Luis Chicharo · Felix Müller
Nicola Fohrer *Editors*

Ecosystem Services and River Basin Ecohydrology

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Foreword

The visionary UN document "The future we want", formulated at Rio +20, defines sustainability as a strategic goal for humanity in the twenty-first century, which means harmonization of social needs with biosphere potential. Such a broadly accepted approach focuses on the human being as the central reference point for sustainability efforts.

However, such approach has been steadily contributing to habitat degradation and disruption of ecological cycles, and to the decline of ecosystem services and overall biosphere potential.

The dynamic and diversified status of the biosphere is best described by the Greek expression *panta rei*, which means that biological evolution is, primarily, the function of the process of which the major driver has been the water cycle. Therefore, the critical condition for harmonizing the biosphere potential with increasing demography and consumption, is the understanding of the "water-biota interplay" as the basis for the enhancement of the ecological potential of ecosystems modified by humans.

Consequently, the major goal of ecohydrology as a "problem-solving science", is the enhancement of the ecosystems carrying capacity based on the understanding of the "dual regulation" interactions between water-biota, for the regulation of the hydrological cycle, as a way to facilitate the ecosystems adaptation to global change (Zalewski 2006).

Coastal zones are critical for biogeosphere sustainability because they are inhabited by more than half of humanity (Chicharo et al. 2009). Such interfaces between continental land masses and oceans are vulnerable to climatic changes owing to sea-level increase, acidification and habitat degradation. Ecotone zones between continental masses and oceans present a high potential for the biological productivity, biodiversity and resilience. This book explores key issues for achieving "the future we want" by providing fundamentals of the knowledge about the dependence of ecosystem services on three-dimensional interactions: oceans/coastal, terrestrial and freshwater ecosystems. The subsequent chapters of *Ecosystem Services and River Basin Ecohydrology* provide sound examples for assessing and enhancing ecosystem services for solving the sustainability problem, using the different disciplines of environmental science, which in turn provides a new, holistic perspective for enhancement of ecosystem potential and its harmonization with society needs.

Finally, this book will encourage scientists and practitioners acting at multi-dimensional and multi-scale levels at the continuum river basin-coastal areas, to consider a synergetic integrative approach between the biological, physical and chemical processes with the services provided by the ecosystems, in order to generate a sustainable coexistence between the humanity and the biogeosphere.

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References

Zalewski M (2006) Ecohydrology: an interdisciplinary tool for integrated protection and management of water bodies. Archiv für Hydrobiologie. Supplement. Large Rivers 16(4):613–622

Chicharo L, Ben Hamadou R, Amaral A, Range P, Mateus C, Piló D, Marques R, Morais P, Chícharo MA (2009) Application and demonstration of the Ecohydrology approach for the sustainable functioning of the Guadiana estuary (South Portugal). Ecohydrol Hydrobiol 9(1):55–71

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Introduction

Luis Chicharo, Felix Müller, and Nicola Fohrer

Abstract Ecohydrology is a new challenging approach to aquatic ecosystems management that considers dual regulation between biota and hydrology, at the entire river basin scale, as the key to restoration and long term sustainability of aquatic ecosystems health. Managing human-environmental systems, such as river basins, is strongly accompanied by several decisions about the intensity of human actions, which potentially affect natural or nature-near ecological systems. At river basin, upstream uses of water impact on the ecohydrologic functioning of downstream ecosystem, and on the tradeoffs between provision of upstream and downstream ecosystem services. This chapter introduces the the importance of integrating the concept of ecosystem goods and services with the ecohydrology approach, for the management of water resources from riverine do estuarine and coastal ecosystems.

Keywords Ecosystem services • Ecohydrology • River basin management

The Gaia hypothesis considers that the earth is a living organism and that life on the planet is maintained by biogeochemical cycles, supported by the biologic component. The hydrologic cycle is a focal determinant for the distribution of plants and animals, nutrient transport, water availability for all living organism, but also for all domestic and industrial activities. Intensive use of agricultural fertilizers, untreated discharge of industrial and domestic pollutants into water bodies, river damming, excessive groundwater abstraction or sea level rise, among others, are impacting water quality and quantity negatively, at the entire river basin scale and consequently, the goods and services provided by aquatic ecosystems.

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Ecosystems services were defined as the benefits people obtain from ecosystems (Millennium Ecosystem Assessment 2005). Despite the anthropocentric perspective subjacent to this definition, the fact that ecosystems need to be kept functional to support human needs implies a positive feedback on ecosystems conservation and restoration. In fact, the provision of benefits from ecosystems requires that the ecosystem carrying capacity is respected and that it is kept at sustainable levels.

The ecohydrology approach was developed throughout the UNESCO international Hydrologic Programme (IHP). It was founded based on the knowledge on the evolutionary established processes of aquatic ecosystems, and on the dual regulation between hydrology and biota. This approach aims to restore the ecosystems' carrying capacities, so that they can become resilient on the long-term and adaptable to impacts at the entire river basin scale.

With this book we aim to bring together the concepts of ecohydrology and ecosystem services. The ecohydrology approach is a concept to promote and sustain the provision of ecosystem services. Both, ecosystem services and ecohydrologic solutions need to account for the impacts on water quantity and quality at the river basin scale, as uses of ecosystem services upstream affect the uses of ecosystem services downstream, as well as ecohydrological functions and solutions need to be integrated at the river basin scale, from the river to the coast (Fig. 1).

Ecosystem services can be used to support decision making processes about the fate of human-environmental systems. They are well-suited to demonstrate man's dependence on the integrity of the surrounding ecosystems. There are several indicator based approaches for their quantification and consistent framework

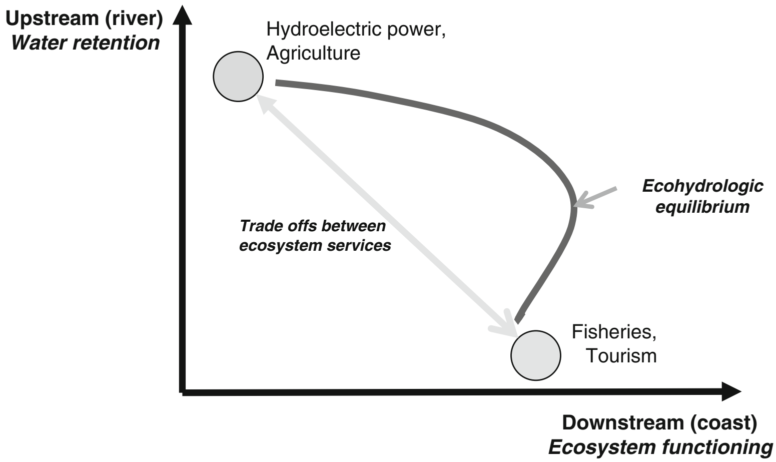

Fig. 1 High water retention in upstream river basin areas reduce the ecosystem functioning of downstream ecosystems, as estuaries and coastal waters. The ecohydrologic equilibrium indicates the best ecological balance between water retention upstream with the sustainability of downstream ecological functions. The ecohydrologic equilibrium also corresponds to the best trade-offs between upstream (eg. damming for hydrolectric power production or agriculture) and downstream ecosystems services (eg. coastal productivity and fisheries or tourism)

models to integrate ecosystem services in management activities. The literature also provides interesting approaches to apply the ecosystem service concept in the context of ecohydrological science and application. Some of these developmental directions will be documented in the following chapters.

The book is structured in 15 chapters, starting with an introduction to ecosystem services and ecohydrology, followed by a critical analysis of the methodologies and finalizing with cases studies on the multiple relations between ecohydrology and ecosystem services, from different world regions and covering both upstream to downstream areas of river basins.

Chapters "The Basic Ideas of the Ecosystem Service Concept", "Cultural Services in Aquatic Ecosystems", "The Importance of Hyporheic Zone Processes on Ecological Functioning and Solute Transport of Streams and Rivers", and "Marine and Coastal Ecosystems: Delivery of Goods and Services, Through Sustainable Use and Conservation" introduce ecosystem services. In chapter "The Basic Ideas of the Ecosystem Service Concept", Müller et al. define the ecosystem services concept as a set of environmental properties deriving from ecosystem structures and processes which are arranged from an anthropocentric point of view. Ecosystem services are the benefits people obtain from ecosystems, and from an economic viewpoint they can be understood as "flows of value to human societies as a result of the state and quantity of the natural capital" (Wallis et al. 2011). Sustaining ecosystem services at the river basin scale requires understanding, maintaining and restoring the basic ecological functions of aquatic ecosystems. Ecohydrology is a concept that aims to restore and maintain the healthy functioning of aquatic ecosystems, based on the interrelations between biota and hydrology, at the entire river basin. Thus, sustainable ecosystem services are dependent of sustainable ecohydrologic functioning of ecosystems. Consequently, tradeoffs between different water uses, at different ecosystems in the river continuum, from upstream to downstream, need to be harmonized and integrated.

In chapter "Cultural Services in Aquatic Ecosystems", Rodrigues highlights the importance of non-material benefits obtained from ecosystems, as cultural services. Cultural ecosystem services are often undervalued, underprotected, and neglected from ecosystem services studies. This arises from difficulties in their operation such as uncertainties on their generation and on people's demand for cultural ecosystem services.

Also, few studies were undertaken on open marine and deep water ecosystems services, as indicated by Borja et al., in chapter "The Importance of Hyporheic Zone Processes on Ecological Functioning and Solute Transport of Streams and Rivers". Because of that, despite various legislations implemented worldwide, the authors propose the precautionary principle to be followed and they advocate protection, prevention and restoration activities regarding these ecosystems and the respective benefits for human societies.

As ecosystems services are increasingly demanded by the human population, changes in demography may cause shifts on the pressures on the services and their sustainable availability. These aspects are analyzed in detail in the chapter by Haase and colleagues in chapter "Marine and Coastal Ecosystems: Delivery of Goods and Services, Through Sustainable Use and Conservation".

To quantify the special potentials for service provisions as well as the flows of services themselves, we need variables that provide aggregated information on these phenomena. Such indicators are selected to support specific management purposes, with an integrating, synoptic value, functioning as depictions of qualities, quantities, states or interactions that are not directly accessible. Rode et al., in chapter "Terrestrial Ecosystem Services in River Basins: An Overview and an Assessment Framework", indicate that solute transport links terrestrial and aquatic systems and upstream and downstream aquatic systems and may be used as an indicator of the effects of anthropogenic disturbance on catchment properties. They call attention to the need to consider the functions of the hyporheic zone, that may be the bottleneck for ecosystem integrity of riverine ecosystems. The ability of streams to assimilate or degrade pollutants from point or nonpoint sources is a highly important ecosystem service and depends on the maintenance of ecosystem integrity.

Most of the ecosystem services are not limited to terrestrial ecosystems. They can be provided both by terrestrial and aquatic ecosystems. In river basins, natural conditions, land use structures and processes are very important, particularly in view of hydrological aspects as water runoff, water retention, groundwater recharge or water pollution. To assess ecosystems in river basins, particularly in view of their multifunctional use and sustainable development, Bastian et al. (Chapter "Quantifying, Modelling and Mapping Ecosystem Services in Watersheds") propose an application of the EPPS assessment framework (Ecosystem Properties, Potentials and Services), an approach that can be used in terrestrial as well as aquatic ecosystems.

Also considering the complexity and multiple ecosystem functions occurring in watersheds, Nedkov et al. (Chapter "A Methodology for Quantifying and Mapping Ecosystem Services Provided by Watersheds") propose a matrix approach that links different land cover types within watersheds, to different ecosystem functions and services. Authors sustain that this approach allows a more comprehensive ecosystem service quantification, modelling and mapping in watersheds.

The accurate representation of the delivery of many water-related ecosystem services requires an understanding of natural hydrologic connectivity as well as how it has been modified by human actions. Villamagna et al., in chapter "Assessing the Impact of Land-Use Changes on Providing Hydrological Ecosystem Functions (ESF) and Services (ESS) – A Case-Study Experience Based Conceptual Framework", direct the attention to the need of new methods to measure and monitor the dynamics of aquatic ecosystem services, to assess their sustainability, and to guide their management. In their Chapter, they present methods to represent ecosystem services based on the integration of physicochemical, biological, and social processes the across entire watershed areas.

In the same line, Fürst and Flügel, in chapter "Valuation of Ecosystem Services Regarding the Water Framework Directive on the Example of the Jahna River Catchment in Saxony (Germany)", propose a conceptual framework for assessing the impact of land use and land cover change on the provision of hydrological ecosystem functions and services. Their approach has the advantage to consider land-use dynamics within hydrological units, thus contributing to a better understanding of ecological processes, their relation to functions and impacts on services over different scales.

The next set of chapters present results from different case studies around the world.

Schmalz et al, in chapter "Water-Related Ecosystem Services – The Case Study of Regulating Ecosystem Services in the Kielstau Basin, Germany", present a study case from the Kielstau catchment, a UNESCO demosite for ecohydrology in Northern Germany, where they linked the SWAT model results on temporal changes in erosion with involvement of stakeholders for the assessment of ecosystems services and for the participation in decision making processes. This case study highlights the contribution that different quantification methods such as modelling, can provide for decision making processes on the spatio-temporal variations in ecosystem services.

At the Great Ruaha River, Tanzania, demography, excessive withdrawals of water, land use changes, exotic species invasions and climate change are impacting major aquatic ecosystems. Kaaya and colleagues, in chapter "Aquatic Ecosystem Services and Management in East Africa: The Tanzania Case", propose to apply the Integrated Water Resource Management framework as a legal basis for the management of the river and the sustainability of the services provided by the ecosystem.

Still in Africa, Agboola and Shakirudeen refer, for Ghana and Nigeria, in chapter "Coastal Watershed Ecosystem Services Management in West Africa: Case of Ghana and Nigeria", similar pressures on ecosystem functions and services. In this case study they present scenarios for the management of coastal watershed ecosystem services, based on identifying and merging socio-economic and climate change drivers.

Brodie et al. present a case study from the Great Barrier reef in Australia (Chapter "Management of Agriculture to Preserve Environmental Values of the Great Barrier Reef, Australia") demonstrating the effects resulting from changes in continental land use due to agriculture, and from discharge of polluted river waters, in the coral reefs, with consequences on ecosystem services such as tourism and fishing activities.

