was lost in the 20th century and many other wetlands are heavily degraded (Millennium Ecosystem Assessment 2005; Zedler et al. 2005). One of the main reasons for wetland loss is drainage, especially in agricultural regions (Zedler et al. 2005). More than 70% of global freshwater withdrawals are used for agriculture (World Water Assessment Programme 2009). Drainage and water diversion are both among the top threats for Ramsar wetlands (Supporting Information (SI), section 8).

Life Cycle Assessment (LCA) is an approach for quantifying the total environmental impacts potentially occurring through the life cycle of a product or process (ISO 2006). Even though several methods exist for assessing the impacts of water use on terrestrial and aquatic ecosystems, on global (e.g.,Hanafiah et al. 2011; Milà i Canals et al. 2009; Pfister et al. 2009) as well as on regional scales (e.g.,van Zelm et al. 2011), there is so far no generally applicable method which assesses the associated impacts on wetlands. Since wetlands are specialized ecosystems on which many species are dependent, it is important to develop characterization factors specifically for this kind of ecosystem.

To quantify the impact on surface water-fed wetlands, hydrological models are needed. Several such models have been developed (e.g., Alcamo et al. 2003; WATCH 2011). However, the resolution of these models is generally too coarse (e.g. WaterGAP (WATCH 2011) 0.5 arc-degree resolution) to grasp smaller wetlands.

The focus of this paper lies on changes in wetland area due to changed water consumption rates. Wetland area change is regarded as a key parameter of wetland quality and is translated into potential biodiversity loss with respect to waterbirds, non-residential birds, amphibians, reptiles and water-dependent mammals in a parallel publication (Verones et al. submitted). The Fate Factors (FF) developed in this publication describe the area change in m^2 per m^3/yr of water consumed. To quantify FF, we calculated global surface water flows at a resolution of 0.5 arc-minutes for surface water-fed wetlands. For groundwater-fed wetlands we used local hydrogeological parameters and equations from pumping wells. Wetland area changes were calculated as the product of the total amount of water consumed (provided by the Life Cycle Inventory Analysis) and FF developed here. These area changes are compatible with land use inventory flows, which are typically reported in units of $m^2 \cdot yr$ as well. They need to be combined with an effect factor to derive results on an endpoint level (Verones et al. submitted).

4.2. Methods

4.2.1. Wetland distribution and classification

The wetlands assessed here were designated Ramsar wetlands, with information available from the Ramsar Sites Information Service (RIS) (Ramsar Sites Information Service 2012). If no information was available, scientific and other literature (e.g. National park websites) were used. All wetlands, which were listed and contained data within the Annotated Ramsar List (Ramsar Convention 2012) on 17 August 2012 were included. We focused on inland wetlands and excluded wetlands containing coastal and marine wetland types (SI, S2). For water balancing, coastal wetlands pose increased difficulties since shortages of freshwater can often be compensated by seawater, compromising water quality (due to salinity) (Verbrugge et al. 2012) but not necessarily affecting volume and area.

We classified the wetlands with information from the RIS or the literature according to the main water source feeding the wetland: surface water (SW, 1033 wetlands) or groundwater (GW, 151 wetlands). We assumed that, apart from precipitation, either GW or SW is the only source for GW-fed and SW-fed wetlands, respectively. Wetlands which are mainly fed by precipitation were classified as SW-fed wetlands.

Temporary wetlands were treated like permanent wetlands. In permanent wetlands water is always present, while in temporary wetlands the exact timing of water presence differs though time and cannot be determined exactly. Once water is present a unique flora and fauna are sustained in many cases (Waterkeyn et al. 2008). For the sake of simplicity we treated them like permanent wetlands, potentially overestimating the impacts, since wetland flora and fauna are not present at all times.

4.2.2. Wetland areas

We modeled wetlands as circular cones of water (for reasoning and sensitivity, see SI S3). The area was per default the Ramsar area, since good information for this was available from the RIS (Ramsar Sites Information Service 2012). We assumed that these areas correspond to the natural areas of the wetlands. The radius and cone volume were calculated from area and water depth. We calculated for each wetland an angle α between the axis of the water depth (Figure 4.1A and B), and the slope of the embankment (SI, S3). For 72 GW-fed wetlands and 485 SW-wetlands no information for water depth was found. In these cases we assumed a depth of 0.15 m for marsh-like wetlands (USGS 2006) and 1 m for other wetlands. This is a precautious assumption since the average depth for GW-fed wetlands was 3.8 m and for SW-fed wetlands 8 m (shallow wetlands tend to lose a larger area per m^3/yr consumed).

4.2.3. Calculation of fate factors (FF)

The FF [$m^2 \cdot yr/m^3$] relates the loss of wetland area to water consumption. We define water consumption as the volume of water that is incorporated in products, evaporated, or diverted from the individual wetland's catchment even though it may remain in the same major watershed. The FF is calculated as shown in Equation 4.1, where i indicates the type of wetland (SW or GW). $A_{reported}$ [m^2] is the wetland area reported in the RIS[5] and A_{new} is the new wetland area after water consumption x_i [m^3/yr]. The term x_i corresponds to the net marginal water consumption rate affecting wetland type i. The default value suggested here is an additional consumption of 1000 m^3/yr from either SW or GW. This value is arbitrarily chosen and is changed to 1'000'000 m^3/yr for large wetlands and 10 m^3/yr for small wetlands (where 1'000'000 m^3/yr would lead to complete depletion) in the sensitivity analysis (see below). We model all wetlands as being in steady-state conditions and only take the change of water consumption x_i into account. This is acceptable since impacts are measured as $m^2 \cdot yr$ and it is not important in the context of LCA whether the damage happens immediately or distributed over time.

$$FF_i = \frac{(A_{reported,i} - A_{new,i})}{x_i}$$

Equation 4.1

All SW consumption upstream of SW-fed wetlands was considered, while for GW consumption an "area of relevance" close to GW-fed wetlands was defined (see below). For SW-wetlands, GW withdrawals were assumed to be irrelevant, making the simplifying assumption that there is no connection to groundwater. For GW-fed wetlands consumption of water from the aquifer in an "area of relevance" (AoR) surrounding the wetland (see definition below), as well as direct water abstraction from the wetland were considered. FFs were provided for every wetland individually. Figure 4.1 shows an overview for the calculation procedure of the FFs.

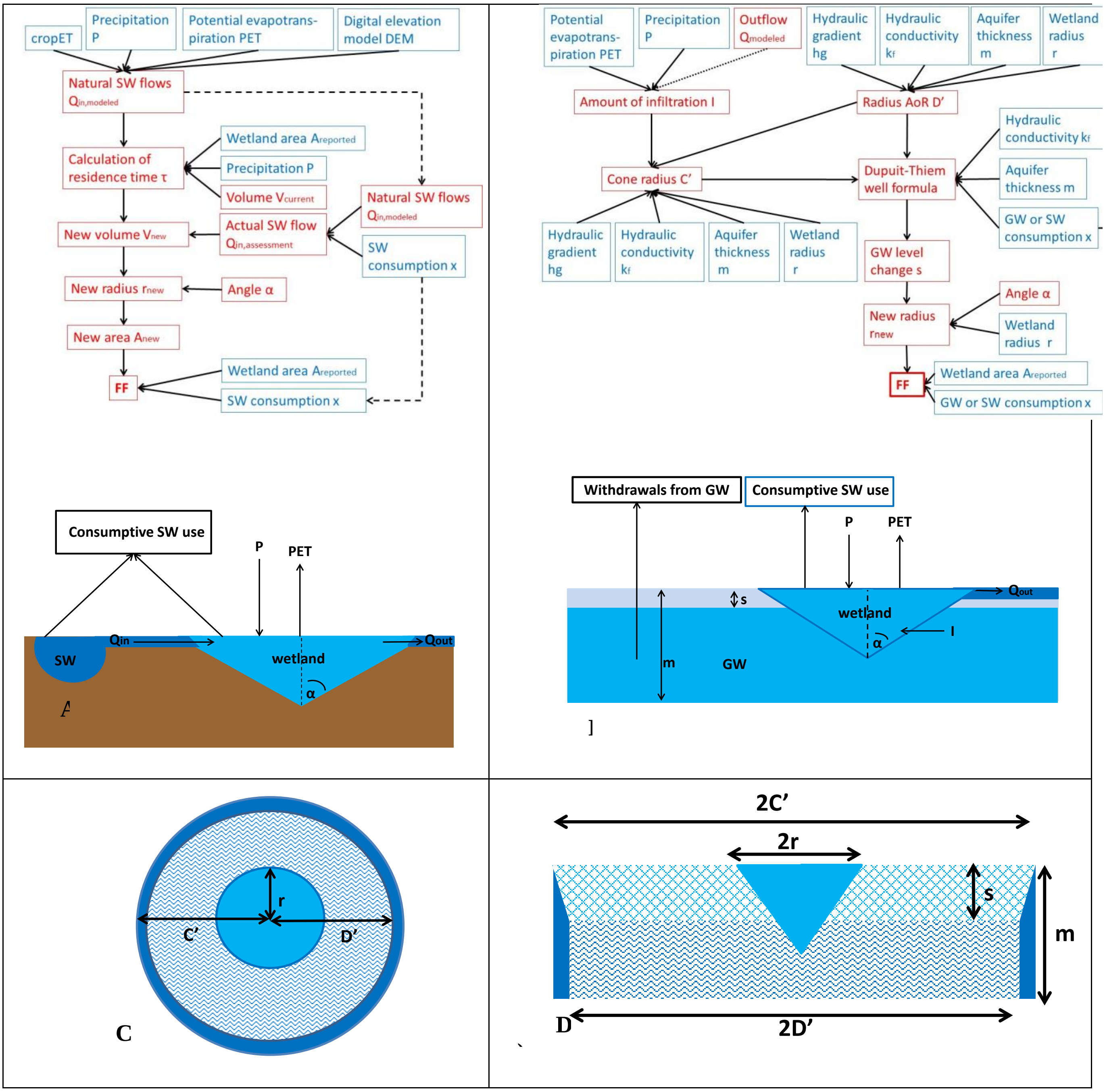


Figure 4.1: Overview of the calculation procedure for the FF and schematic representation of A) SW-fed wetlands and B) GW-fed wetlands. The dashed line indicates that the two boxes are the same. The dotted line indicates that this parameter is only required for some cases. Terms are as explained in the text. Blue boxes denote empirical data for each wetland, red boxes show what is modeled. C) Approach for calculating groundwater drawdown. The Ramsar site's radius is r. The area of relevance with radius D' is the assumed hypothetical well, the radius of the depression cone from pumping is estimated with radius C'. D) Cross section of the situation in C), with aquifer thickness m. The depression cone within the well itself is planar, i.e. follows a horizontal line. Since the whole area of relevance is considered as one well, this is the drawdown in the area. The wetland is shown as a blue triangle.

4.2.4. Fate factors for surface water-fed wetlands

The new wetland area A_{new} [m^2] after consumption of x_i was calculated with Equation 2, assuming the wetlands were circular cones with wetland radius r_{new} [m] and angle α at the embankment of the wetland (see Figure 1A and B).

$$A_{new} = r_{new}^2 \cdot \pi = \left(\left(\frac{3}{\pi} \cdot V_{new} \cdot \tan(\alpha)\right)^{\frac{1}{3}}\right)^2 \cdot \pi$$

Equation 4.2

The new volume V_{new} [m^3], including consumption rate x_i, was estimated with equations 3-5. Natural surface water flow volumes $Q_{in,modeled}$ and river courses were determined with a resolution of 0.5 arc-minutes through a digital elevation model (USGS, May 2012) and empirical data on precipitation (New et al. 2002), actual evapotranspiration (Zhang et al. 2010) and consumptive irrigation water use (Pfister et al. 2011) (details see SI, S4). The natural residence time τ [yr] (Equation 4.5) was calculated with current volumes $V_{current}$ (based on wetland area and water depth), precipitation P[21] on wetland areas $A_{reported}$ and $Q_{in,modeled}$ assuming steady-state conditions.

$$V_{new} = \tau \cdot \left(Q_{in,assessment} + P \cdot A_{reported}\right)$$

Equation 4.2

$$Q_{in,assessment} = Q_{in,modeled} - x_i$$

Equation 4.3

$$\tau = \frac{V_{current}}{Q_{in,modeled} + P \cdot A_{reported}}$$

Equation 4.4

The error introduced by assuming residence time τ and precipitation P to remain constant was smaller than 1% (SI, S6). $Q_{in,modeled}$ is natural SW flow and is thus in many cases larger than current SW flow, since withdrawals increased in many parts of the world. Ramsar areas started to be designated in the 70's, and in many cases the originally defined areas are still reported today although waterbody areas might have changed. Therefore, natural SW flows correspond better to many designated sites than current flows.

4.2.5. Groundwater-fed wetlands

The amount of infiltration (I) into the wetland was determined via simplified water balances (SI, S7). GW-fed wetlands are schematically modeled as pumps with evapotranspiration as the driving force for water depth drawdown, causing groundwater infiltration (SI, S8). The water that reaches the "pump" comes from an area of relevance (AoR) around the wetland, which depends on soil transmissivity and wetland water consumption. We calculated the radius of this relevant area (D' [m], Equation , see **Error! Reference source not found.**C) based on infiltration I in an aquifer having a thickness m, a saturated hydraulic conductivity of the soil k_f, and a threshold hydraulic gradient (hg). The AoR is at least the area of the wetland itself. In cases where D (Equation) is smaller than r, radius D’ is defined equal to r. The assumption of AoRs around wetlands is a simplification, since, depending on recharge rates and hydrogeological properties of aquifers, wetlands can suffer from pumping wells which are hundreds of kilometers away (e.g. loss of pressure in the confined Great Artesian Basin leads to reduced spring flows over large distances) (Fensham et al. 2003). However, no global hydrogeological models are currently available and thus simplifying assumptions were made to account for areas with relevant influence. Circular depression cones were simplifying assumptions since flow directions of aquifers were unknown. In reality, cones would be distorted, due to the circumfluent GW flow.

$$D = \frac{I}{k_f \cdot 2 \cdot \pi \cdot m \cdot hg} \quad \begin{cases} if\ D > r \ \rightarrow \ D' = \dfrac{I}{k_f \cdot 2 \cdot \pi \cdot m \cdot hg} \\ if\ D < r \ \rightarrow \ D' = r \end{cases}$$

Equation 4.6

The thickness m is an aquifer’s fully saturated vertical thickness and k_f is a measure of how well water can pass through available pores. We assumed that the soil’s k_f is a valid proxy for the underlying aquifer, since $k_{f,aquifer}$ were unknown. k_f was determined for each wetland individually from soil properties (SI, S4, (Batjes 2006; Saxton et al. 2006)). Literature values for k_f-values vary for unconsolidated sediments between 10^{-13} m/s and 10^{-2} m/s, for sandstone or karst between 10^{-10}-10^{-6} m/s and 10^{-6} and 10^{-2} m/s, respectively (Domenico et al. 1990). Our values vary between 10^{-7} and 10^{-5} m/s (average $9.4 \cdot 10^{-6}$ m/s), are in the middle range of sediments and do therefore not capture extreme conditions.

The hydraulic gradient (hg) is the slope between two points of a water level, along which groundwater flows. The threshold for hg was chosen as 0.0001, i.e. the wetland will not influence groundwater levels significantly in areas where hg is smaller. Natural hg ranges between 0.001 and 0.0001 for most regional groundwater flows (San Diego State University 2005). With the lower value for hg we adopt a conservative value, i.e. water consumption in a larger AoR is considered to affect the wetland.

The AoR with D’ was considered as one large, conceptual well, as proxy for many small distributed wells (Figure 4.1C), where x_{GW} [m^3/yr] was pumped. This is based on the assumption of steady-state conditions, under which the depression cones of individual wells would finally connect and build one joint depression cone. Unknown numbers of wells and pumping rates influence the FF considerably (SI, S7), because with larger number of wells, smaller amounts of water are pumped per well and depression cones decrease. The assumption of one well is a practical, but not conservative, solution and areal changes will, if anything, be underestimated. Wells are often designed for one or few households and for irrigation of adjacent fields without large transport distances, justifying our assumption of larger numbers of small wells (approximated as one conceptual well in steady-state conditions) instead of few large wells. The total number of wells in the US, for instance, was estimated as $15.9 \cdot 10^6$ wells (National Ground Water Association 2010), with an annual total

GW withdrawal of $1.09 \cdot 10^{11}$ m^3/yr in 2005 (Kenny et al. 2009), resulting in an average pumping rate of 6900 m^3/yr per well. However, in areas with intensive groundwater consumption and water transfers drawdowns might be underestimated.

The radius of the depression cone (C') with hg 0.0001 has to be slightly larger than D', since the wetland needs to pump not only I, but also x_{GW} (Equation 4.7). By calculating C' we estimate the additional area of relevance required to recharge x_{GW}. If C was smaller than the wetland itself, C' was calculated as r plus the difference between D and C.

$$C = \frac{I + x_{GW}}{k_f \cdot 2 \cdot \pi \cdot m \cdot hg} \quad \begin{cases} if\ C > r \ \rightarrow \ C' = \dfrac{I + x_{GW}}{k_f \cdot 2 \cdot \pi \cdot m \cdot hg} \\ if\ C < r \ \rightarrow \ C' = r + (C - D) \end{cases}$$

Equation 4.7

With the well formula of Dupuit-Thiem for unconfined aquifers and steady-state conditions (Stelzig 2012) (Equation 4.8) the depth of the depression cone s [m], resulting from x_{GW}, was determined. Since the water level within a well is horizontal (hg=0), the drawdown in the well equaled the groundwater drawdown in the AoR and the wetland (Figure 4.1D). We iteratively solved Equation 4.9 by adapting s to reach x_{GW}=1000 m^3/yr.

$$x_{GW} = k_f \cdot \pi \cdot \frac{m^2 - (m - s)^2}{ln \frac{C'}{D'}}$$

Equation 4.8

For GW-fed wetlands new radii r_{new} [m] (Equation 4.9) and A_{new} [m^2] were calculated based on groundwater drawdown s.

$$r_{new} = -s \cdot tan(\alpha) + r$$

Equation 4.9

4.2.6. Sensitivity analysis

The Ramsar wetland's area (default area) and the area of the water bodies within the site often did not match. In order to test the FFs sensitivity to area changes, FFs were also calculated for estimated waterbody areas. Whenever possible, waterbody areas were taken from the RIS (Ramsar Sites Information Service 2012) or the scientific literature. If no information was available, waterbodies were estimated by measuring the areas from Google Earth (2011) and analysis with GE-Path (Sgrillo 2012). If the satellite pictures did not allow measuring water bodies (e.g. because no water bodies are visible, for instance due to plant coverage), we assumed that waterbody areas were 60% of the Ramsar area, corresponding to the global average share of waterbody area in Ramsar sites.

Further sensitivity tests were conducted for wetland geometry (S3), and water depth, which we varied by ±50%. In wetlands with an assumed depth of 1 m, we additionally changed this to 8 m (SW-fed) and 3.8 m (GW-fed), respectively (average water depths of the Ramsar wetlands). GW or SW consumption was changed to 1'000'000 m^3/yr. For smaller wetlands that were depleted entirely when using this amount, a smaller consumption of 10

m^3/yr was assumed to quantify the sensitivity of the FF. We refrained from using 10 m^3/yr for all wetlands, since in large wetlands the lost volume was extremely small and led to numerical errors.

For GW-wetlands we changed the k_f-value to more extreme conditions (factor 100 larger). We did not test smaller k_f-values since an aquifer has to be present for feeding the wetlands and lower k_f-values would lead to almost impervious sediment layers and poor hydraulic conductivity.

For the SW-wetlands, $Q_{in,modeled}$ was changed to natural WaterGAP-values (WATCH 2011) to test the influence of SW flows and resolution. Estimating global flows is difficult and variations between models, such as different WaterGAP (Alcamo et al. 2003; WATCH 2011) versions and our estimation, are large. Runoff in our case included subsurface runoff, as it does in WaterGAP.

4.2.7. Coupling with the inventory

From inventory databases, information about the water source used (SW or GW), as well as the amount of water consumed within an AoR (for GW-fed wetlands) or upstream of a wetland (for SW-fed wetlands) should be obtainable. GW consumption in areas without GW-fed wetlands does not create impacts, since no Ramsar wetland is impacted. The FF is multiplied with the amount of water consumed to calculate total area lost. As an example, we calculated the wetland area loss due to the production of one rose grown in greenhouses in Kenya and the Netherlands. A rose consumes 3.4 l SW and 0.7 l GW in Kenya (Mekonnen et al. 2010) and 1.7 l SW (plus 1.7 l from precipitation) in the Netherlands (Anton 2013, Torrellas et al. 2012).

4.3. Results

4.3.1. Fate factors

The global average value for FF_{SW} and FF_{GW} were 0.53 $m^2 \cdot yr/m^3$ and 1.51 $m^2 \cdot yr/m^3$, respectively. Global median values for FF_{SW} and FF_{GW} were 0.33 $m^2 \cdot yr/m^3$ and 0.05 $m^2 \cdot yr/m^3$, respectively. For individual wetland results see SI. For the SW-wetlands, the largest FF_{SW} is 22 $m^2 \cdot yr/m^3$ (Wadi El Rayan Protected Area, Egypt), and the smallest is $9.1 \cdot 10^{-6}$ $m^2 \cdot yr/m^3$ (Les Rapides du Congo-Djoué, Congo). The large variation reflects the greatly varying water availability in humid and hyper-arid regions. FF_{GW} varies between 86.5 $m^2 \cdot yr/m^3$ (Point Pelee, Canada) and $1.4 \cdot 10^{-4}$ $m^2 \cdot yr/m^3$ (La Alberca de los Espinos, Mexico), reflecting the different hydrogeology of the sites. Aquifers with low hydraulic conductivities (and small thickness m) retard the infiltration of water from the aquifer to the wetland, thus resulting in a larger FF. FFs of the individual wetlands are shown in **Error! Reference source not found.** and the SI.

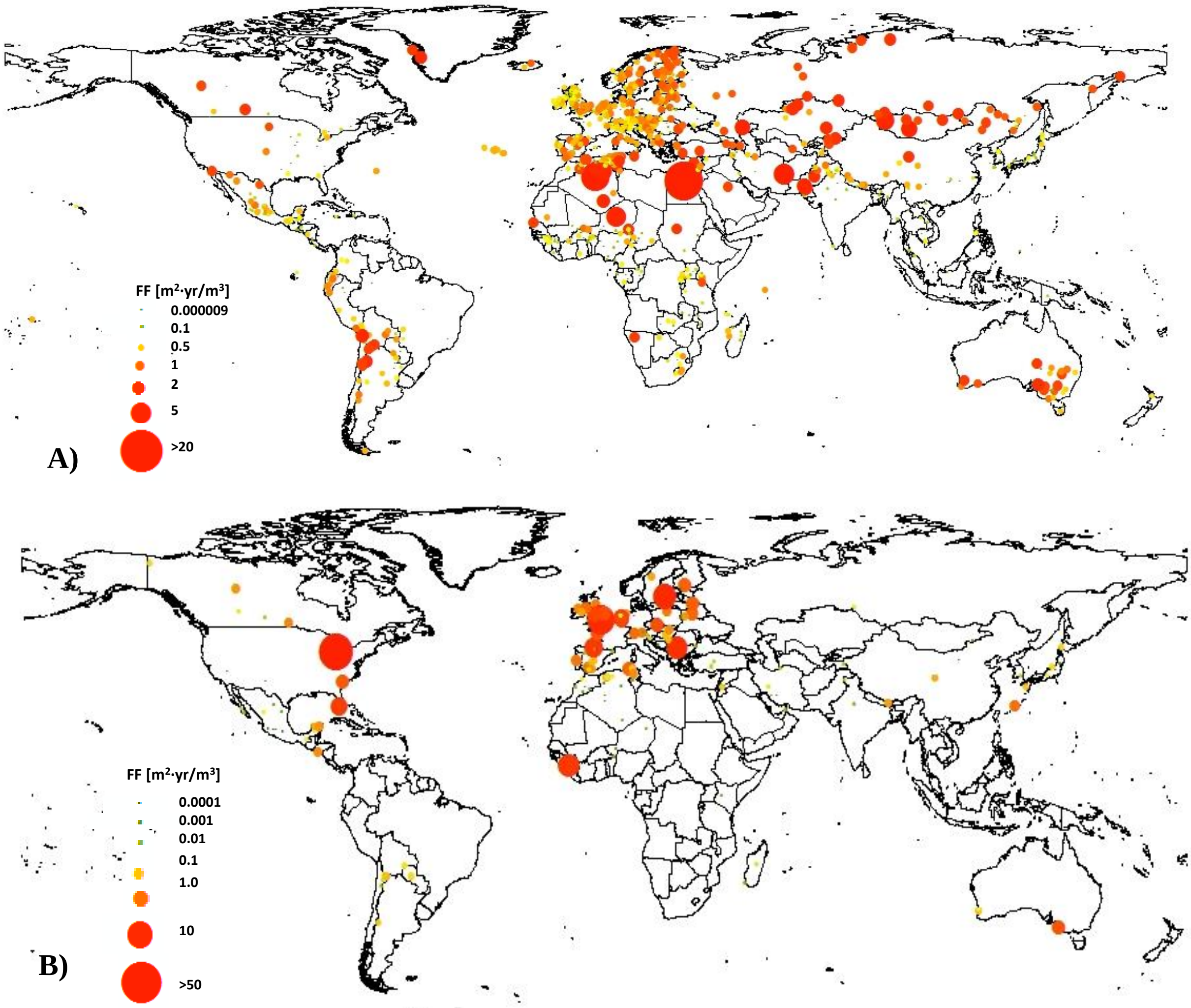


Figure 4.2: Maps showing the FF for each wetland. The values shown are considered as default FF values, calculated with Ramsar areas, the original water depth and 1000 m^3/yr water consumption. A) FF_{SW} for SW-fed wetlands, B) FF_{GW} for GW-fed wetlands. The size and color are relative to the calculated FF and not relative to the Ramsar area. Base map of administrative units adapted from ESRI (2009).

4.3.2. Sensitivity analysis

Some of the largest sensitivities for FF_{SW} were due to the surface water flows modeled (Figure 4.3A). With $Q_{in,modeled}$ from WaterGAP, the largest decreases were recorded in the Indus basin, and the largest increase was in Nicaragua. Differences in flow (e.g., Indus basin: WaterGAP $2\cdot10^{10}m^3$/yr, own estimate $39\cdot10^{10}m^3$/yr), delineation of the river network and resolution caused the differences in FF (SI, S10 and Excel). Under- or overestimation of flows as well as individual river courses and width of the modeled river, i.e. cell resolution, influence FFs considerably.

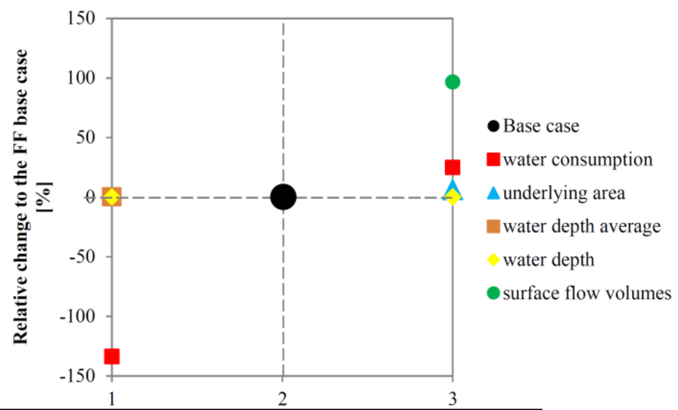
Changing the water consumption to 1’000’000 m3/yr, where possible, led to only small changes in FF. However, it caused a complete loss of 158 SW-fed wetlands within one year, which is unrealistic. For these wetlands, a smaller withdrawal of 10 m^3/yr was assumed to evaluate the difference in FF. For these small amounts, the change is not linear anymore, thereby contributing more to sensitivity.

Other parameters, such as changes in underlying area or water depth, have a much smaller impact on the median FF. However, changes in individual FFs can be substantial. Influences from changes in wetland geometry are negligible (S3).

FF_{GW} are highly non-linear (Figure 4.3B) and, hence, changing the amount of water consumed within the AoR had a very large influence. FF_{GW} increased by four orders of magnitude on average when changing consumption to 1 Mio m^3/yr due to the much larger, non-linear groundwater drawdown with such a large increase in consumption, leading to the complete loss of 71 wetlands within one year. For the latter wetlands we applied 10 m^3/yr, resulting in comparatively smaller changes of around 100%. Similarly, water depth and hydraulic conductivity k_f may lead to a change in median FF_{GW} of up to 100%.

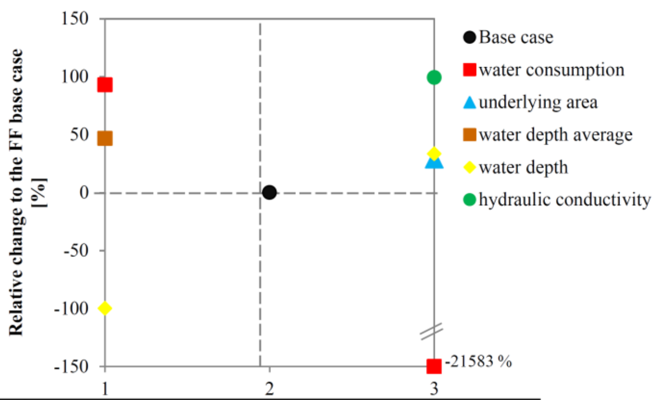
These are results for median global values. Results of the sensitivity analysis for each climate zone are shown in SI, S11 and show essentially the same pattern of sensitivities as the global median values.

A



parameter	1	2 (base case)	3
Water consumption [m³/yr]	10	**1000**	1'000'000
underlying area [ha]	-	**Ramsar area**	waterbody area
water depth average [m]	8 m (if 1m was assumed)	**data from literature**	-
water depth [m]	-50%	**data from literature**	+50%
surface flow volumes [-]	-	**own model**	WaterGap

B)



parameter	1	2 (base case)	3
Water consumption [m³/yr]	10	**1000**	1'000'000
underlying area [ha]	-	**Ramsar area**	waterbody area
water depth average [m]	3.8 m (if 1m was assumed)	**data from literature**	-
water depth [m]	-50%	**data from literature**	+50%
Hydraulic conductivity [m/s]	-	**calculated**	*100

Figure 4.3: Overview of the sensitivity analysis for FFs for A) SW-fed wetlands and B) GW-fed wetlands. The values shown here are the relative changes of the median global FF values for each parameter change. In the table below the

x-axis, the changed parameters and respective values for calculating the new FF for the sensitivity analysis are shown. Parameter values listed under number 1 are shown in the graph where number 1 is indicated on the x-axis. The same applies for numbers 2 and 3. Values outside the y-axis are indicated separately. 1'000'000 m^3/yr was used for 875 SW-fed wetlands and 80 GW-fed wetlands. The other 158 SW-fed and 71 GW-fed wetlands were calculated with 10 m^3/yr instead, since they would be completely lost with a consumption of 1'000'000 m^3/yr (see also main text).

4.3.3. Case study

Rose production in Kenya is widespread around lake Naivasha (SW-fed). Lake Elmenteita (GW-fed) is nearby with its AoR stretching up to lake Naivasha, and we apply both FFs. In the Netherlands roses are e.g. grown in the region of Bleiswijk, South Holland (Anton 2013). The closest Ramsar site is Haringvliet, near Rotterdam. By multiplying SW amounts from the inventory with the FF (see SI) we found that per rose produced lake Naivasha lost 0.003 $m^2 \cdot yr$ and Haringvliet 0.001 $m^2 \cdot yr$. For GW consumption, lake Elmenteita lost $5 \cdot 10^{-6}$ $m^2 \cdot yr$. FF_{SW} are almost equal for both sites, but the area loss is circa double in Kenya due to lower water consumption in the Netherlands. We used the FF for 1000 m^3/yr for all wetlands. It is important to choose the local FF that is closer to the actual water consumption, since, for GW-fed wetlands, area loss is sensitive to the water amount used (SI, S10). The impact of draining wetlands completely in one year is not included, since in LCA the typical focus is on marginal changes.

4.4. Discussion

4.4.1. Fate factors

FF_{SW} were dependent on the availability of natural SW flows as shown by analyses per humidity zone (SI, S10) where the largest factor occurs on average for hyper-arid areas and the smallest for humid areas. For GW-fed wetlands there was no connection to humidity zones. GW-fed wetlands were dependent on local aquifer thickness and hydraulic conductivity.

Steady-state conditions were assumed in the calculation of FF_{GW} in order to use the commonly used Dupuit-Thiem well formula. This assumption is common in simplified hydrological modeling and is often justified since groundwater flows vary relatively slowly(Committee on Ground Water Modeling Assessment et al. 1990). Depending on the aquifer's characteristics and independent of pumping rates(Pettyjohn 1987), steady-state conditions can be achieved within a couple of days or months, but especially in arid regions steady-state conditions might never be reached(Pettyjohn 1987). From an LCIA perspective, assuming steady-state is acceptable as primarily the magnitude of damage matters and not the time when it is reached. It is also consistent with fate models of other impact categories, e.g. toxicity(Rosenbaum et al. 2008).

4.4.2. Modeling approach

The core of this approach involved simplified hydrological modeling. It was not possible to apply more complex hydrologic models without raising data demand beyond data availability and hence significantly increasing the uncertainty of data input. Furthermore, many details cannot be included due to linearity of effects generally assumed in LCIA. Most currently available models for wetland water balances and interactions of SW and GW are site-specific, as, e.g., in Dietrich et al. (2007) and discussed in Jolly et al. (2008) SW-GW interactions are spatially and temporally highly dynamic and strongly controlled by local geomorphology[42] and site conditions. This varying relevance of individual water balance components leads to diverse modeling approaches for individual wetlands.[41] Modeling of a water balance's groundwater components is especially difficult, because these are often small and difficult to determine (Jolly et al. 2008). A site-specific approach is not feasible for our work since it would include establishing 1184 models and the exact knowledge of site conditions is mostly lacking. One approach does exist that attempts to model groundwater on a larger scale with global datasets, but it has only been tested for the Rhine-Meuse watershed(Sutanudjaja et al. 2011). Abstraction of GW cannot be modeled, as the model is not suitable for areas under heavy human water extraction, which would be relevant for this paper. The authors also state that hydrogeological data is a bottleneck and one of the main reasons for not including GW components in current large-scale hydrological models(Sutanudjaja et al. 2011).

4.4.3. Sensitivity analysis and applicability of FF

We conclude that uncertainties of FF_{GW} are much larger than for FF_{SW} (see also figure in SI, section 9), since more assumptions were made and the factors showed larger sensitivities. The largest sensitivity was the strong variation of FF_{GW} with the amount of water consumed, due to the strongly non-linear character of the Dupuit-Thiem well formula. The drawdown calculation is heavily influenced by the total water consumption within an AoR since a larger consumption leads to a much larger drawdown. Therefore, in addition to the source and location of water consumption (including the number of pumping wells and depression cones thereof), the total consumption in AoRs should be known for avoiding large uncertainties of several orders of magnitude. Also, aquifers are very complex and hydrogeological parameters may vary widely, as shown with the influence of changed k_f-values. This compromises the general application of FF_{GW} in LCA, as typically this information is not available. Although uncertainties are also comparatively large in other impact categories (e.g., toxicity), standard use of the FF_{GW} cannot be recommended at this point. The methodology may be used for global screening in cases where groundwater consumption is expected to have a large influence.

To evaluate the model, we quantified FFs for the Azraq Oasis and Lake Chad, for which information about area lost in the past was available. The magnitude of area loss was predicted in good agreement by our model (SI, S9). However, this study and also the global screening should always be complemented with an uncertainty analysis and, for those areas which show to be of potential relevance, local data on key parameters should be gathered to reduce the uncertainties.

Sensitivity of FF_{SW} to $Q_{in,modeled}$ was assessed based on our natural flow model and WaterGAP. The more arid the climate, the larger the difference between our estimation and WaterGAP (WATCH 2011). Differences between flow volumes of both WaterGAP models (Alcamo et al. 2003, WATCH 2011) and our model are in the same order of magnitude (SI, S4). The delineation of the river network, which we accomplished with TopoToolbox (Schwanghart et al. 2010), and spatial resolution are also important. The difference in resolution may lead to different results as the example in the Indus basin and Nicaragua illustrate. Better understanding and higher resolution justify the use of our own model for the current purpose. However, modeling of real natural flows of large rivers on a global scale is challenging, and hopefully better hydrological models alleviate this problem in future.

Since data for the areas of the Ramsar sites were much more reliable than for waterbody sizes, we recommend using the Ramsar areas as a basis for calculation. The differences between FF_{SW} modeled with Ramsar and waterbody areas were limited (see sensitivity analysis) and were mainly due to the different, natural SW flow values. The natural surface inflow was derived by extracting the maximum natural flow on $A_{reported}$. Values extracted for waterbody or Ramsar area may differ because the larger Ramsar area might extend into an area of higher river flows (i.e. a larger river). Thus, basing FFs on Ramsar areas seems advantageous since larger areas with possible river flows are covered.

No uncertainty information was given for data in the RIS, thus no further analysis was performed on uncertainties in the underlying data. Other sources of uncertainty and variation are e.g. the spatial variation of k_f and m, but without detailed models (not available on a global scale) this issue was not analyzed further.

A changing climate will influence the calculated FFs as wetlands are susceptible to changes in water quantity as well as to the timing of the flows (Erwin 2009). It is e.g. projected that river flows and water availability will decrease in the drier regions and increase in high elevations and parts of the wet tropics.[46] Thus, some wetlands may disappear while others might be less affected. However, due to the diversity of wetland types and their individual properties, actual impacts of climate change can only be determined on a case by case basis (Erwin 2009). Our approach could also serve for quantifying impacts of these climate change effects on wetland area via changes in SW flow and GW levels.

4.4.4. Ramsar wetlands and data

The distribution of Ramsar sites partly reflects the countries' economic means and political will to protect their wetlands (e.g., almost 50% are in Europe). Countries with extensive wetland areas but only few designated sites, as e.g. in Uganda and Kenya, highlight the discrepancy between wetland occurrence and number of Ramsar sites (SI, section 8). Taking Ramsar sites into account does therefore not purely address ecosystem quality but human-perceived quality as well. For non-designated wetlands global data availability was extremely scarce and they cannot be included in our assessment yet. It is thus important to note that the developed approach is dealing uniquely with designated Ramsar wetlands. However, the methodological approach can also be applied to other wetlands, once the necessary parameter values are available.

Information for wetlands was mostly collected from one data source (Ramsar Sites Information Service 2012) but not all necessary parameters were available for all wetlands. The determination of the main water source is partly subjective and not always explicit in the available data. Designating only one primary water source for feeding wetlands is a strong simplification since the separation of SW and GW within the water cycle is an artificial construct. There are few wetlands which are uniquely dependent on only one water source, but for most wetlands the degree of dependency on different water sources is still unknown. Therefore and to simplify analyses, we assumed strict dependencies on one water source. For GW-wetlands which showed a precipitation surplus we checked the plausibility of GW-dependency by contacting the contributors to the RIS where e-mail addresses were available. 9 replies were received, all of whom confirmed the dependency or non-dependency on groundwater as determined in our analysis. Overall, research in many wetlands regarding bathymetry, waterbody areas, water sources and depths is still in its infancy inducing large uncertainties.

For the calculation of natural flows, data from different sources were used for DEM (USGS 2011), P (New et al. 2002), AET (Zhang et al. 2010) and crop evapotranspiration (Pfister et al. 2011). All sources contained modeled data and used different assumptions and resolutions. Thus, they were not fully consistent and corrections (Equation 4.) were necessary.

4.4.5. Practical implications

For applying FFs, it is necessary that in the inventory the net water consumption of the respective process distinguishes the source of withdrawal. For SW consumption, both FFs can be used for direct SW consumption from GW-fed wetlands and for SW consumption in the watershed upstream of SW-fed wetlands. GW consumption is only relevant for GW-fed wetlands within their AoR. Research software that can deal with such a level of spatial differentiation is becoming openly available (Mutel et al. 2012), but is not yet standard among LCA practitioners. Neither is inventory data with an appropriate spatial resolution reported (e.g. Ecoinvent 2013), although e.g. agricultural inventories have been published in research papers with more detailed spatial resolution (Pfister et al. 2011). Therefore, while the application of the method presented here is currently limited to research projects and single case studies, we anticipate that it will become possible to deploy it more generally in the near future.

Multiplying a negative net consumption with the FF indicates a benefit for the wetland. This can be, e.g., the case if SW is used extensively for irrigation, recharges the aquifer and increases groundwater levels as e.g. shown in Verones et al. (2012)

The FFs can act as a global screening methodology for quantifying the area losses of wetlands due to water consumption. The translation of water consumption to area loss, as demonstrated in the example, makes it compatible with land use inventories where area losses are reported. However, the use of this methodology should always be accompanied by uncertainty analyses and, in relevant areas, complemented with case studies including local key data such as total water consumption.

FFs for different consumption and underlying area are provided in the SI. The FFs are a first step towards a complete characterization factor for these highly vulnerable and important types of ecosystems, since area losses of different ecosystems should not simply be summed without accounting for ecological quality. The next steps for enhancing this approach, i.e. providing characterization factors on a global level and an application example, are presented in ref.[15] The most important wetlands, designated as Ramsar wetlands, are covered here. In the future, with the inclusion of national wetland inventories, the global coverage of this approach will improve and allow consistent assessment of wetland impacts within LCIA.

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5. Effects of consumptive water use on biodiversity in wetlands of international importance

Francesca Verones#, Dominik Saner#, Stephan Pfister#, Daniele Baisero$, Carlo Rondinini$, Stefanie Hellweg#

#ETH Zurich, Institute of Environmental Engineering, 8093 Zurich, Switzerland

$Global Mammal Assessment program, Department of Biology and Biotechnologies, Sapienza, Università di Roma, 00185 Rome, Italy

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5.1. Introduction

Wetlands cover an area larger than 1'280 million hectares worldwide (Millennium Ecosystem Assessment 2005) and are among the most complex ecosystems in the world, due to a combination of different aquatic and terrestrial conditions. Many species have adapted to these mosaic ecosystems, leading to high varieties of all major groups of animals and plants (Halls 1997). Wetlands can contribute massively to biodiversity within a landscape, often exhibiting high levels of alpha and beta diversity (i.e. species diversity and turnover between habitats) (Flinn et al. 2008). They are frequently used by migratory birds for resting and can be important drinking water sources, especially in semi-arid and arid regions.

At the same time, wetlands are among the most threatened ecosystems on our planet (Lambert 2003). They are degraded and converted to human uses more rapidly than any other ecosystem, and the status of freshwater species is deteriorating faster than for other species (Millennium Ecosystem Assessment 2005; WWF 2012a). Globally, more than 50% of wetland areas were lost during the 20th century (Millennium Ecosystem Assessment 2005), mostly due to conversion and drainage (WWF 2012b). Since wetlands are essentially characterized by hydrologic conditions, changes in water volumes and timing of flows are major threats (Zedler et al. 2005).

Life Cycle Assessment (LCA) is a methodology for assessing the total environmental impacts of a product or service through its entire life cycle (ISO 2006). LCA has rapidly developed over the last years, and life cycle impact assessment (LCIA) methods have started to include impacts from water and land use. LCIA methods are available globally for assessing the impact of water use on ecosystems in general (Milà i Canals et al. 2009; Pfister et al. 2009), for the impact of surface water consumption on aquatic ecosystems (fish species) below 42° North (Hanafiah et al. 2011), and regionally for impacts of groundwater use on plant species in the Netherlands (van Zelm et al. 2011). Yet, no global methodology currently exists in LCA to determine the effects of surface and groundwater consumption specifically on wetland ecosystems. Furthermore, impacts on ecosystems and biodiversity are commonly calculated in PDF (potentially disappeared fraction of species) (Goedkoop et al. 2009; Goedkoop et al. 1999; Hanafiah et al. 2011; Pfister et al. 2009; van Zelm et al. 2011; Verones et al. 2012). However, PDF does not account for absolute variations in species richness. The same relative impact (e.g. a PDF of 0.5) is considered equivalent in a species poor ecosystem and a species rich ecosystem, although in the latter case more species are lost. Moreover, PDF does not take vulnerabilities and distribution ranges of species into consideration, treating all species equally whether critically threatened and endemic or widespread and common. For a review of existing practices and shortcomings in the assessment of biodiversity in LCA, see Curran et al. (2011) The aim of this paper is to develop a new approach for quantifying impacts on biodiversity due to anthropogenic water consumption. In this approach, effect factors (EF) measure the absolute loss of species due to wetland area loss, including the species' vulnerability. The existing fate factors(Verones et al. 2013) that quantify the loss of wetland area due to water consumption are combined with the EFs in order to calculate characterization factors.

5.2. Methods

5.2.1. Biodiversity data and maps

We took into account all inland wetlands which were classified under the Ramsar convention on 17 August 2012 (1184 wetlands) (Ramsar Convention 1994). 73% of these wetlands were considered to be important for waterbirds for different life stages, and 67% were important for birds in general. 25%, 26% and 49% were important for amphibians, reptiles and mammals, respectively (Supporting Information (SI)). Bird distribution data were available from BirdLife & NatureServe (BirdLife International et al. 2011), data for amphibians and reptiles were from IUCN (IUCN 2012a, c, d) (see SI for data sources and species numbers). For each species, map-files of the individual geographic extent of its distribution were available, including information about the presence, origin and season. Waterbirds were chosen for their obvious connection with water and were defined as birds whose primary habitat was, according to BirdLife, "wetland (inland)" or "artificial landscapes (aquatic)" (BirdLife International 2010). Residential birds whose habitat was not "wetland (inland)" or "artificial landscapes (aquatic)" were excluded since there is limited connection to wetlands and they are therefore considered irrelevant for estimating species loss in wetlands. Non-residential birds (excluding seasonal category "resident") were chosen because among them are migratory birds, which require staging and resting grounds during their migration. We assumed that wetland area loss would be a severe drawback for them along their migration routes. Non-residential waterbirds are included in the waterbird category to avoid double counting. We included all amphibian species and those reptile species whose habitat was defined from IUCN as "wetlands (inland)"(IUCN 2012a) and also contained data. For waterbirds, reptiles and amphibians all seasonal categories (SI) were included. The origin of species, i.e. whether they are native or introduced was not considered when calculating current species richness values. With the software Matlab (MathWorks 2011), species with presence categories "extant", "probably extant" and "possibly extant" (see SI) were identified, and the maps were transformed to raster files with a 0.05 decimal degree resolution with input datum WGS84. These were added up, resulting in global species richness maps for waterbirds, non-residential birds, reptiles and amphibians.

For mammals, habitat suitability models are available (Rondinini et al. 2011) and were used to refine their IUCN range. We considered only water-dependent mammals for calculating species-richness maps with 5 km resolution using the WGS84 datum. Other taxa (e.g. plants, fish) are not considered.

5.2.2. Calculation of effect factors

We assume that the aim of biodiversity assessment in LCA is to quantify and minimize the risk of global extinction of species of different taxa. We developed effect factors (EF) that quantify the contribution to potential global extinction of species due to a loss in wetland area that is caused by water consumption. The EF combines three parts based on global maps: (1) potential species loss, (2) vulnerability of present species communities and (3) habitat loss risk (graphical summary in SI, S9). The potential species loss in numbers of species-equivalents lost (S_{lost}) per area changed is derived from the species-area relationship that has been used in LCA before (Goedkoop et al. 2009; Koellner et al. 2008; Schmidt 2008), based on an original area $A_{reported}$ and a new area A_{new} (both from Verones et al. (2013), as well as the original species richness $S_{original}$ from global species richness maps (Equation 5.5, further explanations in SI, S8).

$$S_{lost} = \left(1 - \left(\frac{A_{new}}{A_{reported}}\right)^{z}\right) \cdot S_{original}$$

Equation 5.5

The exponent z is the slope of the species-area relationship and was derived from Drakare et al. (2006) for birds (0.37), mammals (0.34), amphibians (0.2) and reptiles (0.33) as described in SI. $S_{original}$ was derived from current species richness maps (SI) and therefore does not represent pristine species richness, but is appropriate because we were also using current wetland area data as a reference.

The vulnerability of species communities is quantified with a vulnerability score (VS) as an indicator for global extinction risk. VS is a function of the area of the extent of occurrence (EOO, encompassing the outermost geographic limits of all areas where a species occurs)(Gaston et al. 2009) as a predictor for the susceptibility to anthropogenic disturbance (because species with a small range are intrinsically rare) and a threat level (TL) indicating already occurring threats. VS is calculated as global maps for each species i in taxon p, and each pixel j (0.05°x0.05°, for mammals 5kmx5km) as the area of the respective pixel ($EOO_{i,j}$) where species i occurs divided by the total EOO of the species and multiplied with $TL_{i,j}$ (Equation 5.6). The EOO may include discontinuous areas and areas which may be unsuitable as habitat (IUCN 2001). Thus, it is larger than the actual area of occupancy of a species. TL represents discrete values ranging from 1 to 5 on a linear scale (1-least concern, 2-near threatened, 3-vulnerable, 4-endangered, 5-critically endangered) from the IUCN Red List of threatened species (IUCN 2012b). Total $VS_{p,j}$ of each taxon p per pixel is obtained by summing all values for all species i which occur in pixel j and dividing by the number of species of the taxon present in pixel j ($S_{p,j}$, Equation 5.6).

$$VS_{p,j} = \frac{\sum_{i=1}^{n} \frac{TL_{i,j} \cdot EOO_{i,j}}{\sum_{j=1}^{m} EOO_{i,j}}}{S_{p,j}}$$

Equation 5.6

As third part for the EF calculation the habitat loss risk index (CpA) was calculated on a global map (SI, S6). CpA is the weighted and scaled waterbody count per area, which was used as a weighting factor for the density of the network of waterbodies (as proxy of habitat rarity in the region). It accounts for the number of river sections (one point per section)(Lehner et al. 2008) and the number of data points from the global lakes and wetland database (Lehner et al. 2004) in each sub-watershed (Lehner et al. 2008). This was divided by the area of sub-watersheds and weighted with an aridity index (precipitation P (New et al. 2002)/potential evapotranspiration PET (Trabucco et al. 2009)), resulting in an index per pixel. The aridity index was included because in wet regions with relatively high P and low PET, wetlands are less relevant since alternative and temporary water supplies are more frequent. The CpA was scaled, so that 1 was the global maximum value in order to have an index that is relative to the smallest habitat loss risk. This led to lower values in more arid regions, i.e. the habitat type "wetland" was scarcer and thus more critical. Thus, the reciprocal of CpA is used in the EF calculation (Equation 5.7).

$$EF_{p,k} = \underbrace{\frac{\left(1-\left(\frac{A_{new,k}}{A_{reported,k}}\right)^{z}\right)\cdot S_{p,original}}{\left(A_{reported,k}-A_{new,k}\right)}}_{\textbf{Potential loss of species per area loss}} \cdot \underbrace{VS_{p,k}}_{\substack{\textbf{Threat status}\\ \textbf{and occurrence}\\ \textbf{of species}}} \cdot \underbrace{\frac{1}{CpA_k}}_{\textbf{Habitat loss risk}}$$

Equation 5.7

The aforementioned parts are combined to one formula for calculating the effect factor (EF) for each wetland k and each taxon p. The unit of the EF is species-eq/m^2, that is, global species-equivalents lost per square meter of wetland lost. For wetlands which are smaller than a pixel, parameters $S_{original}$, VS and CpA were extracted from global maps. In case the wetland was larger than one pixel, the parameter values were extracted from all relevant pixels and averaged over the Ramsar or waterbody area ($A_{reported}$) of wetland k. A_{new} is the calculated new area after water consumption (calculated as described in Verones et al. (2013)).

5.2.3. Calculation of characterization factors

Characterization factors (CFs) were calculated as a combination of effect (EF) and fate factor (FF, relating water consumption to wetland area loss) (Verones et al. 2013) (**Error! Reference source not found.**) for each taxon p and for each wetland k. The unit of the CF is species-eq·yr/m^3 water consumed and shows the loss of species-eq. due to water consumption.

$$CF_{p,k} = FF_k \cdot EF_{p,k}$$

Equation 5.8

For each SW-fed wetland, we assumed that water consumption upstream would reduce the inflow into the wetland and thus cause biodiversity damage. This upstream area was determined in Matlab (MathWorks 2011) by selecting, on a hydrologically corrected digital elevation model (DEM) (USGS 2011) (resolution 0.05°x0.05°), the parts of a watershed above and at the same altitude as the center of the wetland. Isolated areas with no connection to the wetland were removed from this selection. Where catchments of different wetlands overlapped, CFs were summed (SI). Therefore, consuming water in the upper part of a watershed is potentially worse than at the mouth, since consumed water impacts all wetlands located downstream but has no impact on upstream areas.

For GW-fed wetlands CFs were applicable on the respective area of relevance (AoR). The AoR surrounds the wetland and is the area from which water infiltrates into the wetland (see Verones et al. (2013) for further explanations). Due to geohydrological conditions, these conceptual areas can be large, and it is possible that the whole watershed is regarded as the AoR. For some cases this is realistic, e.g. in the Great Artesian Basin in Australia, where pumping of water reduces spring flows which are large distances away (Fensham et al. 2003). Due to a lack of good global maps of aquifer presence, we used the surface watershed borders to define the maximal size of AoRs. In areas with overlapping CFs from different GW-fed wetlands, values were summed since multiple wetlands are influenced if pumping occurred in that region (SI).

5.2.4. Sensitivity analysis

For the sensitivity analysis of EFs of birds, amphibians and reptiles, the presence category “possibly extinct” (category 4) was included in the species richness maps, since the respective species may still occur in very low numbers in those areas. Furthermore, high number of potentially extinct species might be an indicator of increased vulnerability of the region.

Additionally, the Ramsar area was changed to the reported waterbody area (as done in the FF calculations)(Verones et al. 2013) in order to test the sensitivity of EFs and CFs towards area differences. As discussed in Verones et al. (2013) the aim of the suggested CF is to take complete wetland areas into account and not only open waterbodies. Some wetlands (water-logged soil, waterbodies overgrown with vegetation) are invisible on satellite pictures, making the Ramsar area, which takes into account invisible waterbodies important to biodiversity, more suitable as the base area. For water-dependent mammals, extent of suitable habitat was used as a proxy for area of occupancy (AOO). The AOO is the area within the EOO where species actually occur (Gaston 1991). In order to test the sensitivity of the EF for water-dependent mammals, we calculated it once with EOO and once with AOO. The influence of other relevant parameters from the FFs (Verones et al. 2013), such as amount of water consumed, hydraulic conductivity, water depth and SW flow volumes, were integrated into the sensitivity analysis of the CFs.

We also checked using the Spearman’s rank-order correlation whether there were correlations between CpA, S and VS and how they correlate to the EF. Since both FF and CpA contain information about precipitation and potential evapotranspiration, we also tested their correlation.

For each taxon, we calculated CFs with the unit PDF·yr/m^3 in order to compare them to the values with species-eq·yr/m^3. For calculating CFs with PDF units, we omitted VS and CpA from the EF calculation.

5.2.5. Application example

We calculated the impact from water consumption for the production of a bunch of 10 roses in Kenya and in the Netherlands. Roses are the dominant cut flower in Kenya, accounting for almost 88% of cultivation (Kenya Flower Council 2012). One of the largest production areas (1911 ha in 2006) (Mekonnen et al. 2010) is close to lake Naivasha (SW-fed), which is listed under the Ramsar Convention as being internationally important. One average rose (25 g) consumed 4.1 l of irrigation water: 3.4 l surface water and 0.7 l groundwater (Mekonnen et al. 2010). Rose production in the Netherlands (region of Bleiswijk, South Holland) was assumed to require only 1.6 l/stem irrigation water from surface water, in addition to 1.6 l/stem precipitation (Antón 2013; Torrellas et al. 2012). CF_{SW} and CF_{GW} were extracted from the CF-maps at the location of lake Naivasha and Bleiswijk. This example serves as an illustration and does not aim at calculating a full LCA.

5.3. Results

5.3.1. Biodiversity

Maps of species richness and VS for all taxa are shown in SI for presence categories 1-3 and 1-4. Maximum species richness was 284, 112, 27, 134, 15 for waterbirds, non-residential birds, reptiles, amphibians and water-dependent mammals, respectively. Differences between bird richness maps and VS with presence 1-3 and 1-4 were zero or very small. The maximal differences in wetland regions were 5 species for amphibians, 3 for waterbirds and 1 for non-residential birds, reptiles and mammals. For reptiles, mammals and amphibians, maximum species richness and overall global distribution is smaller than for birds, and, in contrast to birds, they were not present in all wetlands (SI).

5.3.2. Effect factors

EFs of wetlands vary over many orders of magnitude, underlining the importance of spatial differentiation in LCIA of water consumption (see individual EFs in SI). The mean, minimal and maximal EFs for all wetlands are displayed in Table 1, and the averages for SW and GW are often similar. The dominant EFs were in the majority of cases calculated for waterbirds, being often the most species-rich taxon, closely followed by non-residential birds and amphibians. EFs for reptiles and water-dependent mammals were often the smallest, since species richness was often the smallest of all taxa. EFs for reptiles, amphibians and water-dependent mammals may be zero, since not all wetlands harbor these taxa. 121, 168 and 43 SW-fed wetlands show absences of mammals, reptiles and amphibians, respectively. In groundwater-fed wetlands, there are no mammals, reptiles and amphibians in 28, 25 and 6 wetlands, respectively. Reasons for the difference between EFs of SW-fed and GW-fed wetlands are related to the different locations of the wetlands, as well as the different changes in area that are influencing the non-linear species-area relationship.

Table 5.1: Summary of non-zero effect factors (EF) [species-eq/m^2] and characterization factors (CF) [species-eq·yr/m^3] for waterbirds, non-residential birds, amphibians, reptiles (all with presence categories 1 to 3) and water-dependent mammals (based on EOO), based on the area of the Ramsar sites. The last lines show combined (summed) EFs and CFs, assuming the same weight for all species. CV is the coefficient of variation.

	EF [species-eq/m^2]		CF [species-eq·yr/m^3]	
	SW	GW	SW	GW
waterbirds min	3.2E-14	3.1E-13	1.6E-15	5.0E-15
waterbirds max	1.6E-05	1.8E-06	1.1E-05	2.1E-07
waterbirds mean	3.6E-08	1.8E-08	1.2E-08	4.2E-09
CV	19	8	28	5
non-residents min	1.5E-15	1.4E-13	5.3E-17	6.9E-15
non-residents max	9.1E-06	1.5E-06	2.2E-07	3.0E-06
non-residents mean	1.9E-08	1.6E-08	1.2E-09	2.1E-08
CV	18	7	8	11
water-dep. mammals min	8.9E-16	5.0E-15	3.5E-17	1.2E-16

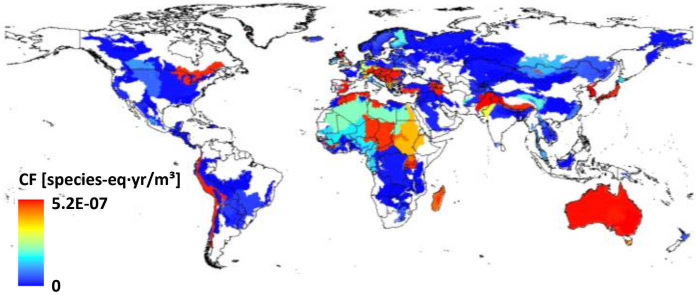
water-dep. mammals max	2.0E-06	2.0E-07	3.3E-08	5.3E-08
water-dep. mammals mean	2.4E-09	2.5E-09	1.3E-10	4.8E-10
CV	26	7	10	9
wetland reptiles min	2.74E-16	8.54E-15	1.22E-17	4.63E-17
wetland reptiles max	4.37E-07	1.13E-05	5.11E-07	9.53E-07
wetland reptiles mean	8.69E-10	7.69E-08	6.18E-10	7.58E-09
CV	16	12	26	10
amphibians min	1.17E-15	5.32E-15	4.38E-17	5.82E-16
amphibians max	6.47E-05	7.71E-07	4.56E-05	9.56E-08
amphibians mean	8.50E-08	1.23E-08	4.58E-08	1.75E-09
CV	24	5	31	5
combined taxa min	1.58E-13	9.34E-13	2.33E-15	1.57E-14
combined taxa max	8.07E-05	1.15E-05	5.68E-05	3.27E-06
combined taxa mean	1.43E-07	1.26E-07	6.01E-08	3.51E-08
CV	19	8	29	8

Spearman's rank-order correlations ρ (SI) between S, VS and EF are largest for amphibians and reptiles, indicating that VS has a large influence on EF. CpA is always negatively correlated to EF. Most mutual correlations between CpA and VS, as well as CpA and S were low or non-existent, illustrating that they are complementary to each other. Correlations between S and VS were large, since VS is related to the number of species present and assigns them a weight according to their vulnerability. Histograms for species richness show that waterbirds are usually the most species-rich taxon and reptiles the taxon with lowest richness (SI).

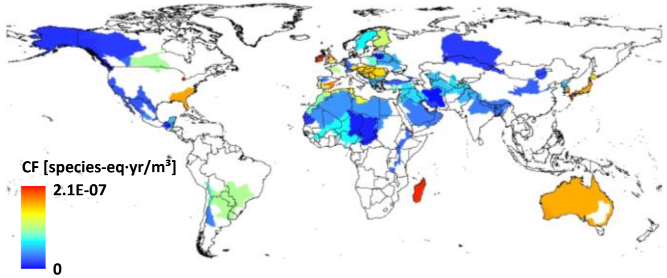
The median difference between EFs calculated with Ramsar area or waterbody area was a factor of 1.67 for all taxa in SW-fed and GW-fed wetlands (SI). Differences in S, VS and CpA were mostly low (less than 2%), since in many cases there was no or little difference between S, VS and CpA on Ramsar or waterbody area. Nevertheless, the underlying area was very important due to the non-linearity of the species-area relationship.

5.3.3. Characterization factors

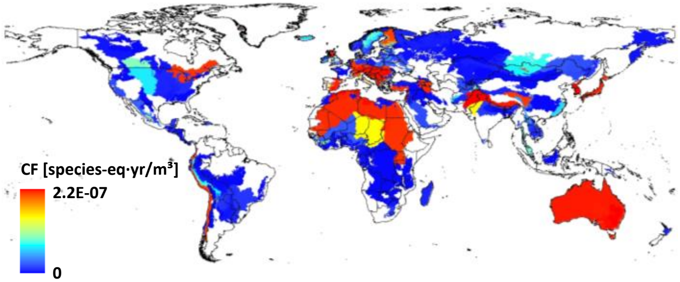
Characterization factors (CFs) were calculated for all wetlands (see SI) and result in global CF maps (**Error! Reference source not found.**). Impacts for waterbirds were mostly larger than for other taxa. In the majority of the cases, magnitudes of impacts on non-residential birds were next, closely followed by amphibians and then mammals and reptiles. Where $CF_{non\text{-}residential\ birds}$ was larger than $CF_{waterbirds}$, it was mostly due to higher vulnerability scores (VS) and only in few cases due to higher species richness (S). $CF_{mammals}$ was never larger than $CF_{waterbirds}$ and only in few cases larger than $CF_{non\text{-}residential\ birds}$, uniquely due to larger VS. $CF_{amphibians}$ and $CF_{reptiles}$ were larger than $CF_{waterbirds}$ in 86 wetlands and 2 wetlands, respectively (out of 1184), and this was always accompanied by a larger VS. Not all wetlands had CFs for mammals, reptiles or amphibians (CF=0) and therefore the coverage of CFs was smaller than that for the bird species (**Error! Reference source not found.**). CF_{SW}s covered 153 out of 238 major watersheds for birds, containing 76% of global land surface (LS). Analogously, CF_{SW}s for mammals, reptiles and amphibians covered 135 (64% LS), 129 (61% LS) and 146 (67% LS) major watersheds, respectively. CF_{GW}s of birds, mammals, reptiles and amphibians covered 70 (37% LS), 57 (30% LS), 60 (28% LS) and 68 (35% LS) major watersheds, respectively.



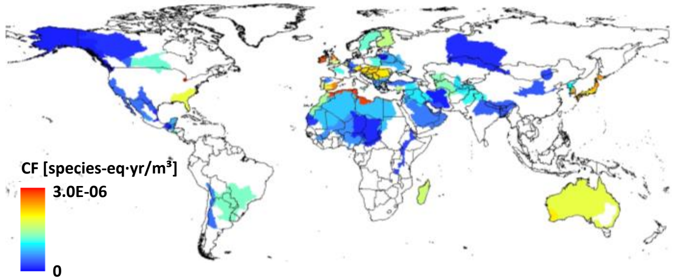
A **SW, waterbirds**



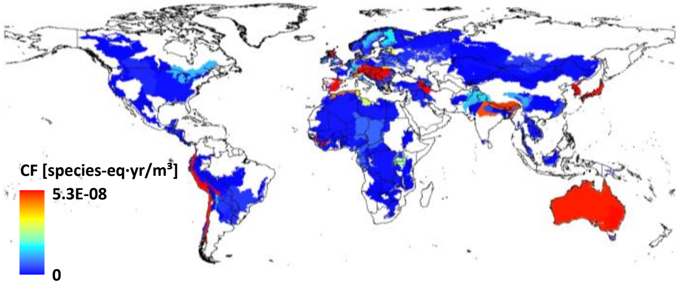
E **GW, waterbirds**



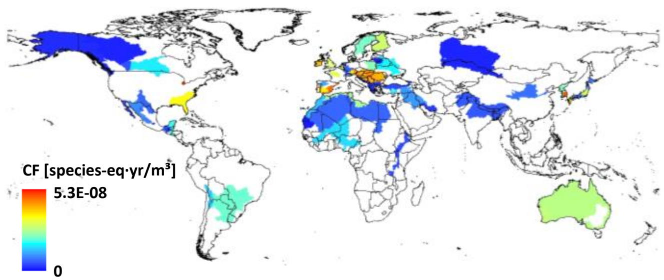
C **SW, Non-residential birds**



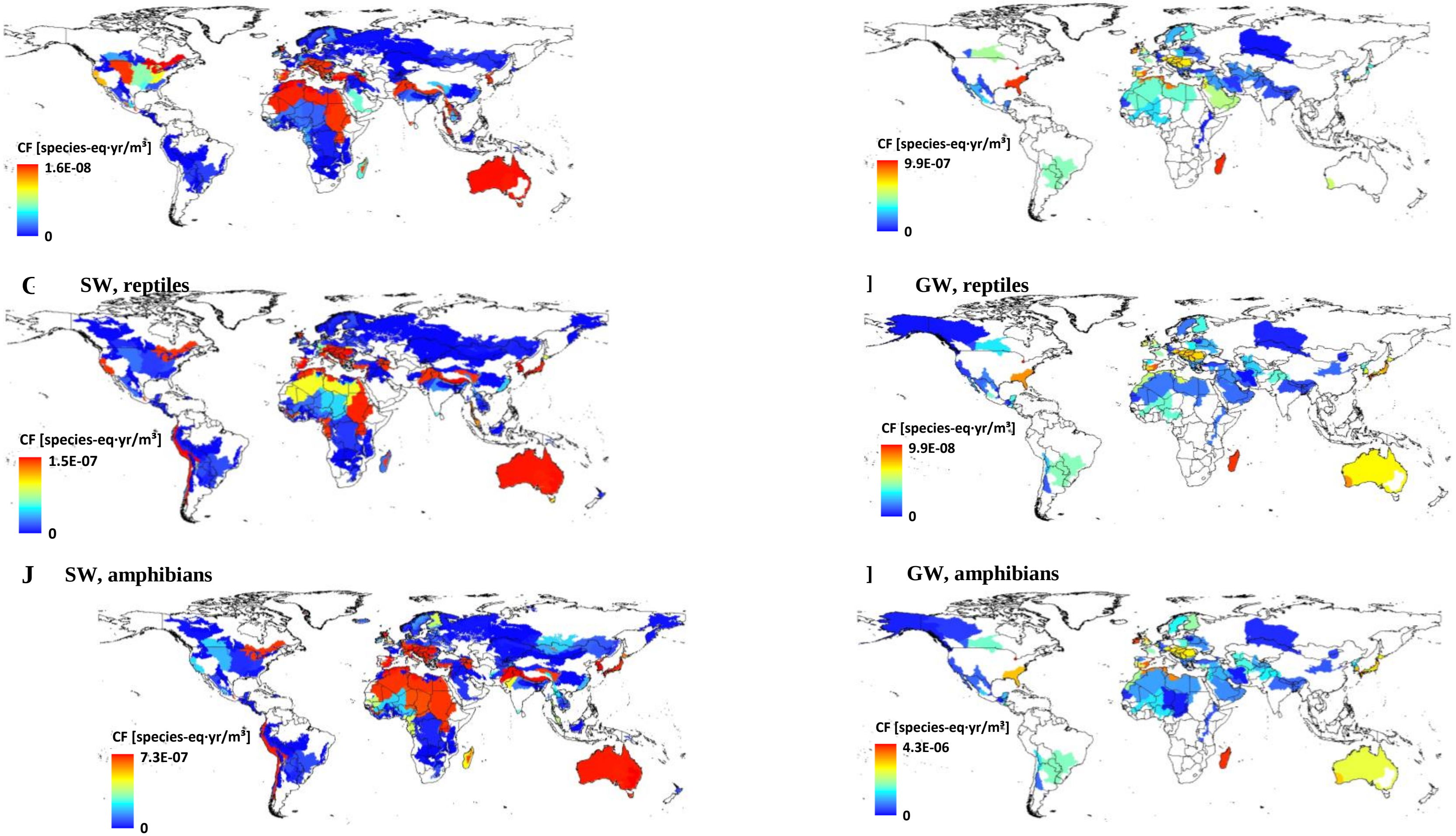
] **GW, Non-residential birds**



l **SW, water-dependent mammals**



] **GW, water-dependent mammals**



L) SW, all taxa combined **M) GW, all taxa**

Figure 5.4: Global maps of CFs, with indications of wetland type (SW – surface water-fed, GW – groundwater-fed) and taxon. White areas are areas where no impact on a Ramsar wetland is perceived. Base map with country boundaries adapted from ref.(ESRI 2009) A) CFs for waterbirds and SW consumption in SW-fed wetlands. B) CFs for waterbirds and GW consumption in GW-fed wetlands. C) CFs for non-residential birds and SW consumption in SW-fed wetlands. D) CFs for non-residential birds and GW consumption in groundwater-fed wetlands. E) CFs for water-dependent mammals and SW consumption in SW-fed wetlands. F) CFs for water-dependent mammals and GW consumption in GW-fed wetlands. G) CFs for wetland reptiles and SW consumption in SW-fed wetlands. H) CFs for wetland reptiles and GW consumption in GW-fed wetlands. J) CFs for amphibians and SW consumption in SW-fed wetlands. K) CFs for amphibians and GW consumption in GW-fed wetlands. L) CFs for all taxa combined and SW consumption in SW-fed wetlands. M) CFs for all taxa combined and GW consumption in GW-fed wetlands. Note that the CFs for groundwater-fed wetlands should only be used for marginal changes or cases studies with good data coverage, due to the uncertainty in the FFs (Verones et al. 2013).

The variation of CFs is shown in **Error! Reference source not found.**. The large sensitivity of CFs to changes in underlying area (Ramsar area or waterbody area) was determined by the sensitivity of FFs (Verones et al. 2013) and by the non-linearity of the species-area relationship for the EFs (SI). It is suggested to use Ramsar-area based CFs, since these underlying areas contain less uncertainty (see also Verones et al. (2013)), even though the larger areas are not a conservative assumption.

Additionally, the CFs are sensitive to the amount of water consumed, surface water flows (SW-fed wetlands) and hydraulic conductivity (GW-fed wetlands). Including species which were “possibly extinct” or changing from EOO to AOO (mammals) had little influence on most CFs (SI). We found no or small correlation between CpA and FF for SW and GW wetlands. Spearman’s rank order coefficients between CF_{SW} and CF_{GW} of different taxa were large (ρ>0.5) between most taxa (SI).

CF_{SW} and CF_{GW} with the unit PDF·yr/m^3 were strongly correlated to CFs (in species eq·yr/m^3) of waterbirds and amphibians, and CF_{GW} was also correlated to non-residential birds. An example for the differences between CF in PDF and in species-eq. is given in SI, showing that species richness, vulnerability and CpA matter.

5.3.4. Application example

We calculated the impact of water consumption during the cultivation of 10 roses in Kenya and the Netherlands (Table 5.2). The largest impact was caused in both cases for waterbirds. In Kenya, impacts from SW consumption were for all taxa larger than those from GW consumption. The impact in the Netherlands is only from SW consumption, since no groundwater is consumed and 50% of the required water for growing roses is from precipitation. Assuming the same weight for all taxa, we added the CFs for all taxa. In total, the impact in Kenya is 67 times larger than the one in the Netherlands due to larger species richness, larger VS, smaller CpA and a larger amount of irrigation water required (4.1 l/stem vs. 1.6 l/stem).

Table 5.2: Calculated impacts from water consumption for the production of a bunch of 10 roses in Kenya and the Netherlands. Impacts are reported for each taxon separately, before summing them to total impacts due to SW consumption or GW consumption. There is no GW consumption at the case study site in the Netherlands.

	Kenya [species-eq·yr]		The Netherlands [species-eq·yr]
Species	**SW**	**GW**	**SW**
Waterbirds	4.75E-12	1.29E-14	3.30E-14
Non-residential birds	2.80E-13	8.65E-16	5.23E-14
mammals	7.46E-14	2.20E-16	2.14E-15
reptiles	3.87E-15	1.14E-17	3.06E-16
amphibans	1.06E-12	9.25E-15	4.79E-15
TOTAL	6.17E-12	2.33E-14	9.25E-14

We refrained from calculating a complete LCA because our units of biodiversity loss are not compatible to existing methods, which are mostly based on PDF.

5.4. Discussion

5.4.1. Effect factors

The inclusion of ecosystem rarity (through CpA) as well as vulnerability of biodiversity (with VS) is a considerable improvement in comparison to existing LCIA methodologies for the assessment of ecosystem damages. A similar approach was presented by Michelsen (2008) for land use impacts on biodiversity from forestry operations in Norway. He defined ecosystem scarcity and ecosystem vulnerability factors which "give information on the intrinsic biodiversity value of an area"(Michelsen 2008), as well as a criterion for including present conditions for biodiversity in an area. He thus addressed scarcity and vulnerability on an ecosystem level. With VS, we focus on vulnerability and rarity at species level while CpA, being an ecosystem scarcity index, goes in a similar direction as Michelsen's work.

The combination of TL and EOO into VS considers that impacts in some regions, where rare species reside, may be worse than in regions with more common species, despite the fact that the absolute loss of species in the area might be similar. For example, absolute species loss for waterbirds in "Reserva Natural Lagunas de Campillos" (Spain) and "Luknajno Lake Nature Reserve" (Poland) are similar, but VS and CpA are one order of magnitude larger and smaller, respectively (i.e. higher vulnerability and higher habitat loss risk) in the Spanish wetland. Waterbird species richness was 70 in the Spanish wetland and 91 in the Polish wetland, but the EF of the former was 30 times larger.

Both TL and EOO are necessary for complete vulnerability scores. The Red List criteria based on geographic distribution (i.e. EOO or AOO) are important since 21% of the threatened bird species are listed as threatened solely on the basis of geographic distribution and 40% of threatened bird species qualified as threatened under at least one criterion containing geographical distribution (Gaston et al. 2009). Mammals were listed as threatened uniquely due to changes in geographic distribution in 35% of the cases, and qualified as threatened under at least one criterion of geographical distribution in 44% of the cases (Gaston et al. 2009). Yet range size informs only one of the five Red List criteria, and it is not sufficient alone for a species to qualify as threatened (indicators of decline in addition to the small range are indeed necessary for this). The TL highlights the species' imminent threats, as considered by the Red List, and reflects changes in EOO and population size. EOO itself gives only an indication about geographic distribution, not about threat. However, it provides additional information for non-threatened, small range species which are more likely to become threatened in future due to fewer possibilities of evasion of impacted regions.

The reference situation commonly used in land use assessment is the potential natural vegetation, even though there are ongoing debates about appropriate reference states (Koellner et al. in press). Due to the lack of data availability, we used the current situation as reference. One might assume that most wetlands have historically decreased in size, but information about past developments is incomplete. Hence, it is more logical to apply current species richness values instead of historical values. Therefore, we excluded confirmed extinct species in the EF calculation.

5.4.2. Characterization factors

The inclusion of ecosystem rarity (through CpA) as well as vulnerability of biodiversity (with VS) is a considerable improvement in comparison to existing LCIA methodologies for the assessment of ecosystem damages. A similar approach was presented by Michelsen (2008) for land use impacts on biodiversity from forestry operations in Norway. He defined ecosystem scarcity and ecosystem vulnerability factors which "give information on the intrinsic biodiversity value of an area"(Michelsen 2008), as well as a criterion for including present conditions for biodiversity in an area. He thus addressed scarcity and vulnerability on an ecosystem level. With VS, we focus on vulnerability and rarity at species level while CpA, being an ecosystem scarcity index, goes in a similar direction as Michelsen's work.

The combination of TL and EOO into VS considers that impacts in some regions, where rare species reside, may be worse than in regions with more common species, despite the fact that the absolute loss of species in the area might be similar. For example, absolute species loss for waterbirds in "Reserva Natural Lagunas de Campillos" (Spain) and "Luknajno Lake Nature Reserve" (Poland) are similar, but VS and CpA are one order of magnitude larger and smaller, respectively (i.e. higher vulnerability and higher habitat loss risk) in the Spanish wetland. Waterbird species richness was 70 in the Spanish wetland and 91 in the Polish wetland, but the EF of the former was 30 times larger.

Both TL and EOO are necessary for complete vulnerability scores. The Red List criteria based on geographic distribution (i.e. EOO or AOO) are important since 21% of the threatened bird species are listed as threatened solely on the basis of geographic distribution and 40% of threatened bird species qualified as threatened under at least one criterion containing geographical distribution (Gaston et al. 2009). Mammals were listed as threatened uniquely due to changes in geographic distribution in 35% of the cases, and qualified as threatened under at least one criterion of geographical distribution in 44% of the cases (Gaston et al. 2009). Yet range size informs only one of the five Red List criteria, and it is not sufficient alone for a species to qualify as threatened (indicators of decline in addition to the small range are indeed necessary for this). The TL highlights the species' imminent threats, as considered by the Red List, and reflects changes in EOO and population size. EOO itself gives only an indication about geographic distribution, not about threat. However, it provides additional information for non-threatened, small range species which are more likely to become threatened in future due to fewer possibilities of evasion of impacted regions.

