Assessing the environmental impact of water consumption by energy crops grown in Spain

Montserrat Núñez, Stephan Pfister, Assumpció Antón, Pere Muñoz, Stefanie Hellweg, Annette Koehler and Joan Rieradevall

Address correspondence to: Montserrat Núñez. IRTA, Sostenipra. Ctra. de Cabrils, Km 2. E-08348 Cabrils, Barcelona, Spain. Montserrat.nunez@irta.cat.

Abstract

The environmental impact of water consumption of four typical crop rotations grown in Spain, including energy crops, were analyzed and compared against Spanish agricultural and natural reference situations. The life cycle assessment (LCA) methodology was used for the assessment of the potential environmental impact of blue water (withdrawal from water bodies) and green water (uptake of soil moisture) consumption. The latter has so far been disregarded in LCA. To account for green water, two approaches have been applied: the first accounts for the difference in green water demand of the crops and a reference situation. The second is a green water scarcity index, which measures the fraction of the soil-water plant consumption and available green water. Our results show that, if the aim was to minimize environmental impacts of water consumption, the energy crop rotations assessed in this study were most suitable in basins in the northeast of Spain. In contrast, the energy crops grown in basins in the southeast of Spain were associated with the greatest environmental impacts. Further research into the integration of quantitative green water assessment in LCA is crucial in studies of systems with a high dependence on green water resources.

Keywords

bioenergy crops; green water scarcity index (GWSI); life cycle assessment (LCA); life cycle impact assessment (LCIA); water footprint

<heading level 1> Introduction

<heading level 2> Water consumption and energy crops

Freshwater is an essential natural resource for human and ecosystem health. While water is abundant globally, the resource is under increasing pressure in many parts of the world. Agriculture is by far the largest water-use sector, accounting for around 70% of the water withdrawn worldwide from rivers and aquifers for agricultural, domestic and industrial purposes (FAO 2008). In Europe, agricultural water use is a serious concern especially in southern and southeastern regions such as Spain, where water is scarce and highly variable during the year and from year to year (European Union 2010).

In Europe, an increasing share of the agricultural land is now used for the cultivation of biomass for energy production (EEA 2007). The use of bioenergy in Europe offers significant opportunities to replace conventional fossil fuel energy sources, reduce greenhouse gas emissions and improve energy supply security. However, it is clear that the increasing demand for energy crops will lead to further pressure on water resources if their cultivation leads to an increased share of land dedicated to irrigated crops. The water footprint associated with growing a particular crop, which is defined as the total volume of freshwater used, whether directly or indirectly, during the complete crop production cycle and expressed as the volume of water per harvested yield (Hoekstra et al. 2011), was recently quantified for Spain as 1.03-1.20 m^3 kg^{-1} for irrigated wheat, 1.03-1.08 m^3 kg^{-1} for irrigated barley and 0.59 m^3 kg^{-1} for irrigated maize (Aldaya and Llamas 2009). All these figures include both the blue (irrigated water from surface and groundwater withdrawal, BW) and green (water from precipitation, GW) components of

the water footprint, but not the gray (polluted water) one. Average irrigation water consumption in Spanish production was quantified as 0.05-0.22 m^3 kg^{-1} for wheat, 0.04-0.17 m^3 kg^{-1} for barley and 0.05-0.17 m^3 kg^{-1} for maize (Pfister et al. 2011). In the case of energy crops, Sevigne and colleagues (2011) found that the water footprint of a short rotation poplar forestry located in the Ter basin (northeast Spain) was about 1.25-1.50 m^3 kg^{-1}, of which the volume attributable to irrigation water was 0.23-0.34 m^3 kg^{-1}, with the volume depending on the density of the plantation. This irrigation water requirement is underlined in M. Gasol and colleagues (2009) as the determining factor restricting the spread of the short rotation poplar forestry in Spain. This example highlights the importance of selecting a balanced combination of crops and locations that minimize water stress.

<heading level 2> Water use and life cycle assessment

A number of water consumption analyses using different hydrological models (Hoff 2010) for many different agricultural products (Gerbens-Leenes et al. 2009; Hoekstra and Hung 2005; Pfister et al. 2011) have been published recently. Most of these studies focus on the quantification of water use while ignoring the environmental impact of water consumption on available freshwater resources, ecosystem quality and human health. Furthermore, water use and water resource depletion have gradually gained greater importance in life cycle assessment (LCA), a methodology for quantifying the environmental impacts of products and activities. However, there is still only preliminary scientific consensus on the parameters to consider and the methodology to follow to account for water use-related impacts.

