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Mapping water provisioning services to support the ecosystem–water–food–energy nexus in the Danube river basin

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ABSTRACT

Water, food and energy are at the core of human needs and there is a boundless complex cycle among these three basic human needs. Ecosystems are in the center of this nexus, since they contribute to the provision of each component, making it imperative to understand the role of ecosystems in securing food, water and energy for human well-being. In this study we aimed to map and assess water provisioning services and associated benefits to support the ecosystem–water–food–energy nexus by taking into account environmental flow requirements for riverine ecosystems using the hydrological model Soil and Water Assessment Tool (SWAT). We developed a framework that includes indicators of renewable water (capacity of ecosystem to provide water) and water use (service flow) and we applied it in the Danube river basin over the period 1995–2004. Water scarcity indicators were used to map the possible water scarcity in the subbasins, and analyze the spatial match of water availability and water use. The results show that modelling is instrumental to perform the integrated analysis of the ecosystem–water–food–energy nexus; and that spatial mapping is a powerful tool to display environmental availability of water provisioning and regulatory services delivered by ecosystems, and can support the nexus analysis.

1. Introduction

In the coming decades population growth and changes in diet will increase the global food demand and consequently the water demand for agricultural production (de Fraiture and Wichelns 2010). Water, food and energy are at the core of human needs and there is a boundless complex cycle among these three elements which has been recently referred to as the water–food–energy nexus. To produce food, water and energy are needed; while to produce energy, water is required; and to access water, energy is almost always needed (i.e. to run pumps). Due to the complexity of relationships among these three elements, there is a need for them to be considered simultaneously in decision-making (Bazilian et al., 2011; Howells et al., 2013).

Ecosystems are at the center of this nexus since they are involved in the production of water, food and energy, making it imperative to understand their role in providing these benefits for human well-being. The benefits that humans derive from ecosystems are referred to as ecosystem services (MEA 2005; TEEB, 2010). To indicate that ecosystems have a crucial role in providing water, food and energy, and they have to be taken into account in the nexus, we refer to the ecosystem–water–food–energy nexus.

According to Costanza et al. (2014) ecosystem services contribute at least 125–145 trillion US $ per year to the global economy and to the livelihood of more than a billion poor people in the world. Due to the value of ecosystem services to humans, governments around the world are beginning to recognize the importance of investing in safeguarding ecosystems as opposed to industrialized solutions to their problems. Ecosystem services are increasingly being incorporated into environmental policies, especially in Europe. For instance, the EU Biodiversity Strategy to 2020 explicitly includes in its targets the importance of ecosystems in delivering ecosystem services (European Commission, 2011). Action 5 of Target 2 of the Strategy asks Member States to map and assess the state of ecosystems and their services in their national territory by 2014 with the assistance of the European Commission (2011). In addition, there are plans to consider ecosystem services in the implementation of the EU Water Framework and Floods directives, as suggested by the Blueprint to

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safeguard Europe's water resources (European Commission, 2012), to include the understanding of services provided by water bodies. To implement these policies, spatial information on ecosystem services must be generated and robust methods for mapping and assessing them must be developed.

In response to the various policies, maps of ecosystem services are being generated in Europe and around the world. The number of mapping studies produced is increasing significantly from one year to the next (Egoh et al., 2012; Liquete et al., 2013). Recently a number of studies have been performed to evaluate and map the supply and demand of different ecosystem services at local or regional (Nelson et al., 2009; Willemen et al., 2010; Burkhard et al.,. 2012a; La Notte et al., 2012), national (Egoh et al., 2008; Nedkov and Burkhard 2012 and Burkhard et al., 2012b), and continental or global scale (Naidoo et al., 2008; Schuol et al., 2008; Bateman et al., 2011; Schulp et al., 2012; Crossman et al., 2013; La Notte et al., 2015). In addition, tools for mapping ecosystem services are also being developed such as INVEST, ARIES, SOLVES (Tallis et al., 2011; Villa et al., 2009; Sherrouse and Semmens 2010). Several indicators are also being proposed as proxies that could be used as stand-alone or that could feed into the modelling tools to generate spatial information on various ecosystem services. In Europe, in response to Action 5 of Target 2 of the EU biodiversity strategy for 2020, there is also an on-going initiative "Mapping and Assessment of Ecosystems and their Services (MAES)" aimed at developing a harmonized analytical framework for mapping ecosystem services across EU Member States (Maes et al., 2013). As a first step, MAES has collected possible indicators for mapping ecosystem services across Europe (Maes et al., 2014).

While all the mapping work is ongoing, much attention has been focused on the regulating and provisioning ecosystem services and several indicators have been used to map these services. The most frequently mapped regulating service at present is climate regulation (e.g. carbon storage) and services related to water provision and regulation. The water-related services have received increased attention as water is vital to life on Earth and its value is easily appreciated by humans. Freshwaters were highlighted as an ecosystem that provides different services by the Millennium Ecosystem Assessment, one of which is water provisioning, now well beyond levels that can sustain current and future demands (MEA, 2005) (for definition of terms see also Table S1 in the Supplementary material). Studies that have mapped water provisioning have used proxies such as rivers, lakes and other open water bodies and distance to assess these water bodies (e.g. Helian et al., 2011; Brenner et al., 2010; Shi et al., 2009), outputs from hydrological models such as runoff (e.g. Van Jaarsveld et al., 2005; Egoh et al., 2008; Van Wilgen et al., 2008), precipitation (e.g. Chan et al., 2006) or models that includes precipitation, stream-flow, vegetation and other environmental parameters (e.g. Lara et al., 2009). A few studies have considered both water supply and demand (Naidoo et al., 2008; De Roo et al., 2012) and few of these studies have mapped water provisioning at a continental scale. In addition, most mapping approaches for water provisioning do not necessarily take into consideration environmental flow requirement, that is, the fraction of water required for the maintenance or sustainability of freshwater-dependent ecosystems (GWSP, 2008a; Smakhtin and Anputhas, 2006); also indicated as ecosystem water requirement.

The integrated analysis of the ecosystem–water–food–energy nexus and the risk of water scarcity can only be established by evaluating the levels of water available from the ecosystem versus the different uses, while taking into consideration the water also for the ecosystem. This corresponds as well to the analysis of water provisioning services (water provided by the ecosystem for different uses).

This study aims to assess water provision for major sectors considering the ecosystem–water–food–energy nexus. The assessment is spatially explicit, and includes the appreciation of the role of the ecosystem to support the delivery of the services. The approach is developed in the Danube river basin. In this study, we first developed a conceptual framework for mapping and assessing water provisioning services. Second, we mapped the service of water provisioning at subbasin level, using indicators of renewable water and water use (thus moving away from the traditional practice of using coarse scale proxies representing whole watershed or country). In the analysis, the renewable water was estimated by the hydrological model SWAT (Soil and Water Assessment Tool). Third, we used water scarcity indicators to map the presumptive water scarcity in the subbasins, also reserving an estimated proportion of water for aquatic ecosystems (environmental flow requirements). Lastly, we mapped the cost of the water used by the different sectors (public, agriculture, industry).

2. Method

2.1. Study area

The Danube river basin has a surface area of 802,500 km^2 covering 10% of the territory of continental Europe making it the most international drainage basin shared by as many as 19 countries (Fig. 1). However, only 14 countries contain more than 2000 km^2 of the basin area and are considered as Danube countries, being contracting parties of the International Commission for the Protection of the Danube River (ICPDR) (Schreiber et al., 2003; ICPDR, 2009). The ICPDR is an international organization

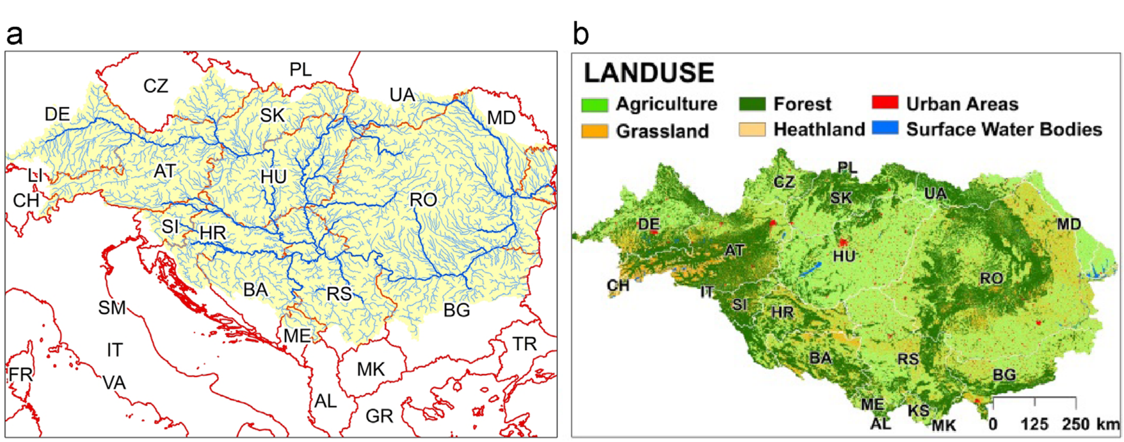

Fig. 1. Map of the Danube river basin: (a) countries in the river basin; (b) land-cover (CLC, 2006).

Table 1
Area share of countries in the Danube river basin and percentage occupied by the Danube river basin in each country.

