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Using ecosystem services to represent the environment in hydro-economic models.

Andrea Momblanch[1], Jeffery D. Connor[2], Neville D. Crossman[2], Javier Paredes-Arquiola[1] and Joaquín Andreu[1]

[1] Research Institute of Water and Environmental Engineering – Universitat Politècnica de València, Camino de Vera s/n, 46022 – Valencia, Spain

[2] Commonwealth Scientific and Industrial Research Organisation – Land and Water, PMB 2, Glen Osmond, SA, 5064, Australia

Abstract

Demand for water is expected to grow in line with global human population growth, but opportunities to augment supply are limited in many places due to resource limits and expected impacts of climate change. Hydro-economic models are often used to evaluate water resources management options, commonly with a goal of understanding how to maximise water use value and reduce conflicts among competing uses. The environment is now an important factor in decision making, which has resulted in its inclusion in hydro-economic models. We reviewed 95 studies applying hydro-economic models, and documented how the environment is represented in them and the methods they use to value environmental costs and benefits. We also sought out key gaps and inconsistencies in the treatment of the environment in hydro-economic models. We found that representation of environmental values of water is patchy in most applications, and there should be systematic consideration of the scope of environmental values to include and how they should be valued. We argue that the ecosystem services framework offers a systematic approach to identify the full range of environmental costs and benefits. The main challenges to more holistic representation of the environment in hydro-economic models are the current limits to understanding of ecological functions which relate physical, ecological and economic values and critical environmental thresholds; and the treatment of uncertainty.

Keywords: Water resources management; Hydro-economic models; Environmental impacts; Ecosystem Services Framework

Highlights:

- Representation of the environment in hydro-economic models (HEMs) is limited
- There is no systematic inclusion of the environmental costs and benefits in HEMs
- The ecosystem services approach identifies the full range of environmental values
- Accurate ecological functions and uncertainty analysis are key challenges for HEMs

1. Introduction

Adequate flows of fresh water in rivers support food and energy production, other economic activities such as river navigation and productive fisheries, as well as clean water provision through processes such as dilution and biological degradation (Momblanch *et al.*, 2015). All these uses compete for water resources with diverse use rights (Babel *et al.*, 2005), and different opportunities and costs associated with adapting to less water availability (Booker, 1995).

The 1972 amendment to the US Clean Water Act established national water quality standard to preserve aquatic life, recreational uses, and their values (Copeland, 2010). Since then, there has been an increased focus on understanding the environmental and socio-economic benefits of leaving water in streams, rivers and aquifers rather than extracting it for consumptive use. For example, in the Murray-Darling Basin in Australia, Connor (2008) found that additional flows in the river could significantly reduce costs of salinity damage through dilution, and Crossman *et al.* (2015) documented substantial carbon sequestration, tourism, and freshwater quality values, among others, from reducing water extraction. Grossmann and Dietrich (2012) assessed carbon sequestration, boating, habitat and biodiversity values of different water management options for the Spreewald Wetland in Germany. These studies used the ecosystem services (ES) concept to report on the benefits. The core ES notion is that a wide range of natural ecosystem processes help sustain and fulfil human life (Daily *et al.*, 1997), and that these services can be translated into economic values. Many ES are only substitutable at high economic costs, and in some cases cannot be replaced (Brauman *et al.*, 2007; Costanza *et al.*, 1997). For example, wetlands have the capacity to purify water by means of biochemical processes (Turner *et al.*, 2008) with capacity being a function of wetland condition and health. The degradation of wetland ecosystems could increase treatment costs of the water extracted for consumptive use (Maltby and Barker, 2009) and/or a reduce the recreation potential (Kahil *et al.*, 2015) leading to loss of income for the tourism industry.

According to the 5th assessment report of the Intergovernmental Panel on Climate Change (2014), renewable fresh water resources are likely to decrease over the 21st century, most significantly in arid and semi-arid regions where increased frequency of drought occurrence is expected (Schwabe *et al.*, 2013). Additionally, water demand is expected to grow with global population growth (UN 2015), resulting in more waste generation, pollution and land use expansion, which increases the pressure on land and water resources (Shama, 2004). Less water availability and lower quality, together with larger water demands, has led to increasing conflicts among water uses. Examples include conflicts between hydropower production and fisheries in the Mekong River in China (Ringler *et al.*, 2004); irrigation and urban water uses in the Jucar and Vinalopó rivers in Spain (Andreu *et al.*, 2009); and environmental and irrigation water uses in the Murray Darling Basin in Australia (Qureshi *et al.*, 2007) and the Colorado River Basin in the United States (Booker and Young, 1991).