Two regionalized methodologies dealing with how to assess consumptive water use in LCA studies have recently been published: Milà i Canals and colleagues (2009)

suggested direct environmental impact indicators, or midpoint indicators, while Pfister and colleagues (2009) developed environmental damage indicators, or endpoint indicators, for human health, ecosystem quality and resource attributes of specific importance or concern. Both methodological approaches are compliant with the need for the water footprint concept (Hoekstra et al. 2011) to generate life cycle inventory (LCI) data. They also both limit their focus to the environmental consequences of blue water consumption at the life cycle impact assessment (LCIA) stage. However, Pfister and colleagues (2009) recognized that neglecting potential changes in green water flows due, for example, to different vegetation types, implies making a simplification and that any related effect should be addressed in future research. Milà i Canals and colleagues (2009) determined changes in green water flows as a consequence of land use changes, as these aspects are strongly related. The methods of Milà i Canals and colleagues (2009) and Pfister and colleagues (2009) take into account the globally unequal distribution of freshwater resources by the spatial differentiation of impacts at the watershed level.

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

In this paper, we assessed the environmental impact associated with water consumption of several energy crop rotation systems grown in Spain. The aims of the study were to (i) assess the environmental impact of blue water consumption in real agricultural settings with a recently developed life cycle impact assessment (LCIA) method (Pfister et al. 2009), examining a large number of case-study sites distributed all over Spain, (ii) propose a characterization approach to account for the environmental

impact of green water consumption, (iii) compare environmental impacts caused by water consumption of the tested energy crops against reference agricultural and natural situations in the country, (iv) identify appropriate production areas and energy crop rotations to minimize the environmental effects of water consumption for growing energy crops in Spain, and (v) quantify the land and water trade-offs between rainfed and irrigated rotations.

<heading level 1> Materials and methods

<heading level 2> Agricultural systems and water consumption

<heading level 3> Crop rotation systems

Water consumption in crop rotation systems was estimated in 117 plots located throughout Spain, covering 23 out of the 47 Spanish watersheds of the WaterGAP 2 global model (Alcamo et al. 2003) (figure 1). More information on the characteristics of the studied plots is available on section 1 of the online supplementary material.

Figure 1

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

This was taken into consideration in our assessment of water consumption associated with four possible energy crop production systems in Spain (table 1): three systems for producing annual energy crops in a three-year crop rotation and a short rotation of a

perennial forestry crop: poplar, with a life-span of 15 years, cut once every three years. The rotations were of crops grown to meet food and energy requirements.

<heading level 3> Agricultural and natural reference situations for modeling water use impacts

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

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

The second reference was the potential natural land situation. In the classification systems proposed by many authors (Bailey 1998; Folch et al. 1984; Olson and Dinerstein 2002), the potential natural vegetation in Spain (except along a narrow coastal strip in the north of the country) is Mediterranean forest (311 classes of broad-leaved forest and 313 classes of mixed broad-leaved and coniferous forest, according to the CORINE land cover classification; EEA 2000). The most representative tree species

found in this kind of forest usually include evergreen varieties with small and/or leathery leaves to better withstand summer droughts: holm oaks (*Quercus ilex* L.) and cork oaks (*Quercus suber* L.); wild olive trees (*Olea europaea* L.); Lusitanian oaks (*Quercus faginea* Lam.); and algarroba trees (*Ceratonia siliqua* L.). Aromatic plants such as rosemary (*Rosmarinus officinalis* L.), sage (*Salvia officinalis* L.) and Corsican mint (*Mentha requienii* Benth) are widespread under the tree canopy. As the majority of the energy crop rotations are located in the Mediterranean area, the Mediterranean forest (MF) was chosen as the natural reference for all the plots, with a MF-MF-MF reference accounting for the water consumption over a three-year period.

<heading level 3> Irrigation schemes of the rotation systems

Crops of the studied rotations (table 1) were grown under three different irrigation schemes, depending on the water supplied in relation to their requirements for optimal production and yield, under the given climatic conditions. Barley (B), wheat (W), oilseed rape (R) and sunflower (SF) are non-irrigated crops. Both sorghum (SG) and poplar (P) were grown under support-irrigated schemes (deficit irrigation). In this situation, the water supplied is below plant requirements, so maximum yield is not achieved. Crops with deficit irrigation have higher water-use efficiencies (obtained yield per unit of water consumed) than rainfed crops. This type of crop management is typical in low precipitation and water-limited areas such as Mediterranean countries. Maize (M) was the only crop where irrigation completely satisfied the crop water requirements, i.e., standard conditions and maximum yield. Finally, neither reference situation (B-W-F and MF-MF-MF) were irrigated.

Table 1

<heading level 3> Calculation of the water consumption in each rotation system

The total water consumption of each rotation is the sum of the respective evapotranspiration losses (ET_c) of the crops in the rotation. The ET_c of each crop was calculated based on the FAO approach (equation 1, Allen et al. 1998), in which crop coefficients (K_c) are used to relate ET_c to potential evapotranspiration (ET_0).