Country	Area share of countries in the Danube river basin in km^2 (%)	% of Danube river basin in the country	Country	Area share of countries in the Danube river basin in km^2 (%)	% of Danube river basin in the country
Montenegro	7075 (0.88)	51.2	**Slovak Republic**	47,084 (5.86)	96.0
Moldova	12,834 (1.54)	35.6	**Bulgaria**	47,413 (5.87)	43.3
Slovenia	16,422 (2.01)	81.0	**Germany**	56,184 (7.01)	16.8
Czech Republic	21,688 (2.70)	27.5	**Serbia**	81,560 (9.46)	92.3
Ukraine	30,520 (3.75)	5.40	**Austria**	80,423 (10.05)	96.1
Bosnia and Herzegovina	36,636 (4.71)	74.9	**Hungary**	93,030 (11.59)	100.0
Republic of Croatia	34,965 (4.76)	62.5	**Romania**	232,193 (29.00)	100.0

consisting of 14 cooperating states and the European Union with the aim to develop joint management plans relevant to socio-economic and environmental issues in the Danube area.

Among these Danube countries Romania and Hungary are entirely located in the basin with the highest share of surface area: 29% and 11.5%, respectively (Table 1). The countries with the smallest area are Montenegro (0.8%) and Moldova (1.5%). In the IPCDR the Danube river basin is divided into 20 major subbasins (ICPDR, 2012).[1]

Dominant land cover types in the basin are agriculture (42%) and forest (35%) (Fig. 1). The rest of the basin is either covered by grasslands and heathlands (16%), urban areas (5%) or water bodies (less than 2%) (EEA, 2013). Most of the basin is in the lowlands (67%) (areas below 500 m according to the approach of Bertrand et al. 2011). Precipitation in the basin is variable with the highlands receiving the highest annual average precipitation between 1000–3200 mm per year and the lowlands between 350 and 600 mm per year (Schreiber et al., 2003; ICPDR, 2013a).

2.2. SWAT hydrological model

Outputs of models have been used to map water related ecosystem services. Examples include GREEN (La Notte et al., 2015; Maes et al., 2012), ACRU (Egoh et al., 2008) and SWAT (Notter et al., 2012). In this study, we used the SWAT hydrological model, developed in the USA (Arnold et al., 1998) and widely used around the world (e.g. Abbaspour et al., 2015; Lin et al., 2015; Scherer et al., 2015; Yen et al., 2015; Pagliero et al., 2014; Gassman et al., 2007), to map the service of water provisioning. We selected SWAT because it is an integrated model used particularly for simulating water quantity, water quality and soil erosion in river basins, and for testing best management practices (Arnold et al., 1998; Gassman et al., 2007; Prochnow et al., 2008; Schilling et al., 2008; Schuol et al., 2008; Notter et al., 2012; Pagliero et al., 2014). SWAT requires detailed spatial information on various environmental variables. The hydrological processes can be calibrated per subbasin. The model inputs (Arnold et al., 1998) include the digital elevation model (DEM), land use, land cover, soil type, soil hydrological properties, time series of climate data, reservoirs, and land management. For calibration, the model requires a time series of water discharge and water quality data. Among the outputs, SWAT provides average daily flow, groundwater recharge, surface runoff, subsurface flow, concentration of sediments, and the amount of nitrogen, phosphorus and pesticides transported with water in each time step (day, month or year) (Arnold et al., 1998).

[1] The 20 major subbasins of the ICPDR are: Upper Danube, Inn, Austrian Danube and Drava in the western part; Sava, Velika Morava, Danube in the southern part; Morava, Vah-Hron-Ipel, Tiza, Prut in the northern part; Pannoian Danube, Middle Danube, Jiu, Olt, Arges-Vedea, Siret, central part of Tiza in the central part and Buzau-lalomita, Banat, Delta-Liman Dobrogea-Litoral in the eastern part (ICPDR, 1999).

Table 2
The geo-datasets (Bouraoui and Aloe, 2009) used in the SWAT modelling.

Data	Main Source	Time	Resolution-scale
SWAT model inputs			
DEM	SRTM	2009	100×100 m
Soil	HWSD	2008	1×1 km
Landuse	CAPRI (2012)	2004	1×1 km
	CSGE	2008	
	CLC	2000	
	GLC	2005	
River network	CCM2	2007	
Subbasins	HydroEurope geodatabase	2009	Average size ≈ 180 km^2
Reservoirs & lakes	GLWD, CCM2, GranD	2004	
Climate data	MARS	1980–2009, daily time series	25×25 km
	EFAS-METEO		5×5 km

SRTM: Shuttle Radar Topography Mission; HWSD: Harmonized World Soil Database (Nachtergaele et al., 2012); CAPRI: Common Agricultural Policy Regionalized Impact Modeling System (Britz, 2004); CSGE: Center for Sustainability and Global Environment (Monfreda et al., 2008); CLC: Corine Land Cover; GLC: Global land Cover (Bartholome and Belward, 2005); CCM2: Catchment Characterization Modeling Version 2 (Vogt et al., 2007); GLWD: Global Lakes and Wetland Database (Lehner and Doll, 2004);GranD: Global reservoir and Dams database (Lehner et al., 2011); MARS: Monitoring Agricultural Resources (Rijks et al., 1998); EFAS-Meteo: A European daily high-resolution gridded meteorological data set for 1990 – 2011 (Ntegeka et al., 2013).

Furthermore, the model estimates evaporation from river and reservoir surfaces, available soil water content, evapotranspiration, water loss from stream-bed or reservoir-bed by transmission as intermediate output.

For this study we used the results of Pagliero et al. (2014), who set and calibrated the model in the Danube region. In Pagliero et al. (2014) the Danube basin was divided into 4663 subbasins, with average size of 180 km^2, according to the catchment discretization of Europe of Bouraoui and Aloe (2009), based on the CCM2 (Vogt et al., 2007) (Table 2) Reservoirs or lakes greater than 20 km^2 were included in the modelling. Pagliero et al. (2014) divided the Danube river basin in regions with similar hydrological characteristics by cluster analysis. The SWAT hydrological parameters were calibrated in subbasins of different clustered regions for which measurements were available, using Sequential Uncertainty Fitting, ver. 2 (SUFI-2; Abbaspour et al., 2004) and SWAT Calibration Uncertainty Programs (SWAT-CUP; Abbaspour, 2007). Then the calibrated set of parameters of each hydrological cluster was extrapolated to the ungauged subbasins within the same cluster. This approach was necessary to represent the spatial variability of hydrological properties in the wide region of the Danube basin, considering the limited availability of measured data. According to the authors, the results of the model simulation of water flow for the whole basin are fine consistent both in time

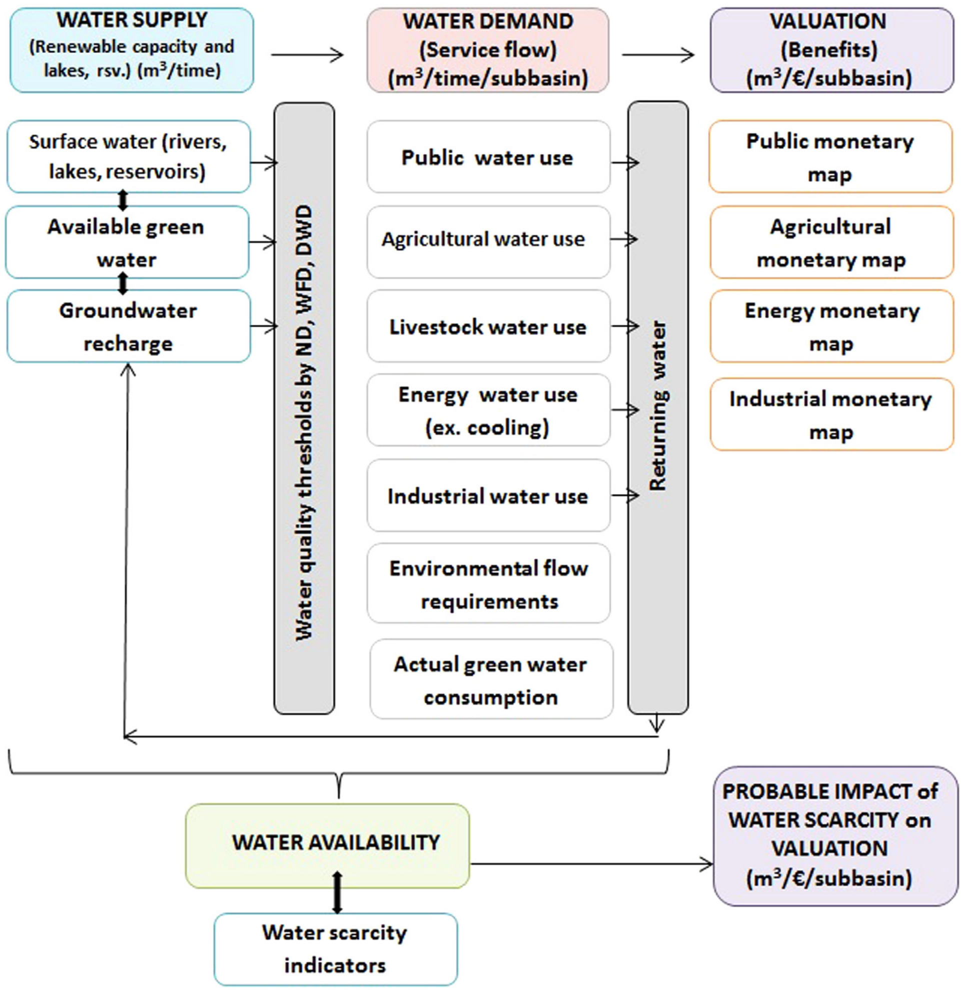

Fig. 2. Conceptual framework for assessing water provisioning services in the ecosystem–water–food–energy nexus adopted in this study (see also Grizzetti et al., 2015) (ND: Nitrate directive; WFD: Water Framework Directive; Drinking Water Directive)..

and in volume with respect to the observed discharge ($R^2_{avg}=0.78$, Nash–Sutcliffe efficiency (NSE_{avg})=0.73 for calibration 1995–2000; $R^2_{avg}=0.77$, $NSE_{avg}=0.63$ for validation 2000–2004). The details of the biophysical modelling can be found in Pagliero et al. (2014).