Integrated water resources management, defined as the coordinated development and management of water, land and related resources to maximise economic and social welfare without compromising the sustainability of vital ecosystems (GWP 2000), can inform decisions about water sharing in the face of competing water demands and increasing scarcity (Booker *et al.*, 2012). Hydro-economic models (HEMs) are one of the main tools used for integrated water resources management (Booker *et al.*, 2012; Harou *et al.*, 2009). HEMs combine hydrological and water infrastructure representation of water resources systems with economic demand functions for key water uses in order to allocate water subject to physical and institutional constraints (Heinz *et al.*, 2007). HEMs typically use a node network structure with nodes representing points of diversion, inflow, outflow, storage or treatment and links between nodes representing river reach processes (Harou *et al.*, 2009). HEMs can use optimisation or simulation approaches, but typically have the goal of allocating water among multiple uses to optimize economic value (Brouwer and Hofkes, 2008). HEMs have been used to solve water management problems for more than 50 years, and have evolved from analysing single-water use problems at water supply scale (Lefkoff and Gorelick, 1990; Wilchfort and Lund, 1997) to integrated multiple-demand and multiple-source problems at single river basin scale (Davidson *et al.*, 2013b; Divakar *et al.*, 2011) and multi-basin scale (Bekchanov *et al.*, 2015c; Fisher *et al.*, 2002). Groundwater representation and its connection to the surface water system have also featured in HEMs (Daneshmand *et al.*, 2014; Pulido-Velazquez *et al.*, 2006; Pulido-Velazquez *et al.*, 2008a).

Several studies have reviewed HEMs. For example, Harou *et al.* (2009) focus on methodological aspects of HEMs, such as model formulation and design, economic valuation methods for the different water uses, and major applications. Heinz *et al.* (2007) discuss the role of economic approaches in water management to address the European Water Framework Directive (EC 2000) objectives, analysing diverse assessment and performance criterion, water policies and management options. Booker *et al.* (2012) review the advances in economic representation, policy objectives and water institutions, and level of integration and complexity of HEMs.

Consistent across reviews of HEMs is the conclusion that representation of environmental costs and benefits in HEMs is patchy and limited. For example, Harou *et al.* (2009) conclude that environmental water uses are rarely represented with economic value functions in HEMs, although minimum-flow constraints are included more often. They also highlight the importance of incorporating water quality processes and values which are mostly lacking in HEMs. Booker *et al.* (2012) argue for the expansion of HEMs to jointly tackle environmental, economic, hydrologic and institutional water resources management problems. Other reviews highlight the limited representation of environmental in-stream uses and processes in HEMs (Ringler and Cai, 2006; Ward and Pulido-Velazquez, 2009; Ward and Pulido-Velázquez, 2008), and the dearth of HEMs which account for water management changes on non-market values provided by ecosystems (Griffin and Hsu, 1993; Kragt, 2013).

There has not yet been any attempt at systematic cataloguing and critical assessment of the range of environmental impacts and values included in HEMs. Here we address this gap by: i) reviewing the range of environmental impacts included in HEMs; ii) documenting the methods used to represent the economic value of environmental impacts in HEMs, and; iii) making recommendations to improve the inclusion of environmental impacts and values in HEMs.

We use ES as an organising framework because it offers a systematic way to analyse the potential environmental impacts of changes to water management using the environment-economy connection. This connection is best demonstrated by the ES cascade (Potschin and Haines-Young, 2011) which shows the causal links from a change in biophysical state as a result of altered management, to the ecosystem change and then the change to ES, economic values and human well-being (Figure 1). In recent years there has been a proliferation of ES frameworks (Haines-Young and Potschin, 2013; MA 2005; TEEB 2008; UK NEA 2011). Common to all ES frameworks is the *provisioning* category, which are directly consumed ES products. An example is fish production in rivers that people value as food. All ES frameworks also include the *regulating* category for ES that arise from maintenance and moderation of environmental conditions. The capacity of wetlands to purify water by means of biochemical processes (Turner *et al.*, 2008) is an example. Also common to ES frameworks is a category for non-consumptive values such as recreational, educational, aesthetic and spiritual. The major difference between ES frameworks is how intermediate ecosystem processes are treated. Some frameworks only include end-products or services consumed or valued directly by humans (MA 2005; Wallace, 2007), while other frameworks include environmental processes which only indirectly contribute to human welfare, such as decomposition and nutrient cycling (Boyd and Banzhaf, 2007; Costanza, 2008; Fisher and Turner, 2008). We use the Common International Classification of Ecosystem Services (CICES) (Haines-Young and Potschin, 2013) as the reference framework to classify the environmental impacts addressed by our reviewed studies. CICES supports the System of Environmental-Economic Accounting (United Nations *et al.*, 2012) and includes only final ES (Haines-Young and Potschin, 2013) which leads to clear environment-economy links consistent with the need of HEMs to include economic demand functions.

2. Methods

Our review involved four stages: 1) identifying a comprehensive set of HEM studies that included representation of the environment; 2) cataloguing the selected studies according to a set key summary attributes; 3) applying the CICES to classify the types of environment representation in the HEMs, and; 4) cataloguing the methods used to value the environment in the HEMs.