In chapter "Ecohydrology: A New Approach to Old Problems for Sustainable Management of Aquatic Ecosystem of Bangladesh for Ecosystem Service Provision", Sohel indicates several ecosystem services from Bangladesh, as fisheries or shrimp aquaculture, that are threatened by a degradation of aquatic ecosystem health, mainly caused by pollution, and river water diversions. In this chapter, Sohel proposes several ecohydrologic measures to resolve or mitigate the negative impacts on aquatic ecosystem health and therefore, to sustain the ecosystem services provided.

From the methodological discussions and case studies presented it is possible to conclude that ecosystem services are being threatened around the world by similar factors. Aquatic ecosystems respond in similar ways, so similar solutions and approaches can be implemented. Ecohydrology is an ecological based approach to support the sustainability of aquatic ecosystems and the services they provide. Both approaches show several interrelations.

The chapters also demonstrate that a high level of knowledge and methodology has been reached concerning the young approach of ecosystem services. But it is possible to conclude that there is still a lot of space for methodological improvements and that the creation of adapted frameworks is crucial for the harmonization of the application of the concept, worldwide and in dealing with multi stressor factors.

References

MEA [Millennium Ecosystem Assessment] (2005) Ecosystems and human well-being: synthesis. Island Press, Washington, DC

Wallis C, Séon-Massin N, Martini F, Schouppe M (2011) Onema meeting: implementation of the water framework directive – when ecosystem services come into play. Brussels. ISBN: 979-10-91047-06-7

The Basic Ideas of the Ecosystem Service Concept

Felix Müller, Nicola Fohrer, and Luis Chicharo

Abstract In this chapter we discuss the different definitions, advantages and limitations of the concept of ecosystem services. Ecosystem services are sets of environmental properties derived from ecosystem structures and processes which are arranged from an anthropocentric point of view. They are the benefits people obtain from ecosystems, and thus they can be used to represent the environmental interrelations between the three sectors of sustainability. The managing human-environmental systems, such as river basins, is strongly accompanied by several decisions about the intensity of human actions, which potentially affect natural or nature-near ecological systems. At river basins, upstream uses of water impact on the ecohydrologic functioning of downstream ecosystems, and thus provoke tradeoffs between the provision of upstream and downstream ecosystem services. Ecosystem services can be classified in many ways depending on the objectives of the observer but in all recent classifications, three groups of services are distinguished: regulating, provisioning and cultural services. The Identification of ecosystem services can be used to quantify the impacts on ecological systems, to contribute to the identification of gaps and to provide policy-relevant information on a sustainable use of these services to maintain their capacities for future generations. The ecosystem service concept has been developed very fast during the last years and a methodological framework has been created that can be applied in environmental management. Incorporating this concept into assessment frameworks, such as the DPSIR approach, makes it more broadly applicable. Despite some limitations, ecosystem services have the advantage to demonstrate the enormous human dependence from nature by focusing on the critical roles of ecosystem functions and structures for sustaining human life and well-being.

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Keywords Ecosystem service definition • Ecosystem service classification • Ecosystem service quantification and indication • Ecosystem service valuation • Uncertainties

1 What Are Ecosystem Services?

Ecosystem services are sets of environmental properties deriving from ecosystem structures and processes which are arranged from an anthropocentric point of view: They describe those products and outcomes from complex ecological interrelations which are useful and necessary for human well-being. Ecosystem services are the benefits people obtain from ecosystems, and thus they can be used to represent the environmental interrelations between the three sectors of sustainability. From an economic viewpoint they can be understood as "flows of value to human societies as a result of the state and quantity of the natural capital" (Wallis et al. 2011, 24).

These definitions seem to be very similar, but they do differ in some points: On the one hand, some authors make a difference between goods and services (e.g. Costanza et al. 1997) while in most cases both aspects are unified in the term service. On the other hand, in some definitions the ecosystem services inevitably have to be based on ecosystem functions and biological processes, thus some sections of the natural capital (e.g. mineral resources, wind, solar radiation) are not taken into account (see for example CICES.EU, Haines-Young and Potschin 2013). These features of the mainly abiotic sphere of nature can be assigned as parts of the environmental services, which more or less represent the overall natural capital (Wallis et al. 2011).

Furthermore, in some definitions the ecological processes, conditions, or functions which produce the services are significant elements while in others the resulting benefits are the focal point of view. Also human investments and the combination of natural and human capital are components of some definitions while others do not consider these inputs.

Another problem which appears in several discussions (there is a nice sequence of papers in the literature basing on the papers from Boyd and Banzhaf 2007; Wallace 2007, 2008; Costanza 2008; Fisher and Turner 2008) is related to the challenge of double counting: Some ecological processes have an indirect effect on those ecosystem services which are finally consumed (and paid). Pollination is an example which in many cases is investigated as a significant service, although the final products are the fruits which can be harvested. In an accounting system there is the danger that these components are added, thus producing an unbalanced result. Consequently, some authors propose to concentrate only on the end-products, the so-called final ecosystem services and to neglect the intermediate services, which are not directly consumed. Other scholars state that also the intermediate products/producers are services, and that they can play major roles in assessments: "as long as human welfare is affected by ecological processes or functions, they are services, be it direct or indirect" (Fisher and Turner 2008, 1,168).

This discussion is a detailed continuation of the decline of supporting services, which have been defined by the Millennium Ecosystem Assessment (MEA 2005) as ecosystem components which are not directly consumed but which contribute to the output of those services which provide such a final product. Supporting services (e.g. primary production, soil fertility,…) were understood as basic necessities for the production of all other ecosystem services. A reflection of this concept makes clear that all ecosystem processes consequently can be called supporting services. And of course then all structural items (e.g. biodiversity) would also fit into this category.

To solve this double counting dilemma, service and function have to be distinguished. While the functional quality of an ecosystem can be described un-valued by integrity variables or state indicators (Müller and Burkhard 2007), ecosystem services have to provide a contribution to human well-being; there must be a demand for the results of the respective environmental processes (Haines-Young and Potschin 2010a). This demand can be formulated rather easy for ecosystem goods or cultural contributions, but is becomes difficult if the results of ecological regulations are discussed. For example the storage of carbon compounds in the soil is the result of typical and complex ecological processes. At a first glance there is no obvious direct demand. But the demand is formulated from the viewpoint of global climate change: To reduce the greenhouse effect, we should attempt to store as much carbon as possible. Therefore, carbon sequestration is related with positive influences on human well-being. Thus, the CO_2 fixation can be assigned to benefits for human society and consequently be understood as an ecosystem service. The difference is a matter of recognition, and thereby the threshold between function and service becomes a little diffuse: it is dependent on the societal perception, and although many services are not known by the public (or even by science), they are existing. And even if we follow Boyd and Banzhaf (2007) in concentrating on final services, we should be aware that the accounted final end-products are connected within a complex network of ecological interrelations that have to be supported if the demanded services shall be used by human society.

To overcome these problems and challenges, many authors have constructed conceptual frameworks for ecosystem service assessments. Figure 1 shows the so-called ecosystem service cascade after Haines-Young and Potschin (2009) which is the most frequently used framework today. It demonstrates a functional hierarchy of ecosystem processes and structures which is ordered to focus on the (known) contributions of ecosystem relations for providing human benefits: All the multiple objects of ecological investigations can be characterized by certain isolated ecosystem properties. They may refer to the structures as well as the processes in an ecosystem. These items are bundled in the set of ecosystem functions, which are able to derive the potentials of an ecosystem to provide a certain service as a result of intensive interactions between structural units and processes. The functionality of an ecosystem can be indicated by its health or integrity or sets of other state variables (Jörgensen et al. 2010).

The functions are turned into services if they are utilized to produce a benefit related to social, economic or personal well-being factors. Consequently, services are groups of functions that are selected due to their utility for human society. If

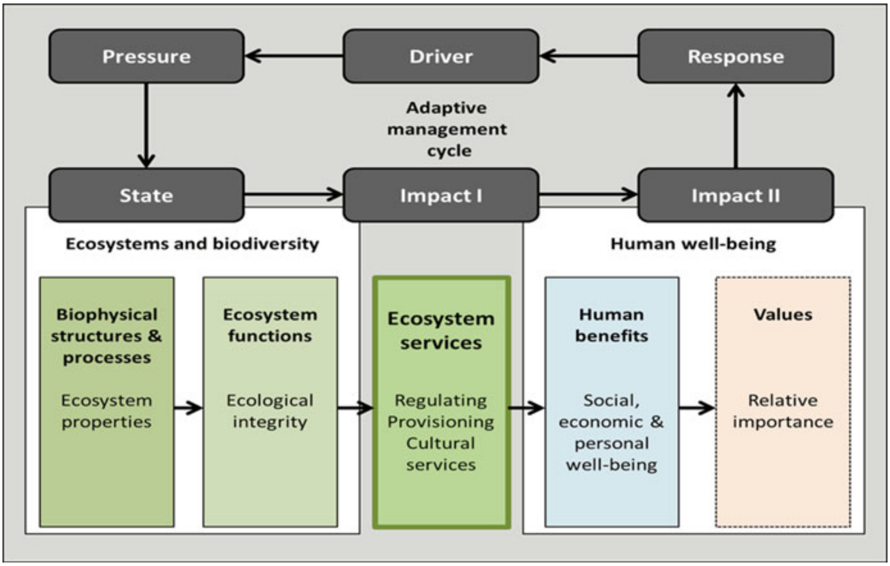

Fig. 1 The 'ecosystem service cascade' embedded in the adaptive DPSIR indicator and management cycle after Haines-Young and Potschin (2009, 2010b) and de Groot et al. (2010b)

these services have a high significance they will receive a high societal value, and their relative importance will be highly considered in human-environmental trade-offs. These values will be different at different places as the demands for the mentioned benefits are varying spatially due to the special site conditions. They will be of different significance for different groups of people due to their specific objectives and backgrounds. They will furthermore be different due to varying degrees of ecological comprehension, and there will be temporal differences due to the dynamics of special pressures on sustainable developmental pathways.

Thus, the categorization of a certain set of functions as ecosystem services is a relatively subjective result of the observer's comprehension and the respective targets of the investigation. Of course, from a systems viewpoint, the selected set of services should be capable of fulfilling the demands of a holistic analysis: All service classes should be represented, and the number of involved services should be as high as possible although that requirement may lead to a huge pile of different tasks for the managing staff. But we should keep in mind that trade-offs can only be analyzed and solved satisfactory if the multiple components of service provisions are really known.

2 Why Do We Need Ecosystem Services?

Managing human-environmental systems, such as river basins, is strongly accompanied by several decisions about the intensity of human actions, which potentially affect natural or nature-near ecological systems. Any of these decisions must be based on a thorough assessment of the potential outcomes of alternative activities and a valuation of the respective chances, risks and potentials (European Commission 2006; Helming et al. 2008; OECD 2010). For such a valuation it is necessary to utilize a conjoint decision basis, a common set of criteria, thresholds and arguments which reflect the targets, borderlines and paradigms that are defined by the society. Therefore, the argumentation within the decision making process should have an ethical fundament: It should be based on a "set of accepted norms, values and informal rules that guide individual and collective behavior" (Jax et al. 2013). But which are those guidelines to measure the accordance of management outcomes and normative settings if human-environmental trade-offs have to be assessed?

As we are discussing about socio-ecological systems, one potential (and very general) answer could be: we have to foster the sustainability of the respective development. For this purpose we have to look for solutions which "meet the needs of the present without compromising the ability of future generations to meet their own needs" (United Nations Assembly 1987), from an environmental, a social and an economic viewpoint. Consequently, the key variables which can be used for the decision making process have to take into consideration key parameters from all three mentioned sustainability pillars, comprising arguments about the environmental situation, the social welfare and the economic conditions.

There are two approaches to do so: on the one hand we can look for separate indicators from the three sectors and ask how they change as a consequence of different management measures (European Commission 2006; Mander et al. 2010). This will lead to a semi-integrative viewpoint because the sectors are observed in isolation. But as we are discussing about human-environmental *systems*, the *interrelations* between the three pillars have to be placed in the focus of the decision making process. Thus, we need integrative models and indicator concepts which demonstrate how the social, economic and environmental sub systems are linked. Although the basic concept of sustainable development is a totally anthropocentric approach, the environmental items should play a focal role. Thus, from an ecological viewpoint (e.g. de Groot et al. 2002; Grunewald and Bastian 2015; Jax et al. 2013), the variables used for the aspired evaluation should

– *demonstrate the enormous human dependence from nature*: Throughout decision making processes it has to be clarified that natural resources are vital for human welfare, endangered by human actions and restricted due to the limited natural potentials and resilience (Gunderson and Holling 2003). The significance of ecosystem provisions for human quality of life should be a major argument within the valuation concept, i.e. because all sustainability approaches and sectors are based on the availability of natural structures, functions and products.
– *focus on the critical roles of ecosystem functions and structures for sustaining (human) life and well-being*: The described human demands for natural products and processes can only be fulfilled if the structural components of ecosystems (the ecosystem processors) are interacting in a healthy manner (Müller et al. 2012), thus supporting functional process bundles that fulfill human requirements e.g. for food, beverages, clean air, or effective shelters. These items in general can be provided on the long-term if the ecological integrity of the respective ecosystems is not affected (Müller 2005).

- *allow intersubjective judgements about human actions with respect to natural structures and processes*: The variables used as indicators have to provide an objective picture of the potential outcomes of the decisions in question. This demand includes the feedbacks from provoked environmental pressures to social and human issues. Therefore, a transition of potential externalities should be attempted, formulated in way that is also comprehensible from social and economic attitudes (Wiggering and Müller 2003).
- *link scientific state descriptions with management demands and normative items*: Very often the decision made is based on normative arguments. Scientific descriptions are helpful tools in the management process providing the best possible objectivity and scenario technology. They can be accomplished by normative values, if the basis of this standardization is consensual, thus, if it refers to a description of sustainability items and if all participants agree with the unavoidable normative loadings.
- *be usable as a communication tool between different interest groups*: During assessment processes several stakeholder groups with varying backgrounds and interests have to participate in the discussions of the focal item. Therefore, a joint language level and a transparent, easy understandable level of valuation with a high general identifiability should be used to find the optimal outcome of the trade-offs (Jax et al. 2013).
- *be expressible in different dimensions, e.g. as monetary and non-monetary values, capable of making environmental externalities implicit*: To cope with the distinct origin of the participants in an assessment process, it makes sense to express the outcomes of decisions in physical units as well as economic or social values (de Groot et al. 2010a) in order to implement and illuminate the preferences people have for a benefit amongst a set of alternatives (Haines-Young and Potschin 2009).
- *be able to demonstrate synergies, trade-offs and conflicts*: In assessment processes it is important to show the consequences of potential decisions related to all sectors of sustainability. Thus, the framework should include the possibility to demonstrate the potential losses and gains of all scenarios in a transparent manner (Verburg et al. 2006).
- *provide a principle motivation from concerns about an undervaluation of biodiversity*: Of course, from an ecological viewpoint the conservation and development of biodiversity items is of central interest. Thus, also the discussed issues of the trade-off procedures should have a high relation to biodiversity features (MEA 2005).
- *fulfill the general demands of sustainable development*: The resulting approaches have to contain long-term and multi-scale strategies; they should be interdisciplinary in character, holistically based, nature-oriented and theory-based (Wiggering and Müller 2003). In this context it is especially important that the variables used can be applied in long-term scenarios to fulfill the temporal demands of the sustainability approach.