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

ET_0 describes the evapotranspiration from a reference surface. More information about equation 1 and the data sources used to obtain ET_c values is provided in section 2 of the supporting material.

For rainfed and support-irrigated crops grown in the Mediterranean region, real crop evapotranspiration deviates from ET_c in equation 1, due to the water shortage conditions in which they grow. To take this into account, the standard crop coefficients of equation 1 have been adjusted in our study (table 1SM, following Allen et al. 1998) by technical experts from the IRTA-Experimental Station Mas Badia Foundation (Salvia 2009).

Monthly green water consumption by crops was calculated as shown in equation 2. It can be seen that the amount of green water consumed is limited by effective precipitation (Pr'), defined as the share of total rainfall (Pr) available for uptake by plants. Equation 2 determines the maximum amount of water that a rainfed crop consumes during a month:

$$\begin{aligned} &\text{If} \quad Pr' \geq ET_c \quad \text{GW consumption} = ET_c \\ &\text{If} \quad Pr' < ET_c \quad \text{GW consumption} = Pr' \end{aligned} \qquad (2)$$

Monthly blue water consumption varies between crops. For rainfed crops, it is zero. For irrigated crops, it is equal to the plant water deficiency – the difference between ET_c and Pr' –, and for support-irrigated crops, irrigation is restricted to a predetermined threshold, which depends on climate and crop type (table 1).

Effective precipitation Pr' depends on specific soil characteristics (soil texture and structure, land slope, vegetation types and crop management). This spatial variability was taken into consideration in the estimation of plant water consumption by calculating the Pr' of each plot and crop on a monthly basis. This was done using the runoff curve number method adapted to Spanish conditions (Ferrer 1993; MOPU 1990, see section 3 of the supporting material for more information).

In summary, the LCI data used to derive the monthly water consumption for each crop were: i) monthly potential evapotranspiration; ii) monthly adjusted crop coefficients, to obtain crop evapotranspiration values per month; iii) monthly precipitation; iv) land use (classification of MOPU 1990); v) soil texture; and vi) land slope, to obtain the effective precipitation per month, i.e., the green water available for evapotranspiration.

The FAO approach used for crops could not be used to calculate evapotranspiration of the natural reference vegetation (i.e., MF), since K_c values have only been obtained for crops. Instead, the evapotranspiration of MF was derived on an annual basis from a method proposed by Piñol and colleagues (1999), where ET_0, Pr and ET_c in Mediterranean forest catchments are related as shown in equation 3:

$$ET_c = \left(\frac{\left(\frac{Pr}{ET_0}\right)^k}{1+\left(\frac{Pr}{ET_0}\right)^k} \right)^{\left(\frac{1}{k}\right)} \times ET_0 \qquad (3)$$

where k=2.0 is a group parameter of the non-climatic catchment characteristics relevant for the water balance. More information on this approach is reported in section 4 of the supporting material.

<heading level 2> Environmental assessment

<heading level 3> Basis for comparison

As agricultural systems are multifunctional, two different approaches were used to compare them:

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

Productive scope. Crops are grown for food, feed, fiber or biomass for bioenergy. The objective is to minimize the environmental impacts per unit of product (Nemecek et al. 2008). In terms of water, it means reducing the quantity of water applied per unit output of, for example, harvested dry matter, raw protein yield or edible energy yield, which denote different productive functions for comparison. A metric ton of harvested dry matter was the functional unit used to identify rotations with the highest water-use efficiency. The physical unit of the productive function was the overall m^3 of water consumed per metric ton of harvested yield, as dry matter per area, during the three-year rotation. Water consumed per crop was allocated to the dry matter of harvested grains for barley, wheat, oilseed rape, sunflower and maize, and to the whole biomass for sorghum and poplar, considering these as the products from each crop. The

Mediterranean forest output was related to its annual aboveground biomass production. To simplify the analysis, we assumed that one metric ton of product was equivalent in function for each crop type, independently of the specific crop type and the harvested fraction (i.e., crop grain or whole biomass).

Current yield data of barley, wheat, oilseed rape, sunflower and maize, regionalized at a province level, were from the Spanish Ministry of the Environment and Rural and Marine Affairs, for the period 2003-2006. Province yield data were assigned to the watershed resolution according to the province area share within a specific watershed. Information on the harvested dry matter for crop and rotation in each watershed is provided in table 6SM. Average whole dry biomass production was taken as 7.50 $tha^{-1}y^{-1}$ for sorghum and 13.30 $tha^{-1}y^{-1}$ for poplar in all watersheds (Salvia 2009), as regional yield data of these energy crops are not yet available. These yield values come from deficit-irrigated experimental plots cultivated in Spain to produce energy from biomass (*SSP On Cultivos*, www.oncultivos.es). The annual net aboveground primary production of the Mediterranean forest was taken as 5.65 $tha^{-1}y^{-1}$ (Ibàñez et al. 1999).