The reference situation commonly used in land use assessment is the potential natural vegetation, even though there are ongoing debates about appropriate reference states (Koellner et al. in press). Due to the lack of data availability, we used the current situation as reference. One might assume that most wetlands have historically decreased in size, but information about past developments is incomplete. Hence, it is more logical to apply current species richness values instead of historical values. Therefore, we excluded confirmed extinct species in the EF calculation.

5.4.3. Data quality and sensitivity

Taking the waterbody area instead of the Ramsar area had the most substantial influence on the EF. Most often, waterbody areas were smaller than Ramsar site areas, and the EF calculated with the smaller area was larger, showing the susceptibility of smaller areas to change. CFs reacted sensitively to changes in underlying area, which is mostly due to the sensitivity of the FF (discussed in Verones et al. (2013)) and the non-linear species-area relationship (see SI). As discussed in Verones et al. (2013) FFs depend strongly on the water consumption (especially for GW-fed wetlands) and this is propagated also to the CF (SI). Hydraulic conductivity and surface water flows also contribute to the sensitivity of the CF.

Global data coverage was available for all taxa, but data quality (i.e. the accuracy and completeness of the range maps) (Jetz et al. 2011) differed within and between taxonomic groups. For example, there are 6285 amphibians species listed in the IUCN red list (IUCN 2012a). Map and threat level data (both are needed) are available for 6021 species (IUCN 2012a, c), i.e. for 264 amphibian species no data was available. Reptile map-data is available for 3087 species (IUCN 2012d). Only 348 reptile species live in wetlands (IUCN 2012a), and of those 80 do not have map data, meaning we could consider 268 species. Many reptile species are also still unknown, in contrast to birds and mammals which have more complete records (Groombridge et al. 2002). There is presently no uncertainty information associated to the primary species richness data, impeding a quantification of the associated uncertainty of EF. However, we are confident that the error is small especially for birds and mammals, since most of their species are well-known. Differences between species richness maps with different presence categories were low, indicating basic robustness to data completeness and that the general patterns likely reflect reality. Taking EOO data for most species groups and assuming no seasonal influence on presence/absence (i.e. migration), especially for birds, is likely to have overestimated richness data. Therefore, it is more useful to think of S as the maximum potential species richness of recorded species. For water-dependent mammals there was little difference in species richness when the habitat suitability map (AOO) or the outermost geographical limits of the EOO were taken (SI, S6). AOOs for other species are not yet available, and sensitivities to changes in habitat area cannot be assessed. However, as data becomes available on AOO through distribution models (Elith et al. 2006), expert opinion or related procedures, the situation is likely to improve (Jetz et al. 2011). Although we only considered a subset of all taxa (exclusion of e.g., plants, fish, insects) because of the global data availability the chosen taxa act as surrogates for the species community present. Still, comparisons to other methods that take other taxa into account have to be applied cautiously (e.g. land use).

5.4.4. Practical Implications

Required inventory information includes the amount of surface water and groundwater consumed at a certain location. For agriculture, for example, water requirement ratios (SI) for determining consumptive water use are available, and the values of Döll et al.(2011) can be applied for estimating the share of groundwater and surface water consumption. For the background system, global databases can be used (e.g. available for most crops from Pfister et al. (2011)), although the spatial information given is currently rather coarse. If the region of water consumption is unknown, global average CFs and related variability are available (**Error! Reference source not found.**), but they should be used with caution and include uncertainty estimates. Matching spatially resolved LCI and LCIA data is still a challenge and not yet included in any standard software, but its feasibility has already been demonstrated in research software and publications (Mutel et al. 2012). Therefore, we expect that it is only a matter of time until such analyses become standard practice.

Our approach is well-suited as a screening methodology on a global scale, but it will never replace modeling on local levels, where more details are included and analyzed. We suggest a tiered procedure, using the method of this paper for identifying potential hotspots in the life-cycle of products and processes which should be further investigated with local analysis.

Comparability with other currently existing LCA methodologies is difficult because of the different units used (PDF, species-eq). As mentioned, two wetlands with the same impact in PDF can have quite different species richness. Consequently, the absolute species loss is different and the overall CFs not the same (SI). Research is ongoing to also derive compatible CFs for other ecosystem impact categories, such as the impacts of land use and eutrophication.

Another reason for difficulties in comparisons with existing methodologies is that these do not take species' vulnerabilities into account. The introduction of TL give threatened species a weight that is up to 5 times larger than for a species of least concern. In ecology, it is common to use linear scales, such as the one proposed here (Purvis et al. 2005). In the ReCiPe project, value choices are made to include either threatened species only (egalitarian perspective) or all species with equal importance (individualist and hierarchist perspectives) (De Schryver et al. 2008). In our approach, threat status' of species are taken into account with VS, incorporating the current threat level and the geographical distribution. In addition, species that are not yet threatened but have a small distribution area and are therefore potentially more vulnerable if their habitat is damaged are included, while they are currently not considered if only threatened species are considered. ReCiPe (Goedkoop et al. 2009) takes into account 1.5 million terrestrial species and 100'000 freshwater species to calculate global species densities of $1.38 \cdot 10^{-8}$ 1/m^2 and $7.89 \cdot 10^{-10}$ 1/m^3. We take 10'110 species into account that specifically occur in wetlands. The calculated wetland species density varies between $2.4 \cdot 10^{-9}$ 1/m^2 and $4.8 \cdot 10^{-2}$ 1/m^2 with an average of $1.1 \cdot 10^{-4}$ 1/m^2, and is thus on average 4 orders of magnitude larger than the one from ReCiPe. ReCiPe (Goedkoop et al. 2009) converts PDF into the unit species·yr, but this unit is not directly comparable to our species-eq·yr, since we are targeting global extinction and include weighting according to the threat and distribution of species as well as the threat for habitat loss. As a result, straightforward comparisons are impossible at the moment. This shows the importance of taking local species richness (via richness maps) into account, specifically for wetlands, since they are hotspots of biodiversity. Species density is highly variable and cannot be represented by one global value.

5.4.5. Outlook

Impacts calculated are *only* impacts on Ramsar wetlands, keeping in mind the irregular distribution of these wetlands. Since most countries currently lack wetland inventories (Zedler et al. 2005), taking the Ramsar database (Ramsar Sites Information Service 2012) is ensuring currently the best global coverage of wetlands. We cover ~10% of all inland wetlands globally (134'216'253 ha in August 2012) (Ramsar Sites Information Service 2012), among them some of the most important ones. Improved coverage can be facilitated in the future by making use of national wetland inventories and applying the presented approach by using the maps of global species richness calculated here. This allows for deriving generic CFs for wetlands for a specific country or region. In order to more adequately represent biodiversity damages in the future, further research is planned for enhancing the comparability between our approach and other LCIA methodologies for the assessment of ecosystem damages and for a weighting scheme between taxa.

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6. Modeling the local biodiversity impacts of agricultural water use: case study of a wetland in the coastal arid area of Peru

Francesca Verones[1], Karin Bartl[2], Stephan Pfister[1,3], Ricardo Jiménez Vílchez[4], Stefanie Hellweg[1]

[1] ETH Zurich, Institute of Environmental Engineering, 8093 Zurich, Switzerland
[2] ETH Zurich, Institute of Agricultural Sciences - Animal Nutrition, 8092 Zurich, Switzerland
[3] Terra Nuova, Perú
[4] UC Santa Barbara, Bren School of Environmental Science & Management, Santa Barbara, CA 93106-5131, USA

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6.1. Introduction

Water withdrawal for irrigation constitutes 70% of all freshwater withdrawals (World Water Assessment Programme 2009). In northern Africa, India, the western United States and the South American Pacific coastline, more than 75% of the agricultural production depends on irrigation (World Water Assessment Programme 2009). Both groundwater and surface water are used for irrigation and there are numerous examples of aquifers being depleted and of reduced surface water levels in rivers due to agricultural water abstractions. The related impacts on ecosystems are manifold. In this work we concentrate on wetland ecosystems only. It is estimated that globally more than 50% of all wetlands have been lost, mostly due to agricultural drainage (OECD/IUCN 1996). Assessing the impacts and balancing wetland conservation and local socioeconomic development are thus crucial for the persistence of remaining wetlands.

Life cycle assessment (LCA) is a method for assessing the total environmental impact that is created through the life cycle of a product or process (ISO 2006). Approaches for assessing environmental impacts from water use on ecosystems are so far scarce. Pfister et al. (2009) have developed a global methodology for assessing the impact from freshwater consumption on ecosystems. However, assessing damage to a local scale is difficult with this method, since it is based on remote sensing data and does not distinguish surface water from groundwater consumption. An approach for assessing the local impacts of groundwater abstractions on terrestrial ecosystems has been published by van Zelm et al (2011). This method is specific to The Netherlands and the developed factors and the methodological approach cannot be easily applied to other regions, due to the complexity of the models and the large data demands. Thus, there is a need for additional regionalized studies regarding the impact of water use on ecosystems, as there is for assessing the applicability of global, water-use related approaches to regional levels.

We have selected a case study area in an arid region, where agricultural activities require irrigation and areas with native vegetation are restricted to small wetlands. The studied wetland, “Santa Rosa”, harbors a large variety of fauna and flora and belongs to the most biodiverse wetlands in the coastal region of Peru (Gobierno regional de Lima, nd; Jiménez Vilchéz et al. 2009). Agriculture constitutes both the most important water source, supplying the wetland with infiltrated irrigation water, and its largest menace (Ramirez et al. 2010), mainly due to contamination with nutrients and agrochemicals. Quality changes are not assessed in this paper.

The aims of this study were (a) to develop characterization factors (CFs) for the environmental impact of agricultural water use on natural wetland vegetation, taking into account local conditions, (b) to check the validity of existing generic characterization factors by comparing them to the local ones developed here, and (c) to provide a basis for environmental decision-making for agricultural management in the region. CFs are developed in terms of potentially disappeared fractions of species (PDF)(Goedkoop et al. 1999) in the wetland itself by differentiating the source and amount of water used and the consumptive shares (i.e., the amount of water that is evaporated, incorporated into products, or diverted to other watersheds or the sea (Falkenmark et al. 2004)). PDF is also used in other life cycle impact assessment (LCIA) methods for ecosystem damage (e.g., Goedkoop et al. 2009) and thus our factors can be compared with other approaches. The CF values are calculated in five steps: (1) Different scenarios and respective water balances are modeled. (2) The hydrological changes which can be expected in Santa Rosa for each scenario are calculated and used to determine fate factors. (3) The ecological effects of hydrological changes are modeled for each scenario. (4) Fate and effect factors for each scenario are combined to derive scenario-specific CF values. (5) CFs are generalized for application in LCA.

6.2. Methods

6.2.1. Case Study Area Description

The relevant area for the case study is the lower part of the watershed of the river Chancay-Huaral in the province of Huaral, Peru, at 77° W and 11° S. The size of the total watershed is 3095 km^2 (Bernabé et al. 2001). The case study area stretches from the river measuring station Santo Domingo to the coast (1245 km^2), covering the main aquifer extent and the area where most agricultural activities and population are located. Santo Domingo is the only measuring station for river flow in the valley and is used as the starting point for inflow of the water balance. The agricultural area in the lower valley covers about 260 km^2 with a total of 64 different crops (Gobierno Regional de Lima 2011b). Main crops grown in 2010 were yellow maize, mandarins, potatoes, and apples. Additionally, there are large numbers of cows and chicken. In higher regions outside the study area livestock is also present in smaller numbers, mostly on natural pastures. Almost the entire agricultural area in the lower valley is irrigated, and all crops except asparagus are irrigated with gravity and furrow irrigation. Asparagus is assumed to have a drip irrigation system (Sanchez Vigo 2005).

For calculating groundwater level changes only the area with an extensive aquifer is of importance (569 km^2). More details on the region are given in the Supporting Information (SI), sections S2, S3, and S5.

The wetland Santa Rosa has a size of 36 ha of which 10 ha are open water (Alcántara Medrano et al. 2005) with a maximum depth of 3 m (Martinez 2010). It is the most important area with natural vegetation and wildlife in Chancay-Huaral. Moreover, compared with other coastal Peruvian wetlands, Santa Rosa is particularly species rich (Gobierno regional de Lima, nd), harboring 51 plant and 73 bird species (Jiménez Vilchéz et al. 2009). The wetland is almost exclusively fed by exfiltrating groundwater and has no outflow.

6.2.2. Establishing the Water Balance

A water balance is determined through its inputs, outputs, and storage within the system for a certain time period. It is always specific to one region, but principles and procedures remain the same everywhere. The steady-state water balance equation for the project area is shown in Equation 6.1.

$$\mathrm{Q} = \mathrm{I} + \mathrm{P} + \mathrm{GWI} - \mathrm{Ex} - \mathrm{ET} + \mathrm{GWR_S} + \mathrm{GWR_I} + \mathrm{RF} - \mathrm{RWW} - \mathrm{GWW} \quad \textbf{Equation 6.1}$$

where Q is the river flow to the Sea, I is the river inflow into the area, P is precipitation, GWI is groundwater inflow to the area, and GWR is groundwater recharge within the area from river seepage (subscript S) and excess irrigation (subscript I) (see also Figure 6.1 and section S1 of the SI for flows and acronyms). Ex is the exfiltration of groundwater at the coast, ET is evapotranspiration, and RF is the return flow to the river from domestic and industrial groundwater use, such as discharged wastewater. RWW is water withdrawal from the river, and GWW is water withdrawal from groundwater. While measurements can provide data for precipitation and river flows, groundwater flow interactions are complex and require modeling.

6.2.3. Parameterization of the Water Balance

Precipitation (P) was adopted from New et al. (2002). The river Chancay-Huaral is fed by precipitation and snowmelt which is stored in 33 lakes in the Andes and flows from 4800 m above sea level into the Pacific Ocean. There are no glaciers in the basin which could contribute to river flow (León Luna 2011). Flow changes related to glacier melt due to climate change will thus not affect the watershed. The stakeholders of the region demonstrate farsightedness by controlling the outflow of most lakes, a measure which is proposed for adaptation to climate change in the Andes (Bradley et al. 2006). River flows were available on a daily basis from 1960 to 2008 from the measuring station Santo Domingo (ALA Chancay-Huaral, 2010). For the water balance the annual long-term mean of 16.1 m^3/s was used as inflow (I). The groundwater inflow into the area (GWI) and the groundwater recharge from the river within the lower valley (GWR_S) were estimated with Philip's formula (Jury et al. 1993).

In 2010 more than 99% of the total irrigation water of 245 million m^3/a was withdrawn from the river (Gobierno regional de Lima 2011a), the rest was abstracted from the aquifer. The area-specific amount of water applied to each crop was calculated as a share of the total irrigation water according to the middle crop coefficient (K_c) of each plant, i.e. plants with a higher K_c are allocated a larger amount of water per hectare than those with a lower K_c.

For livestock, 0.7×10^6 m^3 water/a were required, with a groundwater share of 98% (Gobierno Regional de Lima 2011). Additional withdrawals for domestic, industrial, and mining purposes amounted to 4.7×10^6 m^3/a from the river and 0.6×10^6 m^3/a from groundwater (Gobierno regional de Lima 2011a). Water withdrawals for livestock were assumed to be entirely lost from the system, since the largest part is used for drinking and cleaning the stables, ending up in the milk or being evaporated. We assumed that 20% of the amount withdrawn for mining, industrial and domestic purposes was lost, while 80% returned to the river (e.g., as sewage). This is based on information for return flows from domestic and industrial uses, which vary between 60% (Seckler et al. 1998) and 90% (World Water Council, nd) of total water withdrawal.

Evaporation and evapotranspiration (ET) take place from bare soil, water bodies, and vegetation in the lower valley, and all of them require adequate calculation procedures. The actual evaporation from bare, unproductive soil was calculated with the formula of Turc (Koch, 2003), while the potential evaporation for the open water surface was estimated with the formula of Thornthwaite (Koch, 2003). Evapotranspiration from the crops was calculated according to the FAO Penman-Monteith method (Allen et al. 1998). For this, crop coefficients (K_c, indicating the difference in evapotranspiration between reference surface and respective crop) (Allen et al. 1998) and stress factors (K_s, accounting for water use efficiency and thus water losses, which will not reach the plant) were estimated. The following parameters are required for calculating evapotranspiration. The mean annual air temperature in Lima, used as proxy for Chancay-Huaral (about 60 km north of Lima), is 19.5 °C and varies between 14 °C and 28 °C (METEOTEST 2011). The average precipitation in the lower valley is 34 mm/a (New et al. 2002). Wind speed and dew point temperature were taken from METEOTEST (2011).

An important component in the water balance was the infiltration of excess irrigation water (GWR_I), which was estimated for each crop as the difference between the irrigation water applied and the crop evapotranspiration. Precipitation was assumed to be insignificant for groundwater recharge in the lower valley.

The groundwater flow to the sea (Ex) was estimated over the whole coastline by assuming a mean aquifer thickness of 25 m. For further details on parametrization and inputs to the water balance, see SI, sections S3-S9.

6.2.4. Relation between Water Level and Environmental Habitat Quality

The dominant plant species are *Schoenoplectus americanus, Typha dominguensis*, *Distichlis spicata*, *Paspalum vaginatum*, and *Sporobolus* species (Jiménez Vilchéz et al. 2009, Alcántara Medrano et al. 2005). Thirty-seven species of the flora in Santa Rosa are terrestrial and 14 species are aquatic (floating, submerged)(Jiménez Vilchéz et al. 2009). The terrestrial species are assumed to grow from the edge of the wetland to a water depth of 1.5 m. The area from 1.5 to 3 m is considered as the aquatic plant species' habitat.

It is assumed that plants can be indicators for the suitability of an impacted wetland for birds. A change in vegetation diversity is likely to have an effect on birds as well, since "mobile" species like birds are generally dependent on vegetation and surface water bodies (Hatton et al. 1998). In contrast to the wetland, the river has a low biodiversity. Four phytoplankton species, one zooplankton species, and three benthic macroinvertebrates were registered, but not a single fish species (CESEL Ingenieros 2010). However, the biodiversity of the river was assessed only once at one point in the lower valley and does not enter the calculation of any CF.

6.2.5.Scenario Development

Scenarios were developed for the year 2020 and compared to the base year 2010. The base year is 2010, since data for the crops, agricultural area, and water withdrawals were available for that year.

All proposed scenarios were developed and discussed in a workshop with local stakeholders, including farmers, biologists, hydrologists, and local authorities, held in Chancay in February 2011, which was crucial for the design of the scenarios. It turned out that land is the limiting factor in the lower valley and a further expansion of agricultural area is not an option (León Luna 2011). However, crops in existing areas may change at any time. During the workshop several crops that are likely to expand in cultivated areas during the next years were identified, as shown in Table 6.1. The changes projected in the scenarios are based on the development of the respective crops during the last 10 years, assuming past growth rates to continue until 2020. Changes in water use in the scenarios are thus not more pronounced than changes in the past. Mandarins are very important for the region and their production area has been rapidly increasing recently. This was not the case for asparagus and strawberries, but local stakeholders expect that the cultivation of these crops will increase as they are very cost-effective and demand for export is high. Since no further water rights will be allocated (Le'on Luna 2011), stakeholders try to increase the water use efficiency by adapting more water efficient irrigation techniques for perennial crops. Therefore, we considered in the "increase drip-irrigation" scenario a shift to more drip irrigation on areas with crop changes. Experiments with more water efficient irrigation techniques are undertaken at the INIA (Instituto Nacional de Investigación Agraria) and a shift to other techniques seems realistic. With rather strict water rights, the region seems able to prevent the depletion of water resources, which has, for example, occurred in Ica, the major asparagus producing region of Peru (Hepworth et al. 2010). The calculated change in water use in the scenarios is incomplete because it only incorporates crop development and the water use of other sectors has not been varied. In the "multicrop" scenario, an unchanged continuation of the growing or decreasing trend of the last 10 years of the most common crops as well as of asparagus and strawberries, identified as crops with a high future potential due to economic reasons (53.3% of total crop area in 2010) (gobierno regional de Lima 2011b), is considered. In order to investigate the importance of the water source for water-use related impacts, a hypothetical scenario for mandarin production with more groundwater irrigation was calculated.

Table 6.1: Overview of the Scenarios, the Increased Crop Areas (in hectares [ha]) and the Irrigation Water Sources (RW=river water, GW=groundwater)[a]

scenario name	increase asparagus	increase mandarin	increase strawberry	increase drip-irrigation	increase multicrop	increase mandarin, more GW	no irrigation
crops in 2010 (cultivated area, ha)	asparagus (161)	mandarin (3415)	strawberry (580)	avocado (1676) asparagus (161) mandarin (3415) strawberry (580)	avocado (1676) asparagus (161) mandarin (3415) strawberry (580) yellow maize(4660) potato (3212)	mandarin (3415)	no crops
crops in 2020 (cultivated area, ha)	asparagus (1612)	mandarin (5314)	strawberry (976)	avocado (2397) asparagus (1612) mandarin (5314) strawberry (976)	avocado (2090) asparagus (994) mandarin (4506) strawberry (807) yellow maize (6344) potato (5439)	mandarin (5314)	no crops
irrigation sources	RW (99%) GW (1%)	RW (99%) GW (1%)	RW (99%) GW (1%)	RW (99%) GW (1%)	RW (99%) GW (1%)	RW (75%) GW (25%)	no irrigation

[a] In the scenario "increase drip-irrigation", drip irrigation is assumed to be present on all areas with crop changes, while unchanged areas retain the original furrow irrigation system.

Areal increases imply that areas of other crops need to be reduced. Crops that showed a decreasing trend over the last 10 years (Gobierno Regional de Lima 2011b) are assumed to proportionally decrease further, in order to allow an expansion of other crops in the scenarios (Table 6.1). Since all crops have individual water requirements, the amount of irrigation water needed, and hence the water balance and groundwater level, all change. All nonconsumptively used groundwater is assumed to return to the aquifer and can be neglected. It was supposed that, except for the "increase drip-irrigation" scenario, the irrigation system will stay the same as in 2010. Crop water consumption of the scenarios presented are calculated with crop coefficient values (K_c) from the FAO (Allen et al. 1998), since the majority of required K_c values were available from this source. For crops with no indicated K_c value, the value of a similar crop was used. Since the default K_s values ($K_{s,1}$), which are also used for calculating GWR_I, are widely varying, it was decided to apply a mean value of 0.5. Details about all crops are given in the SI (section S5) and results for scenarios with other K_c and K_s combinations are shown in the SI (section S13).

6.2.6. Calculation of Characterization Factors for the Scenarios

As commonly done in LCA, characterization factors (CF, Equation 6.2) for each scenario x are calculated as the product of a fate factor (FF, Equation 6.3) and an effect factor (EF, Equation 6.4 and 6.5).

$$CF_{x,wetland} = EF_x \cdot FF = (EF_{x,terr} + EF_{x,aq}) \cdot FF$$ **Equation 6.2**

$$FF = \frac{ISR_x - ISR_{base\ year}}{(ARWW_x - ET_{c,x}) - (ARWW_{base\ year} - ET_{c,base\ year})}$$

Equation 6.3

$$EF_{x,terr} = \frac{1 \cdot (A_{terr,base\ year} - A_{terr,x}) + \left(1 - \frac{S_{terr,x}}{S_{terr,base\ year}}\right) \cdot A_{terr,x}}{ISR_x - ISR_{base\ year}}$$

Equation 6.4

$$EF_{x,aq} = \frac{1 \cdot (A_{aq,base\ year} - A_{aq,x}) + \left(1 - \frac{S_{aq,x}}{S_{aq,base\ year}}\right) \cdot A_{aq,x}}{ISR_x - ISR_{base\ year}}$$

Equation 6.5

The agricultural river water withdrawal and evapotranspiration (consumptive water use) per kilogram of crop are collected in the inventory. Surface water (and groundwater) withdrawals are already today present in many life cycle inventory databases, such as ecoinvent (Ecoinvent Centre, 2012). Crop evapotranspiration can be estimated with a variety of formulas (e.g., Penman-Monteith) or tools (e.g., CropWat (FAO, 2012)), if necessary. For 160 crops this data is also available from Pfister et al. (2011). The net balance of these inventory flows is then multiplied with the CF to arrive at impacts/benefits.

The fate factor (Equation 6.3) reflects the relation between infiltration into the wetland (ISR) and the net groundwater flow, i.e., groundwater recharge from irrigation surplus water (inventory flow with positive sign) or groundwater consumption (negative sign). Water withdrawals from and recharge to an aquifer lead to changes in the groundwater table. Groundwater recharge is calculated as the difference of agricultural river water withdrawal (ARWW) and total crop evapotranspiration (ET_c). On the basis of infiltration rates and the corresponding infiltration area, the amount of water annually infiltrating into the wetland (ISR_x) is determined for each scenario x as a function of the groundwater level. Since the soil in the project region is sandy with high permeability, we assumed that the whole aquifer area is homogeneous and affected evenly. As Santa Rosa is exclusively groundwater-fed, the amount of infiltrating water determines the change in water level and surface area of the wetland (A_x). The resulting groundwater levels for the scenarios are then compared to the base year.

The effect factor (Equations 6.4 and 6.5) quantifies the potential disappearance of species (PDF) due to changes in wetland infiltration for the terrestrial (index terr) and aquatic (index aq) ecosystem, thus quantifying a change in biodiversity. The wetland is modeled as a circular cone with a maximum depth of 3 m. For the new wetland area the size of the aquatic and terrestrial plant zones and corresponding species numbers are calculated. $A_{terr,x}$ [m^2] and $A_{aq,x}$ (m^2) are in each scenario x the areas of the terrestrial and aquatic zone, respectively. The difference ($A_{base\ year}$ –A_x) is for both zones the area of wetland which is lost or gained in each scenario. $S_{base\ year}$ is the original species number in Santa Rosa, and S_x is the number of species in the remaining area for both terrestrial and aquatic plants (Liménez Vilchéz et al. 2009). S_x is calculated with $S_{base\ year}$ and the corresponding wetland areas (Equation 6.) for both ecosystems

$$S_x = S_{base\ year} \cdot \left(\frac{A_x}{A_{base\ year}}\right)^z$$

Equation 6.6

where z indicates the slope of the species-area relationship and is different for terrestrial and aquatic species (SI, section 10). The numerators of equations 6.4 and 6.5 were inspired by the framework set up by Eco-Indicator 99 for land use (Goedkoop et al. 1999). The left part of the numerator describes lost wetland area, where 100% of species disappear, i.e., PDF=1. The right part targets potential loss of species in the remaining area, i.e., PDF=1-(S_x/$S_{base\ year}$).

The CF for the wetland (Equation 6.) indicates the benefit of river water irrigation on Santa Rosa. Note that this does not include the impact on the river system, for which a separate CF, denoted CF_{river} here, is needed. The infiltration of surplus irrigation water from surface sources leads to groundwater recharge and a benefit for Santa Rosa. Consumptive groundwater use diminishes the water sources for Santa Rosa and has the opposite sign in the inventory analysis than groundwater recharge, hence resulting in an environmental damage. All groundwater withdrawals that are not consumptive infiltrate back into the aquifer in the case study region and are thus of no relevance to the impact (see example in Table 6.1).

6.2.7. Calculation of Total Environmental Impacts

Impacts on Santa Rosa can be calculated by multiplying the CF with the net infiltration into or consumption of the groundwater. In the inventory, the agricultural river water use (ARWW) and consumed amount of water per kilogram of crop (ET_c) should be reported. In order to calculate the net impact/benefit of river water use on Santa Rosa, the difference of the river water withdrawn and the consumed amount of water is multiplied with $CF_{wetland}$. For the impact of water withdrawal on the river's biodiversity CF_{river} according to Hanafiah et al. (2011) is used, resulting in impacts in terms of water volume and not surface area. For calculating the impact of water consumption associated with groundwater use, $CF_{x,wetland}$ is multiplied with the consumptive fraction of groundwater withdrawal. Since we are assessing wetland area loss and not volume loss, the unit (PDF·m^2·yr) is directly comparable with the units used in land use assessment in LCA (Goedkoop et al. 1999). The calculation of these total environmental impacts is illustrated in Table 6.3 for the case of asparagus and mandarin production for different irrigation techniques and irrigation water sources.

6.2.8. Sensitivity Analysis

Crop coefficient (K_c) values given in the literature differ and we used additional K_c (denoted $K_{c,ElRiego}$, mostly from southern Spain (el Riego.com, 2011)) for the sensitivity analysis. Additionally, the sensitivity of the results to an alternative stress factor ($K_{s,2}$) and the estimation of groundwater recharge was tested with an empirical formula from Turc (1954). Thus, each scenario has six subscenarios combining three different infiltration amounts and two alternative crop coefficients. For testing CF sensitivity to different base cases, a second base case, "no irrigation", was modeled, assuming an absence of irrigated agriculture. Additionally, we tested the sensitivity of the CF to the z-values.

6.3. Results

3.3.1. Water Balance

The water balance is schematically shown in Figure 6.1 with results based on the mean $K_{s,1}$ value and K_c values from FAO (allen et al. 1998). The mean annual discharge to the sea is 382 × 10^6 m^3/a. This is the largest outflow of the system, followed by crop evapotranspiration.

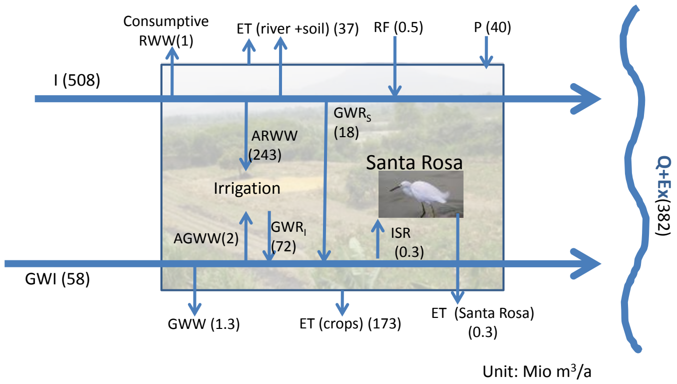

Figure 6.1: Scheme of the water balance for the lower valley of Chancay-Huaral with the respective magnitude of the flows. River and aquifer have inflows from the upper valley. The arrows "consumptive RWW" and "GWW" encompass water withdrawals for industrial, livestock, mining, and domestic purposes. Acronyms are introduced in the main text and Equation 6., 6.4 and 6.5 and are summarized in SI (section S1).

3.3.2. Scenario Results

Except for "increase asparagus", all scenarios require less irrigation water and all scenarios lead to a decrease of wetland water level and area, because reduced river water withdrawal for irrigation decreases groundwater recharge (Table 6.2). Negative impacts and CF highlight a benefit for the wetland's area and biodiversity, as shown for all scenarios compared to base case "no irrigation". The additional water withdrawals for irrigation lead in these cases to advantages for Santa Rosa. The CF_xs for all scenarios are very similar in value (Table 6.2) showing that they are not very specific to the scenarios. Only the CF for “increase drip-irrigation” is different. Therefore, it is reasonable to use the median CF for all subsequent LCA studies for this region. This facilitates the assessment because it means that we can assign an impact to the amount (m^3) of water used, irrespective of the functional unit, without consideration of regional agricultural development (crop choice, irrigation development). The generalized, median value for scenarios with base case “2010” for $CF_{wetland}$ is -3.19×10^{-4} PDF·m²·yr/m³.

Table 6.2: Results for the Scenarios for Santa Rosa with $K_{s,1}$ and K_c Values from FAO with Both Base Case “2010” and “no irrigation”. [a]

base case "2010"	increase asparagus	increase mandarin	increase strawberry	increase drip-irrigation	increase multicrop	increase mandarin, more GW	no irrigation
total AWW (10^6 m³/a)	246.1	239.5	244.9	217.4	241.6	239.5	0
change water level SR (mm)	0.06	-3	-0.4	-18	-5	-60	-70
area change Santa Rosa (%)	0.004	-0.2	-0.03	-1.21	-0.3	-4.1	-4.8
modeled impact (PDF·m²·yr)	-19.7	785.5	117.7	5'556.6	1'372.2	18'779.0	21'797.2
$CF_{x,wetland}$ (PDF·m²·yr/m³)	-3.19×10^{-4}	-3.19×10^{-4}	-3.19×10^{-4}	-4.45×10^{-4}	-3.18×10^{-4}	-3.11×10^{-4}	-3.10×10^{-4}
base case "no irrigation"	**increase asparagus**	**increase mandarin**	**increase strawberry**	**increase drip-irrigation**	**increase multicrop**	**increase mandarin, more GW**	**2010**
total AWW (10^6 m³/a)	246.1	239.5	244.9	217.4	241.6	239.5	245.05
change water level SR (m)	0.077	0.074	0.077	0.070	0.072	0.011	0.077
area change Santa Rosa (%)	5.08	4.89	5.04	4.62	4.75	0.70	5.07
modeled impact (PDF·m²·yr)	24162.7	-23270.7	24010.5	-22019.9	-22620.7	3342.0	24140.9
$CF_{x,wetland}$ (PDF·m²·yr/m³)	-3.44×10^{-4}	-3.43×10^{-4}	-3.44×10^{-4}	-3.81×10^{-4}	-3.43×10^{-4}	-3.35×10^{-4}	-3.44×10^{-4}

[a] AWW stands for agricultural water withdrawal and SR for Santa Rosa. The modeled impact presented is the impact of both water withdrawal and consumption on the terrestrial and aquatic part of Santa Rosa. Note that the scenario in the last column is different for the two base cases.

6.3.3. Application of CFs in LCA (illustrating example)

Table 6.3 shows an example for the production of 1 kg of asparagus and mandarins, illustrating the use of the developed CF. Mandarins are assumed to be irrigated with traditional furrow irrigation systems, once with river water (RW) and once with groundwater (GW). Asparagus is assumed to be always irrigated with river water, once with furrow irrigation and once with drip irrigation. It is assumed that with drip irrigation no water will infiltrate into the aquifer and thus everything will be consumed. In all examples except for “mandarin, GW” more water is abstracted from the river than consumed and hence groundwater is recharged (e.g., by 0.1 m³ in “mandarin, RW”). In the “mandarin, GW” case, the same volume (0.4 m³) is withdrawn from the groundwater of which 0.3 m³ is lost through evapotranspiration. The 0.1 m³ left reinfiltrates to the groundwater, without an effect on Santa Rosa. Hence, the complete groundwater withdrawal itself is not relevant. The mean $CF_{wetland}$ is negative, meaning that a net inflow into the groundwater ($ARWW-ET_c$), as calculated from the inventory flows, reveals benefits for Santa Rosa (impact with negative sign), while a negative net balance (i.e., groundwater consumption) leads to damage (impact with positive sign).

Table 6.3: Example for the Production of 1 kg of Asparagus or Mandarin.[a]

	asparagus		mandarin	
	furrow	drip	RW	GW
withdrawal surface water (ARWW)	1.9	0.9	0.4	0
withdrawal groundwater	0	0	0	0.4
net freshwater consumption (ETc)	1.4	0.9	0.3	0.3
net flow into groundwater	0.5	0	0.1	-0.3
impacts on Santa Rosa [b] (PDF·m^2·yr/kg)	-1.59×10^{-4}	0	-3.19×10^{-5}	9.56×10^{-5}
impacts on the river [c] (PDF·m^3·yr/kg)	5.61×10^{-4} [d]	2.66×10^{-4} [d]	1.18×10^{-4} [d]	0.00 [d]

[a] Asparagus is irrigated with river water, once with furrow irrigation and once with drip-irrigation. Mandarins are assumed to be irrigated traditionally with furrow irrigation, once with river water (RW) and once with groundwater (GW). The impact is calculated via the net balance between withdrawal and consumption (subtracting consumptive water use from withdrawals). Note that the impact on the river for the same volume shows a different impact for a different ecosystem, since it is focusing on fish species only and is calculated per m^3 and not per m^2. [b] negative=benefit for Santa Rosa, positive=damage for Santa Rosa. [c] On the basis of Hanafiah et al.[33] [d] withdrawn from river → consumptive use for river

The impact on the river is calculated here for the amount of water withdrawn in each case. From the river's perspective, all of this volume is lost (i.e., consumptive), since principally none is returned to the river. The average potential impact on the river's fish biodiversity, estimated according to Hanafiah et al. (2011), is 2.95×10^{-4} PDF·m^3·yr/kg and should not be directly compared with $CF_{wetland}$ because of different units. Hence, it should be considered as a separate impact category in a full LCA.

We compared the water-related impacts for asparagus and mandarins, as presented in Table 6.3, with other impact categories adapted from Stoessel et al. (2012), in units of ReCiPe points (Goedkoop et al. 2009). ReCiPe is an established LCIA method quantifying environmental impacts for 16 different impact categories. Water-related impacts are only responsible for a relatively small share of the total ecosystem impact, which is dominated by climate change and agricultural land occupation. However, the irrigation system and the source of irrigation water have an influence on the total water-use-related impacts (SI, section S14).

6.3.4. Sensitivity Analysis

The streamflow to the sea varies between 11.4 and 14.1 m^3/s for all water balances with different K_c and K_s values. This variation is caused by differences in crop coefficients, leading to different evapotranspiration and groundwater recharge. The contribution of different K_s values to the changes of the water balances is larger than that of K_c values. Despite the sensitivity of the water balance to variations in both crop coefficients, the CFs sensitivities to changes in crop coefficients were generally small, except for changes in irrigation systems. Changes in the CFs for alterations in the z-values are small, considering the change of the z-value itself, namely around 10% for increasing the original z-values to a maximal value of 0.34 and 16% for decreasing them to a minimal level of 0.03 (SI, sections S10, S11, S13).

6.4. Discussion

6.4.1. Impacts

The modeled impact on the aquatic ecosystem in Santa Rosa was more pronounced than on the terrestrial ecosystem, since the area losses of open water body are relatively larger than that of terrestrial area. The different z-value of aquatic and terrestrial species also leads to stronger modeled impacts on aquatic species than on terrestrial ones.

The CF from the top-down approach of Pfister et al. (2009) for the region investigated is 2.1 $PDF \cdot m^2 \cdot yr/m^3$, constituting one of the highest values globally. Commonly it is assumed that more water use always creates damage. However, the opposite is true for Santa Rosa when river water is used. Santa Rosa is special since it is exclusively fed by groundwater, which is enriched by surplus irrigation water as the soil is sandy, facilitating rapid infiltration. Less river water irrigation means less infiltration and thus creates ecological damage. The top-down results are not able to catch these local peculiarities, although they may be well-suited for a broader and large-scale assessment. If we broaden the view and include the river, we see conflicting results. There is an impact on the river from water withdrawals, and it would be beneficial for the river and potentially the coastal zone to decrease the amount of water withdrawn. However, extracting more groundwater would lead to larger impacts on the wetland. These conflicting results between ecosystems show the importance of distinguishing between the different sources and types of water use. It also shows the importance of balancing impacts between different ecosystems with different biodiversity. This is manageable on a small scale but becomes increasingly difficult at larger scales or when analyzing supply chains of products, thus accentuating the need for local studies alongside generalized top-down approaches.

The CF combines total species loss on the lost area and partial species loss on the remaining area, as commonly done in LCA according to the Eco-Indicator 99 framework. From an ecological perspective this is questionable. We adopted this concept nevertheless, taking into account the compatibility of our approach to other LCA methodologies.

A direct comparison of the CF derived in this paper with the CF for the river according to Hanafiah et al. (2011) is not possible since the latter is based on fish species diversity, while the developed CF values for Santa Rosa refer to flora. CF_{river} was estimated according to Hanafiah et al. (2011) although no fish species were recorded in the river (CESEL Ingenieros 2010). Whether there used to be fish in the river and they disappeared, e.g., due to anthropogenic activities, is unknown. A direct comparison of our impacts with the complete LCA assessment of Stoessel et al. (2012) faces similar drawbacks. The global value for agricultural land occupation is from central Europe (Stoessel et al. 2012) and thus hard to compare with coastal Peruvian conditions, which are naturally (in contrast to central Europe) desertlike. A direct comparison of impacts is often not meaningful because assessments that are able to reflect local conditions are still missing in various parts of LCIA, such as land and water use, or are applied outside of their regional scope (e.g., ReCiPe was developed for Europe).

6.4.2. Sensitivity and Uncertainties

Since there is no measuring station below Santo Domingo, it is not possible to assess the accuracy of the river flow estimate. However, the runoff seems to be in a realistic range according to local experts (Workshop in Chancay with local stakeholders, 2011).

The precipitation for the lower valley from METEOTEST (2011), based on mean monthly values from 1996 to 2005, is 37 mm/a and matches the value estimated from SENAMHI (2010). For the potential evapotranspiration, estimates range from around 1000 to 565 mm/a (Gutierrez Pusoni, 2010; INRENA, 1999) for one station in the lower valley for two years in the 1960s and 1970s. The estimate with Thornthwaite (893 mm) lies between these two values and seems reasonable. Consumptive evaporation losses from the irrigation systems, as discussed in Faist Emmenegger et al. (2011), are only partially taken into account in the sensitivity analysis, as the application of different K_s values makes the estimated evapotranspiration vary by 29%.

Base cases and K_s and K_c values influence the water balance, the calculation of groundwater level, and impacts. The differences between the scenarios with different K_s and K_c values are small, except for the empirically derived Turc formula. However, this formula is questionable for groundwater recharge, since it neglects many factors influencing infiltration, such as soil properties. The difference between the median CFs with different base cases is only 8%, and the CFs are akin for the different scenarios indicating high linearity of the system. Consequently, the generalized values are applicable to the whole region and different crops with an acceptable uncertainty.

With the base case "no irrigation" and different K_s and K_c values, the increase in wetland area is predicted to be between 0.7% and 9.9%. This matches well with observations in the field that Santa Rosa has slowly, but constantly, been growing during the last decades. A comparison between aerial pictures from 1961 and 2006 revealed an area increase between 5.5% and 8.9% (SI, section S13).

An important question concerns the reference for an assessment of Santa Rosa. Since Santa Rosa has been growing, it has actually changed from its natural state (which is often suggested as reference) and the question remains which part of Santa Rosa deserves protection: only the natural part or the wetland as it is today with increased area due to irrigation. Similar discussions exist in land use assessments, where, for example, poor grassland is more biodiverse than forest, often taken as the potential natural vegetation (Koellner et al. in press), .

If birds were included in the impact calculation, the z-value would be different. However, especially migratory birds are difficult to consider in an impact assessment, since damages on one species (e.g., fewer individuals arriving) can be caused by different impacts elsewhere, showing that the ecological value of a region can be larger than the region itself.

An issue for the wetland today is also the water quality, since input of nutrients from agriculture and sewage lead to eutrophication, which is not examined in this work.

6.5. Practical Implications

Generic or top-down assessment methodologies like that of Pfister et al. (2009) are good tools for a first screening assessment of a larger region. However, as our results have shown, the outcomes of these approaches may not be relevant at a local level. This highlights the importance and need for further regionalization in LCA and water footprinting, in order to improve the accuracy and applicability of LCA on local levels. It is suggested that the presented approach should be used for refinement of top-down results, in order to obtain locally valid figures and to capture the peculiarities and differences between surface water and groundwater of a region with a manageable procedure.

The relevance of proactive assessments is illustrated by the example of Ica, a nearby valley of Chancay-Huaral, where asparagus production contributed heavily to aquifer depletion and increased water scarcity (Hepworth et al. 2010). In order to avoid such consequences, it is necessary to have a decision-support tool helping to assess management decisions. Our study provides generalized and scenario-specific CF factors for Chancay-Huaral which can be applied for evaluating different management options and crop choices. The drawn conclusions will be valuable for local decision-makers and international consumers in a North-South context. This involves the production of commodities in the “global South” (developing and emerging economies) while the consumption takes place in the “global North” (industrialized economies). Local decision-makers should be supported for planning future agricultural development and for the simultaneous protection of ecosystems such as wetlands, while international consumers should become aware of the aggregated impacts of their consumption. A classic example from Peru is the production of asparagus, which is mostly exported and not consumed locally.

6.6. Outlook

The narrow geographical scope of the study prevents an application of the presented CFs on a worldwide level. However, the methodological approach is transferable to any other region due to the relative simplicity of the approach and low requirements for input data. Efforts are currently undertaken to provide characterization factors for the most important wetlands/watersheds on a global level in order to be able to consistently calculate impacts from water use on wetlands in a variety of geographical regions.

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7. Biodiversity impacts from salinity increase in a coastal wetland

Maria José Amores[1], Francesca Verones[2], Catherine Raptis[2], Ronnie Juraske[2], Stephan Pfister[2], Franziska Stoessel[2], Assumpció Antón [1,3], Francesc Castells[1], Stefanie Hellweg[2]

[1]Universitat Rovira i Virgili, Departament d'Enginyeria Química, 26 Avda Països Catalans, Tarragona, Spain 43007
[2] ETH Zurich, Institute of Environmental Engineering, 8093 Zurich, Switzerland
[3] Institute for Food and Agricultural Research and Technology (IRTA), S/N carretera de Cabrils, Cabrils, Barcelona, Spain 8348

Note: This chapter has not yet been submitted to a scientific journal and is therefore to be treated with confidentiality.

7.1. Introduction

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

Wetlands are important and vulnerable ecosystems. More than 50% of all wetlands worldwide were destroyed during the twentieth century, more than 60% in Spain and Greece, and more than 70% in Italy (Hollis 1992), mostly due to agricultural drainage (OECD/IUCN 1996). During the last decades, the use of greenhouses instead of traditional farming systems has often been accompanied by additional groundwater withdrawal for irrigation (Rodríguez-Rodríguez, M et al. 2011). In coastal areas, over-pumping of aquifers leads to sea water intrusions, thus increasing the salinity in aquifers. At the same time, coastal wetlands, where fresh water and salt water are often mixed, are among the most productive, valuable, and yet most threatened ecosystems in the world (Agardy et al. 2005). Coastal wetlands in arid and semi-arid zones experience periods of increasing salinity as a consequence of high evaporative conditions, variability of inflows, their proximity to the sea but also due to impacts of human pressures (Jolly et al. 2008), such as overpumping of aquifers. Due to the inflow of salty water, coastal wetlands might experience an increase in salinity, which could potentially be detrimental for the wetland's specific ecological system.

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

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

In order to develop a methodology for salinity impacts, we selected a coastal wetland in Spain called "Albufera de Adra" as an exploratory case study in order to learn (since water-related impacts are quite complex) how to include the impacts from salinity increases due to water consumption. It is located in a semi-arid region in Almería (South-East of Spain), where agricultural activities require substantial irrigation and areas with native vegetation and fauna are restricted to small patches and wetlands (Molina Vázquez 2006). The aims of this study were (a) to develop a characterization factor (CF) in terms of potentially affected fraction of species (PAF) (Pennington et al. 2004) for salinity impacts based on a new effect factor and a locally specific new fate factor, and (b) to apply it to a local case study and compare the impact of salinity with commonly used ecosystem quality impact categories.

7.2. Methods

7.2.1. Description of the wetland Albufera de Adra

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

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

Figure 4.1: Albufera de Adra (Spain) composed by the larger, coastal Nueva Lagoon and inland Honda Lagoon enclosed in the blue circle. The red line delimits the agricultural area of the study which occupies 899.2 ha(Fernández Sierra et al. 2004) of greenhouses area. The main economic activity in that area is protected horticulture being from the maximum production per ha to the minimum: tomato, cucumber, aubergine, watermelon, pepper, zucchini, melon and green bean(Andalucia, 2012; Andalucia, 2011; Tolón Becerra et al., 2010). (*Source Google earth (2011)*).

7.2.2. Developing the Characterization Factor

As commonly done in LCIA, Characterization Factors (CF, Equation 7.1) are calculated as the product of a Fate Factor (FF, Equation 7.2) and an Effect Factor (EF, Equation 7.9). The FF models the salinity increase in the wetland due to increased water consumption rate (in $g \cdot l^{-1} \cdot m^3 \cdot yr \cdot m^{-3}$) and the EF relates an ecological damage to the increased salinity measured as Potentially Affected Fraction (PAF) of species (in $PAF \cdot l \cdot g^{-1}$). The units from the characterization factor are based on g/l which comes from salinity, m^3 in numerator comes from the units of Nueva volume, the period time is for one year (9 wet months, 3 dry months) and the m3 in denominator comes from ET_{crop}. The CF for the salinity impact in this coastal wetland is therefore defined as the change in PAF of species due to a change in groundwater consumption, which is affecting the salinity content via altered amounts of groundwater and seawater infiltration into the wetland. This can be translated into the effect per m^3 of water consumed.

$$CF = FF \cdot EF \; [\cdot \, \mathrm{PAF} \cdot \mathrm{yr} \, \cdot]$$ **Equation 7.1**

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

7.2.3. Fate Factor

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

$$FF = [\mathrm{g} \cdot\cdot \, \mathrm{yr} \, \cdot]$$ **Equation 2**

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

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

Equation 7.2 is calculated for three time spans, 1983-2003, 2003-2008 and1983-2008. We focused on these years because between 2003 and 2008 the salinity constantly increased from 4.5 to 7.5 g/l(Rodríguez-Rodríguez, M et al. 2011), and changes in irrigation techniques occurred, along with a trend of greenhouse extension out of the delta valley. We took 1983 as a year with a situation as natural as possible in order to compare to 2003 and 2008 since from 1975 until 1983 the salinity remained constant in Albufera de Adra (Pulido et al. 1988).

Table 1: Constant parameters in Nueva Lagoon for the years 1983, 2003 and 2008. Climatic parameters were provided by Adra weather station(IFAPA 2012) and water bodies, salinities and morphometric characteristics by existing literature (Rodríguez-Rodríguez et al. 2011).

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

	groundwater(Molina Vázquez 2006)					groundwater in the Adra River Delta Aquifer from the literature.
	Salinity in the Mediterranean Sea	g/l	37.6	37.6	37.6	Assuming an average of the Mediterranean Sea from literature.
	Salinity in Nueva agoon	g/l	2.6	4.50	7.50	Salinity averages from 2003 and 2008 according to Rodriguez et al.(Rodríguez-Rodríguez, M et al. 2011) using as conversion 1mS=0.64 $g \cdot l^{-1}$
E	Nueva evapotranspiration in dry months from IFAPA weather station(IFAPA 2012)	m^3/ month	59,325	39,432	41,098	Average evapotranspiration from dry months between 2003 to 2008. The values were taken from literature per area of 25 ha.
E	Nueva evapotranspiration in wet months from IFAPA weather station(IFAPA 2012)	m^3/ month	24,331	21,337	23,737	Average evapotranspiration from wet months between 2003 to 2008. The values were taken from literature per area of 25 ha.
	Precipitation in 3 dry months(IFAPA 2012)	m^3/ month	1,308	183.3	33.3	Average precipitation from dry months between 2003 to 2008 considering 25 ha of surface. The values were taken from literature.
	Precipitation in 9 wet months(IFAPA 2012)	m^3/ month	5,419	8,933	9,911	Average precipitation from wet months between 2003 to 2008 considering 25 ha of surface. The values were taken from literature.
E	Evapotranspiration from Nueva(IFAPA 2012)	m^3/ month	33,079	25,860	28,076	Evapotranspiration's average from 2003 to 2008 taking into account the 25 ha surface of the lagoon.
	Precipitation in Nueva(IFAPA 2012)	m^3/ month	4,392	6,745	7,442	Precipitation's average from 2003 to 2008 taking into account the 25 ha of surface of the lagoon.
E	Crop Evapotranspiration(Fernández, MD. et al. 2001)	m^3/ month	146,368	306,849	320,984	Taking into account the harvested areas for 2003 and 2008 from literature(Andalucia 2012)

In order to obtain the unknown variables, FGW_Y, FGW_X, SGW_Y and SGW_X, we developed several equations for salt and water balances considering wet (X) and dry (Y) seasons.. Equation 7.3 shows the salt balance for dry months, taking into account fresh groundwater inflows (FGW), subterranean sea water intrusions (SGW, henceforward called sea groundwater inflows) and groundwater outflow (GW_o) from Nueva Lagoon to the neighboring aquifer. Equation 4 shows the salt balance for wet months with inflows from fresh groundwater and sea groundwater, as well as groundwater outflow. Equation 5 is the yearly salt balance, incorporating both dry (Y) and wet (X) months and reflects the steady-state assumption, that the yearly balance is equal to zero. Equations 7.6, 7.7 and 7.8 are the water balances for dry (Y) and wet (X) months, respectively, as well as the yearly balance. The difference between evapotranspiration (ET) and precipitation (P) in dry months is greater than or equal to the sum of the respective inflows, which consists of fresh and sea groundwater inflows. Concerning the wet months, Equation 7.7 shows the difference between evapotranspiration and precipitation being lower than or equal to the sum of inflows and the groundwater outflow. To solve the system of equations we assume the algebraic sign in Equation 7.6 and Equation 7.7 to be equality instead of inequality. The yearly water balance (Equation 7.8) is zero due to the steady-state assumption.

Salt Balances

$= S\cdot +FG\cdot +SG\cdot -G\cdot$ **Equation 7.3**

$= S\cdot +FG\cdot +SG\cdot -G\cdot$ **Equation 7.4**

$0 = Y\cdot S\cdot +X\cdot S\cdot +Y\cdot FG\cdot +X\cdot FG + Y\cdot SG\cdot + X\cdot SG\cdot -X\cdot G\cdot -Y\cdot G\cdot$

Equation 7.5

Water Balances

$E - \geq S + FG + SG - G$ **Equation 7.6**

$E - \leq S + FG + SG - G$ **Equation 7.7**

$0 = -E\cdot 12 + \cdot 12 + Y\cdot S + X\cdot S + Y\cdot FG + X\cdot FG + Y\cdot SG + X\cdot SG - X\cdot G - Y\cdot G$

Equation 7.8

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

A sensitivity analysis was carried out by changing different constant parameters, such as salinities ($C_{N,1983}$, $C_{N,2003}$, $C_{N,2008}$,), the number of wet (X) and dry (Y) months and the amount of precipitation and evapotranspiration. Several assumptions were made in this section (see Supporting Information, sectionS2.1.).The confidence intervals for the FF were calculated by taking into account the maximum and minimum FF from the sensitivity analyses and assuming a normal distribution.

7.2.4. Effect Factor

Data describing the effect of salinity for various endpoints (e.g. survival, growth inhibition) on 18 species (plants, fish, algae and a crustacean) native to the "Albufera de Adra" wetland were collected from literature (Ahmed et al. 2008; Barman et al. 2005; Bartolomé et al. 2009; Bright et al. 2002; Calheiros et al. 2011; Cambrollé et al. 2011; Espinar et al. 2005; Glenn et al. 1995; Greenwood ; Guma et al. 2010; Hutchinson ; Lillebø et al. 2003; Lissner et al. 1997; Mufarrege et al. 2011; Schuytema et al. 1997; Van Wijk et al. 1988; Wang et al. 1997) and are shown in the Supporting Information (Table S1). This work focused on the indigenous species and the associated damages only. As is common in LCA, only negative impacts are considered and potential benefits and changes in species composition are not included.

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

The 50% effective concentration (EC50) due to salinity is the concentration where a 50% reduction in a given endpoint (e.g. growth) is observed compared to the control. EC50s were either calculated by fitting the log-logistic function to the salinity concentration-response plots (see figure S1), or were taken directly from literature.

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

The SSD for salinity was constructed by fitting the log-normal cumulative distribution function to the EC50s for native species (see Table S1). Equation 7.9 describes the effect factor (EF_{Sal}), which is the average change in the potentially affected fraction of freshwater aquatic species (ΔPAF_{Sal}) due to the change in salinity (ΔSal). The effect factor is calculated as the average gradient at the 50% hazardous concentration ($HC50_{Sal}$), defined as the concentration at which 50% or more of the species included in the SSD are exposed to concentrations above their EC50 (Rosenbaum et al. 2008).

$$= = \ [\mathrm{PAF} \cdot \mathrm{l} \cdot] \qquad \textbf{Equation 9}$$

A 95% confidence interval for the hazardous concentration was estimated by parametric bootstrapping and this uncertainty was propagated to the Effect Factor by taking Equation 9 into account.

7.2.5. Calculation of impact scores

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

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

We considered the area of greenhouses from the municipality of Adra (Fernández, C. et al. 2004) for the 8 main crops from 26 different vegetables that are produced in Almería. Amounts of production from the province of Almería (Andalucia 2012) in 2008 were downscaled to the area around Albufera de Adra.

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

7.3. Results and discussion

7.3.1. Fate Factor

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

Table 7.2: Unknown variables in Nueva Lagoon for the years 1983, 2003 and 2008. The values of the variables are obtained by solving the equation system of Equation 3 to Equation 8 with GAMS.(GAMS 2012)

Parameter	Definition	Unit	Value 1983	Value 2003	Value 2008	Comments
FGW_Y	Fresh groundwater inflow to	m^3/month	$3.96\times10^{+4}$	$3.92\times10^{+4}$	$3.96\times10^{+4}$	Calculated from Equation 3 to Equation 8

	Nueva Lagoon in dry months					
FGW_X	Fresh groundwater inflow to Nueva Lagoon in wet months	m^3/ month	8.63 $x10^{+4}$	2.65 $x10^{+4}$	2.18 $x10^{+4}$	Calculated from Equation 3 to Equation 8
SGW_Y	Sea groundwater infiltration into Nueva Lagoon in dry months	m^3/ month	0	1.49 $x10^{+1}$	0	Calculated from Equation 3 to Equation 8
SGW_X	Sea groundwater infiltration into Nueva Lagoon in wet months	m^3/ month	0	0	0	Calculated from Equation 3 to Equation 8
$GW_{o,y}$	Groundwater outflow from Nueva Lagoon in the dry months	m^3/ month	0	0	0	Calculated from Equation 3 to Equation 8
$GW_{o,x}$	Groundwater outflow from Nueva Lagoon in thewet months	m^3/ month	6.12 $x10^{+4}$	1.41 $x10^{+4}$	7.46 $x10^{+3}$	Calculated from Equation 3 to Equation 8

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

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

Results for the sensitivity analysis are shown in the SI Table S2. The maximum value of the FF occurs between 2008 and 2003 when the wetland salinity is increased by 20%and the other parameters are kept constant, which gives a $FF_{2008\text{-}2003}$ of 6.72 $g{\cdot}l^{-1}{\cdot}m^3{\cdot}yr{\cdot}m^{-3}$. On the other hand, the minimum value of the FF results by decreasing the salinity by 20% for the period between 2003-1983 with a FF of $2.50{\times}10^{-1} g{\cdot}l^{-1}{\cdot}m^3{\cdot}yr{\cdot}m^{-3}$. These two extreme scenarios were taken as minimum and maximum for the uncertainty assessment and for establishing the distribution function.

7.3.2. Effect Factor

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

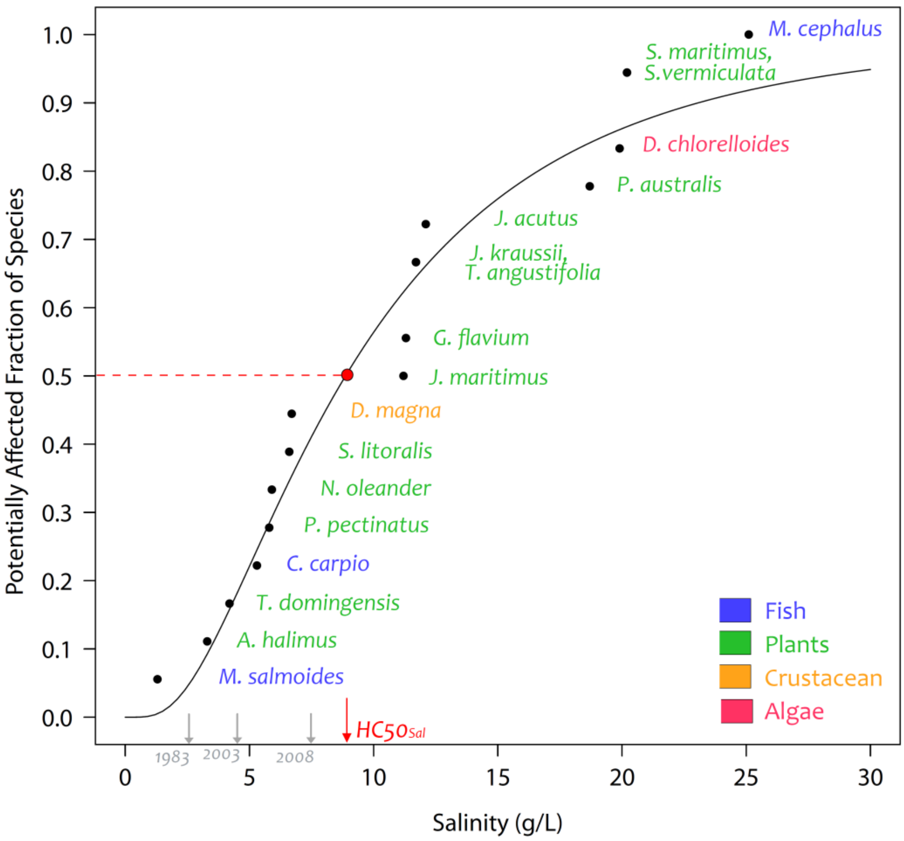

Figure 2: Species Sensitivity Distribution (SSD) for salinity for 18 species native to Nueva Lagoon. The grey arrows indicate the salinity in 1983 (2.6$g{\cdot}l^{-1}$), 2003 (4.5 $g{\cdot}l^{-1}$) and 2008 (6.5 $g{\cdot}l^{-1}$).

Salinity in the Nueva Lagoon increased from 4.5 $g{\cdot}l^{-1}$ in 2003 to 7.5 $g{\cdot}l^{-1}$ in 2008 (Table 7.1), which, according to Figure 7.2, corresponds to an increase of approximately 20% of species potentially affected in this period. The absolute number of species estimated to be found in the wetland is 30 (Humpesch 1985),taking into account plants, fish, algae and crustaceans. The increase in salinity and eutrophication over the past years has already resulted in the disappearance of a specie in Albufera de Adra, namely *Scirpus lacustri,* which was cited by Losa and Rivas Godo in 1968 (Losa et al.) and Sagredo in 1987 (Sagredo 1987) but is no longer found in the lagoon today. From Figure 2 we observe that two fish species*,* *M. salmoides* (EC50 = 1.3 $g{\cdot}l^{-1}$) and *C. carpio* (EC50 = 5.3 $g{\cdot}l^{-1}$), are particularly sensitive to salt stress, and another fish species, *M. cephalus* (EC50 = 25.1 $g{\cdot}l^{-1}$), is the least sensitive of all the native species included in this study. The EC50 of the algae included in the SSD, *D.*

chlorelloides is 19.9 $g \cdot l^{-1}$ and is above $HC50_{sal}$ but the crustacean, *D.magna*is located below $HC50_{sal}$ with an EC50of 6.6 $g \cdot l^{-1}$. Plants, the most abundant taxonomic group are distributed throughout the SSD, with 5 species lying below $HC50_{sal}$ and 8 species lying above it.

Data availability permitted the consideration of 18 species from Nueva Lagoon (13 plants, 3 fish, 1 algae and 1 crustacean) in this work. So, given a total of 30 species reported to be found in the wetland(Humpesch 1985), with these 18 species we cover 60% of the species in the wetland. The calculated EF could, with some caution, be applied to other wetlands, assuming their native species composition is not entirely dissimilar to the one encountered in Nueva Lagoon.

For instance, Punta Entinas is an endorheic wetland, located in the arid southeast of Spain very close to Albufera de Adra surrounded by Mediterranean ecosystems. Both wetlands, Nueva Lagoon and Punta Entinas (focus on salt marsh and marshland) have species in common, for instance *A.halimus, S.vermiculata, S.maritimus, J.maritimus, J.acutus, P.australis, N.oleander, P.pectinatus* and *D.chlorelloides*(Humpesch 1985)*)*, amounting to 50% of the species included in the EF for Albufera de Adra. Therefore, we might consider the EF to be applicable for the marshland of Punta Entinas.

7.3.3. Characterization Factor

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

7.3.4. Impact score

The impact score is calculated as a product of the characterization factor (CF) developed for Nueva Lagoon (3.16 $x10^{-1}PAF \cdot m^3 \cdot yr \cdot m^{-3}$) and the crop evapotranspiration (ET_{crop}) (Equation 7.10). Crop evapotranspiration was obtained from “Las Palmerillas” experimental station close to Albufera the Adra (constructed to improve the efficiency in agricultural production)(Fernández, MD. et al. 2001)and was converted to $m^3 \cdot yr^{-1}$ taking into account the cultivation area from the study(Fernández, C. et al. 2004) close to Adra. Table 7.3 shows the Impact Score ($m^3 \cdot PAF \cdot yr$) for each crop per unit of area and per tonne of production ($m^3 \cdot PAF \cdot tonne^{-1} \cdot yr$) considering the local productions in that area (Andalucia 2012).

$$IS = CF \cdot E \qquad \textbf{Equation 10}$$

Table 3:Characteristics and impact scores for the 8 main crops in the area of study: greenhouse area (GH_{Area}), crop evapotranspiration (ET_{crop}), impact score per area ($IS_{1,per}$area)and its assigned percentage ($IS_{1,\%}$), crop's yield (Y_c) and Impact Score per tonne ($IS_{2,tonne}$) .

Crops	GH_{Area}	ET_{crop}	$IS_{1, per\ area}$	$IS_{1,\%}$	Y_c	$IS_{2,tonne}$

	Ha	$m^3 \cdot ha^{-1}$	$m^3 \cdot PAF \cdot yr$	%	tonne·ha^{-1}	$m^3 \cdot PAF \cdot tonne^{-1} \cdot yr$
Tomato	216.1	7.50x10^{+5}	1.98x10^{+4}	29.6	100.6	9.08 x10^{-1}
Pepper	180.5	6.82x10^{+5}	1.80 x10^{+4}	26.9	61.7	1.61 x10^{0}
Cucumber	95.1	1.53x10^{+5}	4.03 x10^{+3}	6.0	87.0	4.87 x10^{-1}
Zucchini	97.9	2.75x10^{+5}	7.24 x10^{+3}	10.9	54.5	1.36 x10^{0}
Watermelon	101.1	1.81x10^{+5}	4.76 x10^{+3}	7.1	69.6	6.76 x10^{-1}
Melon	115.0	3.42x10^{+5}	9.02 x10^{+3}	13.5	36.0	2.18 x10^{0}
Aubergine	33.7	9.58x10^{+4}	2.52 x10^{+3}	3.8	72.6	1.03 x10^{0}
Green bean	59.7	5.19x10^{+4}	1.37 x10^{+3}	2.1	15.3	1.50 x10^{0}

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

7.3.5. Application in LCA studies

This work derived the first CF for salinity impacts in a coastal wetland defined as the change in the Potentially Affected Fraction (PAF) of species due to a change in salinity related to the extraction of groundwater for crop irrigation. This case study takes the expectation away that this is a fully applicable approach for the whole world and it proved to be very relevant indeed. The impacts on wetland biodiversity due to the irrigation of the existing crops close to the study area, were calculated using the proposed CF.

A comparison between the salinity impacts of the main crops tomato, cucumber, zucchini, melon and aubergine with other impact categories was carried out in order to investigate the relative importance of salinity impacts. For this comparison we used the endpoints of several categories within the area of protection "ecosystem quality" of the ReCiPe methodology (Goedkoop et al. 2009). Experimental data for these crops were adapted from Stoessel et al.(2012) taking into account a local yield (Table 7.3) in Adra greenhouses. The crop-specific impact scores presented in Table 7.3 ($PAF \cdot m^3 \cdot yr \cdot tonne^{-1}$) were converted into $species \cdot yr \cdot kg^{-1}$considering the recommended freshwater species density (Goedkoop et al. 2009)(7.89×10^{-10} $species \cdot m^{-3}$) and the conversion (Goedkoop et al. 2009)dPDF/dPAF = 1 (Table 7.4).

Table 4.Endpoint Impacts (*species·yr·kg*$^{-1}$) according to the ReCiPe methodology and the contribution of each category to the total ecosystem quality impact. No data was available for green beans and watermelon, and thus these crops are neglected in this comparison.

Category impact	Tomato	Cucumber	Zucchini	Melon	Aubergine

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

The results for all crops show that, if the generic freshwater species density from ReCiPe is used (Table 4), the impact of salinity (due to water use) for the total damage to ecosystems is in the range of freshwater eutrophication for tomato (7.16x10^{-13} species.yr·kg^{-1}) and cucumber (3.84x10^{-13} species·yr·kg^{-1}), in the range of terrestrial ecotoxicity for zucchini (1.07x10^{-12} species·yr·kg^{-1}) and melon (1.72x10^{-12} species·yr·kg^{-1}) and in the range of freshwater ecotoxicity for aubergine (8.14x10^{-13} species·yr·kg^{-1}). The relative contribution to the total impact score is dominated by climate change for all the crops with approximately 70% in tomato, cucumber and aubergine and around 40% in zucchini and melon. The climate change ecosystems impact category considers fertilizing (ammonium nitrate, single superphosphate and potassium sulphate) and electricity consumption for the irrigation system, which is the highest contributing impact when capital goods are not considered.

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

7.3.6. Outlook

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

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

7.4. References

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8. Estimating water consumption of potential natural vegetation on global dry lands: building a LCA framework for green water flows

Montse Núñez[1,2], Stephan Pfister[3], Philippe Roux[2], Assumpció Antón[4]

[1]INRA, UR050, Laboratoire de Biotechnologie de l'Environnement, Avenue des Etangs, Narbonne, F-11000, France
[2]Irstea – UMR-ITAP, 361 rue Jean-François Breton BP5095, F-34196, Montpellier, France
[3]ETH Zurich; Institute of Environmental Engineering, 8093 Zurich, Switzerland
[4]IRTA (Institute for Food and Agricultural Research and Technology), Ctra. Cabrils km 2, 08348 – Cabrils, Barcelona, Spain

8.1. Introduction

Land use and the water cycle are intricately linked. During the past century, the expansion of the total land cover dedicated to agriculture has led to a decrease in evapotranspiration and to an increase in runoff due to usually lower water requirements of crops compared to natural forest lands (Calder, 2004; Rockström and Gordon 2001, Scanlon et al. 2006). Furthermore, depending on the planted crop type and the production system (e.g., if the crop is grown in the wet or dry season), the consumption of water stored in the soil profile may vary significantly for a given location (Mekonnen and Hoekstra, 2010; Núñez et al. 2012). In urban areas, sealed soils reduce significantly evaporation, prevent transpiration and infiltration and drain most of the rainfall through surface runoff. Soil water balance influences the water availability of rivers and aquifers downstream available for allocation to competing uses, including human activities and ecosystems.

Life cycle assessment (LCA) is a method that provides information on the environmental impacts that inventoried resource use and emissions by a production system cause along its entire life-cycle. LCA methodology has rapidly progressed over the last few years to better account for the ecological impacts of water use. This is especially important for irrigated agriculture, i.e., the economic sector currently responsible for ~70% of global water withdrawal (UNESCO, 2009) and ~85% of global water consumption (Shiklomanov et al., 2003) taken from surface and ground water bodies. Moreover, global (rain-fed and irrigated) crop production consumes 4-5 times more rainfall water (stored in the soil) than irrigation water (Hoff et al., 2010). Recently, new databases and methods have been developed to integrate the use and consumption of surface and ground water (the so-called blue water) both in the life cycle inventory (LCI, e.g., ref Boulay et al., 2011; Pfister et al., 2011) and in the life cycle impact assessment (LCIA) steps (e.g., ref Milà i Canals et al., 2009; Pfister et al., 2009). So far, consumption of rain water stored in the soil profile (the so-called green water) through plant evapotranspiration and other evaporation routes (e.g., rainwater harvested and reused in production systems) has been for the most part disregarded. This is because green water consumption is often considered less environmentally relevant from a pure water consumption perspective. Yet, soil moisture consumption linked to a specific type of land use does contribute to changing regional water availability in rivers and aquifers, which can be especially critical when water is scarce as in the case of dry lands.

Despite the potential environmental impacts of water losses via evapo(transpi)ration, only Milà i Canals et al. (2009) have attempted to provide a set of LCI default values estimating land-use effects on the water cycle. The values given report on the percentage difference of useful water for ecosystems under different land-use types (infiltration + runoff for non-sealed lands and infiltration for sealed lands). They are compared to useful water availability values for potential natural land state in Europe with no human intervention, defined as potential natural vegetation (PNV, Chiarucci et al., 2010), namely forest. Although it is clear that PNV water consumption depends on water availability at each location of the earth, natural states outside Europe were disregarded in the values provided by Milà i Canals et al. (2009). Even for Europe, the values are very approximate, as the specific characteristics of PNV within each region are obviated. Assessing site-generic PNV instead of the regional natural reference situation leads to the inaccurate accounting of the effects of land-related changes in the water balance.

In this paper, we estimated the long-term annual contemporary water consumption of the PNV fitted to local bio-geographic conditions on global dry lands. Results are reported at different spatial aggregation levels: 10 arcmin grid-cell, ecoregion and biome scales as well as countries and continents. Compared to previous assessments of quantification of green water consumption by terrestrial natural vegetation, we computed total losses by evapotranspiration, and not only by transpiration as in Gerten et al. (2005). In addition, instead of emphasizing global water consumption values (Gerten et al., 2005) or only reporting evapotranspiration for some extremely approximate major biome units (Rockström and Gordon 2001) we provided results at various ecological and administrative geographic scales. Therefore, PNV water consumption flow can be compared at different spatial aggregation levels for different detail levels in LCA.

In LCA, PNV related water consumption can be used to estimate the quantitative effect on water availability of changes in direct water uptake under human land occupation when compared to the potential natural situation. The difference in these green water flows represents a lack (or excess) of groundwater recharge and surface water runoff. The environmental effects should be assessed by characterization factors (CFs) in the LCIA step. The regionalized natural reference flows of green water consumption are used to build proper LCI, i.e., to quantify changes in environmental flows. In practice, total green water use per crop and country are available (e.g., Pfister et al. 2011, see the SI XLS file) and therefore by providing the reference water flow, we are able to compute changes for green water flows. Figure A8.1 in the annex graphically describes the environmental cause-effect chain, i.e., environmental impacts arising from human-activity related soil-water uptake. The figure also shows an overview of the interrelationships between this impact pathway and other water and land-use related impacts.

8.2. Methods

8.2.1. Methods for evapotranspiration estimation

The water consumption of local natural reference vegetation was estimated by combining results from two assessment methodologies: 1) an empirical equation, based on potential evapotranspiration (ET_0), and precipitation (P) data 2) an empirical model, using remote-sensing data sets. Further method description details are provided below and table A8.1 in the annex offers a brief description of the methodology used.

8.2.1.1. *Approach 1: equation-based*

A number of simple models have been developed to calculate annual evapotranspiration of vegetation using measurable climatic and catchment water balance data (Budyko 1974; Milly, 1994; Zhang et al., 2001). The practical application of these models can vary as a function of the amount of data required and data accessibility. From among these models, we selected the approach proposed by Piñol et al. (1991) because of its simplicity and accuracy. According to these authors, annual water consumption of the natural vegetation that would potentially grow in a given catchment ($ET_{PNV,i}$, in mm/y) can be calculated from the ratio of P to ET_0 in the catchment *i* (equation 8.1). ET_0 represents the water demand of the atmosphere and can be obtained from meteorological data (Allen et al., 1998).

= ×   Equation 8.1

The exponent (dimensionless) *k* takes into account certain characteristics of individual catchments. The physical significance of *k* appears to be related to the type of vegetation cover involved, the rainfall partition between runoff and evapotranspiration and the seasonal rainfall distribution throughout the year (Piñol et al., 1991). Prior to the application of the model on a specific catchment, we must set a suitable k value by using local empirical data. For given P and ET_0 values, lower values of *k* indicate lower evapotranspiration losses. For example, in areas lacking a seasonal distribution of precipitation, greater values of *k* and evapotranspiration losses are to be expected.

Equation 8.1 is only applicable in catchments where evapotranspiration of natural vegetation significantly correlates with annual precipitation, while, at the same time, a significant correlation between runoff and precipitation has not been established. This is the case for dry lands. In wetter climates, the opposite is true, runoff is linearly dependent on precipitation, and evapotranspiration linked to natural vegetation is fully satisfied and considered as constant. The contrasting behavior of rainfall partition in arid and humid regions is captured by the P/ET_0 ratio. According to Piñol et al. (1991) the model is valid for dry climates (areas with, approximately, $P/ET_0 \leq 0.75$, almost 50% of the terrestrial surface). Therefore, water consumption of PNV in humid climates ($P/ET_0 > 0.75$) cannot be estimated using this approach. The two important assumptions of the model are that (1) deep water losses are negligible, i.e., water does not percolate down to recharge aquifers (2) soil water content is identical at the beginning and at the end of the hydrological year (1 October and 30 September). This means that the model does not account for soil moisture accumulation from one month to the next, therefore $ET_{PNV,i}$ cannot be calculated on a monthly time scale.

In this paper, we applied equation 8.1 over a set of 6788 data points with natural vegetation scattered over the dry lands of the world (see approach 2 below). For every data point, twenty-seven scenarios of $ET_{PNV,i}$ were developed by modifying the value of the exponent *k* of equation 8.1 between 1.03 and 2.40 (k_j), which is the data range for the *k* parameter given in Piñol et al. (1991). The P and ET_0 data needed to run the model were obtained from two 10 arcmin (~20 km resolution at the Equator) resolution maps (New et al., 2002).

8.2.1.2. Approach 2: model-based

Since approach 1 relies on a simple empirical equation requiring a single parameter to be calibrated, an additional independent approach is applied to test the robustness and to calibrate the k parameter described in approach 1. This calibration enables adjustments of the empirical equation to regional conditions. Based on a continuous satellite-derived record of actual land evapotranspiration (AET) on the Earth's total land surface using a 8 kilometer resolution and land cover maps, annual evapotranspiration of actual natural vegetation in dry lands was estimated using an evapotranspiration algorithm (Zhang et al., 2010). For the analysis, we selected areas with natural coverage located in dry ecosystems. In our paper, water consumption estimates obtained with this second approach are called AET_i (in mm/y). Pastures, croplands and settlements were considered non-natural areas and excluded from the assessment. To be consistent with the initial equation-based approach, the same P/ET_0 ratio threshold was used to identify dry lands, i.e., $P/ET_0 \leq 0.75$. We obtained 6788 data points with corresponding estimates of actual vegetation evapotranspiration. Water consumption by actual natural vegetation is assumed to be a good proxy in order to estimate PNV evapotranspiration, as, in theory, actual natural vegetation should be close to the potential natural state, although some discrepancies are to be expected (e.g., succession areas). Furthermore, remaining natural areas within an ecoregion might not be representative of other areas, which means that evapotranspiration of the remaining natural areas might be not comparable to evapotranspiration that would occur in other human-modified systems.

8.2.2. Calibration of the equation using model-based results

The site-generic expression of equation 8.1 was calibrated using the AET_i estimation data set points, as shown in equation 8.2. With the calibration process, we found a best-estimate of the exponent *k* adjusted to the local conditions in point *i*. We thus regionalized the *k* exponent and consequently equation 8.1. The optimized *k* value ($k_{opt,i}$) was selected from the scenario analysis where k_j varied between a minimum of j=1.03 and a maximum of j=2.40, as justified before.