– *have a systems-based fundament in the ecosystem approach* (CBD 2000): To realize the demands of the ambitious 12 "Malawi Principles"[1] of the CBD, focus of all biodiversity related strategies should be put on the relationships and processes within ecosystems enhancing benefit-sharing, e.g. by valuing the products and outcomes of ecosystem processes due to their long-term usefulness for societies (Table 1).

Table 1 Some definitions of ecosystem services

Daily (1997)	Ecosystem services are the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life
Costanza et al. (1997)	Ecosystem goods (such as food) and services (such as waste assimilation) represent the benefits human populations derive, directly or indirectly, from ecosystem functions
Boyd and Banzhaf (2007)	(Final) Ecosystem services are components of nature, directly enjoyed, consumed, or used to yield human well-being
Fisher and Turner (2008)	Ecosystem services are the aspects of ecosystems utilized (actively or passively) to produce human well-being
Millennium Ecosystem Assessment and IUCN Commission on Ecosystem Management[a, b] (2005)	Ecosystem services are the benefits people derive from ecosystems
	Ecosystem services are the benefits people obtain from ecosystems and also the processes that produce or support the production of ecosystem goods
TEEB (2010)	Ecosystem services are the direct and indirect contributions of ecosystems to human well-being. The concept "ecosystem goods and services" is synonymous with ecosystem services
Haines-Young and Potschin (2009)	Ecosystem services are the contribution which the biotic and abotic components of ecosystems jointly and directly make to human well-being; an 'end-product' of nature
Burkhard et al. (2012)	Ecosystem services are the contributions of ecosystem structure and function – in combination with other inputs – to human well-being
UK National Ecosystem Assessment[c]	Ecosystem services are the benefits provided by ecosystems that contribute to making human life both possible and worth living
Villamagna and Angermeier (this book)	The term "ecosystem service" applies (only) to those ecological processes and features that confer clear benefits to people
Pinto and Marques (this book)	Ecosystem services can be defined as the functions of ecosystems with value for human well-being
Sohel (this book)	"Ecosystem services" is a collective term for the goods and services produced by ecosystems that benefit humankind

[a]http://www.iucn.org/about/union/commissions/cem/cem_work/cem_services/
[b]http://www.unep.org/maweb/en/index.aspx
[c]http://uknea.unep-wcmc.org/EcosystemAssessmentConcepts/EcosystemServices/tabid/103/Default.aspx

[1] See also http://www.cbd.int/ecosystem/principles.shtml

This list shows some of the focal demands on indicator and valuation frameworks to carry out a sustainability assessment. On the following pages the ecosystem service approach will be sketched in order to check if these requirements can be fulfilled by that concept. Therefore, the focal task of this chapter is to provide a general introduction into the ecosystem service approach and a critical reflection of its potentials and limitations. For these purposes we start with a presentation of ecosystem service definitions, introduce one possibility for service classification and try to relate the services to components of human well-being. Thereafter, some general introductions will be made concerning the methodology of the approach and its position in assessment studies. Finally we ask for the special features of the application of the concept in river basin hydrology and for the problems that might evolve from an analysis of the potentials and the requirements mentioned above.

3 Which Ecosystem Services Are Relevant for River Basin Ecohydrology?

This volume focuses on ecosystem services and river basin ecohydrology. Therefore, in the following articles, special emphasis will be put on those ecosystem services which are related to hydrological structures and processes as well as the ecological interrelations in aquatic ecosystems. Due to the differences in living conditions, scales, land use patterns and habitat features it is useful to make a distinction between those services provided by the terrestrial environment of the watershed and those services provided by the aquatic ecosystems themselves. The first group – *watershed ecosystem services* – are the benefits people obtain from ecosystems in the area of land, which drains into a certain stream watershed (Smith et al. 2006). They can easily be reflected by the lists in Tables 2, 3, and 4. The watershed service analyses can furthermore be characterized by special focal objectives, e.g. stressing ecohydrological aspects, water flow, nutrient and erosion regulation, flood protection. In the literature, general concepts for watershed related ecosystem services and their classifications can be found e.g. in the papers of Postle et al. (2005), Lele (2009), Ojea et al. (2012) or Townsend et al. (2012), while water and resource management related questions are foci of the articles from Jewitt (2002), Gordon et al. (2010), Pert et al. (2010) or Posthumus et al. (2010). Specific services with relevance to watershed questions are e.g. discussed in Veloz et al. (1985), Nedkov and Burkhard (2012), Wang et al. (2010), and Loomis et al. (2000), Johnson and Baltodano (2004), Dong et al. (2011) provide works on economic valuation techniques for watershed ecosystem services.

Instream ecosystem services are the benefits people obtain from the river streams and their ecosystems (Chetverikova 2012). They are specifically classified at the right sides of Tables 2, 3, and 4. Instream services are strongly interlinked with watershed services, as e.g. irrigation water is applied in the terrestrial environmental but taken from the aquatic systems. Nevertheless, consumptive services such as water supply for irrigation, for potable water, for demands from domestic, municipal,

Table 2 List of important regulating services after Kandziora et al. (2012) (In the original version also indicators for these services are listed)

Regulating service	Definition	Relevance for aquatic ecosystems
Global climate regulation	Long-term uptake and storage of greenhouse gases in ecosystems providing reduced pressures from atmospheric CO_2 concentrations	
	Exemplary benefits: Deceleration of global climate change dynamics	
Local climate regulation	Regulation of local climate components like wind, precipitation, temperature, or radiation due to ecosystem properties and control processes	
	Exemplary benefits: Optimization of local living conditions	
Air quality regulation	Capture, adsorption and filtering of air particles, dust, chemicals and gases due to eco-chemical processes	
	Exemplary benefits: Cleaning the air to improve people's health	
Water flow regulation	Control of processes of the water cycle (e.g. water storage and buffer, natural drainage, irrigation and drought prevention)	
	Exemplary benefits: Providing usable quantities and ratios of water and water products	
Water purification	Control of chemical compositions in waters, e.g. operating sediments, pesticides, disease-causing microbes and pathogens	
	Exemplary benefits: Providing usable quantities and ratios of water and water products	
Nutrient regulation	Recycling, metabolization and storage of nutrients, e.g. N, P, K	
	Exemplary benefits: Quality of drinking water and aquatic ecosystems	
Erosion regulation	Soil retention and avoidance of soil erosion and landslides	
	Exemplary benefits: Optimization of soil fertility and water quality	
Natural hazard protection	Protection and mitigation of floods, storms (hurricanes, typhoons…), fires and avalanches	
	Exemplary benefits: Risk reduction for the human population	
Pollination[a]	Assistance of plant reproduction and fruit growth by bees, birds, bats, moths, flies, wind	
	Exemplary benefits: Food provision and biodiversity of plants	
Pest and disease control[a]	Control of pests and diseases due to genetic variations of plants and animals making them less disease-prone and by actions of predators and parasites	
	Exemplary benefits: Human health	
Regulation of waste[a]	Control of filter and decomposition processes concerning organic material in water and soils	
	Exemplary benefits: Secure storage and degradation of human wastes	

[a]These ecosystem services are listed here because they can be of high importance in some ecosystems though the potential of double-counting must be noted

Table 3 List of important provisioning services after Kandziora et al. (2012)

Provisioning service	Definition	Relevance for aquatic ecosystems
Crops	Cultivation of edible plants and harvest of these plants on agricultural fields and gardens that are used for human nutrition	
Biomass for energy	Plants used for energy conversion (e.g. sugar cane, maize)	
Fodder*	Cultivation and harvest of fodder for domestic animals	
Livestock (domestic)	Production and utilization of domestic animals for nutrition and use of related products (dairy, wool)	
Fibre	Cultivation and harvest of natural fibre (e.g. cotton, jute sisal, silk, cellulose) for e.g. cloths, fabric, paper	
Timber	Wood used for construction purposes	
Wood fuel	Wood used for energy conversion and/or heat production	
Fish, seafood and edible algae	Catch of seafood/algae for food, fish meal and fish oil	
Aquaculture	Harvest of seafood/algae from marine and terrestrial aquaculture farms	
Wild food, semi-domestic livestock and ornamental resources	Harvest of berries, mushrooms, (edible) plants, hunted wild animals, fish catch from recreational fishing, semi-domestic animal husbandry and collection of natural ornaments (e.g. seashells, leaves and twigs for ornamental or religious purposes)	
Biochemicals and medicine	Natural products used as biochemicals, medicine and/or cosmetics	
Freshwater	Used freshwater (e.g. for drinking, domestic use, industrial use, irrigation)	
Mineral resources**	Minerals excavated close from surface or above surface (e.g. sand for construction, lignite, gold)	
Abiotic energy sources**	Sources used for energy conversion (e.g. solar power, wind power, water power and geothermic power)	

**Potential double-counting when fodder is used for feeding on the same farm
***These services are often not acknowledged as ecosystem services (cp. Haines-Young and Potschin 2010a, b); but they can be of high importance for policy decisions, land use management strategies and scenarios on local and regional scales

industrial or household levels are stressed in the literature. Other important provisioning services are fish production or aquaculture. Furthermore special non-consumptive provisions use types derive from transport, shipping and navigation as well as hydropower or cooling water demands. Besides the very significant regulating services of water purification, sediment and erosion control, among the cultural services swimming, recreational fishing and boating are most often discussed. In the literature, instream ecosystem services are analysed from basic viewpoints e.g. in Bouwes and Schneider (1979), Pattanayak (2004), Willaarts et al. (2012), economic aspects can be found in the papers of Wilson and Carpenter (1999) or Doyle and

Table 4 List of important cultural services after Kandziora et al. (2012)

Cultural service	Definition	Relevance for aquatic ecosystems
Recreation and tourism	Opportunities for outdoor activities and tourism in the environment or landscape, including forms of sports, leisure and outdoor pursuit	
Landscape aesthetic, amenity and inspiration	Visual qualities of ecosystems and ecosystem complexes which influence human well-being, providing a source of inspiration for art, folklore, national symbols, architecture, advertising and technology	
Knowledge systems	The potential for environmental education, i.e. out of a formal schools context, and the knowledge in terms of traditional knowledge and specialist expertise arising from living in a particular environment	
Religious and spiritual experience	Spiritual or emotional benefits that people attach to local environments or landscapes due to religious and/or spiritual experience	
Cultural heritage and cultural diversity	Benefits that humans obtain from on the maintenance of historically important (cultural) landscapes and forms of land use (cultural heritage)	
Natural heritage and natural diversity	The existence value of nature and species themselves, beyond economic or human benefits, support of bequest and existence values	

Yates (2010), the recreational potential of rivers is described e.g. in Sanders et al. (1991) or Duffield et al. (1992), and specific applications, e.g. referring to community aspects, have been reported by Johnston et al. (2011), Holmlund and Hammer (1999), Hoeinghaus et al. (2009), and Xiang et al. (2010).

As water flows from rivers and streams to estuaries and coasts, the upstream use of water impacts on the downstream uses and on ecological functioning. River damming is reducing more than 30 % of world river discharge to estuaries and coastal areas. Such trapping of water upstream supports water uses for agriculture or hydroelectric production, but reduces estuarine and coastal productivity, changes biodiversity and causes shifts in coastal fisheries (Chícharo et al 2006; Morais et al 2009, 2010, 2012; Wolanski et al 2006). Thus, the tradeoffs between both uses of water need to be considered, harmonizing the ecohydrologic functioning of the downstream ecosystem with the services provided by the water both in the upstream and downstream areas.

4 Which Ecosystem Services Can We Distinguish?

In the literature, several ecosystem service classification frameworks can be found (e.g. Costanza et al. 1997; Boyd and Banzhaf 2007; Costanza 2008; Wallace 2008; Fisher and Turner 2008; Daily et al. 2009; De Groot et al. 2002, 2010a; Staub et al. 2011; MEA 2005; Burkhard et al. 2009; TEEB 2010; Haines-Young and Potschin

2010a, b, 2013; CICES 2013[2]). All these distinctions are based on different structures and viewpoints. They demonstrate that there are many useful ways to classify ecosystem services, depending on the objectives of the observer. In the following one example (after Kandziora et al. 2012) for one of these classifications will be presented. As in all recent classifications, three groups of services are distinguished: regulating, provisioning and cultural services.

Regulating services are the benefits people obtain due to the regulation of natural processes and the control or modification of biotic and abiotic factors (see also de Groot et al. 2002; Fu et al. 2011; Dale and Polasky 2007; Nedkov and Burkhard 2012). Being hardly visible and comparably difficult to understand, these services are not widely acknowledged by the society. This undervaluation displays an enormous error: As all produced goods or enjoyed structures depend on the healthy coordination of ecological controls and feed backs, the regulations in ecological systems are the very basic requirements for any ecosystem service. Therefore – in the opposite with the public recognition – they have to be listed at first due to their enormous significance.

As the regulations can hardly be measured by tangible products, they are often understood as indirect or intermediate services. Due to the double counting challenge, three of the services from Table 2 have been highlighted with a remark on this point. All the others are prominent benefits of natural systems for the sake of human environmental management. They are basic requirements for adequate human living conditions and – from that perspective – extremely important services.