<heading level 3> Assessment of blue water consumption

The LCIA method developed by Pfister and colleagues (2009) was used for evaluating the ecological impacts of the area-based and the yield-based systems. This method includes the use of a water stress index (WSI) as a characterization factor for a midpoint water deprivation category as well as an assessment of damage to resources, human health and ecosystem quality that is compatible with the Eco-indicator 99 (EI99) framework (Goedkoop and Spriensma 2001). The WSI, ranging from 0.01 to 1, is based on the ratio of freshwater withdrawals for different users to blue water availability in each watershed, and indicates the portion of consumptive water use that deprives

downstream users of freshwater. Damage to resources are measured using surplus energy units (MJ), damage to human health using the disability adjusted life years concept (DALY), and damage to ecosystems using the potentially disappeared fraction of species (PDF). In developed countries such as Spain, water scarcity does not affect human health, for instance causing malnutrition or diarrhea, as these countries are able to compensate for reduced freshwater availability by for example, water desalination. The midpoint water deprivation category, the endpoint factors for resources and ecosystem damage, as well as the aggregated EI99 single-scores (Pfister et al. 2009) were applied.

<heading level 3> Assessment of green water consumption

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

Spatial agricultural management scope. For the area-based assessment, the green water scarcity index (GWSI) defined by the Water Footprint Network (http://www.waterfootprint.org) was applied (equation 4). Data collected for the LCI were used to obtain results for this index, which provides information at the plot level.

$$GWSI = \frac{GW}{Pr'} \qquad 0 \leq GWSI \leq 1 \qquad (4)$$

Where GWSI is dimensionless, Pr' is the effective precipitation per area during the three years of the rotation (m^3m^{-2} $rotation^{-1}$) and GW is the amount of Pr' consumed by the plant in this same area and time unit, that is, green water consumption (m^3m^{-2} $rotation^{-1}$). GWSI indicates the aridity stress where crops grow. Low values are more favorable for the environment, as they indicate less stress upon available soil water. As effective precipitation and green water depend on vegetation, the green water scarcity index is particularly useful for deciding the spatial location for rainfed rotations. It may also serve to compare the green water demands of the rainfed rotations with those of the

natural reference systems of a particular site. GWSI could be used as an additional indicator in environmental assessments of agricultural and forestry systems, so supplementing the results of LCA studies by ranking different cultivation areas/regions in regard to the absolute aridity stress.

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

It is possible to base selection of the most appropriate places and rotations to minimize the environmental impact of water consumption when growing energy crops in Spain on assessments on blue water and green water consumption. As we did not define a single-score indicator for simultaneous consideration of blue and green water scores, suitable locations and rotations depended on the type of water (i.e., blue or green) analyzed.

<heading level 3> Assessment of land use

As (blue and green) water use and land use are closely linked, we conducted a screening assessment of the impacts due to land occupation by the different rotations studied. In order to compare land and water use impacts, this analysis was carried out using the same endpoint perspective: the damage-oriented EI99 LCIA method (Goedkoop and Sprinsma 2001). Ecosystem quality damage and aggregated EI99 results were calculated. The default normalization and weighting factors (hierarchist perspective, average weighting, EI99HA) were used to calculate single-scores.

<heading level 1> Results

<heading level 2> Water use assessment

<heading level 3> Life cycle inventory

The blue and green water consumption of the crop rotations regionalized at river basin level is shown in table 2SM. Irrigated and deficit-irrigated rotations were more water-intensive per area but more water-use efficient per product yield than rainfed rotations (figure 2, ET_c m^3 m^{-2} $rotation^{-1}$ and ET_c m^3 t^{-1}), due to higher yields of irrigated and deficit-irrigated crops. More results may be found in section 5 of the supporting material.

Figure 2

<heading level 3> Blue water life cycle impact assessment

The potential environmental damage of blue water consumption for five relevant Spanish basins covering different regions of the country is shown in table 2. For a complete list of all basins see table 3SM.

Using the WSI as assessment indicator for water deprivation, most watersheds in the south and southeast of Spain have the greatest environmental implications, both under the spatial agricultural management scope and the product life cycle assessment. For the yield-based assessment, basins in these areas scored between 11-times higher for the M-SG-M rotation and five-times higher for the P-P-P rotation than basins with the lowest deprivation impacts, located in the north and northeast (table 3SM). The fact that in 18 out of the 23 basins the WSI was >0.5 indicates the severe water stress affecting a large part of the basins in the country, mainly those in the south, southeast and east.

Damage to resources and ecosystems according to the endpoint factor analyses as well as the aggregating EI99 method were generally higher for the M-SG-M rotation system than for the P-P-P rotation (table 3SM). For the resources category, there was no water depletion in 13 out of the 23 assessed basins. These watersheds are scattered throughout the country. Major regional water depletion was recorded in southeastern basins.