= for Equation 8.2

According to equation 8.2, the optimized *k* value in the point *i* was the value for which the absolute difference of both estimations of water consumption by the local PNV was minimized. Using the regionalized $k_{opt,i}$ and local P and ET_0 records obtained from New et al. (2002) for each point *i*, optimized evapotranspiration of local PNV ($ET_{PNVopt,\ i}$ in mm/y) was then estimated with equation 8.1. We also identified the differences between $ET_{PNVopt,\ i}$ and AET_i and removed from further analysis those data points for which the difference was higher than 20% of the AET_i, since this indicates the equation does not fit. This might be induced by ecosystems that are fed by groundwater or surface water (wetlands). 2026 points were therefore excluded from the assessment.

8.2.3. Spatial aggregation of data point results

In LCA, the selection of the appropriate spatial aggregation scale for inventory flows as well as the impact assessment methods is a crucial step (Koellner et al., 2012; Mutel et al., 2012), as data uncertainty and study outcomes are closely linked to the regionalization scale chosen. Instead of selecting the optimal spatial scale of aggregation, which also depends on the detail level of an LCA, we provided optimized data point results ($k_{opt,\,i}$ and $ET_{PNVopt,\,i}$) on different ecological and administrative spatial aggregation units: (1) local aridity index, (2) ecoregions and biomes and (3) countries and continents. Indeed, we believe that the LCA databases currently under development should implement routines to automatically link the supply chain to the spatial information available in the LCI, thus enabling greater precision for the presented approach.

8.2.3.1. Aggregation by the local aridity index

The local aridity index, defined as the relation between P and ET_0, has an important role in determining the share of rainfall water that can be stored in the soil profile. The lower the aridity index, the greater the aridity of the area, and thus the lower the soil-water availability for uptake by plants, which is due to the high evaporative demand of the atmosphere. Regionalization using the local aridity index was carried out by calculating the P/ET_0 ratio for each data point i and by allocating it to one of the eight aridity index categories created, with values ranging from 0.00 to 0.75. All the categories correspond to dry climates, and are based on the same criterion that we used throughout the research (i.e., $P/ET_0 \leq 0.75$ for dry regions). As a result, an average ± standard deviation k_{opt} was defined for each aridity index class ($k_{opt,\,ai}$). The aridity index specific $k_{opt,\,ai}$ values were then spatially distributed over a 10 arcmin resolution map by adopting the aridity index for each pixel through the overlapping of a P and an ET_0 data layers at the same grid cell resolution (New et al., 2002). With these input data, we applied equation 8.1 and obtained a water consumption estimate map of PNV (ET_{PNVopt} calculated with $k_{opt,\,ai}$) for the arid regions of the world at a 10 arcmin pixel resolution.

8.2.3.2. Aggregation by ecoregions and biomes

From among the many existing approaches to global regionalization of reference natural vegetation units (e.g., Holdridge's life zones system (Holdridge 1947), Bailey's ecoregions (Bailey, 1998)), we chose the classification system developed by Olson et al. (2001) for terrestrial biomes and ecoregions to report evapotranspiration of PNV, because it is the classification system for natural terrestrial regions that is most often recommended for LCA in land use applications (Koellner et al., 2012; de Baan et al., 2012). Olson et al. (2001) subdivided the terrestrial world into 14 biomes, in which 867 ecoregions are nested. From the total number of ecoregions, 501 (i.e., 58%) are partially or entirely located within areas with a P/ET_0 ratio≤0.75. Ecoregions are sizeable units of land containing a distinct assemblage of natural communities and species, with boundaries that approximate the original extent of natural communities prior to land-use change (Olson et al., 2001). Ecoregions have often been used to represent PNV patterns and, at times, to characterize regional differences in water quality as well (Omernik, 1987).

Regionalization using ecoregions was performed by assigning each data point i to the natural region in which it was located. Next, points within the same ecoregion were aggregated and an average ± standard deviation $k_{opt,\ r}$ was obtained. With this procedure, we obtained $k_{opt,\ r}$ for 146 dry ecoregions, representing 60% of the surface of the dry lands of the world. There were certain ecoregions for which we had no data point to carry out a calibration with equation 8.2 (355 regions representing the remaining 40% of the dry area). As we could not compute a regionalized exponent k for these ecoregions, $k_{opt,\ r}$ was approximated with the weighted $k_{opt,\ r}$ of the neighboring ecoregions, as a function of the shared length of the ecoregions' boundaries, following an iterative procedure. As with the aridity index aggregation, we allocated $k_{opt,\ r}$ to the corresponding ecoregion and derived a global raster layer of PNV water consumption (ET_{PNVopt} calculated with $k_{opt,\ r}$) at a 10 arcmin resolution, by using the P and ET_0 gridded maps from New et al. (2002). For each ecoregion, we also calculated average P and ET_0 for the dry areas. This allowed us to provide an average default value of PNV water consumption for the dry areas located in each ecoregion of the world.

At the broadest spatial scale, i.e., the biome resolution level, an average value of PNV water consumption was derived by allocating ecoregions to biomes and by averaging $k_{opt,\ r}$, P and ET_0 for the ecoregions within a biome (ET_{PNVopt} calculated with biome-aggregated $k_{opt,\ r}$).

8.2.3.3. Aggregation by countries and continents

Regionalization of natural processes such as water consumption using administrative units makes much less sense than using ecological borders. However, LCI data are often only available at the country level, and in certain cases, only the continent of origin can be identified. In a worst case scenario, the LCI may have no spatial information at all, thus a global default value should be made available as well for processes occurring at unknown location. We calculated PNV evapotranspiration for each country possessing arid lands, by aggregating the arid grid cells of the PNV water consumption map we derived at the 10 arcmin resolution level. The same procedure was repeated for continents and for unspecific locations (i.e., world).

8.2.4. Inclusion of the soil-moisture consumption flow in the LCI of an LCA

Based on previous research where attempts were made to include the change in soil-water balance in LCA (Milà i Canals et al., 2009; Núñez et al., 2012), we suggest a spatially differentiated framework to include soil-water consumption in the inventory stage. The method consists in including the regionalized "net soil-water consumption" $NET_{soil\text{-}water}$, with units of volume soil-water consumed per area and time unit (equation 8.3), as the elementary flow. Regionalized net soil-water consumption is defined as the difference between evapo(transpi)ration under human land occupation x at a given location i ($ET_{soil\text{-}water,\ x,\ i}$) and evapotranspiration of the regional natural reference vegetation (ET_{PNVopt}) on any of the spatial aggregation scales for which it was derived. The method is intended to be used in contexts of direct soil-moisture uptake by agriculture as well as for rainwater harvested and reused in production systems. The analysis has to account for a full year of land use, as ET_{PNVopt} is an annual flow. If, during the year, there is more than one land use (e.g., two consecutive crop periods), $NET_{soil\text{-}water}$ may be allocated to the different land uses as a function of the soil-water consumed per each land use.

Like with the consumption of water withdrawn from water bodies, the environmental significance of the volume of consumptive soil-water depends on the amount of water held in the soil. This is captured by the aridity index ratio (P/ET_0). We assumed that, in humid climates ($P/ET_0 > 0.75$), soils have sufficient available moisture for optimum plant growth and that, at the same time, this plant water uptake does not amplify water shortage downstream. On the other hand, in dry areas ($P/ET_0 \leq 0.75$), soil-water consumption by plants may aggravate water scarcity in water bodies downstream from the location of soil-water consumption or cause soil-moisture depletion. Accordingly, the regionalized net soil-water consumption ($NET_{soil\text{-}water}$) is estimated in each situation as shown in equation 8.3.

If = 0

If = - Equation 8.3

In LCA, all the flows recorded in the inventory must be scaled to the quantitative output of the study, i.e., the functional unit. This means that the $NET_{soil\text{-}water}$ flow expressed on the area-time unit may need further adaptation.

8.2.5. Case study

We have demonstrated the applicability of the LCI framework by calculating the $NET_{soil\text{-}water}$ consumption of wheat grown for bread production in three case studies and at different levels of detail for LCI data availability. In terms of world cereal crop production, wheat ranks the second after maize (Mekonnen and Hoekstra, 2010). The global average soil-water consumption per yield unit of wheat (i.e., green water footprint) was recently estimated to be 1200-1300 m^3 Mg^{-1} (Mekonnen and Hoekstra, 2010; Pfister et al., 2011; consult the SI XLS file). In case study 1, we know the exact location in longitude and latitude for the dry land in Spain where the cultivation of wheat takes place. In case study 2, the cultivation region (the South of Spain), is known, but not the exact location of the plot. In case study 3, we only know that wheat can come from Spain, France or Italy, which contributed 1%, 6% and 1.2% to global wheat production respectively for the 1996-2005 period (Mekonnen and Hoekstra, 2010). Soil-water consumption per area for wheat production (mm y^{-1}) was estimated for the case study 1 plot with the FAO approach adapted to Spain (Núñez et al., 2012). For case study 2, we chose the same method used in case study 1, but as the exact location was unknown, we calculated the average soil-water consumption for several plots located in Southern Spain (province of Malaga). Climate data needed (P, ET_0) for case studies 1 and 2 came from regional statistics for the 2002-2011 period (Junta de Andalucia, 2012). For case study 3, the country averages of soil-water consumption per yield (m^3 Mg^{-1}, consult the SI XLS file) were converted to soil-water evapotranspiration per area using Monfreda et al. (2008)'s wheat yield data (Mg ha^{-1}). Since spatial data quality available for each case study differed, different spatial aggregation levels for calculating PNV evapotranspiration were used.

8.3. Results

8.3.1. Soil-water consumption regionalized using the local aridity index

Table 8.1 lists optimized average ± standard deviation k values ($k_{opt,\ ai}$) to be applied in equation 8.1 for the eight differentiated dry aridity index classes. Upper and lower limits of the exponent k are set to 1.03 and 2.40 values (Piñol et al., 1991). These values should be taken into consideration when standard deviation k values of table 8.1 are used in equation 8.1, as it results in a skewed distribution. Figure 8.1a shows water consumption of PNV in dry lands calculated with equation 8.1 at 10 arcmin pixel level resolution by using optimized $k_{opt,\ ai}$ and climate data from New et al. (2002). High water consumption values (more than 1100 mm water consumption per year) were obtained in Central America and North of Australia, in relatively humid areas ($P/ET_0 \geq 0.65$). On the other hand, lower water consumption figures were recorded in the cold and hot deserts of Greenland, Africa and the Pacific coast of South America, where the aridity index is close to zero due to the low precipitation.

Table 8.1: $k_{opt,\ ai}$ values at the grid cell level, aggregated by the local aridity index (P/ET_0). Lower and upper limits of k applied in equation 8.1 should be 1.03 and 2.40.

Aridity index (AI)	$k_{opt,\ ai}$_avg (-)	$k_{opt,\ ai}$_sd (-)
0 ≤ AI < 0.05	1.67	0.73
0.05 ≤ AI < 0.15	1.58	0.70
0.15 ≤ AI < 0.25	1.60	0.68
0.25 ≤ AI < 0.35	1.73	0.66
0.35 ≤ AI < 0.45	1.71	0.64
0.45 ≤ AI < 0.55	1.81	0.63
0.55 ≤ AI < 0.65	1.83	0.52
0.65 ≤ AI < 0.75	1.62	0.52

8.3.2. Soil-water consumption regionalized at the ecoregion level

Optimized average ± standard deviation k values ($k_{opt,\ r}$) for the 501 ecoregions located in the dry lands of the world are listed in the SI XLS file and shown in figure 8.1b. Note that, in most cases, standard deviation figures of $k_{opt,\ r}$ are equal to zero. However, $k_{opt,\ r}$ has been approximated with the weighted $k_{opt,\ r}$ of the neighboring ecoregions, sharing the same value. Therefore, the k values for these ecoregions have low variability but high uncertainty. We also reported in the XLS file an average default value of PNV water consumption on the ecoregion scale, including average and standard deviation of P and ET_0 data for the dry areas of every ecoregion. Overall, results of water consumption of PNV on the ecoregion level were, as expected, very similar to those obtained for the aridity index level of aggregation (figure A8.2 in the annex). However, on the ecoregion scale, the highest water-demand levels for potential natural vegetation communities were found on tropical and temperate forests of the East of Australia, with water consumption figures ranging between 1000-1140 mm/year and a $k_{opt,\ r}$ ~2.40.

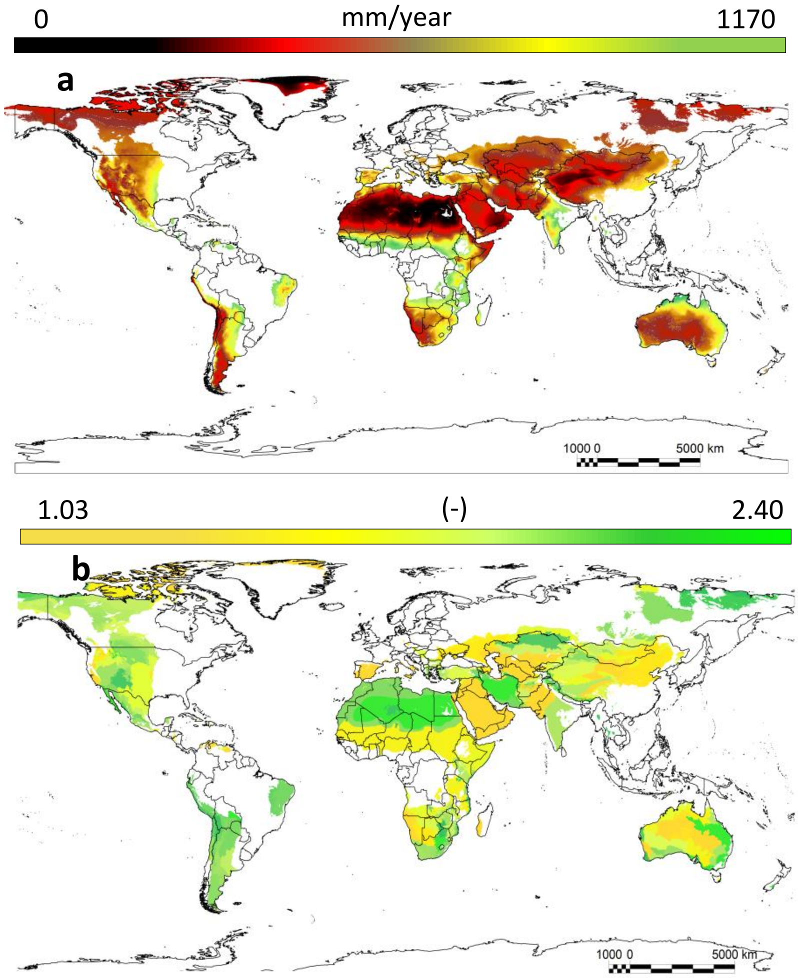

Figure 8.1: a) Water consumption of PNV (mm/year) in global dry lands, calculated with k optimized by the local aridity index (ET_{PNVopt} with $k_{opt,\ ai}$). b) Regionalized spatial distribution of $k_{opt,\ r}$ at the ecoregion level ($k_{opt,\ r}$=0 in ecoregions characterized by rock, ice and lakes). The blank areas on both maps represent humid lands, for which we did not estimate PNV water consumption.

8.3.3. Soil-water consumption regionalized at the biome level

Average water consumption of PNV at the biome level, calculated with within-biome $k_{opt,\ r}$ and average P and ET_0 are summarized in the SI XLS file.

8.3.4. Soil-water consumption regionalized at the country and continental level

The SI XLS file provides average ET_{PNV} (mm/y) for countries with arid lands, and arid area percentages for each country. World and continent averages are given as well for the assessment of processes occurring in unknown location. The largest PNV water consumption values are found in countries around the Equator (Honduras: 931 mm/y and Guinea: 912 mm/y) and the lowest values are reported for desert-like countries such as Egypt (26 mm/y) and Greenland (40 mm/y). When the practitioner knows that the production system is located on an arid area, reported values per country can be directly used. We do, however, recommend using either the gridded or the ecoregion detail level whenever possible. When the country is known but the exact location is unknown (and cannot be assumed), there are two options: all of the surface of the country can be considered as arid (worst case) and the XLS file values directly applied. Otherwise, the decision should be based on the percentage of the country's arid areas, as we did not calculate PNV evapotranspiration for humid areas. For example, assuming an arid situation if more than 50%, 70% or 30% is arid (for hierarchical, individualist, and egalitarian perspective, respectively).

8.3.5. Case study

Net soil-water consumption was calculated with equation 8.3 for the three case studies (table A8.2 in the annex). The spatial uncertainty of inventory spatial data for each case study differed, thus affecting the uncertainty of the $NET_{soil\text{-}water}$ inventory flow. For case study 1 spatial data quality was excellent. Exact location and local P and ET_0 were available. Hence, ET_{PNV} and then $NET_{soil\text{-}water}$ could be calculated at the most detailed grid-cell level. For case study 2 the exact location was unknown, which meant we could not use the gridded $k_{opt,\ ai}$ values. PNV water consumption was calculated with the $k_{opt,\ r}$ value of the ecoregion where the cultivation area was located (Southwest Iberian Mediterranean sclerophyllous and mixed forests) and P and ET_0 for this area. For case study 3, inventory spatial data uncertainty was very significant. Therefore, ET_{PNV} was only roughly estimated by using average values per country (SI XLS file) and a weighted $NET_{soil\text{-}water}$ was calculated based on the proportion of wheat production in each of the three countries. Calculations for this latest case study were made assuming the worst case scenario (i.e., countries 100% of arid surface and the water consumption values per country reported in the SI XLS file directly applied). Results show that the largest water savings made when compared to the local adapted PNV was obtained in case study 3 ($NET_{soil\text{-}water}$= -320±42 mm y^{-1}), with a saving from 1.1 to 1.8 times the amount of soil-water consumed in case studies 1 and 2. However, this is the most uncertain value and can vary significantly depending on the assumptions made (e.g, % wheat production and % arid area in each country).

8.4. Discussion

8.4.1. Methodological approach, limitations and uncertainties

In this study, we addressed the issue of soil-water consumption in LCA by developing a method designed to quantify evapotranspiration of the natural reference vegetation in dry lands as a function of location. In our approach, water consumption of PNV is used in the LCI stage to quantify the net change in soil-water availability under the production system compared to the natural reference situation. A soil-water consumption decrease (which is typically the situation on agricultural lands) can be considered a water saving, and the resulting positive or negative environmental effects should be subsequently assessed by CFs for water consumption during the LCIA step. We did not address the specific impacts of green water change other than on the total water balance. This remains the major drawback of this work. However, it is in line with the state-of-the art: data availability does not allow for better assessment. Hence, soil moisture impacts on other environmental consequences such as greenhouse gas emissions of soils were excluded from the assessment. In addition, the focus on dry lands, where water scarcity is most relevant, is another major drawback of the approach, since changes in soil-water consumption leads to changed runoff patterns in all climates, and can therefore be disastrous in wet areas through floods and increased erosion. Furthermore, the potential effects of global warming on the global water cycle and on the subsequent changes in global vegetation patterns were outside the scope of the study.

As the quality of the geographical information linked to inventory flows may well differ, water consumption of the regionalized natural vegetation has been reported on several spatial aggregation levels, which facilitates application in LCA studies. Whenever possible, $NET_{soil\text{-}water}$ should be estimated at the most detailed grid-cell resolution level, for which k and ET_{PNV} values were aggregated based on the local aridity index. Unlike for ecoregions and countries, the aridity index is a climatic parameter influencing soil evapo(transpi)ration. It is therefore a more relevant regionalization scale for the assessment of green water consumption flows. Consequently, the ecoregion and country aggregation levels should only be applied for background processes or LCA studies where soil-water consumption is not an important issue. For large countries or countries with very diverse climate regions, both the average and the standard deviation, or the minimum and maximum range values of $NET_{soil\text{-}water}$ should be reported and used in sensitivity analysis.

The combination of the two methods to estimate water consumption of the potential natural vegetation has allowed us, on the one hand, to capitalize on information available on evapotranspiration of the actual natural vegetation for the entire land area and, on the other hand, regionalize a site-generic equation for the estimation of global PNV evapotranspiration based on precipitation and potential evapotranspiration data.

The equation-based approach was successfully applied in Núñez et al., (2012) from a LCA perspective to estimate PNV evapotranspiration in Spain, i.e., the Mediterranean forest, in more than 100 points located throughout the country. A spatial generic value of k=2 was used for all the studied points, as this is the value suggested for catchment comparisons when local empirical data are not available (Piñol et al., 1991). In this paper, based on empirical actual evapotranspiration data, we obtained optimized k values ranging from 1.03 and 2.40 for the world's dry lands.

We stress that results obtained from applying our method in LCA must be interpreted with caution. Nevertheless, we feel our study has bridged an important methodological gap regarding soil-water consumption assessment performed in LCA oriented methods. Obviously, limitations and uncertainties in results remain due to insufficient information on different input data used. One source of uncertainty concerns weather stations and the related measurement inaccuracies of the meteorological variables P and ET_0 used in equation 8.1 and the AET satellite-derived algorithm. Another is the different time period for which P, ET_0 and AET data are available. For the climatic variables, worldwide statistics were recorded between 1961-1990, while for the remote-sensing AET, worldwide data were made available from 1986 to 2006. Both data sets cover many years, thus uncertainty of ET_{PNV} is more closely correlated with a possible change in trends than with intrinsic variability of vegetation. Other potential sources of uncertainties in the equation-based and the model-based evapotranspiration results are provided in Piñol et al. (1991) and Zhang et al. (2010), respectively. Additional data may favour alternative delineations of biomes and ecoregions. Likewise, if climate data on a more detailed spatial scale were available, ET_{PNV} could be regionalized accordingly.

Intra-annual temporal variability was not included in the estimation of PNV water consumption, as values are on a yearly basis. Monthly assessments might be required to increase model details. However, due to the complex interactions of groundwater, soil moisture and evaporation processes, it is recommended to always analyze soil moisture use of the studied system over a complete year or even including multi-annual rotations in the case of crops, as explained in Núñez et al. (2012). Therefore, annual assessments might be more practical and not lead to additional uncertainties.

For regionalization at the ecoregion scale, we used the Olson et al. (2001) classification system. The corresponding global map of biomes and ecoregions is built upon widely recognized biogeographic maps, which enhances scientific acceptance and its utility in different regions. In addition, a recent LCA-related publication aimed at standardising regionalisation of land-use flows has recommended this classification system as the basis for registering the geographic information during the LCI phase (Koellner et al., 2012). It has also been used to report regionalized impact factors for land use (de Baan et al., 2012; Müller Wenk and Brandao, 2010; Pfister et al., 2010). The strong link between land use and soil-water consumption would suggest using the same system for regionalisation of flows, since the watershed perspective is relevant for blue water but generally green water (soil moisture) is a local land use feature. However, ecoregions are units of spatial aggregation that are less relevant for soil-water consumption. This would suggest that finding a common regionalization approach for land and water use impact pathways is not a straightforward task. One solution to address this issue could be to base the ecoregion level assessment on both the gridded aridity index and ecoregion approaches. Combining both aggregation levels shows that ecoregions with low $k_{opt,r}$ values (~<1.63) would result in higher ET_{PNV} losses than those reported in this study. This is due to the larger values of the combined k, which was calculated as the arithmetic average of $k_{opt,ai}$ and $k_{opt,r}$ for a given ecoregion. On the other hand, ecoregions with large $k_{opt,r}$ values (~>1.63) would result in lower ET_{PNV} rates (figure A8.3 in the annex). Regional differences are therefore compensated on the global scale.

8.4.2. Life cycle impact assessment step

Although the definition of CFs for the consumption of soil-water was not the main objective of this study, we suggest three screening frameworks that could be further investigated to better account for the effects that soil-water-use dependent land uses have on the environment. The three indicators we propose are to express the effects of the soil-water consumption at the classical midpoint level of impact categories such as acidification or ozone depletion. The first proposal entails using an adjusted inventory flow that simply expresses the total amount of net soil-water consumed. This is done by setting the characterization factor value at one (CF=1, with units of volume, e.g., m^3/m^3), which should be interpreted as if all water uses occurred in the highest water stress situation. This CF, however, does not reflect the ecological impacts of the net change in soil-water consumption. For the second approach, we propose adopting any of the regionalized water scarcity indices used as CFs in midpoint assessment methods for water consumption withdrawn from water bodies (e.g., freshwater ecosystem impact in Milà i Canals et al., 2009; water stress index in Pfister et al., 2009). They would then be used to evaluate $NET_{soil\text{-}water}$. Nevertheless, using these water scarcity indices to make the soil-water consumption assessment LCA compatible is not entirely relevant as such indices do not account for soil-moisture reserves, but only for freshwater reserves in rivers and aquifers (Núñez et al., 2012). Moreover, there is no evidence that additional evapotranspiration of 1 m^3 of soil moisture leads to a corresponding loss of 1 m^3 downstream water. This depends very much on local conditions and only soil moisture may decrease while water flow in downstream rivers and aquifers may not be affected at all. On the other hand, as blue water availability is determined by the consumption of green water upstream, we have considered that CFs for blue water consumption are still suitable until more sophisticated approaches become available. The third suggestion consists in weighting $NET_{soil\text{-}water}$ as a function of natural water availability, taking either the aridity index ratio (P/ET_0) or the ET_{PNV} as an indicator of the available local soil-water. This indicator serves to inform about the severity of the net soil-water consumption in function of the local water reserves. The structure of the indicator is in line with the critical flow concept used in the ecological scarcity method (Frischknecht et al., 2006).

8.4.3. Outlook

Further research is required at the intersection of land use and water consumption issues. As highlighted before, no impacts apart from those relating to water scarcity are explicitly captured. Addressing how to treat reduced soil-water consumption (negative values of $NET_{soil\text{-}water}$) is a key issue since it can have negative effects, as already illustrated in Milà i Canals et al. (2009): saturated soils are contributing to fast runoff as well as impervious areas, which can be highly problematic as they contribute to increased levels of erosion, waterlogging, flooding, salinization and even water cycle disturbances, which might affect precipitation patterns and hence water availability. As a rule, natural vegetation has higher soil-water requirements than crop lands, but they are also more complex ecosystems with high level of biodiversity and deliver fundamental ecosystem services, such as carbon sequestration or erosion regulation. There is therefore a real need to evaluate land use effects other than changes in soil evapotranspiration patterns to obtain a comprehensive environmental perspective. From the methodology proposed here for proper accounting of LCI soil-water consumption flows, the next research step should focus on developing LCIA models and CFs for a complete integration of green water consumption in LCA. Figure A8.4 in the annex illustrates this research need and the effects that land-use changes have on both changes in the soil ecological quality, and direct soil-water uptake. Finally, better climate data and hydrological models are required on a global scale to drive the development of CFs for green water consumption.

8.5. References

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9. Annex

9.1. Annex to Chapter 1

Estimation of river volumes

Estimation of the river volume was based on data from Xenopoulos et al. (2005), Alcamo et al. (2003), Hugueny (1989), Fekete et al. (2000), Döll et al. (2003), and EarthTrends Watersheds of the World (2007). The water volume of a river can be calculated by:

$$V_i = Q_i \cdot \tau_i \tag{9.1.1}$$

where V_i is the water volume of river i (m^3), Q_i is the average discharge of river i ($m^3 \cdot s^{-1}$) and τ_i is the average residence time of the water in river i (s).

The average river discharge was calculated by:

$$Q_i = \frac{Q_{mouth,i}}{2} \tag{9.1.2}$$

where $Q_{mouth,i}$ is the discharge at the mouth of river i ($m^3 \cdot s^{-1}$) available from WaterGap (Alcamo et al. 2003). The average distance travelled by each raindrop will depend on the river network pattern. By dividing $Q_{mouth,i}$ by 2 to estimate the spatially averaged discharge, we assume that the average distance travelled is half of the river's total length.

The average residence time (in s) was obtained from the river's total length and the average river water velocity:

$$\tau_i = \frac{L_i/2}{v_i} \tag{9.1.3}$$

where L_i is the length of river i (m) and v_i is the average velocity of river i ($m \cdot s^{-1}$). Again, we assumed that the average distance travelled of the water is half of the river's total length.

Based on Allen et al. (1994), a typical river velocity can be derived from river discharge data via:

$$v_i = 1.067 \cdot Q_i^{0.1035} \tag{9.1.4}$$

where v_i is the river velocity ($m \cdot s^{-1}$) and Q_i is the average river discharge in river basin i ($m^3 \cdot s^{-1}$).

Feeding equation 9.1.4 into 9.1.3, and equations 9.1.2 and 9.1.3 into equation 9.1.1 reveals that:

$$V_i = \frac{Q_{mouth,i}}{2} \cdot \frac{L_i/2}{1.067 \cdot \left(\frac{Q_{mouth,i}}{2}\right)^{0.1035}} = 0.47 \cdot \left(\frac{Q_{mouth,i}}{2}\right)^{0.90} \cdot L_i \tag{9.1.5}$$

Derivation of $dQ_{mouth,i}/dTEMP$

The derivation of $dQ_{mouth,i}/dTEMP$ for all the 214 river basins was taken as a starting point in the calculation of the effect factor for global warming, using year 2100 as a future reference year. The river basin-specific $dQ_{mouth,i}/dTEMP$ was calculated by dividing the discharge at the mouth of each river basin with the global mean temperature change in 2100. As reported in IPCC (2001) and Millennium Ecosystem Assessment (2005), global mean temperature changes are projected within the range of 1.9 to 4.4 by the year 2100, depending on the scenario chosen (see Table 9.1.1). The effect factors were calculated for five global climate scenarios to project freshwater fish species loss for the year 2100 by multiplying $dQ_{mouth,i}/dTEMP$ with $dPDF_i/dQ_{mouth,i}$ over all river basins included.

Table 9.1.1. Summary of the five global climate scenarios considered in the present study (IPCC 2001 and Millennium Ecosystem Assessment 2005).

Scenario	**Summary**	**Global mean temperature change in 2100 (°C)**
A2	A heterogeneous world with continuously increasing population growth rate. Regionalized and fragmented economic growth and slow technological change.	4.4
B2	A world with intermediate levels of economic and population growth, and emphasize on local solutions to economic, social, and environmental sustainability. Technological change is faster than A2.	3.2
FW	Regionalized and fragmented world. Reactive approach to the global environmental problems. High population growth with low economic development and technological change. The gap between rich and poor countries increases over time.	3.3
GO	Strong global action with emphasis on trade and economic growth. Offer an equal access on public goods and services. Reduce poverty by improving human well-being. Reactive approach to the global environmental problems.	3.5
TG	Strong global action, with emphasis on green technology. High economic growth. Proactive approach to the global environmental problems using technology and market-oriented institutional reform. Focusing on economic, education and human well-being. Symbiotic benefits for both the environment and economy.	1.9

Influence of including river basins located above 42°

Greenhouse gas emissions

Figure 9.1.1 shows the effect factors for greenhouse gases for five global scenarios in 214 and 297 river basins. The average effect factor for 214 river basins included is $2.04 \cdot 10^9$ PDF·.m^3·°C^{-1}. When including other river basins that located at the higher latitude (> 42°), the average effect factor increases to $2.07 \cdot 10^9$ PDF·m^3·°C^{-1}. A relatively high potential freshwater fish species loss is reported in

B2 scenario per degree of temperature increase compared to the other future scenarios. This finding can be explained by the fact that in the B2 scenario the decrease in water discharge is predicted is due to the low water discharge in this scenario compared to other scenarios in rivers with the highest effect factors, i.e. the rivers below 42 degrees latitude with the highest river length. This results in a relatively high value for *dPDF/dQ* in the B2 scenario.

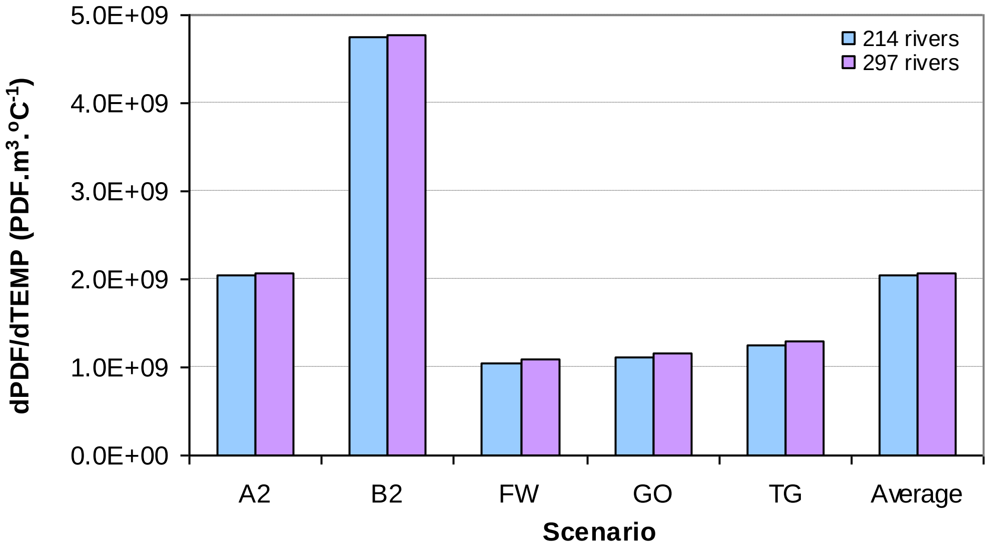

Figure 9.1.1. The effect factors for greenhouse gas emissions (PDF·m^{3}·°C^{-1}) based on IPCC and MA scenarios for 214 and 297 river basins, respectively.

Normalization factors

Characterization factors, water consumption in year 1995 and normalization factors for water consumption for 112 river basins were included. Due to lack of data, we were not able to derive normalization factors for all river basins considered in this study. To derive normalization factors for water consumption, water withdrawal data for households, irrigation, industry, and livestock sectors representative for year 1995 were taken from WaterGap model (Alcamo et al. 2003a, 2003b) as a starting point. We converted from water withdrawal to water consumption by using continent-specific water withdrawal-consumption ratios derived from Shiklomanov (1999), i.e. for Europe = 43%, North America = 34%, Africa = 72%, Asia = 62%, South America = 53% and Australia = 58%. The total population for the 112 river basins included is 2.65·10^{9}. Normalization factors were expressed in unit of the potentially disappeared fraction of fish species for river-specific water consumption (PDF·m^{3}/capita). The total normalization factor for direct water consumption (NF$_{wc}$) was calculated by:

$$NF_{wc} = \frac{\sum_i W_i \cdot CF_i}{\sum_i N_i} \tag{9.1.6}$$

where W_i is the water consumption in river basin *i* (m^{3}·yr^{-1}), CF_i is the characterization factor for river basin *i* (PDF·m^{3}·yr·m^{-3}) and N_i is the number of capita in river basin *i* (capita).

The normalization factors for global warming were based on the global greenhouse gas emissions for year 2000. The total population numbers in the world in 2000 is 6.1·10^{9} (U.S. Cencus Bureau 2010). Normalization factors were expressed in unit of the potentially disappeared fraction of fish species over a certain river volume due to global greenhouse gas emissions in 2000 ($PDF \cdot m^3$/capita). The total normalization factor for greenhouse gas emissions (NF_{ghg}) was calculated by:

$$NF_{ghg} = \frac{\sum_x M_x \cdot CF_x}{N_{world}} \quad (9.1.7)$$

where M_x is the emitted quantity of a substance x (kg), CF_x is the characterization factor for substance x ($PDF \cdot m^3 \cdot yr \cdot kg^{-1}$) and N_{world} is the total number of capita in the world in year 2000 (capita).

Table 9.1.2. River characteristics for 214 river basins below 42 degrees latitude (1 – 6) and river-specific effect factor for global warming. Due to increased precipitation, the river discharge rate is predicted to increase in some areas (Rosenzweig et al. 2007). River basins with increased discharge were excluded in the calculation of the effect factor for global warming.

* - River basins with increased discharge.

River basins below 42 degrees latitude	River length (km)	Average river discharge at the mouth ($km^3 \cdot yr^{-1}$)	Calculated river volume (m^3)	Effect factor for global warming ($PDF \cdot m^3 \cdot {}^oC^{-1}$)
Nil (Af., int.)	5909	75.87	1.60E+09	*
Senegal (Guinée-Sénégal)	1680	9.94	7.34E+07	6.12E+06
Gambia (Guinée-Gambie)	745	6.76	2.31E+07	1.29E+06
Tominé ou Rio Corubal (Guinée-Guineé Bissau)	463	17.82	3.42E+07	9.67E+05
Konkouré (Guinée)	303	13.08	1.69E+07	2.70E+05
Kolenté (Guinée, Great Scarcies)	240	28.05	2.66E+07	4.07E+05
Jong (Sierra Leone)	249	17.85	1.84E+07	1.09E+05
Sewa (Sierra Leone)	240	17.41	1.73E+07	1.24E+05
Moa (Guinée-Sierra Leone)	425	26.14	4.42E+07	2.52E+05
Mano (Libéria)	276	10.79	1.30E+07	2.66E+04
Loffa (Guinée-Libéria)	349	15.81	2.31E+07	4.37E+04
St Paul (Libéria)	410	36.46	5.75E+07	2.26E+04
Nipoué (Cess, Libéria-RCI)	332	16.94	2.34E+07	*
Cavally (Libéria-RCI)	379	25.70	3.88E+07	*
Dodo (aka Déo) (RCI)	89	19.35	7.04E+06	*
San Pédro (RCI)	193	2.55	2.49E+06	*
Sassandra (RCI)	569	30.61	6.82E+07	*
N'Zo (a. Sassandra) (RCI)	243	3.25	3.91E+06	*
Boubo (RCI)	130	2.69	1.76E+06	*
Bandama (RCI)	692	23.26	6.48E+07	*
Yani (s.a. Bandama) (RCI)	167	23.26	1.56E+07	*
Marahoué (a. Bandama) (RCI)	249	12.76	1.36E+07	*
N'Zi (a. Bandama) (RCI)	472	6.86	1.48E+07	*
Kan (s.a. Bandama) (RCI)	629	23.26	5.89E+07	*

River basins below 42 degrees latitude	River length (km)	Average river discharge at the mouth ($km^3 \cdot yr^{-1}$)	Calculated river volume (m^3)	Effect factor for global warming ($PDF \cdot m^3 \cdot {}^oC^{-1}$)
Agnébi (RCI)	281	2.76	3.89E+06	*
Comoé (RCI-Burkina)	750	5.92	2.06E+07	*
Bia (RCI-Ghana)	260	4.32	5.38E+06	5.53E+03
Volta (Ghana-Burkina)	1301	32.76	1.66E+08	*
Black Volta (Burkina-Ghana) (a. Volta)	1352	8.11	4.93E+07	*
Nasia (a. White Volta) (Ghana)	219	8.52	8.33E+06	3.00E+04
Daka (a. Volta) (Ghana)	106	21.45	9.23E+06	*
Mono (Togo)	412	3.47	7.01E+06	*
Ouémé (Bénin)	480	6.22	1.38E+07	*
Ogun (Nigéria)	410	5.81	1.11E+07	*
Niger (Afr. Int.)	4200	147.43	2.06E+09	*
Niandan (Guinée) (a. Niger)	344	18.69	2.65E+07	4.59E+05
Bénoué (Nigéria-Cameroun) (a. Niger)	1400	68.68	3.46E+08	*
Sokoto (a. Niger) (Nigeria)	275	29.01	3.14E+07	5.24E+04
Cross (Nigéria-Cameroun)	480	59.93	1.05E+08	*
Mungo (Cameroun)	13	8.27	4.82E+05	3.29E+02
Dibamba (Cameroun)	150	18.78	1.16E+07	1.48E+04
Wouri (Cameroun)	160	18.78	1.24E+07	1.58E+04
Sanaga (Cameroun)	803	63.81	1.86E+08	3.69E+05
Nyong (Cameroun)	402	22.14	3.60E+07	8.34E+04
Lokoundjé (Cameroun)	185	3.12	2.87E+06	2.85E+03
Kribi ou Kienké (Cameroun)	100	3.12	1.55E+06	1.54E+03
Lobé (Cameroun)	80	3.12	1.24E+06	1.23E+03
Ntem (Cameroun-Gabon-Guinée équat.)	356	19.21	2.81E+07	1.50E+04
Ogôoué (Gabon)	815	155.06	4.18E+08	9.00E+05
Niari-Kouilou (Congo)	481	28.35	5.38E+07	6.76E+05
Zaïre (Afr., Int.)	4339	1348.39	1.55E+10	5.11E+06
Cunene ou Kunene (Namibie-Angola)	828	8.80	3.24E+07	6.18E+05
Kasaï (a. Zaïre) (Zaïre-Angola)	2153	573.15	3.57E+09	*

River basins below 42 degrees latitude	River length (km)	Average river discharge at the mouth ($km^3 \cdot yr^{-1}$)	Calculated river volume (m^3)	Effect factor for global warming ($PDF \cdot m^3 \cdot {}^oC^{-1}$)
Chari (Lac Tchad)	1733	25.45	1.76E+08	*
Ubangi (a. Zaïre) (Congo-RCA)	2300	177.98	1.34E+09	*
Zambezi (Mozambique-Zambie-Angola)	2693	120.36	1.10E+09	6.78E+07
Tana (Kénya)	671	7.31	2.23E+07	1.13E+05
Rufiji (Tanzanie)	809	30.49	9.66E+07	2.33E+05
Limpopo (Botswana-Mozamb.-Rhodésie-RSA)	1800	9.12	7.28E+07	3.45E+06
Pongolo ou Maputo (RCA-Mozambique)	347	4.73	7.80E+06	1.42E+05
Shire (a.) (Malawi-Mozambique)	1200	119.70	4.88E+08	3.00E+07
Kafue (a. Zambèze) (Zambie)	960	17.84	7.09E+07	3.80E+06
Ruaha (a. Rufiji) (Tanzanie)	475	30.49	5.67E+07	1.37E+05
Evros-Mariça (Grèce-Turquie-Bulgarie)	415	11.01	1.99E+07	5.58E+05
Nesta-Nestos (Grèce-Bulgarie)	230	1.91	2.29E+06	1.08E+05
Strymon-Strouma (Grèce-Bulgarie)	389	3.40	6.50E+06	2.45E+05
Agly (France)	82	1.38	6.12E+05	4.48E+03
Minho (Portugal-Espagne)	350	11.20	1.70E+07	1.20E+04
Lima (Portugal)	108	3.23	1.72E+06	1.63E+03
Cavado (Portugal)	135	3.23	2.15E+06	2.04E+03
Douro (Portugal-Esp.)	555	23.10	5.17E+07	5.06E+05
Vouga (Portugal)	148	1.67	1.31E+06	2.13E+03
Mondego (Portugal)	234	2.78	3.27E+06	7.06E+03
Sado (Portugal)	175	1.28	1.22E+06	1.22E+04
Mira (Portugal)	145	0.33	3.01E+05	2.74E+03
Guadiana (Portugal-Esp.)	766	7.86	2.71E+07	4.20E+05
Raisin (Canada)	217	178.89	1.27E+08	1.05E+05
Sydenham (Canada)	165	162.07	8.81E+07	7.69E+04
Grand river (Canada)	280	196.50	1.78E+08	1.28E+05
Thames (Canada)	270	4.90	6.26E+06	*
Missisipi (USA)	4185	530.64	6.47E+09	8.21E+06
Rio Grande (USA-Mexique)	2219	8.00	7.98E+07	2.10E+05

River basins below 42 degrees latitude	River length (km)	Average river discharge at the mouth ($km^3 \cdot yr^{-1}$)	Calculated river volume (m^3)	Effect factor for global warming ($PDF \cdot m^3 \cdot {}^oC^{-1}$)
Pecos (a. Rio Grande)	1490	6.55	4.48E+07	*
Canadian (s. a. Missisipi) (USA)	1223	4.59	2.67E+07	3.05E+05
Colorado (USA-Mexique)	1750	1.35	1.28E+07	3.03E+04
San Juan (a. Colorado) (USA)	375	15.72	2.47E+07	7.52E+03
Zuni (s. a. Colorado) (a. Little Colorado)	145	15.05	9.19E+06	3.42E+03
San Francisco (a. Gila) (USA)	2212	0.04	7.17E+05	*
Gila (a. Colorado)	1044	0.68	4.13E+06	*
Ohio river (a. Missisipi)	2102	240.85	1.60E+09	*
Scioto River (a. Ohio)	372	83.41	1.09E+08	*
Big Darby Creek (s. a. Ohio) (a. Scioto)	135	3.89	2.55E+06	6.29E+02
Wabash River (a. Ohio)	764	147.34	3.75E+08	*
Little Wabash River (a. Wabash)	320	147.34	1.57E+08	*
Embarras River (a. Wabash)	298	12.70	1.62E+07	1.93E+04
St Joseph River (s.a. Wabash)	160	2.84	2.27E+06	6.79E+03
Elk river (s. a. Ohio) (a. Kanawha)	277	9.56	1.17E+07	*
Cumberland river (a. Ohio)	1106	93.04	3.59E+08	*
Green river (a. Ohio)	1175	118.14	4.73E+08	*
Kanawha river (a. Ohio)	156	65.60	3.70E+07	*
Tennessee River (a. Ohio)	1049	240.85	7.99E+08	*
Muskingum River (s.a. Ohio) (a. Allegheny)	179	36.46	2.51E+07	*
Allegheny river (a. Ohio)	523	10.61	2.42E+07	*
Little Miami river (a. Ohio)	170	93.83	5.56E+07	*
Hocking river (a. Ohio)	153	47.46	2.72E+07	*
Kinniconick river (a. Ohio)	159325	85.56	4.80E+10	*
Licking River (a. Ohio)	65	93.83	2.13E+07	*
Little Scioto river (a. Ohio)	65	2.60	8.54E+05	8.11E+01
Ohio Brush Creek (a. Ohio)	102	83.41	3.00E+07	*
Olentangy River (a. Little Scioto)	98	1.91	9.78E+05	2.24E+02
Paint Creek (a. Scioto river)	153	6.97	4.85E+06	*

River basins below 42 degrees latitude	River length (km)	Average river discharge at the mouth ($km^3 \cdot yr^{-1}$)	Calculated river volume (m^3)	Effect factor for global warming ($PDF \cdot m^3 \cdot {}^oC^{-1}$)
Scioto Brush Creek (a. Scioto)	57936	83.41	1.71E+10	*
Symmes River (a. Ohio)	97	67.74	2.37E+07	*
Tygart Creek (a. Ohio)	257	2.72	3.52E+06	*
Bear Creek	46	1.17	2.94E+05	*
Apalachicola (USA)	180	24.24	1.75E+07	1.07E+04
Klamath (USA)	318	19.77	2.57E+07	2.26E+05
Mobile (USA)	72	60.65	1.59E+07	*
Potomac (USA)	297	11.02	1.42E+07	*
Sabine (USA)	564	12.92	3.12E+07	5.05E+05
Sacramento (USA)	927	36.79	1.31E+08	1.81E+06
Savannah (USA)	457	11.18	2.22E+07	*
Susquehanna (USA)	514	33.01	6.59E+07	*
Connecticut river (USA)	497	17.64	3.63E+07	*
Missouri (USA)	3767	192.83	2.35E+09	8.42E+06
Arkansas river (USA)	2364	547.14	3.76E+09	3.28E+06
Red river (USA)	2188	522.08	3.33E+09	4.00E+06
Altamaha (USA)	449	13.33	2.55E+07	9.55E+03
Balsas (Mexico)	706	24.85	7.02E+07	1.34E+06
Panuco (Mexico)	490	17.15	3.49E+07	6.76E+05
Sucio (a. Lempa) (San Salvador)	25	13.42	1.43E+06	7.85E+04
Paz (San Salvador)	134	4.47	2.86E+06	1.52E+05
San Tiguel (ou Miguel) San Salvador)	145	1.30	1.02E+06	1.07E+05
Paraguay (Brésil-Arg.-Paraguay) (a. Parana)	2549	539.87	4.00E+09	*
Uruguay (Brésil-Arg.-Uruguay)	1424	181.85	8.43E+08	*
Magdalena (Colombie)	1271	218.38	8.87E+08	9.30E+06
Rio Negro (a. Amazone) (Colomb.-Venez.-Brésil)	1112	4067.95	1.07E+10	8.24E+07
Parnaiba (Brésil)	1192	26.62	1.26E+08	5.25E+06
Madeira (a. Amazone) (Brésil-Bolivie)	3239	5010.21	3.75E+10	2.93E+08
Orinoco (Vénézuela-Colombie)	1970	1096.40	5.84E+09	1.85E+08

River basins below 42 degrees latitude	River length (km)	Average river discharge at the mouth ($km^3 \cdot yr^{-1}$)	Calculated river volume (m^3)	Effect factor for global warming ($PDF \cdot m^3 \cdot {}^{o}C^{-1}$)
Parana (Brésil-Paraguay-Argentine)	2748	601.89	4.76E+09	*
Tibagi (Bresil)	550	11.51	2.74E+07	1.69E+04
Amazon (Br. Mère Maranon) (Pérou-Brésil)	4327	6394.15	6.23E+10	6.17E+08
Maroni (Guyane-Surinam)	445	57.17	9.34E+07	6.07E+06
Oyapock (Guyane-Brésil)	291	40.17	4.45E+07	2.28E+06
Approuague	270	10.68	1.26E+07	5.98E+05
Sinnamary (Guyane)	250	12.16	1.31E+07	6.78E+05
Kourou (Guyane)	112	6.90	3.53E+06	1.75E+05
Vakhsh ou Vachs (fSU) (a. Amu Darya)	1976	51.29	3.76E+08	*
Surkhandarya ou Surchandarya (fSU)	175	54.58	3.52E+07	*
Zeravshan (a. Syr Darya) (fSU)	1615	59.28	3.50E+08	*
Naryn (a. Syr Darya) (fSU)	807	16.27	5.49E+07	*
Tarim (Chine)	1227	2.23	1.40E+07	*
Murgab ou Murghab ou Mourbab (fSU-Afghanistan) Endo	850	2.78	1.19E+07	3.62E+04
Kabul (a. Indus) (Afghanistan-Inde)	700	84.80	2.09E+08	*
Salween (Tibet-Chine-Birmanie-Thaï)	2576	98.52	8.80E+08	*
Mae Khlong (Thaïlande)	145	21.06	1.24E+07	*
Chao Phrya (Menam) (Thaïlande)	710	27.48	7.72E+07	*
Mekong (Asie Sud-Est, Int.)	3977	421.80	5.01E+09	*
Kelani Ganga(Sri Lanka)	145	3.85	2.71E+06	6.55E+03
Kalu Ganga (Sri Lanka)	129	3.85	2.41E+06	5.82E+03
Gin Ganga (Sri Lanka)	116	2.38	1.41E+06	*
Nilwala Ganga (Sri Lanka)	72	4.32	1.49E+06	3.10E+03
Mahaweli Ganga (Sri Lanka)	335	3.44	5.66E+06	7.77E+04
Brahmapoutre ou Tsangpo (Inde-Bengladesh-Tibet)	2897	1186.94	9.22E+09	1.06E+08
Indus (Tibet-Inde-Pakistan)	2382	121.17	9.80E+08	2.34E+06
Gange (Inde)	2221	397.83	2.65E+09	9.78E+07
Ob (fSU)	3977	413.18	4.91E+09	*
Yangzi Jiang (Tibet-Chine)	6380	955.40	1.67E+10	*

River basins below 42 degrees latitude	River length (km)	Average river discharge at the mouth ($km^3 \cdot yr^{-1}$)	Calculated river volume (m^3)	Effect factor for global warming ($PDF \cdot m^3 \cdot {}^{o}C^{-1}$)
Gandaki river (a. Gange) (nepal)	630	1186.94	2.00E+09	3.87E+07
Sakaria (Turkey)	506	7.78	1.78E+07	4.82E+05
Rakaïa river (New-Zealand)	150	4.74	3.38E+06	*
Fly (Nlle-Guinée)	678	135.37	3.08E+08	4.34E+06
Sepik-Ramu (Nlle-Guinée)	285	100.67	9.93E+07	1.76E+06
Kapuas (Bornéo)	569	174.16	3.24E+08	*
Murray-Darling (Australie)	1767	11.14	8.55E+07	2.83E+06
Yellow (Huang He, Huang Ho, China)	4168	56.53	8.66E+08	*
Yangtze (Chang Jiang, Yangtze Kiang, China)	4734	955.94	1.24E+10	*
Xi Jiang River (Pearl River, Chu Chiang, Zhu, Southeast China)	1696	270.52	1.43E+09	*
Tsengwen River (Southwestern Taiwan)	130	1.29	9.12E+05	1.42E+03
Tigris (Southeast Turkey and Iraq)	1950	34.43	2.60E+08	4.97E+06
Tanshui (Northern Taiwan)	328	2.47	4.12E+06	*
Tano (West Africa)	400	4.52	8.63E+06	1.77E+04
Saloum (West Africa)	105	0.53	3.30E+05	2.48E+04
Saint John (West Africa)	616	25.02	6.16E+07	*
Rokel River (Seli River, West Africa)	386	13.16	2.17E+07	2.34E+05
Purus (Northwest central South America)	3379	2888.58	2.39E+10	1.14E+08
Pra River (West Africa)	245	7.14	7.96E+06	2.38E+04
Pilcomayo (South central South America)	2500	86.50	7.60E+08	1.14E+06
Pará-Tocantins (Brazil)	2234	376.40	2.54E+09	4.67E+07
Orange (South Africa)	1840	8.44	6.95E+07	3.40E+06
Ombrone (Tuscany, Western Italy)	130	1.56	1.08E+06	5.20E+03
Okavango (Southwest central Africa)	1600	23.78	1.53E+08	6.29E+06
Marañon (Peru)	1415	5.37	3.56E+07	1.09E+05
Little Scarcies (West Africa)	280	14.87	1.76E+07	2.53E+05
Kwando River (Southwest Africa/Namibia)	731	34.09	9.65E+07	8.25E+06
Kura (Russia and Turkey)	796	22.00	7.09E+07	4.63E+05
Krishna (Karnataka, India)	1091	107.26	4.02E+08	3.44E+06

River basins below 42 degrees latitude	River length (km)	Average river discharge at the mouth ($km^3 \cdot yr^{-1}$)	Calculated river volume (m^3)	Effect factor for global warming ($PDF \cdot m^3 \cdot {}^{o}C^{-1}$)
Kogon (Guinea, West Africa)	256	10.75	1.22E+07	2.01E+05
Kaoping River (Southern Taiwan)	171	4.29	3.52E+06	1.28E+04
Irrawaddy River (Irawadi, Central Myanmar Burma)	1781	564.35	2.91E+09	*
Godavari (Central India)	950	107.26	3.50E+08	2.99E+06
Géba (Guinea Bissau, West Africa)	547	3.95	1.04E+07	4.04E+05
Ganges (Ganga, North and northeast Indian subcontinent)	2221	1045.01	6.30E+09	8.31E+07
Fatala (West Africa)	205	13.09	1.15E+07	1.83E+05
Euphrates (Firat Nehri, Al-Furat, Southwest Asia)	2289	19.60	1.84E+08	4.42E+06
Erhjen River (Southern River)	36	4.29	7.40E+05	5.34E+03
Chobe River (Southwest Africa/Namibia)	1500	34.09	1.98E+08	1.69E+07
Chittar (Tamil Nadu, India)	80	0.00	2.13E+03	6.22E+01
Cauvery (Karnataka, India)	627	7.59	2.15E+07	1.31E+06
Casamance (West Africa)	320	3.49	5.47E+06	2.83E+05
Brahmaputra (Dyardanes, Oedanes, Tsangpo, Zangbo, Tibet, China, NE India and Bangladesh)	2948	1045.48	8.37E+09	1.10E+08
Araguaia (Araguaya, Central Brazil)	2627	183.47	1.57E+09	2.32E+07
Athi-Galana-Sabaki River Drainage System (Kenya, from Nairobi eastward to Mombasa)	962	3.99	1.85E+07	3.18E+04

Table 9.1.3. River characteristics for 83 river basins above 42 degrees latitude (Xenopoulos et al. 2005, Alcamo et al. 2003, Hugueny 1989, Fekete et al. 2000, Döll et al. 2003, and EarthTrends Watersheds of the World 2007) and river-specific effect factor for global warming.

* - River basins with increased discharge.

River basin above 42 degrees latitude	River length (km)	Average river discharge at the mouth ($km^3 \cdot yr^{-1}$)	Calculated river volume (m^3)	Effect factor for global warming ($PDF \cdot m^3 \cdot {}^oC^{-1}$)
Scorff (a. Blavet) (France)	75	1.9098	7.47E+05	*
Seine (France)	451	17.124	3.21E+07	*
Lot (a. Garonne) (France)	481	19.359	3.82E+07	2.33E+05
Garonne (France-Espagne)	484	21.098	4.15E+07	2.72E+05
Dordogne (a. Garonne)	483	30.929	5.84E+07	3.26E+05
Po (Italie)	500	52.048	9.64E+07	3.71E+05
Rhin (Suisse-All.-Neth.)	1018	79.748	2.88E+08	1.97E+05
Meuse (France-Belg.-NL)	565	12.816	3.10E+07	*
Nida (a. Vistule) (Pol.)	151	35.927	2.09E+07	9.02E+03
Pilica (a. Vistule) (Pol.)	319	18.886	2.48E+07	6.36E+04
Warta (a. Oder) (Pol.)	808	19.755	6.54E+07	9.21E+04
Lyna ou Lava (Pol.)	264	4.599	5.78E+06	*
Bzura (a. Vistule) (Pol.)	166	30.83229	2.00E+07	1.37E+04
Raba (Pol.)	137.4	7.484	4.66E+06	2.28E+04
Vistula (Pol.)	1014	35.927	1.40E+08	6.06E+04
Morava (a. Danube) (Tch.-Autriche)	354	65.878	8.43E+07	2.13E+05
Volga (fSU)	2785	234.33	2.07E+09	1.50E+06
Danube (Int.)	2222	218.517	1.55E+09	1.65E+07
Loire (France)	839	31.714	1.04E+08	*
Yèrres (a. Seine)	6	8.385	2.25E+05	*
Yonne (a. Seine) (France)	292	4.605	6.40E+06	*
Touques (France)	104	1.4128	7.91E+05	*
Dives (France)	105	2.02	1.10E+06	*
Vire (France)	128	0.5467	4.16E+05	*
Doubs (s.a. Rhône) (a. Saône) (France)	453	14.108	2.71E+07	1.17E+04
Gudena (Danemark)	158	1.4956	1.26E+06	2.33E+04
Wye (Severn estuary) (Wales)	297	5.679	7.86E+06	*
Tees (Britain)	132	2.223	1.51E+06	*
Glama (Norvège)	490	21.935	4.36E+07	*

River basin above 42 degrees latitude	River length (km)	Average river discharge at the mouth ($km^3 \cdot yr^{-1}$)	Calculated river volume (m^3)	Effect factor for global warming ($PDF \cdot m^3 \cdot {}^oC^{-1}$)
Dunajec (a. Vistule) (Pologne-Slovaquie)	251	7.484	8.51E+06	4.17E+04
Hérault (France)	148	3.154	2.31E+06	5.76E+03
Orb (France)	136	3.154	2.12E+06	5.29E+03
Tarn (a. Garonne)	381	4.106	7.54E+06	2.18E+04
Allier (a. Loire) (France)	421	9.768	1.81E+07	7.43E+03
Ain (a. Rhône)	190	15.143	1.21E+07	3.21E+04
Isère (a. Rhône)	286	45.661	4.91E+07	9.82E+04
Sorgues (s. a. Rhône)	46.4	48.389	8.38E+06	1.56E+04
Ardèche (a. Rhône)	125	48.389	2.26E+07	4.21E+04
Cèze (a. Rhône)	128	48.389	2.31E+07	4.31E+04
Gard (a. Rhône)	133	54.374	2.67E+07	5.50E+04
Rhône (France-Suisse)	637	54.3377	1.28E+08	2.65E+05
Saône (a. Rhône)	473	32.608	6.00E+07	8.81E+04
Durance (a. Rhône)	324	54.374	6.50E+07	1.34E+05
Arve (a. Rhône)	102	9.282	4.19E+06	2.08E+04
Fier (a. Rhône)	71.9	11.742	3.65E+06	1.24E+04
Bourbre (a. Rhône)	72.2	15.143	4.60E+06	1.22E+04
Eyrieux (a. Rhône)	83	45.661	1.42E+07	2.85E+04
Drôme (a. Rhône)	110	45.661	1.89E+07	3.78E+04
Willamette (a. Columbia) (USA)	301	216.664	2.08E+08	*
St Laurent (Canada)	3175	366.784	3.53E+09	*
Moisie river (Canada)	343	14.071	2.05E+07	*
Ganaraska (Canada)	49.6740 093	200.547	3.21E+07	2.60E+04
Humber (Canada)	100	1.826	9.57E+05	*
Credit (Canada)	1500	1.164	9.59E+06	*
Au Sable (Canada)	240	0.8954	1.21E+06	*
Maitland (Canada)	150	2.083	1.62E+06	*
Saugeen (Canada)	160	2.927	2.34E+06	*

River basin above 42 degrees latitude	River length (km)	Average river discharge at the mouth ($km^3 \cdot yr^{-1}$)	Calculated river volume (m^3)	Effect factor for global warming ($PDF \cdot m^3 \cdot {}^oC^{-1}$)
South Nation (Canada)	175	53.092	3.44E+07	*
Mackenzie (Canada)	3679	267.295	3.08E+09	*
Yukon (Canada-U.S.A.)	2716	187.187	1.65E+09	*
Amu Darya (fSU)	1976	50.257	3.69E+08	*
Syr Darya (fSU)	1615	21.326	1.40E+08	*
Talas (fSU)	661	3.938	1.26E+07	*
Chu ou Tchou (fSU)	1067	3.995	2.06E+07	*
Ili (Chine-fSU) (Lac Balkhach)	1400	4.1855	2.82E+07	*
Léna (fSU)	4387	540.007	6.89E+09	*
Amour (fSU-Chine)	5061	330.454	5.12E+09	*
Dvina (ex-fSU)	1441	101.23877	5.05E+08	*
Neva (ex-fSU)	911	3.38614	1.52E+07	*
Dniepr (ex-fSU)	1544	48.18512	2.78E+08	2.42E+06
Don (ex-fSU)	1401	29.6661	1.63E+08	2.27E+05
Anadir (ex-fSU)	1150	32.17743	1.44E+08	*
Kamtchatka (ex-fSU)	626	28.88501	7.12E+07	*
Yukon	2716	187.20142	1.65E+09	*
Yenisei-Angara (Yenisey, Enisei, Russia)	4803	597.30829	8.26E+09	*
Ural (Russia)	1411	9.509	5.93E+07	7.15E+04
Ob-Irtysh	3977	413.183	4.91E+09	*
Nelson-Saskatchewan	2045	78.713	5.71E+08	*
Lena (East central Russia)	4387	539.918	6.89E+09	*
Kolyma (Russia)	2091	115.24	8.22E+08	*
Dneper (West and southwest Russia)	1544	48.185	2.78E+08	2.42E+06
Amur (Hei-lung chiang, Northeast Asia	5061	330.454	5.12E+09	*
Amudar'ya (Oxus, Jayhun, Amy; Amyderya; Dar'yoi Amu; Jaihun, Central and west Asia)	1976	50.257	3.69E+08	*

Table 9.1.4. Characterization factors, water consumption and normalization factors for water consumption. Characterization factors were calculated for 214 river basins. The data for water consumption, representative for the year 1995, were available for 112 river basins from the WaterGap Model (Alcamo et al. 2003a, 2003b).

River basin	Characterization factor ($PDF \cdot m^3 \cdot yr \cdot m^{-3}$)	Water consumption 1995 ($m^3 \cdot yr^{-1}$)	Normalization factor ($PDF \cdot m^3$)
Nil (Af., int.)	8.42E-03	5.41E+09	4.56E+07
Senegal (Guinée-Sénégal)	2.96E-03	4.34E+08	1.28E+06
Gambia (Guinée-Gambie)	1.36E-03	1.46E+08	1.99E+05
Tominé ou Rio Corubal (Guinée-Guineé Bissau)	7.67E-04	7.75E+07	5.94E+04
Konkouré (Guinée)	5.18E-04	3.34E+07	1.73E+04
Kolenté (Guinée, Great Scarcies)	3.79E-04		
Jong (Sierra Leone)	4.12E-04		
Sewa (Sierra Leone)	3.98E-04	1.01E+07	4.04E+03
Moa (Guinée-Sierra Leone)	6.76E-04	2.80E+07	1.89E+04
Mano (Libéria)	4.82E-04	5.43E+06	2.61E+03
Loffa (Guinée-Libéria)	5.85E-04	3.47E+06	2.03E+03
St Paul (Libéria)	6.30E-04	3.52E+07	2.22E+04
Nipoué (Cess, Libéria-RCI)	5.53E-04		
Cavally (Libéria-RCI)	6.04E-04	1.24E+07	7.48E+03
Dodo (aka Déo) (RCI)	1.46E-04		
San Pédro (RCI)	3.91E-04	2.39E+06	9.33E+02
Sassandra (RCI)	8.91E-04	6.58E+07	5.86E+04
N'Zo (a. Sassandra) (RCI)	4.81E-04		
Boubo (RCI)	2.62E-04		
Bandama (RCI)	1.11E-03	8.88E+07	9.90E+04
Yani (s.a. Bandama) (RCI)	2.68E-04		
Marahoué (a. Bandama) (RCI)	4.27E-04		
N'Zi (a. Bandama) (RCI)	8.63E-04		
Kan (s.a. Bandama) (RCI)	1.01E-03		
Agnébi (RCI)	5.64E-04		
Comoé (RCI-Burkina)	1.39E-03	6.84E+07	9.53E+04
Bia (RCI-Ghana)	4.99E-04	1.76E+07	8.76E+03
Volta (Ghana-Burkina)	2.02E-03	4.34E+08	8.79E+05
Black Volta (Burkina-Ghana) (a. Volta)	2.43E-03		
Nasia (a. White Volta) (Ghana)	3.91E-04		
Daka (a. Volta) (Ghana)	1.72E-04		
Mono (Togo)	8.08E-04	2.75E+07	2.22E+04
Ouémé (Bénin)	8.86E-04	6.19E+07	5.48E+04
Ogun (Nigéria)	7.62E-04	1.17E+08	8.89E+04
Niger (Afr. Int.)	5.59E-03	7.84E+08	4.38E+06
Niandan (Guinée) (a. Niger)	5.66E-04		
Bénoué (Nigéria-Cameroun) (a. Niger)	2.02E-03		
Sokoto (a. Niger) (Nigeria)	4.33E-04		
Cross (Nigéria-Cameroun)	7.01E-04	1.73E+08	1.21E+05
Mungo (Cameroun)	2.33E-05	1.13E+07	2.63E+02
Dibamba (Cameroun)	2.47E-04		
Wouri (Cameroun)	2.64E-04	2.47E+07	6.52E+03
Sanaga (Cameroun)	1.17E-03	1.33E+08	1.55E+05
Nyong (Cameroun)	6.51E-04	2.66E+07	1.73E+04
Lokoundjé (Cameroun)	3.67E-04	4.34E+05	1.59E+02

River basin	Characterization factor (PDF·m^3·yr·m^{-3})	Water consumption 1995 (m^3·yr^{-1})	Normalization factor (PDF·m^3)
Kribi ou Kienké (Cameroun)	1.98E-04		
Lobé (Cameroun)	1.59E-04		
Ntem (Cameroun-Gabon-Guinée équat.)	5.85E-04	9.48E+06	5.54E+03
Ogôoué (Gabon)	1.08E-03	4.33E+07	4.68E+04
Niari-Kouilou (Congo)	7.59E-04	4.20E+06	3.19E+03
Zaïre (Afr., Int.)	4.59E-03		
Cunene ou Kunene (Namibie-Angola)	1.48E-03	7.81E+07	1.15E+05
Kasaï (a. Zaïre) (Zaïre-Angola)	2.49E-03	5.55E+07	1.38E+05
Chari (Lac Tchad)	2.77E-03	5.09E+08	1.41E+06
Ubangi (a. Zaïre) (Congo-RCA)	3.00E-03		
Zambezi (Mozambique-Zambie-Angola)	3.66E-03	8.00E+08	2.93E+06
Tana (Kénya)	1.22E-03	2.46E+08	3.00E+05
Rufiji (Tanzanie)	1.27E-03	6.84E+07	8.67E+04
Limpopo (Botswana-Mozamb.-Rhodésie-RSA)	3.20E-03	2.82E+09	9.02E+06
Pongolo ou Maputo (RCA-Mozambique)	6.59E-04	1.77E+08	1.17E+05
Shire (a.) (Malawi-Mozambique)	1.63E-03		
Kafue (a. Zambèze) (Zambie)	1.59E-03		
Ruaha (a. Rufiji) (Tanzanie)	7.44E-04		
Evros-Mariça (Grèce-Turquie-Bulgarie)	7.22E-04	2.83E+09	2.04E+06
Nesta-Nestos (Grèce-Bulgarie)	4.80E-04	2.03E+08	9.72E+04
Strymon-Strouma (Grèce-Bulgarie)	7.65E-04	8.20E+08	6.27E+05
Agly (France)	1.77E-04		
Minho (Portugal-Espagne)	6.08E-04	2.86E+08	1.74E+05
Lima (Portugal)	2.13E-04		
Cavado (Portugal)	2.67E-04	8.89E+07	2.37E+04
Douro (Portugal-Esp.)	8.95E-04	3.47E+09	3.11E+06
Vouga (Portugal)	3.13E-04	6.10E+07	1.91E+04
Mondego (Portugal)	4.70E-04	1.45E+08	6.83E+04
Sado (Portugal)	3.81E-04	1.23E+08	4.70E+04
Mira (Portugal)	3.63E-04	9.74E+06	3.53E+03
Guadiana (Portugal-Esp.)	1.38E-03	1.97E+09	2.72E+06
Raisin (Canada)	2.83E-04		
Sydenham (Canada)	2.17E-04		
Grand river (Canada)	3.62E-04		
Thames (Canada)	5.11E-04		
Mississipi (USA)	4.88E-03	3.92E+09	1.91E+07
Rio Grande (USA-Mexique)	3.99E-03	5.28E+09	2.11E+07
Pecos (a. Rio Grande)	2.74E-03		
Canadian (s. a. Mississipi) (USA)	2.33E-03		
Colorado (USA-Mexique)	3.79E-03	4.08E+09	1.54E+07
San Juan (a. Colorado) (USA)	6.29E-04		
Zuni (s. a. Colorado) (a. Little Colorado)	2.44E-04		
San Francisco (a. Gila) (USA)	6.85E-03	1.68E+09	1.15E+07
Gila (a. Colorado)	2.42E-03		
Ohio river (a. Mississipi)	2.66E-03	9.77E+09	2.60E+07
Scioto River (a. Ohio)	5.25E-04		
Big Darby Creek (s. a. Ohio) (a. Scioto)	2.62E-04		
Wabash River (a. Ohio)	1.02E-03		
Little Wabash River (a. Wabash)	4.26E-04		
Embarras River (a. Wabash)	5.11E-04		

River basin	Characterization factor (PDF·m^3·yr·m^{-3})	Water consumption 1995 (m^3·yr^{-1})	Normalization factor (PDF·m^3)
St Joseph River (s.a. Wabash)	3.20E-04		
Elk river (s. a. Ohio) (a. Kanawha)	4.89E-04		
Cumberland river (a. Ohio)	1.54E-03		
Green river (a. Ohio)	1.60E-03		
Kanawha river (a. Ohio)	2.26E-04		
Tennessee River (a. Ohio)	1.33E-03		
Muskingum River (s.a. Ohio) (a. Allegheny)	2.75E-04		
Allegheny river (a. Ohio)	9.14E-04		
Little Miami river (a. Ohio)	2.37E-04		
Hocking river (a. Ohio)	2.29E-04		
Kinniconick river (a. Ohio)	2.24E-01		
Licking River (a. Ohio)	9.06E-05		
Little Scioto river (a. Ohio)	1.31E-04		
Ohio Brush Creek (a. Ohio)	1.44E-04		
Olentangy River (a. Little Scioto)	2.05E-04		
Paint Creek (a. Scioto river)	2.78E-04		
Scioto Brush Creek (a. Scioto)	8.18E-02		
Symmes River (a. Ohio)	1.40E-04		
Tygart Creek (a. Ohio)	5.17E-04		
Bear Creek	1.01E-04	1.43E+09	1.44E+05
Apalachicola (USA)	2.89E-04	1.48E+09	4.28E+05
Klamath (USA)	5.21E-04	3.18E+08	1.65E+05
Mobile (USA)	1.05E-04	1.18E+09	1.24E+05
Potomac (USA)	5.17E-04	1.56E+09	8.06E+05
Sabine (USA)	9.66E-04	3.24E+08	3.13E+05
Sacramento (USA)	1.42E-03	1.08E+10	1.54E+07
Savannah (USA)	7.94E-04	4.31E+08	3.42E+05
Susquehanna (USA)	7.99E-04	1.94E+09	1.55E+06
Connecticut river (USA)	8.24E-04	1.11E+09	9.11E+05
Missouri (USA)	4.88E-03	4.73E+09	2.31E+07
Arkansas river (USA)	2.75E-03		
Red river (USA)	2.55E-03		
Altamaha (USA)	7.66E-04	4.33E+08	3.32E+05
Balsas (Mexico)	1.13E-03	1.37E+09	1.55E+06
Panuco (Mexico)	8.15E-04	1.04E+09	8.51E+05
Sucio (a. Lempa) (San Salvador)	4.26E-05	4.51E+07	1.92E+03
Paz (San Salvador)	2.56E-04	6.20E+06	1.59E+03
San Tiguel (ou Miguel) San Salvador)	3.15E-04	9.00E+06	2.83E+03
Paraguay (Brésil-Arg.-Paraguay) (a. Parana)	2.97E-03	2.43E+09	7.22E+06
Uruguay (Brésil-Arg.-Uruguay)	1.85E-03	1.49E+09	2.77E+06
Magdalena (Colombie)	1.62E-03	1.84E+09	2.98E+06
Rio Negro (a. Amazone) (Colomb.-Venez.-Brésil)	1.05E-03	3.38E+06	3.54E+03
Parnaiba (Brésil)	1.89E-03		
Madeira (a. Amazone) (Brésil-Bolivie)	2.99E-03	3.84E+07	1.15E+05
Orinoco (Vénézuela-Colombie)	2.13E-03	1.12E+09	2.40E+06
Parana (Brésil-Paraguay-Argentine)	3.16E-03	2.05E+09	6.50E+06
Tibagi (Bresil)	9.53E-04	1.30E+08	1.23E+05
Amazon (Br. Mère Maranon) (Pérou-Brésil)	3.90E-03	5.43E+06	2.12E+04
Maroni (Guyane-Surinam)	6.53E-04	6.49E+06	4.24E+03
Oyapock (Guyane-Brésil)	4.43E-04	1.16E+06	5.14E+02

River basin	Characterization factor (PDF·m^3·yr·m^{-3})	Water consumption 1995 (m^3·yr^{-1})	Normalization factor (PDF·m^3)
Approuague	4.71E-04	1.00E+06	4.72E+02
Sinnamary (Guyane)	4.31E-04	6.86E+05	2.95E+02
Kourou (Guyane)	2.05E-04	4.22E+05	8.63E+01
Vakhsh ou Vachs (fSU) (a. Amu Darya)	2.93E-03		
Surkhandarya ou Surchandarya (fSU)	2.58E-04		
Zeravshan (a. Syr Darya) (fSU)	2.36E-03		
Naryn (a. Syr Darya) (fSU)	1.35E-03		
Tarim (Chine)	2.52E-03	1.33E+10	3.36E+07
Murgab ou Murghab ou Mourbab (fSU-Afghanistan) Endo	1.71E-03	1.06E+10	1.81E+07
Kabul (a. Indus) (Afghanistan-Inde)	9.86E-04		
Salween (Tibet-Chine-Birmanie-Thaï)	3.57E-03	1.63E+09	5.83E+06
Mae Khlong (Thaïlande)	2.36E-04	3.03E+08	7.16E+04
Chao Phrya (Menam) (Thaïlande)	1.12E-03	4.51E+09	5.07E+06
Mekong (Asie Sud-Est, Int.)	4.75E-03	8.70E+09	4.13E+07
Kelani Ganga(Sri Lanka)	2.81E-04		
Kalu Ganga (Sri Lanka)	2.50E-04		
Gin Ganga (Sri Lanka)	2.37E-04		
Nilwala Ganga (Sri Lanka)	1.38E-04		
Mahaweli Ganga (Sri Lanka)	6.58E-04	1.26E+09	8.31E+05
Brahmapoutre ou Tsangpo (Inde-Bengladesh-Tibet)	3.11E-03		
Indus (Tibet-Inde-Pakistan)	3.23E-03	4.95E+10	1.60E+08
Gange (Inde)	2.67E-03		
Ob (fSU)	4.76E-03	2.01E+09	9.57E+06
Yangzi Jiang (Tibet-Chine)	7.00E-03	3.38E+10	2.37E+08
Gandaki river (a. Gange) (nepal)	6.76E-04		
Sakaria (Turkey)	9.13E-04	1.90E+09	1.74E+06
Rakaïa river (New-Zealand)	2.85E-04	5.86E+07	1.67E+04
Fly (Nlle-Guinée)	9.10E-04	5.88E+06	5.35E+03
Sepik-Ramu (Nlle-Guinée)	3.95E-04	7.36E+06	2.90E+03
Kapuas (Bornéo)	7.44E-04	3.97E+07	2.96E+04
Murray-Darling (Australie)	3.07E-03	5.21E+09	1.60E+07
Yellow (Huang He, Huang Ho, China)	6.13E-03	3.25E+10	1.99E+08
Yangtze (Chang Jiang, Yangtze Kiang, China)	5.19E-03		
Xi Jiang River (Pearl River, Chu Chiang, Zhu, Southeast China)	2.12E-03		
Tsengwen River (Southwestern Taiwan)	2.82E-04		
Tigris (Southeast Turkey and Iraq)	3.02E-03	2.48E+10	7.47E+07
Tanshui (Northern Taiwan)	6.66E-04		
Tano (West Africa)	7.63E-04		
Saloum (West Africa)	2.50E-04		
Saint John (West Africa)	9.85E-04	1.24E+07	1.22E+04
Rokel River (Seli River, West Africa)	6.60E-04		
Purus (Northwest central South America)	3.30E-03		
Pra River (West Africa)	4.46E-04		
Pilcomayo (South central South America)	3.52E-03		
Pará-Tocantins (Brazil)	2.70E-03		
Orange (South Africa)	3.29E-03	2.24E+09	7.38E+06
Ombrone (Tuscany, Western Italy)	2.77E-04		
Okavango (Southwest central Africa)	2.57E-03	8.30E+07	2.14E+05

River basin	Characterization factor (PDF·m^3·yr·m^{-3})	Water consumption 1995 (m^3·yr^{-1})	Normalization factor (PDF·m^3)
Marañon (Peru)	2.65E-03		
Little Scarcies (West Africa)	4.72E-04		
Kwando River (Southwest Africa/Namibia)	1.13E-03		
Kura (Russia and Turkey)	1.29E-03		
Krishna (Karnataka, India)	1.50E-03	3.50E+10	5.25E+07
Kogon (Guinea, West Africa)	4.53E-04		
Kaoping River (Southern Taiwan)	3.28E-04		
Irrawaddy River (Irawadi, Central Myanmar Burma)	2.06E-03	9.58E+08	1.98E+06
Godavari (Central India)	1.31E-03		
Géba (Guinea Bissau, West Africa)	1.06E-03		
Ganges (Ganga, North and northeast Indian subcontinent)	2.41E-03	1.29E+11	3.11E+08
Fatala (West Africa)	3.51E-04		
Euphrates (Firat Nehri, Al-Furat, Southwest Asia)	3.75E-03		
Erhjen River (Southern River)	6.90E-05		
Chobe River (Southwest Africa/Namibia)	2.32E-03		
Chittar (Tamil Nadu, India)	3.31E-04		
Cauvery (Karnataka, India)	1.13E-03		
Casamance (West Africa)	6.27E-04		
Brahmaputra (Dyardanes, Oedanes, Tsangpo, Zangbo, Tibet, China, NE India and Bangladesh)	3.20E-03	8.47E+09	2.71E+07
Araguaia (Araguaya, Central Brazil)	3.42E-03	1.83E+08	6.26E+05
Athi-Galana-Sabaki River Drainage System (Kenya, from Nairobi eastward to Mombasa)	1.86E-03		

Table 9.1.5. Characterization factors, emissions in year 2000 and normalization factors for 63 greenhouse gas emissions, based on 100-year time horizon. The emissions in year 2000 were taken from Sleeswijk et al. (2008). Due to the data availability, we provide the normalization factors for 21 greenhouse gas emissions.