Provisioning services comprise all material outputs from ecosystem processes that are used use for human nutrition, processing and energy use. These products can be traded and consumed or used directly, thus they are the desired 'end-products' of nature providing clearly visible benefits to society. Provisioning services can be divided into the subcategories of food, materials and energy (de Groot et al. 2010a, b; Haines-Young and Potschin 2010b). In Table 3 some non-ecological goods (which are not products of recent ecosystem processes) are listed as well, because these facets of natural capital can play major roles in environmental management.

Cultural ecosystem services are the intangible benefits people obtain from ecosystems in form of non-material spiritual, religious, inspirational and educational experience (Table 4). These services provide benefits for human recreation and mental and physical health, experience by tourism, aesthetic appreciation and inspiration for culture, art and design, spiritual experience and sense of place.

The listed ecosystem services from Tables 2, 3, and 4 can be activated by definition, if they produce benefits for individuals or human societies. Therefore, also the recipients of the ecological products – expressed as the criteria of human welfare – have to be considered. Table 5 represents a list of some of these criteria. Interrelating these categories with the potential products of the services makes clear that between these items a complex network is built up. All services can contribute to these components, and also the contribution of indirect relations becomes clear if this analysis is undertaken. Also the role of regulating services – which sometimes are catego-

[2] Cices.eu.

Table 5 Some components of human well-being, reflecting societal benefits of ecosystem services after Kandziora et al. (2012), Burkhard and Müller (2008a, b)

Components of human well-being	Definitions
Economic well-being	
Income	Disposable income, i.e. the income available to individuals for meeting their respective needs and the material basis available to each individual for participating in social life
Employment	Availability, diversity and security of jobs within the region, linked to the overall regional employment and unemployment ratios
Housing	Availability, quantity and quality of different housing options
Infrastructure	Availability, quantity and quality of infrastructural items for energy, water and material supply, transport and telecommunication
Security	Reduction of threats from environmental, social and economic crisis or catastrophe, extreme events (e.g. storms, fires, floods, droughts) and endangering human activities
Social well-being	
Nutrition	Availability and quality of food to optimize people's nutritional state
Demography	Dynamic changes of population numbers and composition
Health	Access to health infrastructure to optimize people's overall health status
Education	Availability, quantity and quality of all forms of education and training
Leisure	Quantity and quality of individual leisure and cultural activities (in- and outdoor) and provision of a respective infrastructure
Social relations	Personal stability resulting from social networking and interchange
Personal well-being	
Personal well-being	Subjective determinants of quality of life; an integration of all other issues

rized as intermediate services – can be illuminated by consequent interrelations between Tables 2 and 5: For example global climate regulation by forestry can provide employment and income for foresters, the products may be used for the construction of houses or infrastructural facilities, and a reduction of the atmospheric CO_2 concentrations will enhance people's security from extreme events. The forest will provide the opportunity for nutrition, it will improve people's health, e.g. simply by cooling down the air temperature, and it can be a place for leisure and social relations. For several persons forests provide cultural services which also enhance the personal quality of life.

This example illuminates the fact that the production of ecosystem services is a very complex and complicated process. That impression may even be enhanced if we take a look into the interrelations between ecosystem services, which should be a basic item in assessment processes. Table 6 shows the interrelations between ecosystem services based upon the conditions of a Northern German agricultural landscape or watershed. The relations represent the topical answers for the trade-off-questions: If I want to increase the provision of a certain service (here this 'active' service is to be found at the vertical y-axis), which will be the consequence

Table 6 A matrix showing the direct interrelations between the ecosystem services listed in Tables 2, 3 and 4, after Kandziora et al (2012)
The matrix has to be read from the vertical columns to the horizontal rows and it represents (i) a strong probability for exclusion and competition ↘, (ii) a strong probability for mutual support ↗ or (iii) no significant direct interaction

Ecosystem services (X) ⇗ Ecosystem services (Y)		Regulating services								Provisioning services												Cultural services					
		1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26
Regulating services	1 Global climate regulation	¤	↗	↗	↗		↗	↗	↗	↘	↗	↘	↘	↘	↗	↗											
	2 Local climate regulation	↗	¤	↗	↗		↗	↗	↗		↗		↘	↘	↗							↗	↗				
	3 Air quality regulation	↗	↗	¤	↗	↗	↗						↘	↘	↗	↘			↗			↗	↗				↗
	4 Water flow regulation	↗	↗	↗	¤			↗	↗						↗						↗						
	5 Water purification					¤	↗	↗		↘	↘	↘	↘	↘	↗	↗	↗	↗	↗		↗	↗	↗				↗
	6 Nutrient regulation				↗	↗	¤	↗	↗	↘	↘	↘	↘	↘	↗	↗	↗		↗	↗	↗	↗	↗				↗
	7 Erosion regulation	↗	↗		↗			¤	↗	↘	↘	↘	↘	↘	↗	↗			↗		↗	↗	↗			↗	↗
	8 Natural hazard protection	↗	↗	↗	↗	↗	↗	↗	¤	↘	↘	↘	↘	↘	↗	↗		↘			↗	↗	↗	↗	↗	↗	↗
Provisioning services	9 Crops					↘	↘	↘		¤	↘	↘	↘	↘	↘	↘	↘	↘	↘	↘	↘		↗	↗		↗	↘
	10 Biomass energy	↗		↘		↘	↘	↘	↗	↘	¤	↘	↘	↘	↘	↘	↘	↘	↘	↘	↘	↘	↘			↘	↘
	11 Fodder	↘				↘	↘	↘		↘	↘	¤	↗	↘	↘	↘	↘	↘	↘	↘	↘					↗	↘
	12 Livestock	↘		↘		↘	↘	↘	↘	↘	↘	↗	¤	↘	↘	↘	↘	↘	↘	↘	↘	↗	↗	↗		↗	↘
	13 Fiber	↘				↘	↘	↘		↘	↘	↘	↘	¤	↘	↘	↘	↘	↘	↘	↘					↗	↘
	14 Timber	↗	↗	↗	↗	↗	↗	↗	↗	↘	↘	↘	↘	↘	¤	↗		↘	↗	↗	↘	↗	↗		↗	↗	↗
	15 Wood fuel	↗	↗	↘	↗	↗	↗	↗	↗	↘	↘	↘	↘	↘	↗	¤		↘	↗	↗	↘	↗	↗				
	16 Fish and seafood					↗	↗	↗									¤		↗			↗				↗	
	17 Aquaculture		↗		↗	↗	↗	↗		↘	↘	↗	↘	↘	↘	↘	↘	¤	↘		↘		↗			↗	
	18 Wild foods									↘	↘	↘	↘	↘	↗	↗			¤			↗				↗	↗
	19 Biochemicals									↘	↘	↘	↘	↘	↗	↗		↗	↗	¤		↗			↗		↗
	20 Freshwater		↗		↗	↗	↗	↗		↘	↘	↘	↘	↘	↗	↗		↘			¤	↗					
Cultural services	21 Recreation & tourism	↘		↗		↗	↗	↗	↗						↗	↗	↗		↗		↗	¤	↗	↗	↘	↗	↗
	22 Landscape aesthetics		↗	↗		↗	↗	↗			↘				↗	↗		↗				↗	¤	↗	↗	↗	↗
	23 Knowledge systems	↗						↗	↗	↗	↗	↗	↗	↗	↗	↗	↗	↗	↗	↗	↗	↗	↗	¤		↗	↗
	24 Religious experience								↗														↗		¤	↗	↗
	25 Cultural heritage		↗					↗	↗	↗	↗	↗	↗	↗	↗	↗	↗	↗	↗	↗	↗	↗	↗	↗	↗	¤	↗
	26 Natural heritage	↗	↗	↗	↗	↗	↗	↗	↗													↗	↗	↗	↗	↗	¤

for the other services (the ‘passive’ services, which are listed at the horizontal x-axis)? The possibilities are a reciprocal competition or exclusion (↘), a mutual support (↗) or no direct reaction at all.
Assigning these conditions in an interrelation matrix, there is an interesting dominance of positive relationships, thus the overall mutual support of different services is higher than mutual exclusion. On the other hand, ecosystem services are negatively correlated to each other in several cases, as well. These conditions can be distinguished for the different ecosystem service classes: Between the regulating services only positive relations can be found. Regarding provisioning services, exclusions and competitions are dominating, and looking at cultural services, the internal budget of this group is highly positive. The overall budget shows similar results: Regulating services provide a high number of supporting functions. The strongest competition is based on provisioning services. Thus, in the middle of the matrix, negative relations are dominating. Also the negative correlations between regulation and provision in the left part of the matrix show conflicts due to the inputs and disturbances of agricultural land use regimes.

These conditions show that there are several negative interrelationships possible between ecosystem services. Another class of “negative” provisions of nature are the so-called disservices. Lyytimäki and Sipilä (2009) or Dunn (2010) show that natural structures and processes also can provide unwelcome pressures on human well-being. Starting with slippery street surfaces due to litter fall, smells originating in decomposition processes, these disservices can end in strongly harming and killing human beings, e.g. by being bitten by venomous snakes, being besieged by flies, mosquitoes, ticks, fleas and bedbugs, eaten up by predators, or by being killed by pathogens. In fact these processes do not include too many benefits for humans.

5 How Can We Quantify Ecosystem Services?

In the tables shown above, several bundles of ecosystem functions which provide societal benefits have been listed as ecosystem services. Layke (2011) characterizes ecosystem service indicators as policy-relevant representations to identify gaps and communicate trends for information on sustainable use of these services and benefits to maintain them for future generations. For a satisfactory utilization of these tools, certain demands have to be fulfilled. These requirements (see Table 7) of course also have to be set for ecosystem service indicators.

Following the concept of TEEB (2010) a utilization of ecosystem service indicators should be proceeded in three steps:

- identification of human activities on ecosystems (“recognizing value”);
- quantification of ecosystem services (“demonstrating value”) by applying ecosystem service indicators;
  - non-monetary physical/biological quantification
  - monetary quantification

Table 7 Demands for indicator development, after Wiggering and Müller (2003)

Scientific demands for indicator selection	*Good indicator sets should provide…..*
	A clear representation of the indicandum by the indicator
	A clear proof of relevant cause – effect relations
	An optimal sensitivity of the representation
	Information for adequate spatio-temporal scales
	A very high transparency of the derivation strategy
	A high degree of validity and representativeness of the data
	A high degree of comparability in and with indicator sets
	An optimal degree of aggregation
	A good fulfilment of statistical requirements
Management related demands for indicator selection	*Good indicator sets should provide…..*
	Information and estimations of the normative loadings
	High political relevance concerning the decision process
	High comprehensibility and public transparency
	Direct relations to management actions
	An orientation towards environmental targets
	A high utility for early warning purposes
	A satisfying measurability
	A high degree of data availability
	Information on long – term trends of development

- integration of results into natural resource management decisions ("capturing value") and alignment with other features, e.g. state indicators.

In this context, the quantification seems to be the most difficult item, while the two other points can be assigned to the general steps of assessments (see Chapter "Marine and Coastal Ecosystems: Delivery of Goods and Services, Through Sustainable Use and Conservation"). Hereby we have to make a distinction between different indicanda: Very often quantifying studies are describing the potential to provide ecosystem services of the investigated ecosystems or landscapes. These values are derived from ecological factors and very often are related to land use structures. On the other hand it is also possible to measure the direct flows of ecosystem services, which are active utilizations of natural capital issues. For example, the potential for crop production of a site can be estimated on the base of agricultural production models, land-based look-up tables, algorithms for production estimation or expert judgements. The respective outcome will work out distinctions of production capacities e.g. based on soil parameters, climate variables, hydrological relations, geomorphological influences or specific demands arising from the crops under investigation. The result does not reflect the actual yield, which can be assigned as a real flow of ecosystem services. Quantifications of flows and potentials can reveal quite different results, thus it should be avoided to mix these types of ecosystem service variables.

Depending on the selected indicator and the respective indicandum different methods are used to characterize the significance of the indicated ecosystem service. These non-monetary approaches can be qualitative or quantitative in character. While the first can be carried our rather easy and fast, the latter always takes a high amount of time and methodological investment. A physical and ecological characterization can be carried out by

- direct measurements of ecosystem service flows, e.g. by agricultural yield analyses;
- indirect application and interpolation of statistical data, e.g. for the assessment of agricultural productions in greater regions;
- application of models (e.g. InVEST, ARIES) or service algorithms to determine ecosystem service provisions;
- use of transfer functions; e.g. using the universal soil loss equation for assessing erosion risks;
- mapping of ecosystem potentials and flows to demonstrate the regional distribution of service characteristics;
- expert judgements and classifications of ecosystem service capacities;

In many cases the information produced by these methods is sufficient to support the environmental assessment of trade-off processes. One possible disadvantage is that different services may be represented by variables with different dimensions, thus the weighting of the significance of the single services in comparison with the others has to be carried out during the assessment. For example it has to be decided if a high crop yield is as important as a high avoidance of erosion events, a high potential for soil carbon sequestration or a high support of landscape beauty.

That step – which might cause long and intensive discussions – can be neglected if it is assumed and accepted that the economic dimension of money is able to represent the extents of these benefits. In many instances, then the respective prices can be defined, observed or derived. Thereafter a monetary comparison of the scenario outcomes can be used as a guideline for ecosystem provisions to support the decision making process. But, this procedure only makes sense if the participants really are convinced that all benefits of nature can be expressed in terms of money. Many critical actors do not do so. They argue that the intrinsic value of nature must take precedence and a price cannot be placed on the priceless.