2.3. Assessing water provisioning services in the ecosystem–water–food–energy nexus

We used the outputs of the SWAT model that are related to water quantity to map water provisioning services. The spatial units of the analysis were the 4663 subbasins (as described in Pagliero et al., 2014, see previous paragraph) and the temporal frame refers to average values for the period 1995–2004.

Conceptual framework for assessing water provisioning services was developed in four steps according to Grizzetti et al. (2015) (Fig. 2):

1. The first step consisted of mapping the *capacity* of the ecosystem to provide water (Ecosystem service capacity). We computed the renewable water (or natural water yield), which is defined as the sum of the long-term average annual flow of rivers and the recharge of aquifers generated from precipitation (FAO, 2014) (see also Table S1 in Supplementary material for the definitions of terms used in this article). The average annual stream flow and groundwater recharge simulated by the SWAT model were used as input to map the water yield, and the available soil water content simulated by SWAT was considered as proxy for the potential green water.
2. In the second step we quantified the water provisioning for different sectorial uses, which represents the actual *flow* of the ecosystem service (Ecosystem service flow). For the purposes of this study, we assumed the use of water per sector to be equivalent to the actual water withdrawals. Therefore we considered the withdrawals of water for cities, industry, energy, irrigation, and livestock to represent the water uses. We also considered the environmental flow requirements and the green water (water directly used and evaporated by non-irrigated agriculture, pastures and forests, GWSP, 2008b) to cover the whole spectrum of water uses within the ecosystem–water–food–energy nexus.
   An important concern in water provisioning services is the quality of water (Kremen, 2005; Kremen and Ostfeld, 2005), since each use has specific water quality standards. In the conceptual framework, we included a filter for water quality, based on nitrogen and phosphorus concentration, but it was not considered in the analysis, as the water quality parameters were not calibrated in Pagliero et al. (2014).
3. In the third step, two *water scarcity indicators*, the Falkenmark water scarcity indicator (FLK) (Falkenmark, 1989) and the Water Exploitation Index Plus (WEI +) (EEA, 2010; De Roo et al., 2012), were calculated to analyze the water availability. These

indicators combine the supply and demand side of the water provisioning services. To include the role of the ecosystem in supporting the delivery of the services we added the environmental flow requirements for riverine ecosystems in the computation of the water scarcity.

4. In the last step we made a simplified attempt to *value* water provisioning services within the Danube river basin using the market price of supplied water per country.

The detailed methodology for each step is described in the following paragraphs.

2.3.1. Renewable water (ecosystem service capacity)

We first analyzed the capacity of the ecosystem to produce water (renewable water, Fig. 2) based on SWAT long-term annual average estimations of surface and groundwater recharge in each of the 4663 subbasins. The total renewable water (TRW) was calculated using the annual average deep aquifer recharge and the entering stream flow in each subbasin (average of the period 1995–2004).

We then computed the total renewable water per land cover type by aggregating the values per major land use classes. We also addressed the renewable water per country of the Danube basin to represent the proportion of water generated in their territory.

Amount of water held in the soil between the field capacity and permanent wilting point is considered to be available soil water for the plant uptake. It is controlled by soil type and physical properties, e.g. texture, slope, infiltration rate and soil temperature (Neitsch et al., 2009). As available soil water content is important for sustaining vegetation, we considered it as potential green water that can be used by natural plants and agricultural crops. Since available soil water content is continuously varying depending upon climatic conditions in time, evaluating and displaying it in a time series would be the most appropriate. Nevertheless, in order to emphasize its importance on ecosystems, the annual mean of available water content was assumed as potential green water in this study. Potential green water also plays a role on irrigated land, because irrigation water is supplied especially where precipitation is not sufficient to sustain optimal crop growth.

2.3.2. Water use (ecosystem service flow)

Water provisioning for the different uses in the ecosystem–water–food–energy nexus (water use, Fig. 2) was assessed using water withdrawal maps for sectorial uses. For each sector, water withdrawals were assumed to be equivalent to the water use with the assumption that once water is withdrawn it is delivered directly to the sector without leakages. The water use was calculated in volumetric unit in each subbasin per each sector, including public, industrial, energy, livestock and agricultural sectors, using the maps developed by Vandecasteele et al. (2013), Mubareka et al. (2013) and Bouraoui and Aloe (2012) at the European scale (Table 3).

Table 3
The geo-datasets used in mapping water uses (Source data: Bouraoui and Aloe, 2012; Vandecasteele et al., 2013; FAO, 2012; Mubareka et al., 2013).

Data	Main Source	Time	Spatial resolution
Irrigation water use	JRC	Annually average (1995–2006)	10 × 10 km
Public, industrial, energy water use	JRC	Monthly (2006)	5 × 5 km
Livestock water use	FAO JRC	Daily (2006)	5 × 5 km

In Vandecasteele et al. (2013) public water withdrawals were mapped by allocating national statistics on actual water use per capita (EUROSTAT data) to the spatial location of users, defined by the land use (Refined Corine, Batista e Silva et al., 2013) and population density maps (Batista e Silva et al., 2013). In addition, a tourism density map was computed and a proportionally higher water use assigned per (Vandecasteele et al., 2013) tourist. Industrial water withdrawals were mapped by disaggregation of the country-level statistics to the industrial land use classes (Vandecasteele et al., 2013). In a similar way, country-level energy withdrawals were disaggregated using the locations of thermal power stations (EPRTR dataset), which were assumed to account for the majority of water use in the sector (Vandecasteele et al., 2013). Livestock water uses were provided by Mubareka et al. (2013), where withdrawals are mapped considering the livestock density and water requirements per livestock type. Finally, since we used the SWAT model results without irrigation, we considered the water use for irrigation computed by the EPIC model by Bouraoui and Aloe (2012).

Most water abstracted is given back to the water system, although in a different location, which may generate problems of water scarcity locally and issues related to the degradation of the quality of the resource. According to the water footprint approach (Vanham and Bidoglio, 2013), the quantity of abstracted water in Europe that is lost before returning to the water system is around 10% for public water, 5% for the energy sector, and about 50% for agriculture, which means that large part of the water withdrawals returns back to the system. This is considered in the conceptual model, which represents the water cycle calibrated against the water flow observed in various points of the basin.

To include the role of the ecosystem, we added the environmental flow requirements for riverine ecosystem in the water uses. The environmental flow requirement (EFR) is an indicator referring to the fraction of water required for the maintenance or sustainability of freshwater-dependent ecosystems in river basins (GWSP, 2008a; Smakhtin and Anputhas, 2006). In order to estimate the EFR, the Q90 percentile (Smakhtin 2001) of river discharge was calculated at the outlets of 25 subbasins (randomly selected) based on the SWAT model simulation for period of 1995–2004. The flow computed in this way represented on average 15% of the total flow, and we considered this fraction as the minimum flow level to be reserved for EFR of the riverine ecosystem.

To consider the water consumed by natural plants and crops, we used actual green water consumption as a proxy (Fig. 2). Green water consumption (GWSP, 2008b; Rockstrom et al., 2009) by natural plant and also by agricultural crops was calculated and mapped using the evapotranspiration estimated by SWAT from the (non-irrigated) model outputs (1995–2004).

2.3.3. Water scarcity indicators

To analyze the water availability by integrating the different uses of the ecosystem–water–food–energy nexus, two indicators of water scarcity were used: the Falkenmark indicator (FLK) and Water Exploitation Index Plus (WEI+). The first indicates the water available per capita, while the second addresses the total sectoral pressure to the water resource. Both indicators were computed per each subbasin including the environmental flow requirement in the calculation.

According to FLK indicator, water stress is defined as a situation where there is not enough water for human water requirements (Falkenmark, 1989; UN-WBCSD, 2006; Brown and Matlock, 2011). Defining thresholds for stress in terms of available water per capita requires assumptions about water use and its efficiency. Nevertheless, it has been proposed that when annual water availability per capita is less than 1700 m^3, countries begin to experience

Table 4
Thresholds of water stress and water scarcity (Brown and Matlock, 2011) for the FLK (Falkenmark, 1989) (left) and the WEI+ (EEA, 2010; De Roo et al., 2012).

FLK (m^3 per capita)	Category /Condition	WEI+(%)	Category /Condition
> 1700	No stress	< 10	No stress
1000–1700	Stress	10–20	Low stress
500–1000	Scarcity	20–30	Moderate stress
< 500	Absolute scarcity	30–40	Scarcity
		> 40	Severe scarcity

periodic or regular water stress. Below 1000 m^3, water scarcity begins to hamper economic development and human health and well-being (Falkenmark, 1989; UN-WBCSD, 2006). The FLK indicator (m3 water per capita) was calculated for each subbasin considering the total renewable water (TRW, m^3), environmental flow requirement (EFR, m^3) for riverine ecosystems, and population per subbasin (Eq. (1)):

$$FLK = [(TRW) - (EFR)]/\text{Population} \tag{1}$$

where TRW is the sum of groundwater recharge and runoff of the subbasins. The values of FLK were classified using the thresholds of water stress and water scarcity proposed by Falkenmark (1989) to map potential water stress and environmental water scarcity in the Danube's subbasins (Table 4).