Substance	Characterization factor ($PDF \cdot m^3 \cdot yr \cdot kg^{-1}$)	Emission in year 2000 (kg)	Normalization factor ($PDF \cdot m^3$)
CO_2	8.53E-05	2.85E+13	2.43E+09
CH_4	1.69E-03	2.99E+11	5.04E+08
N_2O	2.78E-02	1.15E+10	3.19E+08
CFC-11	4.43E-01	4.06E+07	1.80E+07
CFC-12	1.02E+00	1.01E+08	1.02E+08
CFC-13	1.35E+00		
CFC-113	5.72E-01	3.86E+06	2.21E+06
CFC-114	9.37E-01	2.07E+06	1.94E+06
CFC-115	6.87E-01	8.73E+05	6.00E+05
Carbon tetrachloride	1.31E-01	4.17E+05	5.44E+04
Methyl bromide	4.48E-04		
Methyl chloroform	1.37E-02	3.57E+05	4.87E+03
HCFC-22	1.69E-01	3.00E+08	5.06E+07
HCFC-123	7.23E-03		
HCFC-124	5.68E-02	3.93E+06	2.23E+05
HCFC-141b	6.76E-02	1.66E+08	1.12E+07
HCFC-142b	2.16E-01	5.09E+07	1.10E+07
HCFC-225ca	1.14E-02		
HCFC-225cb	5.55E-02		
Halon-1211	1.76E-01	4.82E+06	8.48E+05
Halon-1301	6.66E-01	9.26E+05	6.17E+05
Halon-2402	1.53E-01	2.96E+05	4.54E+04
HFC-23	1.38E+00		
HFC-32	6.29E-02		
HFC-43-10mee	1.53E-01		
HFC-125	3.27E-01	7.40E+06	2.42E+06
HFC-134a	1.33E-01	1.30E+08	1.73E+07
HFC-143a	4.17E-01	5.40E+06	2.25E+06
HFC-227ea	3.01E-01		
HFC-245fa	9.65E-02		
HFC-152a	1.16E-02		
HFC-236fa	9.15E-01		
HFC-365mfc	7.41E-02		
Sulphur hexafluoride	2.13E+00	5.22E+06	1.11E+07
Nitrogen trifluoride	1.68E+00		
PFC-14	6.90E-01		
PFC-116	1.14E+00		
PFC-218	8.24E-01		

Substance	Characterization factor (PDF·m^3·yr·kg^{-1})	Emission in year 2000 (kg)	Normalization factor (PDF·m^3)
PFC-318	9.57E-01		
PFC-3-1-10	8.26E-01		
PFC-4-1-12	8.54E-01		
PFC-5-1-14	8.67E-01		
PFC-9-1-18	7.01E-01		
Trifluoromethyl sulphur pentafluoride	1.66E+00		
HFE-125	1.39E+00		
HFE-134	5.89E-01		
HFE-143a	7.05E-02		
HCFE-235da2	3.25E-02		
HFE-245cb2	7.51E-02		
HFE-245fa2	6.15E-02		
HFE-254cb2	3.68E-02		
HFE-347mcc3	5.37E-02		
HFE-347pcf2	5.39E-02		
HFE-356pcc3	1.02E-02		
HFE-449sl	2.86E-02		
HFE-569sf2	5.31E-03		
HFE-43-10pccc124	1.75E-01		
HFE-236ca12	2.64E-01		
HFE-338pcc13	1.40E-01		
PFPMIE	9.61E-01		
Dimethylether	3.96E-05		
Methylene chloride	8.15E-04		
Methyl chloride	1.20E-03		

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9.2. Annex to Chapter 2

Uncertainties

Water balance model (WTA)

The WaterGAP model is not explicitly addressing uncertainties in quantitative ways but provides uncertainty classes based on modeling performance on a global map (Alcamo et al. 2003). From the distribution and the assumption of data quality, we decided to quantify uncertainty based on Human Ddevelopment Indicators (HDI) values per watershed (Figure 9.2.1, derived from Pfister et al. 2009) for the water withdrawals between k values of 1.2 and 1.5 as depicted below. The rationale for usingHDI is to account for limited data quality due to restricted fund for monitoring in low-income countries and limited capacity in counries with low educational performance; both represented in the HDI. The same uncertainty function is also applied to agricultural water use estimates in the fate modelling.

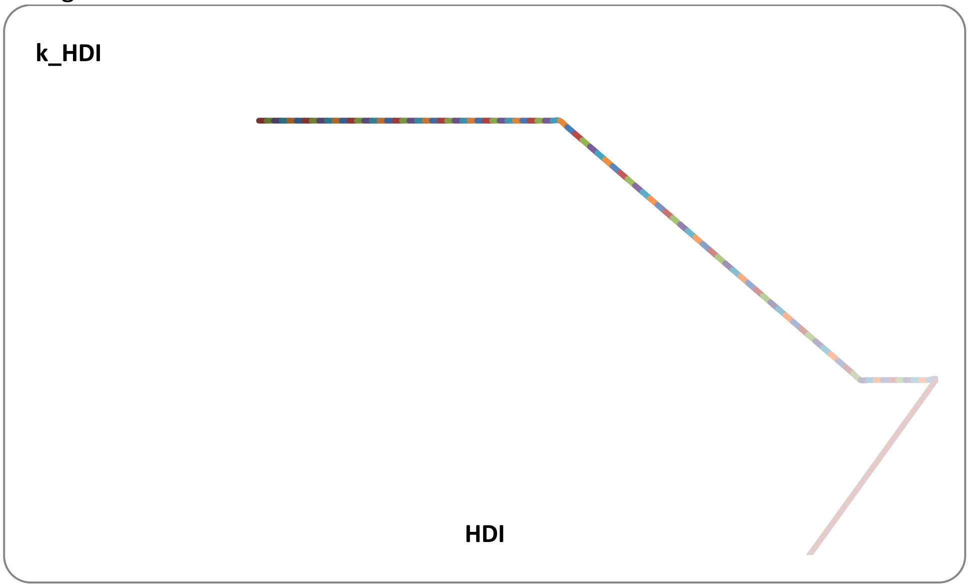

Figure 9.2.1. Estimated k-values for water availa bility

For water availability, we use data provided by Fekete et al. (2004) for their water balance model. In their Figure 4b and 7b they provide relative precipitation ranges of the main water balance models and sensitivity of these to Runoff modeling, respectively. We assumed the range of the four models covers 90% of all cases and derived coefficient of variation (CV), based on z-tables of normal distribution:

$$CV = Range_{90\%} / 1.645 \qquad (9.2.1)$$

Based on this CV, we calculated k values for precipitation (k_P):

$$k_P = \exp(\ln(CV^2 + 1)^{0.5})^{1.96} \qquad (9.2.2)$$

By applying the sensitivity of runoff to precipitation (S_P), we derived the k value for the water availability (k_A) using error propagation based on taylor series expansion according to MacLeod et al. (2002):

$$k_A = \exp(\ln(k_P)^2 \cdot S_P{}^2)^{0.5} \qquad (9.2.3)$$

Uncertainty of variation factor adjustment for WTA

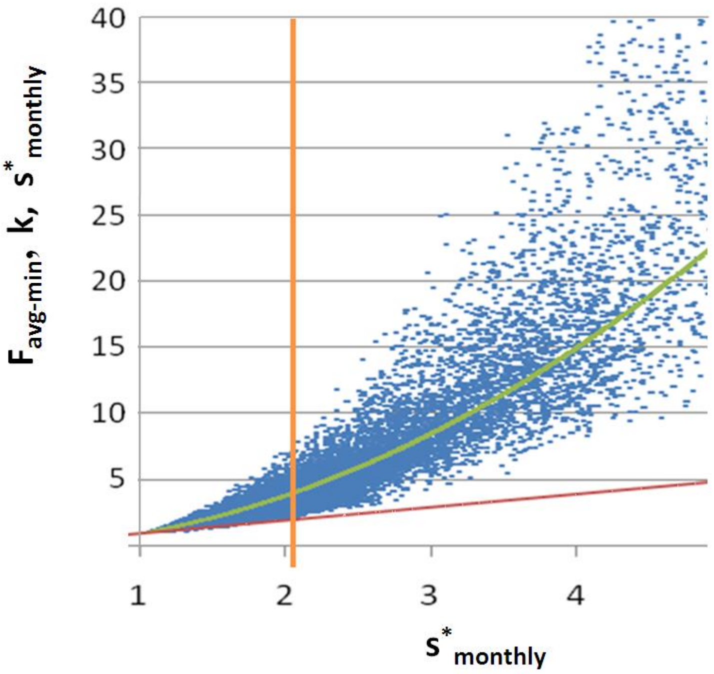

Figure 9.2.2. This graph shows the ratio of average to minimum monthly precipitation ($F_{avg\text{-}min}$; as surrogate for water availability) as blue points, versus the monthly geometric standard deviation ($s^*_{monthly}$). $F_{avg\text{-}min}$ represents the ratio of the month with maximal water stress versus the waters stress based on annual average (assuming constant withdrawal). To account for the increased water stress due to variability over the year we applied $s^*_{monthly}$ (represented by the red line) as main part of the variation factor (VF). The uncertainty is therefore also VF as at the 2.5% percentile we estimate that the impact od availability is zero because the demand follows the supply Variability, while ${s^*_{monthly}}^2$ (k; green line) represents the 97.5% where the demand is even higher in the dry months and therefore the ratio of average monthly withdrawal and minimum monthly water availability gives an good estimate for it.

Uncertainty of the human development factor (HDF)

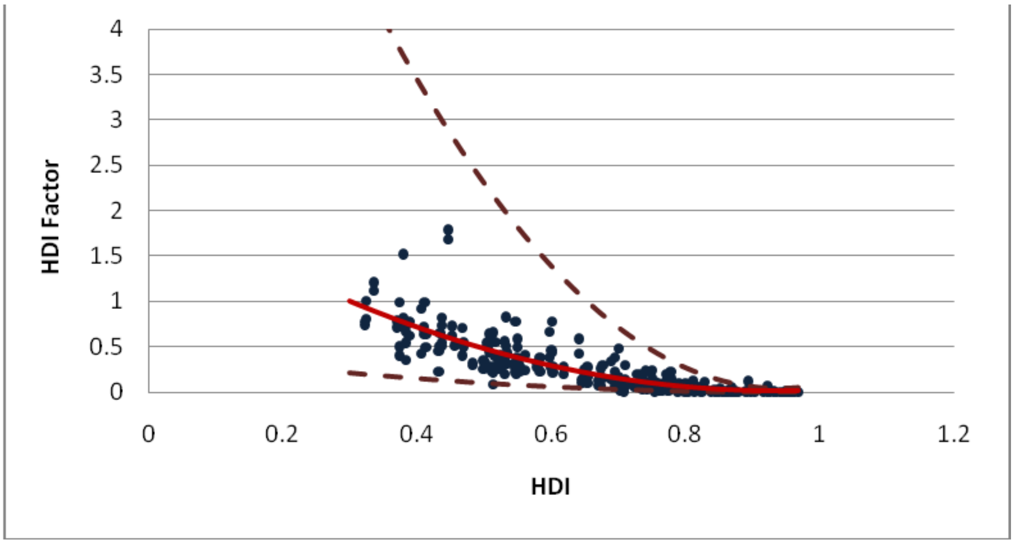

Figure 9.2.3. HDI and HDI factor for each country. The blue dots represent each country's relative malnutrition impact compared to the prediction with the factor (regression). The solid red line is the HDI factor and the dotted lines indicate the borders applying the k of 4.83 including 95% of all countires.

Uncertainty of Water requirements (effect).

While Pfister et al. (2009) used the water requirement per person, we analysed upper and lower linear regression based on comparing agricultural per-capita water use (FAO 2008) and corresponding malnutrition percentage (WHO 2007) in every country (Figure 9.2.4). The low estimate was including all vulnerable countries, while the high estimate only includes a selection of these which already have low agricultural water use (below 1000 m^3/cap^*year). We assume that these two regressions represent the 95% interval, resulting in a k-factor of 3.

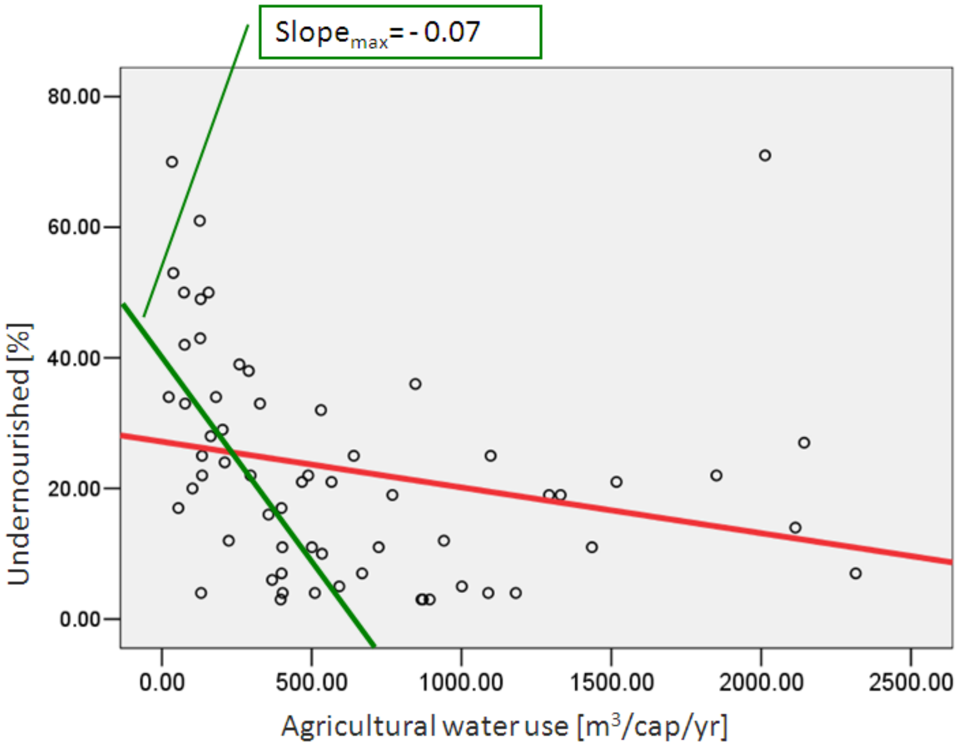

Figure 9.2.4. Linear regression between agricultural water use and malnourished population share for each vulnerable country (red line). The green line considers only countries with already low agricultural water use (below 1000 m^3/cap^*year).

Basis for DALY per malnutrition case (damage)

Daly per malnourished in the different world regions as reported by WHO (2007) are shown in Figure 9.2.5. The average is roughly 0.02 DALYs per malnutrition case per year with a variation between 0.01 and 0.04, resulting in a k-value of 2.0 as we assumed these damage factors represent the 95% interval

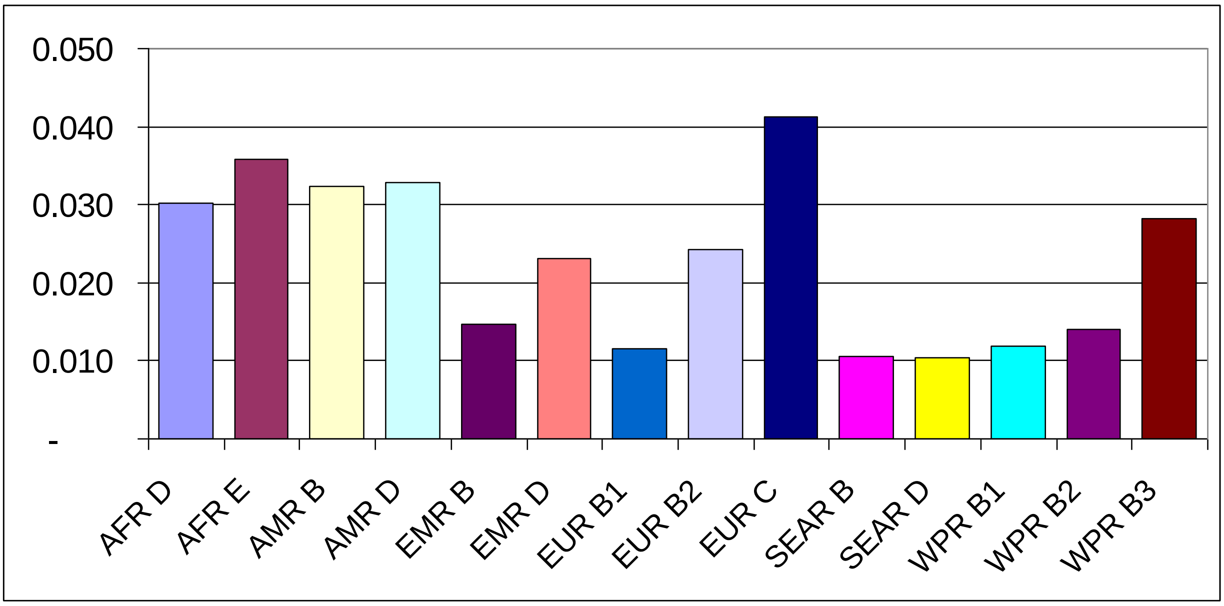

Figure 9.2.5. This graph shows the DALYs per malnutrition case per year for the WHO regions African Region (AFR), South East Asian Region (SEAR), Region of the Americas (AMR), European Region (EUR) Eastern Mediterranean Region (EMR), Western Pacific Region (WPR); distinct by health condition classes (from A = high level to D = low level). Based on WHO (2007).

Additional results

Uncertainty of water avilability

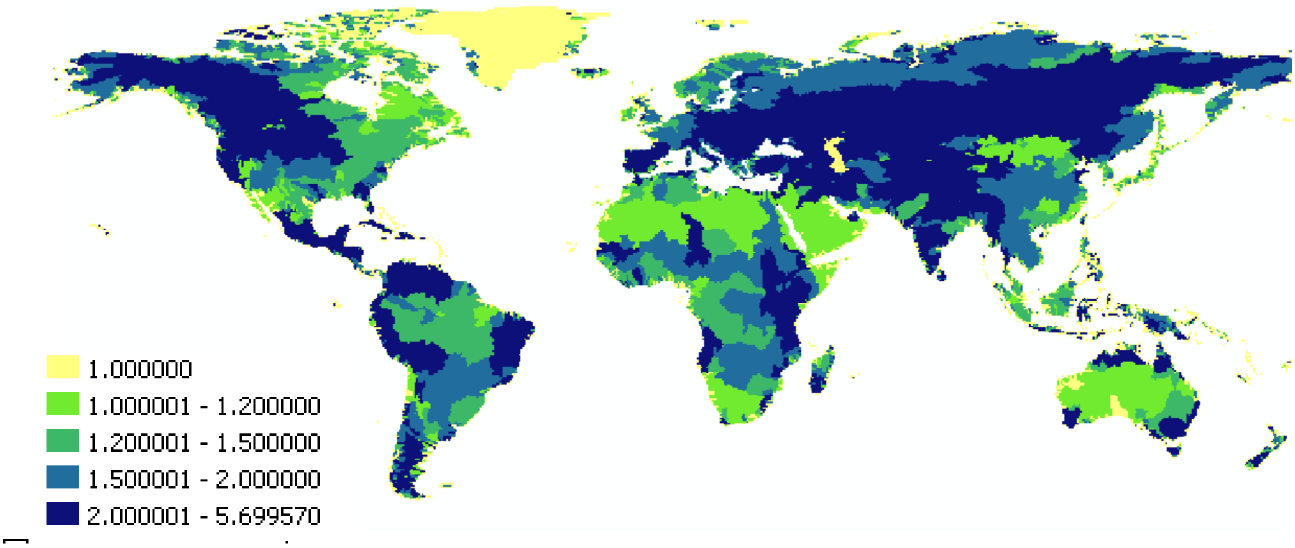

Figure 9.2.6. Resulting k-value from analyzing the different water availability models.

Uncertainty due to aggregation on country level

Table 9.2.1. The average k factor due to country aggregation of watershed characterization factors is 19.2. If it is weighted by Dalys, the average k factor results to be 16.7.

Country NAME	Average DALY/m3 (unweighted)	CV	k-value	Ratio max/mean	ISO country code
Nepal	**0.00000289789**	1.64676	9.43688	6.807988	NPL
Yemen	**0.00000257760**	2.387085	14.92411	16.21485	YEM
India	**0.00000223811**	1.536908	8.656248	6.625099	IND
Pakistan	**0.00000218025**	1.830166	10.76634	5.881069	PAK
Bangladesh	**0.00000213502**	2.256494	13.93884	10.28557	BGD
Morocco	**0.00000197269**	1.661863	9.54521	9.398486	MAR
Egypt	**0.00000176751**	2.02439	12.20078	8.946993	EGY
Afghanistan	**0.00000160508**	2.814526	18.16546	16.07957	AFG
Lesotho	**0.00000158780**	0.543599	2.71091	1.648419	LSO
Mali	**0.00000154801**	7.140199	49.19882	68.0493	MLI
Tajikistan	**0.00000129184**	2.203268	13.53855	9.797111	TJK
South Africa	**0.00000118553**	2.925721	19.00957	50.59298	ZAF
Nigeria	**0.00000117783**	3.597582	24.08465	49.87307	NGA
Botswana	**0.00000113567**	2.723723	17.476	18.43449	BWA
Uzbekistan	**0.00000109009**	1.55877	8.810515	6.848554	UZB
Iraq	**0.00000105164**	1.564014	8.847602	7.311209	IRQ
Syria	**0.00000096802**	1.533476	8.632079	5.894271	SYR
Sri Lanka	**0.00000093968**	1.848573	10.90126	8.056703	LKA
Tunisia	**0.00000090761**	1.312553	7.110873	4.748048	TUN
Turkmenistan	**0.00000089400**	2.749322	17.67037	19.97651	TKM
Somalia	**0.00000087578**	4.082028	27.69634	32.00937	SOM
Kyrgyzstan	**0.00000081533**	2.08752	12.67141	12.14671	KGZ
Ethiopia	**0.00000075923**	5.09646	35.08411	72.88485	ETH
Algeria	**0.00000074711**	2.366495	14.76851	13.7921	DZA
Iran	**0.00000071881**	2.740829	17.60589	20.35189	IRN
Armenia	**0.00000065960**	1.248681	6.685787	4.776584	ARM
Peru	**0.00000065329**	2.922666	18.98639	34.15559	PER
Niger	**0.00000062344**	3.017618	19.70674	21.78807	NER
Korea, Democratic People's Republic of	**0.00000062116**	3.078277	20.16657	15.37841	PRK
Azerbaijan	**0.00000062077**	0.952618	4.831077	3.777911	AZE
Saudi Arabia	**0.00000055684**	8.617695	58.80695	156.923	SAU
Lebanon	**0.00000055282**	0.589549	2.916818	1.648237	LBN
Turkey	**0.00000053916**	2.078597	12.60477	14.38703	TUR
Eritrea	**0.00000051181**	4.221881	28.72953	23.62443	ERI
Sudan	**0.00000049065**	4.783928	32.8349	77.558	SDN
Mozambique	**0.00000048466**	6.809277	46.98114	91.80675	MOZ
Libya	**0.00000047103**	3.623052	24.27572	30.41753	LBY
Bolivia	**0.00000046813**	6.765755	46.6876	87.82314	BOL
Georgia	**0.00000045711**	2.362288	14.73672	11.8649	GEO
Jordan	**0.00000043935**	2.522402	15.94848	10.96709	JOR
Vietnam	**0.00000038942**	2.840004	18.3589	14.15677	VNM
Oman	**0.00000038717**	2.877101	18.64053	16.39826	OMN
Zimbabwe	**0.00000035871**	2.138662	13.05392	16.12687	ZWE
China	**0.00000031290**	2.776631	17.87773	29.34259	CHN
Kazakhstan	**0.00000024587**	5.944165	41.06192	135.0627	KAZ
Djibouti	**0.00000023042**	1.621769	9.258131	4.851377	DJI
Macedonia	**0.00000021921**	0.91203	4.594673	3.07631	MKD

United Arab Emirates	**0.00000021468**	3.431708	22.83755	15.01453	ARE
West Bank	**0.00000021095**	1	5.113106	2	
Ecuador	**0.00000019779**	3.322626	22.01498	21.48099	ECU
Indonesia	**0.00000019746**	5.154815	35.50136	48.41488	IDN
Haiti	**0.00000019094**	2.072421	12.55868	7.61501	HTI
Mexico	**0.00000016843**	3.087029	20.23289	34.71753	MEX
Thailand	**0.00000015934**	2.874952	18.62422	22.11279	THA
Philippines	**0.00000015442**	3.088695	20.24552	17.63841	PHL
Swaziland	**0.00000015426**	0.146683	1.331052	1.126157	SWZ
Chile	**0.00000015389**	2.949081	19.18683	17.8642	CHL
Ukraine	**0.00000014472**	4.20989	28.64111	43.03602	UKR
Ghana	**0.00000012734**	7.205629	49.63434	62.84677	GHA
Dominican Republic	**0.00000012321**	2.718741	17.43819	10.78441	DOM
Qatar	**0.00000012098**	1.414218	7.801937	3.000008	QAT
Venezuela	**0.00000011527**	5.592784	38.60579	62.583	VEN
Israel	**0.00000011033**	1.356797	7.409562	4.019533	ISR
Madagascar	**0.00000010863**	4.060768	27.5389	48.11657	MDG
Cuba	**0.00000010139**	1.59587	9.073576	6.661052	CUB
Trinidad and Tobago	**0.00000009869**	0.649869	3.200634	1.649864	TTO
Sierra Leone	**0.00000009382**	0.91858	4.632492	3.628514	SLE
Chad	**0.00000008415**	1.680481	9.679063	12.71911	TCD
Angola	**0.00000008253**	1.901381	11.28962	11.42053	AGO
Bulgaria	**0.00000008128**	1.679417	9.671408	6.79454	BGR
Guinea	**0.00000007296**	0.94976	4.81427	5.392427	GIN
Guinea-Bissau	**0.00000007061**	1.132854	5.935529	3.918059	GNB
Mauritania	**0.00000007057**	3.133922	20.58811	17.54996	MRT
Tanzania, United Republic of	**0.00000006707**	2.55702	16.21093	26.04476	TZA
Cambodia	**0.00000006705**	4.687458	32.13572	35.10124	KHM
Senegal	**0.00000006620**	5.175727	35.65068	40.5279	SEN
Benin	**0.00000006144**	2.362665	14.73958	14.54892	BEN
Liberia	**0.00000006090**	0.402278	2.136334	2.72439	LBR
Kenya	**0.00000005996**	2.515468	15.89593	21.31634	KEN
Albania	**0.00000005671**	2.095842	12.73358	7.641725	ALB
Malawi	**0.00000005579**	1.523621	8.562773	6.591358	MWI
Russia	**0.00000005371**	10.88134	72.69802	495.1069	RUS
Burundi	**0.00000004877**	1.865812	11.02783	5.941027	BDI
Zambia	**0.00000004798**	3.812239	25.69097	47.42601	ZMB
Burkina Faso	**0.00000004089**	2.921449	18.97715	26.18533	BFA
Bhutan	**0.00000004019**	1.781682	10.41215	6.817248	BTN
Argentina	**0.00000003616**	5.811816	40.14039	78.37131	ARG
Moldova	**0.00000003599**	2.53142	16.01684	11.12627	MDA
Nicaragua	**0.00000003500**	3.230061	21.31562	20.66781	NIC
Rwanda	**0.00000003391**	1.582486	8.978494	4.550669	RWA
Uganda	**0.00000003384**	2.660914	16.99919	15.3079	UGA
Gambia, The	**0.00000003248**	0.625125	3.082392	1.625125	GMB
Togo	**0.00000003021**	1.299261	7.021801	5.746448	TGO
Greece	**0.00000002798**	3.987863	26.9982	26.37295	GRC
Cote d'Ivoire	**0.00000002681**	2.605274	16.57695	19.51363	CIV
Namibia	**0.00000002678**	2.811649	18.14361	21.49815	NAM
Myanmar (Burma)	**0.00000002415**	4.645839	31.83337	49.62195	MMR
El Salvador	**0.00000002368**	1.034694	5.32354	2.950064	SLV
Honduras	**0.00000002347**	1.930234	11.50257	8.135153	HND
Brazil	**0.00000001953**	7.56184	51.98861	138.8279	BRA

Laos	**0.00000001798**	2.095546	12.73136	13.76327	LAO
Congo (Republic of the)	**0.00000001746**	6.588281	45.48601	50.35525	COG
Guatemala	**0.00000001647**	2.440395	15.32734	11.4511	GTM
Colombia	**0.00000001461**	7.936627	54.43569	117.7506	COL
Mongolia	**0.00000001427**	8.603945	58.71964	120.7912	MNG
Cameroon	**0.00000001334**	2.574903	16.34655	19.61539	CMR
Congo (Democratic Republic of the)	**0.00000001318**	8.121112	55.62923	171.5155	COD
Guyana	**0.00000000997**	3.258741	21.53242	17.54252	GUY
Jamaica	**0.00000000992**	0.504124	2.541367	1.711972	JAM
Costa Rica	**0.00000000920**	1.051791	5.428414	3.415288	CRI
Fiji	**0.00000000908**	0.735598	3.629109	1.748299	FJI
Suriname	**0.00000000709**	5.24651	36.15529	28.76154	SUR
Serbia and Montenegro	**0.00000000698**	2.422188	15.18957	16.058	SCG
Kuwait	**0.00000000643**	2.059675	12.4636	6.22917	KWT
Panama	**0.00000000638**	1.179173	6.232153	4.28394	PAN
Romania	**0.00000000625**	3.210179	21.16525	32.47345	ROU
Belarus	**0.00000000491**	3.057528	20.00931	29.63496	BLR
Belize	**0.00000000478**	0.943268	4.77618	2.519683	BLZ
Croatia	**0.00000000441**	2.601497	16.54829	13.68214	HRV
Hungary	**0.00000000439**	1.050253	5.418949	5.339867	HUN
Slovenia	**0.00000000439**	0.571959	2.836925	1.931355	SVN
Austria	**0.00000000431**	1.608817	9.165742	9.729215	AUT
Slovakia	**0.00000000421**	0.844097	4.210447	4.457937	SVK
Malaysia	**0.00000000421**	3.017148	19.70317	20.76131	MYS
Paraguay	**0.00000000371**	2.332362	14.51076	12.87763	PRY
Bosnia and Herzegovina	**0.00000000340**	1.662466	9.549545	5.924349	BIH
Central African Republic	**0.00000000249**	5.051759	34.7639	54.80748	CAF
Uruguay	**0.00000000231**	2.61691	16.66524	16.66817	URY
United States	**0.00000000213**	15.67105	99.46657	640.2087	USA
Poland	**0.00000000196**	1.582321	8.977322	12.06303	POL
Czech Republic	**0.00000000127**	1.933646	11.52778	9.984145	CZE
Gabon	**0.00000000057**	3.275824	21.66152	23.99654	GAB
Lithuania	**0.00000000056**	2.036705	12.29244	11.93966	LTU
Germany	**0.00000000050**	3.217038	21.21713	30.35119	DEU
Latvia	**0.00000000046**	2.925752	19.0098	17.16275	LVA
Estonia	**0.00000000041**	1.617221	9.225669	7.403866	EST
Western Sahara	**0.00000000021**	3.141194	20.64317	10.93316	ESH
Switzerland	**0.00000000009**	3.999998	27.08828	17.00001	CHE
Finland	**0.00000000004**	3.002115	19.58916	21.51537	FIN
Equatorial Guinea	**0.00000000003**	2.645753	16.88412	8.000015	GNQ
French Guiana	**0.00000000000**	2.97993	19.42089	11.4	GUF

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9.3. Annex to Chapter 3

NHI modeling

Figure 9.3.1 shows the NHI groundwater model extent and all 872 major groundwater wells in the Netherlands.

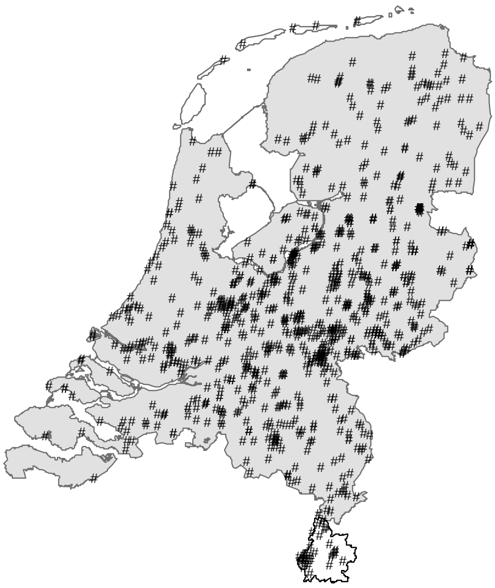

Figure 9.3.1. Overview of all 872 major groundwater wells in the Netherlands. The grey area indicates the NHI groundwater model extent.

Terrestrial plant species

Table 9.3.1 shows the terrestrial plant species included in the effect factor calculations and their κ-values, which express the probability of occurrence related to measured observations. It is indicated whether a species is on the red list in the Netherlands.

Table 9.3.1. All terrestrial plant species included in the effect factor calculations, with their κ-values.

	Plant species	Scientific name	κ - value	Red list?
	Sycamore (Sycamore Maple)	*Acer pseudoplatanus*	0.1	
	Sneezewort	*Achillea ptarmica*	0.15	
	Sweetflag (Flagroot)	*Acorus calamus*	0.15	
	Baneberry	*Actaea spicata*	0.1	x
	Moschatel	*Adoxa moschatellina*	0.25	
	Ground Elder	*Aegopodium podagraria*	0.15	
	Fool's Parsley	*Aethusa cynapium*	0.05	
	Agrimony (Churchsteeples, Cocklebur, European Grovebur)	*Agrimonia eupatoria*	0.35	x
	Velvet- + Brown Bent	*Agrostis canina +agrostis vinealis*	0.05	x
0	Silver Hair-grass	*Aira caryophyllea (subsp. multiculmis)*	0.15	
1	Bugleweed	*Ajuga reptans*	0.1	
	Garlic Mustard	*Alliaria petiolata/officinalis*	0.2	

2 :	Field Garlic	*Allium oleraceum*	0.05	x
3 :	Bear's Garlic (Ramsons, Wild Garlic)	*Allium ursinum*	0.05	
4 :	Grey Alder	*Alnus incana*	0.1	
5 :	Orange Foxtail	*Alopecurus aequalis*	0.1	x
6 :	Bulbous Foxtail	*Alopecurus bulbosus*	0.15	x
7 :	Blackgrass (Slender Meadow Foxtail, Mouse Foxtail, Slender Foxtail)	*Alopecurus myosuroides*	0.1	x
8 :	Meadow Foxtail	*Alopecurus pratensis*	0.3	
9 :	Common Marsh Mallow	*Althaea officinalis*	0.25	x
0 :	Hybrid Marram	*Calammophila x baltica (Ammocalamagrostis x baltica)*	0.1	
1 :	Scarlet Pimpernel	*Anagallis arvensis subsp. arvensis (Anagallis phoenicea)*	0.05	
2 :	Bog Pimpernel	*Anagallis tenella*	0.1	x
3 :	Alkanet (Common Bugloss)	*Anchusa officinalis*	0.05	
4 :	Bog Rosemary	*Andromeda polifolia*	0.25	x
5 :	Wood Anemone	*Anemone nemorosa*	0.05	
6 :	Angelica Root	*Angelica archangelica*	0.1	
7 :	Mountain Everlasting	*Antennaria dioica*	0.05	x
8 :	Corn chamomile	*Anthemis arvensis*	0.05	x
9 :	Annual Vernal-grass	*Anthoxanthum aristatum/puelli*	0.1	x
0 :	Bur-chervil	*Anthriscus caucalis/scandicina*	0.1	x
1 :	Cow Parsley (Wild Chervil, Bur Chervil, Keck)	*Anthriscus sylvestris*	0.15	x
2 :	Kidney-vetch	*Anthyllis vulneraria*	0.05	x
3 :	Field parsley-piert	*Aphanes arvensis*	0.1	x
4 :	Slender parsley piert	*Aphanes australis/inexpectata/microcarpa)*	0.05	x
5 :	Celery	*Apium graveolens*	0.1	x
6 :	Fool's Water-cress	*Apium/helosciadium nodiflorum*	0.25	
7 :	Thale Cress	*Arabidopsis thaliana*	0.05	
8 :	Greater Burdock	*Arctium lappa/majus*	0.05	
9 ,	Lesser Burdock	*Arctium minus/pubens*	0.05	
0 ,	Thrift (Thrift Seapink, Sea Pink, Common/Sea/Western Thrift)	*Armeria maritima (Statice armeria elongata)*	0.4	x
1 ,	Lamb Succory	*Arnoseris minima*	0.1	x
2				

3	Field Wormwood	*Artemisia campestris subsp. maritima*	0.05	
4	Sea Wormwood	*Artemisia/Seriphidium maritima*	0.3	x
5	Mugwort	*Artemisia vulgaris*	0.1	
6	Lords-and-ladies (Cuckoo Pint, Adders Tongue, Calves Foot, Sweethearts)	*Arum maculatum*	0.4	
7	Asparagus (Garden-Asparagus)	*Asparagus officinalis (Asparagus prostratus)*	0.1	
8	Asparagus (Asparagus Fern, Wild Asparagus)	*Asparagus officinalis (Asparagus prostratus)*	0.05	x
9	Sweetscented bedstraw	*Galium odoratum*	0.15	
0	Wall Rue	*Asplenium ruta-muraria*	0.15	
1	Grass-leaved Orache	*Atriplex littoralis*	0.2	x
2	Black Horehound	*Ballota alba/nigra subsp. Meridionalis/foetida*	0.05	
3	Common Winter-cress	*Barbarea vulgaris*	0.1	
4	Barberry	*Berberis vulgaris*	0.15	x
5	Hoary Alison	*Berteroa incana*	0.15	
6	Nodding Bur-marigold	*Bidens cernua*	0.1	
7	London bur-marigold	*Bidens connata*	0.2	
8	Beggar Ticks	*Bidens frondosa*	0.05	
9	Trifid Bur-marigold	*Bidens tripartita*	0.1	
0	Hard Fern (Deer Fern)	*Blechnum spicant*	0.05	x
1	Common Moonwort (Moonwort Grape-Fern, Moonwort Grapefern)	*Botrychium lunaria*	0.05	x
2	Tor-grass (Heath False Brome)	*Brachypodium pinnatum*	0.3	x
3	(Slender) False-brome	*Brachypodium sylvaticum*	0.05	
4	Black Mustard	*Brassica nigra*	0.05	x
5	Smooth Brome	*Bromopsis/Bromus inermis*	0.1	x
6	Barren Brome	*Anisantha/Bromus sterilis*	0.05	x
7	Drooping Brome	*Anisantha/Bromus tectorum*	0.05	
8	Slender Hare's-ear	*Bupleurum tenuissimum*	0.05	x
9	Sea Rocket	*Cakile maritima*	0.1	
0	Wood Small-reed (Chee Reed Grass, Feathertop)	*Calamagrostis epigeios*	0.2	
1	Narrow Small-reed	*Calamagrostis stricta/neglecta*	0.05	
2	Heather	*Calluna vulgaris*	0.4	x
3	Marsh Marigold	*Caltha palustris*	0.2	

4	Rampion bellflower	*Campanula rapunculus*	0.1	x
5	Harebell	*Campanula rotundifolia*	0.2	x
6	Nettle-leaved Bellflower	*Campanula trachelium*	0.15	
7	Large Bitter-Cress, Cardamine amère	*Cardamine amara*	0.05	
8	Wavy Bitter-cress (Bittercress, Woodland Bittercress)	*Cardamine sylvatica/flexuosa/hirsuta subsp. Sylvatica*	0.05	
9	Nodding Thistle	*Carduus nutans*	0.1	x
0	Slender Tufted-sedge	*Carex acuta/gracilis)*	0.2	
1	Lesser Pond-sedge	*Carex acutiformis*	0.15	
2	Fibrous Tussock-sedge	*Carex appropinquata*	0.1	x
3	Water Sedge	*Carex aquatilis*	0.15	x
4	Spring Sedge	*Carex caryophyllea*	0.25	x
5	White Sedge	*Carex curta/canescens*	0.15	x
6	Yellow Sedge	*Carex serotina/scandinavica/tumidicarpa/demissa/viridula/ oederi subsp. oedocarpa*	0.1	
7	Distant Sedge	*Carex distans (vikingensis)*	0.15	x
8	Brown Sedge	*Carex disticha*	0.25	
9	Elongated Sedge	*Carex elongata*	0.2	
0	Long-bracted Sedge	*Carex extensa*	0.2	x
1	Hairy Sedge	*Carex hirta*	0.15	
2	Tufted Sedge	*Carex elata/hudsonii*	0.15	x
3	Slender/Woolly-Fruit Sedge	*Carex lasiocarpa*	0.15	x
4	False Fox-sedge	*Carex otrubae/cuprina*	0.1	
5	Oval Sedge	*Carex ovalis/leporina*	0.1	
6	Pale Sedge	*Carex pallescens*	0.1	x
7	Greater Tussock Sedge	*Carex paniculata*	0.15	
8	Pill Sedge	*Carex pilulifera*	0.25	
9	Cyperus Sedge	*Carex pseudocyperus*	0.15	
00	Flea Sedge	*Carex pulicaris*	0.15	x
01	Remote Sedge	*Carex remota*	0.15	
02	Great Pond-sedge	*Carex riparia*	0.15	
03	Bottle Sedge	*Carex rostrata*	0.2	
	Spiked/Prickly Sedge	*Carex spicata*	0.05	

04:	Wood sedge (European Woodland Sedge)	*Carex sylvatica*	0.25	
05:	Three-nerved Sedge	*Carex trinervis*	0.15	x
06:	Bladder Sedge	*Carex vesicaria*	0.1	
07:	Carline Thistle	*Carlina vulgaris*	0.1	x
08:	Hornbeam	*Carpinus betulus*	0.15	
09:	Caraway	*Carum carvi*	0.05	x
10:	Sweet Chestnut	*Castanea sativa*	0.05	
11:	Whorl-grass	*Catabrosa aquatica*	0.05	x
12:	Cornflower	*Centaurea cyanus*	0.15	x
13:	Greater knapweed	*Centaurea scabiosa*	0.3	x
14:	Seaside Centaury	*Centaurium littorale*	0.1	x
15:	Common Centaury (European Centaury, Bitter Herb)	*Centaurium erythraea/minus*	0.1	
16:	Branching Centaury	*Centaurium pulchellum*	0.15	
17:	Chaffweed	*Centunculus minimus (Anagallis minima)*	0.1	x
18:	Field/Mouse-Ear/Starry Chickweed	*Cerastium arvense*	0.2	
19:	Sea Mouse-ear	*Cerastium diffusum/tetrandrum/atrovirens*	0.05	x
20:	Sticky Mouse-ear	*Cerastium glomeratum*	0.1	
21:	Rough Chervil	*Chaerophyllum temulum/temulentum*	0.15	
22:	Greater Celandine	*Chelidonium majus*	0.05	
23:	Oak-leaved Goosefoot	*Chenopodium glaucum*	0.15	
24:	Many-seeded Goosefoot	*Chenopodium polyspermum*	0.05	
25:	Red Goosefoot	*Chenopodium rubrum*	0.1	x
26:	Oxeye/Field Daisy/Moon/Shasta Daisy (Marguerite)	*Leucanthemum vulgare (Chrysanthemum leucanthemum)*	0.2	
27:	Corn Marigold/Chrysanthemum (Corndaisy)	*Glebionis/Chrysanthemum segetum*	0.05	
28:	Opposite-leaved Golden-saxifrage	*Chrysosplenium oppositifolium*	0.05	
29:	Yellow Centuary	*Cicendia filiformis*	0.1	x
30:	Chicory	*Cichorium intybus*	0.1	
31:	Enchanter's-nightshade	*Circaea lutetiana*	0.15	
32:	Dwarf/Stemless/Picnic Thistle	*Cirsium acaule*	0.2	x
33:	Creeping/Californian/Canadian/Field/Perennial Thistle	*Cirsium arvense*	0.25	
34:	Meadow Thistle	*Cirsium dissectum*	0.1	x

35	Marsh thistle	*Cirsium palustre (Carduus palustris)*	0.25	
36	Great Fen-Sedge ((Swamp) Sawgrass)	*Cladium mariscus*	0.05	x
37	Spring Beauty	*Claytonia/Montia perfoliata*	0.1	
38	Danish scurvygrass (Early Scurvygrass)	*Cochlearia danica*	0.1	
39	Common Scurvygrass	*Cochlearia officinalis subsp. Officinalis*	0.15	x
40	Marsh Cinquefoil	*Comarum palustre (Potentilla palustris/comarum)*	0.2	x
41	Lily-of-the-Valley	*Convallaria majalis*	0.15	
42	Swine-cress	*Coronopus squamatus*	0.1	
43	Bird in a bush	*Corydalis solida*	0.05	x
44	Hazel	*Corylus avellana*	0.15	
45	Grey Hair-grass	*Corynephorus canescens*	0.3	x
46	Midland Hawthorn	*Crataegus oxyacantha/laevigata (Mespilum oxyacantha)*	0.15	
47	Rough Hawk's-beard	*Crepis biennis*	0.15	x
48	Smooth Hawksbeard	*Crepis capillaris*	0.15	
49	Marsh Hawk's-beard	*Crepis paludosa*	0.1	x
50	Beaked Hawk's-beard	*Crepis polymorpha/vesicaria subsp. taraxacifolia/subsp. haensleri)*	0.15	x
51	Clover dodder	*Cuscuta epithymum*	0.05	x
52	Greater Dodder	*Cuscuta europaea*	0.05	
53	Bermuda grass	*Cynodon dactylon*	0.1	
54	Bog Hair-grass	*Deschampsia setacea*	0.1	x
55	Maiden Pink	*Dianthus deltoides*	0.2	x
56	Foxglove	*Digitalis purpurea*	0.05	
57	Smooth Finger-grass	*Digitaria ischaemum*	0.2	
58	Perennial Wall-rocket	*Diplotaxis tenuifolia*	0.1	
59	Wild Teasel	*Dipsacus fullonum/sylvestris*	0.15	
60	Broad Buckler-fern	*Dryopteris dilatata/austriaca*	0.2	
61	Male Fern	*Dryopteris filix-mas*	0.05	
62	Narrow Buckler-fern	*Dryopteris carthusiana/spinulosa*	0.25	
63	Marsh fern	*Thelypteris palustris (Dryopteris thelypteris)*	0.25	x
64	Lesser Water-plantain	*Baldellia ranunculoides subsp. ranunculoides (Echinodorus/Alisma ranunculoides)*	0.15	x

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66	Viper's bugloss	*Echium vulgare*	0.1	
67	Few-flowered Spike-rush	*Eleocharis quinqueflora/pauciflora (Scirpus quinqueflorus/pauciflorus)*	0.1	x
68	Slender Spike-rush	*Eleocharis uniglumis/palustris subsp. uniglumis)*	0.1	x
69	Lyme-grass	*Leymus arenarius (Elymus arenarius)*	0.1	
70	Sea Couch	*Elytrigia atherica (Elymus athericus/pycnanthus, Agropyron littorale/pungens)*	0.2	
71	American willowherb	*Epilobium ciliatum/adenocaulon*	0.05	
72	Rosebay Willow-herb	*Chamerion angustifolium (Epilobium angustifolium/spicatum)*	0.1	
73	Dark-green Willow-herb	*Epilobium obscurum*	0.05	
74	Marsh Willowherb	*Epilobium palustre*	0.1	x
75	Hoary Willowherb	*Epilobium parviflorum*	0.1	
76	Broad Leaved Helleborine	*Epipactis helleborine/latifolia*	0.05	
77	Marsh Helleborine	*Epipactis/Helleborine palustris*	0.25	x
78	Shore Horsetail	*Equisetum litorale(x)/arvense x fluviatile*	0.05	
79	Variegated Horsetail	*Equisetum variegatum*	0.2	x
80	Blue Fleabane	*Erigeron acer (Erigeron acris)*	0.1	x
81	Canadian Horseweed	*Conyza/Erigeron canadensis*	0.1	
82	Common Cottongrass	*Eriophorum angustifolium/polystachion*	0.25	x
83	Hare's-tail Cottongrass	*Eriophorum vaginatum*	0.25	x
84	Common Stork's-bill	*Erodium cicutarium subsp. cicutarium*	0.1	
85	Sticky Stork's-bill	*Erodium lebelii/glutinosum*	0.1	x
86	Common Stork's-bill	*Erodium cicutarium subsp. dunense*	0.2	
87	Common Whitlowgrass	*Erophila/Draba verna*	0.15	
88	Field eryngo	*Eryngium campestre*	0.15	x
89	Sea Holly	*Eryngium maritimum*	0.1	x
90	Treacle Mustard	*Erysimum cheiranthoides*	0.05	
91	Cypress Spurge	*Euphorbia cyparissias*	0.05	
92	Marsh spurge	*Euphorbia palustris*	0.05	x
93	Petty Spurge	*Euphorbia peplus*	0.05	
94	Red Bartsia	*Odontites vernus/litoralis/verna/rubra (Euphrasia odontites)*	0.05	x
95	Giant Fescue	*Festuca gigantea*	0.1	
	Hybrid Fescue	*Festulolium x loliaceum/x braunii*	0.15	

96	Small Cudweed	*Filago/Logfia minima*	0.1	x
97	Wild Strawberry	*Fragaria vesca*	0.1	x
98	Snake’s head fritillary	*Fritillaria meleagris*	0.05	x
99	Common Fumitory	*Fumaria officinalis*	0.05	
00	Snowdrop	*Galanthus nivalis*	0.1	x
01	Bifid Hemp-nettle	*Galeopsis bifida/tetrahit subsp. Bifida*	0.1	
02	Large-flowered Hemp-nettle	*Galeopsis speciosa*	0.05	x
03	Shaggy-soldier	*Galinsoga quadriradiata/ciliata*	0.05	
04	Crosswort	*Cruciata laevipes (Galium cruciata)*	0.05	x
05	Slender Bedstraw	*Galium pumilum*	0.35	x
06	Lady's Bedstraw	*Galium verum (subsp. maritimum)*	0.35	x
07	Petty Whin	*Genista anglica*	0.1	x
08	Hairy Greenweed	*Genista pilosa*	0.15	x
09	Dyer's Greenweed	*Genista tinctoria*	0.1	x
10	Autumn Gentian	*Gentianella/gentiana amarella*	0.2	x
11	Chiltern Gentian	*Gentianella/gentiana germanica*	0.15	x
12	Marsh Gentian	*Gentiana pneumonanthe*	0.15	x
13	Cut-leaved Crane's-bill	*Geranium dissectum*	0.1	
14	Small-flowered Cranesbill	*Geranium pusillum*	0.05	
15	Herb-Robert	*Geranium robertianum*	0.2	
16	Wood Avens	*Geum urbanum*	0.2	
17	Sea-milkwort	*Glaux maritima/generalis*	0.2	
18	Small Sweet-grass	*Glyceria declinata (Glyceria notata/plicata subsp. declinata)*	0.05	
19	Reed Sweet Grass	*Glyceria maxima/aquatica*	0.2	
20	Plicate Sweet-grass	*Glyceria notata/plicata*	0.05	
21	Weedy cudweed	*Gnaphalium/Pseudognaphalium/Helichrysum luteo-album*	0.05	x
22	Heath Cudweed	*Gnaphalium/Omalotheca sylvaticum*	0.05	x
23	Marsh Cudweed	*Gnaphalium uliginosum (Filaginella uliginosa)*	0.05	
24	Fragrant Orchid	*Gymnadenia conopsea*	0.2	x
25	Annual Sea Purslane	*Atriplex/Halimione/Obione pedunculata*	0.1	x
26	Bog Orchid	*Hammarbya/Malaxis paludosa*	0.05	x

27	Ivy	*Hedera helix*	0.25	
28	Downy Oat-grass	*Helictotrichon/Avenula pubescens*	0.2	
29	Smooth Rupturewort	*Herniaria glabra*	0.2	
30	Common Hawkweed	*Hieracium vulgatum/lachenalii/levicaule/maculatum/ argillaceum/diaphanoides)*	0.05	x
31	Smooth Hawkweed	*Hieracium laevigatum/tridentatum/rigidum)*	0.1	
32	New England Hawkweed	*Hieracium sabaudum*	0.05	
33	Leafy Hawkweed	*Hieracium umbellatum subsp. hollandiae)*	0.2	
34	Holy Grass	*Hierochloë odorata (Anthoxanthum nitens)*	0.1	x
35	Seabuckthorn	*Hippophae rhamnoides*	0.25	
36	Yorkshire-fog	*Holcus lanatus*	0.35	
37	Sea Sandwort	*Honkenya peploides*	0.05	x
38	Sea Barley	*Hordeum marinum*	0.05	x
39	Wall Barley	*Hordeum murinum*	0.05	
40	Meadow Barley	*Hordeum secalinum/pratense/nodosum*	0.15	x
41	Hop	*Humulus lupulus*	0.2	
42	Marsh Pennywort	*Hydrocotyle vulgaris*	0.25	
43	Marsh St. John's-wort	*Hypericum elodes*	0.15	x
44	Trailing St. John's-wort	*Hypericum humifusum*	0.1	
45	Spotted St. John's-wort	*Hypericum maculatum*	0.05	
46	St. John's wort	*Hypericum perforatum*	0.3	
47	Slender St John's-wort	*Hypericum pulchrum*	0.25	x
48	Square-stalked St. John's-wort	*Hypericum tetrapterum/quadrangulum*	0.05	
49	Common Holly	*Ilex aquifolium*	0.1	
50	Coral-necklace	*Illecebrum verticillatum*	0.1	x
51	Touch-me-not Balsam	*Impatiens noli-tangere*	0.15	
52	Small Balsam	*Impatiens parviflora*	0.1	
53	Meadow Fleabane	*Inula britannica*	0.15	x
54	Ploughman's Spikenard	*Inula conyzae*	0.15	
55	Yellow Flag	*Iris pseudacorus*	0.25	
56	Sharp-flowered Rush	*Juncus acutiflorus*	0.2	
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58	Jonc des grenouilles	*Juncus ambiguus (Juncus bufonius subsp. ambiguus/subsp. ranarius)*	0.1	
59	Northern Green Rush	*Juncus anceps/alpinoarticulatus subsp. atricapillus*	0.15	x
60	Arctic rush	*Juncus balticus/arcticus subsp. Balticus*	0.15	x
61	Toad Rush	*Juncus bufonius/minutulus)*	0.05	
62	Round-fruited Rush	*Juncus compressus*	0.1	
63	Compact Rush (Common Rush)	*Juncus conglomeratus/subuliflorus*	0.2	
64	Thread Rush	*Juncus filiformis*	0.1	x
65	Northern Green Rush	*Juncus alpinoarticulatus subsp. alpinoarticulatus*	0.15	x
66	Saltmarsh Rush	*Juncus gerardii*	0.2	
67	Hard Rush	*Juncus inflexus/glaucus*	0.1	
68	Sea Rush	*Juncus maritimus*	0.1	x
69	Heath Rush	*Juncus squarrosus*	0.15	
70		*Juncus tenageia*	0.05	x
71	Slender Rush	*Juncus tenuis*	0.05	
72	Common Juniper	*Juniperus communis*	0.05	x
73	Field Scabious	*Knautia arvensis (Scabiosa arvensis)*	0.2	x
74	Crested Hair-grass	*Koeleria macrantha/albescens/gracilis/cristata*	0.25	x
75	Prickly Lettuce	*Lactuca serriola/scariola*	0.05	
76	White Dead-nettle	*Lamium album*	0.1	
77	Hen-bit Dead-nettle	*Lamium amplexicaule*	0.2	
78	Yellow Archangel	*Lamiastrum/Lamium galeobdolon (Galeobdolon luteum)*	0.4	
79	Spotted Deadnettle	*Lamium maculatum*	0.15	
80	Red Dead-nettle	*Lamium purpureum*	0.1	
81	Nipplewort	*Lapsana communis*	0.1	
82	Marsh Pea	*Lathyrus palustris*	0.15	x
83	Meadow Vetchling	*Lathyrus pratensis (Pisum pratense)*	0.15	
84	Tuberous Pea	*Lathyrus tuberosus*	0.05	
85	Rough Hawkbit	*Leontodon hispidus*	0.3	x
86	Lesser Hawkbit	*Leontodon saxatilis/taraxacoides/nudicaulus (Thrincia hirta)*	0.1	
87	Summer Snowflake	*Leucojum aestivum*	0.1	x
88	Wild Privet	*Ligustrum vulgare*	0.15	

89	Mudwort	*Limosella aquatica*	0.05	x
90	Ivy-leaved Toadflax	*Cymbalaria muralis (Linaria/Antirrhinum cymbalaria)*	0.1	
91	Dwarf Snapdragon	*Chaenorhinum/Antirrhinum minus (Linaria minor)*	0.2	
92	Common Toadflax	*Linaria vulgaris*	0.1	
93	Fairy Flax	*Linum catharticum*	0.35	x
94	Fen Orchid	*Liparis loeselii*	0.1	x
95	Common Gromwell	*Lithospermum officinale*	0.1	x
96	Shoreweed	*Littorella uniflora*	0.4	x
97	Italian Rye Grass	*Lolium multiflorum*	0.05	
98	Common honeysuckle	*Lonicera periclymenum*	0.2	
99	Narrow-leaved Bird's-foot Trefoil	*Lotus glaber/tenuis/corniculatus subsp. tenuifolius*	0.15	
00	Hairy Wood-rush	*Luzula pilosa*	0.15	
01	Great Wood-rush	*Luzula sylvatica*	0.1	
02	Marsh Clubmoss	*Lycopodiella inundata (Lycopodium inundatum)*	0.1	x
03	Annual Bugloss	*Anchusa/Lycopsis arvensis*	0.1	x
04	Yellow Pimpernel	*Lysimachia nemorum*	0.1	
05	Creeping-Jenny	*Lysimachia nummularia*	0.15	
06	Purple Loosestrife	*Lythrum salicaria*	0.25	
07	May Lily	*Maianthemum bifolium*	0.15	
08	Common Mallow	*Malva neglecta*	0.2	
09	Common/High Mallow	*Malva sylvestris*	0.05	
10	Scentless Mayweed	*Tripleurospermum maritimum (Matricaria indorata, Chamaemelum inodorum)*	0.1	
11	Spotted Medick	*Medicago arabica*	0.1	
12	Sickle medic	*Medicago falcata/sativa subsp. falcata*	0.35	
13	Lucerne	*Medicago sativa*	0.05	
14	Common Cow-wheat	*Melampyrum pratense*	0.05	x
15	White Campion	*Silene pratensis/alba/latifolia subsp. alba (Melandrium album, Lychnis alba/vespertina)*	0.05	
16	Red Campion	*Silene dioica (Melandrium rubrum/dioicum)*	0.2	
17	Wood Melick	*Melica uniflora*	0.15	
18	White Melilot	*Melilotus alba*	0.2	
19	Whorled Mint	*Mentha x verticillata*	0.05	

20	Dog's Mercury	*Mercurialis perennis*	0.15	
21	Medlar	*Mespilus germanica*	0.05	
22	Wood millet	*Milium effusum*	0.2	
23	Three-nerved Sandwort	*Moehringia trinervia*	0.15	
24	Purple Moor-grass	*Molinia caerulea (subsp. arundinacea)*	0.35	
25	Wall Lettuce	*Mycelis/Lactuca muralis*	0.05	
26	Field Forget-me-not	*Myosotis arvensis/intermedia*	0.05	
27	Changing Forget-me-not	*Myosotis discolor*	0.1	
28	Early Forget-me-not	*Myosotis ramosissima/hispida*	0.2	
29	Water Forget-me-not	*Myosotis scorpioides/palustris*	0.2	
30	Wood Forget-me-not	*Myosotis sylvatica*	0.05	
31	Water Chickweed	*Myosoton/Stellaria aquaticum*	0.05	
32	Mousetail	*Myosurus minimus*	0.05	x
33	Bog Myrtle	*Myrica gale*	0.1	x
34	Mat-grass	*Nardus stricta*	0.15	x
35	Bog Asphodel	*Narthecium ossifragum*	0.1	x
36	Water-cress	*Nasturtium officinale (Rorippa nasturtium-aquaticum)*	0.05	x
37	Fine-leaved Water-dropwort	*Oenanthe aquatica*	0.1	
38	Tubular Water-dropwort	*Oenanthe fistulosa*	0.1	x
39	Parsley Water-dropwort	*Oenanthe lachenalii*	0.05	x
40	Common Eveningprimrose	*Oenothera biennis*	0.05	
41	Common Restharrow	*Ononis repens subsp. repens*	0.15	
42	Spiny Restharrow	*Ononis repens subsp. spinosa (Ononis spinosa/campestris)*	0.15	x
43	Adder's Tongue	*Ophioglossum vulgatum*	0.05	x
44	Early Marsh-Orchid	*Dactylorhiza incarnata*	0.1	x
45	Broad-leaved Marsh Orchid	*Dactylorhiza majalis (Orchis majalis/fistulosa/latifolia, Dactylorhiza fistulosa)*	0.05	x
46	Military orchid	*Orchis militaris*	0.05	x
47	Green-winged Orchid	*Anacamptis morio (Orchis morio)*	0.1	x
48	Southern Marsh-orchid	*Dactylorhiza praetermissa/majalis subsp. praetermissa*	0.15	x
49	Wild Marjoram	*Origanum vulgare (Thymus origanum)*	0.4	
50	Star of Bethlehem	*Ornithogalum umbellatum*	0.1	

51	Bird's-foot	*Ornithopus perpusillus*	0.15	
52	Clove-scented Broomrape	*Orobanche caryophyllacea*	0.1	x
53	Royal Fern	*Osmunda regalis*	0.4	x
54	Wood-sorrel	*Oxalis acetosella*	0.1	
55	Upright Yellow-sorrel	*Oxalis stricta/fontana/europaea*	0.1	
56	Cranberry	*Vaccinium macrocarpon (Oxycoccus macrocarpos)*	0.35	
57	Prickly Poppy	*Papaver argemone*	0.05	
58	Long-headed Poppy	*Papaver dubium*	0.1	
59	Common Poppy	*Papaver rhoeas*	0.05	
60	Sea Hard-grass	*Parapholis strigosa*	0.25	x
61	Herb-Paris	*Paris quadrifolia*	0.2	x
62	Grass of Parnassus	*Parnassia palustris*	0.2	x
63	Wild Parsnip	*Pastinaca sativa*	0.15	
64	Marsh Lousewort	*Pedicularis palustris (subsp. opsiantha)*	0.1	x
65	Lousewort	*Pedicularis sylvatica*	0.1	x
66	Water-purslane	*Lythrum/Peplis portula*	0.05	
67	Butterbur	*Petasites hybridus*	0.05	
68	Milk Parsley	*Peucedanum carvifolia*	0.1	x
69	Sand Cat's-tail	*Phleum arenarium*	0.15	
70	Black Rampion	*Phyteuma spicatum subsp. nigrum (Phyteuma nigrum)*	0.1	x
71	Oxtongue Hawkweed	*Picris hieracioides*	0.15	
72	Pillwort	*Pilularia globulifera*	0.05	x
73	Greater Burnet-saxifrage	*Pimpinella major*	0.15	
74	Burnet Saxifrage	*Pimpinella saxifraga*	0.25	
75	Greater Plantain	*Plantago major subsp. intermedia*	0.05	
76	Greater Plantain	*Plantago major subsp. major*	0.1	
77	Sea Plantain	*Plantago maritima*	0.25	x
78	Hoary Plantain	*Plantago media*	0.2	x
79	Lesser Butterfly-Orchid	*Platanthera bifolia (subsp. bifolia)*	0.1	x
80	Flattened Meadow-grass	*Poa compressa*	0.1	
81	Wood Meadow-grass	*Poa nemoralis*	0.15	

82	Swamp Meadow-grass	*Poa palustris*	0.05	x
83	Tufted Milkwort	*Polygala comosa*	0.2	x
84	Heath Milkwort	*Polygala serpyllifolia*	0.05	x
85	Angular Solomon’s seal	*Polygonatum odoratum/officinale*	0.1	x
86	Common Bistort	*Persicaria/Polygonum bistorta*	0.05	
87	Copse Bindweed	*Fallopia/Polygonum/Bilderdykia dumetorum*	0.05	
88	Water-pepper	*Persicaria/Polygonum hydropiper*	0.15	
89	Small Water-pepper	*Persicaria/Polygonum minor*	0.05	
90	Tasteless Water-pepper	*Persicaria/Polygonum mitis*	0.05	
91	White Poplar	*Populus alba*	0.1	
92	Grey Poplar	*Populus x canescens*	0.05	
93	Black Poplar	*Populus nigra*	0.05	x
94	Aspen	*Populus tremula*	0.1	
95	Trailing Tormentil	*Potentilla anglica/procumbens*	0.1	
96	Silverweed	*Potentilla anserina*	0.15	
97	Hoary Cinquefoil	*Potentilla argentea*	0.15	
98	Barren Strawberry	*Potentilla sterilis/fragariastrum (Fragaria sterilis)*	0.05	x
99	Spreading Cinquefoil	*Potentilla supina*	0.1	x
00	Spring Cinquefoil	*Potentilla tabernaemontani/verna/neumanniana*	0.1	x
01	Oxlip	*Primula elatior*	0.2	
02	Cowslip	*Primula veris/officinalis*	0.2	x
03	Wild Cherry	*Prunus/Cerasus avium*	0.15	
04	Bird Cherry	*Prunus padus*	0.15	
05	Bracken	*Pteridium aquilinum*	0.15	
06	Reflexed Saltmarsh-grass	*Puccinellia distans*	0.1	
07	Borrer's Saltmarsh-grass	*Puccinellia fasciculata/pseudodistans*	0.1	x
08	Common Fleabane	*Pulicaria dysenterica*	0.1	
09	Round-leaved Wintergreen	*Pyrola rotundifolia*	0.15	x
10	Sessile oak	*Quercus petraea/sessilis*	0.1	x
11	Allseed	*Radiola linoides*	0.15	x
12	Goldilocks Buttercup	*Ranunculus auricomus*	0.1	

13	Bulbous Buttercup	*Ranunculus bulbosus*	0.2	x
14	Lesser Spearwort	*Ranunculus flammula*	0.2	
15	Greater Spearwort	*Ranunculus lingua*	0.1	x
16	Hairy Buttercup	*Ranunculus sardous*	0.1	
17	Cursed Crowfoot	*Ranunculus sceleratus*	0.15	
18	Wild Radish	*Raphanus raphanistrum*	0.05	
19	Wild Mignonette	*Reseda lutea*	0.05	
20	Buckthorn	*Rhamnus cathartica*	0.05	
21	Greater Yellow Rattle	*Rhinanthus angustifolius/serotinus/major*	0.2	x
22	Yellow Rattle	*Rhinanthus minor*	0.15	x
23	Black Currant	*Ribes nigrum*	0.15	
24	Red currant	*Ribes rubrum/sylvestre/vulgare*	0.05	
25	Gooseberry	*Ribes uva-crispa*	0.1	
26	Great Yellow-cress	*Rorippa amphibia (Sisymbrium amphibium)*	0.15	
27	Marsh Yellow-cress	*Rorippa palustris/islandica*	0.05	
28	Creeping Yellow-cress	*Rorippa sylvestris (Nasturtium sylvestre)*	0.05	
29	Burnet Rose	*Rosa pimpinellifolia/spinosissima*	0.15	x
30	Ramanas Rose	*Rosa rugosa*	0.1	
31	Common Sorrel	*Rumex acetosa*	0.3	
32	Hybrid Dock	*Rumex pratensis(x)/x acutus/crispus x obtusifolius)*	0.15	
33	Sharp Dock	*Rumex conglomeratus*	0.05	
34	Curly Dock	*Rumex crispus*	0.15	
35	Golden Dock	*Rumex maritimus*	0.05	x
36	Marsh Dock	*Rumex palustris*	0.05	x
37	Wood Dock	*Rumex sanguineus*	0.1	
38	Narrow-leaved sorrel	*Rumex thyrsiflorus (Acetosa thyrsiflora, Rumex acetosa subsp. thyrsiflorus)*	0.15	
39	Annual Pearlwort	*Sagina apetala*	0.05	
40	Sea Pearlwort	*Sagina maritima*	0.3	
41	Knotted pearlwort	*Sagina nodosa*	0.15	x
42	Procumbent Pearlwort	*Sagina procumbens*	0.1	
43	Eared Willow	*Salix aurita*	0.05	

44	Goat Willow	*Salix caprea*	0.05	
45	Grey Willow	*Salix cinerea subsp. cinerea*	0.25	
46		*Salix dasyclados*	0.1	
47	Bay Willow	*Salix pentandra*	0.2	
48	Purple Willow	*Salix purpurea*	0.05	
49	Almond Willow	*Salix triandra/amygdalina*	0.1	
50	Basket willow	*Salix viminalis*	0.1	
51	Meadow Clary	*Salvia pratensis*	0.25	x
52	Common Elder	*Sambucus nigra*	0.3	
53	Red-berried Elder	*Sambucus racemosa*	0.05	
54	Brookweed	*Samolus valerandi*	0.1	x
55	Salad Burnet	*Sanguisorba minor subsp. minor*	0.25	x
56	Great Burnet	*Sanguisorba officinalis*	0.15	x
57	Sanicle	*Sanicula europaea*	0.1	x
58	Soapwort	*Saponaria officinalis*	0.05	
59	Scotch Broom	*Cytisus/Sarothamnus scoparius*	0.05	
60	Basil Thyme	*Clinopodium/Calamintha/Satureja/Thymus acinos (Acinos arvensis)*	0.05	x
61	Wild Basil	*Clinopodium vulgare (Satureja vulgaris, Calamintha clinopodium)*	0.15	x
62	Rue-leaved Saxifrage	*Saxifraga tridactylites*	0.1	
63	Pincushion Flower	*Scabiosa columbaria*	0.4	x
64	Bluebells	*Hyacinthoïdes non-scripta*	0.05	
65	Common Club-rush	*Schoenoplectus lacustris (Scirpus lacustris subsp. lacustris)*	0.05	x
66	Sea Club-rush	*Bolboschoenus/Scirpus maritimus*	0.2	x
67	Flat-sedge	*Blysmus compressus (Scirpus cariciformis/planifolius)*	0.15	x
68	Saltmarsh Flat-sedge	*Blysmus/Scirpus rufus*	0.15	x
69	Bristle Club-rush	*Isolepis setacea (Scirpus setaceus)*	0.05	
70	Wood Club-rush	*Scirpus sylvaticus/expansus*	0.05	
71	Grey Club-rush	*Schoenoplectus tabernaemontani (Scirpus lacustris subsp. tabernaemontani/glaucus)*	0.15	x
72	Annual Knawel (German Knotgrass)	*Scleranthus annuus subsp. annuus*	0.05	x
73	Perennial Knawel	*Scleranthus perennis*	0.25	x
	Water Figwort	*Scrophularia auriculata/balbisii/aquatica*	0.15	

74	Common Figwort	*Scrophularia nodosa*	0.05	
75	Common Skullcap	*Scutellaria galericulata*	0.1	
76	White Stonecrop	*Sedum album*	0.15	
77	Reflexed Stonecrop	*Sedum rupestre/reflexum*	0.1	x
78	Tasteless Stonecrop	*Sedum sexangulare/boloniense*	0.25	x
79	Marsh Ragwort	*Jacobaea aquatica (Senecio aquaticus)*	0.1	x
80	Marsh fleawort	*Tephroseris palustris (Senecio congestus/palustris/tubicaulis)*	0.05	x
81	Broad-leaved Ragwort	*Senecio sarracenicus/fluviatilis*	0.05	x
82	Wood Ragwort	*Senecio nemorensis/ovatus/fuchsii*	0.1	
83	Fen Ragwort	*Jacobaea paludosa (Senecio paludosus)*	0.15	x
84	Sticky Groundsel	*Senecio viscosus*	0.1	
85	Green foxtail	*Setaria viridis*	0.05	
86	Sand Catchfly	*Silene conica*	0.05	x
87	Nottingham Catchfly	*Silene nutans*	0.15	x
88	Spanish Catchfly	*Silene otites*	0.1	x
89	Bladder Campion	*Silene vulgaris*	0.1	
90	Charlock	*Sinapis arvensis*	0.05	
91	Tall Rocket	*Sisymbrium altissimum*	0.05	
92	Hedge Mustard	*Sisymbrium officinale*	0.1	
93	Lesser Water-parsnip	*Berula erecta*	0.15	x
94	Great Water-parsnip	*Great water-parsnip*	0.1	x
95	Goldenrod	*Solidago virgaurea*	0.05	x
96	Marsh Sow-thistle	*Sonchus palustris*	0.05	x
97	Least Bur-reed	*Sparganium natans/minimum*	0.1	x
98	Corn Spurrey	*Spergula arvensis*	0.15	
99	Pearlwort Spurrey	*Spergula morisonii*	0.25	
00	Greater Sea-spurrey	*Spergularia media (subsp. Angustata) (Spergularia marginata/maritima, Arenaria marina)*	0.2	x
01	Sand Spurrey	*Spergularia rubra/campestris*	0.05	
02	Salt Sandspurry	*Spergularia salina/marina*	0.25	
03	Field Woundwort	*Stachys arvensis*	0.25	x
04				

05	Bog Stitchwort	*Stellaria uliginosa/alsine*	0.05	
06	Lesser Stitchwort	*Stellaria graminea*	0.1	
07	Greater Stitchwort	*Stellaria holostea*	0.05	
08	Lesser Chickweed	*Stellaria pallida (Stellaria media subsp. pallida/subsp. apetala)*	0.15	x
09	Marsh Stitchwort	*Stellaria palustris*	0.1	x
10	Devil's-bit Scabious	*Succisa pratensis/praemorsa (Scabiosa succisa)*	0.2	x
11	Dandelion	*Taraxacum celticum*	0.05	x
12	Dandelion	*Taraxacum obliquum*	0.05	
13	Dandelion	*Taraxacum palustre*	0.05	x
14	Yew	*Taxus baccata*	0.15	x
15	Shepherd's Cress	*Teesdalia nudicaulis*	0.15	x
16	Wild Thyme	*Thymus serpyllum*	0.15	x
17	Small-leaved Lime	*Tilia cordata/ulmifolia/parvifolia*	0.15	
18	Large-leaved Lime	*Tilia platyphyllos*	0.05	
19	Upright Hedge-parsley	*Torilis japonica*	0.05	
20	Hare's-foot Clover	*Trifolium arvense*	0.05	
21	Hop Trefoil	*Trifolium campestre*	0.1	
22	Strawberry clover	*Trifolium fragiferum*	0.15	x
23	Swedish Clover	*Trifolium hybridum*	0.05	
24	Marsh Arrowgrass	*Triglochin palustris*	0.05	x
25	Yellow Oat-grass	*Trisetum flavescens*	0.25	x
26	Coltsfoot	*Tussilago farfara (Tussligo generalis)*	0.05	
27	Small-leaved Elm	*Ulmus minor/campestris/carpinifolia/procera*	0.15	
28	Small Nettle	*Urtica urens*	0.1	
29	Bilberry	*Vaccinium myrtillus*	0.2	
30	Cowberry	*Vaccinium vitis-idaea*	0.2	x
31	Marsh Valerian	*Valeriana dioica*	0.15	x
32	Common Cornsalad	*Valerianella locusta*	0.05	
33	Dark mullein	*Verbascum nigrum*	0.05	
34	Mullein	*Verbascum thapsus*	0.05	
35	Vervain	*Verbena officinalis*	0.05	

36	Green Field-speedwell	*Veronica agrestis*	0.1	
37	Blue Water Speedwell	*Veronica anagallis-aquatica*	0.05	x
38	Wall Speedwell	*Veronica arvensis*	0.2	
39	Brooklime	*Veronica beccabunga*	0.05	
40	Pink Water-speedwell	*Veronica catenata (Veronica anagallis-aquatica subsp. aquatica)*	0.05	
41	Ivy-leaved Speedwell	*Veronica hederifolia*	0.1	
42	Long-Leaf Speedwell	*Veronica longifolia*	0.2	x
43	Wood Speedwell	*Veronica montana*	0.1	
44	Marsh Speedwell	*Veronica scutellata*	0.05	
45	Thyme-leaved Speedwell	*Veronica serpyllifolia*	0.05	
46	Large speedwell	*Veronica austriaca subsp. teucrium*	0.05	x
47	Guelder Rose	*Viburnum opulus*	0.15	
48	Narrow-leaved Vetch	*Vicia sativa/angustifolia*	0.1	
49	Hairy Tare	*Vicia hirsuta*	0.1	
50	Spring Vetch	*Vicia lathyroides*	0.1	
51	Narrow-leaved Vetch	*Vicia sativa subsp. sativa*	0.05	
52	Smooth Tare	*Vicia tetrasperma subsp. tetrasperma*	0.05	
53	Dwarf Periwinkle	*Vinca minor*	0.1	
54	Field Pansy	*Viola arvensis (Viola tricolor subsp. arvensis)*	0.25	
55	Heath Dog-violet	*Viola canina*	0.15	x
56	Seaside Pansy	*Viola curtisii (Viola tricolor subsp. curtisii/subsp. maritima)*	0.15	x
57	Hairy Violet	*Viola hirta*	0.15	x
58	Marsh Violet	*Viola palustris*	0.15	
59	Early Dog-violet	*Viola reichenbachiana/sylvestris*	0.2	
60	Common Dog Violet	*Viola riviniana*	0.1	
61	Teesdale Violet	*Viola rupestris*	0.1	
62	Fen Violet	*Viola persicifolia/stagnina*	0.1	x
63	Heartsease	*Viola tricolor*	0.05	
64	Rat's-tail Fescue	*Vulpia myuros*	0.1	
65	Cat's tail	*Phleum nodosum/hubbardii/bertolonii/pratense subsp. serotinum/subsp. Nodosum*	0.05	
66	Marsh Marigold	*Caltha palustris*	0.1	x

67	Common Mouse-ear	*Cerastium holosteoides/fontanum subsp. holosteoides/subsp. Glabrescens*	0.05	x
68	Blue fescue (Hard fescue)	*Festuca brevipila/trachyphylla/cinerea/ovina subsp. cinerea*	0.1	x
69	Narrow-leaved Meadow-grass	*Poa angustifolia (Poa pratensis subsp. angustifolia)*	0.1	
70	Velvet Bent	*Agrostis canina*	0.25	
71	Brown Bent	*Agrostis vinealis/stricta/pusilla/coarctata/canina subsp. montana/var. arida*	0.25	x
72		*Carex x timmiana*	0.15	
73	Smooth Brome	*Bromus racemosus*	0.1	x
74	Blackberry	*Rubus fruticosus*	0.35	
75	Common Glasswort	*Salicornia europaea/ramosissima/brachystachya*	0.15	
76	Saltwort	*Salicornia procumbens/stricta/dolichostachya*	0.15	
77	Broad-leaved Marsh Orchid	*Dactylorhiza fistulosa/majalis subsp.majalis (Orchis majalis/fistulosa/latifolia)*	0.15	x
78	Square-stalked Willowherb	*Epilobium tetragonum/adnatum/lamyi*	0.05	
79	Sweet Briar	*Rosa rubiginosa/eglanteria*	0.1	x
80	Narrow-leaved Ragwort	*Senecio inaequidens*	0.1	
81	Oat	*Avena sativa*	0.15	
82	Rape	*Brassica napus*	0.1	
83	Barley	*Hordeum vulgare*	0.2	
84	Rye	*Secale cereale*	0.1	
85	Wheat	*Triticum*	0.05	
86	Norway Maple	*Acer platanoides*	0.05	
87	Horse Chestnut	*Aesculus hippocastanum*	0.05	
88	Red Oak	*Quercus rubra/borealis*	0.15	
89	Locust tree	*Robinia pseudoacacia*	0.05	
90	Common Elder	*Sambucus nigra cv. laciniata*	0.05	
91	Wych Elm	*Ulmus glabra*	0.05	
92	Common Spike-rush	*Eleocharis palustris (Scirpus palustris)*	0.1	
93	Common Stork's-bill	*Erodium cicutarium*	0.05	
94	Tufted Forget-me-not	*Myosotis laxa subsp. cespitosa*	0.05	
95	Toad Rush/ Frog Rush	*Juncus bufonius/minutulus+Juncus ambiguus*	0.05	
96	Heath Wood-rush	*Luzula multiflora/pallescens*	0.25	
97	Crab Apple	*Malus sylvestris/domestica/acerba (Pyrus malus)*	0.05	

98	Lesser Meadow-rue	*Thalictrum minus*	0.1	x
99	Purple Chokeberry	*Aronia x prunifolia/floribunda/atropurpurea)*	0.05	
00	Early Dog-violet/ Common Dog Violet	*Viola reichenbachiana/sylvestris+Viola riviniana*	0.05	
01		*Rubus corylifolius*	0.05	
02	Rhododendron	*Rhododendron ponticum*	0.05	
03	Snowberry	*Symphoricarpus albus*	0.05	
04	Little Green Sedge	*Carex serotina/scandinavica/tumidicarpa/demissa/oederi/ viridula*	0.1	x
05	Bifid Hemp-nettle / Common Hemp-nettle	*Galeopsis bifida/tetrahit subsp. Bifida+Galeopsis tetrahit*	0.25	
06		*Larix decidua*	0.15	
07		*Larix kaempferi*	0.25	
08		*Picea abies*	0.05	
09		*Picea sitchensis*	0.05	
10		*Pinus nigra*	0.15	
11	Black Poplar	*Populus x canadensis*	0.15	x
12	Douglas Fir	*Pseudotsuga menziesii*	0.1	
13	Potato	*Solanum tuberosum*	0.1	
14	Corn	*Zea mays*	0.05	
15	Rigid Eyebright	*Euphrasia stricta/stricta-stricta/arctica/curta/micrantha/nemorosa/borealis*	0.2	x
16	Red Bartsia	*Odontites vernus/litoralis/verna/rubra (Euphrasia odontites)*	0.15	x
17	Greater Plantain	*Plantago major subsp. major*	0.1	
18	Hairy Rock-cress	*Arabis hirsuta subsp. hirsuta*	0.05	x
19	Thyme-leaved Sandwort	*Arenaria serpyllifolia*	0.15	
20	Yellow-wort	*Blackstonia perfoliata/acuminata (Chlora perfoliata)*	0.1	x
21	Deergrass	*Trichophorum cespitosum subsp. germanicum (Scirpus cespitosus (subsp. germanicus))*	0.15	x
22	Marsh Bedstraw	*Galium palustre/elongatum*	0.3	
23	Leafy spurge	*Euphorbia esula/waldsteinii/x pseudovirgata*	0.15	
24	Green Figwort	*Scrophularia umbrosa/aquatica*	0.05	
25	Goat's-beard	*Tragopogon pratensis subsp. pratensis*	0.1	

Physical-geographical regions and vegetation types included

Table 9.3.2. Combinations of physical geographical regions and vegetation types included in the effect factor calculations.

	nutrient poor grassland	pine forest	spruce forest	deciduous forest	heath
North Sea area	x	x			
Tidal area	x				
Closed estuaries	x				
Rivers	x	x		x	
Hills	x			x	
Urban area	x				
Sea clay	x			x	
Peat	x			x	
Higher sand grounds south	x	x		x	x
Higher sand grounds north	x	x	x	x	x
Dunes	x	x		x	x

9.4. Annex to Chapter 4

9.4.1. Wetland types and criteria of the Ramsar convention

In the Ramsar Convention there are 42 wetland types, which are grouped into the three broad categories “Marine and coastal Wetlands”, “Inland wetlands” and “Human-made wetlands” (Ramsar Convention 2009b). Each wetland can belong to several of these categories (Table 9.4.1). The distribution of the wetands that are used here is shown in Figure 9.4.1.