The economic valuation of ecosystem services can be illuminated from two aspects, the value typology and the valuation methodology. The first attempt is illuminated in Fig. 2. It shows the TEEB concept of a "total economic value" (TEEB 2010), which is subject of exciting discussions in the scientific community. Most of the services which we have listed in Tables 2, 3, and 4 are assigned to the class of "use values" with a distinction between consumptive values (mostly provisioning services), non-consumptive values (mostly deriving from cultural services) and indirect use values, which are in most cases related to regulating services. All of these benefit types can be bundled as they are representing an actual use. On the

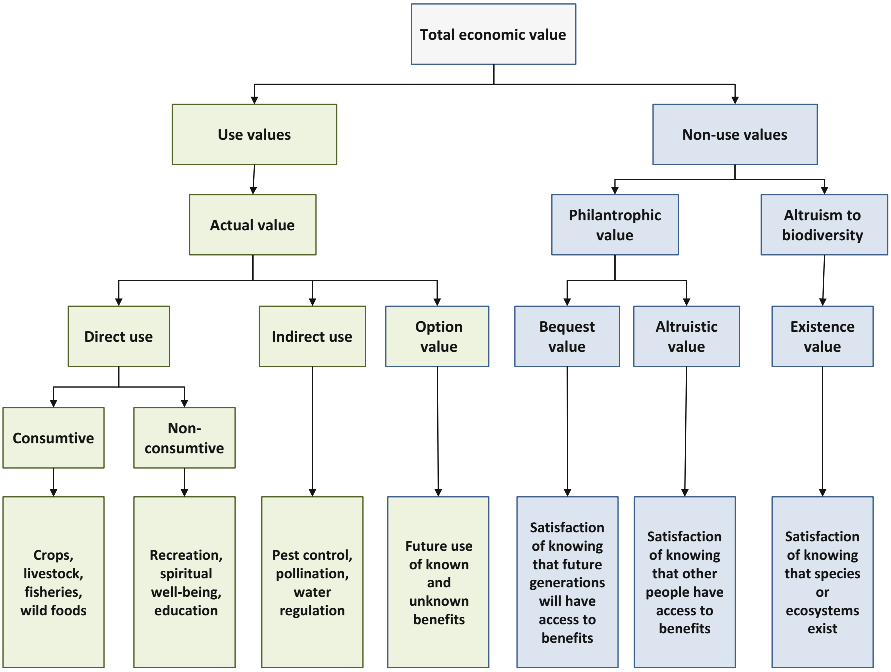

Fig. 2 Types of ecosystem service values referring to the concept of the "total economic value", after TEEB (2010)

other hand, there are also several non-use values that represent the satisfaction of certain groups of persons in knowing that ecological structures, diversities and integrity levels can be sustained, also for future generations. Looking at this value typology critically, it may be asked if all of these value types in fact have the same significance. Some critical voices also say that all the values on the right side of the picture have been added to compensate for the loss of acceptance of the intrinsic values of nature, which have been sufficient for management decisions in the past – before economic aspects have taken over due to the sustainability concept.

A second type of distinction is related to the methodologies that are used for economic valuation (see Fig. 3). The optimal way to determine an economic value can be used for direct market goods which are sold and bought on the real market. Other market goods have to be negotiated as option values, referring to potential future uses of the ecosystem services. The monetary value of indirect market goods can be estimated by the damage costs which really exist after an environmental rare event (e.g. flood damages). Hypothetical indirect monetary values are calculated taking into consideration the costs of a repair, avoidance costs or replacement costs. Non-market goods are much harder to quantify, because the respective public goods are often enjoyed for free by the beneficiaries, while their access can hardly be regulated or controlled. To determine a "shadow price" for these services, two attempts are chosen in ecological economics: The revealed preference method is based on the

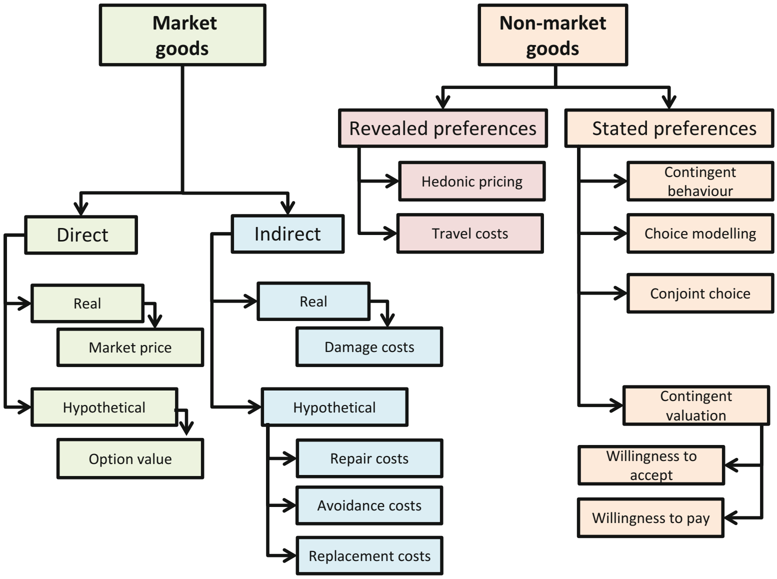

Fig. 3 Some methods to determine the economic value of ecosystem services, after Spangenberg and Settele (2010)

observation of actual consumer or producer behavior, trying to find out how the focal non-market good can influence the market for some other goods (e.g. by estimating travel costs or variations in housing prices that reflect the value of local environmental attributes). On the other hand, the stated preference methods are based on questionnaires and interviews. They try to find out the willingness of people to participate in the benefits of a certain service or make comparisons with other goods to value the significance of the service in question. As these methods are quite complicated, very often the much more simple benefit transfer method is used. It estimates economic values by transferring existing benefit estimates from studies already completed for another location or issue. It is clear that this approach – even if large existing service value data bases are used – is connected with several uncertainties, transformation problems, and thus with several potential mistakes and inaccuracies.

6 How Can Services Be Used in Human-Environmental Assessments?

To apply those quantified services in decision making there are several opportunities to illuminate environmental changes by the potential or realized feedbacks that they have for society. To be efficient, ecosystem services should be incorporated into assessment frameworks, such as the DPSIR approach (Driver – Pressure – State – Impact – Response; see Burkhard and Müller 2008a, b). An application is demonstrated in Fig. 1. In this concept a certain human motivation is understood as a driving force (*driver*) of human actions, e.g. land use activities. Typical direct drivers are the human demand for goods and services, health, social relations, security, education, and freedom, creating certain consumption patterns, life styles and motivations. Indirect drivers are influences or constraints on the direct forces, including e.g. demographical development, economic and social conditions, or political constraints. The resulting inputs into the environmental system provoke a *pressure* on the particular ecosystems and their components, e.g. by release of substances, physical and biological agents, the use of resources and the use of land by human activities. That pressure affects a change of the ecosystem *state,* which can be indicated by quantifiable and qualitative physical, biological and chemical conditions in a defined area. As a consequence, that disturbance may have an *impact* on the ecosystem's performance, human health, ecosystem health, or financial situation. According to the kinds and the degrees of the impacts, decision makers can determine appropriate *responses* to counteract the negative outcomes of pressures.

Within this management cycle, the ecosystem services can be used to quantify the *impacts* on ecological systems, by demonstrating environmental consequences which come up due to changes in the state of the respective ecosystems. That are ecologically based modifications of human well-being criteria which provoke modifications of human welfare variables. Benefits from nature are lost or gained. If these changes succeed a certain threshold they will be perceived, e.g. by the participants of assessments and should lead to respective responses.

7 Recent Problems of Ecosystem Service Assessments

Summarizing this introductory text, we might state that the ecosystem service concept has been developed very fast during the last years, and that besides the interesting basic conceptual ideas, a methodological framework has been created which in fact is ready for utilization and application in environmental management. Ecosystem services can serve as informative indicators, if they are constructed according to the demands from Table 7. But up to now, the ecosystem service framework does not fulfill all of the demands which have been mentioned in this paper with a satisfactory degree (up to now). Going through the list of requirements from the introduction we can state, that ecosystem services in fact are extremely well-suited to demonstrate the enormous human dependence from nature by focusing on the critical roles of ecosystem functions and structures for sustaining human life and well-being. This is the most impressive advantage of the approach, and this fact may provide the most convincing arguments in environmental sustainability assessments. Ecosystem services can allow intersubjective judgements about human actions with respect to natural structures and processes, if the valuation strategy is also based on an intersubjective consensus of the participating stakeholders. In this case it is possible to link scientific state descriptions with management demands and normative items and ecosystem services can be used as a communication tool between different interest groups. Due to their specific layout the service provision may be expressed in different dimensions, e.g. as monetary and non-monetary values, capable of making environmental externalities implicit and demonstrating synergies, trade-offs and conflicts. Ecosystem services – this should not be forgotten – represent the linkages between the three sectors of the sustainability approach, and therefore in practice they should be accompanied by indicators of the sectoral features themselves: It might be dangerous to rely on ecosystem services alone because the aspect of human utility might be overestimated. Much more, arguing from an ecological perspective, objective information on the state of the affected ecosystems – e.g. by using integrity indicators – have to be integrated into participatory trade-offs. Only then the general demands of sustainable development can be better attained by an integration of the ecosystem service concept as a part of the applied ecosystem approach.

In the end of this part of the discussion, one demand has to be mentioned which in fact does provoke some problems: Ecosystem services – that was the central initial requirement – should provide a principle motivation from concerns about an undervaluation of biodiversity. Although the basic concept has been created to demonstrate the performances of biodiversity for human well-being (Costanza et al. 1997; MEA 2005), today there are doubts about the quality and quantity of this relationship. On the other hand, a number of studies have identified ecosystemic key relations which have driven the discussion into a more detailed view: For instance Swift et al. (2004) propose to concentrate on functional groups (e.g. producers, regulators, service providers, ecosystem engineers, keystone species) in agro-ecological investigations instead of structural biodiversity features. Concerning the demanded interrelations, the authors provide an undecided picture: biodiversity can have positive as well as negative influences on ecosystem service provision. As the services often are supported by a small number of key species, in many cases biodiversity does not play a major direct role. In this context, Anton et al. (2010) stress the significance of functional response and effect traits. Similar results have been stated by Díaz et al. (2006): species compositions and the abundance of special organisms are often more important than sole numbers of species. This point has also been exemplified in the recent TEEB report (2010), where the individual performance of different species for several services is exemplified. In the respective article (Elmqvist and Maltby 2010, 55) the recent state of science is characterized as follows: “The above considerations mean that it is not yet possible to account accurately for the role of biodiversity, nor the probable impact of its decline on ecosystem service delivery in general.”

Besides this point, which really needs an intensive scientific clarification, and besides the multitude of uncertainties which are correlated to the ecosystem service approach (and which have to be understood as scientific challenges, see Table 8), there are some conceptual questions which still need a discussion and elaboration in the scientific and environmental community:

Table 8 Different sources of uncertainty in ecosystem service analyses, after Hou et al. (2013)

Uncertainty of systems dynamics	Chaos, catastrophe, and non-linearity can be understood as sources for uncertainties in any complex system
Uncertainty of systems analysis	Indirect, delocalized, or cumulative effects provide substantial difficulties in making prognoses on the dynamics of ecosystems and the service they can provide
Uncertainty of ecosystem dynamics	In assessments, the dynamics of (ideally constant) constraints in context with other scenario assumptions cause uncertainty of developing directions
Uncertainty of modeling methods	In assessments the behavior of ecosystems (and their service related performance) often is modeled; therefore all the uncertainties of modeling are also significant for ecosystem service analyses
Uncertainty of landscape analysis	The investigated landscape analysis can suffer from the implicit problems of heterogeneity, scale mismatches or classification biases
Uncertainty of valuation	Any valuation is correlated with subjectivity, uncertain policy inputs, communication biases, and with the relativeness of monetary values, i.e. in benefit transfer procedures
Uncertainty of natural supply dynamics	In any case we comprise of limited knowledge about the investigated systems and their structural and functional composition and dynamics, even if the data situation is an extraordinary good one
Uncertainty of local demands for services	Demands may change by time but also by population group and societal preferences, and by space. Also the representativeness of inquiries may be limited
Uncertainty of preference settings	Uncertainty arises through the distinct valuation strategies of stakeholders, different responses and biased opinions, uncertain value systems, non-compatible valuation strategies, and communication biases
Uncertainty of indicator identification	Table 7 lists several sources of uncertainty. Nevertheless the biggest insecurity arises from inadequate indicandum – indicator relations
Uncertainty of physical indicator values	All physical, chemical or biological derivation strategies of ecosystem services can be influenced by missing or poor data sources, scaling problems, or measurement failures
Uncertainty of monetary indicator values	The transfer methodology from physical data to societal values is especially problematic if non-market goods have to be expressed in monetary items

Is the basic concept of recent ecosystem service frameworks really structured consistently? After the publications of the Millennium Ecosystem Assessment and the initial service frameworks, several new models have been created to distinguish ecosystem service types and to clarify the question how they are produced. Recently the ecosystem service cascade (Fig. 1) seems to be the dominant approach, but other conceptions are in discussion as well. Therefore, besides the ongoing struggle of approaches, it should be clarified how we can better meet, formulate and calculate the linkages between the components of the cascade, how we can clarify the diffuse pictures at these docking points and how the transfer of dimensions from one category to the next can be better and more objectively managed.

Can we really find clear interrelations between service provision and human well-being? And are these linkages significant for human decision making? It is a basic hypothesis of the ecosystem service approach that reductions of environmental health must cause a loss of ecosystem services. As we could see in Table 6, there are some services which can increase although the integrity of the system is decreasing. Therefore, the overall performance of an ecosystem must be taken into account, a reduction to one or two services cannot help to answer this question. Furthermore, we also have to look at the dynamics of the needs of our society and ask if for instance the usage of a highly elaborated electronic device provides more "services" than a walk through a forest. Thus, especially the linkage between service provision and human well-being should be investigated with a higher emphasis. The factual dependence of human welfare from natural integrity has to be put on a stronger basis of evidence, also in the details.

How can we cope with the large regional differences of ecosystem service valuation? Besides an enormous bundle of methodological questions, the regional distinctions of ecosystem service supply and demand display an interesting ground of heterogeneity. For example the value of a cow will be extremely different if a person is asked in India or Bavaria. Consequently, all the values we are interpreting in monetary terms will change, not only by region but also as a consequence of the time, Zeitgeist or social position of the surveyed persons. These differences can provide extremely interesting research questions.

Can we really compare (and add) the different monetary service values if they have been quantified on the base of different methodologies? Figure 3 shows the different methods of monetary valuation of ecosystem services. Now we have seen before, that it is necessary to work with service bundles in assessment studies, because otherwise the trade-offs cannot be made transparent. For each service type, another method might be optimal, the market price for provisioning services, repair costs for erosion protection, avoidance costs for nutrient retention, travel costs for landscape beauty and willingness to pay for a higher carbon sequestration. All of these specific methods produce specific results. And the question is if they really can be added or be combined with benefit transfer procedures to calculate a total value? In this field comparative studies might be helpful to give us a better fundament for methodological comparison.