The general trend of the EI99 single-score factor revealed that the northern and northeastern basins of Spain appeared to be the best choices for reducing blue water consumption and the ecological impacts associated with cultivating irrigated rotations. On the other hand, these rotations should not be applied in several of the basins in southeast Spain (figure 3a).

<heading level 3> Green water assessment

Spatial agricultural management scope. Looking at the outcomes of the green water scarcity index, no single rotation scored the best in all watersheds (table 2 and table 4SM). Here, results vary between basins because the GWSI does not only depend on green water consumption by plants, but also on conditions of the location, such as

the soil water availability, determined by soil properties (texture, slope, land use), and the amount and distribution of rainfall during the year. Irrigated rotations recorded very similar or even lower GWSI values (indicating lower aridity stress) than rainfed rotations in some water basins, which means that they do not use more soil-water compared to non-irrigated rotations. The Mediterranean forest GWSI was the highest (most arid) in all basins (figure 2, GWSI). Watersheds where crop rotations grow in conditions with more soil green water availability are not clustered in one specific area of the country, but in several small basins in the northeast and south (figure 3b). In these basins, the GWSI score was ≤0.65, indicating less stress upon soil water reserves compared to other watersheds, where the GWSI reveals that nearly 90% of the soil reserves are consumed.

Productive scope. Delta green water deprivation values showed that all rotations affect green water resources per product yield less than the agricultural reference system B-W-F (negative values of weighted dGW in table 2, table 4SM and figure 2 dGW deprivation Agr. ref.). If the potential natural land situation is chosen as the reference, rainfed rotations caused higher deprivation impacts (positive values of weighted dGW in table 2, table 4SM and figure 2 dGW deprivation Nat. ref.). Irrigated crop rotations were those with the lowest impacts on green water. The optimal spatial distribution of rotations fits well with the distribution for minimizing impacts of blue water consumption (figure 3c). Along with basins in the north and the northeast of Spain recommended in the blue water assessment, some central and southern areas had the lowest weighted delta green water values. This is because most of these water basins have the lowest WSI values in Spain and high crop yields.

<heading level 2> Land use assessment

A list of land use damage on the level of ecosystem quality and the aggregated EI99 single-scores is shown in table 2 and in table 5SM.

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

Variation of aggregated impacts of land use occupation was more than 65% between watersheds. The lowest occupation impacts of growing rainfed rotations were in the northern and central areas (figure 3d), whereas watersheds that were more suitable for irrigated rotations were found in central, southern and southeastern regions (figure 3e). Some of the latter areas had the highest blue water use impacts. The lowest land use occupation impacts were found in the regions with the highest crop yields, usually with higher water consumption.

Table 2

Figure 3

<heading level 1> Discussion

<heading level 2> Choice of the basis for comparison

The selection of an appropriate functional unit for LCA of agricultural systems is not an easy task, being currently a key subject of discussion between experts in the field. In this study, the environmental burdens of water consumption were quantified per m^2•rotation as well as per metric ton dry matter produced in each rotation. Whilst the m^2-based assessment is strictly for land management assessment, as a unit, it provides relevant information for comparing the total water consumption and environmental impacts of a specific rotation between watersheds. However, for comparing alternative

crops and rotations from an LCA perspective, neither the m^2 nor the metric ton based assessments are appropriate, as crops have different functions per m^2 and per metric ton and the definition of the functional unit states that all the systems being compared must have a common function. Thus, wheat and barley grain share a similar function, that of making flour to produce bread, while oilseed rape grain is primarily used for animal feed, producing vegetable oil for human consumption and making biodiesel. In contrast, all of the biomass harvested in the short rotation of poplar is for producing electricity by combustion. Furthermore, they provide different habitats. Despite this disparity of functions and harvested fractions, we chose the metric ton of harvested dry matter as the functional unit rather than others that could have better reflected the purpose of energy crops, such as the crop's calorific value, in order to avoid unfair comparisons between food and energy crops. For users who prefer another functional unit for an allocation we also provide the yield values of all crops in table 6SM.

<heading level 2> Green water assessment

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

Firstly, estimates of the use of green water additional to the reference situation have to be considered as the LCA base results so that water basins can be ranked according to their weighted green water impacts. This dGW deprivation indicator is useful for estimating the environmental burdens placed on the additional blue water resources of the watershed that could result from the consumption of green water. The WSI used in the calculation of dGW deprivation (WSI×dGW consumption) is defined by Pfister and

colleagues (2009) as a modified ratio of blue water withdrawal to blue water availability. It is not entirely accurate to use this to make the assessment of the additional green water consumption LCA-compatible, since the ratio is based on blue water flows. We still applied the WSI weighting because dGW also influences the availability of blue water and thus may exacerbate/alleviate water scarcity.