The second indicator of water scarcity that was used in this study is the Water Exploitation Index Plus (WEI+) (EEA, 2010). It has been applied in studies on water scarcity at the European scale, particularly in the Blueprint for Europe's Waters (De Roo et al., 2012). WEI+ is an indicator of the level of pressure that human activity exerts on the natural water resources, helping to identify areas prone to suffer water stress (De Roo et al., 2012).

In this study WEI+ was computed for each subbasin as the ratio between water abstractions minus return water (total sectoral water withdrawals, including domestic, agriculture, livestock, energy and industrial uses, in m^3) and total available water (in m^3). The latter is defined as sum of external inflow and internal flow in the subbasin minus the environmental flow requirement (EFR) for riverine ecosystem, as shown in Eq. (2) (using a formulation similar to the one used by Smakhtin, 2001, and Brown and Matlock, 2011). WEI+ is calculated considering the net consumed water (EEA, 2010; De Roo et al., 2012):

$$WEI+ = (\text{Totalsectoralwaterwithdrawals} - \text{Returns})/[(\text{externalinflow} + \text{internalflow}) - EFR] \tag{2}$$

External inflow is the water coming from the upstream, while *internal flow* is the water generated in the subbasin contributing to the stream. In our computation *external inflow+internal flow* corresponds to the TRW. Thresholds for water scarcity level according to the values of WEI+ are reported in Table 4.

A fraction of sectoral water withdrawal is consumed and is no longer available because it is evaporated, transpired by plants, incorporated into products or crops, or consumed by people or livestock. The rest of the water withdrawal returns to the environment with a different quality. The return water for each sector is calculated by using the ratios from literature. The ratio of return water for each sector considered in this study was: 0.2 for public, 0.15 for livestock, 0.15 for industrial, 0.94 for irrigation water withdrawal in summer season and 0.75 for spring season, and 0.33 for energy water withdrawal when water returns to a surface water body and 0.025 in the other cases (EEA, 2010; De Roo et al., 2012). In addition to the long term annual WEI+, we calculated long term (1995–2004) monthly WEI+ in order to see the seasonal water scarcity variation in the Danube region.

2.3.4. Valuation

Freshwaters can be associated with a wide range of benefits to humankind and they have complex structure to value. Although one may put a value on freshwater based on water use by humans, it is important to note that freshwater provides more services than just usable water (e.g. food provision in terms of fish, habitat provision and recreational value). Some water services have well-known market values, such as water provisioning for different uses, but other key services, such as provisioning of habitat, climate regulation or water purification, do not. Valuing the benefits of water provisioning services should then embrace the market and non-market values. However this was not possible in the context of this study because of limited data availability. For this reason, we limited our analysis to water provisioning services for which there is a market. We then restricted our valuation exercise to the water provisioning service for *industry*, *agriculture* and *households*. In the discussion of the paper we will explain how non-market values could be integrated to this framework.

For agriculture and industry, water is a physical input (or production factor) along with labor, energy, land or capital. It is assumed that those factors are substitutable (to a given degree), so that the choice of how much of each to be used depends on the relative input prices. Adopting a marginalist point of view, the value of water for agriculture and industry is then equal to the marginal contribution to the economic profit resulting from the use of an extra unit of water in the production process. Estimating the marginal value of water requires identifying production technologies of farmers or firms. Production technologies may be represented by production functions, cost functions or profit functions, see (Varian, 2009). In the existing literature, the most common approach for measuring the marginal value of water has been by estimating an agricultural cost function (dual approach) for farms. With the cost function approach, it is assumed that firms choose water and the others inputs in order to minimize the production cost for a given level of output. Some specific applications of the cost function approach for valuing water include Renzetti (1988) and Dupont and Renzetti (2001) in Canada, Reynaud (2003) in France, and Féres and Reynaud (2005) in Brazil. With the production function approach (primal approach), the marginal value of water is obtained from the estimated production function, see Wang and Lall (2002) for a specific application for valuing water used by industrial plants in China. Lastly, the profit function approach consists in estimating a profit function but, to our best knowledge, no empirical application including water as a production factor has been published.

Water is not a production factor for households, it is a good that is consumed and valued as any other good.[2] Water consumption provides some welfare to households that can be approximated by the consumer surplus (Varian, 2009). To estimate the value of water for households, one way commonly used in the empirical economic literature consists in estimating a residential water demand function that relates the quantity of water consumed by households to a set of determinants including the water price paid by households and user's characteristics. The household water demands may be estimated from aggregated or from household-level data, but the estimation process is typically highly data-intensive. Estimations have been undertaken for over 35 countries including the United States, Australia, Cambodia, Saudi Arabia, Italy, France and Portugal (see Gardner, 2011).

These different methodologies for valuing water for agricultural, industrial and residential uses could not be applied because of lack of harmonized data in the Danube region. As a result,

[2] In most countries, water used for satisfying basic needs represents only a small share of the total household water use.

we propose here a valuation method that relies on a *cost-based* approach. It consists in using expenses incurred for the use of the resource as a proxy of its value for those users (Nunes and van den Berg 2001; TEEB, 2010; Brouwer et al., 2013). The underlying assumption is that, to some extent, costs and values are linked. In the specific case of the Danube river basin, Getzner (2009) indicates that such an approach has already been implemented to value water provision and quality regulation services provided by two national parks in Poland and the Slovak Republic. Some important limits of cost-based approaches should however be stressed. First, the cost of providing water may only represent a portion of the full water services provided by the natural resource. Thus, the benefits approximated by costs can be understated. Second, cost-based approaches do not account for heterogeneity in individual preferences and their spatial impact on the willingness to pay.

We have collected local or national water fees and tariff systems to produce maps of costs charged by each potential sectorial water user (households, industries and agriculture). The basic concept behind this is that when people purchase something they disclose that they are willing to pay to have it, while they may pay more depending on their varying preferences (King et al., 2000). When people use water at home they disclose their willingness to pay at least with the money they spent to have the water service. In the same way, farmers or manufacturers optimize the use of water as any input, consuming the resource as long as the marginal gross revenue they get from its use is higher than its cost. Keeping this approach the economic value maps of sectorial water uses have been created for the Danube river basin taking into account the quantity of the water used and the marketing prices charged in each country, using the administrative water service costs of each country available by different data sources (Klarer et al., 1999; Van Den Berg and Danilenko, 2011; OECD, 2009; GDP, 2012) (Table 5).

3. Results

Average annual renewable water was analyzed considering the main land covers: forest, agricultural land, grassland, heathland and urban area (Figs. 1b and 3a), and also in highlands and lowlands (Fig. 3b). The spatial distribution of the runoff coefficient (ratio between runoff and precipitation per subbasin) was calculated to examine the land use effect on the water yield (Fig. 3c). The highest level of water yield (surface and groundwater) originates in the forest areas (73%), in direct proportion to the precipitation they receive. This is not surprising as forests cover 35% of the Danube river basin, receive 58% of precipitation and have higher runoff coefficient compared to other areas of the basin (Fig. 3c). In contrast, only 17% of renewable water is produced in the agricultural lands despite the large surface they cover (42%) and the fact that they receive about one third of the precipitation of the basin. On the other hand agricultural lands have relatively higher available soil water content (37%). The rest of renewable water is provided by grasslands and heathlands or collected in urban areas. As a whole, the major quantity of total renewable water (around 60%) comes from highlands which cover only 33% surface area of the Danube in direct proportion to the precipitation they receives (Fig. 3b).

Considering the geographical distribution of water yield, the model indicates that a large part of surface water is generated in the Upper Danube, Inn, Drava, Sava, Middle Danube, Velika Drova and some subbasins in Olt and Tiza (Fig. 4a). Similarly, a significant amount of groundwater is generated in the Upper Danube, where the Danube starts in German territory, and in some central highlands of the basin, where mostly forested areas are located (Fig. 4b). In total, the greatest amount of water is generated in western and southern subbasins. For example, Austria, located in the western region of the Danube basin, in spite of its small share area (10%) of the Danube, has a 25% share of the total water provisioning capacity. In contrast, Romania, located in the central parts of the basin, occupies 29% of the basin area, but it is only contributing 17% to the total renewable water.

Share of water used by different sectors per land cover type is shown in Fig. 5(a, b) and the spatial distribution of sectorial water uses is presented in Fig. 6. Despite most of water yield is from highlands where forested areas are located, most of the water is used in lowlands (Fig. 5a and b).

In general, all the sectorial water uses concentrate in lowlands following the spatial distribution of major urbanized areas (the correlation is not surprising as the land cover was used for spatially allocating the withdrawals) (Fig. 6a, c, d, and e). A significant amount of water used for the agricultural sector is situated in the Hungarian and Romanian lowlands (Fig. 6b).

Sectorial water uses were aggregated per Danube countries. The water use of Ukraine and Montenegro is less than 1% of the total amount, mostly due to their small area and the lack of withdrawal data in this region in the Danube. The highest industrial water use per unit area is located in Austria (49%), the highest agricultural water use in Romania (45%) and the highest energy use in Hungary (29%) (Fig. 6a, b, d, e). Romania also has the highest livestock water use (27%) (Fig. 6c). The public water use is ranked in descending order by Romania with 24%, Hungary with

Table 5
Sector-specific water prices and GDP of countries in the Danube river basin.