2.1. Literature search and selection

We started with the set of papers reviewed by Heinz *et al.* (2007), Brouwer and Hofkes (2008) and Harou *et al.* (2009) (n = 124). These were supplemented with papers from 2009 to the present using a SCOPUS search containing the key words ‘hydro-economic model’, ‘water management’, ‘optimization’, and ‘network flow’ (n = 877). We then refined the scope to case studies dealing with economic analysis of water management, including environmental aspects, at river basin scale (n = 144). For environmental aspects we considered environmental flows, water quality in water bodies, nature related recreation activities, flood control, and broader concepts such as habitat or vegetation. We screened the titles, abstracts and journals to remove irrelevant papers (n = 135), and then downloaded and read the full papers in order to select the final collection of papers (n = 95).

We classified all papers by year of publication and identified the water use sector to which each HEM was applied. We also documented the spatial scale of analysis, the major water management problem addressed based on the categories established in Harou *et al.* (2009), the assessment criteria used according to the proposal by Heinz *et al.* (2007), and how uncertainty was treated. We assessed whether the environment was considered as a constraint or valued in economic terms. For the papers in the latter group, we extended the review as described in sections 2.2 and 2.3.

2.2. Classifying representation of the environment

We used CICES to classify the representation of the environment in the reviewed studies. CICES uses the three main ES categories of provisioning, regulating and maintenance, and cultural services. Each of these broad ES types is successively split into divisions, groups and classes, following a hierarchical structure. The elements within a level of hierarchy are conceptually similar to one another according to the ways they are used by people (Haines-Young and Potschin, 2013). We identified which ES in CICES are potentially provided by freshwater bodies (Table 1).

2.3. Economic valuation methods

The next step was to identify the economic valuation methods used to estimate environmental values included in the HEMs. We classified valuation methods into the standard typologies common throughout the literature (Costanza *et al.*, 2011; Chee, 2004; de Groot *et al.*, 2002; TEEB 2010; Tietenberg and Lewis, 2009). The typologies we used are:

- Market value: Used when the valued ES is a good that has a market price, e.g. cultivated crop. However, for the ES whose price does not include the impact of abstraction/use on their availability for other users and the environment, e.g. drinking water, the value is derived from the marginal willingness-to-pay using econometric approaches.
- Production-based: Used when the valued ES is a factor of production for a good or service traded on the market, e.g. water for agricultural production. Value is estimated as the contribution to the net revenues obtained from the produced good or service in the market.
- Cost-based: This method approximates the value of the ES based on the costs of replacing it (replacement cost method). This approach is applicable to ES such as mediation of waste. The method can also consider the avoided mitigation damages given the presence of the ES (avoided cost method), e.g. flood protection.
- Revealed preference: Often used to value recreation and amenity values of water. The travel cost method assumes that the value of an ES can be approximated with the expenses incurred to enjoy it. This method is applicable to ES such as aquatic recreation by considering transportation expenses, on site spending and protected area entrance fees. The hedonic price method relates the value of an ES with the price variation of associated goods for different production levels or quality of the ES. A common example is the difference in market prices for real estate with more

and less aesthetic water related amenity, assuming all other variables influencing real estate sales are equal.

- Stated preference: Surveys designed to elicit the values people ascribe to an ES. Respondents are usually asked how much they would be willing to pay for a specific improvement in the ES (contingent valuation method), or they are asked to select one among a number of alternatives for improvement of the ES, where price or cost required to pay for improved ES condition is a key attribute (choice experiment method). This method is applicable to non-consumptive ES such as aquatic biodiversity.
- Benefit transfer (or meta-analysis). Takes estimates of ES value from one site and applies them to another site.

In HEMs, these valuation methods are used to produce a value function for the different water uses and environmental benefits and costs, given the variation in the physical variables such as water flow or volume. In the studies we assessed, these functions are estimated using econometric or statistical methods, or by combining mathematical representations of an ecological production function with a unit production value obtained with one of the valuation approaches.

3. Results

3.1. General features of the studies

As a result of the literature search and selection, 95 papers were reviewed. Table 2 cites the final set of papers which was assigned unique ID numbers for easy citation.

The 95 papers covered the period 1984 to 2015, with less than 2 papers published per year on average prior to 2002 (Figure 2). About 6 papers were published per year on average after 2002, with the most studies in 2013 (n = 13).

We found that the water use sectors most represented by HEMs were urban, agricultural, industrial, and hydropower sectors (Table 3). Other sectors such as livestock, tourism, navigation, and industry were rarely included except within the twelve studies that included five or more sectors. The river basin was the most common spatial scale of the papers we reviewed. We also found that HEMs have been applied to administrative regions (15 papers: 1, 18, 21, 22, 24, 26, 28, 32, 35, 36, 44, 45, 51, 55, and 81), water supply systems (7 papers: 11, 12, 29, 31, 65, 84, and 91), and international regions (5 papers: 17, 86, 88, 89, and 90).