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

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

<heading level 2> Comparing water consumption and land use of rainfed and irrigated energy crop rotations

It can be seen in figure 2 how blue and green water consumption (ET_c) varies depending on the type of assessment. While the m^2-based assessment indicates that attention must be paid to the total water consumption of the irrigated rotations M-SG-M (1.6-2.9 times higher than the other rotations) and P-P-P (1.2-2.1 times higher than the other rotations), these have the highest water-use efficiency per output function (M-SG-M: 0.43-0.52 and P-P-P: 0.21-0.49 times the consumption of the other rotations). These

results indicate that both assessments are required for a sound evaluation and an efficient allocation of water resources, indicating the trade-off between both objectives. For regional resource management, the m^2-based assessment is helpful to alleviate water use intensity.

Considering the blue water assessment, rainfed rotations were the best option to minimize environmental burdens, as they are not irrigated (figure 2, BW deprivation). If irrigation efficiency had been considered in the inventory of blue water consumption, there would have been more differences between irrigated and non-irrigated rotations. It was not considered, as the irrigation system applied for each crop varies both between watersheds and within each individual watershed. We still recommend that irrigation efficiency is considered for each particular case study, where excess irrigation water recharges soil, increasing its green water availability, and may also drain off to groundwater or be evaporated unproductively.

With green water, different trends in impacts were detected depending on the assessment method applied (GWSI and dGW deprivation, figure 2), albeit, with both indices, irrigated rotations had the lowest impacts on the additional blue water flows and on the soil green water availability of the plot.

The land use impact assessment per metric ton of harvested dry matter, showed the land/water trade-off between irrigated and rainfed agriculture within watersheds (figure 4). While in rainfed agriculture there are no impacts on the blue water resources, there is 85% higher damage to ecosystems per product compared to irrigated rotations due to land use, on a nationwide average. This is because of the lower yields and, therefore, lower crop-water productivity of non-irrigated crops. In the vast majority of cases, the scores using EI99 methodology were higher for land use occupation impacts compared to water consumption impacts (figure 4, triangles below the line with slope = 1). The

higher scores for the land use impacts in the EI99 method may have been because the land use impact assessment was based on vascular plant species diversity in the Swiss lowlands and may therefore have been inappropriate for Spain, as Mediterranean, not temperate, forest is its reference system.

Figure 4

<heading level 2> Comparing green water consumption of energy crop rotations and reference systems

When the environmental impact caused by green water consumption of the energy crops were evaluated against the reference agricultural and natural situations, the most harmful effects were generally obtained for the reference systems (figure 2, GWSI and dGW deprivation).

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

Looking at the impacts on the on-site availability of green water (GWSI), the natural reference system (MF-MF-MF) was grown in some basins with up to 40% more severe aridity stress conditions than the other rotations. The higher green water consumption per area of Mediterranean forest compared to the four energy crop rotations was mainly due to the reduction of ET_c in crop stages with residue cover or bare soil in agricultural systems. Many studies have pointed out higher soil water retention capacity, higher water requirements and a decreasing runoff from areas under forest as compared with other plant communities in similar environmental conditions (Calder 2004; Valentíni

2003). This indicates that upstream forest cover does not generally enhance water availability downstream. In dry zone regions found in most parts of Spain, soil moisture seldom reaches the field capacity necessary to percolate water to recharge aquifers, independently of the vegetation cover and the infiltration properties of soils (Bellot and Ortiz de Urbina 2008). Shorter agricultural rotations can therefore be considered beneficial as compared to the natural reference situation, since crop production creates more runoff that increases the streamflow of the river basin. Other environmental issues, for example, carbon sequestration, must be considered to fully compare agricultural and forestry systems.

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

As shown in figure 2, the use of the agricultural or the natural reference system did not influence the relative results of impact scores between rotations, but it may influence the importance attached to water use in the overall assessment. Therefore, choosing the most suitable and consistent reference is a crucial issue in the evaluation of agricultural system alternatives in each ongoing assessment of water use impacts.

Milà i Canals and colleagues (2007a) recommended using natural relaxation, that is, the potential natural vegetation, in attributional LCA. This type of LCA assessment

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

<heading level 1> Conclusions

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

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

Together with the water assessment, studies covering land use impacts on biodiversity and ecosystem services (e.g., carbon sequestration potential, erosion

regulation potential) must be measured, for a full comparison of natural and agricultural systems.

Acknowledgements

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<heading level 1> References

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About the authors

Montserrat Núñez, Assumpció Antón and Pere Muñoz are researchers at the Catalan government Institute of Food and Agricultural Research and Technology (IRTA), Cabrils (Barcelona, Spain). Assumpció Antón is also lecturer at the Chemical Engineering Department, Universitat Rovira i Virgili (URV), Tarragona (Spain). Joan Rieradevall is professor at the Chemical Engineering Department, Universitat Autònoma de Barcelona (UAB), Bellaterra (Barcelona, Spain) and senior researcher at the Institute of Environmental Science and Technology (ICTA), UAB. These four authors are part of the Sostenipra research group. Stephan Pfister is currently post-doctoral fellow at UC Santa Barbara (CA, United States). Stefanie Hellweg is Professor at the Institute of Environmental Engineering, ETH Zurich, Zurich (Switzerland). Annette Koehler is Managing Director at PE International.