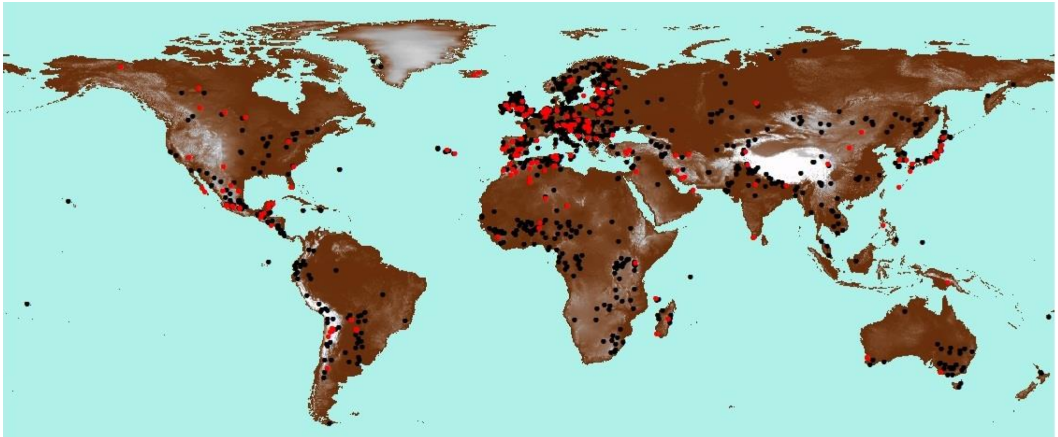


Figure 9.4.1: Distribution of Ramsar wetlands, excluding coastal and marine wetlands. The color code indicates the main water source with black being surface water-fed wetlands and red groundwater-fed wetlands.

Table 9.4.1: Overview of the 42 wetland categories, which are used in the Ramsar Convention (1996).

Broad category	Wetland category	Explanation
Marine and Coastal Wetlands	A	Permanent shallow marine waters in most cases less than 6m deep at low tide; excludes sea bays and straits
	B	Marine subtidal aquatic beds; includes kelp beds, sea-grass beds, tropical marine meadows
	C	Coral reefs
	D	Rocky marine shores; includes rocky offshore islands, sea cliffs
	E	Sand, shingle or pebble shores; includes sand bars, spits and sandy islets; includes dune systems and humid dune slacks
	F	Estuarine waters; permanent water of estuaries and estuarine systems of deltas
	G	Intertidal mud, sand or salt flats
	H	Intertidal marshes; includes salt marshes, salt meadows, saltings, raised salt marshes; includes tidal brackish and freshwater marshes
	I	Intertidal forested wetlands; includes mangrove swamps, nipah swamps and tidal freshwater swamp forest
	J	Coastal brackish/saline lagoon; brackish to saline lagoons with at least one relatively narrow connection to the sea
	K	Coastal freshwater lagoons; includes freshwater delta lagoons
	Zk(a)	Karst and other subterranean hydrological systems, marine/coastal
Inland wetlands	L	Permanent inland deltas
	M	Permanent rivers/streams/creeks; includes waterfalls
	N	Seasonal/intermittent/irregular rivers/streams/creeks
	O	Permanent freshwater lakes (over 8 ha); includes floodplain lakes
	P	Seasonal/intermittent freshwater lakes (over 8 ha); includes floodplain lakes
	Q	Permanent saline/brackish/alkaline lakes

	R	Seasonal/intermittent saline/brackish/alkaline lakes and flats
	Sp	Permanent saline/brackish/alkaline marshes/pools
	Ss	Seasonal/intermittent saline/brackish/alkaline marshes/pools
	Tp	Permanent freshwater marshes/pools; ponds (below 8 ha), marshes and swamps on inorganic soils; with emergent vegetation water-logged for at least most of the growing season
	Ts	Seasonal/intermittent freshwater marshes/pools on inorganic soils; includes sloughs, potholes, seasonally flooded meadows, sedge marshes
	U	Non-forested peatlands; includes shrub or open bogs, swamps, fens
	Va	Alpine wetlands; includes alpine meadows, temporary water from snowmelt
	Vt	Tundra wetlands; includes tundra pools, temporary waters from snowmelt
	W	Shrub-dominated wetlands; shrub swamps, shrub-dominated freshwater marshes, shrub carr, alder thicket on inorganic soil
	Xf	Freshwater, tree-dominated wetlands; includes freshwater swamp forests, seasonally flooded forests, wooded swamps on inorganic soils
	Xp	Forested peatlands; peatswamp forests
	Y	Freshwater springs; oases
	Zg	Geothermal wetlands
	Zk(b)	Karst and other subterranean hydrological systems, inland
Human-made wetlands	1	Aquaculture (e.g. fish/shrimp) ponds
	2	Ponds; includes farm ponds, stock ponds, small tanks; (generally below 8 ha)
	3	Irrigated land; includes irrigation channels and rice fields
	4	Seasonally flooded agricultural land (including intensively managed or grazed wet meadow or pasture)
	5	Salt exploitation sites; salt pans, salines, etc.
	6	Water storage areas; reservoirs/barrages/dams/impoundments (generally over 8 ha)
	7	Excavations; gravel/brick/clay pots; borrow pits, mining pools
	8	Wastewater treatment areas; sewage farms, settling ponds, oxidation basins, etc.
	9	Canals and drainage channels, ditches
	Zk(c)	Karst and other subterranean hydrological systems, human-made

There are nine criteria within the Ramsar Convention of which a wetland site needs to fulfil at least one in order to be designated as a wetland of international importance (Table 9.4.2).

Table 9.4.2: Criteria of the Ramsar convention for the designation of wetlands of international importance. The text of the explanations is directly taken from the Ramsar Convention's website (Ramsar Convention 2009a).

Criterion	Explanation
1	"A wetland should be considered internationally important if it contains a representative, rare, or unique example of a natural or near-natural wetland type found within the appropriate biogeographic region."
2	"A wetland should be considered internationally important if it supports vulnerable, endangered, or critically endangered species or threatened ecological communities."
3	"A wetland should be considered internationally important if it supports populations of plant and/or animal species important for maintaining the biological diversity of a particular biogeographic region."
4	"A wetland should be considered internationally important if it supports plant and/or animal species at a critical stage in their life cycles, or provides refuge during adverse conditions."
5	"A wetland should be considered internationally important if it regularly supports 20,000 or more waterbirds."
6	"A wetland should be considered internationally important if it regularly supports 1% of the individuals in a population of one species or subspecies of waterbird."
7	"A wetland should be considered internationally important if it supports a significant proportion

	of indigenous fish subspecies, species or families, life-history stages, species interactions and/or populations that are representative of wetland benefits and/or values and thereby contributes to global biological diversity."
8	"A wetland should be considered internationally important if it is an important source of food for fishes, spawning ground, nursery and/or migration path on which fish stocks, either within the wetland or elsewhere, depend."
9	"A wetland should be considered internationally important if it regularly supports 1% of the individuals in a population of one species or subspecies of wetland-dependent non-avian animal species."

9.4.2. Wetland geometry

All wetlands are assumed to be circular cones. This is a strong assumption and is of course not realistic in all wetland types (e.g., rivers). However, there is evidence that certain wetland types can be modelled as cones. Playa wetlands were for example modelled as truncated cones (Tsai et al. 2010), 67 lakes in Germany were also assumed to be "ideal cones" (Mehner et al. 2005) and the US Army Corps of Engineers suggest for assessing wetland functions in Florida to use circular cones for depressional wetlands (Potholes, Playa lakes, Vernal poles and cypress domes) (Noble et al. 2004).

We tested the influence of the shape of the cone, by calculating for a subset of wetlands (large, small, deep and shallow wetlands mixed) the change in FF if a more concave wetland geometry was assumed (ellipsoid).

The volume would in that case be calculated as indicated in Equation 9.4.1 with radius r and depth h.

$$V = \frac{4}{9} \cdot \pi \cdot r^2 \cdot h$$

Equation 9.4.1

with the formula for calculating the cosinus of angle α and the Pythagorean theorem (Equation 9.4.2, c being the hypothenuse in the triangle with radius r and depth h, see Figure 9.4.)

$$r^2 + h^2 = c^2$$

Equation 9.4.2

the new depth h_{new} (Equation 9.4.) and new radius r_{new} (Equation 9.4.3) can be calculated via the changed volume V_{new}.

$$h_{new} = \sqrt[3]{\frac{\frac{9}{4} \cdot \frac{V_{new}}{\pi}}{\sqrt{\frac{1}{(\cos(\alpha))^2} - 1}}}$$

Equation 9.4.3

$$r_{new} = \sqrt{\frac{h_{new}^2}{(\cos(\alpha))^2} - h_{new}^2}$$

Equation 9.4.4

For several SW-fed and GW-fed wetlands the FF with this ellipsoid has been calculated as well and has lead to very little changes in the FF (Table 9.4.3). There is the tendency that the smaller the wetland is, there is a larger influence of the geometrical shape of the wetland. However, even in a small wetland with an area of 2 ha, the difference was only 0.2%.

Table 9.4.3: Comparison of FF with different wetland geometry for some SW-fed (source: SW) or GW-fed (source: GW) wetlands. Area A and depth h are indicated, as well as the FF with the origical cone assumption and the assumption of an ellipsoid. The difference between both FFs is indicated in the last column.

Source	Wetland	A [ha]	h [m]	FF cone [$m^2 \cdot yr/m^3$]	FF ellipsoid [$m^2 \cdot yr/m^3$]	Difference [%]
SW	Parque Provincial El Tromen	30000	82	0.8614	0.8614	4.15E-08
SW	Gwydir Wetlands: Gingham and Lower Gwydir Watercourses	823	*0.5*	0.9370	0.9370	1.19E-09
SW	Parc national des Virunga	800000	112	0.4520	0.4520	4.01E-06
SW	Kauhaneva - Pohjankangas National Park	6849	0.15	1.0256	1.0256	4.36E-09
SW	Katano-kamoike	10	2.5	0.0002	0.0002	9.99E-02
SW	Odaesan National Park Wetlands	2	0.15	0.0174	0.0173	2.00E-01
GW	Laguna de Llancanelo	65000	0.3	0.0998	0.0998	5.39E-05
GW	Chippenham Fen	112	0.15	0.5447	0.5447	6.41E-10
GW	Wigry National Park	15085	73	0.0022	0.0022	1.42E-06
GW	Lac et tourbière de Mejen Ech Chitan	7	3	0.0845	0.0845	2.88E-11

Calculation of angle alpha

Each wetland is idealized as a circular cone. The cross section of one half of the cone is shown in Figure 9.4.2.

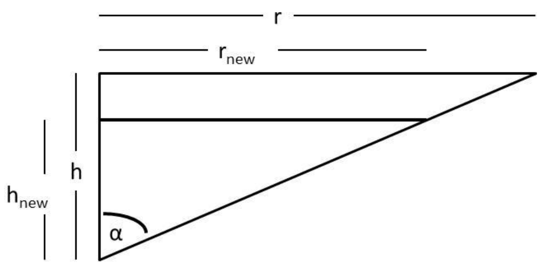


Figure 9.4.2: The cross section of one half of the idealized cone of the wetlands, with depth h and radius r. The ones with "new" are the depth and radius, respectively, after the withdrawal of the water.

The angle α is calculated as indicated in Equation 9.4.. Even if the depth and the radius of the wetland change, the angle will stay the same.

$$\alpha = arctan$$

Equation 9.4.5

9.4.1. Calculation of global surface water flows

Approach

Based on a digital elevation model (DEM, resolution 30 arc-seconds) (USGS 2012), average precipitation (P) (New et al. 2002) for 1961 to 1990 (10 arc-minutes), average satellite-derived actual evapotranspiration (AET) (Zhang et al 2009) for 1983 to 2006 (4 arc-minutes) and consumptive irrigation water use (equaling crop evapotranspiration, cropET, 0.5 arc-minutes) (Pifster et al. 2011), we modeled global natural surface water flows with a resolution of 0.5 arc-minutes. Since the modeled AET already includes irrigated crop evapotranspiration, we corrected it by adding cropET to each 0.5 arc-minute pixel to get the incremental runoff ΔQ_{pixel} of each pixel. Negative pixels were set to zero, since the corrected AET exceeded available water from precipitation (Equation 9.4.13**Error! Reference source ot found.**).

$$\Delta Q_{pixel} = \begin{cases} if\ P - AET + cropET > 0 \rightarrow \Delta Q_{pixel} = P - AET + cropET \\ if\ P - AET + cropET < 0\ \rightarrow \Delta Q_{pixel} = 0 \end{cases}$$

Equation 9.4.6

In areas with groundwater exfiltration, lakes and large rivers available, water was consequently underestimated. On the other hand, rivers were not allowed to diminish in flow or dry up. Based on the topography of the DEM, river courses were determined in Matlab[11] by using TopoToolbox (Schwanghart et al. 2010), and ΔQ_{pixel} were accumulated as surface and river flows. These accumulated flows were applied in the SW cases as inflow into the wetlands: $Q_{in,modeled}$. These were annual, natural SW flows. They are based on empirical data and are the basis for calculating FFs for SW-fed wetlands for different amounts of water consumed, thus leading to changes in the natural inflow to the wetlands.

Comparison to WaterGAP flow volumes

Our natural flow estimation for the Nile watershed was 711 km^3/yr and 390 km^3/yr for the Indus. The two versions of WaterGAP estimated natural flows in the Nile watershed to be 447 km^3/yr (WATCH 2012) (used for FF sensitivity calculation) and actual river flows at 76 km^3/yr.[14] The difference cannot only be attributed to consumption (44 km^3/yr).[13] In the Indus basin natural discharge was estimated at 197 km^3/yr (WATCH 2012), actual discharge at 121 km^3/yr (Alcamo et al. 2003) and consumption at 69 km^3/yr (WATCH 2012). The largest difference between our estimation and WaterGAP (WATCH 2012) is 87% (Huang He), which is comparable to the largest among the two WaterGAP models (taking withdrawals into account) at 82% (e.g. for Nile and Senegal). In our model, flow volumes are not allowed to diminish, since we set negative pixels zero as explained above. This is one reason for the differences between our estimation and WaterGAP.

9.4.2. Determination of k_f

The k_f value is based on a dataset of ISRIC (Batjes 2006) that contains a wealth of soil information (Figure 9.4) for five layers (0-20 cm, 20-40 cm, 40-60cm, 60-80 cm, 80-100cm) with a resolution of 5 arcminutes. For each wetland the sand content (SDTO [%]), clay content (CLPC [%]), total organic carbon content (TOTC [g/kg]), and bulk density (BULK [kg/dm^3]) are extracted in ArcGIS (ESRI 2010). With these indicators it is possible to calculate the hydraulic conductivity k_f based on the soil water characteristics equations from Saxton and Rawls (Saxton et al. 2006). In the equations, organic matter is used instead of total organic carbon. Traditionally, a factor of 1.724 is used for the transformation of organic carbon to organic matter, but that factor can actually vary (Schumacher 2002). Therefore the bulk density is used as control, when calculating k_f: The organic matter value is adapted until the calculated bulk density matches the one from the soil properties database.

The k_f value is calculated for each of the five layers separately and averaged in the end. Since the soil map is quite detailed, several soil types can lie within one Ramsar site. Therefore the k_f is determined for each soil type present and an area-weighted average is calculated.

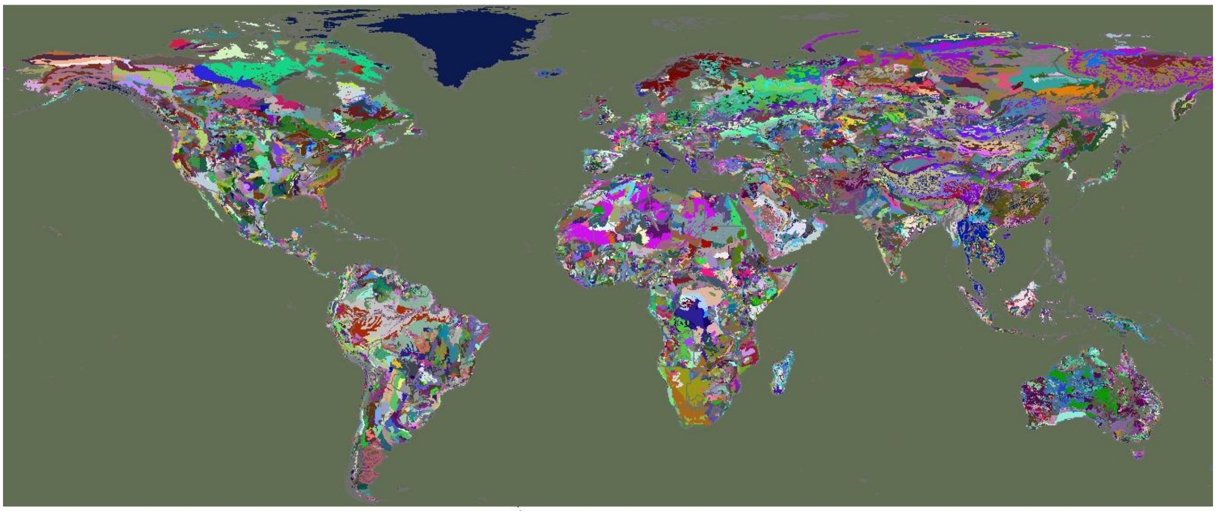


Figure 9.4-3: Soil properties of the world from ISRIC (Batjes 2006) with a resolution of 5 arcminutes.

9.4.3. Residence time and wetland area: explanation for constancy

The residence time for the SW case is calculated under the natural circumstances and it is assumed constant. Thus, the new volume is calculated directly with the residence time. The same procedure was followed for the area on which precipitation falls in the calculation of the new volume (see main manuscript). It is assumed that the original area can be taken for calculating the new volume. The two assumptions introduce only minor inaccuracies as shown in Table 9.4.4.

Table 9.4.4: Percent difference between the residence times calculated with the original volume and area and the ones calculated with the new volume and the new area.

	Difference in residence time [%]
min	4.7E-08
max	11.8
mean	0.02

The minimum and maximum differences in the SW cases are rather extreme. The second largest difference is 1.7% only. These are also the only ones with differences larger than 1%. 1031 of the 1033 SW wetlands show differences in the residence time which are smaller than 1%.

Since in general, apart from a few exceptions, the differences between the old and new residence time are smaller than 1%, it is assumed that this simplification in estimating new wetland areas can be justified.

9.4.4. Further explanations on groundwater-fed wetlands

We assumed that in a natural state, groundwater level and water level in the wetland are alike. Apart from precipitation (P), groundwater is the main source of water for these wetlands. Based on the atmospheric inputs (precipitation P (New et al. 2002)) and outputs (potential evapotranspiration PET (Trabucco et al. 2009) and surface water outflow Q_{out}) we determined whether the wetland required additional groundwater inflow (I), as shown in Figure 9.4.4. If the wetland received more atmospheric inputs than were flowing out, no additional groundwater was required for matching the water balance. Thus, these wetlands were shifted to the category of the SW-dependent wetlands. Q_{out} was taken from the surface water flow modeling, as described in the main manuscript. Since the outflow of one pixel is the inflow into the next one both in- and outflows can be determined.

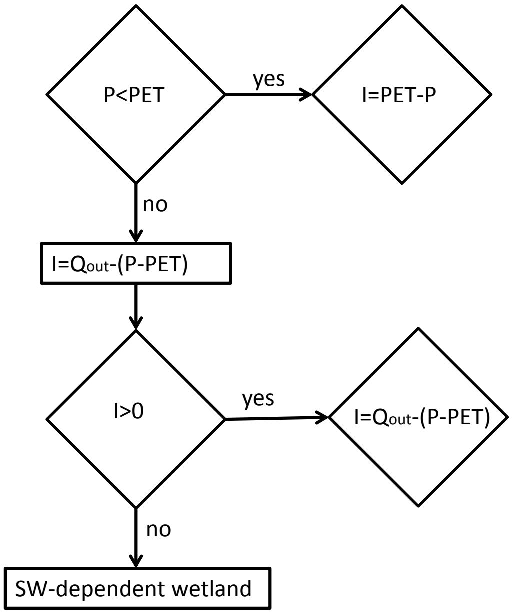


Figure 9.4.4: Decision tree for determining the net amount of infiltration into a groundwater-fed wetland. If there was a large atmospheric input (larger than the outflow), the wetland was not dependent on groundwater infiltration and was moved to the SW-fed wetlands. Abbreviations are explained in the text.

9.4.5. Explanations for the determination of s

The change in the groundwater level s, which is due to pumping an amount x from within the area of relevance is estimated with the help of the Dupuit-Thiem well formula (Stelzig 2012).

Figure 9.4.5 shall help in understanding the reasoning behind the approach taken. Only pumping of water within the area of relevance (Figure 9.4.5A) has an influence on the water level and the GW-fed wetland. Pumping outside the area of relevance will not lead to a noticeable influence on the wetland (simplifying assumption, as discussed in the main document). In order to calculate the number of wells, i.e. the number of depression cones and respective water use, the blue hatched circle of the area of relevance around the wetland can be virtually cut and spread out. Since the circumference of the weltand and the area of relevance are not the same, it results in a trapezoid, whose geometrical shape can be adapted to a rectangle (Figure 9.4.5B). We assume that the cones of depression of each well touch but do not overlap, since the determination of the influence of overlapping cones on the final drawdown is difficult. We can calculate the minimal number of optimally distributed depression cones (i.e. touching, but not overlapping) fitting into this rectangle, by assuming that their cones of depression fill the entire distance between the wetland and the end of the area of relevance. However, this assumption is an arbitrary one. Halving, for instance, the diameter of each circle leads to a fourfold increase in the number of cones

(Figure 9.4.5C). The difference in the FFs calculated with the two different number of wells (based or not on the above-mentioned assumption) is large. Increasing the number of wells leads to a decreasing value of the FF, since each individual well is pumping less and thus shows a smaller drawdown (see example in Table 9.4.5 and Figure 9.4.5D). Note that due to the scale of the picture the depression cones are shown as triangles. The water level in the well itself is however, horizontal and the shape of the cones is in reality funnel-shaped. If we imagine an ever-increasing number of wells, in the end they will the complete area of relevance without gaps with one horizontal water level in the depression cone. Thus, we assume that pumping is occurring on the whole area of relevance as if it were one big well. Pumping from the area of relevance the additional amount x (additional to I) will lead to the area of a depression cone which is slightly larger than the area of relevance itself. The area of the cone is calculated as a slighty larger area with radius C' (Figure 1C in main document). With the formula of Dupuit-Thiem (as shown in the main document) the drawdown in the center of the pumping well (i.e. the area of relevance) is calculated. The calculated drawdown in the well is the same over the whole area of relevance, since the water level in the well is horizontal (Figure 1D in the main document).

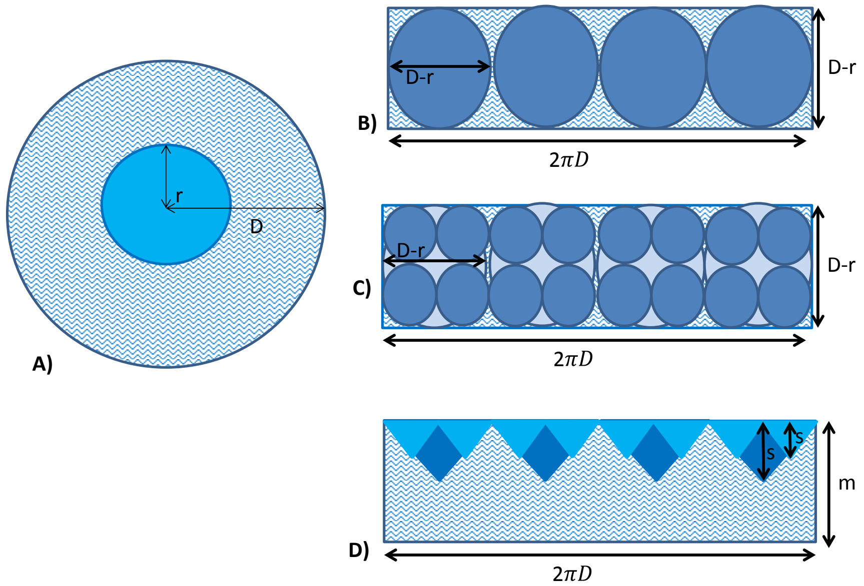


Figure 9.4.5: A) The area of relevance with radius D (blue hatched) and the area of the wetland or Ramsar site (blue) with the radius r. B) Schema for showing how many circles fit maximally into the area of a rectangle, i.e. around the area of the wetland (a trapezoid in reality, which can be converted into a rectangle. Here just for illustration). C) Increasing the number of wells (circles) by halving the diameter. D) Cross section of the area of relevance with an aquifer thickness m. In dark and light blue are the depression cones of each well schematically shown for case B and C respectively (approximation with cylindrical cones). The more wells, the smaller the puming rate and the smaller the cone of depression. Note that the cones are no triangles in reality but funnel shaped, therefore the average drawdown is what we are interested in.

In Table 9.4.5 an example of how the number of wells affects the FF is shown for two wetlands. Each individual well creates smaller disturbances to the groundwater level and thus the FF decreases. Underlying the calculation is also an assumption of a well radius (as required by the Dupuit-Thiem formula). We assumed 0.5 m as a reasonable radius, since this is the average between differen types of wells which are used today (such as dug water wells and drilled water wells in different types of ground) (Bieske et al. 1998). With this assumption a limit is set to the possible area of a cone of depression, since it has to be larger than the well. Therefore, it is not possible that the number of wells increases infinitely. In the example shown below, the "Deurnese Peelgebieden" shows radii of the cones of depression as low as 2.5 m, i.e. it is already quite close to the smallest possible cone radius. Also, increasing the number of wells leads to very unrealistic situations of more than a thousand wells on less than 4000 ha (Deurnese Peelgebieden), thus the FFs calculated like this seem to be unrealistic as well and there is the question, whether such a large number of wells would really still be independent. There is no rule or guidance as to how many wells are reasonable in one area and this would have to be different for each wetland depending on hydrogeological conditions. Therefore the more conceptual approach of one large well over the whole area of relevance is chosen.

Table 9.4.5: The influence of the number of wells on the FF, exemplified with the Wadi Wurayah Nationalpark in the United Arab Emirates (most other wetlands start with well numbers in the same ranges) and the Deurnese Peelgebieden in the Netherlands, as an example of a wetland with a larger number of wells from the outset. The numbers of the wells are calculated as explained before, by halving the individual cone diameter over and over again. The last row shows for both examples the case with one well: the whole area of relevance. This assumption of one well over the area of influence will underestimate the area loss in a wetland, since it is not a consevative assumption.

	No. of wells	FF [$m^2 \cdot yr/m^3$]
Wadi Wurayah Nationalpark	6	7.7E+03
	25	1.9E+03
	101	4.9E+02
	402	1.2E+02
	1609	3.1E+01
	6436	7.7E+00
	25744	1.9E+00
	102975	4.8E-01
	411901	1.2E-01
	1647602	3.0E-02
	whole area as one	1.5E-02
Deurnese Peelgebieden	17	2.6E+02
	67	6.0E+01
	270	1.3E+01
	1079	3.0E+00
	4317	6.6E-01
	17267	1.4E-01
	69069	3.0E-02
	276276	6.1E-03
	1105103.217	1.2E-03
	4420413	2.0E-04
	whole area as one	4.7E+00

9.4.6. Analyses of inland Ramsar sites

The analysis of the information from the Ramsar Sites Information Service (2012) for all inland wetlands leads to the following tables on the relevance of the Ramsar criteria, wetland types, biological importance, the top-twenty threats within the sites and surrounding the sites, as well as indicated social and cultural values, land tenure and ecological changes.

Note that this information was not indicated in its entirety for all wetlands. At the same time, multiple answers were possible in all categories, for all wetlands.

579 (49%) wetlands belong to European countries, 213 (18%) wetlands are in Africa, 170 (14.4%) in Asia, 43 (3.6%) in North America, 71 (6%) in Central America, 65 (5.5%) in South America, 7 (0.6%) in the Near-East and 36 (3%) in Oceania.

Table 9.4.6: Sum of the area of Ramsar sites and waterbody area per water source (SW+P, surface water and precipitation-fed; GW, groundwater-fed) and per continent. The percentage shows how large the share of the individual continent or water source is for the total size.

	Waterbody area		Ramsar site area [ha]	
	[ha]	[%]	[ha]	[%]
Global	7.38E+07	100.0	1.34E+08	100.0
SW+P Global	6.88E+07	93.3	1.24E+08	92.5
GW Global	4.96E+06	6.7	1.01E+07	7.5
Africa	4.30E+07	58.3	7.69E+07	57.3
Asia	6.63E+06	9.0	1.06E+07	7.9
Central America	1.79E+06	2.4	3.01E+06	2.2
Europe	8.04E+06	10.9	1.49E+07	11.1
Near-East	1.07E+05	0.1	1.68E+05	0.1
NorthAmerica	1.80E+06	2.4	3.42E+06	2.5
Oceania	7.89E+05	1.1	2.71E+06	2.0
South America	1.16E+07	15.7	2.26E+07	16.8

Table 9.4.7: Average size of Ramsar sites and waterbodies, depending on the water source and per continent.

Average size	Waterbody [ha]	Ramsar site [ha]
Global	6.23E+04	1.13E+05
SW+P Global	6.66E+04	1.20E+05
GW Global	3.28E+04	6.71E+04
Africa	2.02E+05	3.61E+05
Asia	3.90E+04	6.21E+04
Central America	2.52E+04	4.24E+04
Europe	1.39E+04	2.57E+04
Near-East	1.53E+04	2.41E+04
NorthAmerica	4.19E+04	7.96E+04
Oceania	2.19E+04	7.52E+04
South America	1.79E+05	3.48E+05

Table 9.4.8: Importance of the Ramsar criteria for the wetlands, globally and split according to the water source feeding the wetland.

Criterion	Global	SW+P	GW
1	800	695	105
2	853	741	112
3	730	639	91
4	586	511	75
5	332	298	34
6	355	315	40
7	197	181	16
8	184	177	7
9	11	10	1

Table 9.4.9: Importance of the Ramsar criteria for the wetlands split according to geographical occurrence. The colour code goes from green (frequent Ramsar criterion) to red (rarely used Ramsar criterion).

Criterion	Africa	Asia	Central America	Europe	Near-East	North America	South America	Oceania
1	133	109	39	405	5	33	51	25
2	142	124	65	407	4	34	57	20
3	153	96	40	333	2	29	52	25
4	133	56	41	261	2	33	42	18
5	45	69	13	139	1	24	24	17
6	66	60	3	177	1	16	16	16
7	56	31	23	56	1	10	18	2
8	48	27	13	70	1	10	13	2
9	2	0	0	5	1	2	1	0

Table 9.4.30: Inland wetland types globally and per water source. Each site can contain more than one wetland type. Highlighted in green are the three most frequent types in each category.

Wetland type	Global	SW+P	GW
L	67	67	0
M	527	496	31
N	278	246	32
O	571	525	46
P	169	157	12
Q	111	85	26
R	132	96	36
Sp	80	59	21
Ss	104	72	32
Tp	552	505	47
Ts	414	376	38
U	368	323	45
Va	46	44	2
Vt	11	11	0
W	234	211	23
Xf	318	294	24
Xp	187	168	19
Y	100	75	25
Zg	60	26	6
Zk(b)	5	32	28
Zk(c)	32	3	2
1	67	62	5
2	57	52	5
3	78	71	7
4	134	125	9
5	19	15	4
6	150	140	10
7	40	37	3
8	20	18	2
9	158	142	16

Table 9.4.11: Inland wetland types per continent. Each site can contain more than one wetland type. Highlighted in green are the three most frequent types in each category.

Wetland type	Africa	Asia	Central America	Europe	Near-East	North America	South America	Oceania
L	10	10	0	38	0	7	1	1
M	87	60	32	274	2	25	37	10
N	89	35	33	61	1	12	32	15
O	66	88	32	305	2	25	36	17
P	29	28	18	48	0	11	19	16
Q	19	30	4	31	1	3	18	5
R	39	18	5	37	1	5	21	6
Sp	19	16	6	23	0	5	9	2
Ss	28	17	5	30	0	6	15	3
Tp	98	66	20	286	5	33	32	12
Ts	64	48	21	209	5	22	29	16
U	13	38	6	274	0	13	19	5
Va	4	16	1	15	0	0	9	1
Vt	1	4	0	5	0	0	1	0
W	8	25	6	157	0	18	13	7
Xf	34	26	19	188	1	19	20	11
Xp	5	12	3	156	0	7	2	2
Y	26	5	19	39	0	3	5	2
Zg	7	3	5	12	0	0	5	0
Zk(b)	6	6	18	26	0	1	2	1
Zk(c)	2	1	0	2	0	0	0	0
1	5	13	3	45	0	0	1	0
2	3	16	4	30	0	1	3	0
3	21	23	6	25	0	1	2	0
4	9	12	8	97	0	4	4	0
5	3	3	0	11	1	0	1	0
6	34	29	18	56	1	7	4	1
7	4	1	3	30	0	1	1	0
8	2	1	2	13	0	1	1	0
9	10	14	8	115	1	6	3	1

Table 9.4.12: Top-twenty threats within the Ramsar sites. These are split per geographical region only and not per water source, since the main influence is anthropogenic and thus geopolitical reasons are more important than water sources. There can be more threats than twenty listed here, if there were some with the same importance.

Global threat	o.	Africa threat	o.	Asia threat	o.	Europe threat	o.	North America threat	o.	Central America threat	o.	South America threat	o.	Near-East threat	o.	Oceania threat	o.
habitat loss/fragmentation	13	overgrazing	9	habitat loss/ fragmentation	1	tourism-based	30	invasion exotic plants	3	habitat loss/fragmentation	7	overgrazing	3	invasion exotic animals		invasion exotic animals	0
tourism-based	89	poaching	0	overgrazing	1	eutrophication	19	agricultural runoff		agricultural development	3	poaching	2	invasion exotic plants		eutrophication	
eutrophication	79	loss/reduction of species	9	loss/reduction of species	8	Habitat loss/ fragmentation	3	eutrophication		loss/reduction of species	1	cutting/clearing of vegetation	6	drainage for agriculture/ forestry		infestation native plants	
loss/reduction of species	77	erosion	5	over-fishing	7	drainage	8	habitat loss/fragmentation		domestic sewage	9	tourism-based	6	fluctuation of water levels		salination of soil	
overgrazing	73	over-fishing	5	poaching	6	none reported	1	erosion		water diversion	8	invasion exotic animals	4	excessive hunting		habitat loss/fragmentation	
poaching	54	habitat loss/fragmentation	6	cutting/clearing of vegetation	5	pollution	6	forest exploitation		solid waste	6	mining	4	poaching		tourism-based	
over-fishing	45	sedimentation/siltation	6	erosion	3	cessation of traditional land management	3	industrial waste		acricultural runoff	5	(excessive) hunting	3	tourism-based		drainage for agriculture/ forestry	
domestic sewage	36	cutting/clearing of vegetation	2	sedimentation/ siltation	1	loss/reduction of species	3	invasion exotic animals		cutting/clearing of vegetation	5	erosion	2	agricultural runoff		overgrazing	
invasion exotic plants	31	excessive hunting	2	invasion exotic plants	0	vegetational succession	3	water diversion		erosion	4	habitat loss/fragmentation	2	cutting/clearing of vegetation		sedimentation/ siltation	
excessive hunting	27	water diversion	0	domestic sewage	9	agricultural runoff	9	drainage for agriculture/forestry		invasion exotic animals	4	loss/reduction of species	2	dam: unspecified impacts		domestic sewage	
cutting/clearing of vegetation	26	invasion exotic plants	8	human activities	8	excessive hunting	9	fertilizer		(illegal) forestry	3	egg collection	1	drainage		drainage	
erosion	23	domestic sewage	5	falling water levels	4	over-fishing	9	inappropriate management practices		sedimentation/siltation	3	transport infrastructure development	0	drainage for urban development		forest exploitation	
invasion exotic animals	18	habitat burning (non agricultural)	5	habitat burning (non agricultural)	4	domestic sewage	8	invasion exotic species		invasion exotic species	2	water diversion		erosion		invasion exotic plants	
sedimentation/ siltation	13	agricultural development	1	water diversion	4	human activities	8	tourism-based		pollution	2	eutrophication		habitat burning (non agricultural)		channelization	
water diversion	10	pesticide/herbicide	0	vegetational succession	3	invasion exotic animals	2	agricultural development		urban development	2	transport artery		habitat loss/ fragmentation		erosion	
human activities	09	inappropriate farming practices	9	eutrophication	2	agricultural development	7	channelization		eutrophication	0	domestic sewage		human activities		pesticide/ herbicide	
drainage	08	illegal fishing methods	8	expansion of settlements	2	invasion exotic plants	6	domestic sewage		invasion exotic plants	0	human activities		increased noise		salination of groundwater	

agricultural development	07	mining	8	excessive hunting	2	fluctuations water level	5	drainage		tourism-based	0	invasion exotic plants		industrial waste		cutting/clearing of vegetation	
agricultural runoff	02	over-harvesting natural forest	7	none reported	2	poaching	5	expansion of settlements		expansion of settlements		over-fishing		loss/reduction of species		dam: loss of wetland due to restriction	
pollution	01	persistent drought	6	pesticide/ herbicide	2	overgrazing	4	fluctuations water level		human activities		solid waste		overgrazing		fertilizer	
								infestation native plants		overgrazing				salt industry		grazing encroachment	
								pollution		poaching				water diversion		habitat burning (non agricultural)	
								sedimentation/siltation								vegetational succession	
								tourist/recreational facilities									
								transport artery									
								urban development									

Table 9.4.13: Top-twenty threats surrounding the Ramsar sites. These are split per geographical region only and not per water source, since the main influence is anthropogenic and thus geopolitical reasons are more important than water sources. There can be more threats than twenty listed here, if there were some with the same importance.

Global		Africa		Asia		Europe		North America		Central America		South America		Near-East		Oceania	
threat	**o.**	**threat**	**o.**	**threat**	**o.**	**threat**	**o.**	**threat**	**o.**	**threat**	**o.**	**threat**	**o.**	**threat**	**o.**	**threat**	**o.**
agricultural runoff	31	overgrazing	0	overgrazing	8	agricultural runoff	3	urban development	2	agricultural development	0	mining	0	persistent drought		eutrophication	
agricultural development	29	erosion	0	habitat loss/fragmentation	1	domestic sewage	2	agricultural development		habitat loss/fragmentation	8	tourism-based	0	agricultural development		drainage	
overgrazing	11	agricultural development	6	cutting/clearing of vegetation	4	tourism-based	2	agricultural runoff		urban development	4	agricultural development		domestic sewage		overgrazing	
erosion	7	loss/reduction of species	5	erosion	4	agricultural development	0	drainage for agriculture/forestry		erosion	3	cutting/clearing of vegetation		expansion of settlements		salination of soil	
domestic sewage	2	pesticide/herbicide	3	expansion of settlements	3	pollution	8	fertilizer		domestic sewage	1	poaching		overgrazing		drainage for agriculture/forestry	
urban development	0	sedimentation/siltation	0	agricultural development	2	drainage	3	industrial development		agricultural runoff		transport infrastructure development		pesticide/herbicide		invasion exotic animals	
habitat loss/fragmentation	8	poaching	9	agricultural runoff	1	industrial waste	0	expansion of settlements		expansion of settlements		erosion		urban development		dam: loss of wetland due to restriction	
tourism-based	9	slash&burn agriculture	9	loss/reduction of species	1	drainage for agriculture/forestry	6	habitat loss/fragmentation		fertilizer		invasion exotic plants		agricultural runoff		pesticide/herbicide	
pesticide/herbicide	5	(illegal) forestry	8	tourist/recreational facilities	1	urban development	6	pesticide/herbicide		pesticide/herbicide		loss/reduction of species		air pollution		sedimentation/siltation	
pollution	4	cutting/clearing of vegetation	7	dam: unspecified impacts	0	eutrophication	3	domestic sewage		solid waste		overgrazing		dam: unspecified impacts		agricultural development	

drainage	2	expansion of settlements	7	domestic sewage	0	fertilizer	2	drainage	forest exploitation	slash&burn agriculture	eutrophication	channelization
industrial waste	0	invasion exotic plants	7	drainage	0	forest exploitation	4	eutrophication	grazing encroachment	channelization	fluctuations of water level	domestic sewage
fertilizer	9	mining	7	none reported	0	habitat loss/fragmentation	3	forest exploitation	pollution	pesticide/herbicide	human activities	erosion
loss/reduction of species	2	habitat loss/fragmentation	6	poaching		excessive hunting	3	industrial waste	(illegal) forestry	solid waste	poaching	forest exploitation
mining	1	fertilizer	5	grazing encroachment		none reported	3	invasion exotic animals	industrial development	transport artery	sedimentation/siltation	grazing encroachment
drainage for agriculture/forestry	0	habitat burning (non agricultural)	5	habitat burning (non agricultural)		water diversion	3	mining	industrial waste	habitat loss/fragmentation	tourism-based	infestation native plants
forest exploitation	9	inappropriate farming practices	5	human activities		cessation of traditional lan management	2	pollution	sedimentation/siltation	invasion exotic animals	transport artery	mining
expansion settlements	9	over-fishing	5	over-fishing		transport artery	2	tourist/recreational facilities	cutting/clearing of vegetation	urban development	transport infrastructure development	salination of groundwater
water diversion	6	dam: unspecified impacts	4	solid waste		mining	1	transport artery	human activities	dam:unspecified impacts	water diversion	tourism-based
cutting/clearing of vegetation	6	none reported	4	desertification		dam: unspecified impacts	0	air pollution	inappropriate management practices	domestic sewage		urban development
				forest exploitation		pesticide/herbicide	0	channelization	mining	(illegal) forestry		water diversion
				inappropriate farming practices		transport infrastructure development	0	erosion		grazing encroachment		
				livestock grazing				fluctuations of water level		industrial waste		
				over-harvesting natural forest				human activities		over-fishing		
				pesticide/herbicide				invasion exotic plants		water diversion		
				sedimentation/siltation				tourism-based				
				tourism-based				transport infractructure development				
								water diversion				

Table 9.4.14: Top-ten social and cultural values of the Ramsar sites split per geographical region.

Global		Africa		Asia		Europe		North America		Central America		South America		Near-East		Oceania	
value	o.	Value	o.	Value	o.	Value	o.	Value	o.	Value	o.	Value	o.	Value	o.	Value	o.
Current scientific research	05	Livestock grazing	02	Current scientific resesarch	5	Current scientific research	66	Current scientific research	9	Archaeolo gical/ historical	7	Archaeological/ historical	4	Current scientific research		Non-consumptive recreation	2
Conservation education	69	Agriculture	6	Conservation education	3	Conservation education	48	Conservati on education	8	Agricultur e	1	Livestock grazing	8	Archaeologica l/ historical		Current scientific research	0
Non-consumptive recreation	89	Traditional cultural	1	Tourism	1	Non-consumptive recreation	48	Non-consumptive recreation	6	Tourism	6	Tourism	7	Water supply		Archaeologic al/ historical	9
Tourism	72	Conservatio n education	5	Water supply	1	Archaeological / historical	87	Sport fishing	7	Conservat ion education	4	Current scientific research	4	Non-consumptive recreation		Conservation education	9
Archaeological/ historical site	66	Tourism	5	Unspecified fishing	8	Tourism	79	Archaeolo gical/ historical	6	Traditiona l cultural	4	Traditional cultural	4	Aesthetic		Traditional cultural	
Livestock grazing	59	Current scientific research	4	Non-consumptive recreation	3	Livestock grazing	55	Tourism	6	Current scientific research	3	Conservation education	1	Conservation eucation		Livestock grazing	
Traditional cultural	94	Unspecified fishing	1	Traditional cultural	9	Unspecified fishing	30	Sport hunting	4	Livestock grazing	3	Non-consumptive recreation	5	Honey collection		Tourism	
Unspecified fishing	66	Water supply	6	Aesthetic	7	Aesthetic	27	Aesthetic		Water supply	8	Water supply	5	Livestock grazing		Sport fishing	
Agriculture	44	Archaeologi cal/ historical	3	Agriculture	6	Traditional cultural	11	Livestock grazing		Unspecifi ed fishing	7	Agriculture	4	Tourism		Sport hunting	
Aesthetic	10	Subsistence fishing	9	Livestock grazing	5	Sport fishing	10	Traditional cultural		Non-consumptive recreation	4	Subsistence fishing	4			Water supply	

Table 9.4.15: Indicated ecological changes per geographical region. Several categories might be named per wetland. The least amount of information is available for ecological changes, since not all information sheets contain information concerning the ecological changes. The color code indicates the frequency with green and red meaning frequently named and rarely mentioned, respectively.

Ecological change	Africa	Asia	Central America	Europe	Near-East	North America	South America	Oceania
no negative changes forseeable	29	20	0	133	1	6	10	15
less significant changes likely to occur	23	43	6	87	0	10	11	6
less significant changes occurring	60	29	14	168	2	8	18	12
ecological enhancement since designation	2	2	1	16	0	3	1	0
significant negative changes occurring	22	26	18	37	1	1	11	0
significant negative changes likely to occur	10	14	2	17	0	3	6	2

In Figure 9.4. the distribution of wetlands in Uganda and Kenya according to the World Resources Institute (World Resources Institute 2010a, b) is shown along with the wetlands which are protected under the Ramsar convention. Clearly, only the smallest part of the wetlands is protected, even though many of them will be of great importance for flora and fauna, along with the wetlands which are designated Ramsar sites.

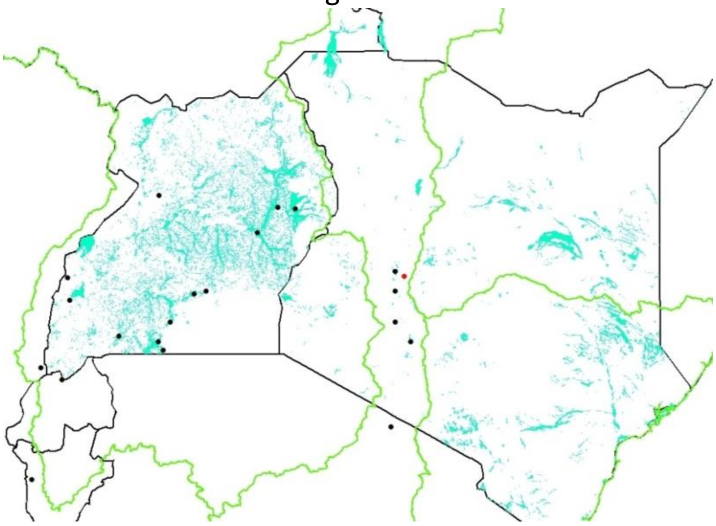


Figure 9.4.6: A snapshot of the area of Uganda and Kenya (country boundaries in black). The present major watersheds are indicated in green (Lehner et al. 2008). Black dots show the SW-wetlands and red dots show the GW-wetlands which are designated as Ramsar sites. The wetland areas of Uganda and Kenya are shown in blue (World Resources Institute 2010a, b).

Azraq Oasis

The Azraq Oasis is a Ramsar site in Jordan. On an area of 7’327 ha (Ramsar 2012) it provided a habitat for numerous aquatic and terrestrial species and was considered as a vital wetland for migratory birds (Shah et al. 2000). Due to heavy groundwater pumping for irrigation and the water supply of Amman, the wetland started to deteriorate and eventually dried up to a large extent (Shah et al. 2000). Small-scale withdrawals had been common during the 1960s and 70s, but larger scale pumping operations in the Azraq basin started in 1980, withdrawing as much as $31.6 \cdot 10^6$ m^3/yr in 1988 (The Royal Society for the

Conservation of Nature 2012). In 1988 one of the two spring systems had already dried up (Ramsar 2013), and in 1993 no surface water body remained (The Royal Society for the Conservation of Nature 2012). The largest part of the oasis was thus lost within a decade. Currently, efforts are underway to recover part of the oasis (The Royal Society for the Conservation of Nature 2012).

The fate factor for the Azraq Oasis, a GW-fed wetland, for 1’000’000 m3/yr consumption was calculated to be FF =64.9 $m^2 \cdot yr/m^3$. On average 18 million m^3/yr were withdrawn from 1981 to 1989 from both government and private wells Ramsar 2013). The water requirement ration from AQUASTAT for Jordan, estimating the efficiency of water consumption, is 49% (FAO 2013). That means that roughly half of the withdrawn water is flowing back to the environment and is also partially recharging the groundwater. With our FF this average water consumption would mean that the oasis is dry within one year. Our FF is thus overestimating the speed of damage occurrence since it took a decade for the wetland to completely dry up. This can be attributed to the steady-state assumption and simplified hydrological modelling, since we do not know the exact subterranean inflows and neglected all possible surface inflows (precipitation, seasonal streams). Using the FF for 1’000 m^3/yr, which is too small for application because the amount consumed is larger than 1’000’000 m^3/yr, the model predicts 87 years until desiccation of the wetland. The values for 1’000 m^3/yr and 1’000’000 m^3/yr thus embrace the actual value and are therefore quantifying reasonable time frames and damages.

Lake Chad

Lake Chad is shared by Cameroon, Chad, Nigeria and Niger. In the 60’s the lake measured about 22’000 km^2 and was shrinking to about 300 km^2 by the 80’s, mostly due to climate variability and irrigation withdrawals (Gao et al. 2011). In the 90’s the lake received from its main tributary, the Chari river, an inflow that was reduced by about 8 km^3/yr due to withdrawals for irrigation upstream of Lake Chad(Gao et al. 2011).

We have FFs for four Ramsar sites for the respective portions of the lake: the “Partie tchadienne du lac Tchad” in Chad, “Lac Tchad” in Niger, “Lake Chad Wetlands in Nigeria” in Nigeria and “Partie Camerounaise du Lac Tchad”. The area-weighted FF for a consumption of both 1’000 m^3/yr and 1’000’000 m^3/yr is 0.53 $m^2 \cdot yr/m^3$. According to these FFs, it would take five years to lose 21’700 km^2 with a consumption of 8 billion m^3/yr. Before the 90s the irrigation withdrawals were likely smaller, thus the time for the area to become desiccated would have been longer. Again, regarding the uncertainties and simplified modelling, the result seems appropriate for global screening and should be refined in a case study.

9.4.7. Further results and analyses

In Figure 9.4. we show the parameters that were varied in the sensitivity analyses for both surface water-fed wetlands and groundwater-fed wetlands. For the surface water-fed wetlands the calculation of new radii r_{new} after water consumption contributed most to the sensitivity of the SW-fed wetlands since this is a non-linear step. The surface water flows (own model or WaterGap (WATCH 2011)) also contributed to uncertainty. GW-fed wetlands have much higher sensitivity, often due to an interrelation of different factors. Area changes (Ramsar area or waterbody area), as well as changes in water depth (especially decreasing depth by 50%) were relatively sensitive. Increases of water depth lead only to moderate sensitivity. For those wetlands for which a water depth of 1 m was assumed, it was changed to 3.8 m but the change in FF was comparably small (74%) for these wetlands. One of the big drivers of the sensitivity for GW-fed wetlands is the non-linear Dupuit-Thiem well formula for estimating the groundwater drawdown and the subsequent calculation of new radii. These steps and thus the amount of water consumed are dominating the sensitivity of the FF_{GW}. The underlying assumption of e.g. a cone for modelling a wetland is not included here.

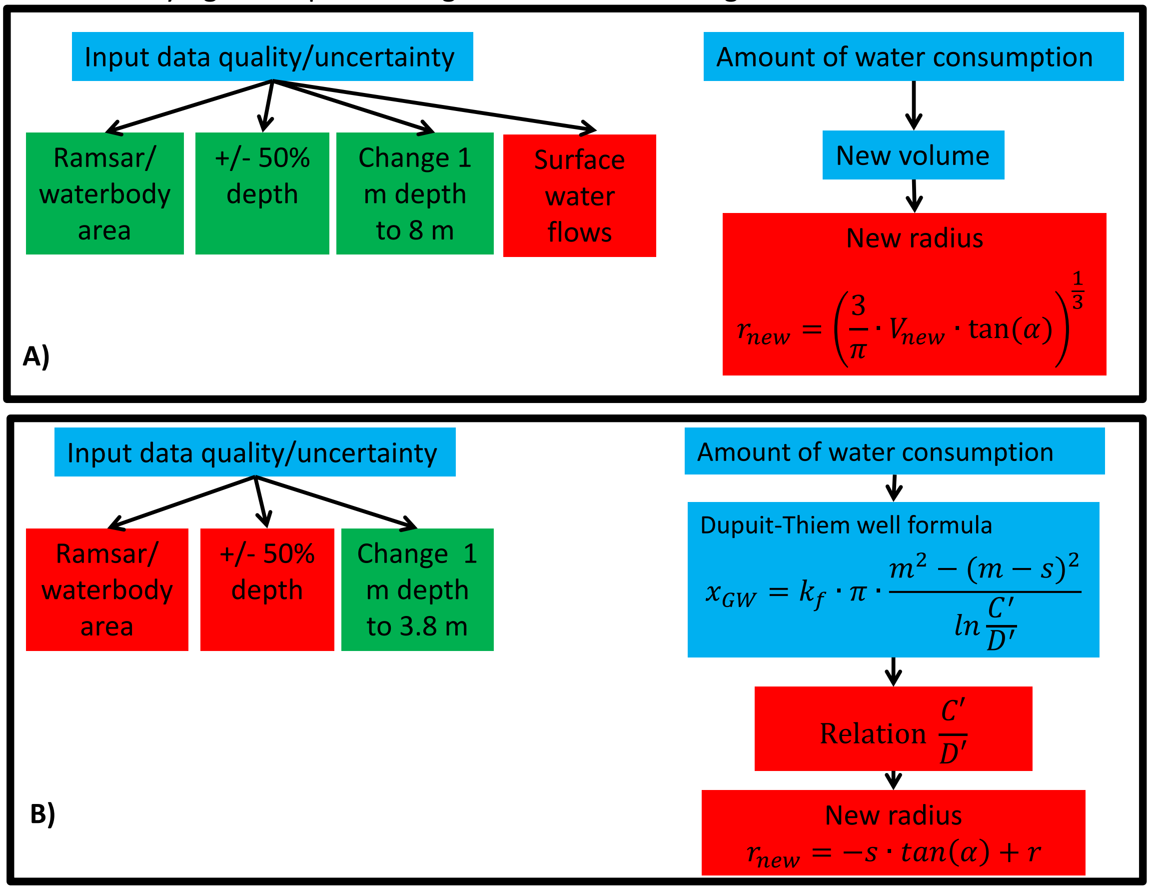
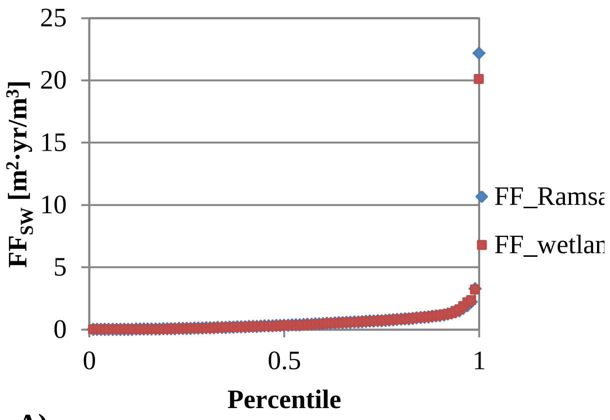
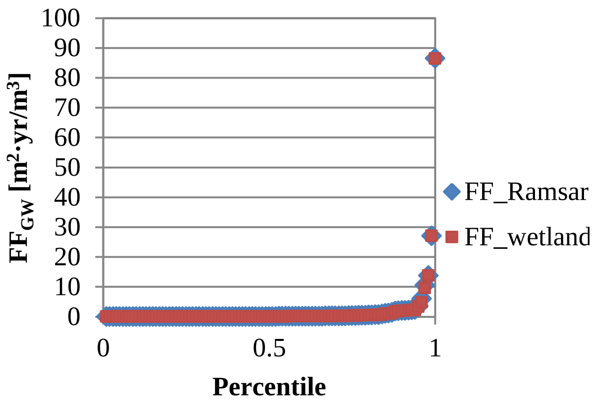


Figure 9.4.7: Drivers for uncertainty and sensitivity. Green boxes show that this particular change had only a limited influence on the sensitivity of the FFs, red boxes show the dominant drivers. Blue boxes show where the uncertain parameters are affiliated (either already as input data or from calculation steps). A) Drivers for surface-water-fed wetlands. B) Drivers for groundwater-fed wetlands.

An analysis by percentiles is shown in Figure 9.4.. Statsitical results per continents are shown in Table 9.4.4. North America also contains Central America, and the Near-East is included in Asia.



A)



B

Figure 9.4.8: A) Distribution of FF_{SW} over the percentiles. B) Distribution of FF_{GW} over the percentiles.

Table 9.4.46: Statistical analysis of the FFs per continent, once with the Ramsar area as basis and once with the wetland area as basis. The absolute number of sites per continent is mentioned in the text in the SI, section S8.

Continent	Statistics	SW			GW	
		FF_{Ramsar} [$m^2 \cdot yr/m^3$]	$FF_{waterbody}$ [$m^2 \cdot yr/m^3$]	$FF_{WaterGAP}$ [$m^2 \cdot yr/m^3$]	FF_{Ramsar} [$m^2 \cdot yr/m^3$]	$FF_{waterbody}$ [$m^2 \cdot yr/m^3$]
AFRICA	average	0.6	0.6	0.7	0.7	0.7
	min	0.00001	0.00005	0.000001	0.0003	0.0003
	max	22.2	20.1	50.7	13.2	13.2
	standard dev.	2.0	1.7	4.1	2.6	2.6
ASIA	average	0.6	0.6	0.2	0.10	0.04
	min	0.0002	0.0001	0.0000001	0.0004	0.0003
	max	6.5	5.2	4.6	1.1	0.2
	standard dev.	1.0	0.9	0.6	0.3	0.1
AUSTRALIA	average	0.7	0.7	0.2	0.8	0.8
	min	0.001	0.0002	0.0003	0.034	0.034
	max	2.6	2.6	2.2	2.2	2.2
	standard dev.	0.6	0.7	0.5	1.2	1.2
CENTRAL AMERICA	average	0.3	0.3	0.1	0.1	0.1
	min	0.0001	0.0001	0.00002	0.0001	0.0001
	max	1.4	1.4	1.3	0.5	0.5
	standard dev.	0.3	0.3	0.3	0.1	0.1
EUROPE	average	0.5	0.5	0.1	1.7	1.7
	min	0.00004	0.00001	0.00001	0.0003	0.0002
	max	3.4	3.4	2.6	35.0	35.0
	standard dev.	0.4	0.4	0.2	5.2	5.3
NEAR-EAST	average	0.4	0.3	0.3	0.1	0.1
	min	0.09	0.05	0.0004	0.015	0.015
	max	0.7	0.8	1.5	0.1	0.1
	standard dev.	0.2	0.3	0.7	0.1	0.1
NORTH AMERICA	average	0.3	0.4	0.1	9.6	9.5
	min	0.0001	0.00004	0.00005	0.001	0.001
	max	1.9	2.3	0.8	86.5	86.3
	standard dev.	0.5	0.6	0.2	27.1	27.1
SOUTH AMERICA	average	0.6	0.5	0.5	0.1	0.1
	min	0.0007	0.0004	0.00003	0.02	0.02

max	2.9	2.4	8.2	0.2	0.2
standard dev.	0.6	0.6	1.3	0.06	0.06

The humidity zones as defined in Deichmann et al. (1991) are shown in Figure 9.4..

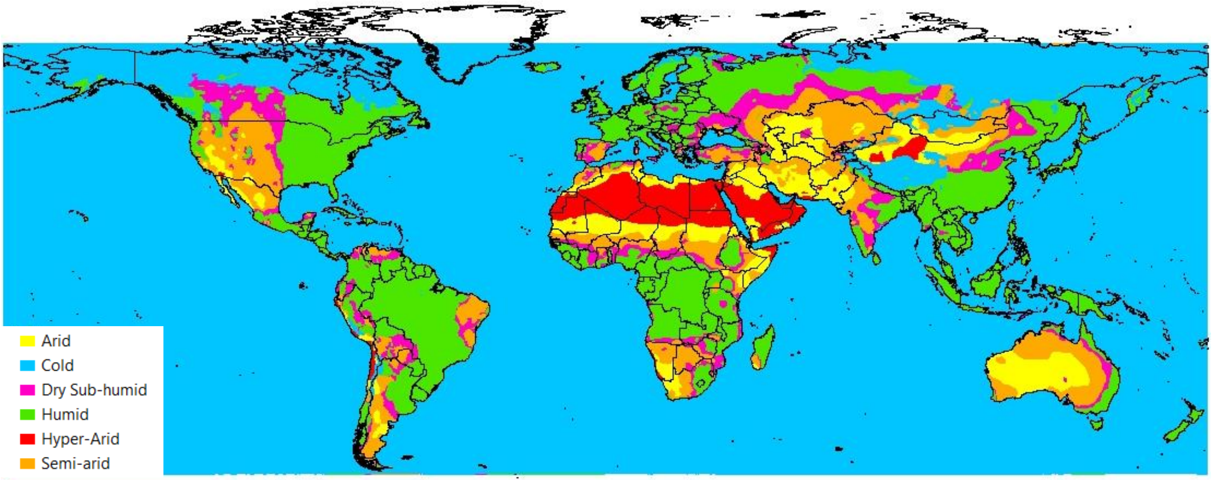


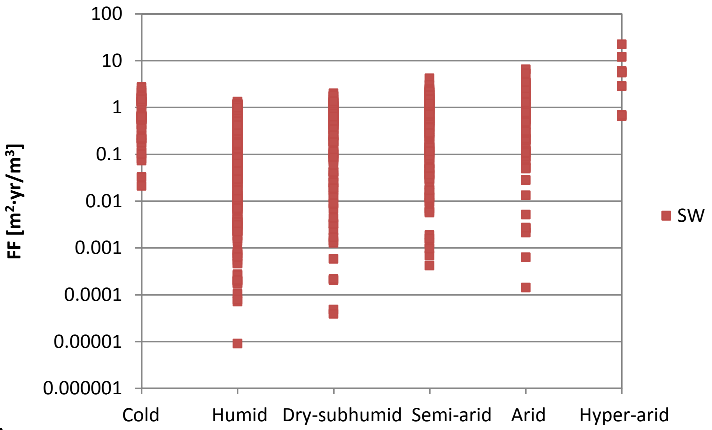
Figure 9.4.9: Humidity zones as defined in Deichmann et al.(1991).

The statistical results for the FFs per humidity zone are shown in Table 9.4..

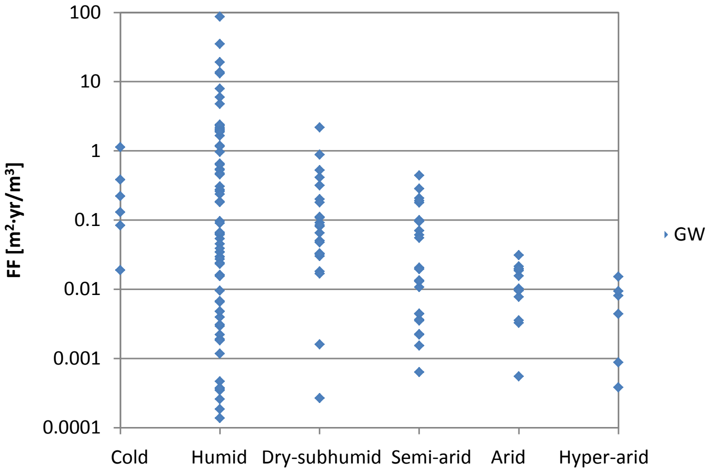
Table 9.4.17: Statistical results and counts of Ramsar sites per humidity zone, once with the Ramsar area as basis and once with the wetland area as basis.

Humidity zone	Statistics	SW FF$_{Ramsar}$ [$m^2 \cdot yr/m^3$]	SW FF$_{waterbody}$ [$m^2 \cdot yr/m^3$]	SW FF$_{WaterGAP}$ [$m^2 \cdot yr/m^3$]	GW FF$_{Ramsar}$ [$m^2 \cdot yr/m^3$]	GW FF$_{waterbody}$ [$m^2 \cdot yr/m^3$]
ARID	count	50	50	50	12	12
	average	1.13	1.08	0.49	0.01	0.01
	min	0.00014	0.00008	0.00001	0.0006	0.0006
	max	6.46	5.25	4.65	0.03	0.03
	standard dev.	1.42	1.35	0.89	0.01	0.01
COLD	count	87	87	87	7	7
	average	0.67	0.63	0.28	0.28	0.15
	min	0.0002	0.0001	0.0000	0.017	0.017
	max	2.70	2.68	2.98	1.13	0.38
	standard dev.	0.57	0.57	0.63	0.39	0.13
DRY SUBHUMID	count	129	129	129	23	23
	average	0.52	0.48	0.13	0.24	0.24
	min	0.00004	0.00002	0.00004	0.0003	0.0003
	max	1.98	2.47	1.23	2.17	2.17
	standard dev.	0.47	0.50	0.24	0.47	0.47
HUMID	count	625	625	625	77	77
	average	0.35	0.35	0.05	2.84	2.75
	min	0.00001	0.00001	0.00000	0.00014	0.00014
	max	1.32	1.32	0.90	86.51	86.34
	standard dev.	0.32	0.32	0.11	10.88	10.85
HYPER-ARID	count	8	8	8	6	6
	average	6.92	5.43	12.98	0.006	0.006
	min	0.645	0.286	0.029	0.0004	0.0004
	max	22.17	20.10	50.70	0.02	0.02
	standard dev.	7.14	6.35	16.66	0.006	0.006
SEMI-ARID	count	134	134	134	26	26
	average	0.68	0.72	0.28	0.07	0.07
	min	0.0004	0.0002	0.0001	0.0006	0.0006
	max	4.15	4.60	2.45	0.44	0.44
	standard dev.	0.75	0.83	0.45	0.11	0.11

In Figure 9.4.5 the relationship between the FFs and the humidity zones is shown. For the FF$_{SW}$ the trend is that the more arid the climate gets, the larger the FFs get, and also the range of the FFs gets larger. For FF$_{GW}$ the largest values lie in the humid zone, followed by the cold zone, and the smallest are found in the hyper-arid zone.



A)



B)

Figure 9.4.50: Relationship between humidity zones and FFs for A) FF_{SW}; B) FF_{GW}. Note that the axis of abscissae is log-scale.

Results for rose production in Kenya and the Netherlands

The calculation of the area loss for wetlands in Kenya and the Netherlands after the production of one rose shows the importance of choosing the correct FF, especially for the GW-fed wetlands. The FFs and the factor between the FFs are shown below. The resulting area loss is shown for the base case (1000 m^3/yr), as well as the minimum (FF divided by k) and maximum (FF multiplied by k) area loss. For the SW-fed wetlands the difference is negligible. For the GW-fed wetland (lake Elmenteita), the difference can be up to 6 orders of magnitude. Therefore, as written in the main manuscript, we do not recommend general use of the factors for GW consumption, without knowledge of the local conditions.

Table 9.4.18: Fate factors for 1000 m^3/yr water consumption and the factor k between the FFs with 1000 m^3/yr and 1'000'000 m^3/yr. None of the wetlands were dry when consuming 1'000'000 m^3/yr).

Wetland	FF [m^2·yr/m^3]	k
lake Naivasha	0.801	1.001
Haringvliet	0.826	1.002
Lake Elmenteita	0.007	980.658

Table 9.4.19: Area loss for three wetlands in Kenya and the Netherlands after producing one rose. The base case is calculated with the FF for 1000 m^3/yr, being the smaller of the two available FFs for each of the wetlands. The difference in possible area loss is calculated with a minimum and maximum scenario, based on the factor between the two FFs for each wetland.

	Wetland	Base case [$m^2 \cdot yr$]	min [$m^2 \cdot yr$]	max [$m^2 \cdot yr$]
SW-fed	lake Naivasha	2.69E-03	2.69E-03	2.69E-03
	Haringvliet	1.36E-03	1.35E-03	1.36E-03
GW-fed	Lake Elmenteita	4.87E-06	4.96E-09	4.77E-03

Comparison of FF_{SW} with our own model and FF_{SW} with WaterGAP

The largest differences for FF_{SW} were recorded in the Indus basin and Nicaragua. In the Indus basin, the wetland with the largest difference is located in the Indus itself in WaterGAP and further away in our estimate. In reality, the wetland is located on the Indus plain about 30 km from the river itself (Ramsar 2012). There were other wetlands in the vicinity where the differences were less striking. For the sake of completenes. There is one wetland that was not situated correctly in our model, but there were three that were not correct in WaterGAP (due to the coarser resolution of WaterGAP). Those two sites that were located by both models correctly also showed the smallest differences between the FF_{SW} ($FF_{SW,\ WaterGAP}$ being 109% and 80% of the FF_{SW} with our model, respectively).

In Nicaragua, the wetland in question was once situated close to a river (our model) and once further away (WaterGAP) with less SW flow (and thus a larger FF). The wetland is in reality a lacustrine system and includes part of a local river (Ramsar 2012).

Table 9.4.20: Information for the wetlands in the Indus basin. Information from the Ramsar Sites Information Service (RIS) is given to control the location of the wetlands. Indicated in green are locations which match with the description according to the RIS, while orange indicates that the underlying model of the surface water flows does not match with the information given by the RIS. Taking the FFSW that is calculated with our model as 100% the difference to the FFSW, WaterGAP is given.

Wetland name	Information from RIS[22]	Location according to WaterGAP	Location according to own model	$FF_{SW,WaterGAP}$ as % of $FF_{SW,own}$
Indus Dolphin Reserve	170 km of the Indus itself	Situated in the Indus	Ca 8 km from Indus	168
Drigh Lake	Situated ca 30 km from Indus	Situated in the Indus	Ca 40 km from Indus	0.0002
Deh Akro-II Desert Wetland Complex	In desert, fed by seepage from irrigation canals and rain	No visible river close-by	No visible river close-by	80
Hub (Hab) Dam	Situated on Hub river	No visible river close-by	Situated on a river next to Indus	205
Kinjhar (Kalri) lake	Fed by a canal and seasonal rivers	Situated in the Indus	No visible river close-by	0.1
Haleji Lake	Fed by canals	No visible river close-by	No visible river close-by	109

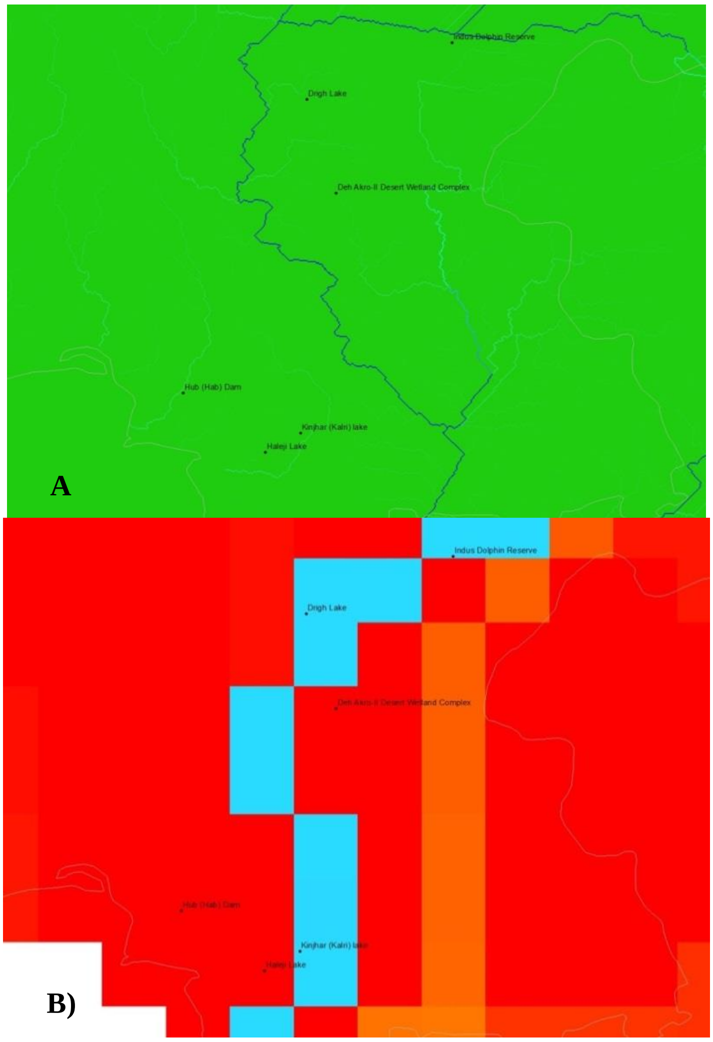


Figure 9.4.11: A) Part of the Indus basin with some SW-fed wetlands in our surface water model. Rivers are indicated in light to dark blue pixels. The resolution is 0.5 arc-minutes. B) Same part of the Indus basin with the map from WaterGAP.[13] The Indus is shown in blue pixels. The resolution is 0.5 arc-degrees.