Country	Price of public water use (€/m^3)	Price of agricultural water use (Irrigation + livestock) (€/m^3)	Price of industrial water use (€/m^3)	GDP (Mio €)	Population	GDP/capita
Moldova	0.566	0.210	1.482	5439	3486,000	1560.2
Ukraine	0.381	0.210	1.482	132,150	45,456,000	2907.2
Bosnia Herzegovina	0.312	0.210	0.840	12,998	3847,000	3378.6
Serbia	1.441	0.520	1.441	28,050	7203,000	3894.2
Montenegro	0.489	0.210	1.702	3210	620,000	5177.4
Bulgaria	0.965	0.079	0.008	38,265	7261,000	5269.9
Romania	0.867	0.004	0.867	127,050	19,858,000	6397.9
Hungary	1.874	0.038	1.175	95,175	9894,000	9619.5
Croatia	1.538	0.107	1.460	42,825	4258,000	10,057.5
Slovenia	1.537	0.030	1.188	34,215	2,956,000	11,574.8
Slovakia	2.294	0.071	0.890	68940	5413,000	12,736.0
Czech Republic	2.455	0.210	1.697	147,075	10,519,000	13,981.8
Germany	3.133	0.021	4.376	2550,750	80,640,000	31,631.3
Austria	3.224	1.225	3.336	298,950	8477,000	35,266.0

GDP: Gross domestic product. GDP/ca: Gross domestic product per capita. IBNET (2011), International SAVA River Basin Commission (2011), OECD (1999,2009), GDP (Official Exchange Rate) (2012).

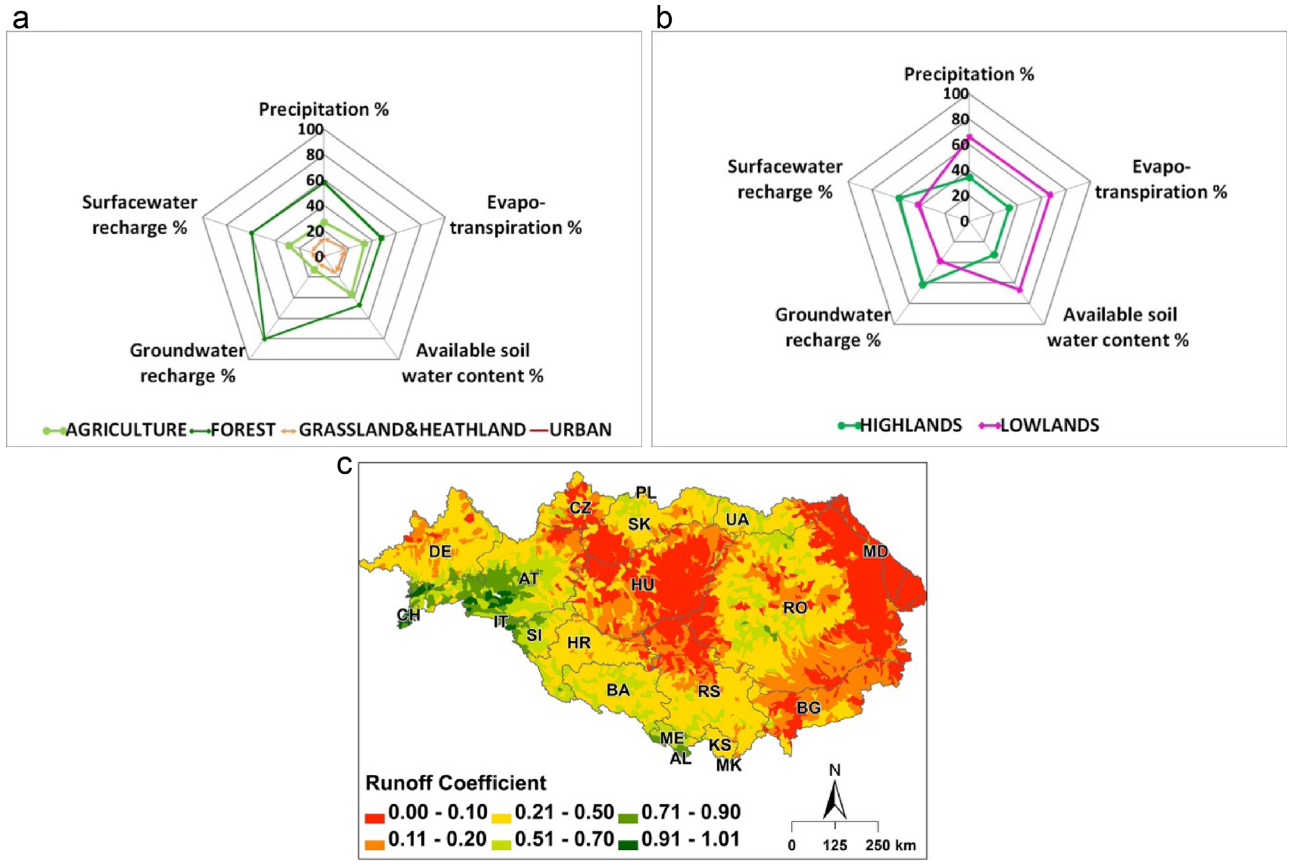

Fig. 3. Distribution of renewable water (percentage) for the whole Danube basin (a) per major land covers and (b) between highlands and lowlands. (c) Runoff coefficient (fraction) distribution in the Danube basin, estimated by the SWAT model by sub-basins used in this study.

11%, Serbia, Austria, and Bulgaria with 10% (Fig. 6a).

Green water consumption, representing the water consumed by agro-ecosystems and natural plants, has the great proportion of Danube freshwater cycle with more than 50% of total (Fig. 6f). The highest available soil water content (considered in this study as the proxy for the available green water) is in Romania with 44% and followed by Hungary with 15% (Fig. 4c). The highest amount of evapotranspiration (considered in this study as the proxy for the actual green water consumption) occurs in Romania and Hungary with 26% and 14%, respectively, in accordance with their share area of the basin and warmer climate conditions, especially in their relatively low elevated territories (Fig. 6f).

Maps of WEI+ and FLK indicators represent the environmental water availability/scarcity that was calculated considering the water yield and use, and deducting the environmental water requirements for riverine ecosystems (Fig. 7a and b). In spite of some differences in several subbasins, these two maps have a similar spatial pattern mostly due to the location of the majority of water withdrawals in urbanized or surrounding areas. The results show that there would be severe or absolute environmental water scarcity in more than 660 subbasins according to FLK (17% of Danube area) and more than 390 subbasins according to WEI+ (12% of Danube area). These subbasins are mostly located in agricultural and urbanized areas (Fig. 7). WEI+ of the long term monthly averages frankly reflects the irrigation water use impact on water scarcity in the Danube region (Fig. 7c). For instance, 22% of Danube area shows environmental water scarcity in the summer season, but only 2% in the winter season.

The subbasins having environmental water scarcity are distributed along the Danube countries in varying rates. More than 500 subbasins, highlighted as at risk of water scarcity, are mostly located within territories of Romania, Hungary, some upward subbasins of Bulgaria, Croatia, Moldova and Ukraine. Without taking into account the dams surrounding some urban subbasins, environmental water scarcity is also reported in Germany, Austria, Hungary and Romania. It is important to note that the analysis and the results reported in Fig. 7 do not consider the water transfers by artificial infrastructures from one subbasin to another.[3]

Economic valuation of public water use shows that the highest amount of money spent per unit area for water provisioning services among the Danube countries are in Germany, Croatia, Austria, Hungary and Romania (Fig. 8a). The amount of money spent in these countries varies between 50 and 90 Mio (million) Euro. The spatial distribution of the value of water is affected both by the quantity of the water used and by the prices charged in each country (see Table 5).

The highest costs per unit area charged for agricultural water use take place in Austria, Serbia and Czech Republic with total expenses varying between 500 and 7000 thousand Euros. Although Romania is the largest consumer of agricultural water in the Danube river basin, Romanians do not bear heavy cost due to very low prices charged to the agricultural sector (Fig. 8b). The highest expenses of industrial water use are found in Austria and Germany in accordance with their significant industrial water use and the relative higher cost of water (Fig. 8c).

[3] According to the Global reservoir and Dams database (GRanD) there are 187 artificial reservoirs and dams in the Danube region of which 40 are used for water supply of urban areas, 24 for irrigation and 92 for producing hydroelectricity (Lehner et al., 2011). For further information on dams in the Danube river basin see also the Danube Basin Analysis (WFD Roof report 2004) http://www.icpdr.org/main/resources/danube-basin-analysis-wfd-roof-report-2004 (accessed on 08-04-2015).