Figure and table captions

Figure 1 Location of the studied plots in the Spanish watersheds. Location of watersheds (black straight lines), political boundaries of provinces and agricultural production plots (black dots), modified and adapted from Alcamo and colleagues (2003) and Trueba and colleagues (2000).

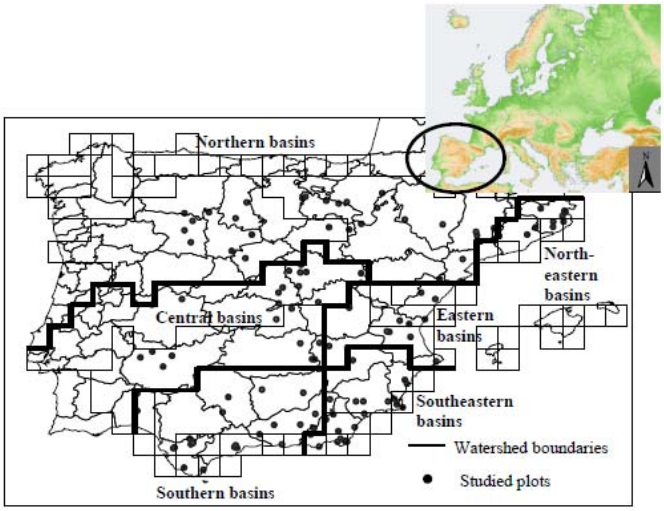
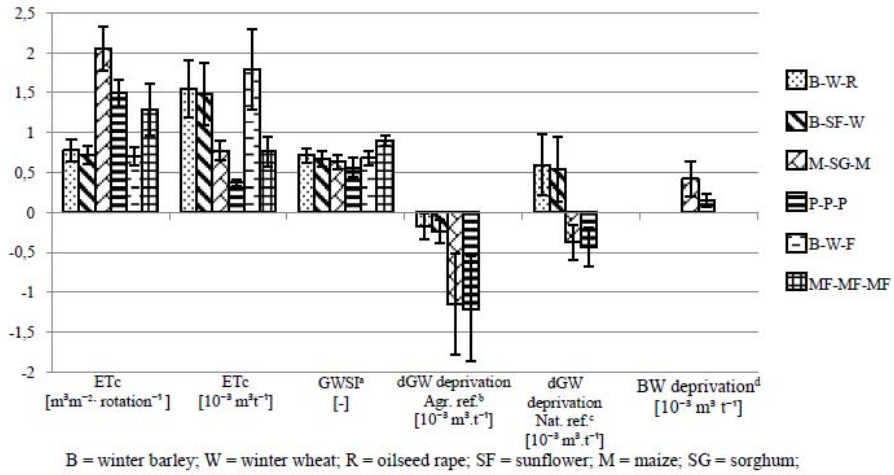


Figure 2 Comparison of water consumption and its environmental effects between energy crop rotations and the reference agricultural and natural systems. Values are the average for all the Spanish watersheds considered in this study. Error bars denote variation between watersheds.



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

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

[b] dGW deprivation under agricultural reference:

WSI×(GW consumption system studied - GW consumption B-W-F rotation)

[c] dGW deprivation under natural reference:

WSI×(GW consumption system studied - GW consumption MF-MF-MF rotation)

[d] BW deprivation: WSI×BW consumption

Figure 3 Location of the most and the least appropriate cultivation watersheds for a) blue water assessment - EI99HA single-score, b) green water scarcity index (GWSI), c) delta green water assessment (dGW), d) land use assessment for rainfed rotations, and e) land use assessment for irrigated rotations. The legend of the figures (high and low) is defined by ranking the impact scores of each watershed.