The major water management problems tackled by the HEMs we reviewed were resource allocation, with emphases on inter-sectoral allocation (n = 48), water institutions (n = 13), and water supply infrastructure (n = 13). Other water issues such as drought or climate change management, trans-boundary water management, conjunctive surface-groundwater use, and land use management were less common. The HEMs used different types of assessment criteria to design and test water management solutions. We identified 58 papers which used a net benefit maximisation approach. Table 4 presents the number of papers addressing each major issue and the type of assessment used.

We found that there were very few studies that treated uncertainty in physical variables and parameters. Uncertainty was analysed by means of probabilistic approaches (2 and 11) and sensitivity analyses in deterministic models (17 and 77). Only two studies (6 and 34) assessed uncertainty of economic parameters.

We distinguished between HEMs that included economic valuation of the environment versus those that accounted for the environment using only biophysical units. We found that 61 papers considered environmental uses as constraints (e.g. 12, 33 and 93) or as decision variables in the optimisation function via ecological functions (e.g. 66 and 72). These studies mainly included minimum flows and, occasionally, water quality as environmental aspects. Some calculated the opportunity costs of environmental constraints, which provided useful economic information for decision making but did not allow comparison of environmental and other water use values.

3.2. Environmental impacts classification

Among the reviewed studies, 34 defined environmental benefits and costs and used economic functions to value these within water management analysis (2, 3, 5, 6, 7, 8, 9, 10, 11, 16, 27, 28, 29, 30, 34, 37, 49, 53, 57, 60, 65, 67, 68, 71, 73, 74, 75, 77, 84, 88, 89, 90, 91, and 94). The aspects of the environment considered were diverse and broadly covered vegetation and fauna, water quality and flood control. Most studies analysed only one (2, 3, 5, 8, 9, 16, 30, 37, 49, 53, 57, 60, 68, 73, 74, 75, 77, 84, and 91) or two (i.e. 6, 7, 11, 27, 28, 29, 34, 65, 71, and 94) environmental aspects, and only five papers covered more than three (10, 67, 88, 89, and 90). Table 5 uses the CICES framework to summarise the environmental impacts included in the HEMs we reviewed. Some HEMs included components of ecosystems which could not readily be allocated to the CICES framework, such as wetlands and environmental flows (16, 27, 28, 29, 34, 60, 73, 74, 88, 89, 90, and 94).

We found no systematic approaches to valuation of the environment and ecosystems in HEMs although there were some recurring methods (Table 6). Production-based valuation methods were more commonly applied for provisioning ES such as commercial fishing. Water quality improvement (e.g. salt dilution and nutrients abatement) was most often valued using cost-based methods (e.g. agricultural production losses due to salinity, and treatment for drinking water). Flood control and carbon sequestration valuation were also valued with cost-based methods. Recreation activity related values were mainly estimated using the travel cost method. When valuing impacts on habitat and biodiversity, in general, or for specific natural vegetation types and native animals, stated preferences techniques were used, but in some cases results were obtained through the benefit transfer method rather than with case specific studies. Among the impacts which cannot be categorised according to CICES, benefit transfer was the main valuation technique for wetlands, and other non-specified or bespoke valuation methods were used for the environment as a general concept. The greatest diversity in valuation methods was found for environmental flows in rivers or volumes in aquifers.

HEMs require demand functions which relate the value of the impacts to water supply. For most in-stream use studies, the values were dependent on river flows (3, 10, 27, 34, 77, 88, 89, and 90), whilst for uses in lakes and reservoirs values relied on the water level or the stored volume (49, 65, 84, and 91). Finally, there were few examples of more complex demand functions which captured the relationship between the value and the ecological response using more than one hydrological variable (67, 71, and 94).

4. Discussion

We selected 95 HEM studies which cover environmental aspects of water management at river basin or comparable scales. The majority of HEMs analysed inter-sectoral water allocation between two or three water use sectors, including environmental, agricultural and urban uses, with the aim of maximising net benefits. The consideration of uncertainty issues was rare. From the initial 95 studies, about two thirds considered environmental aspects in physical terms, mostly as constraints to realising other use values. The third which valued at least one environmental impact in economic terms were mostly limited to a single environmental aspect, or included very broad or vague environmental aspects. Recreation, commercial fishing and salt dilution were the most frequently valued in HEMs. We also found that established and traditional valuation methods were used to assign economic value to the environment, with little deviation from methods recommended in the ES literature (Banerjee *et al.*, 2013; de Groot *et al.*, 2002; TEEB 2010).

We found that the use of the ES framework to identify the aspects of the environment likely affected by alternative water management actions is a systematic and thorough way to select relevant impacts and values. The ES framework should more comprehensively capture the ecological processes, values and interactions in HEMs. To some extent, the ES approach is already influencing the inclusion of environmental and economic values in HEMs. For example, Bryan *et al.* (2013) selected environmental impacts based on the main water demands and the important river ecological and ES components were identified using river basin mapping.