Are ecosystem services sufficient to represent the ecological items throughout assessment? If we only concentrate on the impact side of environmental changes, we concentrate on the linkages between the sustainability sectors alone. A satisfactory information basis must include more facts, e.g. on the changes of environmental, social and economic state variables themselves. This demand is existential because high monetary service values can be related to decreases of the ecological integrity. Therefore, it is important to enlarge the dimension of our conceptual frameworks and to give the services an important position in the DPSIR management cycles, as one focal part of an overall assessment system.

Are we in danger of losing the intrinsic values of nature? This final question is the starting point of many critical remarks concerning the ecosystem service concept. It is a totally anthropocentric approach, and nature is reduced to a utility function for human life support. The value of nature in itself, which is the basic item of many people who are engaged in environmental protection, is not considered in the physical or monetary service calculations. These critical voices should be recognized and we should look for a common concept which is capable of including both approaches, maybe in correlation with nature conservation initiatives or legacies. On the other hand, we also have to be aware that species protection in many instances has to be linked with "unnatural" management measures, stopping successions or creating isolated (artificial) site conditions for the sake of rare species. Finally, of course also the establishment of protected areas can only work if there is a societal consensus on doing so. Thus, the costs of preservation areas and the willingness to abstain from the other potential services of such regions, can be seen as the result of a trade-off, whereby a small amount of overall service provision and a small amount of monetary values are consciously renounced due to a better state of biodiversity.

References

Anton C, Young J, Harrison PA, Musche M, Bela G, Feld CK, Harrington R, Haslett JR, Pataki G, Rounsevell MDA, Skourtus M, Sousa JP, Sykes MT, Tinch R, Vanderwalle M, Watt A, Settele J (2010) Research needs for incorporating the ecosystem service approach into EU biodiversity conservation policy. Biodivers Conserv 19:2979–2994

Bouwes N, Schneider R (1979) Procedures in estimating benefits of water quality change. Am J Agric Econ 61:535–539

Boyd J, Banzhaf S (2007) What are ecosystem services? The need for standardized environmental accounting units. Ecol Econ 63:616–626

Burkhard B, Müller F (2008a) Indicating human-environmental system properties: case study Northern Fenno-Scandinavian Reindeer Herding. Ecol Indic 8:828–840

Burkhard B, Müller F (2008b) Drivers-pressure-state-impact-response. In: Joergensen SE, Fath BD (eds) Ecol indicators, vol [2] of Encyclopedia of ecology, 5 vols. Elsevier, Oxford, pp 967–970

Burkhard B, Kroll F, Müller F, Windhorst W (2009) Landscapes' capacities to provide ecosystem services – a concept for land-cover based assessments. Landsc Online 15:1–22

Burkhard B, de Groot R, Costanza R, Seppelt R, Jørgensen SE, Potschin M (2012) Solutions for sustaining natural capital and ecosystem services. Ecol Indic 21:1–6

CBD [Convention on Biological Diversity] (2000) Ecosystem approach: further conceptual elaboration. SBSTTA 5/11, Montreal. http://www.cbd.int/ecosystem/documents.shtml

Chetverikova N (2012) Evaluation of the ecosystem services based on the SWAT model – case study Kielstau catchment. MS thesis, University of Kiel, Institute of Natural resource Conservation

Chícharo L, Chícharo MA, Ben-Hamadou R (2006) Use of a hydrotechnical infrastructure (Alqueva Dam) to regulate planktonic assemblages in the Guadiana estuary: basis for sustainable water and ecosystem services management. Estuar Coast Shelf Sci 70(1–2):3–18

Costanza R (2008) Ecosystem services: multiple classification systems are needed. Biol Conserv 141:350–352

Costanza R, d'Arge R, de Groot RS, Farber S, Grasso M, Hannon B, Limburg K, Naeem S, O'Neill RV, Paruelo J, Raskin RG, Sutton P, van den Belt M (1997) The value of the world's ecosystem services and natural capital. Nature 387:253–260

Daily GC (ed) (1997) Nature's services: societal dependence on natural ecosystems. Island Press, Washington, DC

Daily GC, Polasky S, Goldstein J, Kareiva PM, Mooney HA, Pejchar L, Ricketts TH, Salzman J, Shallenberger R (2009) Ecosystem services in decision making: time to deliver. Front Ecol Environ 7(1):21–28
Dale VH, Polasky S (2007) Measures of the effects of agricultural practices on ecosystem services. Ecol Econ 6(4):286–296
De Groot RS, Wilson MA, Boumans RM (2002) A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecol Econ 41:393–408
de Groot RS, Fisher B, Christie M, Aronson J, Braat L, Haines-Young R, Gowdy J, Maltby E, Neuville A, Polasky S, Portela R, Ring I (2010a) Chapter 1: Integrating the ecological and economic dimensions in biodiversity and ecosystem service valuation. In: Kumar P (ed) The Economics of Ecosystems and Biodiversity (TEEB): ecological and economic foundations. Earthscan, London, pp 4–9
de Groot RS, Alkemade R, Braat L, Hein L, Willemen L (2010b) Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecol Complex 7:260–272
Díaz S, Fargione J, Chapin FS III, Tilman D (2006) Biodiversity loss threatens human well-being. PLoS Biol 4(8):e277, http://dx.doi.org/10.1371/journal.pbio.0040277 (online)
Dong Z, Yan Y, Duan J, Fu X, Zhou Q (2011) Computing payment for ecosystem services in watersheds: an analysis of the Middle Route Project of South-to-North Water Diversion in China. J Environ Sci 23(12):2005–2012
Doyle MW, Yates AJ (2010) Stream ecosystem service markets under no-net-loss regulation. Ecol Econ 69:820–827
Duffield JW, Neher CJ, Brow TC (1992) Recreation benefits of instream flow: application to Montana’s Big Hole and Bitterroot Rivers. Water Resour Res 29:2169–2181
Dunn R (2010) Global mapping of ecosystem disservices: the unspoken reality that nature sometimes kills us. Biotropica 42(5):555–557
Elmqvist T, Maltby E (2010) Biodiversity, ecosystems and ecosystem services. In: Kumar P (ed) The economics of ecosystems and biodiversity. Earthscan, Washington, DC, pp 41–111
European Commission (2006) Handbook for trade sustainability impact assessment. Brussels
Fisher B, Turner K (2008) Ecosystem services: classification for valuation. Biol Conserv 141:1167–1169
Fu B, Liu Y, Lu Y, Chansheng H, Zeng Y, Bingfang W (2011) Assessing the soil erosion control service of ecosystems change in the Loess Plateau of China. Ecol Complex 8:284–293
Gordon LJ, Finlayson CM, Falkenmark M (2010) Managing water in agriculture for food production and other ecosystem services. Agric Water Manag 97:512–519
Grunewald K, Bastian O (eds) (2015) Ecosystem services – Concept, methods and case studies. Springer, Heidelberg
Gunderson LH, Holling CS (2003) Panarchy. Island Press, Washington, DC
Haines-Young R, Potschin M (2009) Methodologies for defining and assessing ecosystem services. CEM Report 14, University of Nottingham
Haines-Young RH, Potschin M (2010a) Proposal for a common international classification of ecosystem goods and services (CICES) for integrated environmental and economic accounting. European Environment Agency
Haines-Young R, Potschin M (2010b) The links between biodiversity, ecosystem services and human well-being. In: Raffaelli DG, Frid CLJ (eds) Ecosystem ecology: a new synthesis. Cambridge University Press/British Ecological Society, Cambridge, pp 110–139
Haines-Young R, Potschin M (2013) Common International Classification of Ecosystem Services (CICES): consultation on Version 4, August-December 2012. EEA framework contract no EEA/IEA/09/003
Helming K, Soba MP, Tabbush P (eds) (2008) Sustainability impact assessment of land use changes. Springer, Berlin/Heidelberg
Hoeinghaus DJ, Agostinho AA, Gomez LC, Pelicice FM, Okada EK, Latini JD, Kashiwaqui E, Winemiller KO (2009) Effects of river impoundment on ecosystem services of large tropical

rivers: embodied energy and market value of artisanal fisheries. Conserv Biol 23(5):1222–1231
Holmlund CM, Hammer M (1999) Ecosystem services generated by fish populations. Ecol Econ 29:253–268
Hou Y, Burkhard B, Müller F (2013) Uncertainties in landscape analysis and ecosystem service assessment. J Environ Manage, doi.org/10.1016/j.jenvman.2012.12.002
Jax K, Barton DN, Chan KMA, de Groot R, Doyle U, Eser U, Görg C, Gómez-Baggethun E, Griewald Y, Haber W, Haines-Young R, Heink U, Jahn T, Joosten H, Kerschbaumer L, Korn H, Luck GW, Matzdorf B, Muraca B, Neßhöver C, Norton B, Ott K, Potschin M, Rauschmayer F, von Haaren C, Wichmann S (2013) Ecosystem services and ethics. Ecol Econ 93:260–268
Jewitt G (2002) Can integrated water resources management sustain the provision of ecosystem goods and services? Phys Chem Earth 27:887–895
Johnson NL, Baltodano ME (2004) The economics of community watershed management: some evidence from Nicaragua. Ecol Econ 49:57–71
Johnston RJ, Segerson K, Schultz ET, Besedin EY, Ramachandran M (2011) Indices of biotic integrity in stated preference valuation of aquatic ecosystem services. Ecol Econ 70:1946–1956
Jorgensen SE, Xu L, Costanza R (eds) (2010) Handbook of ecological indicators for assessment of ecosystem health, 2nd edn. Taylor & Francis, Baco Raton
Kandziora M, Burkhard B, Müller F (2012) Interactions of ecosystem properties, ecosystem integrity and ecosystem service indicators – a theoretical matrix exercise. Ecol Indic 28:54–78
Layke C (2011) Indicators from the global and sub-global Millennium Ecosystem Assessments: an analysis and next steps. Ecol Indic, http://dx.doi.org/10.1016/j.ecolind.2011.04.025
Lele S (2009) Watershed services of tropical forests: from hydrology to economic valuation to integrated analysis. Curr Opin Environ Sustain 1:148–155
Loomis J, Kent P, Strange L, Fausch K, Covich A (2000) Measuring the total economic value of restoring ecosystem services in an impaired river basin: results from a contingent valuation survey. Ecol Econ 33:103–117
Lyytimäki J, Sipilä M (2009) Hopping on one leg – the challenge of ecosystem disservices for urban green management. Urban For Urban Green 8(4):309–315
MEA [Millennium Ecosystem Assessment] (2005) Ecosystems and human well-being: synthesis. Island Press, Washington, DC
Mander U, Wiggering H, Helming K (eds) (2010) Multifunctional land use – meeting future demands for landscape goods and services. Springer, Berlin/Heidelberg
Morais P, Chicharo MA, Chicharo L (2009) Changes in a temperate estuary during the filling of the biggest European dam. Sci Total Environ 407:2245–2259
Morais P, Babaluk J, Correia AT, Chícharo MA, Campbell JL, Chícharo L (2010) Diversity of anchovy migration patterns in a European temperate estuary and in its adjacent coastal area: implications for fishery management. J Sea Res. doi:10.1016/j.seares.2010.04.001
Morais P, Martins F, Chícharo MA, Lopes J, Chícharo L (2012) Merging anchovy eggs abundance into a hydrodynamic model as an assessment tool for estuarine ecohydrological management. River Res Appl 28:160–176. doi:10.1002/rra.1443
Müller F (2005) Indicating ecosystem and landscape organization. Ecol Indic 5(4):280–294
Müller F, Burkhard B (2007) An ecosystem based framework to link landscape structures, functions and services. In: Mander Ü, Wiggering H, Helming K (eds) Multifunctional land use – meeting future demands for landscape goods and services. Springer, Berlin/Heidelberg, pp 37–64
Müller F, Burkhard B, Kandziora M, Schimming C, Windhorst W (2012) Ecological indicators: ecosystem health. In: Jørgensen SE (ed) Encyclopedia of environmental management, vol 1. CRC Press, Bacon Rouge, pp 599–613
Nedkov S, Burkhard B (2012) Flood regulating ecosystem services – mapping supply and demand, in the Etropole municipality Bulgaria. Ecol Indic 21:67–79
OECD (2010) Guidance on sustainability impact assessment. OECD, Paris
Ojea E, Martin-Ortega J, Chiabai A (2012) Defining and classifying ecosystem services for economic valuation: the case of forest water services. Environ Sci Pol 19–20:1–15

Pattanayak SK (2004) Valuing watershed services: concepts and empirics from southeast Asia. Agric Ecosyst Environ 104:171–184

Pert PL, Butler JR, Brodie JE, Bruce C, Honzak M, Kroon FJ, Metcalfe D, Mitchell D, Wong G (2010) A catchment-based approach to mapping hydrological ecosystem services using riparian habitat: a case study from the Wet Tropics, Australia. Ecol Complex 7:378–388

Posthumus H, Rouquette JR, Morris J, Gowing DJ (2010) A framework for the assessment of ecosystem goods and services; a case study on lowland floodplains in England. Ecol Econ 69:1510–1523

Postle SL, Barton H, Thompson Jr (2005) Watershed protection: capturing the benefits of nature's water supply services. Nat Res Forum 29:98–108

Sanders LD, Walsh RG, McKean JR (1991) Comparable estimates of the recreational value of rivers. Water Resour Res 27(7):1387–1394

Smith M, de Groot D, Perrot-Maîte D, Bergkamp G, (2006) Pay – establishing payments for watershed services. IUCN, Gland, Switzerland. Reprint, Gland, Switzerland: IUCN, 2008

Spangenberg JH, Settele J (2010) Precisely incorrect? Monetizing the value of ecosystem services. Ecol Complex 7(3):327–337

Staub C, Ott W et al (2011) Indicators for ecosystem goods and services: framework, methodology and recommendations for a welfare-related environmental reporting, Environmental studies no. 1102. Federal Office for the Environment, Bern, p 17 S

Swift MJ, Izac A-MN, van Noordwijk M (2004) Biodiversity and ecosystem services in agricultural landscapes – are we asking the right questions? Agric Ecosyst Environ 104(1):113–134

TEEB (2010) The economics of ecosystems and biodiversity: ecological and economic foundations. Earthscan, London

Townsend PV, Harper RJ, Brennan PD, Dean C, Wu S, Smettem KR, Cook SE (2012) Multiple environmental services as an opportunity for watershed restoration. For Policy Econ 17:45–58