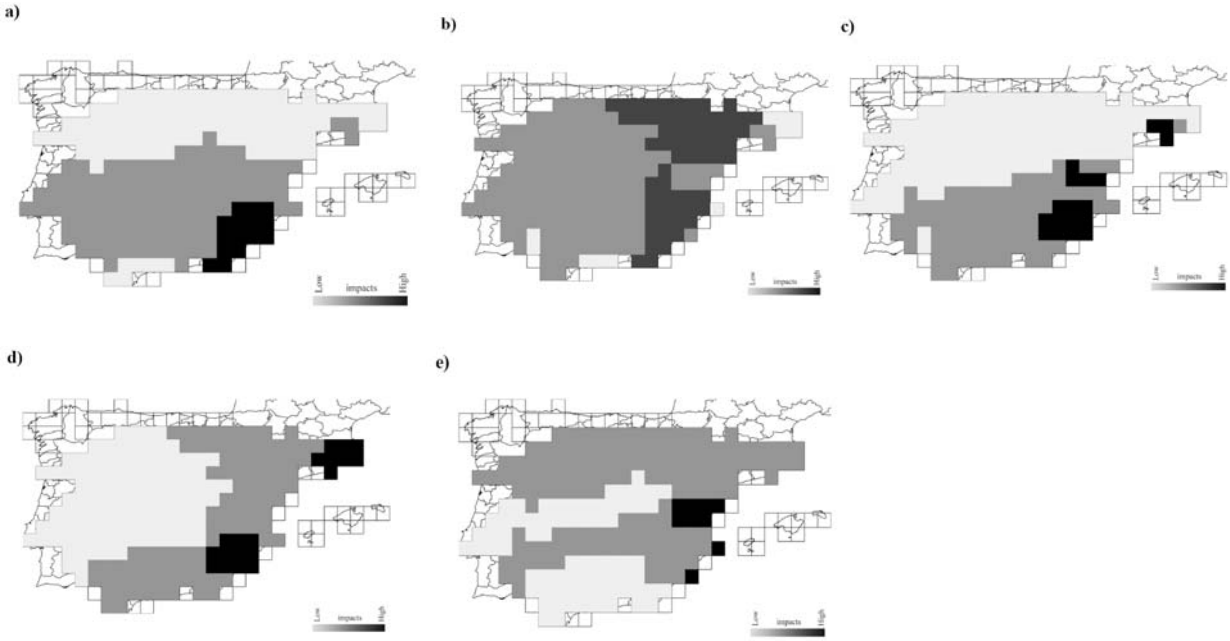


Figure 4 Comparison of land use occupation impacts and blue water consumption impacts based on the aggregated Eco-indicator 99HA scores. Each triangle represents the combination of land/water use damage of a specific rotation in one of the studied watersheds. Triangles below the line (slope=1) indicate that land use impacts are greater than water use impacts for the rotation in the water basin. EI99HA = Eco-indicator 99 hierarchist perspective, average weighting; mP t^{-1} = milipoints t^{-1}; MF = Mediterranean forest.

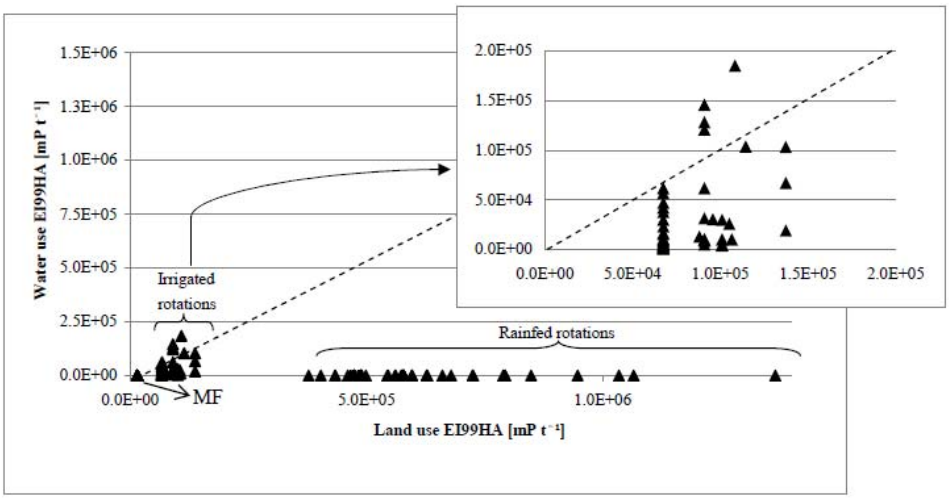


Table 1 Characteristics of the studied rotations and the reference systems. Cells in grey indicate crops produced for potential energy use

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

B = winter barley; W = winter wheat; R = oilseed rape; SF = sunflower; M = maize; SG = sorghum; P = poplar; MF = Mediterranean forest; Pr = precipitation; BW = blue water.

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

Table 2 Impacts of blue water and green water consumption for the assessments per m^2 and per metric ton dry matter, and of land use per metric ton dry matter for 5 relevant Spanish watersheds. Complete list of all basins in tables 3SM to 5SM

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

B = winter barley; W = winter wheat; R = oilseed rape; SF = sunflower; M = maize; SG = sorghum; P = poplar; MF = Mediterranean forest; PDF = potentially disappeared fraction of species; EI99HA = Eco-indicator 99 hierarchist perspective, average weighting.

[a] Basin-ID according to the WaterGAP2 global model (Alcamo et al. 2003).

[b] WSI×BW consumption.

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

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

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