A reason for the poor representation of the environment, especially in economic terms, in many HEMs is the limited availability of data and models characterising relevant environmental processes and associated economic values (Dandy *et al.*, 2013). Although good quality information is complex and costly to obtain, we think that in well studied river basins omissions may be a result of the single issue focus of many studies. Many river basins have a good knowledge base which can be used to include more environmental values. For example, water quality processes related to flow are reasonably well understood and they are not difficult to value using cost-based methods (Keeler *et al.*, 2012; La Notte *et al.*, 2015; Terrado *et al.*, 2016). Similarly, it is possible to estimate values of recreational opportunities related to flow or water level (Grossmann and Dietrich, 2012; Hurd *et al.*, 1999), and obtain values of provisioning services such as fisheries using production functions and market values (Mullick *et al.*, 2013; Ringler and Cai, 2006), although for these ES the difficulty relies on having reliable data about underlying biophysical processes for water bodies in the basin. Environmental impacts can be valued with more than one method and, in agreement with de Groot *et al.* (2002), we suggest that following a rank ordering of valuation methods for each type of ES adds rigour and value comparability. Selection of the appropriate method depends on the data available and on the type of ES. Market valuation methods are generally more suited to provisioning ES or use values, cost-based methods to regulating ES, and revealed and stated preferences methods to cultural ES (TEEB 2010; Turner *et al.*, 2008).

Uncertainty in ES values can be a consequence of the valuation approach and of the quality of the economic data. For instance, revealed and stated preference valuation methods have been criticised for their subjectivity and bias (Bateman *et al.*, 2006; Chee, 2004; La Notte *et al.*, 2015), while market value, production and cost based valuation techniques are more objective. Benefit transfer can increase the range of values included in HEMs when local valuation studies are absent. However, transferring values introduces additional uncertainty if there is inadequate correspondence between the original and new studies (Plummer, 2009). This uncertainty may lead to highly variable results that would prevent decision makers from using HEMs to support decisions.

Expanding the representation of the environment and its values in HEMs will likely increase model uncertainties. Since the number of ES values associated with environmental impacts can be the greatest source of uncertainty (Boithias *et al.*, 2016), there will need to be more systematic incorporation of uncertainty analyses into HEMs, including assessment of implications of uncertainty in decision making (Cai *et al.*, 2002). We show that very few HEMs currently treat uncertainty, a conclusion drawn by a number of other studies (Bateman *et al.*, 2006; La Notte *et al.*, 2015; Lund and Ferreira, 1996). We suggest that Monte Carlo based analysis, an approach used more often in integrated analysis such as integrated assessment of global climate change impacts and adaptation (Gao *et al.*, 2016), be also used to assess uncertainty in HEMs. Monte Carlo analyses consider non-linearities and are probabilistic, which is in line with actual measurement processes (Papadopoulos and Yeung, 2001).

Although there are arguments for expanding the number of monetised environmental values in HEMs, not all the potential environmental values impacted by water management need to be included to support good decisions. It may be the case that inclusion of additional environmental values does not influence the decision path. For example, in the case of a decision that has high net benefit based on the social, economic and environmental benefit values that are already quantified, quantifying additional benefits in monetary terms may add little to the conclusion (Kandulu *et al.*, 2014). It may also be the case that monetised environmental values, such as those characterising productivity of wetlands or environmental flows (e.g. €/ha and €/m^3, respectively), disguise the diverse pluralistic values of the environmental assets. These aggregated values are incompatible with ES classification and valuation, though in some cases they may provide information that could be unpacked into distinct components that could be valued in an ES framework.

Something that is rarely dealt with explicitly in HEMs, despite many studies noting its importance, is the role of critical thresholds and system irreversibilities in the ecosystems response functions (Folke *et al.*, 2002; Folke *et al.*, 2004; Scheffer *et al.*, 2001; Spangenberg *et al.*, 2014). An exception is Kahil *et al.* (2015) who use a piecewise function to consider the shifts in the benefits provided by a wetland depending on inflow critical thresholds. Another aspect, not often considered in valuation functions in HEMs, is the correlation between ecological functions and other biophysical variables apart from water flows and volumes. Water quality has an important bearing on environmental aspects such as fauna and flora, and so water quality should be represented with environmental processes and linked to valuation functions. Although none of our reviewed studies consider the impact of water quality on environmental uses of water, some studies do consider impacts on traditional uses. For instance, Hurd *et al.* (1999) account for the impact of salinity on agricultural, urban and industrial uses. We suggest water quality variables (e.g. salinity, temperature) should be sufficiently detailed in HEMs to assess environmental impacts.

5. Conclusion

We used an ES framework to catalogue how HEMs have represented and valued the environment. Even though water management affects many environmental values, the HEMs we reviewed did not apply any systematic approaches to identify potential environmental impacts. This unsystematic approach to inclusion of the environment in HEMs risks over-looking potential trade-offs (between environment and economy) and unintended ecosystem impacts from water management decisions. The ES framework can be used to screen many environmental impacts that could be more widely applied in setting scope of analysis for water management actions.