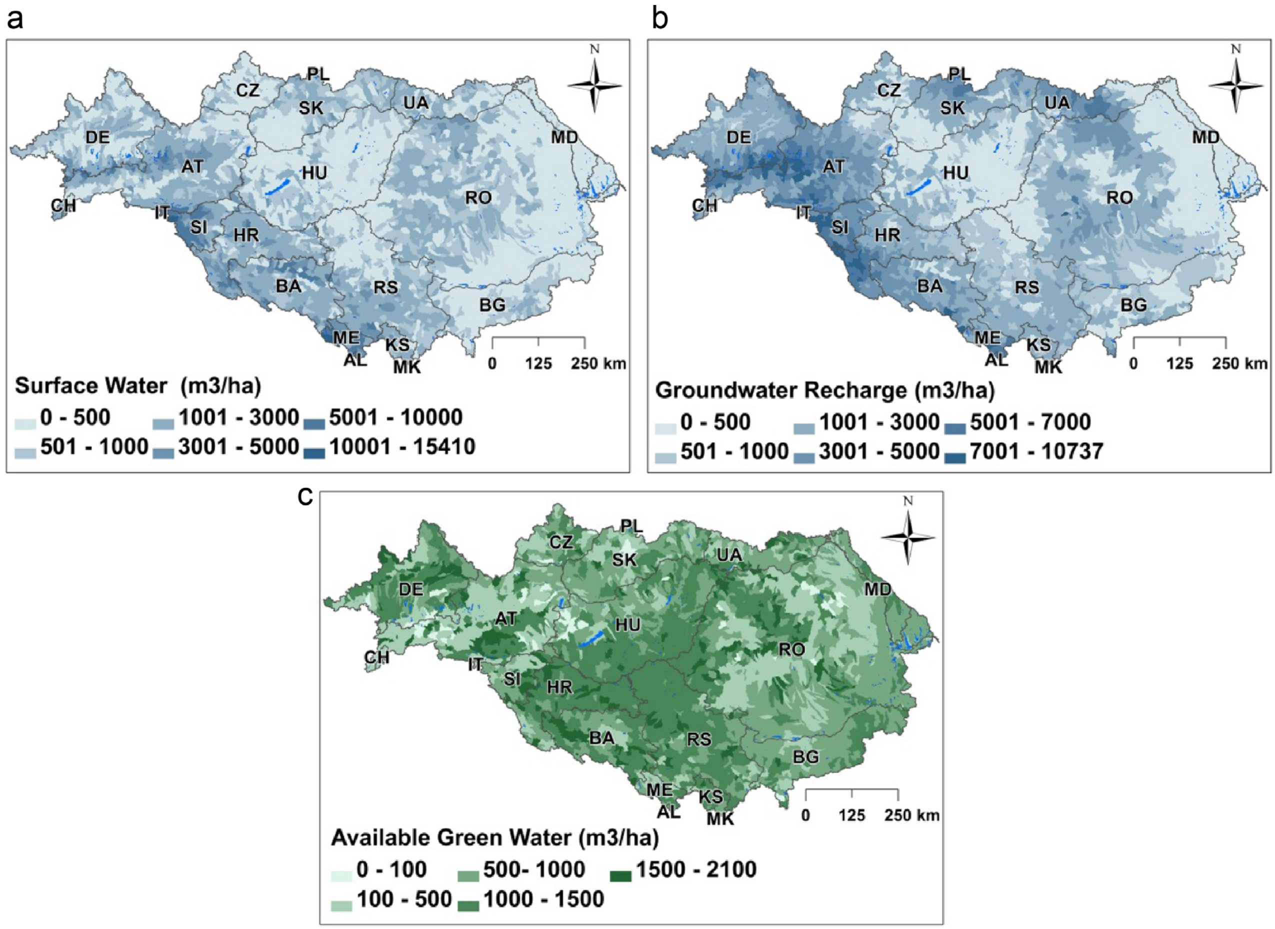

Fig. 4. Maps of renewable water in the Danube river basin estimated by the model SWAT: (a) surface water; (b) groundwater recharge; and (c) available green water. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

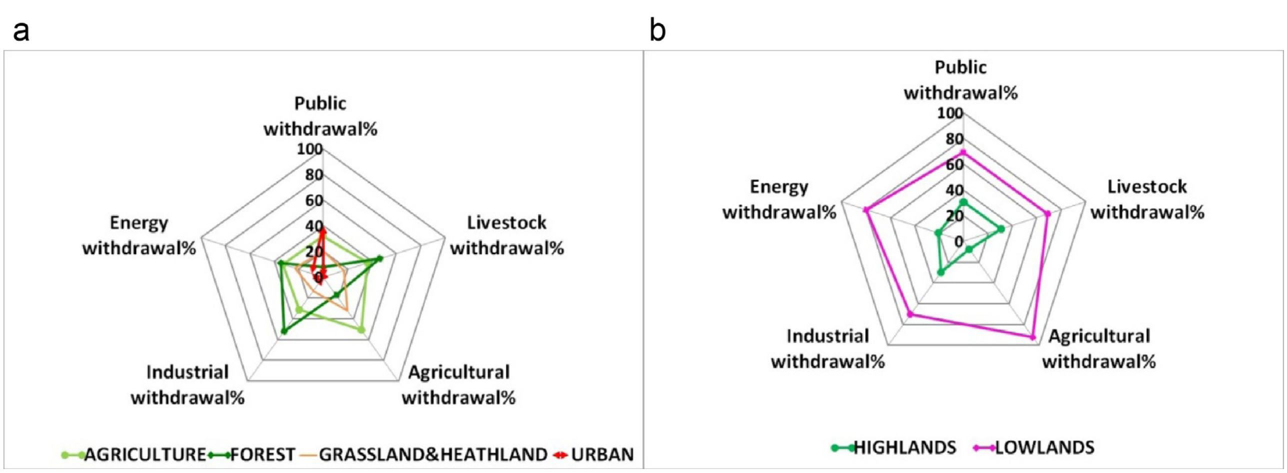

Fig. 5. Distribution of water uses by sectors (a) per different major land covers and (b) between highlands and lowlands.

4. Discussion

4.1. Mapping and assessing water provisioning

Spatially explicit assessments of ecosystem services, which show not only where ecosystem services are produced but where they are used, are necessary for decision making. Nexus approach aims at reducing trade-offs and generating co-benefits for sustainable development in the water, food, energy sectors considering the sustainability of the ecosystems. Considering both diversity of sectors influencing the nexus and complex relationships in between, there is a spatial explicit need for analytically evaluating and reducing trade-offs, generating co-benefits and sharing water in balance. For instance, increasing agricultural water use upstream may create downstream water stress and less water availability for hydropower and ecosystems or vice versa. Assessing these relations requires catchment-based spatially explicit water availability knowledge. In this study we provide spatial

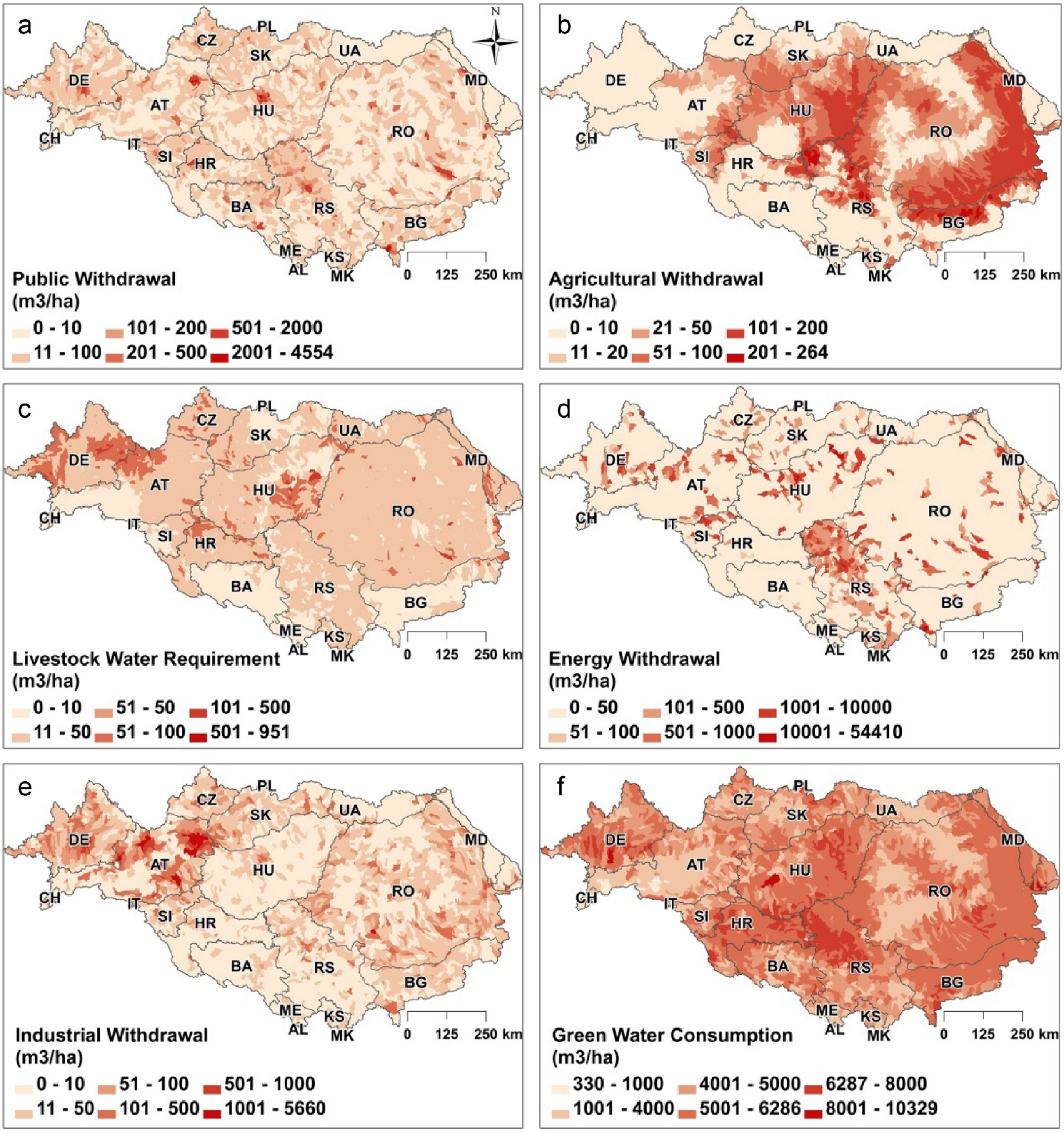

Fig. 6. Maps of water provisioning for different uses in the Danube river basin: (a) public withdrawal; (b) agricultural withdrawal; (c) livestock withdrawal; (d) energy withdrawal; (e) industrial withdrawal; and (f) green water consumed by crops and plants. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

explicit approach to the furtherance of nexus assessment to address its interconnected challenges. Our analysis supports understanding where water resources are available in the river basin and where it is abstracted for different uses. The approach allowes integrating the biophysical and the socio-economic components of water provisioning in a spatially explicit assessment. This is particularly relevant in the case of water resources, which depend on natural basin features and upstream–downstream relationships, rather than land national boundaries.