United Nations General Assembly (1987) Report of the World Commission on Environment and Development: our common future. New York

Veloz A, Southgate D, Hitzhusen F, Macgregor R (1985) The economics of erosion control in a subtropical watershed: a Dominican case. Land Econ 61(2):145–155

Verburg PH, Schulp CJE, Witte N, Veldkamp A (2006) Downscaling of land use change scenarios to assess the dynamics of European landscapes. Agric Ecosyst Environ 114(1):39–56

Wallace KJ (2007) Classification of ecosystem services: problems and solutions. Biol Conserv 139:235–246

Wallace K (2008) Ecosystem services: multiple classifications or confusion? Biol Conserv 141:353–354

Wallis C, Séon-Massin N, Martini F, Schouppe M (2011) Onema meeting: implementation of the Water Framework Directive – when ecosystem services come into play. Brussels. ISBN: 979-10-91047-06-7

Wang G, Fang Q, Zhang L, Chen W, Chen Z, Hong H (2010) Valuing the effects of hydropower development on watershed ecosystem services: case studies in the Jiulong River Watershed, Fujian Province, China. Estuar Coast Shelf Sci 86:363–368

Wiggering H, Müller F (eds) (2003) Umweltziele und Indikatoren. Springer, Berlin, Heidelberg/ New York, 670 pp

Willaarts BA, Volk M, Aguilera PA (2012) Assessing the ecosystem services supplied by freshwater flows in Mediterranean agroecosystems. Agric Water Manag 105:21–31

Wilson MA, Carpenter SR (1999) Economic valuation of freshwater ecosystem services in the United States: 1971–1997. Ecol Appl 9(3):772–783

Wolanski E, Chicharo L, Chicharo M, Morais P (2006) An ecohydrology model of the Guadiana Estuary (South Portugal). Estuar Coast Shelf Sci 70(1–2):85–97

Xiang H, Yahing C, Jianxin M, Yapeng C (2010) Study on change in value of ecosystem service function of Tarim River. Acta Ecol Sin 30:67–75

Cultural Services in Aquatic Ecosystems

João Manuel Garcia Rodrigues

Abstract Many people seek and interact with aquatic ecosystems such as seas, rivers and wetlands to obtain non-material benefits provided by cultural ecosystem services. These services influence the way people live and feel in the world and contribute to the satisfaction of fundamental human needs. However, cultural ecosystem services are often undervalued, underprotected, and neglected from ecosystem services studies. This arises from difficulties in their operation such as uncertainties on their generation and on people's demand for cultural ecosystem services.

This chapter provides an overview of cultural services generated by aquatic ecosystems. It gives insights into their biophysical generation and it explores the relationships between human needs and ecosystem service demand. Furthermore, it illustrates the values of cultural ecosystem services with a case study, and it proposes a driver-pressure-state-impact-response (DPSIR) framework as a management tool for decision-makers.

These topics are fundamental to apply better strategies that can effectively protect and conserve aquatic ecosystems and their cultural service provision.

Keywords Ecosystem services • Environmental valuation • Ecohydrologic integrity • Human needs

1 Introduction

People interact with nature in a myriad of different ways, such as swimming in a sea, worshiping a river, or doing field work in a wetland. While doing so, people obtain non-material benefits from ecosystems. These benefits are a result of the interactions between people and nature and are provided by what was coined as *cultural ecosystem services* (MA 2003). Chan et al. (2011, p. 206) defined

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these services as the "ecosystems' contribution to the non material benefits (e.g., experiences, capabilities) that people derive from human-ecological relations". Cultural services are a category included in the broader ecosystem services concept. The other categories are provisioning services (e.g., drinking water, fish) and regulating services (e.g., carbon sequestration, water purification). *Ecohydrologic integrity* (formerly designated as supporting services by the MA 2003) constitutes the foundation of all services generated by aquatic ecosystems. Ecosystem services are the direct and indirect contributions of ecosystems to human wellbeing (de Groot et al. 2010) and the concept was developed to highlight the importance of ecosystems to human livelihoods (Ehrlich and Mooney 1983; Daily 1997; Costanza et al. 1997; MA 2003).

Several classifications exist for aquatic ecosystem services (Postel and Carpenter 1997; Brauman et al. 2007). Although these are valid classifications, this chapter will use the division of ecosystem services into provisioning, regulating and cultural services, as it can be applied in any type of ecosystem and because it is the most used classification in the literature (e.g., MA 2005; Costanza et al. 1997; Daily 1997). Table 1 describes the broad categories within the culture ecosystem services classification. For indicators that quantify each service see Kandziora et al. (2013).

Each of the main freshwater ecosystems, i.e., rivers and streams, lakes and inland waters, and wetlands, provide innumerable non-material benefits to people through cultural ecosystem services (Aylward et al. 2005; Maltby et al. 2011). River and stream networks, and their connecting groundwaters, are the sinks into which landscapes in a river basin drain. These systems provide many cultural services, as they are part of religious cultures (e.g., Christians worship the River Jordan and Hindus the Ganges), cultural heritage landscapes (e.g., the River Douro in Portugal), and places for recreation (e.g., swimming, rafting, sport fishing). Wetlands are diverse

Table 1 Classification of cultural ecosystem services and respective description (After MA 2005)

Categories	Description
Spiritual and religious	People seek a great variety of ecosystems for spiritual fulfilment. Religions consider many ecosystems and their features sacred places and/or entities
Educational	Ecosystems provide the basis of both formal and informal education in many societies
Inspirational	Many people are artistically and spiritually inspired by ecosystems
Aesthetic	Many people appreciate the beauty of ecosystems and choose the place to live, travel or spend time accordingly
Recreation and tourism	The natural features of a place are many times the main reason for travelling or for leisure time
Societal	Ecosystems shape the social relations in many societies, such as fishing, nomadic or agricultural ones
Cultural heritage	Landscapes and ecosystems provide "memories" of past cultural bonds, expressed through objects, places, practices and cultural behaviours
Sense of place	Many people have emotional bonds and feel attached to a landscape or ecosystem

ecosystems that vary according to a range of hydrological, ecological, geomorphological and economic characteristics (Davis 1994). They are a source of cultural services, such as aesthetic services and inspirational services, as can be appreciated in many paintings; spiritual services (e.g., the mauri for the Maoris); and tourism destinations (e.g., Everglades, Pantanal, and Coto Doñana). Permanent lakes and other inland waters normally occur where precipitation is abundant and where geology is suitable for water-retaining basins. Lakes provide cultural services, namely educational ones, because they are good systems for ecological study as they have distinct boundaries, the water is well mixed, and the bottom relatively homogeneous, making them tractable systems for ecologists (Dodds 2002). Many people seek lakes for recreation activities such as boating, swimming, or bird-watching. Marine and coastal areas are vast and diverse, and include terrestrial ecosystems (e.g., sand dunes, cliffs), areas of fresh and saltwater mixing (e.g., estuaries), near shore coastal areas (e.g., intertidal zones, coral reefs), and open ocean (e.g., seamounts, abyssal plains) (UNEP 2006). Marine ecosystems provide many cultural ecosystem services such as educational services (e.g., technological and medical research), societal services (e.g., coastal areas provide conditions for the establishment of fishing communities) and opportunities for recreation and tourism (e.g., surfing, recreational diving, whale-watching).

Although aquatic ecosystems are essential for human populations, their overall condition is degrading due to human activities (Halpern et al. 2008; Lotze et al. 2006). The state of marine and coastal ecosystems is rapidly deteriorating as a consequence of exploitation, pollution, habitat destruction, climate change and related perturbations of ocean biogeochemistry (Worm et al. 2006). Freshwater ecosystems face similar threats. Population growth and socio-economic development have been occurring at the expense of freshwater ecosystem's exploitation worldwide. This includes diverting water for agricultural, industrial, and domestic needs, and by physically modifying water systems for hydropower generation, flood control, and navigation (Aylward et al. 2005). These perturbations contribute to biodiversity loss and have serious implications for ecosystem service provision because biodiversity is positively related with ecosystem functions and services (Worm et al. 2006). The ecosystem service concept has the potential to tackle anthropogenic threats because it recognises the dependencies people have towards ecosystems, through the benefits derived from them. It is a powerful tool to acknowledge the need for conserving and protecting ecosystems and their services (Brauman et al. 2007).

Ecosystem service research is receiving considerable attention nowadays. However, cultural ecosystem services are clearly underrepresented in the scientific literature (Schaich et al. 2010). This stems from difficulties in their operation, such as absence of exact criteria for quantitative and qualitative assessments, arising from uncertainties about their generation, and issues in understanding the dependencies people have towards cultural services, and how these dependencies are related to fundamental human needs. Following these challenges, this chapter aims at (1) unveiling ecohydrological processes and functions of aquatic ecosystems that supply cultural services, (2) understanding the relationship between human needs and the demand for cultural services, (3) proposing valuation approaches, (4) and suggesting a DPSIR framework as a management tool for decision-makers in human-environmental systems.

2 The Supply of Cultural Services by Aquatic Ecosystems

As with provisioning and regulating services, cultural ecosystem services are ultimately a result of the interactions between ecosystem structures, processes, and functions. The interactions between structures and processes can be physical (e.g., infiltration of water, sediment movement), chemical (e.g., reduction, oxidation), and biological (e.g., photosynthesis, transpiration) (de Groot et al. 2010). These interactions control the flow of energy, matter, and water within and between ecosystems (Müller 2005; de Groot et al. 2002). The outcomes of these interactions are ecosystem functions, which in this context designate a particular capability of an

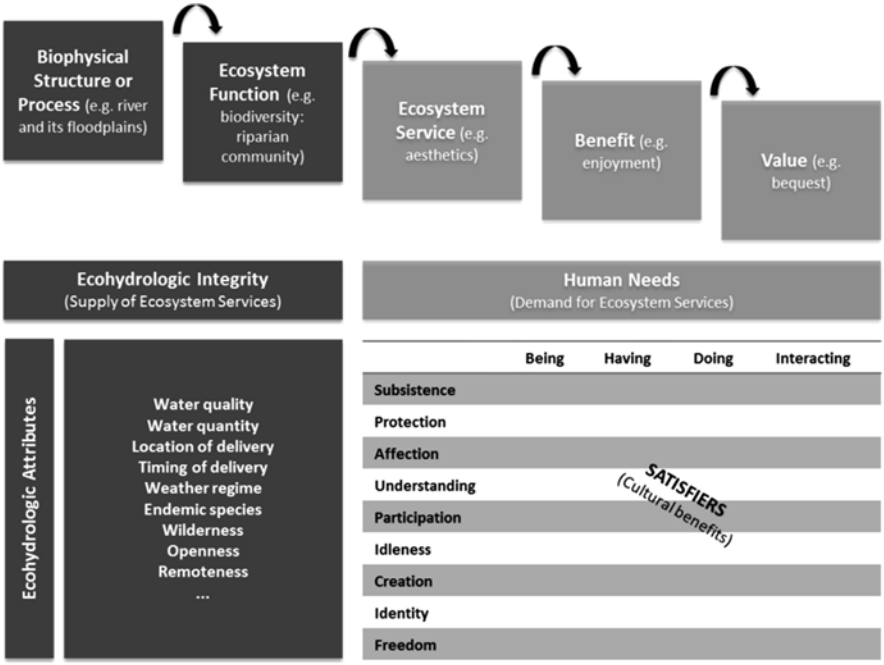

Fig. 1 Conceptual model of the linkages between the "ecosystem service cascade" (After Haines-Young and Potschin 2010), ecohydrologic integrity and attributes, and human needs. Biophysical structures or processes and ecosystem functions supply services that generate benefits to people. The supply of services by aquatic ecosystems depends on their ecohydrologic integrity. Each cultural service has its specific ecohydrologic attributes (Brauman et al. 2007) that influence people's preferences towards them. The cultural benefits people derive from cultural services are satisfiers of fundamental human needs, which is the reason why people demand and depend on ecosystem services. The Human Scale Development matrix (*bottom right side*), developed by Max-Neef et al. (1986, 1989), is created to reveal individual and collective satisfiers of human needs

ecosystem to generate an outcome that is beneficial to people. Ecosystem functions are the suppliers of ecosystem services.

The term "ecosystem service cascades" (Haines-Young and Potschin 2010) was coined to highlight the role ecosystem structures, processes, and functions have in the delivery of services (Fig. 1). People value ecosystem services because they derive benefits from them. To illustrate the generation of services through this "cascade", one can think of a river and its floodplains (biophysical structure), which might provide suitable conditions for a riparian community (ecosystem function) to thrive. People might find this riparian community aesthetically attractive (cultural ecosystem service, i.e., aesthetic services) and enjoy to observe it (benefit), thus wanting to preserve it for future generations (bequest value).

To acknowledge the main ecosystem functions and the magnitude of service provision necessary to generate cultural and other ecosystem services, it is fundamental to characterise the integrity of the system. Ecosystem integrity represents the support and preservation of those processes and structures which are essential prerequisites of the ecological ability for self-organisation (Barkmann et al. 2001). The ecosystem integrity concept was developed having terrestrial ecosystems in mind. Although many aquatic ecosystems share common features with terrestrial ones, they generally have processes and functions that are not major actors in strictly terrestrial ecosystems. For this reason, and because it is suitable for highlighting the role hydrological processes and functions play in aquatic ecosystems integrity, the term *ecohydrologic integrity* is here introduced.

The ecohydrologic integrity indicandum, depicted in Table 2, is similar to the ecosystem integrity indicandum developed by Müller et al. (2000), Müller (2005) and Kandziora et al. (2013), with the exception of *local/global climate* and *natural flow regime/environmental flow*. These were added to better address the fundamental hydrological processes acting on aquatic ecosystems. Potential indicators for each constituent of the indicandum are also displayed.

The *natural flow regime*, with its five key components of variability, magnitude, frequency, duration, timing and rate of change, is a fundamental process in sustaining biodiversity and ecohydrologic integrity in freshwater systems (Poff and Ward 1989; Karr 1991; Richter et al. 1997; Tharme 2003). Therefore, the natural flow regime of freshwater ecosystems is fundamental to generate cultural and other ecosystem services and hence, included in the ecohydrologic integrity indicandum. However, the natural hydrological regime of water resources have been drastically altered globally by the construction of impoundments, diversion weirs, interbasin water transfers, run-of-river abstraction and exploitation of aquifers for irrigated agriculture, hydropower generation, industry and domestic supply (Rosenberg et al. 2000; Tharme 2003). As a result, assessments had to be developed to know how much of the original (natural) flow regime of a watercourse would have to continue flowing to maintain the functioning of freshwater ecosystems. These are *environmental flows*, which are artificially controlled flow regimes. When an ecosystem has altered flow regimes, environmental flow should be used in the indicandum, instead of natural flow regime.