Some important challenges remain. Firstly, the biophysical variables impacted by water management should be better understood in order to undertake a proper impact assessment and valuation. Aggregated environmental indexes which lose information about relevant detailed environmental impact values should then be avoided. Secondly, environmental functions which capture non-linearities and thresholds in ecological processes should be better defined, as should the role that water quality plays in broader aspects of environmental quality. Finally, uncertainty in both biophysical and economic variables should be more often addressed to improve the decision-support capabilities of HEMs.

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Table 1. CICES framework for ecosystem accounting (v4.3) modified with the inclusion of the potential services provided by water bodies.

Section	Division	Group	Class	Water bodies*
Provisioning	Nutrition	Biomass	Cultivated crops	
			Reared animals and their outputs	
			Wild plants, algae and their outputs	√
			Wild animals and their outputs	√
			Plants and algae from in-situ aquaculture	√
			Animals from in-situ aquaculture	√
		Water	Surface water for drinking	√
			Ground water for drinking	√
	Materials	Biomass	Fibres and other materials from plants, algae and animals for direct use or processing	√
			Materials from plants, algae and animals for agricultural use	√
			Genetic materials from all biota	√
		Water	Surface water for non-drinking purposes	√
			Ground water for non-drinking purposes	√
	Energy	Biomass-based energy sources	Plant-based resources	
			Animal-based resources	
		Mechanical energy	Animal-based energy	
Regulation & Maintenance	Mediation of waste, toxics and other nuisances	Mediation by biota	Bio-remediation by micro-organisms, algae, plants, and animals	√
			Filtration/sequestration/storage/accumulation by micro-organisms, algae, plants, and animals	√
		Mediation by ecosystems	Filtration/sequestration/storage/accumulation by ecosystems	√
			Dilution by atmosphere, freshwater and marine ecosystems	√
			Mediation of smell/noise/visual impacts	
	Mediation of flows	Mass flows	Mass stabilisation and control of erosion rates	√
			Buffering and attenuation of mass flows	√
		Liquid flows	Hydrological cycle and water flow maintenance	√
			Flood protection	√
		Gaseous / air flows	Storm protection	
			Ventilation and transpiration	
	Maintenance of physical, chemical, biological conditions	Lifecycle maintenance, habitat and gene pool	Pollination and seed dispersal	√
			Maintaining nursery populations and habitats	√
		Pest and disease control	Pest control	
			Disease control	
		Soil formation and composition	Weathering processes	
			Decomposition and fixing processes	
		Water conditions	Chemical condition of freshwaters	
			Chemical condition of salt waters	
		Atmospheric composition and climate regulation	Global climate regulation by reduction of greenhouse gas concentrations	√
			Micro and regional climate regulation	√
Cultural	Physical and intellectual interactions	Physical and experiential interactions	Experiential use of plants, animals and land-/seascapes	√
			Physical use of land-/seascapes	√
		Intellectual and representative interactions	Scientific	√
			Educational	√
			Heritage, cultural	√
			Entertainment	√
			Aesthetic	√
	Spiritual, symbolic and other interactions	Spiritual and/or emblematic	Symbolic	√
			Sacred and/or religious	√
		Other cultural outputs	Existence	√
			Bequest	√

*** They comprise all the river basin elements that can be affected by water management (quantity and quality): rivers including riverbed and riverbanks; wetlands considering the different types (e.g. US Hydrogeomorphic classification or the simplification proposed by Turner et al. (2008)); aquifers; reservoirs.**

Table 2. Final selection of papers reviewed.