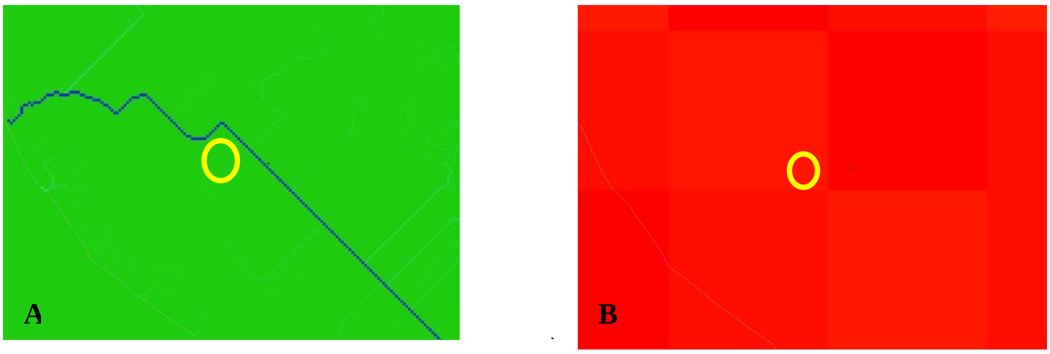
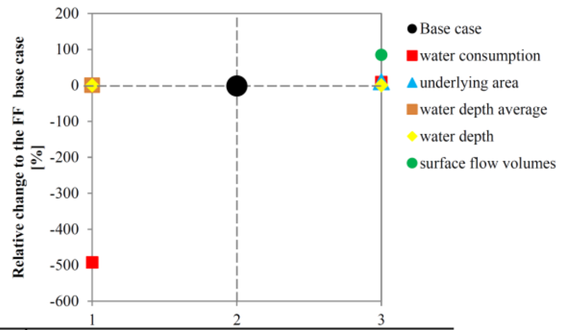


Figure 9.4.62: Location of the wetland „Sistema Lagunar de Tisma" in Nicaragua (black dot within the yellow circle) in our surface water model (A) and WaterGAP[13] (B). In A) green means a low surface flow, while rivers are indicated with different shades of blue. In B) red indicates low flows, more surface flow and rivers are indicated with colors ranging from yellow to blue.

Results for the sensitivity analysis per climate zone

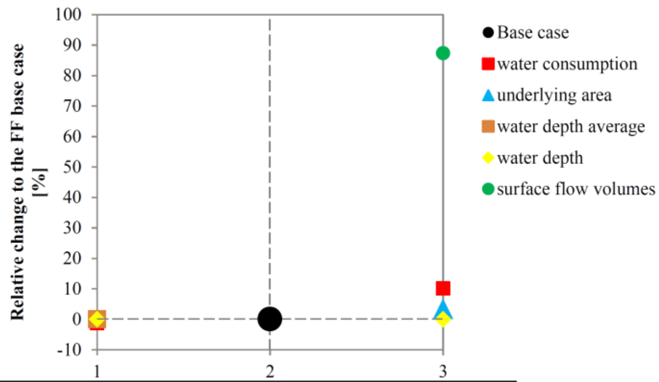
The following graphs for sensitivity analyses according to climate zone show essentially the same picture as the global ones included in the main manuscript. What may change is the magnitude of some of the sensitivities. Related to the change in underlying area, 136 FF_{SW}s remained equal

(A_{Ramsar}=$A_{waterbody}$), 701 had smaller $FF_{SW,waterbody}$ (due to e.g. equal inflows on a smaller waterbody area and depending on the location of wetlands and river networks) and 196 had larger $FF_{SW,waterbody}$ (smaller flows, but also smaller area).



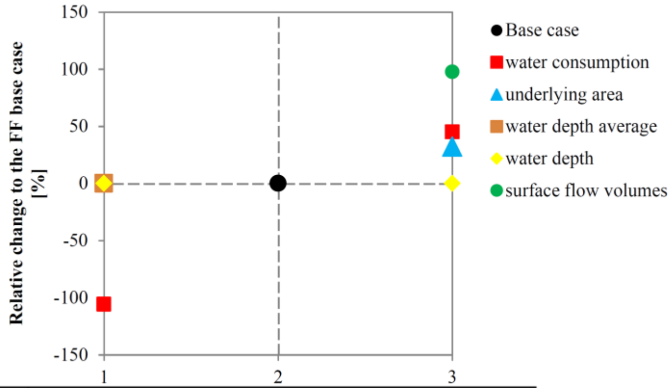
parameter	1	2 (base case)	3
Water consumption [m³/yr]	10	**1000**	1'000'000
underlying area [ha]	-	**Ramsar area**	waterbody area
water depth average [m]	8 m (if 1m was assumed)	**data from literature**	-
water depth [m]	-50%	**data from literature**	+50%
surface flow volumes [-]	-	**own model**	WaterGap

Figure 9.4.73: Overview of the sensitivity analysis for the FF of SW-fed wetlands. The values shown here are the relative changes of the median global FF values for each parameter change in arid climates. In the table below the x-axis the changed parameters and respective values for calculating the new FF for the sensitivity analysis are shown. Parameter values listed under number 1 are shown in the graph where number 1 is indicated on the x-axis. The same applies for numbers 2 and 3.



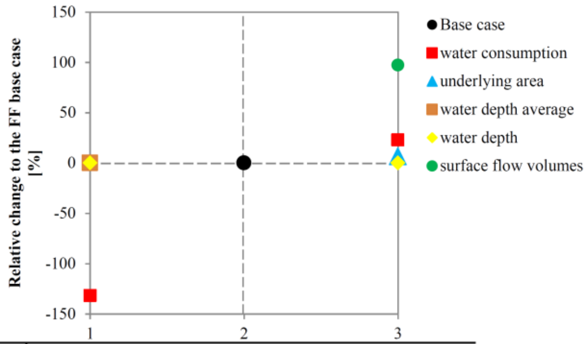
parameter	1	2 (base case)	3
Water consumption [m³/yr]	10	**1000**	1'000'000
underlying area [ha]	-	**Ramsar area**	waterbody area
water depth average [m]	8 m (if 1m was assumed)	**data from literature**	-
water depth [m]	-50%	**data from literature**	+50%
surface flow volumes [-]	-	**own model**	WaterGap

Figure 9.4.14: Overview of the sensitivity analysis for the FF of SW-fed wetlands. The values shown here are the relative changes of the median global FF values for each parameter change in cold climates. In the table below the x-axis the changed parameters and respective values for calculating the new FF for the sensitivity analysis are shown. Parameter values listed under number 1 are shown in the graph where number 1 is indicated on the x-axis. The same applies for numbers 2 and 3.



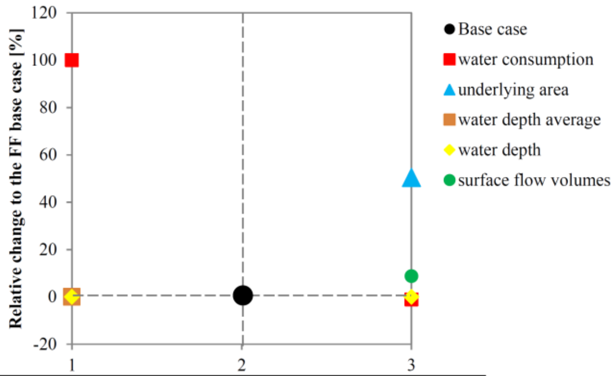
parameter	1	2 (base case)	3
Water consumption [m³/yr]	10	**1000**	1'000'000
underlying area [ha]	-	**Ramsar area**	waterbody area
water depth average [m]	8 m (if 1m was assumed)	**data from literature**	-
water depth [m]	-50%	**data from literature**	+50%
surface flow volumes [-]	-	**own model**	WaterGap

Figure 9.4.8: Overview of the sensitivity analysis for the FF of SW-fed wetlands. The values shown here are the relative changes of the median global FF values for each parameter change in dry-subhumid climates. In the table below the x-axis the changed parameters and respective values for calculating the new FF for the sensitivity analysis are shown. Parameter values listed under number 1 are shown in the graph where number 1 is indicated on the x-axis. The same applies for numbers 2 and 3.



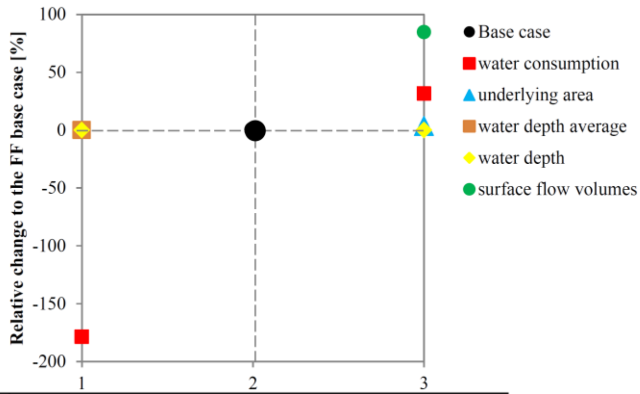
parameter	1	2 (base case)	3
Water consumption [m³/yr]	10	**1000**	1'000'000
underlying area [ha]	-	**Ramsar area**	waterbody area
water depth average [m]	8 m (if 1m was assumed)	**data from literature**	-
water depth [m]	-50%	**data from literature**	+50%
surface flow volumes [-]	-	**own model**	WaterGap

Figure 9.4.16: Overview of the sensitivity analysis for the FF of SW-fed wetlands. The values shown here are the relative changes of the median global FF values for each parameter change in humid climates. In the table below the x-axis the changed parameters and respective values for calculating the new FF for the sensitivity analysis are shown. Parameter values listed under number 1 are shown in the graph where number 1 is indicated on the x-axis. The same applies for numbers 2 and 3.



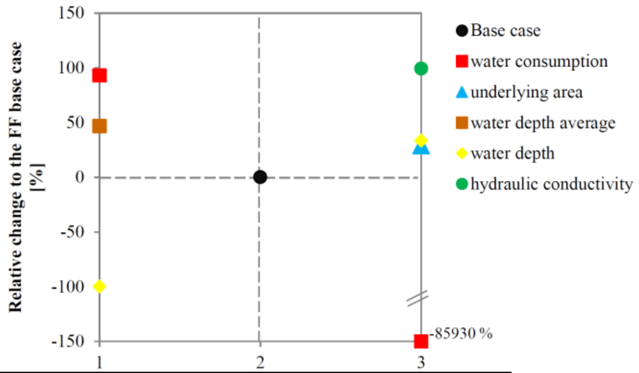
parameter	1	2 (base case)	3
Water consumption [m^3/yr]	10	**1000**	1'000'000
underlying area [ha]	-	**Ramsar area**	waterbody area
water depth average [m]	8 m (if 1m was assumed)	**data from literature**	-
water depth [m]	-50%	**data from literature**	+50%
surface flow volumes [-]	-	**own model**	WaterGap

Figure 9.4.17: Overview of the sensitivity analysis for the FF of SW-fed wetlands. The values shown here are the relative changes of the median global FF values for each parameter change in hyper-arid climates. In the table below the x-axis the changed parameters and respective values for calculating the new FF for the sensitivity analysis are shown. Parameter values listed under number 1 are shown in the graph where number 1 is indicated on the x-axis. The same applies for numbers 2 and 3.



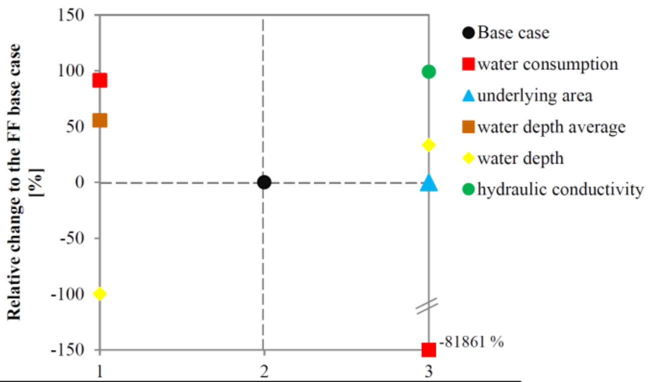
parameter	1	2 (base case)	3
Water consumption [m^3/yr]	10	**1000**	1'000'000
underlying area [ha]	-	**Ramsar area**	waterbody area
water depth average [m]	8 m (if 1m was assumed)	**data from literature**	-
water depth [m]	-50%	**data from literature**	+50%
surface flow volumes [-]	-	**own model**	WaterGap

Figure 9.4.18: Overview of the sensitivity analysis for the FF of SW-fed wetlands The values shown here are the relative changes of the median global FF values for each parameter change in semi-arid climates. In the table below the x-axis the changed parameters and respective values for calculating the new FF for the sensitivity analysis are shown. Parameter values listed under number 1 are shown in the graph where number 1 is indicated on the x-axis. The same applies for numbers 2 and 3.



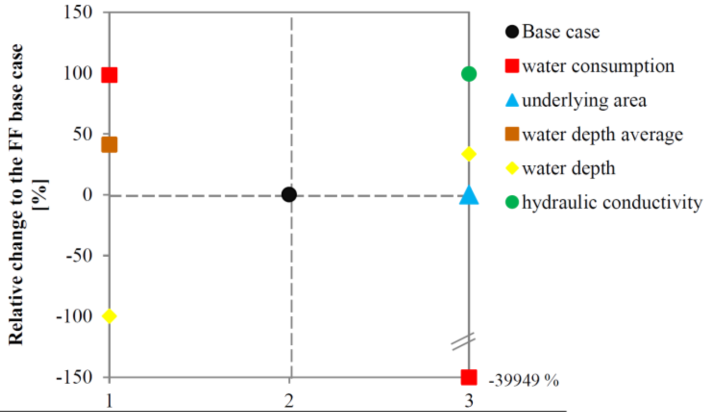
parameter	1	2 (base case)	3
Water consumption [m³/yr]	10	**1000**	1'000'000
underlying area [ha]	-	**Ramsar area**	waterbody area
water depth average [m]	3.8 m (if 1m was assumed)	**data from literature**	-
water depth [m]	-50%	**data from literature**	+50%
Hydraulic conductivity [m/s]	-	**calculated**	*100

Figure 9.4.19: Overview of the sensitivity analysis for the FF of GW-fed wetlands. The values shown here are the relative changes of the median global FF values for each parameter change in arid climates. In the table below the x-axis the changed parameters and respective values for calculating the new FF for the sensitivity analysis are shown. Parameter values listed under number 1 are shown in the graph where number 1 is indicated on the x-axis. The same applied for numbers 2 and 3.



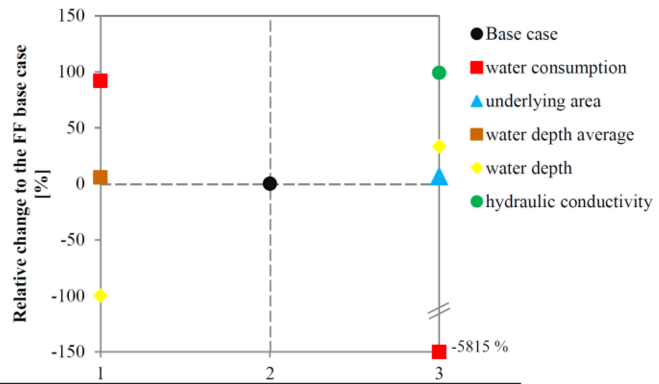
parameter	1	2 (base case)	3
Water consumption [m³/yr]	10	**1000**	1'000'000
underlying area [ha]	-	**Ramsar area**	waterbody area
water depth average [m]	3.8 m (if 1m was assumed)	**data from literature**	-
water depth [m]	-50%	**data from literature**	+50%
Hydraulic conductivity [m/s]	-	**calculated**	*100

Figure 9.4.209: Overview of the sensitivity analysis for the FF of GW-fed wetlands. The values shown here are the relative changes of the median global FF values for each parameter change in cold climates. In the table below the x-axis the changed parameters and respective values for calculating the new FF for the sensitivity analysis are shown. Parameter values listed under number 1 are shown in the graph where number 1 is indicated on the x-axis. The same applies for numbers 2 and 3.



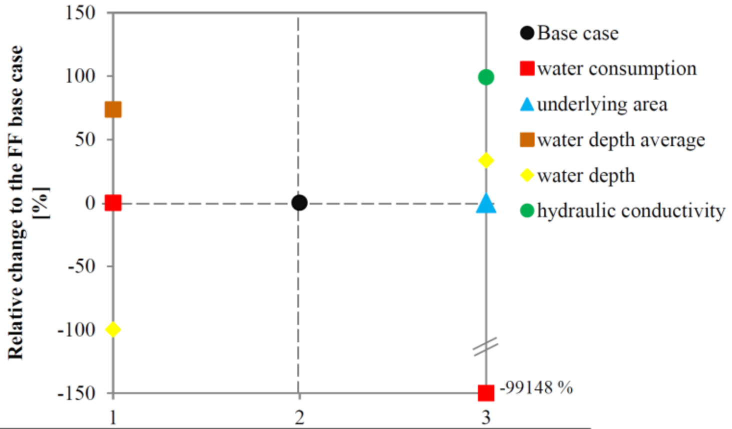
parameter	1	2 (base case)	3
Water consumption [m³/yr]	10	**1000**	1'000'000
underlying area [ha]	-	**Ramsar area**	waterbody area
water depth average [m]	3.8 m (if 1m was assumed)	**data from literature**	-
water depth [m]	-50%	**data from literature**	+50%
Hydraulic conductivity [m/s]	-	**calculated**	*100

Figure 9.4.21: Overview of the sensitivity analysis for the FF of GW-fed wetlands. The values shown here are the relative changes of the median global FF values for each parameter change in dry-subhumid climates. In the table below the x-axis the changed parameters and respective values for calculating the new FF for the sensitivity analysis are shown. Parameter values listed under number 1 are shown in the graph where number 1 is indicated on the x-axis. The same applies for numbers 2 and 3.



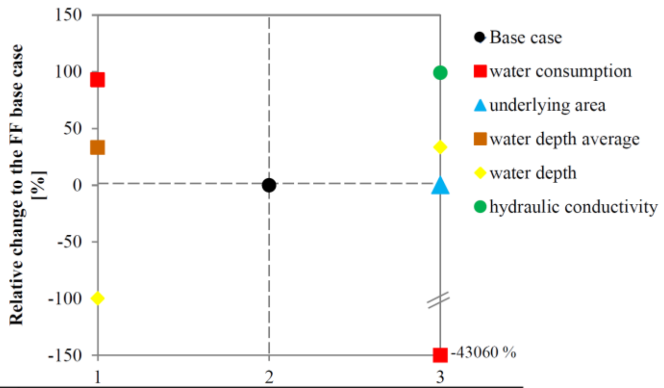
parameter	1	2 (base case)	3
Water consumption [m³/yr]	10	**1000**	1'000'000
underlying area [ha]	-	**Ramsar area**	waterbody area
water depth average [m]	3.8 m (if 1m was assumed)	**data from literature**	-
water depth [m]	-50%	**data from literature**	+50%
Hydraulic conductivity [m/s]	-	**calculated**	*100

Figure 9.4.22: Overview of the sensitivity analysis for the FF of GW-fed wetlands. The values shown here are the relative changes of the median global FF values for each parameter change in humid climates. In the table below the x-axis the changed parameters and respective values for calculating the new FF for the sensitivity analysis are shown. Parameter values listed under number 1 are shown in the graph where number 1 is indicated on the x-axis. The same applies for numbers 2 and 3.



parameter	1	2 (base case)	3
Water consumption [m³/yr]	10	**1000**	1'000'000
underlying area [ha]	-	**Ramsar area**	waterbody area
water depth average [m]	3.8 m (if 1m was assumed)	**data from literature**	-
water depth [m]	-50%	**data from literature**	+50%
Hydraulic conductivity [m/s]	-	**calculated**	*100

Figure 9.4.23: Overview of the sensitivity analysis for the FF of GW-fed wetlands. The values shown here are the relative changes of the median global FF values for each parameter change in hyper-arid climates. In the table below the x-axis the changed parameters and respective values for calculating the new FF for the sensitivity analysis are shown. Parameter values listed under number 1 are shown in the graph where number 1 is indicated on the x-axis. The same applies for numbers 2 and 3.



parameter	1	2 (base case)	3
Water consumption [m³/yr]	10	**1000**	1'000'000
underlying area [ha]	-	**Ramsar area**	waterbody area
water depth average [m]	3.8 m (if 1m was assumed)	**data from literature**	-
water depth [m]	-50%	**data from literature**	+50%
Hydraulic conductivity [m/s]	-	**calculated**	*100

Figure 9.4.104: Overview of the sensitivity analysis for the FF of GW-fed wetlands. The values shown here are the relative changes of the median global FF values for each parameter change in semiarid climates. In the table below the x-axis the changed parameters and respective values for calculating the new FF for the sensitivity analysis are shown. Parameter values listed under number 1 are shown in the graph where number 1 is indicated on the x-axis. The same applies for numbers 2 and 3.

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World Resources Institute. (2010b). Download Uganda GIS Data. Retrieved 8 May, 2012, from http://www.wri.org/publication/uganda-gis-data#wetlands.

Zhang, K.;Kimball, J.S.;Nemani, R.R.;Running, S. A continuous satellite-derived global record of land surface evapotranspiration from 1983 to 2006. *Water Resour. Res.* **2010,** 46: W09522, doi:10.1029/2009WR008800

9.5. Annex to Chapter 5

9.5.1. Inland Ramsar wetlands

The biological importance of the Ramsar sites, as indicated in the Ramsar Sites Information Service (2012) (RIS), is shown in Table 9.5. and Table 9.5.2. Note that each Ramsar site can be named several times within the biological importance category.

Table 9.5.1: Biological importance of the Ramsar sites. The number of sites is given for total sites, surface water-fed sites and groundwater-fed sites. The percentages are related to the respective total (i.e. 1184 for total, 1033 for surface water-fed and 151 for groundwater-fed wetlands).

importance for	Total		Surface water-fed		Groundwater-fed	
	Number of sites[-]	Percent age [%]	Number of sites[-]	Percent age [%]	Number of sites[-]	Percent age [%]
amphibians	300	25	261	25	39	26
birds	795	67	703	68	92	61
critical link in major food chain	106	9	97	9	9	6
crocodilians	77	7	76	7	1	1
fish	489	41	447	43	42	28
flora	809	68	720	70	89	59
invertebrates	331	28	289	28	42	28
mammals	579	49	513	50	66	44
marine turtles	11	1	11	1	0	0
reptiles	304	26	259	25	45	30
waterbirds	862	73	762	74	100	66

Table 9.5.5: Biological importance of the Ramsar sites per geographical region.

importance for	Africa	Asia	Central America	Europe	Near-East	North America	South America	Oceania
amphibians	42	28	31	154	1	15	21	8
birds	153	124	56	346	4	28	59	25
crocodilians	29	10	12	8	0	1	15	2
fish	104	77	45	190	3	24	28	18
flora	153	98	52	431	3	7	36	29
invertebrates	30	40	11	219	1	13	7	10
mammals	140	73	41	236	1	22	53	13
marine turtles	0	2	0	9	0	0	9	0
reptiles	75	36	40	97	2	16	25	13
waterbirds	140	124	44	459	3	24	42	26
critical link in major food chain	28	22	7	25	0	8	10	6

9.5.2. Overview of species and data sources

We have included different taxa for calculating effect factors of water consumption on biodiversity in wetlands. All species combined can act as a proxy for biodiversity. **Error! Reference source not found.** lists all considered taxa.

Table 9.5.3: Overview of taxa, data sources and total number of species. The SI section indicates in which section more information and the calculated maps for the respective taxon can be found.

Taxon	total Species number	data source	SI section	comments
Waterbirds	2119	Birdlife/Nature Serve(BirdLife International et al. 2011)	S3	habitat was according to BirdLife “wetland (inland)” or “artificial landscapes (aquatic)”.
Non-residential birds	1274	Birdlife/Nature Serve(BirdLife International et al. 2011)	S3	seasonal category "resident" excluded during calculation. Non-residential waterbirds excluded
amphibians	6021	IUCN(IUCN 2012a, b)	S5	all amphibians with map and TL data included
reptiles	268	IUCN(IUCN 2012a, c)	S4	only reptiles included whose habitat is "wetland (inland)" and contain TL and map data
water-dependent mammals	123	Global Mammal Assessment(Rondinini et al. 2011)	S6	only mammals included that are directly water-dependent (not only for drinking water)

9.5.3. Bird maps

For each bird species a shape file is available from BirdLife and NatureServe (2011), indicating the range of distribution. Additionally it gives information on Presence, Origin and Season (see

Table 9.5.4) that is equally valid for amphibians and reptiles.

Table 9.5.4: Codes for presence, origin and season of the dataset of BirdLife and NatureServe (2011), that are also valid for amphibians and reptiles.

	Presence	
Code	**Term**	**Explanation**
1	Extant	Occurs presently in area
2	Probably extant	Species presence thought probable
3	Possibly extant	Species may possibly occur
4	Possibly extinct	Species is most likely extirpated from area
5	Extinct	Formerly occurred in area, not recorded since 30 years, almost certainly extinct
6	Presence uncertain	Species formerly there, but now uncertain
	Origin	
Code	**Term**	**Explanation**
1	Native	Native inhabitant
2	Reintroduced	Formerly native range, reintroduced through human activities
3	Introduced	Through human activities to areas outside its natural range
4	Vagrant	Species recorded once or sporadically, not native to area
5	Origin uncertain	May be native, reintroduced or introduced
	Season	
Code	**Term**	**Explanation**
1	Resident	Present throughout the year
2	Breeding season	Occurs regularly during breeding season
3	Non-breeding season	Occurs regularly during non-breeding season, winter
4	Passage	Present during short periods during migration
5	Seasonal occurrence uncertain	Is present but unknown how long/which season

Resulting bird maps for the number of non-residential birds and waterbirds and the respective vulnerability scores are shown in Figure 9.5.111 to 9.5.7. For the definition of waterbirds, non-residential birds and the calculation of the vulnerability scores, see the main document.

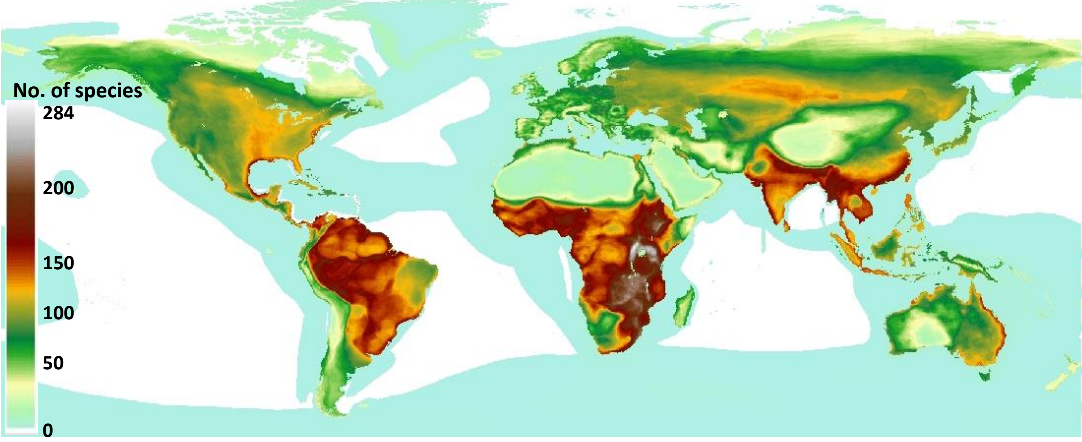


Figure 9.5.111: Bird richness map for the waterbird sample based on data from BirdLife and NatureServe (2011) Presence values are chosen from categories 1 to 3, values for season are at 1 to 5 .

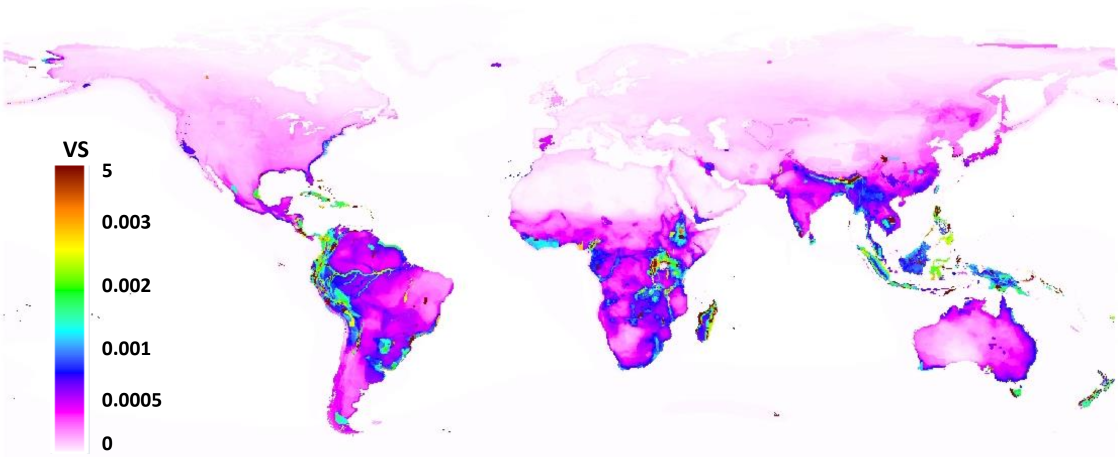


Figure 9.5.2: Bird vulnerability score (VS) map for waterbirds. Presence values are chosen from categories 1 to 3, values for season are at 1 to 5 .

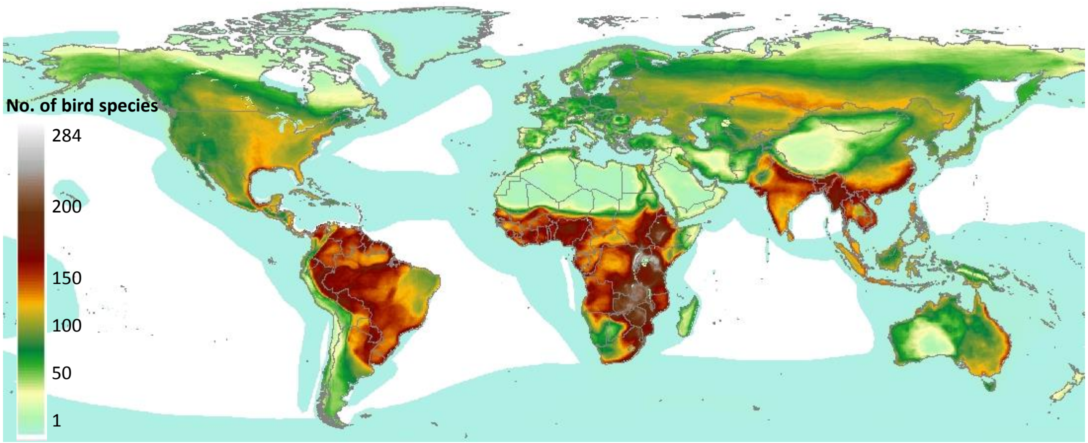


Figure 9.5.3: Bird richness map for the waterbird sample based on data from BirdLife and NatureServe (2011). Presence values are chosen from categories 1 to 4 (instead of 1 to 3), values for season remain at 1 to 5.

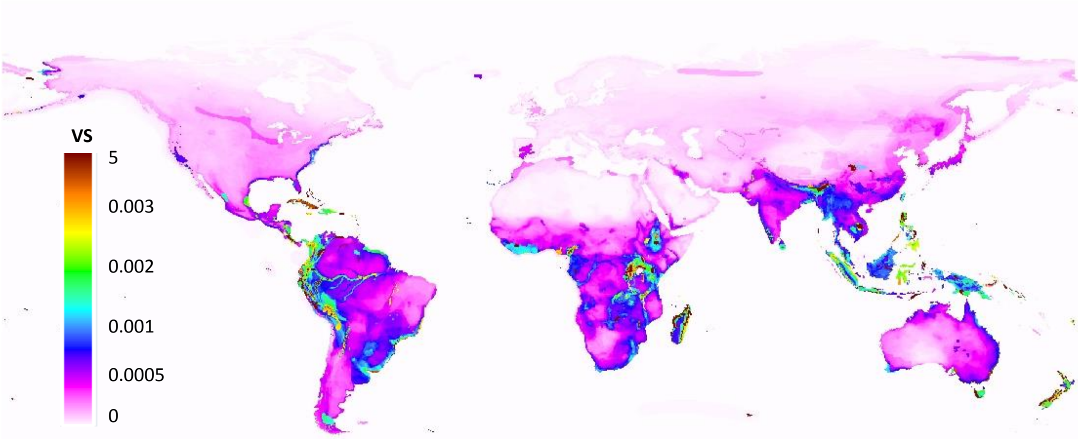


Figure 9.5.4: Bird vulnerability score (VS) map for waterbirds. Presence values are chosen from categories 1 to 4 (instead of 1 to 3), values for season remain at 1 to 5.

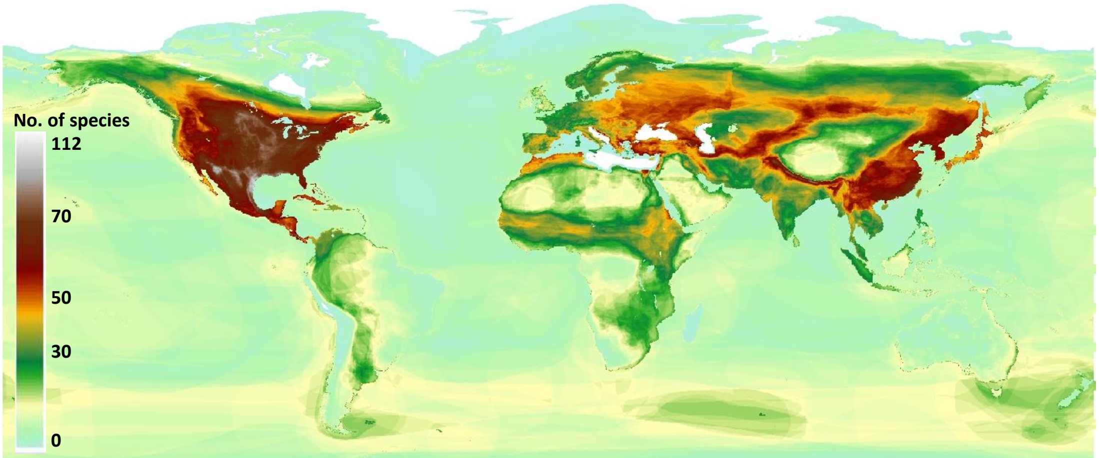


Figure 9.5.5: Bird richness map for the non-residential birds based on data from BirdLife and NatureServe (2011). Presence values are chosen from categories 1 to 3, values for season are at 2 to 5 .

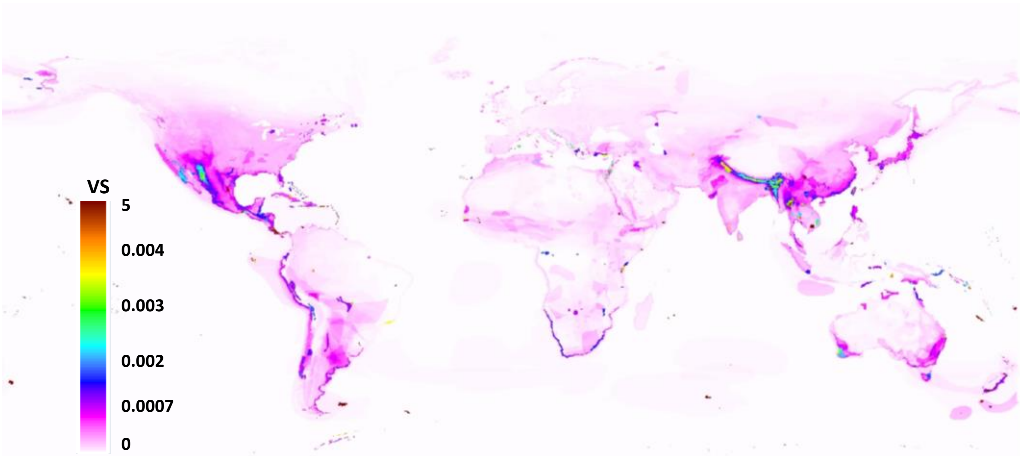


Figure 9.5.6: Bird vulnerability score (VS) map for non-residential birds. Presence values are chosen from categories 1 to 3, values for season are at 2 to 5 .

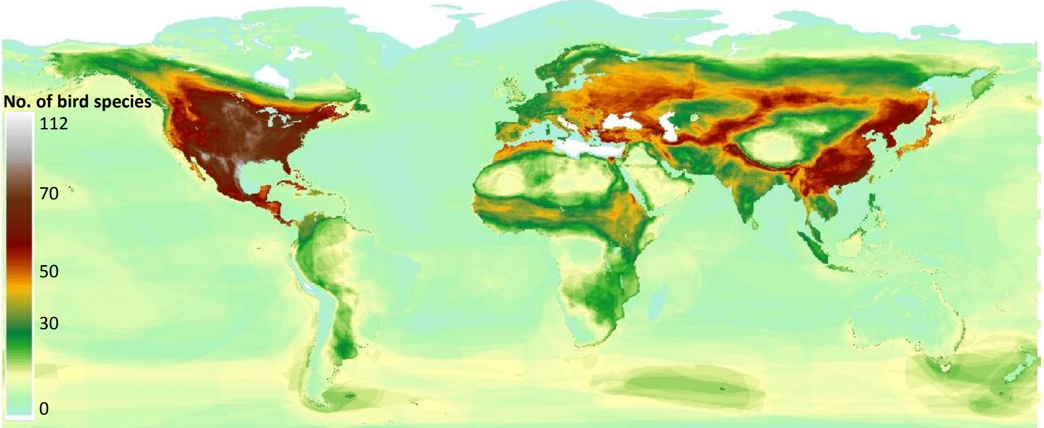


Figure 9.5.7: Bird richness map for the non-residential birds based on data from BirdLife and NatureServe (2011). Presence values are chosen from categories 1 to 4 (instead of 1 to 3), values for season remain at 2 to 5.

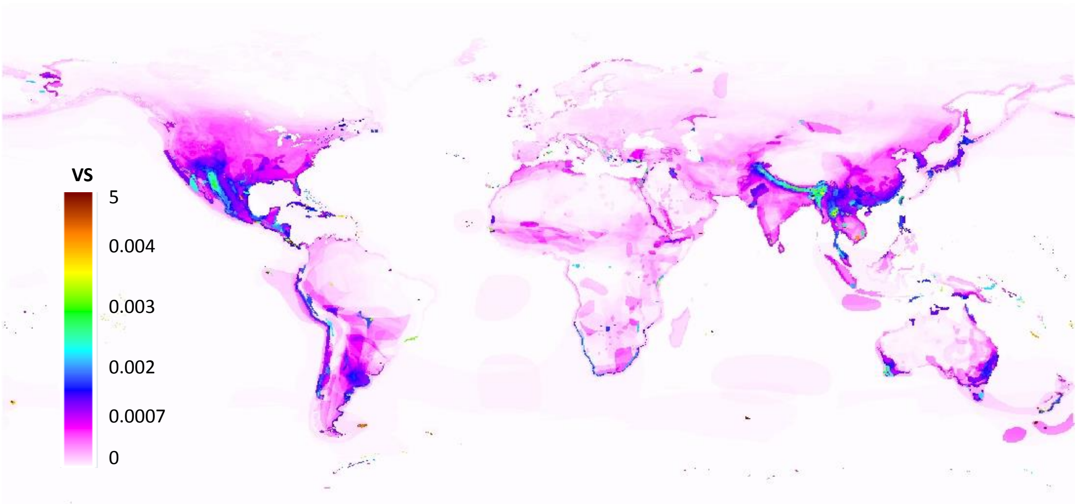


Figure 9.5.8: Bird vulnerability score (VS) map for non-residential birds. Presence values are chosen from categories 1 to 4 (instead of 1 to 3), values for season remain at 2 to 5 .

The largest difference between species richness calculated with presence values 1 to 3, or with presence values 1 to 4 for waterbirds, was 7 birds (The Bahamas), and 6 for non-residential birds (St.Helena). None of them were in areas where one of our 1184 Ramsar wetlands was located.

9.5.4. Reptile maps

Reptile maps were derived based on data from IUCN (2012). We only used those species which were classified as having “Wetland (inland)” as habitat. The categories for presence and seasonality are valid for reptiles as well. All seasonality values were used and for presence categories we changed between 1 to 3 and 1 to 4. The maps and corresponding vulnerability scores (VS) are shown in Figure 9.5.9ff.

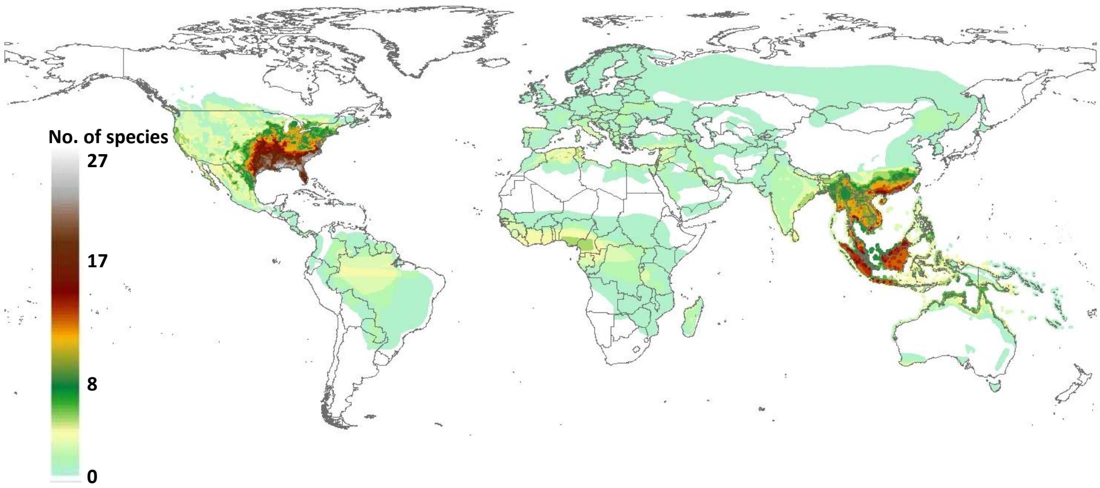


Figure 9.5.9: Species richness map of wetland reptiles based on data from IUCN (2012). Presence values are chosen from categories 1 to 3.

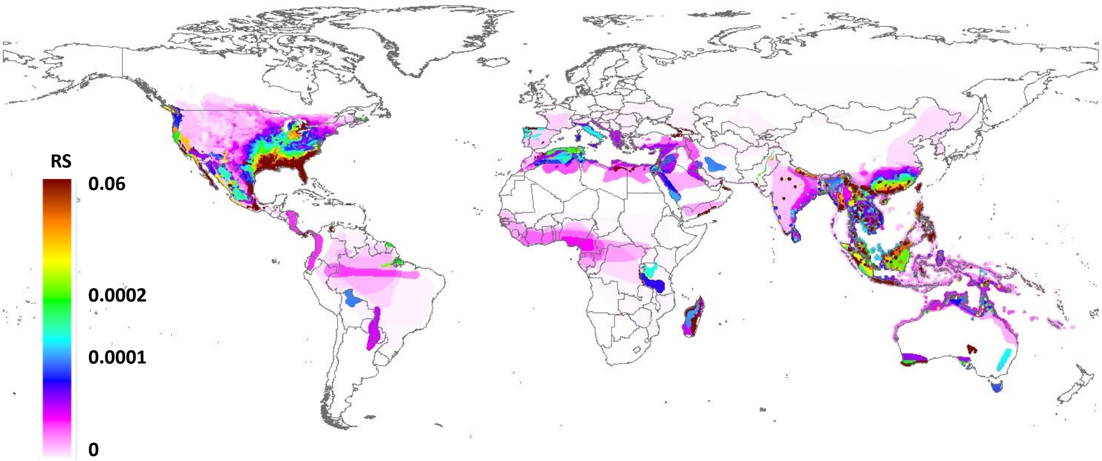


Figure 9.5.1012: Map of the vulnerability score (VS) of wetland reptiles based on data from IUCN (2012). Presence values are chosen from categories 1 to 3.

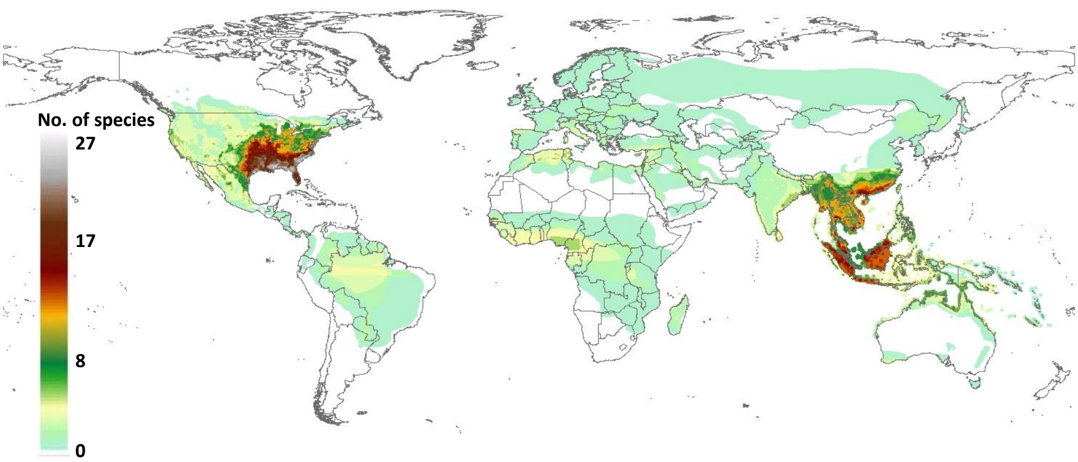


Figure 9.5.11: Species richness map of wetland reptiles based on data from IUCN (2012). Presence values are chosen from categories 1 to 4 (instead of 1 to 3).

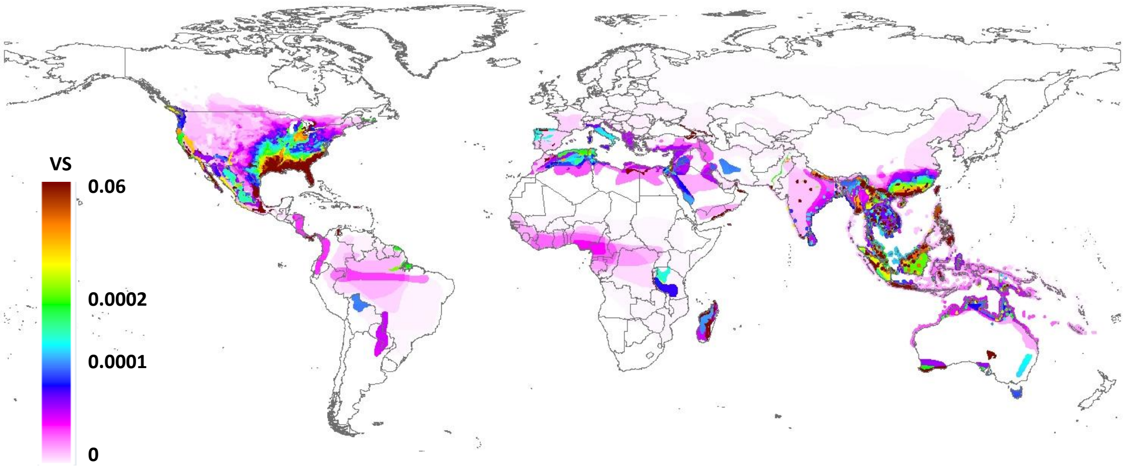


Figure 9.5.12: Map of the vulnerability score (VS) of wetland reptiles based on data from IUCN (2012). Presence values are chosen from categories 1 to 4 (instead of 1 to 3).

9.5.5. Amphibian maps

Amphibian maps were derived based on data from IUCN (2012). All amphibian species were used. The categories for presence and seasonality are valid for reptiles as well. All seasonality values were used, and for presence categories we altered between 1 to 3 and 1 to 4. The maps and corresponding vulnerability scores (VS) are shown in Figure 9.5. to Figure 9.5..

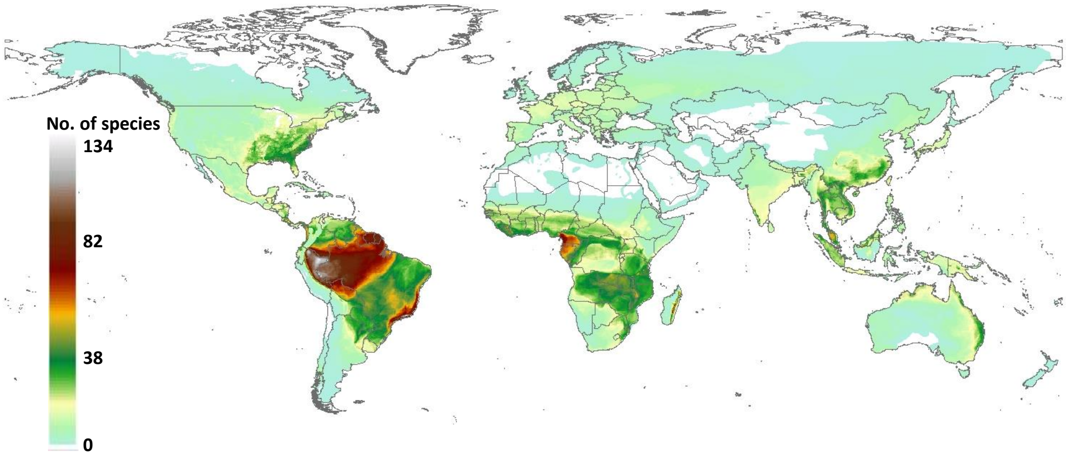


Figure 9.5.13: Species richness map of amphibians based on data from IUCN (2012). Presence values are chosen from categories 1 to 3.

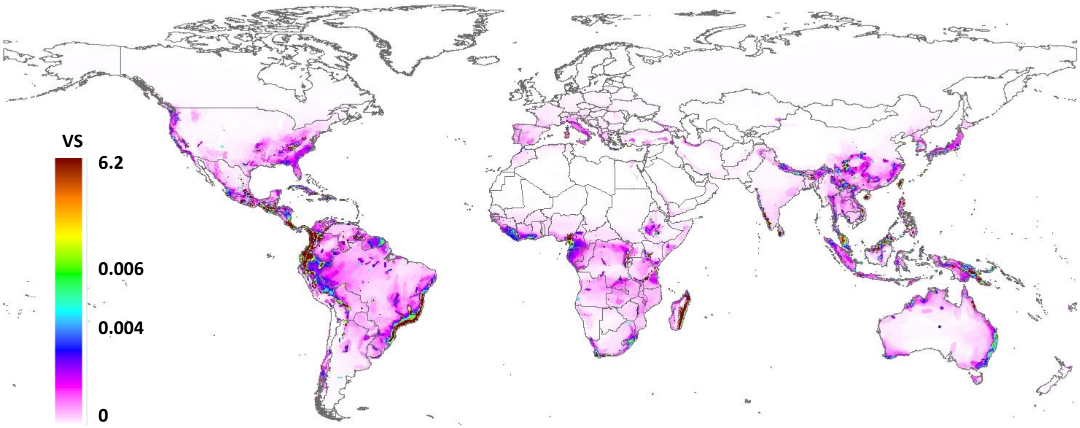


Figure 9.5.1413: Map of the vulnerability score (VS) of amphibians based on data from IUCN (2012). Presence values are chosen from categories 1 to 3.

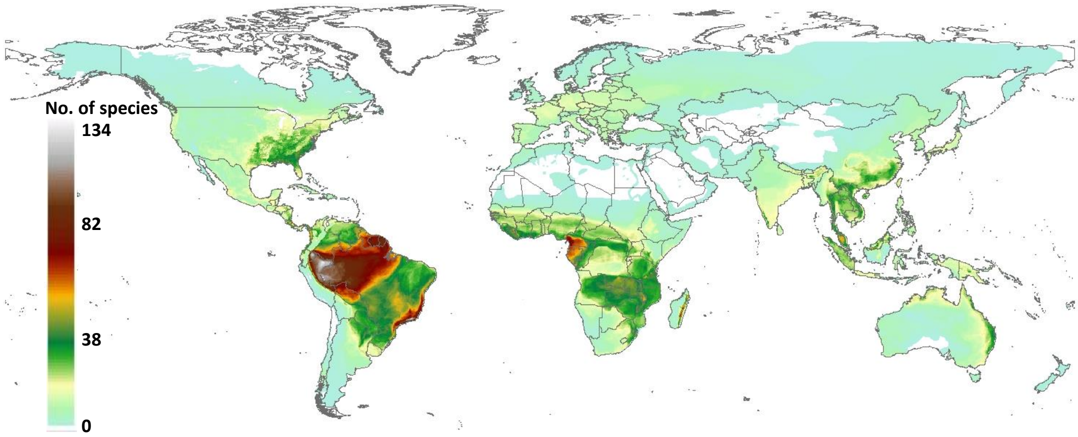


Figure 9.5.15: Species richness map of amphibians based on data from IUCN (2012). Presence values are chosen from categories 1 to 4 (instead of 1 to 3).

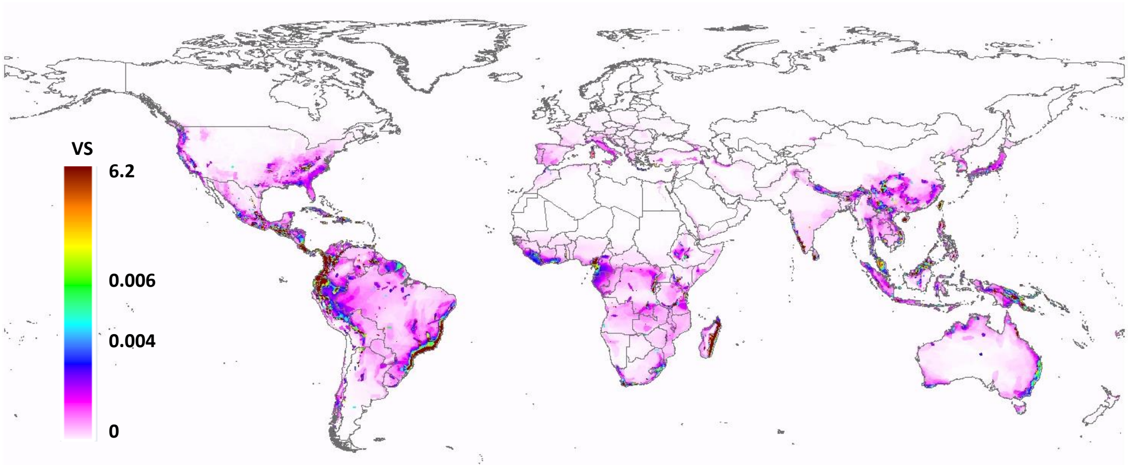


Figure 9.5.16: Map of the rarity score of amphibians based on data from IUCN (2012). Presence values are chosen from categories 1 to 4 (instead of 1 to 3).

9.5.6. Mammal maps

Maps for the number of water-dependent mammals and the respective vulnerability scores are shown in Figures 9.5.17ff.

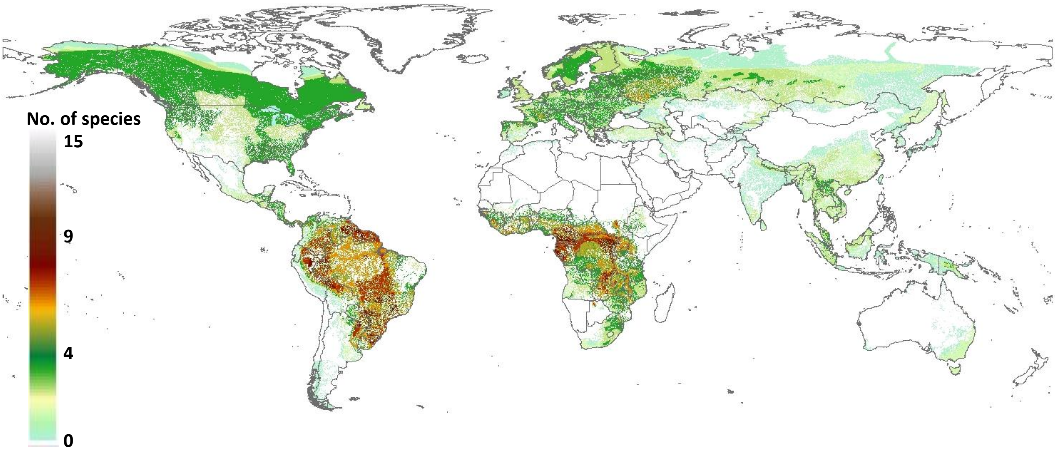


Figure 9.5.17: Species richness of water-dependent mammals based on the extent of occurrence of the mammals.

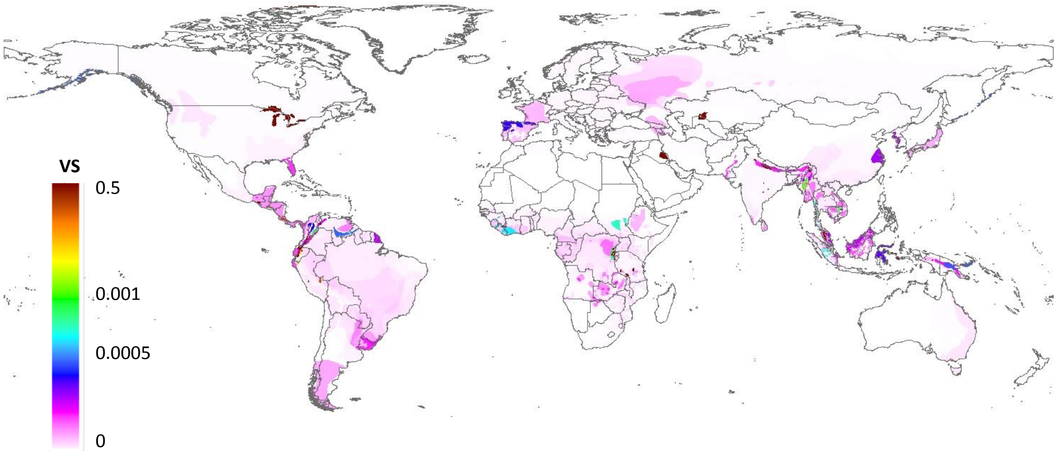


Figure 9.5.18: Vulnerability score (VS) of the water-dependent mammals, based on the extent of occurrence of the mammals.

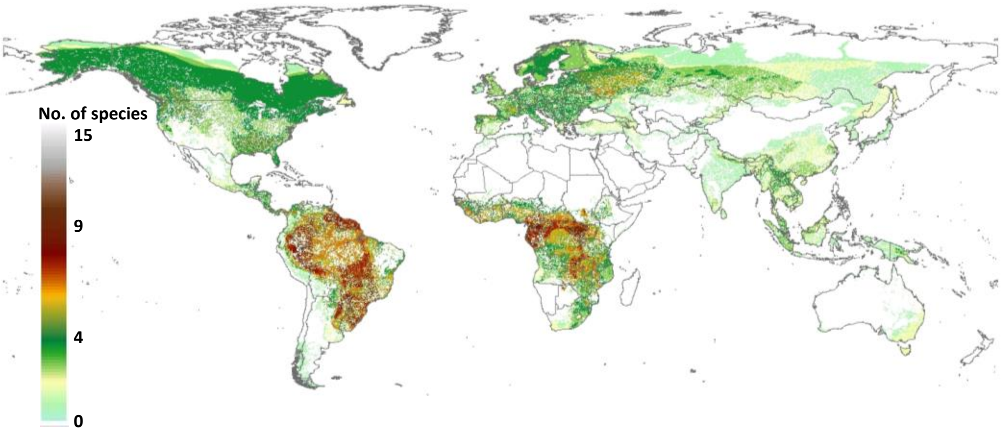


Figure 9.5.19: Species richness of water-dependent mammals based on the suitable habitat of the mammals (AOO).

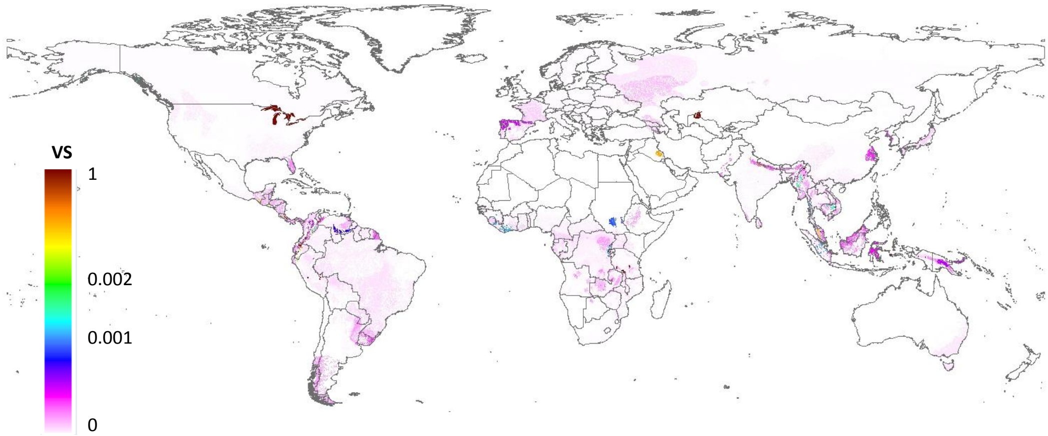


Figure 9.5.20: Vulnerability score (VS) of the water-dependent mammals, based on the suitable habitat of the mammals (AOO).

Difference between species richness map with EOO and AOO is small. The reason is that, as water-dependent mammals are limited to waterbodies, the difference between AOO and EOO is not large, and that AOO represents a nested subset of the EOO data. The geographical outermost boundary can only follow waterbody borders, as in the AOO.

9.5.7. CpA – Waterbody count per area

The waterbody count per area (CpA) data set is derived, as described in the main manuscript, based on the rivers of the world dataset (Lehner et al. 2008) and the global lakes and wetland database (Lehner et al. 2004) by counting how many points (i.e. waterbodies or river sections) fall into each sub-watershed (Lehner et al. 2008). Dividing the absolute number of points by the area of the sub-watershed, multiplying with an aridity index (precipitation (New et al. 2002) divided by potential evapotranspiration (Trabucco et al. 2009)) leads to a value set, that tells us how large the density in waterbodies is (taking into account a potential larger density of temporary pools by multiplying with the aridity index). The largest value is 142607. By dividing all values with this largest value, the CpA is scaled between 0 and 1.

The waterbody count per area (CpA) data set is derived, as described in the main manuscript, based on the rivers of the world dataset (Lehner et al. 2008) and the global lakes and wetland database (Lehner et al. 2004) by counting how many points (i.e. waterbodies or river sections) fall into each sub-watersheds area (N in Equation 9.5.9) (Lehner et al. 2008). Dividing the number of points by the area of the sub-watershed (A in Equation 9.5.9) and multiplying with an aridity index (precipitation P(New et al. 2002) divided by potential evapotranspiration PET(Trabucco et al. 2009)) leads to a value set that tells us for each pixel i how large the habitat loss risk in the network of waterbodies in each pixel is (taking into account a potential larger density of temporary pools by multiplying with the aridity index). By dividing all values with the maximum value, CpA is scaled between 0 and 1 and becomes unitless. If there is little water, the pixel had higher chances of becoming unsuitable as habitat, thus if the CpA is small, the habitat loss risk is large.

$$CpA_i = \frac{\frac{N_{per\ subwatershed,i}}{A_{subwatershed,i}} \cdot \frac{P_i}{PET_i}}{\max CpA_i}$$

Equation 9.5.9

Iceland, Norway, Finland, as well as parts of Sweden and Eastern Russia are not covered in the dataset for the watersheds (Lehner et al. 2008). The closest available CpA values were thus assumed to be valid in the administrative regions of these countries which were missing for calculating the CpA. As they have a high CpA, they are not relevant and this simplification is acceptable. For remote islands for which no P, PET, rivers and lakes data were available in global databases (e.g. Azores), a CpA of 1 was assumed. Since there was no indication about counts of waterbodies, we decided to set CpA to 1 in these cases, although this was not a conservative assumption and the damage is likely to be underestimated. However, this concerns only very few wetlands on individual islands and these small, data deficient wetlands would need a closer look in future. For islands close to the mainland, the closest mainland value was assigned to the island (e.g. Malta received its value from Sicily).

The resolution of the CpA data set (Figure 9.5) is 0.167°x0.167° since this was the resolution of the precipitation dataset (coarsest dataset).

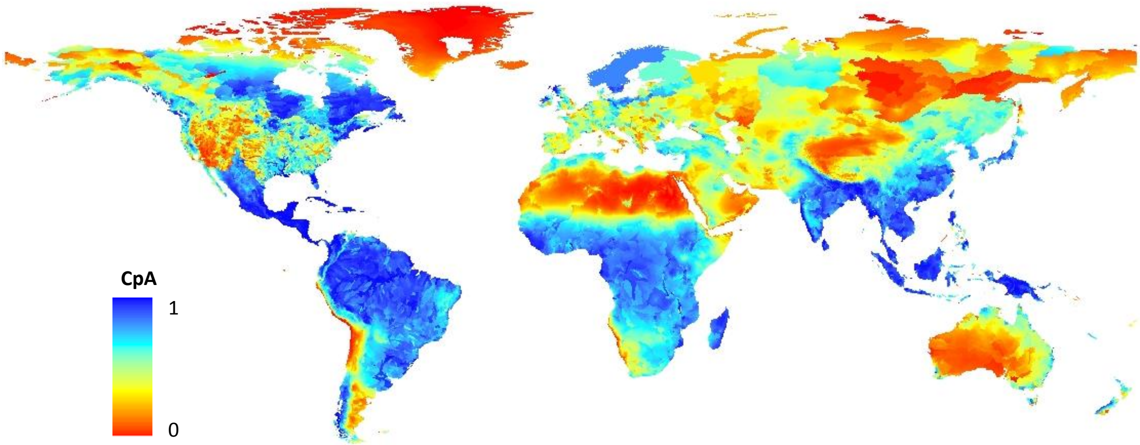


Figure 9.5.21: Waterbody count per area CpA scaled to one.

CpA=1 are areas with a large density of waterbodies, and as the value approaches zero, wetland habitats get more rare. The smaller the CpA, the rarer the waterbodies and the larger the threat of losing the habitat type “wetland/waterbody” in the region, when water is consumed.

9.5.8. z-values

The formula of the species-area relationship is shown in Equation 9.5.10. The species richness S can be predicted from a habitat area A, an exponent z indicating the slope of the species richness curve and a constant c. z is often determined for specific curves, but a common value of 0.25 is often applied (Rosenzweig 1995).

$$S = c \cdot A^z$$

Equation 9.5.10

For a known area change a new species richness S_{new} can thus be predicted based on the original area and species richness, as shown in Equation 9.5.11.

$$S_{new} = \left(\frac{A_{new}}{A_{original}}\right)^z \cdot S_{original}$$

Equation 9.5.11

The number of lost species S_{lost} from the original area is thus (Equation 9.5.12).

$$S_{lost} = S_{original} - S_{new} = S_{original} - \left(\frac{A_{new}}{A_{original}}\right)^z \cdot S_{original} = \left(1 - \left(\frac{A_{new}}{A_{original}}\right)^z\right) \cdot S_{original}$$

Equation 9.5.12

The range of the z values applied for the different taxa are shown in

Table 9.5.5. All values are taken from Drakare et al. (2006) For birds and mammals, we only used the z-values from nested studies because they best represent pure diversity change over different sampling area sizes, are best suited to the power model employed, and are more suited for extrapolation beyond the range of area sizes used to derive the z-value.(Dengler 2009) For reptiles and amphibians, we used z-values from independent (non-nested) studies from the same data source, since no values for nested studies were available. Values across multiple studies for a single taxon were averaged to derive taxon-specific (Drakare et al. 2006) values. All z-values are close to the commonly used z-value of 0.25(Brooks et al. 2002; Rosenzweig 1995; Thomas et al. 2004).

Table 9.5.5: Minimum, maximum and average slopes of the species-area relationship for the different taxa (Drakare et al. 2006).

Taxon	zmin	z max	zavera ge	No. of studies used
birds	0.15	0 .63	0.37	8 (nested)
mamma ls	-0.24	0 .93	0.34	4 (nested)
reptiles	0.08	0 .81	0.33	10 (independent)
amphibi ans	0.04	0 .36	0.20	18 (independent)

9.5.9. Characterization factors – determining individual catchments

All characterization factors (CFs) are applicable on a larger scale than just the wetland itself. The reasoning is explained below for surface water-fed and groundwater-fed wetlands separately.

Surface water-fed wetlands

A surface water-fed wetland is fed by inflowing water from the catchment upstream of the wetland. If water is consumed anywhere in the area upstream of the wetland, inflow will be reduced and the wetland will be damaged. Therefore, the CF for this wetland is applicable in the whole watershed of the wetland (e.g. blue watershed in **Error! Reference source not found.**). A second wetland, which is for instance situated upstream of the first one, may receive water from partly the same sources. But other rivers, for example, may be completely irrelevant for the second wetland, because they do not flow into this specific wetland. Thus, the CF for the second wetland is applicable in another area, which is the individual catchment of the second wetland (e.g. orange area in **Error! Reference source not found.**). However, water which is consumed in the range area does not reach both wetlands and therefore both of them are damaged and the CFs of both are applicable. That means that the CFs are summed in these areas. This procedure is repeated for all 1033 surface water-fed wetlands and leads to the global maps.

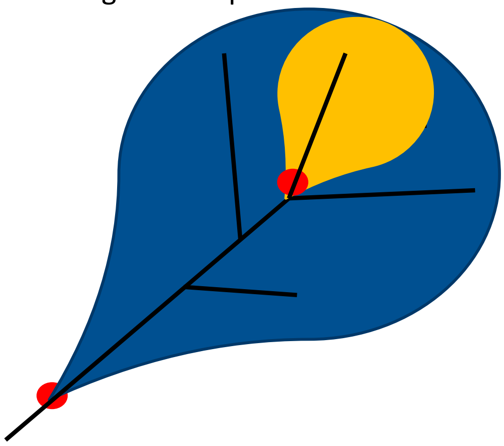


Figure 9.5.22: Schematic representation of two surface water-fed wetlands (red dots) in their respective watersheds (blue and orange). The river network is shown in black.

<u>Groundwater-fed wetlands</u>

Here the relevant area is calculated according to the hydrogeological condition surrounding the wetland (not upstream-downstream as in the surface water-fed wetland cases). The Area of Relevance (AoR) has been used for the calculation of the FF before (for details see Verones et al. (2013)). In principle, we determined circles around the wetlands from which water is being drawn to the wetland (imagining the wetland to act like a pump). The decrease in groundwater level due to pumping anywhere in this Area of Relevance influences the infiltrating amount into the wetland. Thus, any pumping within this area leads to damage and thus the CF is applicable in the whole AoR. If there is a second groundwater-fed wetland and their AoRs overlap, the CFs are summed, because it was assumed that pumping in that region will damage both wetlands.

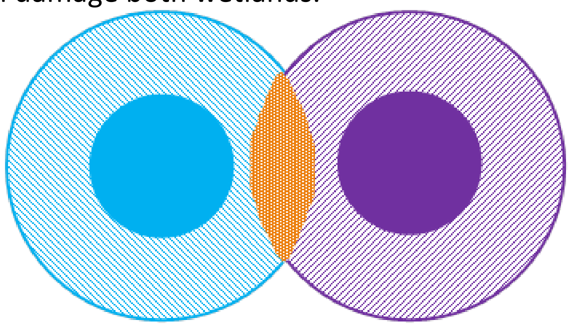


Figure 9.5.23: Schematic representation of two Areas of Relevance (AoR, hatched circles) around two groundwater-fed wetlands (blue and violet circle). The orange part is the area where the AoRs overlap and where CFs are thus summed.

9.5.1. EF and CFs

In Figure 9.5. an overview of all the necessary parameters for the calculation of the EF is shown. As an example all the values of the parameters for lake Naivasha and lake Elmenteita (both in Kenya) are given in Table 9.5.. These two wetlands are used in the application example and have individual catchments. For the location Bleiswijk no example is provided since the used factor at the location consists of a several overlaying catchments of wetlands within the Rhine watershed.

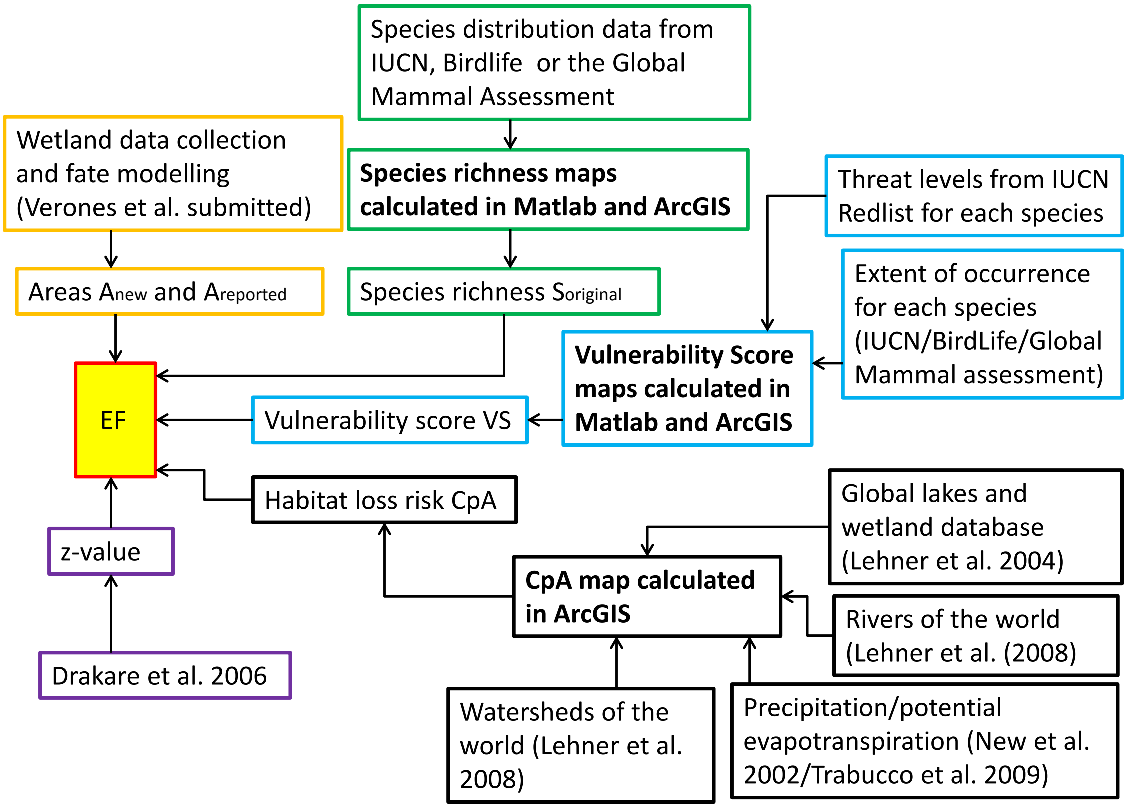


Figure 9.5.24: Overview of the parameters and their origin that are required for calculating the EFs of a wetland.

Table 9.5.6: Examples for all required parameters for calculating EFs for lake Naivasha (SW-fed, Kenya) and lake Elmenteita (GW-fed, Kenya).

	Lake Naivasha	**Lake Elmenteita**	**References/comments**
Area reported [ha]	30000	10880	RIS(Ramsar Sites Information Service 2012)
Area new [ha]	29999	10879	Calculated, see Verones et al.(2013)
CpA [-]	0.064	0.027	Habitat loss risk index (SI, S7)
z-value waterbirds[-]	0.37	0.37	Based on Drakare et al. (2006)
z-value non-residential birds[-]	0.37	0.37	Based on Drakare et al. (2006)
z-value water-dependent mammals[-]	0.34	0.34	Based on Drakare et al. (2006)
z-value reptiles[-]	0.33	0.33	Based on Drakare et al. (2006)
z-value amphibians[-]	0.20	0.20	Based on Drakare et al. (2006)
Species richness waterbirds [no.of species]	250	245	Maps, based on Birdlife/NatureServe data(2011) (SI, S3)
Species richness non-residential birds [no.of species]	32	37	Maps, based on Birdlife/NatureServe data(2011) (SI, S3)
Species richness water-dependent mammals [no.of species]	6	6	Maps, data from global mammal assessment (SI, S6)
Species richness reptiles [no.of species]	1	1	Maps, based on IUCN data(IUCN 2012c) (SI, S4)
Species richness amphibians [no.of species]	29	24	Maps, based on IUCN data(IUCN 2012b) (SI, S5)
Vulnerability score waterbirds [-]	8.5E-06	8.5E-06	Maps, based on Birdlife/NatureServe data(BirdLife International et al. 2011) and IUCN data(IUCN 2012a) (SI, S3)
Vulnerability score non-residential birds [-]	4.1E-06	3.8E-06	Maps, based on Birdlife/NatureServe data(BirdLife International et al. 2011) and IUCN data(IUCN 2012a)

			(SI, S3)
Vulnerability score water-dependent mammals [-]	8.4E-06	6.9E-06	Maps, based on IUCN data(IUCN 2012a) (SI, S6)
Vulnerability score reptiles [-]	2.1E-06	2.1E-06	Maps, based on IUCN data(IUCN 2012a) (SI, S4)
Vulnerability score amphibians [-]	7.8E-05	1.1E-04	Maps, based on IUCN data(IUCN 2012a) (SI, S5)
EF waterbirds [species-eq/m2]	4.74E-11	2.66E-10	Calculated
EF non-residential birds[species-eq/m2]	2.52E-12	1.78E-11	Calculated
EF water-dependent mammals [species-eq/m2]	9.52E-13	4.53E-12	Calculated
EF reptiles [species-eq/m2]	3.58E-14	2.35E-13	Calculated
EF amphibians [species-eq/m2]	2.37E-11	1.90E-10	Calculated

Bird species are present in all wetlands. This does not apply to the other taxa. In Table 9.5. the number of wetlands is shown that do not harbour mammals, reptiles or amphibians.