The approach presented in this study also allowes the integration of all components of the ecosystem–water–food–energy–nexus by taking into consideration major water stakeholders in the assessment, including the aquatic ecosystem, through the EFR, which was considered in the analysis of the water scarcity. Overall, when looking at the water abstractions by the different sectors in the Danube basin, most of water (44%) goes to the energy sector while 26% goes to the food (sum of agricultural and livestock). This implies that any water shortage may have consequences to the economy of the region as these will affect key income generating sectors. The public water use and the industrial water uses are 13% and 17%, respectively. This includes food production in the industrial sector and water for personal use by the public (Fig. 9).

Potential water scarcity is predominantly in Pannonian Danube, in some subbasin of Tiza, Middle Danube and Lower Danube (Fig. 7). Almost in the same area, according to a study of the ICPDR

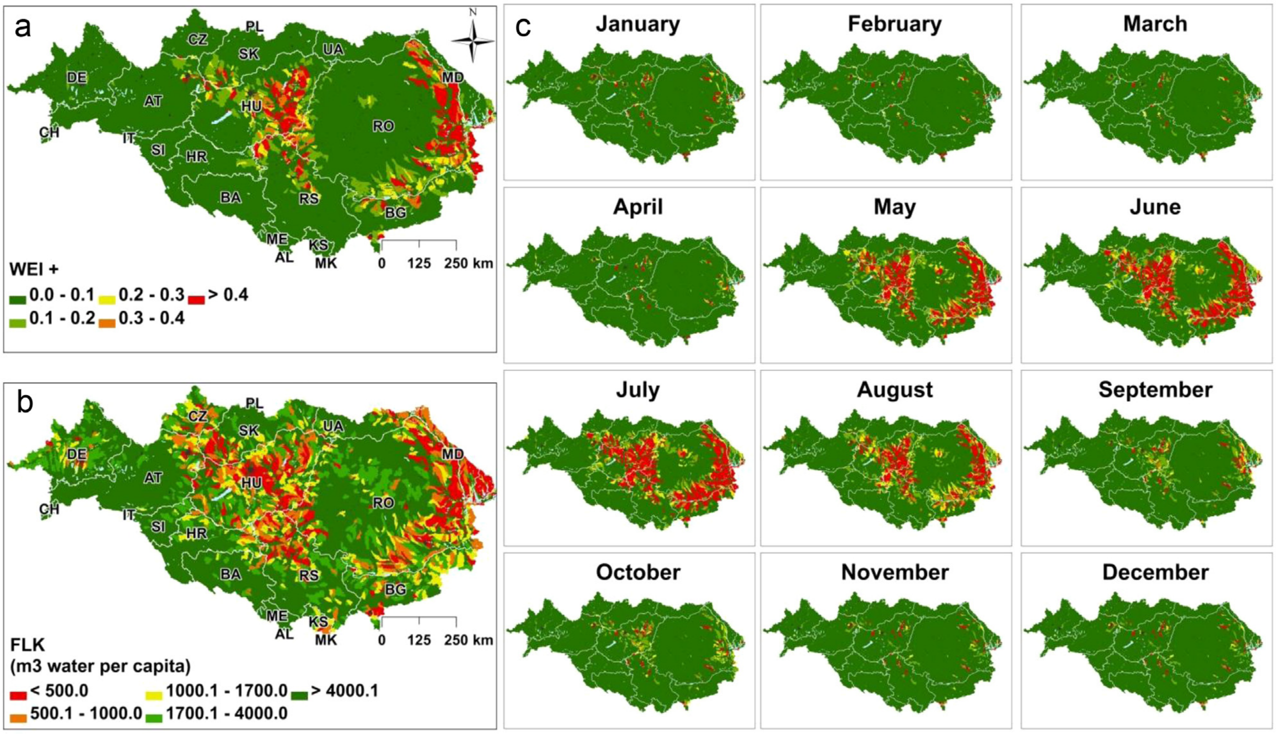

Fig. 7. Maps of water scarcity indexes in the Danube river basin: (a) FLK; (b) WEI+; and (c) Monthly WEI+ of long term average.

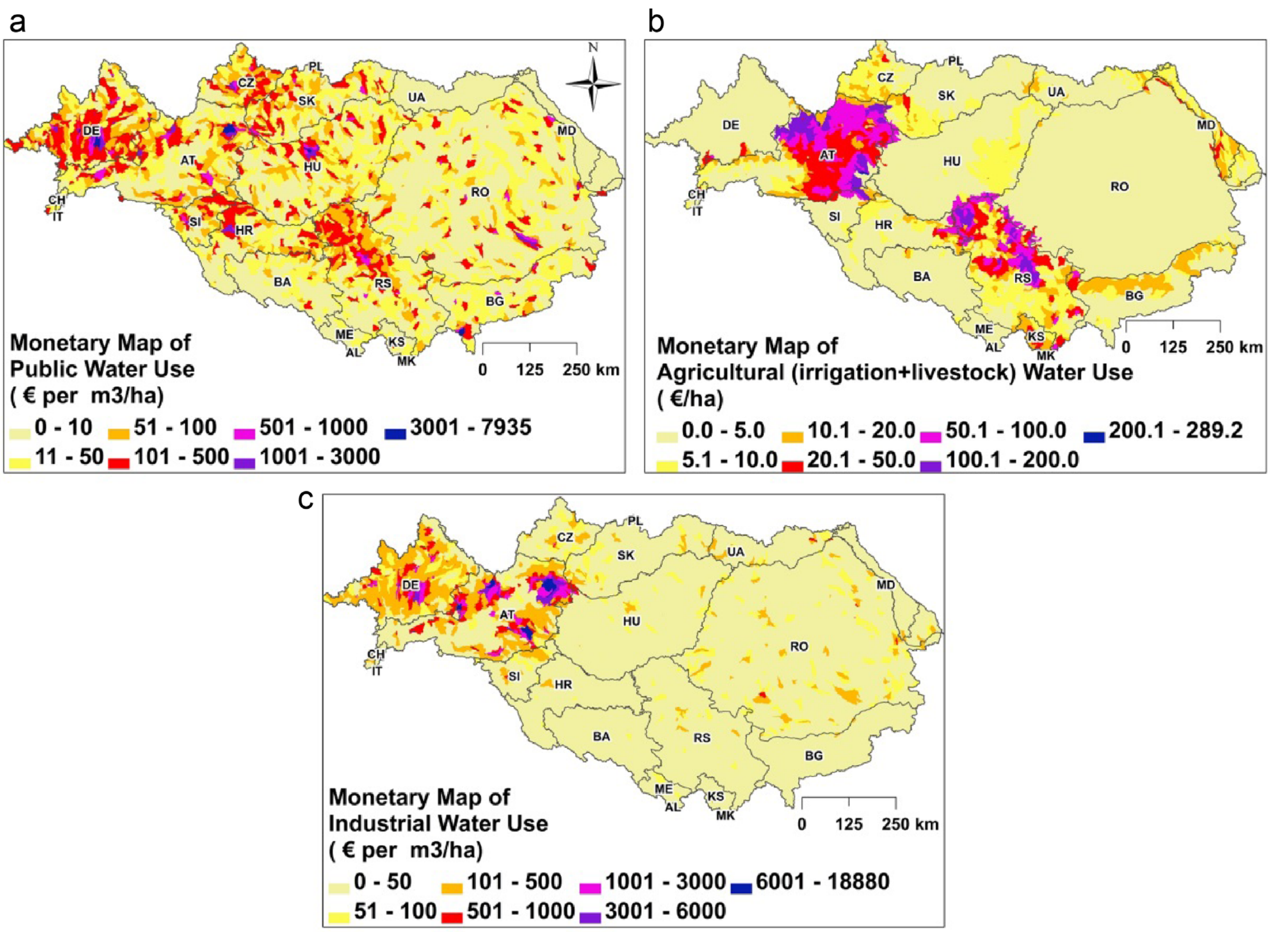

Fig. 8. Maps of monetary value of water uses in the Danube river basin: (a) public water use; (b) agriculture water use and (c) industrial water use.

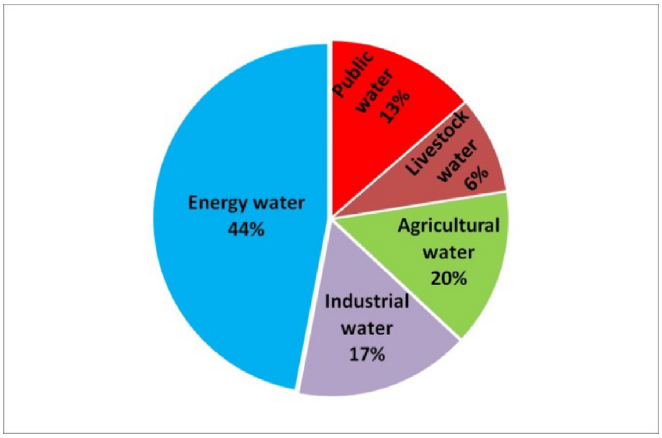

Fig. 9. Distribution of water used by sectors in the Danube basin considering the ecosystem–water–food–energy nexus. The percentages are computed on the total water used.