ID	Citation	ID	Citation	ID	Citation
1	(Vaux and Howitt, 1984)	33	(Pulido-Velazquez *et al.*, 2006)	65	(Yang and Cai, 2011)
2	(Brown *et al.*, 1990)	34	(Ringler and Cai, 2006)	66	(Ahmadi *et al.*, 2012)
3	(Booker and Young, 1991)	35	(Schoups *et al.*, 2006)	67	(Grossmann and Dietrich, 2012)
4	(Diaz *et al.*, 1992)	36	(Tanaka *et al.*, 2006)	68	(Ward and Pulido-Velazquez, 2012)
5	(Booker and Young, 1994)	37	(Ward *et al.*, 2006)	69	(Yang *et al.*, 2012)
6	(Booker, 1995)	38	(Houk *et al.*, 2007)	70	(Blanco-Gutiérrez *et al.*, 2013)
7	(Lund and Ferreira, 1996)	39	(Mainuddin *et al.*, 2007)	71	(Bryan *et al.*, 2013)
8	(Ward and Lynch, 1996)	40	(Medellín-Azuara *et al.*, 2007)	72	(Connor *et al.*, 2013)
9	(Ward and Lynch, 1997)	41	(Qureshi *et al.*, 2007)	73	(Davidson *et al.*, 2013a)
10	(Hurd *et al.*, 1999)	42	(Cai *et al.*, 2008)	74	(Davidson *et al.*, 2013b)
11	(Watkins Jr and McKinney, 1999)	43	(Harou and Lund, 2008)	75	(Divakar *et al.*, 2013)
12	(Jenkins and Lund, 2000)	44	(Medellín-Azuara *et al.*, 2008a)	76	(Geng and Wardlaw, 2013)
13	(Rosegrant *et al.*, 2000)	45	(Medellín-Azuara *et al.*, 2008b)	77	(Mullick *et al.*, 2013)
14	(Bielsa and Duarte, 2001)	46	(Pulido-Velazquez *et al.*, 2008b)	78	(Pulido-Velazquez *et al.*, 2013)
15	(Tisdell, 2001)	47	(Reynaud and Leenhardt, 2008)	79	(Riegels *et al.*, 2013)
16	(Cai *et al.*, 2002)	48	(Volk *et al.*, 2008)	80	(Roozbahani *et al.*, 2013)
17	(Fisher *et al.*, 2002)	49	(Ward and Pulido-Velázquez, 2008)	81	(Wan *et al.*, 2013)
18	(Newlin *et al.*, 2002)	50	(Li *et al.*, 2009)	82	(Yang and Yang, 2013)
19	(Cai *et al.*, 2003a)	51	(Medellín-Azuara *et al.*, 2009)	83	(Daneshmand *et al.*, 2014)
20	(Cai *et al.*, 2003b)	52	(Ward, 2009)	84	(Debnath, 2014)
21	(Draper *et al.*, 2003)	53	(Ward and Pulido-Velazquez, 2009)	85	(Erfani *et al.*, 2014)
22	(Knapp *et al.*, 2003)	54	(Alcoforado de Moraes *et al.*, 2010)	86	(Hasler *et al.*, 2014)
23	(Burke *et al.*, 2004)	55	(Harou *et al.*, 2010)	87	(Yang and Yang, 2014)
24	(Jenkins *et al.*, 2004)	56	(Zoltay *et al.*, 2010)	88	(Bekchanov *et al.*, 2015b)
25	(Letcher *et al.*, 2004)	57	(Divakar *et al.*, 2011)	89	(Bekchanov *et al.*, 2015c)
26	(Pulido-Velazquez *et al.*, 2004)	58	(George *et al.*, 2011a)	90	(Bekchanov *et al.*, 2015a)
27	(Ringler *et al.*, 2004)	59	(George *et al.*, 2011b)	91	(Debnath *et al.*, 2015)
28	(Assimacopoulos *et al.*, 2005)	60	(Grafton *et al.*, 2011)	92	(Erfani *et al.*, 2015)
29	(Babel *et al.*, 2005)	61	(Munoz-Hernández *et al.*, 2011)	93	(Girard *et al.*, 2015)
30	(Booker *et al.*, 2005)	62	(Grafton and Jiang, 2011)	94	(Kahil *et al.*, 2015)
31	(Marques *et al.*, 2006)	63	(Riegels *et al.*, 2011)	95	(Roozbahani *et al.*, 2015)
32	(Null and Lund, 2006)	64	(Varela-Ortega *et al.*, 2011)		

Table 3. Water use sectors considered in the reviewed HEM studies indicating the number of papers and their ID.

Sectors		**Num. papers**	**ID papers**
Single sector	Environmental	2	65, 71
Two sectors	Environmental and agricultural	23	15, 22, 23, 25, 31, 35, 38, 39, 41, 48, 60, 61, 62, 64, 66, 67, 70, 72, 76, 80, 88, 90, 92
	Environmental and hydropower	2	8, 9
	Environmental and urban	3	12, 50, 81
Three sectors	Environmental, agricultural and hydropower	2	14, 89
	Environmental, agricultural and industrial	3	82, 85, 86
	Environmental, agricultural and navigation	1	77

Sectors		Num. papers	ID papers
	Environmental, urban and agricultural	23	1, 18, 21, 24, 26, 28, 33, 37, 40, 43, 45, 46, 47, 49, 51, 52, 53, 55, 68, 69, 78, 93, 94
	Environmental, urban and hydropower	2	84, 91
Four sectors	Environmental, urban, agricultural and hydropower	13	2, 6, 13, 16, 19, 20, 27, 32, 36, 42, 44, 58, 59
	Environmental, urban, agricultural and industrial	9	5, 17, 30, 73, 74, 79, 83, 87, 95
Five or more sectors		12	3, 4, 7, 10, 11, 29, 34, 54, 56, 57, 63, 75

Table 4. Cross relationship between major issues and assessment criteria in the reviewed papers indicating the number of papers and their ID. See Heinz *et al.* (2007) for description of assessment criteria.