Table 9.5.7: Number of wetlands that do not contain a certain taxa in absolute numbers and as percentage of wetland type. SW stands for surface water-fed wetlands and GW for groundwater-fed wetlands.

	Number of wetlands zero		Percentage of wetlands zero	
Taxa	**SW [-]**	**GW [-]**	**SW [%]**	**GW [%]**
Waterbird	0	0	0	0
Non-residential birds	0	0	0	0
water-dependent mammals	121	28	12	19
reptiles	168	25	16	17
amphibians	43	6	4	4

Effect and characterization factors calculated on the basis of the waterbody area are presented in Table 9.5..

Table 9.5.8: Effect factors [species-eq/m^2] and characterization factors [species-eq·yr/m^3] for waterbirds, non-residential birds, water-dependent mammals, wetland reptiles, amphibians and all combined based on the area of the waterbodies within the Ramsar sites. Factors are presented summarized for surface-fed wetlands with surface water consumption (SW) and groundwater-fed wetlands with groundwater consumption (GW). Presence categories are 1 to 3 (birds, reptiles, amphibians). CV is the coefficient of variation.

	EF [species-eq/m^2]		CF [species-eq·yr/m^3]	
	SW	GW	SW	GW
waterbirds min	1.7E-13	7.1E-13	1.7E-15	6.8E-15
waterbirds max	2.4E-05	2.0E-06	1.1E-05	1.3E-06
waterbirds mean	5.8E-08	3.0E-08	1.4E-08	1.3E-08
CV	16	6	25	8
non-residents min	1.9E-15	2.1E-13	5.3E-17	1.2E-14
non-residents max	2.0E-05	1.8E-06	7.4E-06	3.7E-06
non-residents mean	5.6E-08	3.2E-08	1.0E-08	2.9E-08
CV	14	5	23	10
water-dep. mammals min	1.6E-15	1.1E-14	3.4E-17	1.7E-16
water-dep. mammals max	2.0E-06	4.7E-07	3.8E-08	8.4E-08
water-dep. mammals mean	3.9E-09	5.0E-09	3.4E-10	7.9E-10
CV	17	8	7	9
wetland reptiles min	1.29E-16	1.38E-14	2.01E-17	7.91E-17
wetland reptiles max	2.58E-05	1.13E-05	1.72E-05	1.11E-06
wetland reptiles mean	3.53E-08	8.19E-08	1.70E-08	1.54E-08

CV	24	11	32	8
amphibians min	5.02E-16	8.24E-15	5.62E-16	6.74E-16
amphibians max	6.47E-05	9.79E-07	4.56E-05	1.88E-06
amphibians mean	1.29E-07	3.14E-08	6.01E-08	1.50E-08
CV	17	4	24	10
combined taxa min	2.5E-13	1.1E-12	2.3E-15	2.7E-14
combined taxa max	8.1E-05	1.1E-05	5.7E-05	4.7E-06
combined taxa mean	2.8E-07	1.8E-07	1.0E-07	7.4E-08
CV	13	6	21	7

Characterization factors (CFs) calculated with the waterbody areas within the Ramsar area are shown in Figures 9.5.25ff.

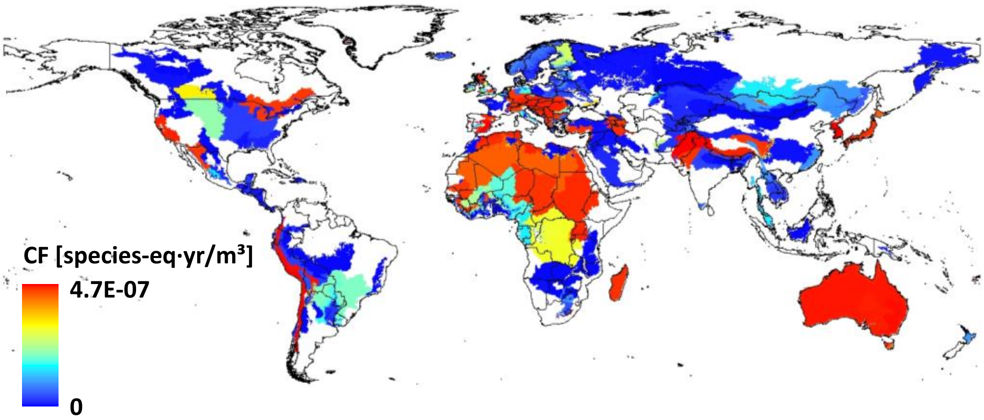


Figure 9.5.25: CFs for surface water-fed wetlands with surface water consumption for waterbirds (presence 1 to 3) based on waterbody areas within the Ramsar areas. Base map with country boundaries adapted from ref.(ESRI 2009)

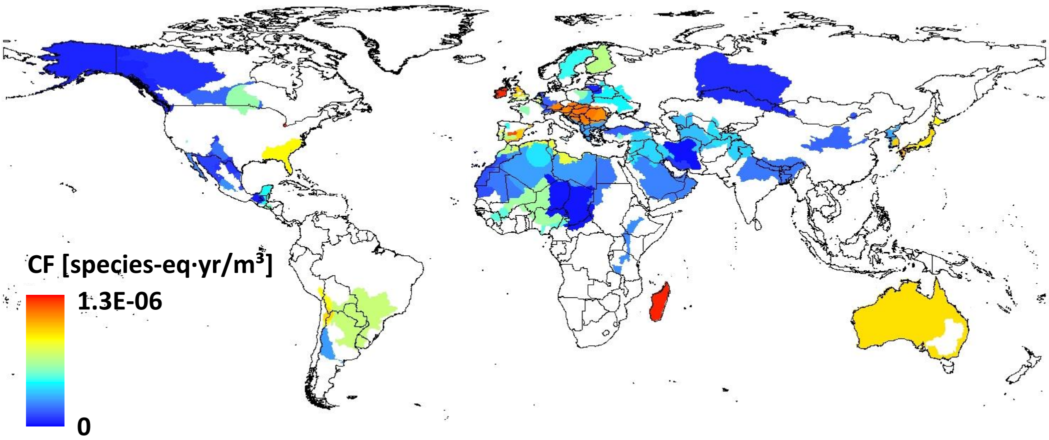


Figure 9.5.26: CFs for groundwater-fed wetlands with groundwater consumption for waterbirds (presence 1 to 3) based on waterbody areas within the Ramsar areas. Base map with country boundaries adapted from ref.(ESRI 2009)

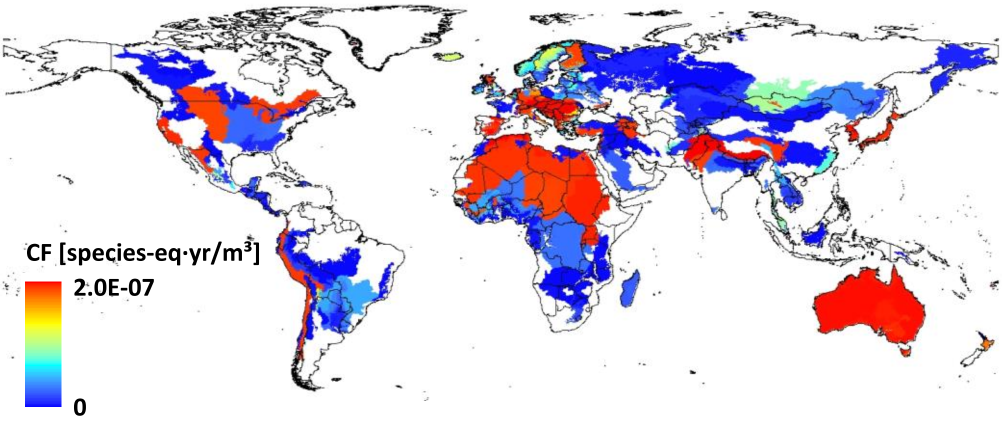


Figure 9.5.27: CFs for surface water-fed wetlands with surface water consumption for non-residential birds (presence 1 to 3) based on waterbody areas within the Ramsar areas. Base map with country boundaries adapted from ref.(ESRI 2009)

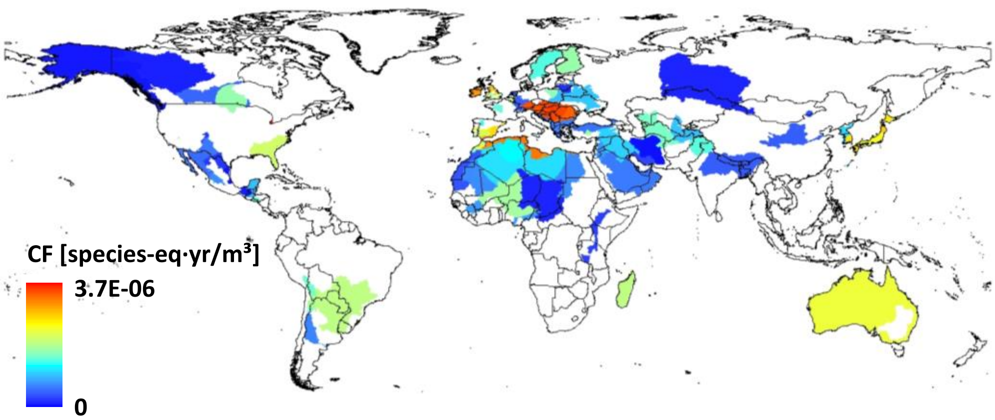


Figure 9.5.28: CFs for groundwater-fed wetlands with groundwater consumption for non-residential birds (presence 1 to 3) based on waterbody areas within the Ramsar areas. Base map with country boundaries adapted from ref.(ESRI 2009)

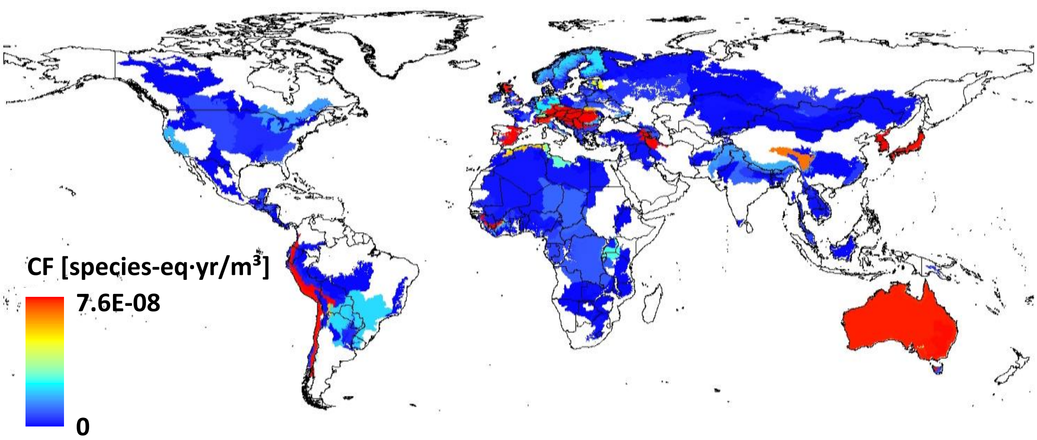


Figure 9.5.29: CFs for surface water-fed wetlands with surface water consumption for water-dependent mammals based on waterbody areas within the Ramsar areas. Base map with country boundaries adapted from ref.(ESRI 2009)

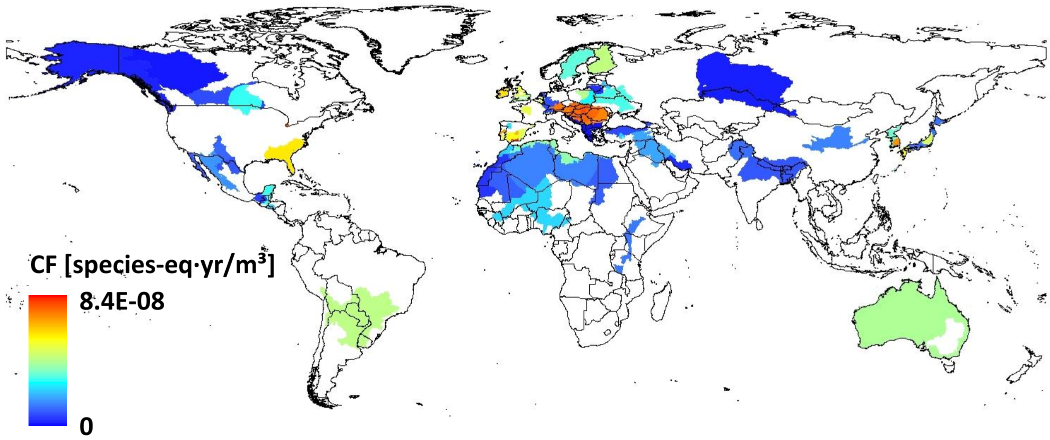


Figure 9.5.14: CFs for groundwater-fed wetlands with groundwater consumption for water-dependent mammals based on waterbody areas within the Ramsar areas. Base map with country boundaries adapted from ref.(ESRI 2009)

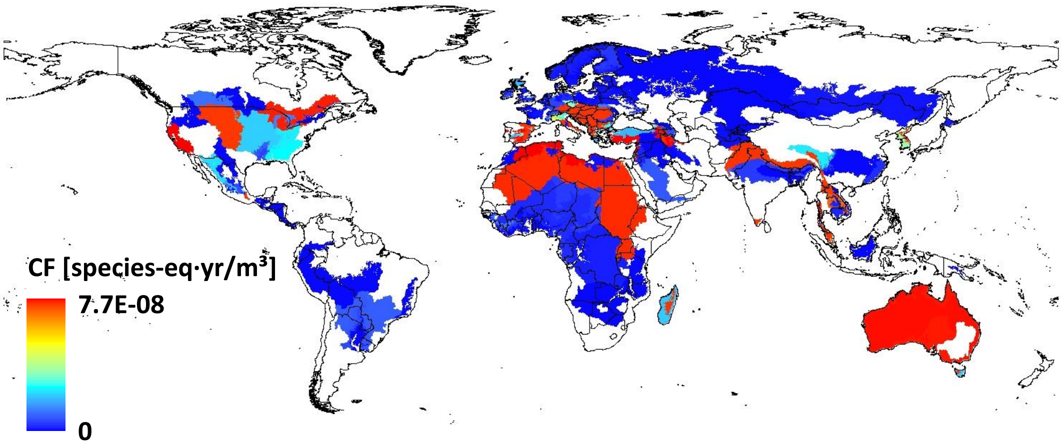


Figure 9.5.31: CFs for surface water-fed wetlands with surface water consumption for wetland reptiles based on waterbody areas within the Ramsar areas. Base map with country boundaries adapted from ref.(ESRI 2009)

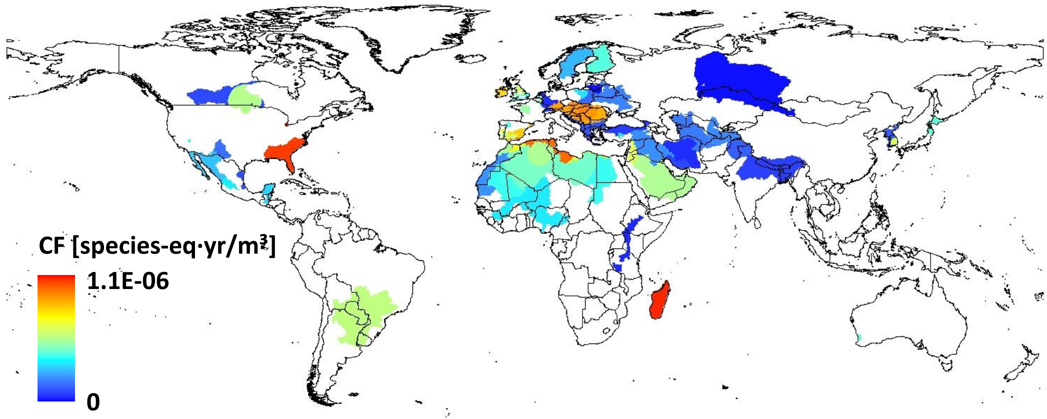


Figure 9.5.32: CFs for groundwater-fed wetlands with groundwater consumption for wetland reptiles based on waterbody areas within the Ramsar areas. Base map with country boundaries adapted from ref.(ESRI 2009)

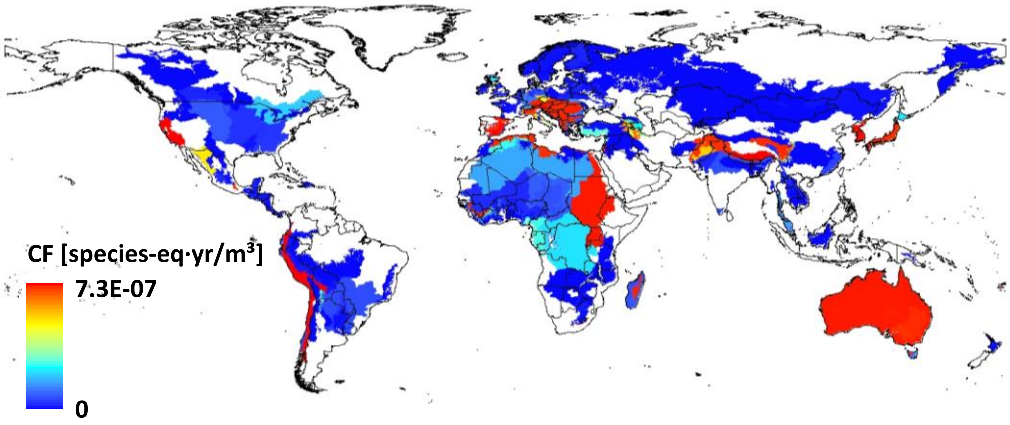


Figure 9.5.33: CFs for surface water-fed wetlands with surface water consumption for amphibians based on waterbody areas within the Ramsar areas. Base map with country boundaries adapted from ref.(ESRI 2009)

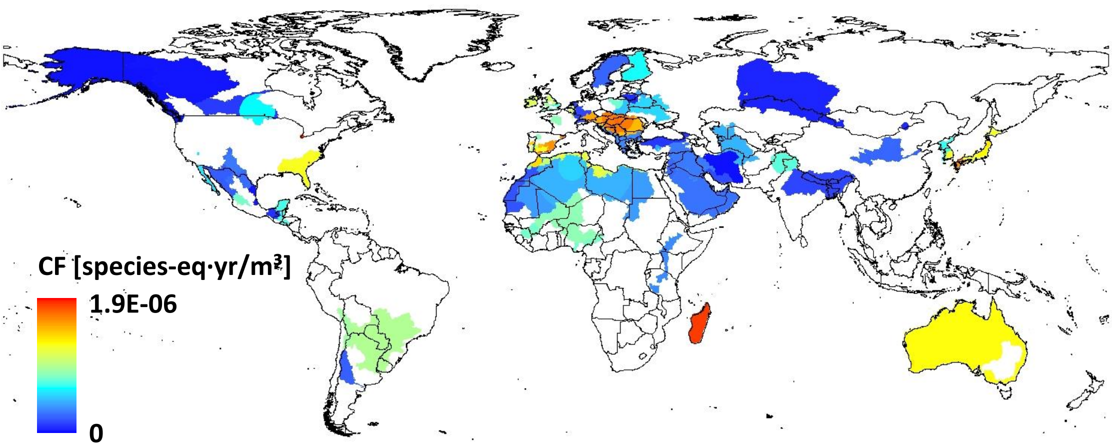


Figure 9.5.34: CFs for groundwater-fed wetlands with groundwater consumption for amphibians based on waterbody areas within the Ramsar areas. Base map with country boundaries adapted from ref.(ESRI 2009)

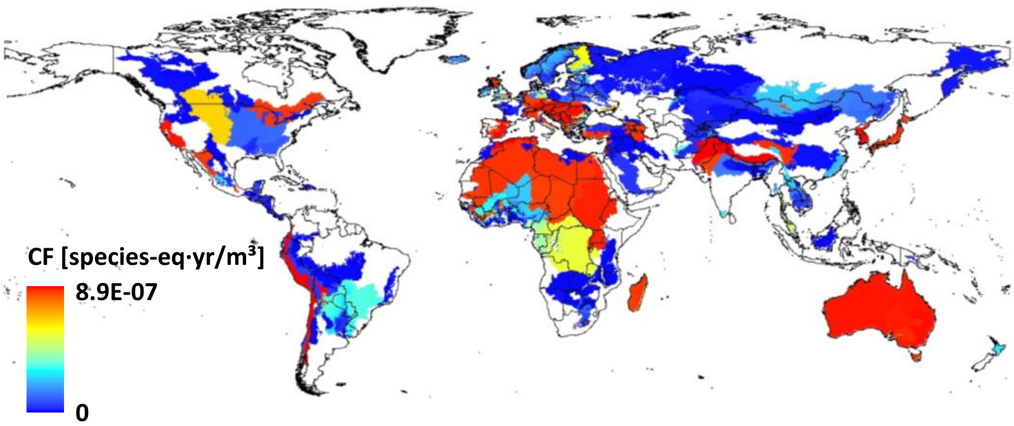


Figure 9.5.35: CFs for surface water-fed wetlands with surface water consumption for all taxa combined, based on waterbody areas within the Ramsar areas. Base map with country boundaries adapted from ref.(ESRI 2009)

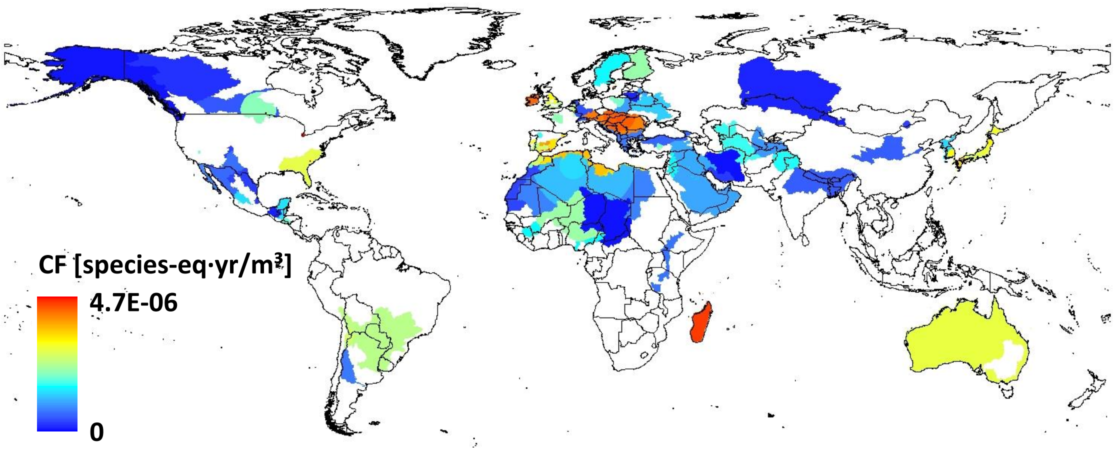


Figure 9.5.3615: CFs for groundwater-fed wetlands with groundwater consumption for all taxa combined, based on waterbody areas within the Ramsar areas. Base map with country boundaries adapted from ref.(ESRI 2009)

9.5.1. Sensitivities and correlations of EF and CF

The correlation between CFs of different taxa and correlations between different parts of the EF and the EF itself are shown in Tables 9.5.9f.

Table 9.5.9: Spearman's rank correlation coefficient ρ between CFs of different taxa and for both surface water- fed (SW) and groundwater-fed (GW) wetlands. The color code indicates whether there is medium (ρ > 0.3, green with green letters) or high correlation (ρ > 0.5, blue with yellow letters).

Water source	Correlation	Spearman ρ [-]
SW	CF waterbirds – CF non-residential birds	0.921
	CF waterbirds – CF mammals	0.653
	CF waterbirds -CF reptiles	0.535
	CF waterbirds – CF amphibians	0.836
	CF non-residential birds – CF mammals	0.606
	CF non-residential birds – CF reptiles	0.569
	CF non-residential birds – CF amphibians	0.801
	CF mammals –CF reptiles	0.490
	CF mammals –CF amphibians	0.689
	CF reptiles –CF amphibians	0.600
GW	CF waterbirds – CF non-residential birds	0.967
	CF waterbirds – CF mammals	0.741
	CF waterbirds – CF reptiles	0.734
	CF waterbirds – CF amphibians	0.907
	CF non-residential birds – CF mammals	0.742
	CF non-residential birds – CF reptiles	0.757
	CF non-residential birds – CF amphibians	0.923
	CF mammals –CF reptiles	0.647
	CF mammals –CF amphibians	0.768
	CF reptiles –CF amphibians	0.711

Table 9.5.10: Correlations between components of the effect factor and effect factor itself (EF), as well as the fate factor (FF). Correlations are calculated for each taxa separately, except for the correlation between CpA and FF, which is the same for all taxa and is thus only calculated for SW-fed and GW-fed wetlands. VS is the vulnerability score, S the species number and CpA is the habitat loss risk index.

Water source	Correlation	Spearman ρ [-]
SW	CpA-FF	-0.263
	CpA-VS waterbirds	0.229
	CpA-VS non-residents	0.175
	CpA-VS mammals	0.037
	CpA-VS reptiles	-0.016
	CpA-VS amphibians	0.063
	S-VS waterbirds	0.429
	S-VS non-residents	0.030
	S-VS mammals	0.311
	S-VS reptiles	0.551
	S-VS amphibians	0.367
	S-CpA waterbirds	0.259
	S-CpA non-residents	-0.054
	S-CpA mammals	0.092
	S-CpA reptiles	0.082
	S-CpA amphibians	0.238
GW	CpA-FF	-0.005
	CpA-VS waterbirds	0.305
	CpA-VS non-residents	0.379
	CpA-VS mammals	0.241
	CpA-VS reptiles	0.025
	CpA-VS amphibians	0.250
	S-VS waterbirds	0.275
	S-VS non-residents	0.199
	S-VS mammals	0.418
	S-VS reptiles	0.791
	S-VS amphibians	0.355
	S-CpA waterbirds	0.562
	S-CpA non-residents	0.404
	S-CpA mammals	0.062
	S-CpA reptiles	0.133
	S-CpA amphibians	0.439
SW	S-EF waterbirds	-0.175
	S-EF Nonresidents	0.161
	S-EF mammals	0.193
	S-EF reptiles	0.440
	S-EF amphibians	0.176
	VS-EF waterbirds	0.136
	VS-EF non-residents	0.194
	VS-EF mammals	0.443
	VS-EF reptiles	0.699
	VS-EF amphibians	0.627
	CpA-EF waterbirds	-0.215
	CpA-EF non-residents	-0.268
	CpA-EF mammals	-0.238
	CpA-EF reptiles	-0.183
	CpA-EF amphibians	-0.189
GW	S-EF waterbirds	-0.016
	S-EF non-residents	0.099
	S-EF mammals	0.521
	S-EF reptiles	0.577
	S-EF amphibians	0.423
	VS-EF waterbirds	0.046
	VS-EF non-residents	0.206
	VS-EF mammals	0.623
	VS-EF reptiles	0.703
	VS-EF amphibians	0.487
	CpA-EF waterbirds	-0.073

CpA-EF non-residents	-0.026
CpA-EF mammals	0.009
CpA-EF reptiles	-0.129
CpA-EF amphibians	0.079

We calculated the histogram of species richness, a dominant factor for the EF, and EFs themselves for all taxa. They are shown for SW-fed and GW-fed wetlands in Figure 9.5.37. In Figure 9.5.37**Error! Reference source not found.**A, mammals are with 912 wetlands highest in the bin category 10-20 species. Also, reptiles and amphibians are mostly represented by 10-20 species (840 wetlands and 539 wetlands, respectively). The largest number of wetlands for non-residential bird species is 235 in the species richness category 40-50 species. Only waterbirds show their maximum in an even higher category (80-90 species in 128 wetlands). The distribution of species richness is widest for waterbirds. In GW-fed wetlands (Figure 9.5.37B), waterbird species are present in 29 wetlands with between 70 and 80 species. Non-residential birds have their maximum with 50-60 species in 35 wetlands, and this is again a bit lower than for SW-fed wetlands. Mammals, reptiles and amphibians are all mostly present with 10-20 species (123, 123 and 83 wetlands, respectively). For the EFs, the most frequent bins for the EFs are those between 10^{-11} species-eq/m^2 and 10^{-8} species-eq/m^2 (Figure 9.5.37C and D).

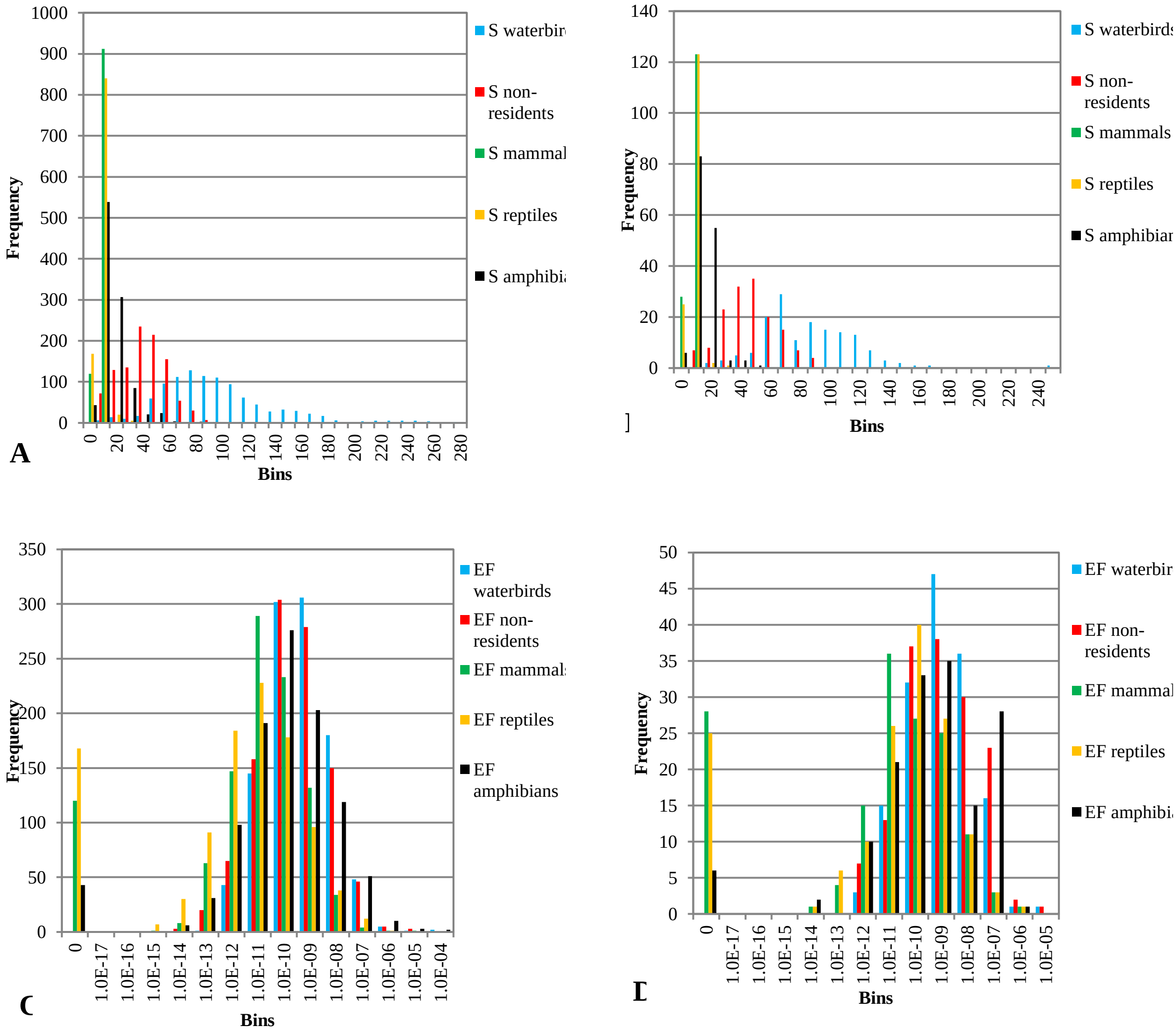


Figure 9.5.37: Distribution of the species richness for A) SW-fed wetlands and B) GW-fed wetlands. The distribution for the EF factors is shown in C) for SW-fed wetlands and D) for GW-fed wetlands.

A large part of the sensitivity of the characterization factor comes from the fate factor (FF). The sensitivity of the FF is discussed in detail in Verones et al. (2013). In Figure 9.5.38, the differences between the FFs for different amounts of water consumption is shown as factor of the FF with 10 m^3/yr consumption divided by the FF with 1'000'000 m^3/yr consumption. For the SW-fed wetlands, the differences are small, since the factor varies over the whole globe only between 1 and 1.167. For the GW-fed wetlands, the non-linearity of the well formula shows in the much larger differences between the FFs, distributed over the world. As stated in Verones et al. (2013), caution should thus be applied when using the factors for GW-fed wetlands.

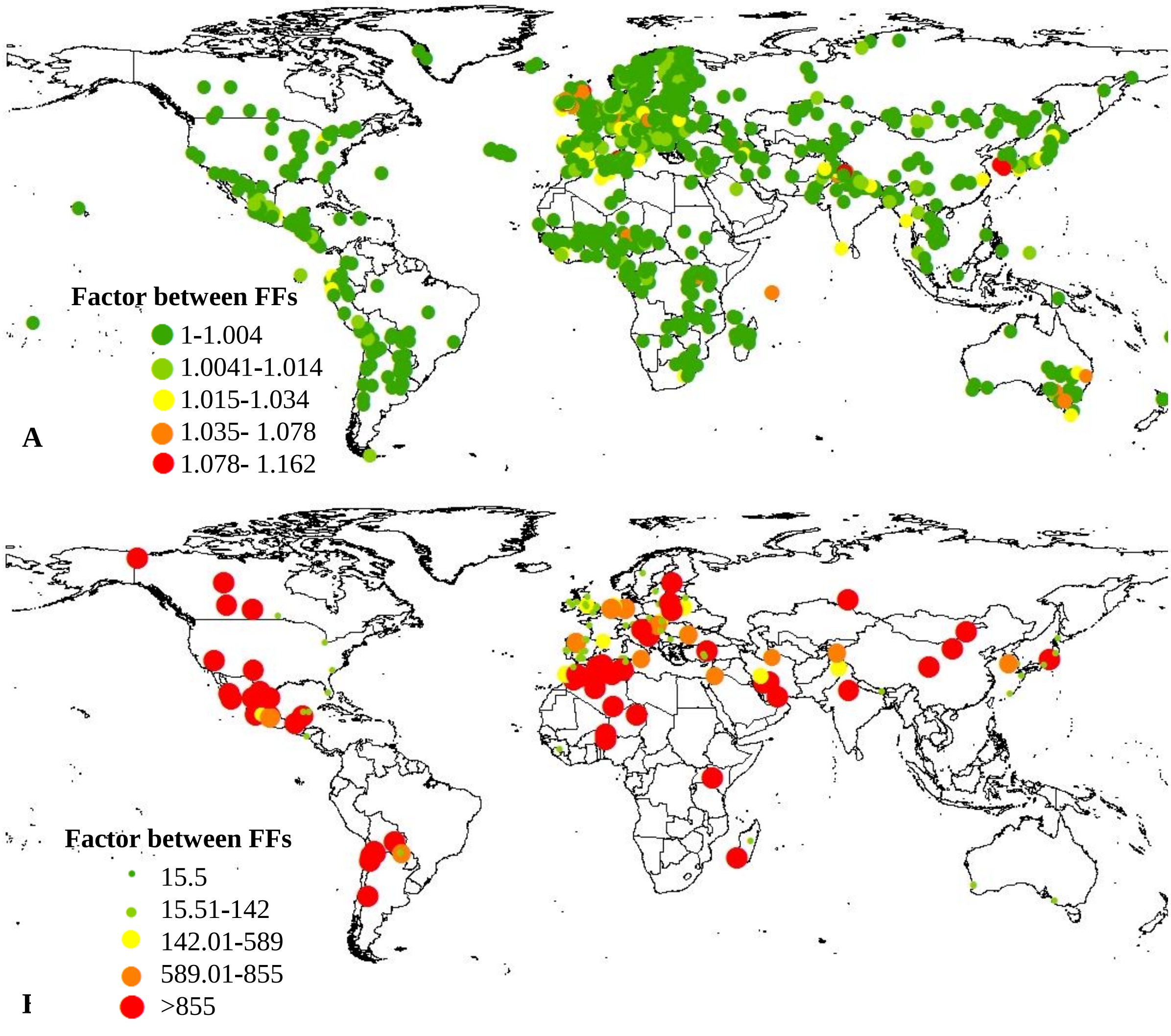


Figure 9.5.38: Factor between FFs with different amounts of water consumption. A) for SW-fed wetlands. B) for GW-fed wetlands. Base map with country boundaries adapted from ref.(ESRI 2009)

Whether the CF is calculated with the Ramsar area or the waterbody area can make a substantial difference, as is shown in Table 9.5.11 exemplarily for waterbirds (being the most dominant taxon). For the other taxa the trend is similar. The table shows the relation between values that are once calculated (CF, EF, FF) or extracted from maps (S, VS, CpA) based on Ramsar areas and once based on waterbody areas. The median, as well as the 2.5% and 97.5% percentile (between them are 95% of the data), are shown for these factors between Ramsar and waterbody areas based results. The difference between species richness, CpA and the vulnerability score is very low. However, the difference in the EF is larger, which is due to the non-linearity of the species-area relationship. In 95% of the data, the difference of the FF is smaller than for the EF. For the GW-fed wetlands, the difference between the FFs regarding waterbody or Ramsar area is negligible, because differences between the areas are often smaller. This is probably a coincidence. Bear in mind that the sample for the SW-fed wetlands is almost 10 times larger than for the GW-fed wetlands. Since the differences for the FFs and the underlying area are the same, they are not repeated. The minor differences from S, VS and CpA are not indicated.

Table 9.5.11: Factors between different parameters calculated based on the waterbody and the Ramsar area of each wetland, respectively. Between the 2.5% percentile and 97.5 % percentile 95% of the wetlands are found. These values are for waterbirds, as this is the most dominant taxon and covers all wetlands.

Factor between waterbody and Ramsar value	SW-fed wetlands			GW-fed wetlands		
	median	2.5% percentile	97.5% percentile	median	2.5% percentile	97.5% percentile
CF	1.25	0.92	70.14	1.60	0.64	62.44
EF	1.67	0.97	55.24	1.67	1.00	63.23
FF	0.93	0.12	10.53	1.00	0.19	1.00
Underlying wetland area (A)	0.60	0.02	1.00	0.60	0.02	1.00
Species richness (S)	1.00	0.96	1.04	1.00	0.98	1.02
Habitat loss risk (CpA)	1.00	0.92	1.09	1.00	0.91	1.09
Vulnerability score (VS)	1.00	0.94	1.06	1.00	0.98	1.03

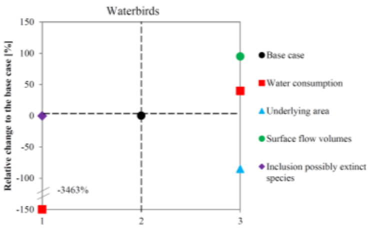
Table 9.5.12: Factors for the taxa other than waterbirds between CF and EF calculated based on the waterbody and the Ramsar area of each wetland, respectively. Between the 2.5% percentile and 97.5 % percentile 95% of the wetlands are found. Factors between the FFs or the underlying areas are the same as for waterbirds.

Taxa	Factor between waterbody and Ramsar value	SW-fed wetlands			GW-fed wetlands		
		median	2.5% percentile	97.5% percentile	median	2.5% percentile	97.5% percentile
non-residential birds	CF	1.24	0.88	73.31	1.60	0.64	61.74
	EF	1.67	0.97	58.29	1.67	1.00	63.43
water-dependent mammals	CF	1.22	0.90	53.65	1.50	0.67	32.49
	EF	1.67	0.99	34.12	1.67	1.00	41.31
reptiles	CF	1.20	0.93	55.82	1.58	0.57	23.13
	EF	1.67	1.00	42.12	1.67	1.00	28.27
amphibians	CF	1.24	0.85	57.64	1.57	0.63	44.07
	EF	1.67	0.96	49.39	1.67	1.00	51.32

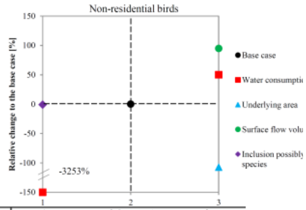
Overviews of the sensitivity analysis for the characterization factors (for each taxon and water source separately) are shown in **Error! Reference source not found.** and **Error! Reference source not found.**. The parameters that were important in the sensitivity analyses of the fate factors

(FF)(Verones et al. 2013) were used here again. In addition, the influence of including possibly extinct species (presence category 4, see

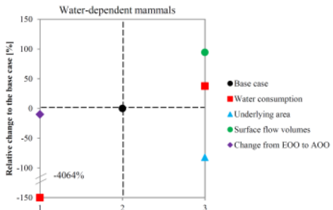
Table 9.5.) was checked (for mammals: area of occupancy instead of extent of occurrence). As for the FFs, the amount of water consumed, surface water flow volumes, and hydraulic conductivity were most relevant. However, due to the non-linear species-area relationship, the underlying area is now relevant for both SW-fed and GW-fed wetlands, while for the FF it was only relevant for some GW-fed wetlands. The influence of including possibly extinct species is in the majority of cases negligible. For the mammals, the change from the extent of occurrence (EOO) to the actual area of occupancy (AOO) of the species had a slightly larger influence, showing that for future developments the derivation of AOOs is relevant.



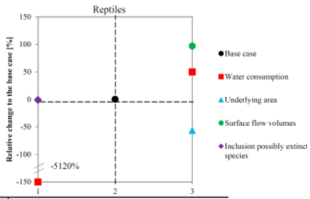
parameter	1	2 (base case)	3
Water consumption [m³/yr]	10	**1000**	1'000'000
underlying area [ha]	-	**Ramsar area**	waterbody area
surface flow volumes [-]	-	**own model**	WaterGap
Inclusion possibly extinc species [-]	Presence 1-4	**Presence 1-3**	-



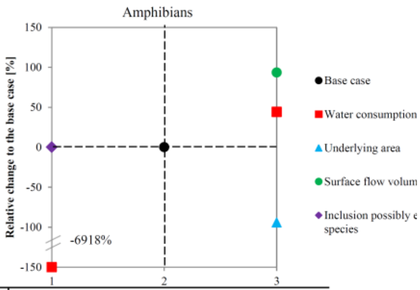
parameter	1	2 (base case)	3
Water consumption [m³/yr]	10	**1000**	1'000'000
underlying area [ha]	-	**Ramsar area**	waterbody area
surface flow volumes [-]	-	**own model**	WaterGap
Inclusion possibly extinc species [-]	Presence 1-4	**Presence 1-3**	-



parameter	1	2 (base case)	3
Water consumption [m³/yr]	10	**1000**	1'000'000
underlying area [ha]	-	**Ramsar area**	waterbody area
surface flow volumes [-]	-	**own model**	WaterGap
Change from EOO to AOO [-]	AOO	**EOO**	-

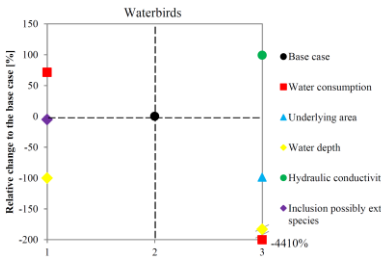


parameter	1	2 (base case)	3
Water consumption [m³/yr]	10	**1000**	1'000'000
underlying area [ha]	-	**Ramsar area**	waterbody area
surface flow volumes [-]	-	**own model**	WaterGap
Inclusion possibly extinc species [-]	Presence 1-4	**Presence 1-3**	-

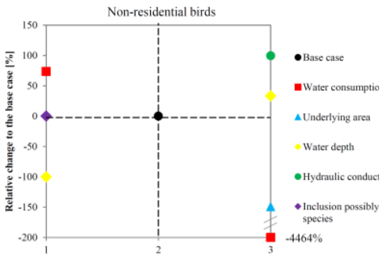


parameter	1	2 (base case)	3
Water consumption [m³/yr]	10	**1000**	1'000'000
underlying area [ha]	-	**Ramsar area**	waterbody area
surface flow volumes [-]	-	**own model**	WaterGap
Inclusion possibly extinc species [-]	Presence 1-4	**Presence 1-3**	-

Figure 9.5.39: Sensitivity analysis overview for SW-fed wetlands and all taxa separately. The change in each parameter is given for the global median values, parameters are listed below the graphic. Too large changes are indicated next to the marker.



parameter	1	2 (base case)	3
Water consumption [m³/yr]	10	**1000**	1'000'000
underlying area [ha]	-	**Ramsar area**	waterbody area
water depth [m]	-50%	**data from literature**	+50%
Hydraulic conductivity [m/s]	-	**calculated**	*100
Inclusion possibly extinct species [-]	Presence 1-4	**Presence 1-3**	-



parameter	1	2 (base case)	3
Water consumption [m³/yr]	10	**1000**	1'000'000
underlying area [ha]	-	**Ramsar area**	waterbody area
water depth [m]	-50%	**data from literature**	+50%
Hydraulic conductivity [m/s]	-	**calculated**	*100
Inclusion possibly extinct species [-]	Presence 1-4	**Presence 1-3**	-

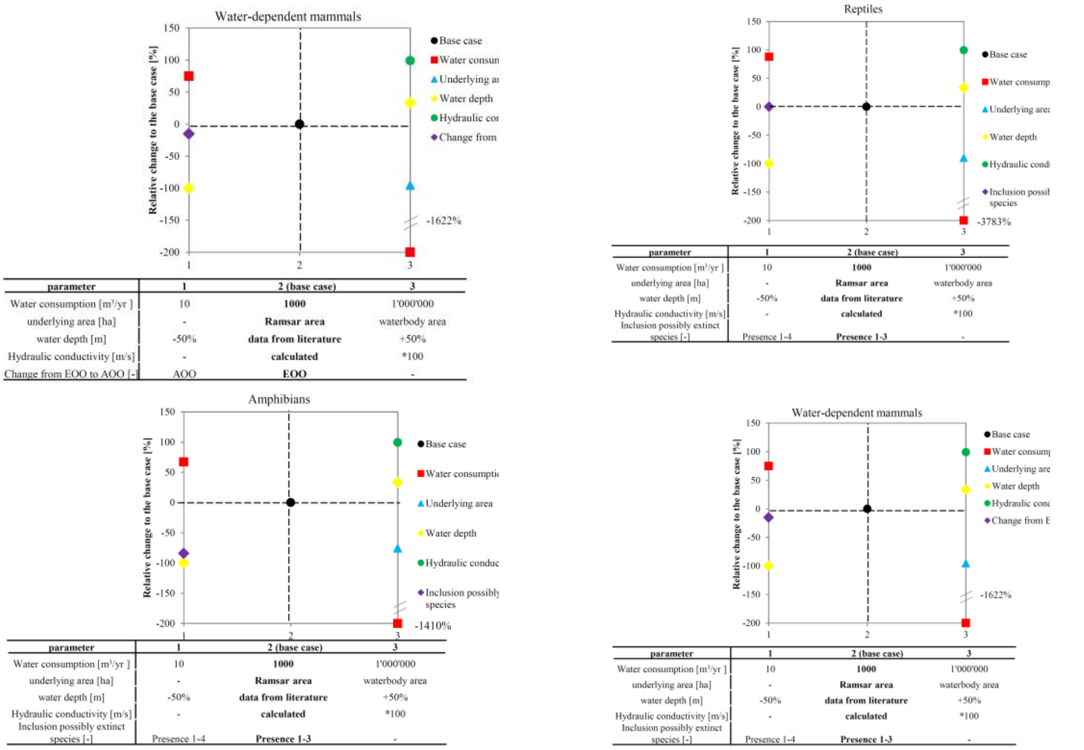


Figure 9.5.4016: Sensitivity analysis overview for GW-fed wetlands and all taxa separately. The change in each parameter is given for the global median values, parameters are listed below the graphic. Too large changes are indicated next to the marker.

9.5.2. Example of comparison of CFs calculated with PDFs and species-eq.

The CF in PDF·yr/m^3 of the two SW-fed wetlands "Cheyenne Bottoms" (USA) and "Lake Ånnsjön" (Sweden) is the same ($3.38 \cdot 10^{-9}$ PDF·yr/m^3). However, the number of waterbird species present is very different, being 112 in "Cheyenne Bottoms" and 49 in "Lake Ånnsjön". Absolute species loss was one order of magnitude smaller in the Swedish wetland. VS is the same order of magnitude in both wetlands ($2.5 \cdot 10^{-6}$ for Cheyenne Bottoms and $1.1 \cdot 10^{-6}$ for Lake Ånnsjön) and CpA one order of magnitude lower in "Cheyenne Bottoms", showing that the wetland habitat loss risk is larger than in "Lake Ånnsjön". CFs are consequently different for those two wetlands in the species-eq approach (CF for "Cheyenne Bottoms" $1.83 \cdot 10^{-10}$ species-eq·yr/m^3, for "Lake Ånnsjön" $5.8 \cdot 10^{-12}$ species-eq·yr/m^3).

9.5.3. Agricultural water requirement ratio

The consumptive share of the water use can be used for estimating water consumption amounts from withdrawn water amounts (for agriculture). It is based on data from AQUASTAT (FAO 2012) for 93 developing countries and on data from Döll and Siebert(2002) for other world regions and is shown in Figure 9.5..

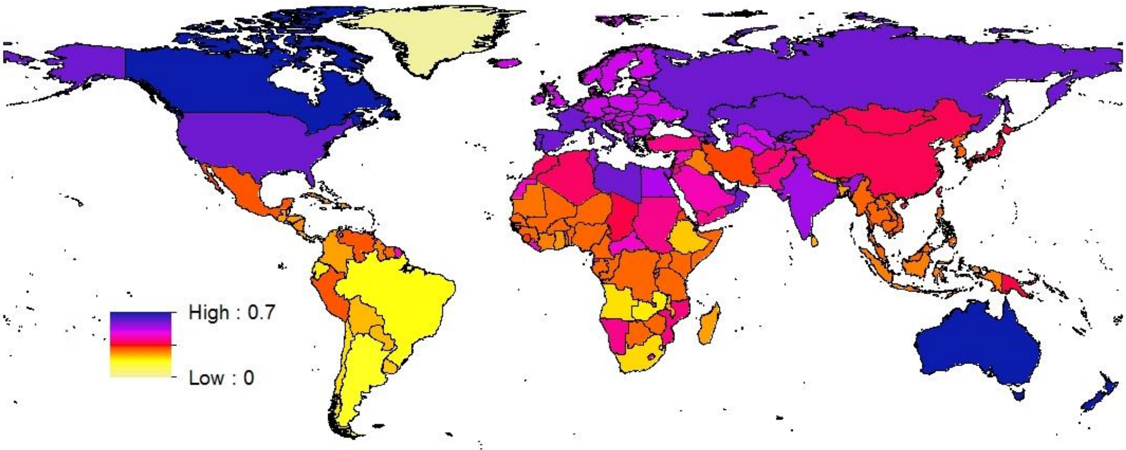


Figure 9.5.41: Water requirement ratios based on data from AQUASTAT(FAO 2012) and Döll and Siebert (2002).

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9.6. Annex to Chapter 6

9.6.1. Location of the case study area

The case study is located in Peru, in the province of Huaral (Figure 9.6.1). The relevant area is the lower part of the valley, where most agricultural activities take place and which measures approximately 1200 km^2 (see also main text).

Figure 9.6.1: The small picture shows Peru. The blow-up shows the Lima region in Peru. The province Huaral is shown in red and the relevant case study area is shown in a blue circle (Emilia 2011).

9.6.2. Hydrogeological map of the case study area

The hydrogeology of the watershed of the river Chancay-Huaral shows that a reasonably well developed aquifer is not present throughout the whole catchment, which coincides mostly with the area of the district of Chancay-Huaral (Figure 9.6.2). It is mainly the blue part (Qh-c) which can be considered relevant as an aquifer (area: 568.7 km^2). It is described as “Acuiferos generalmente extensos, con productividad elevada (permeabilidad elevada)” (extensive aquifers with elevated production (elevated permeability) (INGEMMET, 2009). The other most important layer types are described as follows:

Ks-mzgr/gdl-sr	“Formaciones generalmente sin acuíferos (pemeabilidad muy baja)” (formations without aquifers (very low permeability))
Ks-gd/to-pa	“Formaciones generalmente sin acuíferos (pemeabilidad muy baja)” (formations without aquifers (very low permeability))
PN-vs	“Acuíferos locales, en zonas fracturadas o meteorizadas en formaciones consolidadas, sin excluir acuíferos cautivos más productivos (permeabilidad baja a muy baja)” (local aquifers in fractured zones or weathered in consolidated formations, without excluding trapped but more productive aquifers (low to very low permeability))
Ki-di/gb	“Formaciones generalmente sin acuíferos (pemeabilidad muy baja)” (formations without aquifers (very low permeability))
Ks-mzgr/gdi-sr	“Formaciones generalmente sin acuíferos (pemeabilidad muy baja)” (formations without aquifers (very low permeability))

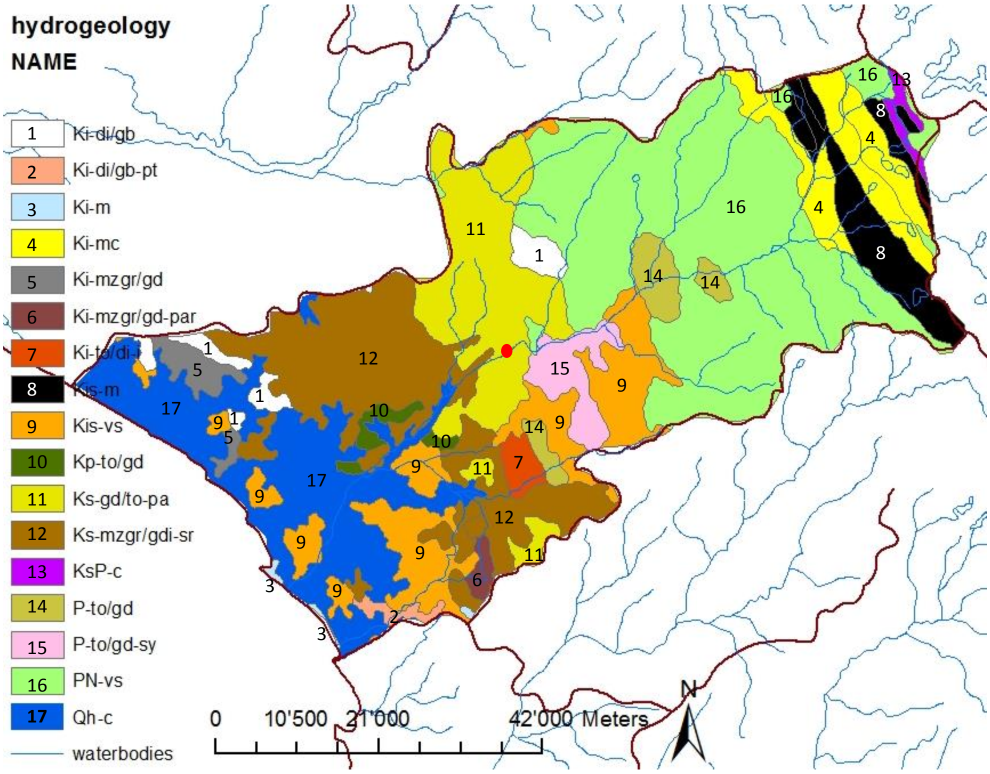

Figure 9.6.2: Hydrogeological map of the area of Chancay-Huaral (adapted from INGEMMET (2009)) which is mostly identical with the watershed of the river Chancay-Huaral. The blue area is the relevant Aquifer area. The red point indicates the approximate location of Santo Domingo. For a description of the most important abbreviations, see text.

The Aquifer itself consists primarily of quaternary, alluvial material. That means that these deposits are composed of mixtures of blocs, cobblestones, pebbles, gravel, sand and loam, forming layers of different thickness. The thickness is given with 30 to 50 m in most areas for the extensive aquifers. The maximum thickness is 84 m (Prieto Celi et al. 2001).

The soil is assumed to be medium grained sand, which is confirmed by own observations and Vera et al.(2006).

9.6.3. Climatic parameters

Air temperature

The average monthly minimum and maximum air temperature are aggregated from hourly measurements from Lima from 2005, as reported in METEOTEST (2011) and shown in Table 9.6.1. Mean monthly dew point temperature and mean air temperature are average monthly values from 1996-2005 (METEOTEST, 2011). In CEPES (nd) the mean annual temperature for three stations is given as 19.0-19.2°C. However, since not more data was available directly for the region of the lower valley of Chancay-Huaral and the distance between Huaral (the main town) and Lima is only approximately 60 km it was assumed that values for the latter can be applied to

Chancay-Huaral. The difference between 19°C and 19.6°C for mean annual temperatures is small and justifies this assumption for the purpose of this study.

Table 9.6.1: Mean monthly values of maximum, minimum, mean and dew point temperatures for the weather station in Lima (METEOTEST, 2011) and average monthly wind speed values u from METEOTEST (2011).

Month	Tmax [°C]	Tmin [°C]	Tmean [°C]	Tdew [°C]	Wind speed u [m/s]
Jan	27.1	19.3	22.1	18.9	5.0
Feb	27.6	20.1	23.1	19.5	4.5
Mar	27.2	19.6	23	19	4.2
Apr	26.3	18.0	21.4	17.7	3.9
May	24.4	16.4	19.4	15.9	3.7
Jun	24.9	16.0	18.6	14.7	3.5
Jul	24.6	15.6	18.2	14.1	3.6
Aug	23.3	15.6	17.6	13.9	3.8
Sep	22.0	14.7	16.8	14	4.0
Oct	20.8	14.2	16.9	14.4	4.2
Nov	22.1	15.6	18	15.5	4.5
Dec	24.5	16.9	20	17.4	4.7
mean	24.6	16.8	19.6	16.3	4.1

Precipitation (P)

The precipitation for the region above Santo Domingo was available from Bernabé et al. (2001) as monthly averaged values (with Thiessen-polygon method) over different stations for 1960 to 2000. The annual sum of averages over all these years was used as precipitation in the upper area. The annual precipitation is 444 mm in this area. This is confirmed with pixel data of New et al. (2002) with a resolution of 10 Minutes, which gives a precipitation of 487 mm/a in the same area (see Figure 9.6.3).

Precipitation in the lower area (Santo Domingo to coast, 1245 km^2) is very low, however only few values were available. Data was finally taken from New et al. (2002), from a GIS Raster data set with a resolution of 10 minutes (see Figure 9.6.3) with average values from 1961-1990. The precipitation is 33.7 mm per year (amounting for 0.37 Mio m^3/a in this area).

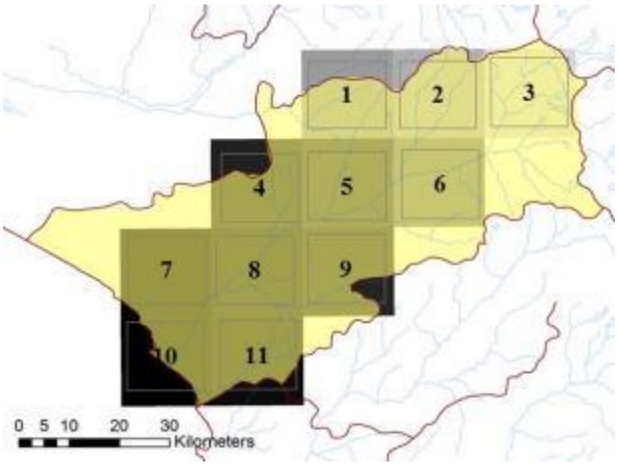

Figure 9.6.3: Mask of the area Chancay-Huaral (yellow) with the 10 minute resolution Pixels with precipitation. Pixels, 7, 8, 10 and 11 are attributed to the lower part of the valley after Santo Domingo.

Wind

The wind speed is derived from hourly wind speed measurements from Lima (60 km South-East, similar topology) from METEOTEST (2011) for the year 2005 (see Table 9.6.1). It is used for the calculation of crop evapotranspiration.

9.6.4. Agriculture

There is a variety of 64 crops grown in Chancay-Huaral (see Table 9.6.2) on totally almost 260 km^2. For the crop coefficients (K_c) we used different sources. The two main sources are FAO (Allen et al. 1998) and a website on irrigation for Córdoba in southern Spain (ElRiego.com (2011)), which represents similar climate. For Asparagus K_c values from Sanchez Vigo (2006) were used. In the case of "ElRiego.com" as main source for K_c values, the values for banana, pumpkin, cucumber and gherkin, onion, broccoli and celery stem from irrigation scheduling from Baja California from Snyder et al. (2007). In the same case, Alfalfa is taken from Koch (2003). For some crops no crop coefficients were available but it was tried to allocate the crop coefficients of "similar" crops (i.e. either crops from the same family or crops which grow similarly and e.g. both have waxy leaves, see Table 9.6.3 and 9.6.4).

Crops were grouped into classes, according to their K_c values and the crop areas in each class summed. A scaling factor, scaling all K_c values to the lowest value was introduced, in order to take into account the different water needs. If the lowest K_c value is 0.6, a plant with a K_c value of 0.9 will receive 1.5 times as much water as the plant with K_c value 0.6. According to that and the area in each class, the amount of water for each class was calculated and again transformed to irrigation water amounts per hectare and class. I.e. all crops with the same K_c values (in both cases with values from FAO and ElRiego, respectively) have the same irrigation water amount per hectare.

Table 9.6.2: Crops which were grown in Chancay-Huaral in 2010. The area is taken from information from the regional government (Gobierno Regional de Lima 2011b). The crop

coefficients are determined mainly from FAO (Allen et al. 1998) and ElRiego.com. (2011). According to the crop coefficients and the planted area, the water amounts allocated to the plant were determined (different for the two cases considered in the table). The colors help to visually group the crops into different categories, with different K_c values with violet being the highest water using plants and red the lowest ones. The value for total irrigation water (sum of last row) was taken from information from the local government (Gobierno regional de Lima 2011a).

Crops	Area [ha]	%of area	based on FAO		based on ElRiego.com	
			K_c	Irrigation water allocated [m^3/a]	K_c	Irrigation water allocated [m^3/a]
Alfalfa	256	1.0	1.0	3'116'848	0.85	2'891'963
Apple	2109	8.2	0.9	23'109'724	0.8	22'423'345
Artichoke	127	0.5	0.9	1'391'624	0.6	1'012'719
Asparagus	161	0.6	0.9	1'764'185	0.9	1'925'760
Avocado	1676	6.5	0.8	16'324'491	0.55	12'250'972
Banana	18	0.1	1.1	241'069	1.0	239'225
Beans	160	0.6	0.9	1'753'227	0.6	1'275'866
Beans (other crop type)	157	0.6	0.9	1'720'354	0.6	1'251'944
Brocoli	118	0.5	0.9	1'293'005	0.8	1'254'602
Cabbage	127	0.5	0.9	1'391'624	0.7	1'181'505
Caigua (pumpkin family)	97	0.4	0.8	944'795	0.8	1'031'325
Capsicum	161	0.6	0.9	1'764'185	0.8	1'711'787
Carrot	696	2.7	0.9	7'626'537	0.8	7'400'023
Cauliflower	167	0.7	0.9	1'829'931	0.7	1'553'633
Cayenne pepper	144	0.6	0.9	1'577'904	0.8	1'531'039
Celery	54	0.2	1.0	657'460	0.9	645'907
Chirimoya	47	0.2	0.6	343'340	0.55	343'554
Coriander	70	0.3	0.9	767'037	0.8	744'255
Cotton	423	1.6	0.7	3'605'073	0.8	4'497'428
Cucumber	32	0.1	0.8	311'685	0.8	340'231
Garlic	163	0.6	0.9	1'786'100	0.9	1'949'683
Gherkin	154	0.6	0.8	1'499'983	0.8	1'637'361
Grape	435	1.7	0.6	3'177'724	0.6	3'468'761
Grapefruit	8	0.0	0.6	58'441	0.6	63'793
Green Beans	166	0.6	0.9	1'818'973	0.6	1'323'711
Guanabano (fruit tree)	4	0.0	0.6	29'220	0.6	31'897
Leek	14	0.1	0.9	153'407	0.8	148'851
Lettuce	142	0.6	0.9	1'555'989	0.7	1'321'053
Lima Bean	82	0.3	0.9	898'529	0.6	653'881
Lime	2	0.0	0.6	14'610	0.6	15'948
Lucuma	309	1.2	0.6	2'257'280	0.55	2'258'682
Maize (maiz amilaceo)	128	0.5	0.8	1'246'739	0.8	1'360'924
Maize (maiz chala)	882	3.4	0.8	8'590'812	0.8	9'377'615
Maize (maiz choclo)	303	1.2	0.8	2'951'265	0.8	3'221'562
Maize (maiz morado)	230	0.9	0.8	2'240'235	0.8	2'445'410
Mandarins	3415	13.3	0.6	24'946'959	0.55	24'962'451
Mango	702	2.7	0.6	5'128'189	0.55	5'131'374
Marigold	23	0.1	1.0	280'029	0.85	259'825
Medlar	10	0.0	0.9	109'577	0.7	93'032
Oca (a kind of potato)	3	0.0	0.8	29'220	0.8	31'897
Olive	12	0.0	0.7	102'272	0.6	95'690
Olluco (a kind of potato)	8	0.0	0.8	77'921	0.8	85'058
Onion	139	0.5	0.9	1'523'116	0.9	1'662'613
Orange	168	0.7	0.6	1'227'259	0.6	1'339'659
Pacae o guabo (tree)	3	0.0	0.9	32'873	0.6	23'922
Papaya	10	0.0	0.9	109'577	0.6	79'742
Passionfruit	142	0.6	0.6	1'037'326	0.7	1'321'053
Pea	187	0.7	1.0	2'276'760	0.6	1'491'168

Peach	1690	6.6	0.8	16‘460‘854	0.7	15‘722‘390
Pear	21	0.1	0.9	230‘111	0.7	195‘367
Pecan	64	0.2	1.0	779‘212	0.6	510‘346
Plum	7	0.0	0.8	68‘181	0.7	65‘122
Pomegranate	24	0.1	0.6	175‘323	0.7	223‘277
Potato	3212	12.5	0.8	31‘285‘362	0.8	34‘150‘680
Prickly pear	8	0.0	0.6	58‘441	0.55	58‘477
Pumpkin	178	0.7	0.8	1‘733‘747	0.8	1‘892‘535
Quince	18	0.1	0.9	197‘238	0.7	167‘457
Spinach	20	0.1	0.9	219‘153	0.7	186‘064
Strawberry	580	2.3	0.8	5‘649‘287	0.7	5‘395‘850
Sweet potato	236	0.9	0.9	2‘586‘010	0.8	2‘509‘203
Tomato	120	0.5	0.9	1‘314‘920	0.8	1‘275‘866
Watermelon	76	0.3	0.8	740‘251	0.8	808‘048
Yellow maize	4660	18.1	0.8	45‘389‘099	0.8	49‘546‘130
Yucca	134	0.5	0.9	1‘468‘328	0.55	979‘493
Total Area	25692	100		245‘050‘000		245‘050‘000

Table 9.6.3: Allocation of K_c values for crops which have no available coefficient in Allen et al. (1998).

crop	K_c from	reason
Cayenne pepper	Capsicum	same genus (Capsicum)
Caigua (pumpkin family)	Cucumber	same family (Cucurbitaceace), similar growth
Chirimoya	Citrus	deciduous tree
Coriander	Carrot	same family (Apiaceae)
Guanabano	Citrus	waxy leaves
Leek	Tomato	assumed to be like tomato
Lucuma	Citrus	slightly waxy leaves
Mango	Citrus	waxy leaves
Marigold	Grass	waxy leaves
Medlar	Apple	same family (Rosaceae)
Oca	potato	similar growth like potato
Olluco	potato	similar growth like potato
Pacae o guabo (tree)	Apple	deciduous tree (not as efficient as citrus)
Papaya	Apple	deciduous tree (not as efficient as citrus)
Passionfruit	Citrus	waxy leaves
Pea	Bean	same family (Fabaceae)
Pecan	Walnut	same family (Juglandaceae)
Pomegranate	Citrus	waxy leaves
Prickly Pear	Citrus	assumed to be water efficient
Quince	Apple	same family (Rosaceae)
Yuca	Date palms	assumed to be water efficient

Table 9.6.4: Allocation of K_c values to crops for which no values were available on ElRiego.com.(2011) These are more values than in Table 9.6.3, since the latter is based on a more extensive source.

crop	Kc from	Reason
Avocado	Mandarin	waxy leaves
Artichoke	bean	same values in FAO like beans
Cabbage	Lettuce	lettuce and cabbage have similar crop coefficients in FAO
Caigua (pumpkin family)	Cucumber	same family (Cucurbitaceace), similar growth
Cauliflower	Lettuce	Crucifer like cabbage
Cayenne pepper	Capsicum	same genus (Capsicum)
Chirimoya	Mandarin	deciduous tree, assumed to be water efficient
Coriander	Carrot	same family (Apiaceae)
Garlic	Onion	same genus (Allium)
Guanabano	Orange	waxy leaves

Leek	Tomato	assumed to be like tomato
Lucuma	Mandarin	deciduous tree, assumed to be water efficient
Mango	Mandarin	waxy leaves
Marigold	Grass	waxy leaves
Medlar	Plum	same family (Rosaceae)
Oca	potato	similar growth like potato
Olluco	potato	similar growth like potato
Pacae o guabo (tree)	Orange	(slightly) waxy leaves
Papaya	Orange	waxy leaves
Passionfruit	Orange	waxy leaves
Pea	Bean	same family (Fabaceae)
Pecan	Walnut	same family (Juglandaceae)
Pomegranate	Peach	(slightly) waxy leaves
Prickly Pear	Citrus	assumed to be water efficient
Quince	Plum	same family (Rosaceae)
Spinach	Lettuce	assumed like lettuce (large leaves)
Strawberries	Lettuce	assumed like lettuce (large leaves)
Yuca	Date palms	assumed to be water efficient

Further, when applying the water that is withdrawn from river and aquifer (245.05 Mio m^3/a) the irrigation water efficiency and thus the water stress for the crops can vary. There are two scenarios calculated with higher or lower water losses on average. One scenario follows a rather technical approach of calculating the water use efficiency and thus the losses of water (part of the water will never reach the plants). This factor of irrigation efficiency is assumed to be usable as K_s factor and is termed $K_{s,2.}$ It is calculated according to Brouwer et al. (1989), assuming unlined canals in loamy earth with a medium length (200- 2000 m) and surface irrigation (furrow irrigation). Asparagus is the only crop which is assumed to have drip irrigation in this scenario. $K_{s,2}$ is calculated according to Equation 9.3.1:

$$=$$

Equation 9.6.1

e_c conveyance efficiency [%]
e_a application efficiency [%]

This leads to a water use efficiency ($K_{s,2}$) of 45% for furrow irrigation and 67.5% for asparagus, respectively. This is considered a "reasonable" water use efficiency in Brouwer et al. (1989).

The second scenario is a more plant-related one, where the yield and the amount of water use to reach a certain yield are incorporated. $K_{s,1}$ is thus estimated with yields and yield response factors according to Allen et al. (1998) as shown in Equation 9.6.2.

$$= 1 - \cdot$$

Equation 9.6.2

$K_{s,1}$ stress factor
K_y yield response factor
Y_a actual yield
Y_m maximum (expected) yield

Values for yield response factors are available from FAO (Doorenbos et al. 1979) for a variety of crops. If no value was available the yield response factor was assumed to be one. Actual yields are derived as 10-year averages from the crop information from Chancay-Huaral (Gobierno Regional de Lima (2011b) for each crop. The expected maximum yield was estimated from Pfister et al.(2011). Only arid areas like Egypt or Israel were taken into account for comparison. If the indicated value was smaller than the actual yield for Chancay-Huaral or the crop was not available in the table, the maximum yield was approximated via Equation 9.6.2. The calculations lead to a mean $K_{s,1}$ value of 0.5, which is varying widely between 0.1 and 0.9. This means that 10-90% of the crop water requirements are supplied to the plants.

According to the administration for irrigation in Chancay-Huaral (Administración técnica des distrito d riego Chancay-Huaral, n.d.) the irrigation efficiency is 48% which is in line with the values calculated for the water use efficiency of 45%.

Table 9.6.5: Crop yield response factor K_y, actual yield Y_a, estimated maximal annual yield Y_m and the estimated water stress factor $K_{s,1}$. The mean K_s value here is 0.5.

Crops	K_y	Y_a [t/ha]	Y_m [t/ha]	$K_{s,1}$
Alfalfa	0.7	37	42	0.83
Apples	1.0	12	17	0.71
Artichokes	1.0	8.5	19	0.45
Asparagus	1.0	6	9	0.67
Avocado	1.0	9	20	0.45
Banana	1.2	13.5	39	0.46
Beans	1.2	10.5	29	0.45
Beans (other crop type)	1.2	14.5	39	0.45
Brocoli	1.0	17.5	39	0.45
Cabbage	1.0	32	37	0.86
Caigua (pumpkin family)	1.0	15	33	0.45
Capsicum	1.1	25	63	0.45
Carrot	1.0	31	44	0.70
Cauliflower	1.0	37	44	0.83
Cayenne pepper	1.1	7	18	0.45
Celery	1.0	22	49	0.45
Chirimoya	1.0	7	16	0.45
Coriander	1.0	18	40	0.45
Cotton	0.9	2	4	0.41
Cucumber	1.0	11	84	0.13
Garlic	1.0	7	23	0.30
Gherkin	1.0	34	84	0.40
Grapefruit	0.8	11.5	43	0.08
Grapes	0.9	5	19	0.13
Green Beans	1.2	4	11	0.45
Guanabano (fruit tree)	1.0	11	24	0.45
Leek	1.0	22.5	50	0.45
Lettuce	1.0	28	34	0.84
Lima Bean	1.2	8	22	0.45
Lime	0.8	7	20	0.20
Lucuma	1.0	7.5	17	0.45
Maize (maiz amilaceo)	1.3	1.2	17	0.26
Maize (maiz chala)	1.3	100	320	0.45
Maize (maiz choclo)	1.	16.5	19	0.89
Maize (maiz morado)	1.3	4	17	0.39
Mandarins	0.8	19.5	26	0.69
Mango	1.0	8.5	18	0.49
Marigold	1.0	12.5	28	0.45
Medlar	1.0	6.5	14	0.45
Oca (a kind of potato)	1.1	3	33	0.17

Olive	1.0	3	7	0.43
Olluco (a kind of potato)	1.1	3.5	33	0.19
Onions	1.0	19	30	0.64
Oranges	1.0	11	20	0.56
Pacae o guabo (tree)	1.0	5	11	0.45
Papaya	1.0	5	13	0.38
Passionfruit	1.0	12	27	0.45
Peaches	1.0	8.5	10	0.89
Pear	1.0	6.5	11	0.59
Peas	1.2	6	16	0.45
Pecan	1.0	2	3	0.67
Plums	1.0	7.5	12	0.63
Pomegranate	1.0	7.5	17	0.45
Potatoes	1.1	30	35	0.87
Prickly pear	1.0	8.5	19	0.45
Pumpkin	1.0	36	80	0.45
Quince	1.0	7	16	0.45
Spinach	1.0	19	25	0.76
Strawberries	1.0	18	24	0.75
Sweet potatoes	1.1	36	91	0.45
Tomatoes	1.1	34	38	0.90
Watermelon	1.1	12	28	0.48
Yellow maize	1.3	9	17	0.62
Yucca	1.0	10	22	0.45

Production area of crops which show a decreasing trend (Table 9.6.6) is proportionally reduced for the scenarios, in order to allow an expansion of the chosen scenario crops.

Table 9.6.6: Crops which showed over the last 10 years on average a decrease, adapted from the information from the regional government (Gobierno Regional de Lima, 2011b).

Crop	**Average decrease [%/yr]**
Alfalfa	-8.7
Apple	-2.6
Beans	-1.5
Cayenne pepper	-7.8
Cotton	-11
Green beans	-4.5
Lima bean	-1.1
Mango	-0.1
Marigold	-29
Oca (a kind of potato)	-11.1
Olluco (a kind of potato)	-2.2
Orange	-7.4
Pea	-4.8
Peach	-4.1
Sweet potato	-14
Tomato	-6.2

The areas of the expanding crops have been increasing with very different percentages over the last decade (Table 9.6.7). All increases are estimated for Chancay-Huaral from the information from the regional government (Gobierno Regional de Lima 2011b) except for asparagus. The asparagus area in Chancay-Huaral has shown a decreasing trend until 2005 but is steadily increasing since then. In the workshop held in February

2011 in Chancay the participants identified asparagus as one important future crop and therefore a scenario for asparagus is considered as well. For the growth rate per year values from the Ica region, a coastal area and major asparagus producing region in Peru are applied (Portal Agrario Regional Ica, 2011).

Table 9.6.7: Crops which are used in the different scenarios with increasing area. Average increase rate for asparagus is estimated from the increase rate in Ica over the last 20 years (Portal Agrario Regional Ica, 2011), the rest is estimated from 10 year data series from information from the regional government (Gobierno Regional de Lima 2011b).

Crop	**Average increase [%/yr]**
Asparagus	*90*
Mandarin	5.6
Potato	12.1
Avocado	4.3
Yellow Maize	6.3
Strawberry	6.8

The decreasing crops have a total area of 6477 ha in 2010, which is the area available for a further expansion of the increasing crops until 2020 (assuming the total disappearance of the decreasing crops). For the scenarios “increase Asparagus”, “increase Mandarin”, “increase Strawberry”, “increase drip-irrigation” and “increase Mandarin, more GW” the increase in respective crop area can be calculated with the average annual increase from Table 9.6.7. For the scenario “increase multi-crop” the theoretically possible increase in area of the six crops exceeds 6477 ha (according to the crops’ individual annual increase rates). Therefore the six crops increase as a percentage of 6477 ha, according to their overall contribution to the theoretically possible area increase.

9.6.5. Water withdrawals (RWW and GWW)

Water is withdrawn from the river (river water withdrawal, RWW) and the groundwater (GWW). The main water withdrawals stem from the agricultural sector with 245.05 Mio m^3/a. The largest share of the water withdrawn is abstracted from the river (Table 9.6.8).

Table 9.6.8: Annual water withdrawals in the valley of Chancay-Huaral, broken down to water sources and sectors (Gobierno Regional de Lima 2011a).

	Source	
	river [Mio m^3/a]	**groundwater [Mio m^3/a]**
agricultural	243	2.05
domestic	4.57	0.04
industrial	0	0.55
livestock	0.01	0.67
mining	0.11	0

9.6.6. Hydrologic parameters

River (I)

Daily discharges were available from ALA (2010). The long-term average annual discharge of 16.1 m^3/s is used in the model as inflow (I). Figure 9.6.4 shows the average discharge per month. The variability between the months, caused by rainfall events and snow melt in the higher mountains are clearly visible. CEPES (2010) reported that more river water was flowing to the Sea than was measured at the measuring station, situated ca. 38 km inland. It is assumed that part of this additional water is exfiltrated groundwater.