(2012; 2013b) less precipitation in summer season is projected and severe water stress is expected in the lower parts of the Danube basin in near future. From our results, most areas with low natural water yield have higher water use, creating a potential risk for water scarcity. In addition, high population densities in urban areas cause localized water scarcity, e.g. in Germany and Austria and Hungary and Romania (Fig. 7), with available water less than 1000 m^3/capita per year. However, in reality, there could be no water scarcity in urbanized areas in these countries, implying that, to meet the high demand, water is taken from other areas through artificial infrastructures. The artificial water transfers were not considered in this study (see footnote 3).

The approach proposed in this study could be improved further, as there are some shortcomings partially due to lack of data. For example, water quality should be included in the analysis as shown by the conceptual framework (Fig. 2) but it was not implemented (as the model SWAT was not calibrated for the water quality). In fact, each water use has specific quality requirements. Limits set in the Drinking Water Directive could provide information on water quality for domestic use, while nutrient and pesticide loads in water could provide information on water quality for agricultural use (Steyaert and Ollivier, 2007). In addition, some assumptions were made in this study. Concerning the environmental flows, we did not consider the seasonal and regional actual ecological requirements, but used a hydrological indicator based on long-term flow regimes. Moreover, we considered the water use equal to the withdrawals, but water withdrawal is not necessarily an accurate reflection of water demand and not all the water withdrawn is used, there are leakages and partial returns to the system. Also, water is not necessarily consumed in the same location where it is taken. Most water abstracted is given back to the system in a different point, which may generate problems of water scarcity locally and issues related to the degradation of the quality of the resource. This is included in the water exploitation index only by using ratios from literature considering that water was consumed where it was withdrawn.

Despites these limitations, the framework we have developed for this study is an illustration of how water provisioning services could be evaluated and mapped for the ecosystem–water–food–energy nexus using indicators from an river basin hydrological model such as SWAT. These types of assessments could be performed with integrated models, such as LISQUAL (Gentile et al., 2014); or SWAT, whereby all aspects (quantity and quality) are integrated. Using an integrated model would facilitate including scenario analysis to propose measures for ecosystem conservation, climate adaptation and sustainable land use management, simultaneously considering the different components of the ecosystem–water–food–energy nexus and the biophysical and socioeconomic aspects.

4.2. Valuing water provisioning

As explained in Paragraph 2.3.4, the different methodologies for valuing water for agricultural, industrial and residential users could not be applied because of lack of harmonized data in the Danube region. Instead in this study, we considered the expenses incurred for the use of the water resource as a proxy of its value for the different users (*cost-based* approach). The spatial distribution of the cost of water use performed in this study is an attempt to map the benefits of the water provisioning services for industry, agriculture and households using a *cost-based* approach. Since the cost of water is proxied by market prices, this methodology presents some clear limitations that need to be stressed.

First, water prices may be a bad proxy for the cost of providing water to end-users. Indeed, there is a high level of heterogeneity across countries with regards to the implementation of the *cost-recovery principle* through pricing. In many countries, water prices are distorted by subsidies (or cross-subsidies among types of water users) or due to the fact that the water sector is highly regulated and non-competitive. For instance, costs are almost fully recovered in Germany: 99% of drinking water costs and 96% of wastewater costs are directly paid for by the consumers. In Austria, the amount paid by consumers represents 93% of drinking water costs and 78% of wastewater costs (BDEW, 2010). But in Croatia, an analysis for four utility companies showed service prices do not reflect real costs, with a cost-recovery of 77% for drinking water supply and 45% for wastewater (ICPDR, 2005). A similar situation is observed in Romania where an analysis of water and wastewater systems in the Cluj and Salaj counties revealed a recovery of investment cost equal to 38% for water and wastewater (ICPDR, 2005). As a result, water prices may then not reflect the true marginal cost of supplying water. In addition, within a given country, the cost-recovery principle might be applied differently depending upon the sector considered. Industrial tariffs are generally higher than tariffs paid by households, even if it cannot be explained by differences in infrastructure or operation costs. This cross-financing phenomenon is common in the water sector but variable across countries. For instance, the ratio of commercial to household tariffs for water supply is 3.3 for Albania, 1.6 for Croatia and 2 for Montenegro. For sewage, the ratio is 1.5 for Croatia and 2.1 for Albania (REC, 2009). As a result, observed water prices should be used with caution when it comes to defining some mechanisms of allocation of water across countries or river basins, or across sectors.

Second, water price may not reflect the full social cost of providing water to end users. It is often the case that pollution externalities or water scarcity rents are not fully accounted for in the water prices. To get a full picture, water prices should include the non-market values of water ecosystem services. Indeed, freshwater ecosystems (lakes, rivers, groundwater) play a very important role in supporting the delivery of various services, but for reasons developed above they have not been included in the economic valuation exercise. An extension of the current framework would be to map these ecosystem economic values. This would imply the implementation non-market valuation methods (contingent valuation or choice experiment). A recent example of use of these non-market valuation methods in the Danube river basin is Brouwer et al. (2010). These non-market values may then be integrated to observed water prices to get a more accurate measure of the social cost of providing water.

Keeping in mind these two important limitations, we consider however the *price* approach as a pragmatic second-best option that allows for a geographical aggregation of costs at national and international scales. Despite the methodological limitations, the *price* approach offered the possibility to reflect on the benefit side of the water provisioning services. The economic value of a good is a measure of its contribution to human welfare and this value is highly spatially variable in the case of water. This work is therefore valuable to implement a water policy, as policy-makers must justify their decisions in term of welfare improvement. For instance, it may help to design a resource-efficient policy, evaluating the benefits of a better allocation of water resource across sectors and space. Another example is the cost-effectiveness of investments, providing a measure of the economic benefits of such investments that must balance their costs (e.g. in infrastructures). Results highlight the scope of spatial monetary assessment for water governance in the Danube river basin.

Mapping of both biophysical capacity and economic value of water resource is essential information for policy-making, such as planning a transboundary management, mutualizing cost-effective investment or designing instruments to improve resource allocation. This policy framework may be implemented at different levels, from national to international (continental) scales, according to its components. Potential for impact assessment should also be recognized; by estimating the economic value of the service in comparable monetary terms, impact of alternative policy interventions can be evaluated to make efficient trade-offs.

Regarding the benefits of the water provisioning for agriculture, results clearly emphasize the rationale for a solid integrated governance of the Danube. The monetary mapping shows that values are spatially located along the river, irrespective of the political frontiers. However, higher water expenses in Austria, Hungary, Croatia, Serbia and Bulgaria (Fig. 8a and b) should be considered with caution. Countries have implemented different water pricing policies and a direct interpretation of water-related costs as the value of ecosystem services, and this may be misleading. Water ecosystem service values have been proxied by water prices that usually reflect the value of economic activities (higher prices in higher marginal productivity sectors) but value comparisons across countries and across time should not be done on this basis. For instance, relatively higher prices are applied in some upstream countries such as Austria (for potable water and agricultural water) or Germany (for the industry) whereas economies in transition are still processing water policy and pricing reforms. Despite this statement, we can note that benefits are particularly high in downstream countries such as in the Balkans (the Romania's case is more specific since the country did not yet implement water pricing policy which result in low water costs). Economic welfare is distributed across numerous stakeholders and this resource-sharing context supposes national policies to be coordinated. To avoid that disconnected management lead to conflicting planning, mapping the contribution of resource provisioning to economic surplus beforehand any policy implementation help to define the role and responsibilities of respective governmental organizations.

Spatial monetary assessment is a component of the evaluation of trade-offs across sectors and a tool for choice-making of cost-effective infrastructures. In the end, the design of a resource allocation mechanism is the main purpose of a water policy. The basic rule for economic efficiency consists in sharing the resource in such a way that the marginal net benefits the different users get from its use are equalized. In this sense, inefficient allocations may be improved by transferring water from activities resulting in a low value toward more "valuable" activities. From national perspectives, even if the value mapping exercise described in this paper points out potential improvements, inter-sector adjustments (e.g. between agriculture and public water use or public water use and industry) should be based on the marginal value of water for these sectors, estimated with quantitative approaches. The present economic valuation does not provide a mapping of marginal values. Still quantification of an economic indicator (such as total value) for this ecosystem service allows policymakers to consider trade-offs into resource policy design on its geographic scale.

5. Conclusion

Water is a vital service provided by ecosystems, underpinning the ecosystem–water–food–energy nexus for human well-being. Considering the pressures on water resources originating from population growth, food demand and climate change, a spatially explicit assessment of water provisioning services is necessary to support the integrity of the ecosystems while benefiting from them.

In this study, we developed an integrated framework to assess water provisioning services, addressing the complex relationships of the ecosystem–water–food–energy nexus, using the SWAT hydrological model. We applied our framework to the Danube river basin and provided maps of the water provisioning services related to the nexus, analyzing the capacity to provide water, the different water uses and (a proxy of) benefits.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found in the online version at http://dx.doi.org/10.1016/j.ecoser.2015.08.002.

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