Assessment criteria **Major water management problems**	**Cost-Benefit**	**Cost-Effectiveness**	**Net Benefit maximisation**	**Total Cost minimisation**	**Multi-criteria**	**Priority based optimisation and post-economic valuation**	**Total**
Conjunctive use of surface and groundwater	0	0	4 (26, 31, 35, 83)	1 (33)	0	0	**5**
Drought/climate change management	0	0	4 (10, 30, 37, 94)	3 (36, 44, 55)	0	1 (6)	**8**
Inter-sectoral water allocation	4 (52, 58, 59, 67)	3 (38, 64, 93)	28 (3, 4, 8, 9, 13, 14, 15, 20, 25, 39, 41, 42, 47, 50, 54, 56, 57, 60, 61, 62, 69, 70, 77, 79, 84, 89, 90, 91)	1 (46)	11 (29, 65, 66, 71, 75, 76, 80, 81, 82, 87, 95)	1 (28)	**48**
Land use management	0	1 (48)	0	0	0	0	**1**
Trans-boundary management and conflict resolution	0	1 (86)	5 (16, 17, 19, 27, 34)	1 (40)	0	0	**7**
Water institutions (prices, markets, rights)	0	0	12 (1, 5, 18, 22, 23, 49, 53, 63, 78, 85, 88, 92)	0	0	1 (72)	**13**
Water supply, engineering infrastructures and capacity expansion	1 (45)	0	5 (24, 43, 68, 73, 74)	6 (7, 11, 12, 21, 32, 51)	0	1 (2)	**13**
Total	**5**	**5**	**58**	**12**	**11**	**4**	**95**

Table 5. Classification of environmental and non-environmental impacts included in HEM studies (and papers ID) according to the CICES framework.

Section	**Class**	**Number of papers**
Provisioning	Wild animals and their outputs	Commercial fishing (11, 27, 34, 77, 88, 89, and 90)
	Surface water for drinking	Urban demands (2, 3, 5, 6, 7, 10, 11, 16, 27, 28, 29, 30, 34, 37, 49, 53, 57, 68, 73, 74, 75, 84, 91, and 94)
	Ground water for drinking	
	Surface water for non-drinking purposes	Agricultural and/or Hydropower and/or Industrial and/or Navigation and/or Livestock and/or Commercial (2, 3, 5, 6, 7, 8, 9, 10, 11, 16, 27, 28, 29, 30, 34, 37, 49, 53, 57, 60, 67, 68, 74, 75, 77, 84, 88, 89, 90, 91, and 94)
	Ground water for non-drinking purposes	
Regulation & Maintenance	Filtration/sequestration/storage/accumulation by micro-organisms, algae, plants, and animals	Water quality (28) + Wastewater treatment (10)
	Filtration/sequestration/storage/accumulation by ecosystems	
	Dilution by atmosphere, freshwater and marine ecosystems	Salt dilution (2, 3, 5, 6, 10, 57, and 75) + Waste heat (10)
	Flood protection	Flood control (7, 10, and 65)
	Global climate regulation by reduction of greenhouse gas concentrations	Carbon sequestration (67)
Cultural	Experiential use of plants, animals and land-/seascapes	Tourism (88, 89, 90, and 94)

Section	Class	Number of papers
	Physical use of land-/seascapes	Recreation (6, 10, 11, 29, 30, 65, 84, and 91) + Recreational fishing (8, 9, 37, 49, 53, and 68) + Boating (67)
	Symbolic	Habitat (67) + Biodiversity (67) + Natural vegetation (71) + Native animals (71)

Table 6. Valuation methods used in HEM studies for the considered environmental and non-environmental impacts.

		Consumer Surplus	**Production-based**	**Cost-based**	**Revealed preferences**	**Stated preferences**	**Benefit transfer**	**Other**
Provisioning	**Commercial fishing**		+ + +					
	Urban demands	+ + +						
	Agricultural demands	+ +	+ + +					
	Hydropower demands		+ + +					
	Industrial demands	+ + +	+ +	+				
	Navigation demands		+ + +					+ +
	Livestock demands		+ + +					
Regulation & Maintenance	**Water quality**			+ + +				
	Wastewater treat.			+ + +				
	Salt dilution			+ + +				
	Waste heat			+ + +				
	Flood control			+ + +				+ +
	Carbon sequestration			+ + +				
Cultural	**Tourism**		+ +		+ + +			
	Recreation				+ + +		+	+ +
	Recreational fishing				+ + +			
	Boating				+ + +			
	Habitat						+ + +	
	Biodiversity						+ + +	
	Natural vegetation					+ + +		
	Native animals					+ + +		
Not in CICES	**Wetlands**		+ +				+ + +	
	Environmental flows			+ + +		+ + +	+ + +	+ + +
	Environment							+ + +

*** The most used method on which the calculation was based is indicated with +++, the second most with ++, and the third most with +; empty values mean that the method was not used. Based on de Groot *et al.* (2002).**