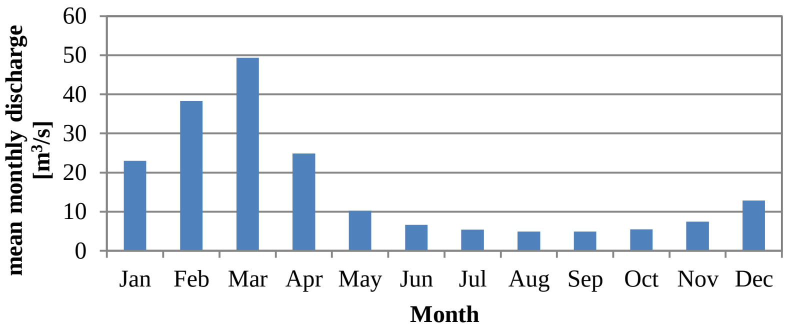

Figure 9.6.4: Mean monthly discharges for the period 1960 to 2007, measured at the measuring station Santo Domingo.

Groundwater

Data on groundwater inflow (GWI) into the lower area was not available and hence had to be estimated. With the formula of Philip (Jury et al. 1993) the infiltration from the river was calculated according to Equation 9.6.3.

$$i = s \cdot + k \cdot t$$

Equation 9.6.3

i=infiltration [m]
s=sorptivity
t=time [d]
k=hydraulic conductivity [m/d]

The sorptivity was neglected, since the area beneath the riverbed is saturated and sorptivity is only important at the beginning of an infiltration. It was also assumed that the hydraulic conductivity of the aquifer corresponds to the one of the riverbed and thus the value of the aquifer can be implemented in Equation 9.6.3.

The main river has a length of 120 km (Bernabé et al. 2001) with a mean estimated width of the riverbed of 53 m (Google Earth, 2009). The length of the river from the measuring station to the Pacific Ocean is 38 km (Google Earth, 2009). With the calculated infiltration for one year and the estimated area of infiltration based on length and width of the river, the amount of groundwater infiltration above and below the measuring station Santo Domingo can be calculated and appear as groundwater inflow (GWI) and groundwater recharge from seepage (GWR_S) in the model.

Transmissivity values and hydraulic conductivity values are available from Prieto Celi et al.(2001) for different sections and averaged ($2{\cdot}10^{-2}$ m^2/s and $8.89{\cdot}10^{-4}$ m/s, respectively). These values correspond to coarse to medium grained sand (e.g. Scheffer et al., 2002). In the lower valley the soil is indeed sandy, which is confirmed by own observations and Vera et al. (2006), which state that the soils along the western coast of South America vary between loamy sand and sand and have a loose structure.

With average hydraulic conductivity values and transmissivity values an average aquifer thickness of ca. 25 m was calculated. This is rather an underestimation of the thickness of the aquifer, which is mostly between 30 and 50 m thick (Prieto Celi et al. 2011). However, since this was calculated with averaged transmissivity and conductivity values it is acceptable.

9.6.7. Infiltration from excess irrigation water (GWR_I)

In order to estimate the infiltrated water from the used irrigation water two different methods are applied.

The first approach is to estimate the infiltration to groundwater by subtracting for each crop the crop evapotranspiration (via Penman-Monteith method, see next section) from the applied irrigation water. The remainder is the amount of water which is recharging the groundwater.

The second approach is carried out with a formula from Turc (1954) (Equation 9.6.4 andEquation 9.6.5).

$$r = p \cdot$$

Equation 9.6.4

$$L = 300 + 25 \cdot T + 0.05 \cdot$$

Equation 9.6.5

r recharge rate [mm/a]
p precipitation [mm/a]
T mean annual temperature [°C]

It was assumed, that precipitation in this formula equals the irrigation water, since precipitation is almost negligible in the lower valley. The irrigation water applied on each crop area was used as precipitation in this formula. Caution has to be applied when using this formula, since it is purely empirical and does not reflect any local soil and environmental conditions.

With these assumptions a groundwater recharge of about 51 Mio m^3/a is calculated. This results in 198-199 mm/a over the whole agricultural area (260 km^2, weighted according to crop areas) with the different crop coefficients. With a total irrigation water use (replacement for precipitation here) of 954 mm (Gobierno Regional de Lima, 2011b), a theoretical crop evapotranspiration of 754-756 mm can be calculated, of which a part can also be evaporated unproductively. This is the difference between the irrigation water applied and the sum of ET_c and infiltrated water.

Comparing this value with the crop ET from the Penman-Monteith approach (470 mm or 672 mm with high and low stress factors, respectively and $K_{c,FAO}$) reveals differences of 60% and 12 %, respectively. However, in the total balance this leads to a smaller difference and considering that the approaches are quite diverging these differences are to be expected. In order to calculate the infiltrated amount of water from both evapotranspiration estimates (with $K_{s,1}$ and $K_{s,2}$ values) the amount of infiltrating water is estimated by subtracting the amount of evapotranspired water ($ET_{c, adj}$) from the totally applied irrigation amount of 245 Mio m^3/a (Gobierno Regional de Lima, 2011b).

9.6.8. Evaporation and Evapotranspiration (ET)

To calculate the evaporation and evapotranspiration several formulas are applied.

Turc

The formula of Turc (Equation 9.6.6) is used to estimate the actual evaporation from bare soil in the lower valley (area which is not covered with water or vegetation) (Gangopadhyaya, 1968, Ahmad, 1959, Burlando, 2012). The formula has proven to be reliable in climatically arid and humid area (Koch, 2003).

$$= P \cdot$$

Equation 9.6.6

$$= 300 + 25 \cdot t + 0.05 \cdot$$

Equation 9.6.7

ET_{act} actual evapotranspiration [mm]
P annual precipitation [mm/a]
t annual mean temperature [°C]

Evaporation from bare soil could also alternatively be estimated with the approach of FAO Penman-Monteith (see below), by assuming a K_c value of 0.1 and applying K_s values as well.

Thornthwaite

For Santa Rosa and the river, the potential evaporation needs to be calculated. This is done with the crop-independent formula of Thornthwaite (Koch, 2003; Stephens et al. 1963; Reed, n.d.) according to Equation 9.6.8 to Equation 9.6.10 for each month.

$$= 16 \cdot\cdot$$

Equation 9.6.8

$$=$$

Equation 9.6.9

$$a = 6.7 \cdot\cdot -7.7 \cdot\cdot +1.8 \cdot\cdot +0.49$$

Equation 9.6.10

N_m monthly correction for the day lengths of the specific month (dep. on latitude)
T_m mean monthly temperature [°C]

I_m heat index for the year

The latitude of Chancay-Huaral is considered roughly as 10°S for determining the N_m values. The actual latitude of Chancay-Huaral is between 11°S and 11°40'. The mean monthly temperature are the long-term average values for Lima based on METEOTEST (2011) as described above.

The river area on which this is applied is 38 km long (measuring station Santo Domingo to coast) and 53 m wide (estimated from Google Earth (2009)), resulting in 2.01 km^2. The evapotranspiration from Santa Rosa is also estimated with this formula for the whole wetland. 10 ha of the wetland is open water body. The other 26 ha are vegetated, with a water level above or just at the surface. Thus it is assumed that potential evaporation can be applied here.

FAO Penman-Monteith

The method of FAO Penman-Monteith (Allen et al. 1998) is used to estimate the evapotranspiration of all crops (Equation 9.6.11).

$$=\cdot\cdot$$

Equation 9.6.11

$ET_{c, adj}$ crop evapotranspiration [mm/day]
K_c crop coefficient [-]
$K_{s,1}$ stress factor [-] (Equation 9.6.2)
ET_o reference crop evapotranspiration [mm/d]

The reference crop evapotranspiration (grass reference, height 0.12 m, albedo 0.23, surface resistance 70 s/m) is calculated according to the Penman-Monteith equation as presented in Equation 9.6.12:

$$=$$

Equation 9.6.12

where
Δ slope of the vapour pressure curve [kPa/°C]
R_n net radiation [MJ/m2day]
G soil heat flux [MJ/m2day]
γ psychrometric constant [kPa/°C]
T mean air temperature [°C]
u_2 mean wind speed at 2m's height [m/s]
e_s saturation vapour pressure [kPa]
e_a actual vapour pressure [kPa]

The ETo was calculated on a monthly basis, i.e. always for the 15th day of the month.

The stress factor is calculated with Equation 9.6.2.

The mean temperature, maximum and minimum temperature, dew point temperature and the average wind speed are derived from METEOTEST (2011). In Table 9.6.2 the crops and the corresponding crop coefficients K_c are listed. Crop yield response factors and stress factors are shown in Table 9.6.5.

In total, the crop evapotranspiration via Penman-Monteith $ET_{c,\ adj}$ varies between 420 mm and 676 mm depending on the crop coefficients K_c and the K_s value (see Table 9.6.9). The non-adjusted ET_c from Penman-Monteith is 1040 mm, which is the evapotranspiration of crops under "standard" conditions, meaning excellently managed and well-watered conditions. The crop evapotranspiration in the so called "Turc" scenarios are calculated as the difference between the applied irrigation water and the calculated recharge to the groundwater (Equation 9.6.4). In the Turc scenario $ET_{c,adj}$ varies between 625 mm and 785 mm. Unproductive ET is not directly accounted for.

Table 9.6.9: Crop evapotranspiration for the different scenarios and base cases. The crop evapotranspiration in the Turc scenarios is calculated as difference from irrigation amount and groundwater recharge, while the other crop's evapotranspiration are derived with the method by Penman-Monteith.

	Crop evapotranspiration ET_c [mm/a], base case 2010					
Scenario	K_{s1} , K_{cFAO}	K_{s1} , $K_{celRiego}$	K_{s2} , K_{cFAO}	K_{s2} , $K_{celRiego}$	Turc,K_{cFAO}	Turc, $K_{celRiego}$
Increase Asparagus	676	625	487	449	758	625
Increase Mandarin	660	605	461	420	745	744
Increase Strawberry	673	617	469	429	756	754
Increase Drip-Irrigation	613	572	455	423	651	657
Increase Multi-crop	676	633	474	442	751	758
Increase Mandarin, more GW	660	605	461	420	745	744
NO irrigation	662	616	467	432	785	785
	Crop evapotranspiration ET_c [mm/a], base case "no irrigation"					
Scenario	K_{s1} , K_{cFAO}	K_{s1} , $K_{celRiego}$	K_{s2} , K_{cFAO}	K_{s2} , $K_{celRiego}$	Turc, K_{cFAO}	Turc, $K_{celRiego}$
Increase Asparagus	676	625	487	449	758	625
Increase Mandarin	660	605	461	420	745	744
Increase Strawberry	673	617	469	429	756	754
Increase Drip-Irrigation	613	572	455	423	656	657
Increase Multi-crop	676	633	474	442	751	758
Increase Mandarin, more GW	660	605	461	420	745	744
2010	672	617	470	429	756	754

9.6.9. Procedure for calculating characterization factors

Subtracting the groundwater recharge (GWR_S and GWR_I) and extraction (GWW) of each scenario and the base case indicates the difference in available groundwater volume, assuming constant discharge to the sea. This amount is distributed over the whole 56'874 ha of aquifer, as it has a high transmissivity (Qh-c, extensive aquifer, see Figure 9.6.2) and groundwater extractions and groundwater recharge through excess irrigation water are distributed over the whole aquifer. Recharge from the higher valley and the river is more concentrated, however, since the soil and aquifer show a rather high permeability the recharged water will distribute over a larger area within the aquifer quickly.

The wetland Santa Rosa is assumed to be shaped like a circular cone with a surface area (A) of 36 ha, the maximum depth (h) of 3 m being the depth at the center of the model in the year 2010. The total radius (r) in the base year 2010 is consequently 338.5 m (see also Figure 9.6.5). For the "no irrigation" base case the area is assumed to be 34.3 ha, the total radius is 330.3 m and the maximum depth h is 2.9 m (see Equation

9.6.13). This is based on the calculation of the "no irrigation" scenario, which indicates that without irrigation the area would be 4.8% smaller.

$$h = -$$

Equation 9.6.13

The infiltration area (A_{inf}), over which new water can infiltrate to the wetland is equal to the lateral surface of the circular cone and is thus calculated as in Equation 9.6.14.

$$= \pi \cdot r \cdot$$

Equation 9.6.14

The angle alpha between the depth (h) and the side slope is calculated via trigonometry to match the depth as 89.5° and is assumed to be constant for both base cases.

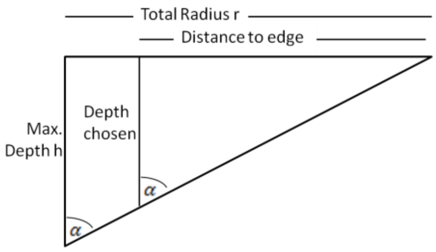

Figure 9.6.5: Half of the idealized circular cone, which is used to model the wetland. The total radius r and maximum depth h of 2010 are used for calculating the infiltration capacity. Since this is essentially a material constant it is assumed to be the same for the "no irrigation" base case. The distance to edge and depth chosen are used to estimate the areas of the four vegetation categories.

From the estimated evaporation for the wetland ($ET_{Santa\ Rosa}$), the amount of water that has to be replaced annually is calculated. If the water situation in the area stays the same, the wetland is neither growing nor shrinking and thus the infiltration capacity stays, also due to geological reasons, the same. From the lateral surface and the necessary amount of water to conserve the wetland ($ET_{Santa\ Rosa}$) the infiltration capacity (IC) was calculated as 0.9 m/a on each square meter of the lateral area and kept constant in all scenarios. Basically, the valid formula for this is shown in Equation 9.6.15

$$A \cdot = IC \cdot$$

Equation 9.6.15

The groundwater level which is changing around the wetland is changing the area of the lateral surface which is in direct contact with the aquifer and contributes to recharging the wetland (A_{inf}). Thus the amount of water which is infiltrating changes and this leads to a change in the water level in the wetland itself. With the changed water level the new total area (A) of the wetland and the lost or gained area can be estimated.

The wetland was separated into an aquatic and a terrestrial zone, given that there are species for both classifications. It is not always clear which species is to be considered as aquatic and which as terrestrial. We adopted the terrestrial or aquatic classification of Jiménez et al. (2009)[31] for the purpose of separating them into two classes, except for *Typha dominguensis*. This species is the one rooted species which grows furthest from the shore in Santa Rosa and we considered its maximum growth depth of 1.5 m as the boundary between terrestrial (rooted) and aquatic habitat. Therefore the terrestrial habitat is considered going from the edge of the wetland up to 1.5 m depth, the remainder from 1.5 to 3 m being the aquatic habitat.

The modeled areas did not match exactly with the estimated areas and therefore scaling factors are required, as shown in Table 9.6.10.

Table 9.6.10: The scaling factor is used to transform the modeled areas in the scenarios back to observed areas.

	Scaling factor calculated/estimated	
Habitat type	Base case „2010“	Base case „no irrigation“
Aquatic habitat	0.9	0.85
Terrestrial habitat	1.03	1.05

For each scenario the area which is lost compared to the base year is calculated and used in the calculation of the characterization factor (see main document).

The species-area relationship (Equation 9.6.16), which forms part of the effect factor calculation, is applied with the commonly used z-value of 0.25 for terrestrial ecosystems (Brooks et al. 2002; Thomas et al., 2004; Rosenzweig, 1995). A positive relationship between the size of the waterbody and the aquatic plant species richness is not necessarily given. Oertli et al. (2002) mention that for some species (among them one each of the genus *Schoenoplectus* and *Typha*, although not the same species as occurring in Santa Rosa) no significant relationship to pond size could be found. However, they still conclude that “for aquatic plants, a positive relationship between pond size and species richness could be a valid generalization”. Jones et al. (2003) found that the area of a waterbody is one of the best predictors of aquatic plant species richness. Also, a correlation between size and depth of a waterbody has been found (Jones et al., 2003). Plants usually grow along specific depth gradients (Spence, 1982) and our plants grow in distinct depth categories in Santa Rosa. Since there is evidence that the depth is correlated with the size of a waterbody and supported by the findings of Jones et al. (2003) and the generalization in Oertli et al. (2002), we assumed that the species-area relationship can be applied for the aquatic flora as well. In general, reported z values for aquatic plants and water bodies are smaller than for terrestrial plants (Oertli et al. 2002; Vestergaard et al., 2000; Archibald, 1949; Weiher, 1999). The reported values vary widely among different publications, e.g. between 0.07-0.13 (Oertli et al. 2002),0.27-0.32 (Møller et al., 1985), 0.03-0.08 (Archibald, 1949), 0.173±0.018 (Jones et al. 2003), and 0.1-0.34 (Weiher, 1999) for aquatic vascular plants. For the aquatic plants we applied the average of the reported z-values as a proxy as no location-specific value was available. The average value among the reported values is 0.2.

$$S = c \cdot A^z$$

Equation 9.6.16

Where S is number of species, c is a constant, A is the relevant area and z is the slope of the species-are relationship.

The Species number $S_{scenario}$ is calculated based on the area in the respective scenario (Equation 9.6.17) and the original species number. The constant c is thus irrelevant. The original species number in 2010 is 37 terrestrial and 14 aquatic species. For the base case “no irrigation” the species number for terrestrial and aquatic plant zones are adapted according to the species-area relationship (analogous to Equation 9.6.17). Thus, the original species number for terrestrial plants is 36.5 and for aquatic plants 13.9.

$$S_{scenario} = S_{base\ year} \cdot \left(\frac{A_{scenario}}{A_{base\ year}}\right)^{z}$$

Equation 9.6.17

9.6.10. Sensitivity and uncertainty

Several procedures have been applied for estimating groundwater recharge from irrigation. One approach is to apply the difference between evapotranspiration and irrigation water with different K_s values. The base value is 0.5 on average ($K_{s,1}$), while the one approximated in the sensitivity analysis is 0.45, as calculated with Equation 9.6.1. Also, there is an empirical formula from Turc[25] which was used in the sensitivity analysis, substituting irrigation water for the precipitation in the formula.

The sensitivity of the z-value is tested with increasing and decreasing the value for both terrestrial and aquatic habitat. From the original values of 0.25 and 0.2 for terrestrial and aquatic habitat, respectively, the z-value was once changed to 0.34 for both values and once to 0.03 for both values. These two are the minimum and maximum of the reported z-values from the literature cited above (for aquatic plants, assuming the same values for the terrestrial species here).

9.6.11. Water balance results

The results for the water balance for the year 2010 were calculated with six different combinations of Ks and Kc values and are shown in Table 9.6.11. The values used for the standard case are $K_{s,1}$ and $K_{c,FAO}$. The inflows of river and groundwater, as well as the precipitation and evaporation stay the same throughout the balances. The differences are due to the different K_s values and crop coefficients, as discussed in section 5 and shown in Table 9.6.2. These changes lead to differences in surface stream flow going to the Ocean, varying between 11.4 m^3/s (Turc, $K_{c,FAO}$) and 14.1 m^3/s ($K_{s,2}$, $K_{c,ElRiego.com}$), i.e. the variation is approximately 20%.

Table 9.6.11: Results of the six water balances for the year 2010. The differences between the results are due to different crop coefficients and hence different evapotranspiration and groundwater recharge from irrigation water. Santo Domingo is the measuring station which denotes the initial point of the water balance model. GW stands for groundwater. The unit of the flows is Mio m^3/a.

	$K_{s,1}$, $K_{c,FAO}$	$K_{s,1}$, $K_{c,ElRiego.com}$	$K_{s,2}$, $K_{c,FAO}$	$K_{s,2}$, $K_{c,ElRiego.com}$	Turc, $K_{c,FAO}$	Turc, $K_{c,ElRiego.com}$
Precipitation (Santo Domingo-coast)	40.37	40.37	40.37	40.37	40.37	40.37
Evaporation (soil, Santo Domingo-coast)	35.06	35.06	35.06	35.06	35.06	35.06
Evaporation (river, Santo Domingo-coast)	1.80	1.80	1.80	1.80	1.80	1.80
Evaporation (Santa Rosa)	0.32	0.32	0.32	0.32	0.32	0.32
groundwater recharge (irrigation)	72.31	86.57	124.36	134.81	50.79	51.22
groundwater recharge (until Santo Domingo)	39.51	39.51	39.51	39.51	39.51	39.51
groundwater recharge (Santo Domingo-coast, seepage)	18.30	18.30	18.30	18.30	18.30	18.30
water consumption agriculture (river) (incl crop ET)	243.00	243.00	243.00	243.00	243.00	243.00
water consumption agriculture (GW) (incl. crop ET)	2.05	2.05	2.05	2.05	2.05	2.05
water consumption livestock (river)	0.01	0.01	0.01	0.01	0.01	0.01
water consumption livestock (GW)	0.67	0.67	0.67	0.67	0.67	0.67
water consumption industry (river)	0.00	0.00	0.00	0.00	0.00	0.00
water consumption industry (GW)	0.55	0.55	0.55	0.55	0.55	0.55
water consumption domestic (river)	0.91	0.91	0.91	0.91	0.91	0.91
water consumption domestic (GW)	0.04	0.04	0.04	0.04	0.04	0.04
water consumption mining (river)	0.02	0.02	0.02	0.02	0.02	0.02
water return flow to river from domestic and industrial GW use	0.47	0.47	0.47	0.47	0.47	0.47
river discharge into area	508.06	508.06	508.06	508.06	508.06	508.06
exfiltration at coast	12.60	12.60	12.60	12.60	12.60	12.60
TOTAL (=river discharge into Ocean)	381.98	396.24	434.03	444.48	360.46	360.89

9.6.12. Scenario and sensitivity results

The scenarios have been calculated for two base cases: base case 2010 and base case “no irrigation”. For each scenario different crop coefficients, water use efficiencies and consequently different groundwater recharges from irrigation are available. Table 9.6.12 and Table 9.6.13 show all results for the scenarios from total agricultural water use, change in overall groundwater level and water level in Santa Rosa, as well as the calculated impacts and characterization factors. In all scenarios except “Increase Mandarin, more GW” the lion’s share of withdrawn water is coming from the river. In “Increase Mandarin, more GW” 25% of the required water is abstracted from the groundwater. However, this is considered unrealistic based on the insights from the stakeholder workshop.

Table 9.6.12: Total results for the scenarios with base case “2010” and the adapted CF values from Hanafiah et al. (2011). GW stands for groundwater.

Scenario	K_{s1}, K_{cFAO}	K_{s1}, $K_{celRiego}$	K_{s2}, K_{cFAO}	K_{s2}, $K_{celRiego}$	Turc, K_{cFAO}	Turc, $K_{celRiego}$
	New total agricultural water use [Mio m^3/a]					
Increase Asparagus	246'111'830	247'788'336	246'111'830	247'788'336	246'111'830	247'788'336
Increase Mandarin	239'504'580	239'907'595	239'504'580	239'907'595	239'504'580	239'907'595
Increase Strawberry	244'857'939	244'767'260	244'857'939	244'767'260	244'857'939	244'767'260
Increase Drip-Irrigation	217'364'358	219'483'165	217'364'358	219'483'165	217'364'358	219'483'165
Increase Multi-crop	241'620'454	247'276'589	241'620'454	247'276'589	241'620'454	247'276'589
Increase Mandarin, more GW	239'504'580	239'907'595	239'504'580	239'907'595	239'504'580	239'907'595
NO irrigation	0	0	0	0	0	0
	Change GW level aquifer [m]					
Increase Asparagus	0.0001	0.0012	-0.0060	-0.0042	0.0010	0.0029
Increase Mandarin	-0.0043	-0.0038	-0.0056	-0.0050	-0.0049	-0.0036
Increase Strawberry	-0.0006	-0.0008	-0.0001	-0.0003	-0.0002	-0.0003
Increase Drip-Irrigation	-0.0308	-0.0248	-0.0418	-0.0424	-0.0188	-0.0156
Increase Multi-crop	-0.0076	-0.0035	-0.0080	-0.0018	-0.0036	0.0025
Increase Mandarin, more GW	-0.1060	-0.1056	-0.1073	-0.1068	-0.1066	-0.1063
NO irrigation	-0.1235	-0.1494	-0.2159	-0.2343	-0.0865	-0.0873
	Change GW level Santa Rosa [m]					
Increase Asparagus	0.0001	0.0007	-0.0036	-0.0025	0.0006	0.0017
Increase Mandarin	-0.0026	-0.0023	-0.0034	-0.0030	-0.0029	-0.0022
Increase Strawberry	-0.0004	-0.0005	-0.0001	-0.0002	-0.0001	-0.0002
Increase Drip-Irrigation	-0.0183	-0.0147	-0.0247	-0.0250	-0.0112	-0.0093
Increase Multi-crop	-0.0045	-0.0021	-0.0048	-0.0011	-0.0021	0.0015
Increase Mandarin, more GW	-0.0620	-0.0618	-0.0627	-0.0625	-0.0623	-0.0622
NO irrigation	-0.0720	-0.0867	-0.1239	-0.1340	-0.0508	-0.0512
	Lost (negative) or won (positive) area [%]					
Increase Asparagus	0.004	0.048	-0.238	-0.166	0.040	0.116
Increase Mandarin	-0.172	-0.151	-0.224	-0.198	-0.196	-0.145
Increase Strawberry	-0.026	-0.030	-0.005	-0.011	-0.009	-0.012
Increase Drip-Irrigation	-1.214	-0.977	-1.640	-1.662	-0.742	-0.616
Increase Multi-crop	-0.300	-0.137	-0.318	-0.070	-0.143	0.099
Increase Mandarin, more GW	-4.090	-4.077	-4.140	-4.122	-4.113	-4.102
NO irrigation	-4.745	-5.699	-8.089	-8.735	-3.356	-3.385
	Modeled impact on terrestrial ecosystem [PDF·m^2·yr]					
Increase Asparagus	-9	-104	517	360	-87	-252
Increase Mandarin	373	327	486	430	425	314
Increase Strawberry	56	66	11	24	19	25
Increase Drip-Irrigation	2'641	2'124	3'572	3'619	1'614	1'340
Increase Multi-crop	652	299	690	152	311	-215
Increase Mandarin, more GW	8'944	8'914	9'052	9'012	8'994	8'970
NO irrigation	10'386	12'430	17'745	19'189	7'259	7'323
	Modeled impact on aquatic ecosystem [PDF·m^2·yr]					
Increase Asparagus	-10	-115	571	398	-96	-278
Increase Mandarin	412	361	537	475	470	347
Increase Strawberry	62	73	12	26	21	28
Increase Drip-Irrigation	2'915	2'345	3'940	3'992	1'782	1'480
Increase Multi-crop	720	330	763	168	343	-238

Increase Mandarin, more GW	9'835	9'803	9'954	9'911	9'890	9'864
NO irrigation	11'412	13'641	19'409	20'970	7'991	8'061
	Modeled impact total [PDF·m²·yr]					
Increase Asparagus	-20	-219	1'088	757	-182	-530
Increase Mandarin	785	689	1'023	904	895	661
Increase Strawberry	118	139	24	50	41	53
Increase Drip-Irrigation	5'557	4'469	7'512	7'612	3'396	2'819
Increase Multi-crop	1'372	629	1'453	320	654	-453
Increase Mandarin, more GW	18'779	18'717	19'006	18'923	18'883	18'834
NO irrigation	21'797	26'070	37'154	40'159	15'250	15'385
	$CF_{wetland}$, terrestrial [PDF·m²·yr/m³]					
Increase Asparagus	-1.52E-04	-1.50E-04	-1.51E-04	-1.51E-04	-1.52E-04	-1.52E-04
Increase Mandarin	-1.51E-04	-1.52E-04	-1.51E-04	-1.51E-04	-1.51E-04	-1.23E-04
Increase Strawberry	-1.52E-04	-1.53E-04	-1.52E-04	-1.52E-04	-1.52E-04	-1.52E-04
Increase Drip-Irrigation	-2.12E-04	-1.51E-04	-1.50E-04	-1.50E-04	-7.16E-04	-2.55E-03
Increase Multi-crop	-1.51E-04	-1.52E-04	-1.51E-04	-1.51E-04	-1.51E-04	-1.52E-04
Increase Mandarin, more GW	-1.48E-04	-1.48E-04	-1.48E-04	-1.48E-04	-1.48E-04	-1.48E-04
NO irrigation	-1.48E-04	-1.47E-04	-1.45E-04	-1.45E-04	-1.49E-04	-1.49E-04
	$CF_{wetland}$, aquatic [PDF·m²·yr/m³]					
Increase Asparagus	-1.67E-04	-1.66E-04	-1.67E-04	-1.67E-04	-1.68E-04	-1.68E-04
Increase Mandarin	-1.67E-04	-1.68E-04	-1.67E-04	-1.67E-04	-1.67E-04	-1.36E-04
Increase Strawberry	-1.67E-04	-1.70E-04	-1.67E-04	-1.67E-04	-1.67E-04	-1.67E-04
Increase Drip-Irrigation	-2.34E-04	-1.67E-04	-1.66E-04	-1.66E-04	-7.91E-04	-2.82E-03
Increase Multi-crop	-1.67E-04	-1.68E-04	-1.67E-04	-1.67E-04	-1.67E-04	-1.68E-04
Increase Mandarin, more GW	-1.63E-04	-1.63E-04	-1.63E-04	-1.63E-04	-1.63E-04	-1.63E-04
NO irrigation	-1.62E-04	-1.61E-04	-1.59E-04	-1.58E-04	-1.64E-04	-1.64E-04
	$CF_{wetland}$, total [PDF·m²·yr/m³]					
Increase Asparagus	-3.19E-04	-3.17E-04	-3.19E-04	-3.19E-04	-3.19E-04	-3.19E-04
Increase Mandarin	-3.19E-04	-3.20E-04	-3.19E-04	-3.19E-04	-3.19E-04	-2.60E-04
Increase Strawberry	-3.19E-04	-3.23E-04	-3.19E-04	-3.19E-04	-3.19E-04	-3.19E-04
Increase Drip-Irrigation	-4.45E-04	-3.17E-04	-3.16E-04	-3.16E-04	-1.51E-03	-5.37E-03
Increase Multi-crop	-3.18E-04	-3.20E-04	-3.18E-04	-3.19E-04	-3.19E-04	-3.19E-04
Increase Mandarin, more GW	-3.11E-04	-3.12E-04	-3.11E-04	-3.11E-04	-3.11E-04	-3.11E-04
NO irrigation	-3.10E-04	-3.08E-04	-3.04E-04	-3.02E-04	-3.13E-04	-3.13E-04
	CF_{river} (adapted from Hanafiah et al.[42]) [PDF·m³·yr/m³][a)]					
Increase Asparagus	2.96E-04	2.95E-04	2.92E-04	2.92E-04	2.98E-04	2.98E-04
Increase Mandarin	2.96E-04	2.95E-04	2.92E-04	2.91E-04	2.98E-04	2.97E-04
Increase Strawberry	2.96E-04	2.95E-04	2.92E-04	2.91E-04	2.98E-04	2.98E-04
Increase Drip-Irrigation	2.95E-04	2.94E-04	2.92E-04	2.91E-04	2.96E-04	2.96E-04
Increase Multi-crop	2.96E-04	2.95E-04	2.92E-04	2.92E-04	2.98E-04	2.98E-04
Increase Mandarin, more GW	2.96E-04	2.95E-04	2.92E-04	2.91E-04	2.98E-04	2.97E-04
NO irrigation	8.76E-08	8.70E-08	8.54E-08	8.50E-08	8.87E-08	8.87E-08

a)CF_{river} is varying since the discharge to the Sea (Q_{mouth} in Hanafiah et al.2011) is varying, depending on the chosen scenario and coefficients.

Table 9.6.13: Total results for the scenarios with base case "no irrigation" and the adapted CF values from Hanafiah et al.[42] GW stands for groundwater.

Scenario	K_{s1} , K_{cFAO}	K_{s1} , $K_{celRiego}$	K_{s2} , K_{cFAO}	K_{s2} , $K_{celRiego}$	Turc , K_{cFAO}	Turc, $K_{celRiego}$
	New total agricultural water use [Mio m3/a]					
Increase Asparagus	246'111'830	247'788'336	246'111'830	247'788'336	246'111'830	247'788'336
Increase Mandarin	239'504'580	239'907'595	239'504'580	239'907'595	239'504'580	239'907'595
Increase Strawberry	244'857'939	244'767'260	244'857'939	244'767'260	244'857'939	244'767'260
Increase Drip-Irrigation	217'364'358	219'483'165	217'364'358	219'483'165	217'364'358	219'483'165
Increase Multi-crop	241'620'454	247'276'589	241'620'454	247'276'589	241'620'454	247'276'589
Increase Mandarin, more GW	239'504'580	239'907'595	239'504'580	239'907'595	239'504'580	239'907'595
2010	245'050'000	245'050'000	245'050'000	245'050'000	245'050'000	245'050'000
	Change GW level aquifer [m]					
Increase Asparagus	0.124	0.150	0.209	0.229	0.087	0.089
Increase Mandarin	0.119	0.145	0.209	0.228	0.081	0.083
Increase Strawberry	0.123	0.148	0.215	0.233	0.085	0.086
Increase Drip-Irrigation	0.113	0.124	0.170	0.191	0.067	0.071
Increase Multi-crop	0.116	0.145	0.207	0.232	0.082	0.089
Increase Mandarin, more GW	0.018	0.043	0.108	0.127	-0.021	-0.020
2010	0.124	0.149	0.215	0.233	0.086	0.086
	Change GW level Santa Rosa [m]					

Increase Asparagus	0.077	0.094	0.132	0.145	0.054	0.055
Increase Mandarin	0.074	0.091	0.132	0.145	0.050	0.051
Increase Strawberry	0.077	0.092	0.136	0.148	0.053	0.053
Increase Drip-Irrigation	0.070	0.077	0.107	0.120	0.041	0.044
Increase Multi-crop	0.072	0.091	0.131	0.147	0.051	0.055
Increase Mandarin, more GW	0.011	0.026	0.067	0.079	-0.013	-0.012
2010	0.077	0.093	0.136	0.148	0.053	0.054
	Lost (negative) or won (positive) area [%]					
Increase Asparagus	5.08	6.20	8.79	9.69	3.52	3.64
Increase Mandarin	4.89	5.98	8.80	9.65	3.28	3.36
Increase Strawberry	5.04	6.11	9.05	9.87	3.47	3.50
Increase Drip-Irrigation	4.62	5.08	7.07	7.99	2.70	2.87
Increase Multi-crop	4.75	5.99	8.70	9.80	3.33	3.62
Increase Mandarin, more GW	0.70	1.73	4.40	5.20	-0.83	-0.78
2010	5.07	6.14	9.05	9.88	3.48	3.51
	Modeled impact on terrestrial ecosystem [$PDF \cdot m^2 \cdot yr$]					
Increase Asparagus	-11'022	-13'425	-18'958	-20'875	-7'670	-7'912
Increase Mandarin	-10'616	-12'963	-18'992	-20'798	-7'136	-7'321
Increase Strawberry	-10'952	-13'243	-19'514	-21'247	-7'559	-7'622
Increase Drip-Irrigation	-10'047	-11'039	-15'299	-17'266	-5'894	-6'250
Increase Multi-crop	-10'320	-12'994	-18'768	-21'105	-7'255	-7'874
Increase Mandarin, more GW	-1'529	-3'766	-9'572	-11'290	1'810	1'719
2010	-11'012	-13'313	-19'526	-21'274	-7'580	-7'649
	Modeled impact on aquatic ecosystem [$PDF \cdot m^2 \cdot yr$]					
Increase Asparagus	-13'141	-16'028	-22'702	-25'023	-9'127	-9'416
Increase Mandarin	-12'655	-15'473	-22'743	-24'929	-8'489	-8'710
Increase Strawberry	-13'058	-15'809	-23'375	-25'474	-8'995	-9'070
Increase Drip-Irrigation	-11'973	-13'162	-18'284	-20'657	-7'007	-7'431
Increase Multi-crop	-12'300	-15'510	-22'472	-25'302	-8'632	-9'371
Increase Mandarin, more GW	-1'813	-4'471	-11'403	-13'464	2'142	2'034
2010	-13'129	-15'894	-23'389	-25'506	-9'019	-9'102
	Modeled impact total [$PDF \cdot m^2 \cdot yr$]					
Increase Asparagus	-24'163	-29'453	-41'661	-45'898	-16'797	-17'328
Increase Mandarin	-23'271	-28'436	-41'735	-45'727	-15'625	-16'030
Increase Strawberry	-24'010	-29'052	-42'889	-46'721	-16'554	-16'693
Increase Drip-Irrigation	-22'020	-24'200	-33'582	-37'923	-12'902	-13'680
Increase Multi-crop	-22'621	-28'503	-41'239	-46'407	-15'887	-17'244
Increase Mandarin, more GW	-3'342	-8'238	-20'974	-24'754	3'952	3'754
2010	-24'141	-29'207	-42'916	-46'779	-16'599	-16'751
	$CF_{wetland}$, terrestrial [$PDF \cdot m^2 \cdot yr/m^3$]					
Increase Asparagus	-1.57E-04	-1.58E-04	-1.59E-04	-1.60E-04	-1.56E-04	-1.56E-04
Increase Mandarin	-1.57E-04	-1.57E-04	-1.59E-04	-1.60E-04	-1.55E-04	-1.57E-04
Increase Strawberry	-1.57E-04	-1.58E-04	-1.60E-04	-1.60E-04	-1.56E-04	-1.56E-04
Increase Drip-Irrigation	-1.74E-04	-1.57E-04	-1.55E-04	-1.59E-04	-1.27E-04	-1.28E-04
Increase Multi-crop	-1.56E-04	-1.57E-04	-1.59E-04	-1.60E-04	-1.55E-04	-1.56E-04
Increase Mandarin, more GW	-1.53E-04	-1.54E-04	-1.56E-04	-1.57E-04	-1.52E-04	-1.52E-04
2010	-1.57E-04	-1.58E-04	-1.60E-04	-1.60E-04	-1.56E-04	-1.56E-04
	$CF_{wetland}$, aquatic [$PDF \cdot m^2 \cdot yr/m^3$]					
Increase Asparagus	-1.87E-04	-1.88E-04	-1.91E-04	-1.92E-04	-1.85E-04	-1.85E-04
Increase Mandarin	-1.87E-04	-1.88E-04	-1.91E-04	-1.92E-04	-1.85E-04	-1.87E-04
Increase Strawberry	-1.87E-04	-1.88E-04	-1.91E-04	-1.92E-04	-1.85E-04	-1.85E-04
Increase Drip-Irrigation	-2.07E-04	-1.87E-04	-1.86E-04	-1.90E-04	-1.51E-04	-1.53E-04
Increase Multi-crop	-1.87E-04	-1.88E-04	-1.91E-04	-1.92E-04	-1.85E-04	-1.85E-04
Increase Mandarin, more GW	-1.82E-04	-1.83E-04	-1.86E-04	-1.87E-04	-1.80E-04	-1.80E-04
2010	-1.87E-04	-1.88E-04	-1.91E-04	-1.92E-04	-1.85E-04	-1.85E-04
	$CF_{wetland}$, total [$PDF \cdot m^2 \cdot yr/m^3$]					
Increase Asparagus	-3.44E-04	-3.46E-04	-3.50E-04	-3.52E-04	-3.41E-04	-3.41E-04
Increase Mandarin	-3.43E-04	-3.45E-04	-3.50E-04	-3.52E-04	-3.40E-04	-3.44E-04
Increase Strawberry	-3.44E-04	-3.46E-04	-3.51E-04	-3.52E-04	-3.41E-04	-3.41E-04
Increase Drip-Irrigation	-3.81E-04	-2.69E-04	-3.41E-04	-3.49E-04	-2.78E-04	-2.81E-04
Increase Multi-crop	-3.43E-04	-3.45E-04	-3.50E-04	-3.52E-04	-3.40E-04	-3.41E-04
Increase Mandarin, more GW	-3.35E-04	-3.37E-04	-3.42E-04	-3.44E-04	-3.32E-04	-3.32E-04
2010	-3.44E-04	-3.46E-04	-3.51E-04	-3.52E-04	-3.41E-04	-3.41E-04
	CF_{river} (adapted from Hanafiah et al.[37]) [$PDF \cdot m^3 \cdot yr/m^3$][a)]					
Increase Asparagus	2.96E-04	2.95E-04	2.92E-04	2.92E-04	2.98E-04	2.98E-04
Increase Mandarin	2.96E-04	2.95E-04	2.92E-04	2.91E-04	2.98E-04	2.97E-04
Increase Strawberry	2.96E-04	2.95E-04	2.92E-04	2.91E-04	2.98E-04	2.98E-04

Increase Drip-Irrigation	2.94E-04	2.94E-04	2.92E-04	2.91E-04	2.96E-04	2.96E-04
Increase Multi-crop	2.96E-04	2.95E-04	2.92E-04	2.92E-04	2.98E-04	2.98E-04
Increase Mandarin, more GW	2.96E-04	2.95E-04	2.92E-04	2.91E-04	2.98E-04	2.97E-04
2010	2.96E-04	2.95E-04	2.92E-04	2.91E-04	2.98E-04	2.98E-04

a)CF_{river} is varying since the discharge to the Sea (Q_{mouth} in Hanafiah et al.[42]) is varying, depending on the chosen scenario and coefficients.

Table 9.6.14 shows the sensitivity of the scenarios to changes in the base case (2010 and "no irrigation"), changes in K_c values ($K_{c,FAO}$ and $K_{c,ElRiego}$) and changes in K_s values ($K_{s,1}$ and $K_{s,2}$) in percent of change between the respective CF values.

Table 9.6.14: Sensitivities due to changes in K_c ($K_{c,FAO}$ to $K_{c,ElRiego}$) values and K_s values ($K_{s,1}$ to $K_{s,2}$), and application of the minimal and maximal z-value of the species-area relationship, respectively. Changes in result with different the base cases ("2010" to "no irrigation") are shown in % of change in the result as well., Sensitivities (in % change of the respective CF) are always related to the scenarios with base case "2010", $K_{s,1}$ and $K_{c,FAO}$. "cons" stands for the consumptive CF.

	Changes in…				
	base case [%]	Kc values [%]	Ks value [%]	z-value, increase [%]	z-value, decrease [%]
Increase Asparagus	7.7	-0.7	-0.1	9.5	-15.8
Increase Mandarin	7.7	0.3	0.0	9.5	-15.8
Increase Strawberry	7.7	1.2	0.0	9.5	-15.8
Increase Drip-Irrigation	-14.4	-28.7	-29.0	9.4	-15.7
Increase Multi-crop	7.7	0.4	0.0	9.5	-15.8
Increase Mandarin, more GW	7.6	0.0	0.0	9.1	-15.4
NO irrigation	10.8	-0.6	-2.1	9.1	-15.3

All sensitivities are relatively small. The average change of the K_c and K_s values for all crops was 12%. The terrestrial z-value is changed from 0.25 to 0.34 and 0.03, the aquatic z-value from 0.2 to 0.34 and 0.03. Due to the larger decrease from 0.25 to 0.03 in the terrestrial z-value, compared to the z-value increase, the change in the resulting CF is larger. For the aquatic z-value the changes are more similar (0.2 to 0.34 and 0.2 to 0.03), thus the contribution to result changes in both cases more alike.

From two aerial pictures, once from Google Earth (2009) for the year 2006 and once from an aerial picture of the Instituto Geográfico Nacional from 1961 the approximate size change of Santa Rosa can be estimated after geo-referencing them in ArcGIS (ESRI, 2009) (Figure 9.6.6 and Figure 9.6.7). Since in both pictures the exact dimensions are not totally clear two values were determined after georectifying both images in ArcGIS and measuring their areas with a polygon. It seems that Santa Rosa has grown by 5.5% to 8.9% since 1961.

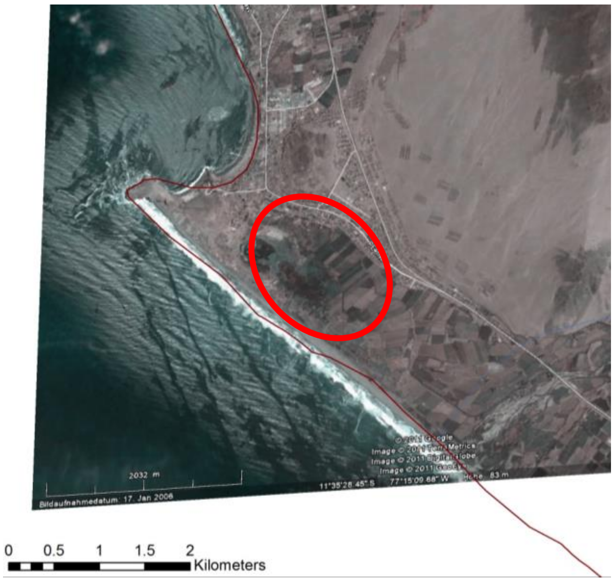

Figure 9.6.6: Overview of the area around Santa Rosa from Google Earth. The distortion is due to the geo-referencing in ArcGIS. The brown line indicates the coastal line and the light blue line is the river Chancay-Huaral. Santa Rosa is located in the red circle.

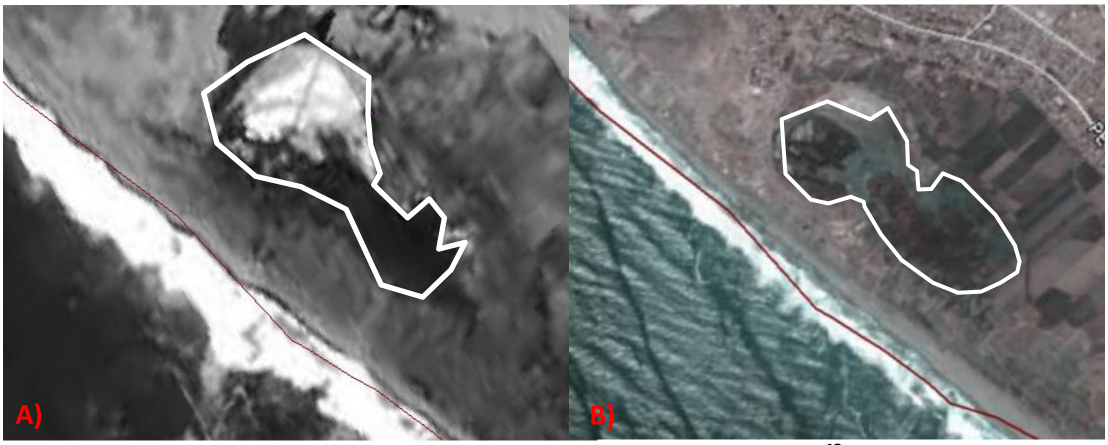

Figure 9.6.7: A) Geo-referenced aerial picture of Santa Rosa of 1961.[43] B) Geo-referenced remote sensed picture of Santa Rosa from 2006.[23] In both pictures the coast is shown with the light brown line. The white line denotes the approximate area of the wetland.

9.6.13. Comparison with other LCA results

The hierarchist perspective, average weighting, and European normalization of the ReCiPe-method (Goedkoop et al., 2009) was used to assess the damage in eco-points, comparing the impacts of water use and consumption of asparagus and mandarin production to other LCA based impacts. The comparison is based on data from Stoessel et al. (2012) and some minor changes to the data of Asparagus and Citrus were applied. The transport distance was shortened to 100km and no air or ship transports were included, since the largest part of the products is transported to Lima (ca. 60 km away) and consumed there. The size of the lorry was slightly decreased from >32t to 16-32t. For irrigation values the calculated generalized characterization factors from the main paper (transformed to ReCiPe points) were applied. Asparagus are assumed to be produced once with traditional irrigation and once with drip irrigation. For the cultivation of mandarins two traditional irrigation options with different water sources, either fully irrigated with river water (RW) or fully irrigated with groundwater (GW), are presented. The water-related inventory used is presented in Table 6.3. It is calculated based on the water amount allocated (Table 9.6.2), yields and planted area from the local government (Gobierno Regional de Lima, 2001a), as well as calculated evapotranspiration (Penman-Monteith equation) from the scenario $K_{s,2}$ and $K_{c,FAO}$. The results for 1 kg crop under different management options are shown in Table 9.6.15. The impacts are dominated by some few categories such as climate change and agricultural land occupation. Considering only ecosystem quality, agricultural land occupation constitutes 80 to 90 % of the impact, followed by climate change with 10 to 16%. However, it has to be noted, that the different impacts affecting the same area of protection have different underlying assumptions and direct comparison should be assessed with caution.

Table 9.6.15: Adapted results based on Stoessel et al.(2012) for asparagus and mandarins with different management options. Water impacts on Santa Rosa and the impact on the river (in italics) are the factors which are calculated based on the generalized CF factors and the inventory as presented in the main document. The first four columns show the results in ReCiPe points, the latter four show the contribution of each impact category to the total impact.

		Asparagus, no drip [Pts]	Asparagus, drip [Pts]	Mandarins, RW [Pts]	Mandarins, GW [Pts]	Asparagus, no drip [%]	Asparagus, drip [%]	Mandarins, RW [%]	Mandarins, GW [%]
Human Health	Climate change Human Health	3.58E-02	3.58E-02	4.49E-03	4.49E-03	31.76	31.76	31.70	31.69
	Ozone depletion	9.69E-06	9.69E-06	9.42E-07	9.42E-07	0.01	0.01	0.01	0.01
	Human toxicity	3.14E-03	3.14E-03	9.34E-04	9.34E-04	2.79	2.79	6.59	6.59
	Photochemical oxidant formation	4.57E-06	4.57E-06	6.55E-07	6.55E-07	0.00	0.00	0.00	0.00
	Particulate matter formation	1.82E-02	1.82E-02	1.96E-03	1.96E-03	16.12	16.12	13.82	13.81
	Ionising radiation	2.92E-05	2.92E-05	8.96E-06	8.96E-06	0.03	0.03	0.06	0.06
Ecosystem Quality	Climate change Ecosystems	3.18E-03	3.18E-03	3.99E-04	3.99E-04	2.82	2.82	2.82	2.82
	Terrestrial acidification	2.85E-05	2.85E-05	2.42E-06	2.42E-06	0.03	0.03	0.02	0.02
	Freshwater eutrophication	4.44E-06	4.44E-06	8.33E-07	8.33E-07	0.00	0.00	0.01	0.01
	Terrestrial ecotoxicity	2.62E-04	2.62E-04	3.50E-05	3.50E-05	0.23	0.23	0.25	0.25
	Freshwater ecotoxicity	1.60E-06	1.60E-06	1.50E-07	1.50E-07	0.00	0.00	0.00	0.00
	Marine ecotoxicity	1.40E-09	1.40E-09	4.43E-09	4.43E-09	0.00	0.00	0.00	0.00
	Agricultural land occupation	2.86E-02	2.86E-02	2.03E-03	2.03E-03	25.42	25.42	14.34	14.33
	Urban land occupation	4.36E-05	4.36E-05	7.87E-06	7.87E-06	0.04	0.04	0.06	0.06
	Natural land transformation	1.27E-04	1.27E-04	1.70E-05	1.70E-05	0.11	0.11	0.12	0.12
Resources	Metal depletion	3.54E-05	3.54E-05	6.13E-06	6.13E-06	0.03	0.03	0.04	0.04
	Fossil depletion	2.32E-02	2.32E-02	4.27E-03	4.27E-03	20.60	20.60	30.18	30.17
Water-related impacts	*Impact on Santa Rosa*	*-5.23E-06*	*-1.12E-21*	*-8.12xE-07*	*2.92E-06*	-0.005	0.000	-0.006	0.021
	Impact on the river	*1.00E-06*	*4.52E-07*	*1.97E-07*	*0.00E+00*	0.001	0.000	0.001	0.000
	TOTAL	1.13E-01	1.13E-01	1.42E-02	1.42E-02				

9.6.14. Threatened species in Santa Rosa

Of the plant species there are none listed in the IUCN Redlist of Threatened Species.[47] Of the 73 listed bird species there are three birds (two residential and one migratory bird) which are “near-threatened”(Table 9.6.16).

Table 9.6.16: Conservation status of birds of Santa Rosa.[47] All other bird species are not threatened.

Species name	Common Name	Conservation status (IUCN)
Phalacrocorax bougainvillii	Guanay Cormorant	near threatened
Pelecanus thagus	Peruvian Pelican	near threatened
Thalasseus elegans	Elegant Tern	near threatened

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9.7. Annex to Chapter 7

9.7.1. Effect factor

Figure 9.7.1 shows the plots of salinity concentration-response curves for the various endpoints (see references for further information). The 4-parameter log-logistic function (Equation 9.7.1) was fitted to all data with the aid of the R package 'drc'[1]:

$$y = c + \frac{d-c}{1+(\frac{x}{e})^b} \qquad \textbf{Equation 9.7.1}$$

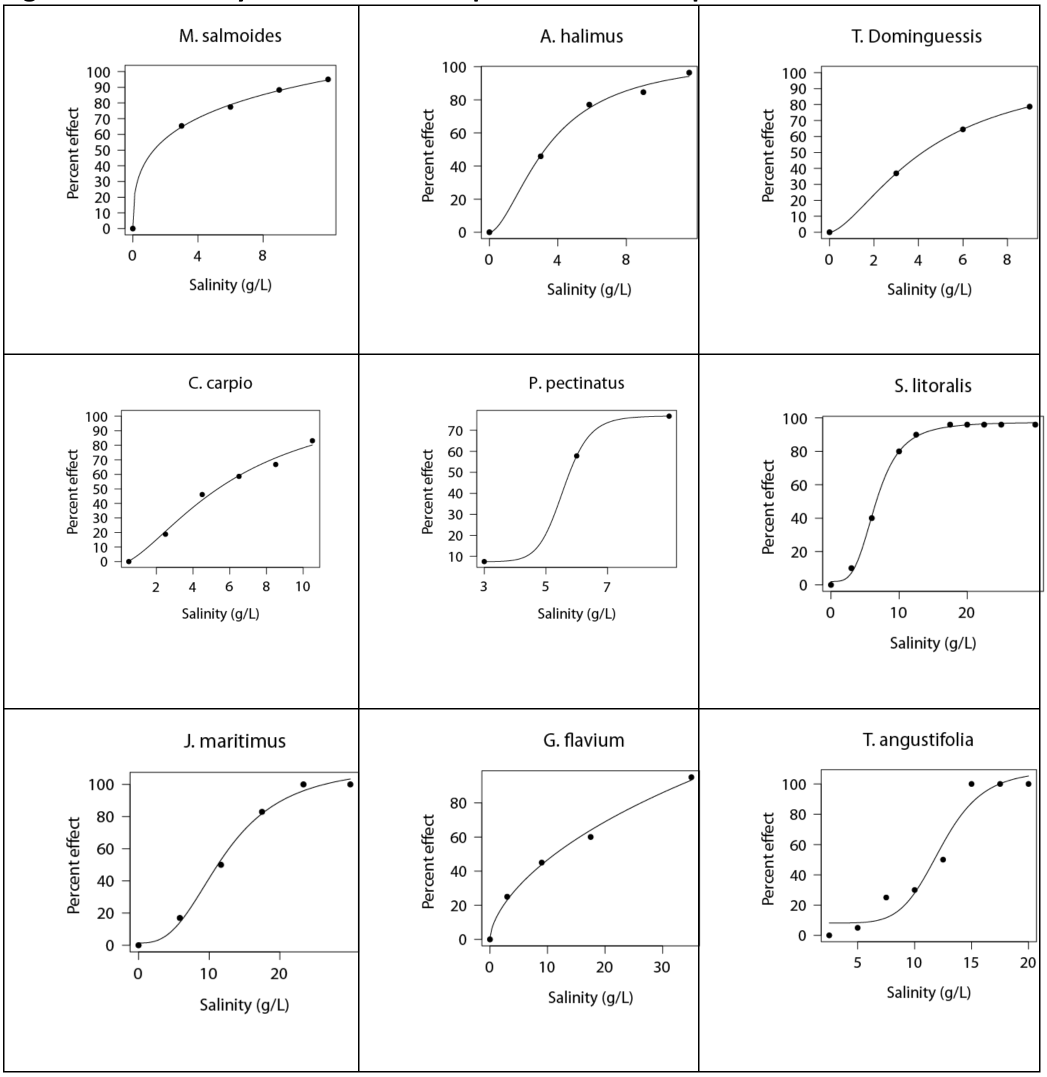
Where y is the response , x the salinity concentration, *c* and *d* are the lower and upper horizontal asymptotes respectively, *b* is the curvature and *e* is the inflection point[2]. The EC50 was estimated from each fitted curve and is displayed for each species in Table 9.7.1, which also displays the EC50s gathered from literature.

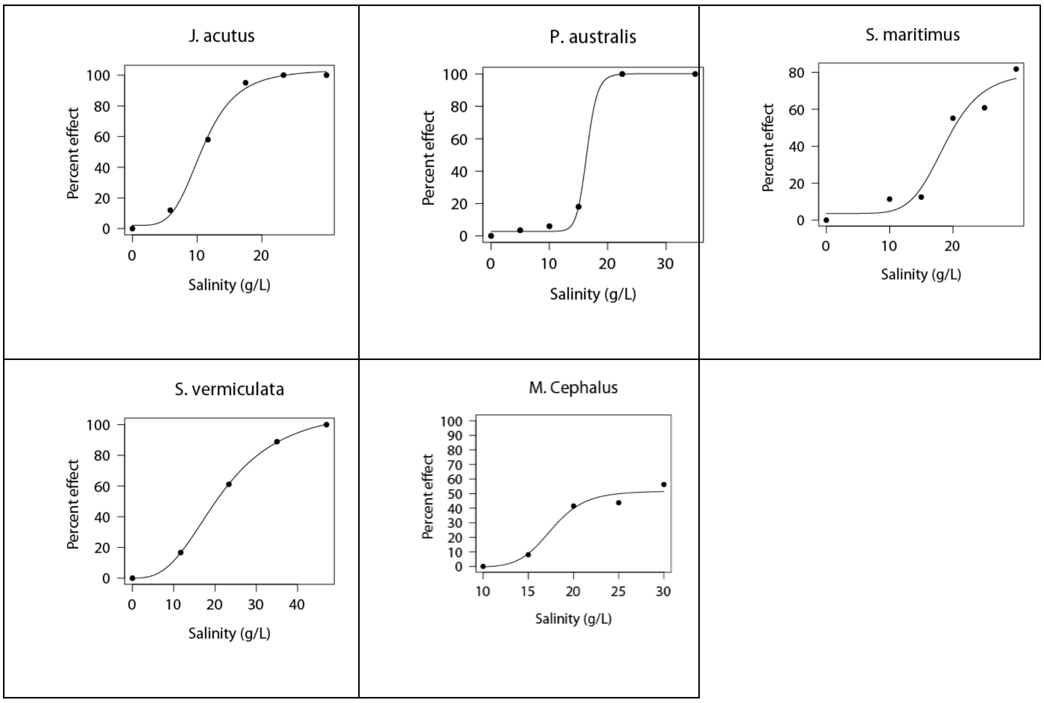
Table 9.7.1. EC50's from species in Albufera de Adra.

Species	EC50	Endpoint	Comments	References
Micropterus Salmoides	1.3	Survival	Calculated	Meador, MR. and Kelso, WE 1989
Atriplex Halimus	3.3	Germination	Calculated	Ahmed, HB. et al.2008
Typha Domingensis	4.2	Height, shoots and dry matter	Calculated	Glenn, E. et al. 1995
Ciprinus Carpio	5.3	Length and weight	Calculated	Wang, JQ. et al.1997
Potamogeton Pectinatus	5.8	Biomass	Calculated	Van Wijk, R.J. et al.1988
Nerium Oleander	5.9	Growth	From literature	Bright, DA and Addison, J 2002
Scirpus Litoralis	6.7	Germination	Calculated	Espinar, JL and Garcia, LV 2005
Juncus Maritimus	11.2	Germination	Calculated	Boscaiou, M et al. 2007
Glacium Flavium	11.3	Growth	Calculated	Cambrollé, J. et al. 2011
Typha Angustifolia	11.7	Survival	Calculated	Hutchinson, I. n.d.
Juncus Kraussii	11.7	Germination	From literature	Greenwood, M.
Juncus Acutus	12.1	Germination	EC50 calculated = 10.6; final EC50	Boscaiou, M et al.2007

			calculated by taking geometric mean of extracted EC50 plus one more value found in literature: 13.9	Greenwood, M. nd
Phragmites Australis	18.7	Survival	EC50 extracted from plot = 16.4; final EC50 calculated by taking geometric mean of extracted EC50 plus two more values found in literature: 18.9 and 21	Lissner, J. and Schierup, HH 1997 Greenwood, M. n.d.
Dictyosphaerium Chlorelloides	19.9	Growth	From literature	Bartolomé, MC 2009
Scirpus Maritimus	20.2	Survival	Calculated	Lillebø, AI et al. 2003
Salsola Vermiculata	20.2	Germination	Calculated	Guma IR. et al.2010
Mugil Cephalus	25.1	Growth	Calculated	Barman, UK.et al. 2005

Figure 9.7.1. Salinity concentration-response curves for species in Albufera de Adra.





9.7.2. General assumptions for the Fate Factor

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

9.7.3. Sensitivity analysis for the Fate Factor

Table 9.7.2: Sensitivity scenarios for the Nueva lagoon of Albufera de Adra. Parameters which were varied are the salinity (C_N) in the wetland in 2003 and 2008, the number of wet (Y) and dry (X) months, as well as the amounts of precipitation (P_N) and evapotranspiration (ET_N) on the Nueva lagoon.

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

9.7.4. References

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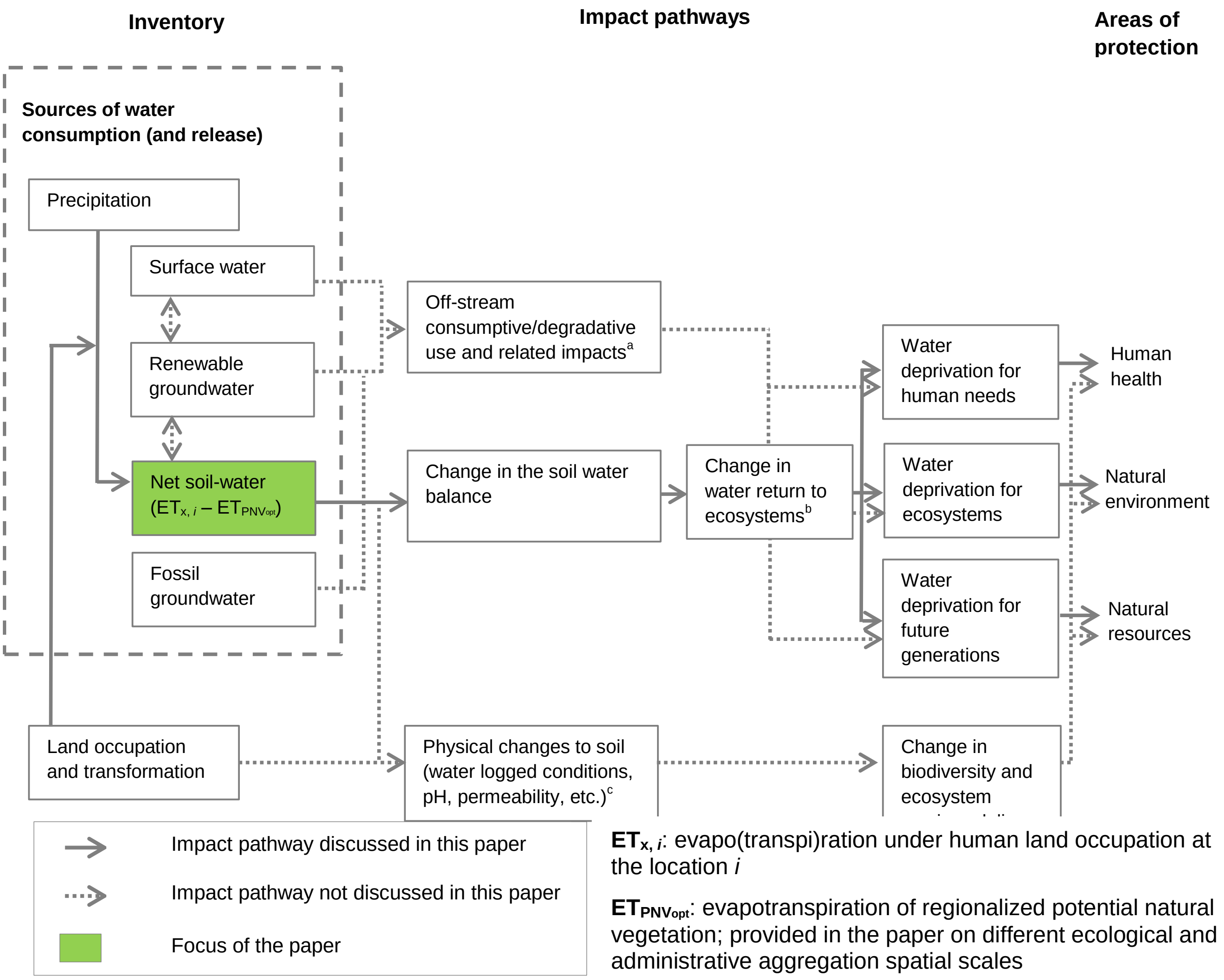
9.8. Annex to Chapter 8

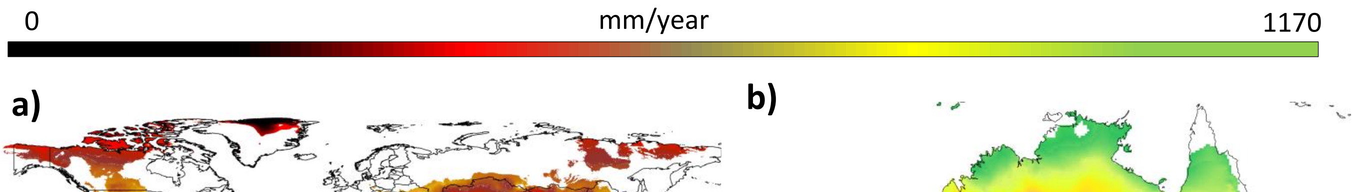


[a] *Cause-effect chains described in Bayart et al., (2010) and Kounina et al., (2013), both addressing water use in LCA.*
[b] *Cause-effect chain related to the freshwater regulation ecosystem service. This ecosystem function was modeled in Milà i Canals et al., (2009) (water-use impacts) and Saad et al., (2011) (land-use impacts), who focus on the land-use effects on rainwater infiltration, while we focus on changes in evapo(transpi)ration due to land occupation.*
[c] *Cause-effect chains described in Milà i Canals et al., (2007) addressing land use in LCA.*

Figure A8.1: graphical description of the impact pathways related to soil moisture consumption (green water) in LCA. Adapted from Kounina et al., (2013) and Milà i Canals et al., (2009).

Table A8.1: **brief description of the proposed method for estimation of a calibrated and regionalized k parameter (k_{opt}) and for potential natural vegetation evapotranspiration (ET_{PNVopt}).**

	Approach 1: equation-based	**Approach 2: model-based**
Input data	Evapotranspiration equation inferred from basin-scale water balance calculations (Piñol et al., 1991)	Evapotranspiration algorithm inferred from empirical observation of the normalized difference vegetation index (NDVI) (Zhang et al., 2010)
Output data	$ET_{PNV,\ i}$: Evapotranspiration of potential natural vegetation at the location *i*. $ET_{PNV,\ i} = f\ (P_i, ET_{0,i}, k)$	AET_i: Evapotranspiration of actual vegetation at the location *i*
Spatial differentiation	Eq. only for dry lands ($P/ET_0 \leq 0.75$) Depending on resolution of P and ET_0. Site-generic $k = 2$	8 kilometers at the global scale
Purpose in this paper	Base results with site-generic parameter *k*	Parameterization of *k* in 6788 data points *i* of natural coverage in dry lands
	Calibrated approach	
Output data	$ET_{PNVopt,\ i}$: evapotranspiration of potential natural vegetation optimized at the location *i*. $ET_{PNVopt,\ i} = f\ (P_i, ET_{0,i}, k_{opt,i})$	
Spatial aggregation of $k_{opt,i}$	Grid-cell: based on the local aridity index ($AI = P/ET_0$) at location *i*: 8 $k_{opt,ai}$ values, one for every AI category created (Table 8.1) Ecoregions: 501 $k_{opt,r}$ values (see XLS file) Biomes: 14 aggregated $k_{opt,r}$ values (see XLS file)	
Spatial aggregation of $ET_{PNVopt,\ i}$	Grid-cell: ET_{PNVopt} map calculated with $k_{opt,ai}$, and P and ET_0 10 arcmin resolution layers[8] (figure 8.1 and GIS file) Ecoregions: - ET_{PNVopt} map calculated with $k_{opt,r}$, and P and ET_0 10 arcmin resolution layers (New et al., 2002) (figure A8.2 and GIS file) - ET_{PNVopt} tabulated values calculated with $k_{opt,r}$, and average P and ET_0 of the dry lands of the ecoregion (see XLS file) Biomes: ET_{PNVopt} tabulated values calculated with biome-aggregated $k_{opt,r}$, and P and ET_0 10 arcmin resolution layers (New et al., 2002) (see XLS file) Countries, continents and earth: ET_{PNVopt} tabulated default values calculated with aggregation of arid grid-cell ET_{PNVopt} values in countries, continents and the earth (see XLS file)	



LC-IMPACT

SEVENTH FRAMEWORK PROGRAMME

Figure A8.2: PNV related water consumption based on the local aridity index approach (ET_{PNVopt} with $k_{opt, ai}$) for a) global dry lands, and b) Australia; PNV related water consumption based on the ecoregion approach (ET_{PNVopt} with $k_{opt, r}$) for c) global dry lands, and d) Australia. Global changes are difficult to see, but become more visible at greater spatial resolution scales. Note the patchiness of the ecoregion approach (d) in comparison to the local aridity index approach (b).

Table A8.2: spatial data available in the LCI, PNV water consumption and net-soil water consumption ($NET_{soil\text{-}water}$) for the three practical examples based on wheat cultivation presented in the main paper.

Spatial LCI data	$ET_{soil\text{-}water, wheat, i}$ (mm/y)	ET_{PNVopt} avg (mm/y)	$NET_{soil\text{-}water}$ avg (mm/y)	% of wheat production	Weighted $NET_{soil\text{-}water}$ avg (mm/y)

		(min, max)	(min, max)	in the case study[a]	(min, max)
Case study 1: known exact location					**-287 (-213, -317)**
-4.63 lon, 36.75 lat	197	484 (410, 514)[b]	197-484= -287 (-213, -317)	100%	=-287*1=-287 (-213, -317)
Case study 2: known the province					**-180 (-128, -230)**
Province of Málaga, South of Spain	204	384[c] (332, 434)[d]	204-384= -180 (-128, -230)	100%	=-180*1=-180 (-128, -230)
Case study 3: known the country mix					**-320 (-278, -361)**
Spain	157	462 (383, 541)[e]	157-462=-305 (-226, -384)	12%	=-305*0.12=-37 (-27, -46)
Italy	216	496 (444, 548)[e]	216-496=-280 (-228, -332)	15%	=-280*0.15=-42 (-34, -50)
France	221	551 (518, 584)[e]	221-551=-330 (-297, -363)	73%	=-330*0.73=-241(-217, -265)

[a] Global wheat production: Spain 1% ; Italy 1.2% ; France 6% (Mekonnen and Hoekstra, 2010).
[b] Precipitation = 547; ET_0 = 1176; Aridity index = 0.47; $k_{opt,\ ai}$ = 1.81±0.63 (table 8.1)
[c] Precipitation = 543; ET_0 = 1237; Ecoregion code = 81221 (Southwest Iberian Mediterranean sclerophyllous and mixed forests); $k_{opt,\ r}$ = 1.03±0.00 (see the XLS file, Ecoregion assessment sheet)
[d] Min. and max. values calculated with the standard deviation of P and ET_0 for the dry areas of the ecoregion (see the XLS file, Ecoregion assessment sheet)
[d] See the XLS file (Country and continent assessment sheet)

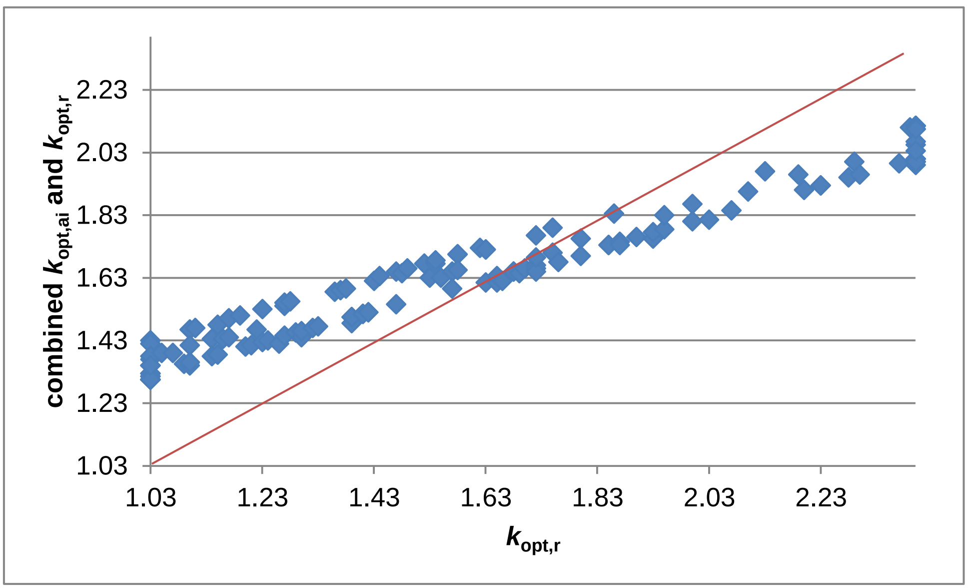


Figure A8.3: comparison of $k_{opt,r}$ (x-axis, ecoregion assessment) versus combined $k_{opt,ai}$ and $k_{opt,r}$ (y-axis, grid-cell and ecoregion assessments). Combined k parameter is the arithmetic average of $k_{opt,ai}$ and $k_{opt,r}$ in a specific ecoregion. Note that for $k_{opt,r}$~<1.63, the combined k is larger than this value (blue point is above the 1:1 line), while for $k_{opt,r}$~>1.63 the trend is the opposite (blue point is below the 1:1 line). This means higher ET_{PNV} losses for the combined approach for ecoregions with $k_{opt,r}$~<1.63 and lower ET_{PNV} for ecoregions with $k_{opt,r}$~>1.63.

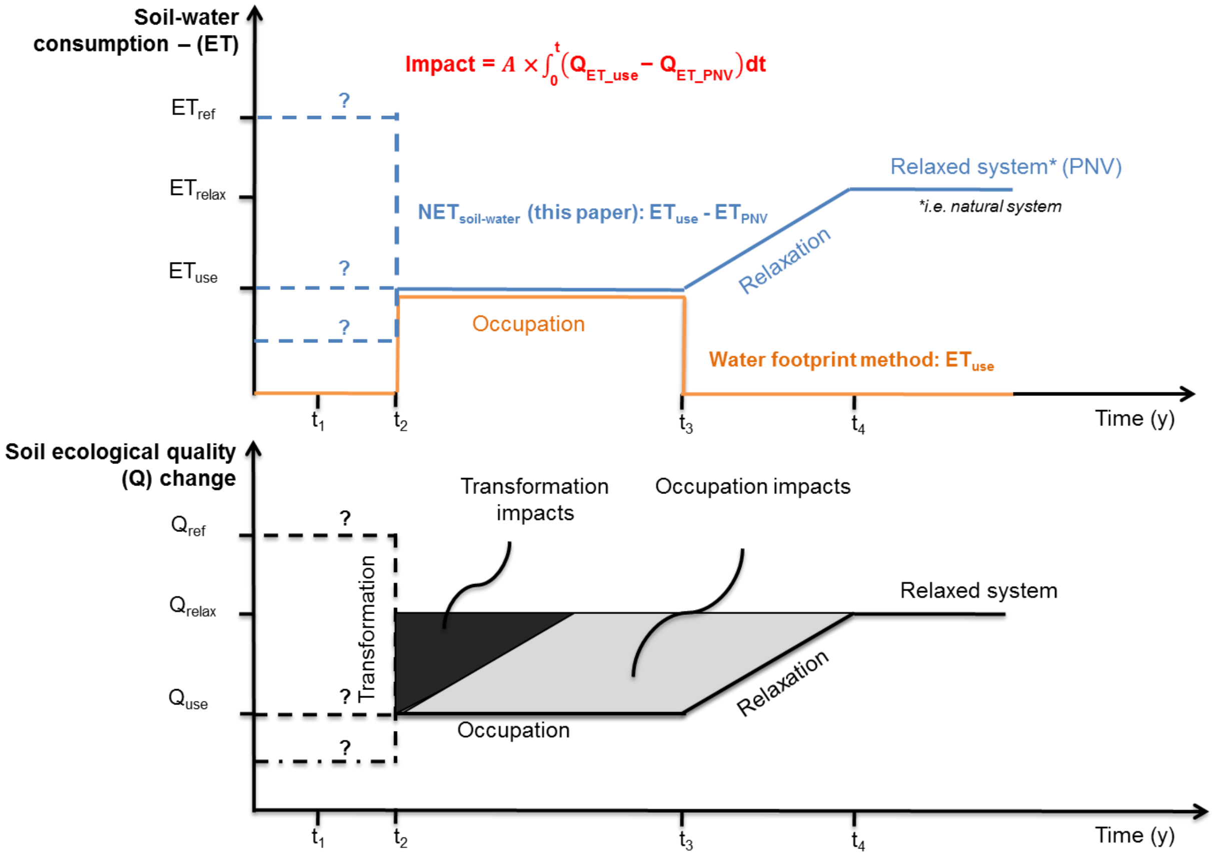


Figure A8.4: illustration of the effects of land-use changes in soil-water consumption and in the ecological quality of soil. While the water footprint method (orange line) accounts for evapo(transpi)raton during the use of the land (e.g.: agriculture), the LCI framework proposed in this paper (blue line, $NET_{soil\text{-}water}$) evaluates the difference in evapo(transpi)ration between the use of the land and the reference system (also known as relaxed system in the LCA literature), that is between, e.g., agriculture and the potential natural vegetation (PNV). Research needs should focus on developing LCIA methods to account for the environmental impact of this evapotranspiration change (equation in red). Adapted from Milà i Canals et al., (2007).

9.8.1. Detailed results table (for application)

In the attached XLS file, green water consumption of 160 crop groups is provided per country together with PNV water consumption estimates (including standard deviation) on ecoregion, biome, country and continental level. Additionally, Optimized average ± standard deviation k values (kopt, r) for the 501 ecoregions located in the dry lands including precitpitation and ET climate data are presented in the following XLS-file:

SI_chapter 8 (green water).xls