The other proposed addition to the indicandum is *local/global climate*. As ecosystems are thermodynamically open systems that exchange energy, matter, and

Table 2 Ecohydrologic integrity indicandum, description of each constituent and respective potential indicators (After Müller et al. 2000; Müller 2005; Kandziora et al. 2013; Brauman et al. 2007)

Ecohydrologic integrity indicandum	Description	Potential Indicators
Heterogeneity	The aptitude of an ecosystem to create diverse and different habitats for species and for processes	Substrate, e.g., particle size (mm)
Biodiversity	The presence and absence of species, functional groups, biotic habitat components or species compositions	Shannon-Wiener Index
Biotic water flows	The water cycling in the ecosystem affected by biotic processes	Transpiration/total evapotranspiration
Metabolic efficiency	The least amount of energy needed to maintain a specific amount of biomass	Respiration/biomass
Exergy capture	The capacity of an ecosystem to produce biomass and structures by using the energy fraction that can be converted in mechanical work, i.e., exergy	Net primary production (t C/ha*a; Kj/ha*a)
Entropy production	Non-convertible energy fractions which are exported into the environment of the system	Respiration (C/year)
Matter cycling	The capacity of an ecosystem to reuse internal matter and to prevent it from leaving the system	Leaching of nutrients, e.g., N (mg/l)
Storage capacity	The aptitude of an ecosystem to store matter, energy, and water when available and to release them when needed	C in biomass (kg/t/a)
Local/global climate	The water cycle in the ecosystem affected by climate conditions such as humidity, precipitation, atmospheric pressure, temperature, wind, etc.	Precipitation, e.g., rain (mm)
Natural flow regime/ environmental flow	The presence or absence of the required quantity, quality, timing and location of water flows for ecosystem functioning	Flow, e.g., mean flow (m^3/s)

water with the exterior (Jørgensen 1992; Müller 2005), some processes are part or a result of other interacting ecosystem processes (e.g., local climate), or external to the ecosystem's boundaries (e.g., global climate), depending on the spatial scale considered. Climatic processes exert enormous influence on aquatic ecosystems. For example, in rivers the local climate affects the amount of water flowing in a watercourse, its seasonality, and the location where water flows. The oceans are more susceptible to the global climate, which influences surface water currents, the thermohaline circulation, downwelling and upwelling movements, among many others. Therefore, global and local climate conditions are added to the ecohydrologic integrity indicandum to take into account the effects they have on the system's hydrologic regime.

Other approaches exist to assess the provision of cultural ecosystem services, such as quantifying the amount of service benefit, or the magnitude of ecohydrologic attributes of the system (Brauman et al. 2007). Each cultural service has its own ecohydrologic attributes (Fig. 1). For example, the recreation possibilities on a water body depend on the ecohydrologic attributes of quantity, quality, location and timing of water flow. In the same way, inspirational services might be only possible according to the remoteness and wilderness of an aquatic setting.

Although the biophysical and abiotic components of ecosystems are fundamental aspects in the generation of cultural ecosystem services, these services are highly contextual, subjective and observer dependent. In the same natural setting one person might feel inspired and other not, one might feel spiritually attached and other not. Individual and collective cultural experiences, habitats and belief systems, traditions of behaviour and judgment, and life styles are important aspects to consider when dealing with cultural ecosystem services (Kumar and Kumar 2008; Gee and Burkhard 2010). Assessing people's perceptions, preferences, and the amount of service benefits derived, through qualitative and quantitative data obtained from questionnaires and social surveys, is a useful approach to understand cultural ecosystem service provision. Furthermore, it is fundamental to understand the demand for these services.

3 Links Between Human Needs and the Demand for Cultural Services

The ecosystem service concept emphasises the role ecosystems play in the generation of benefits for human wellbeing (Costanza et al. 1997; Daily 1997; MA 2003). However, there are ambiguities in the concept of human wellbeing and in what are its constituents. The MA (2005) proposes several constituents for human wellbeing: basic material for a good life, health, good social relations, security, and freedom of choice and action. Kandziora et al. (2013) divide the constituents of human wellbeing into three categories: economic wellbeing (income, employment, housing, infrastructure, security), social wellbeing (nutrition, demography, health, education, leisure, social relations), and personal wellbeing (subjective category that integrates all other indicators). All these classifications are valid and characterise, with more or less extent, human wellbeing and its constituents. However, the constituents of these classifications are only "means", rather than "ends". They do not clearly acknowledge the "ends" that are the universal needs common to all humans, and what are the basic requirements to fulfil them.

Manfred Max-Neef (1986, 1989) proposed a concept where the satisfaction of fundamental human needs is in the centre of human development. It is a holistic approach that proposes nine classes of human needs (axiological needs, i.e., the things we value) and four forms of expression of those needs (existential needs, i.e., the way needs are manifested), which combined constitute the Human Scale Development matrix (Fig. 1). The axiological needs are *subsistence, protection, affection, understanding, participation, creation, idleness, identity,* and *freedom*. The types of expression of each need are *being* (personal or collective attributes), *having* (institutions, tools, norms), *doing* (personal or collective actions) and *interacting* (settings in time and space). Max-Neef (1992) points out that human needs must be viewed as a system where they are interrelated and interactive, without hierarchies, with the exception for the need of subsistence (to remain alive). The author postulates that fundamental human needs are finite, few, and classifiable, and that they are the same in all cultures and in all historical periods, changing only over time and among cultures, the means by which the needs are satisfied, i.e., the satisfiers (for a satisfier typology see Max-Neef 1992 and Church et al. 2011). Following the same line of thought, Wallace (2007) suggests that the constituents of a reasonable quality of life are probably consistent across cultures, but the weighting, specification and means of achieving those constituents vary among cultures and among individuals. Needs are satisfied at the personal, social, and environmental levels, which are contextual upon time, place, and local circumstances (Max-Neef 1992). Furthermore, Max-Neef (1992, p. 199) states that there is no direct correspondence between needs and satisfiers because "a satisfier may contribute simultaneously to the satisfaction of different needs, or conversely, a need may require various satisfiers in order to be met." People and societies have the same needs but different forms of *being, having, doing and interacting*. The satisfaction of the individual and collective needs depends on the right combination and articulation of specific satisfiers (Cruz et al. 2009).

The Human Scale Development matrix is applied in community exercises, which divides participants into groups that analyse the needs and the satisfiers that have constructive or destructive effects in their society (Alkire 2002). This allows them to recognise what has to be changed to improve their livelihoods and encourages them to take action. During environmental planning, Human Scale Development matrices can be used to understand cultural ecosystem service demand and the importance they have for communities. Cultural ecosystem services provide many benefits which are satisfiers of human needs (Church et al. 2011). For example, educational services derived from ecosystems, provide both formal and informal education, which is a satisfier of the need for *understanding*. Interacting with the natural environment in the form of recreation is a satisfier for the fundamental human need of *idleness*. Table 3 provides an example of how a community exercise might look like using the benefits derived from cultural ecosystem services, which act as satisfiers of the human need of *identity*. Sense of place and societal services provide, in many communities, the benefits of sense of belonging, differentiation, integration and recognition of the environmental setting they belong to. Cultural heritage services provide symbols and historic memory. Spiritual and religious services allow to "get to know oneself" and to "recognise oneself".

Table 3 Expressions of human needs for *identity* through the use of cultural benefits as satisfiers (After Max-Neef 1992)

–	**Being** (personal or collective attributes)	**Having** (institutions, tools, norms)	**Doing** (personal or collective actions)	**Interacting** (settings in time and space)
Identity	Sense of belonging, differentiation	Symbols, religion, historic memory	Integrate oneself, get to know oneself, recognise oneself	Setting which one belongs to

With these participatory exercises, communities can identify the cultural benefits which help in satisfying their fundamental needs and that are essential prerequisites for a rich and meaningful life. They can help in revealing the total value of these services to communities.

4 The Values of Cultural Ecosystem Services

The value of ecosystem services is often not easily measurable since a great amount of their produced benefits are not accounted and are not foreseen by conventional markets. Consequently, it is useful to classify the different types of value to apply the most suitable valuation methods. The values of ecosystems stem from two broad categories: the use and the non-use values (MA 2003; TEEB 2010). The use values are classified in direct-use, indirect-use and option values. The direct-use values encompass the straightforward use of ecosystem services, such as extracting water from a river for drinking purposes, or boating on a lake. The direct-use values are normally more susceptible to monetary valuations because they are often traded in conventional markets. The indirect-use values refer to ecosystem services that indirectly benefit people, such as water purification and carbon sequestration. The option values are associated with the preservation of a natural resource that is not used at the present moment but it might be beneficial in the future, such as protecting and conserving a plant community and its habitat, because it might contain undiscovered therapeutic substances.

The non-use values are not captured by market valuations because they are highly subjective and arise from non-material benefits. Their classification is divided in bequest and existence values (Walsh et al. 1984). The bequest values encompass the benefits derived from knowing that future generations will be able to enjoy the same natural settings that exist nowadays. The existence values are derived from the enjoyment that people might experience just by knowing that a particular ecosystem exists, irrespective of its usage.

A recurring issue when dealing with cultural ecosystem services valuation, is that many people obtain these services in bundles because they often overlap, leading to double-counting. For example, people experiencing a pleasant waterscape scenery (aesthetic services) might feel, at the same time, both inspired and spiritually fulfilled, thus getting inspirational and spiritual services from the same waterscape. Recreation and tourism services are also many times associated with aesthetic services or cultural heritage services. These valuation issues may not have a common solution because cultural ecosystem services are many times circumstantial and highly subjective. A possible way to tackle this issue, and avoid double-counting, is the implementation of a local or regional adaptation of the ecosystem service assessment, one that lets stakeholders decide what are the most valuable cultural ecosystem services (Chan et al. 2012a).

4.1 Valuation Dimensions: Ecological, Socio-cultural, and Economic

The valuation of ecosystems services can be approached by considering their main dimensions: ecological, socio-cultural and economic (De Groot et al. 2002). However, many cultural ecosystem services are ill-suited for monetary valuation because they provide non-market benefits. Most of them are public and important to society as a whole, instead to individuals, which contradicts the theoretical fundamentals of economic valuation methodology based on individual preferences and individual utility maximization (Kumar and Kumar 2008). Nevertheless, the economic value of some cultural ecosystem services has been derived. De Groot et al. (2012) estimated the world monetary value of cultural ecosystem services for the main aquatic ecosystems: 6,369 int.$/ha/year for inland wetlands and other freshwater biomes; 2,493 int.$/ha/year for coastal wetlands and other coastal systems; 108,837 int.$/ha/year for coral reefs; and 319 int.$/ha/year for marine biomes. However, only few cultural services were considered and most of the monetary value was attributed to "recreation" and to "aesthetic information".

Most of the cultural ecosystem services importance arises from their ecological value (de Groot et al. 2002). This value is determined by the magnitude of the system's complexity, diversity, and rarity (de Groot et al. 2000). The magnitude of these criteria depends on the system's ecological integrity (Müller et al. 2000), resilience (Holling 1986) and resistance (Booth and Figueira 2008). Understanding and assessing these criteria in ecosystems is crucial to recognise when, or under what circumstances, a regime shift is likely to occur, one that might diminish or enhance the value of ecosystem services (Limburg et al. 2002). Predicting and assessing the outcomes of contrasting state regimes is a worthy approach to derive the ecological value of aquatic ecosystems, in what regards the provision of benefits to people.

Socio-cultural values strongly relate to cultural ecosystem services (de Groot et al. 2002). Many people value ecosystems due their historical, national, ethical, religious, and spiritual significance (MA 2005). They are associated with the provision of physical and mental health, education opportunities, socio-cultural diversity and identity, among many others. These values are determined by individual and social preferences. These preferences are dynamic, evolving, and are greatly influenced by prevailing social and cultural practices (Kumar and Kumar 2008). Qualitative methods have been proposed and applied by Cocks et al. (2012), Chan et al. (2012a), Gee and Burkhard (2010), and Tengberg et al. (2012) to identify and assess socio-cultural values.

Capturing the value of ecosystem services is fundamental to prevent irreparable losses in ecosystem's service provision. These losses occur because ecosystems are

Box 1 Schleimünde case study

In 2012, I did a survey in Schleimünde, a nature reserve located in the German Baltic Coast of Schleswig-Holstein, to reveal cultural ecosystem services' values and preferences. This was achieved by interviewing 110 Schleimünde visitors on their reasons for visiting the reserve, on their willingness-to-pay (WTP) to support and improve the reserve's conservation efforts, and on the underlying reasons that motivated them for that support. These interviews gave insights into the reserve's ecological, socio-cultural, and economic values.

The reserve's ecological value is the fundamental reason why Schleimünde is a protected area. The reserve comprehends a small peninsula with 127 ha of land and 564 ha of adjacent waters. Its landscape ranges from a flat coast with beach dunes and ridges in one side, to grasslands and mudflats in the other (Fig. 2). This is an area with a significant fauna and flora diversity, whose conservation efforts are mainly directed towards water bird species, such as gulls and terns, due to the site's importance as their breeding and resting grounds (Vogel 2011). The natural landscape and biodiversity constitute the reserve's ecological value and are, respectively, the biophysical structure and the ecosystem function from which cultural ecosystem services arise. The benefits obtained from cultural ecosystem services are the reasons why people visit the reserve.

People visit Schleimünde because they value the non-material benefits such as physical and mental wellbeing, learning opportunities, enjoyment, and serenity, among many others. These benefits originate from cultural ecosystem services and are the root of their socio-cultural value, which is determined by individual and social preferences (Kumar and Kumar 2008). To unveil visitors' preferences, the interview contained a group of statements, each one corresponding to a cultural ecosystem service, from which the

Fig. 2 View of Schleimünde Nature Reserve. On the *left*, the beach side and grassland, facing the Baltic Sea. On the *right*, the mudflats and adjacent waters of the estuary

(continued)

Box 1 (continued)

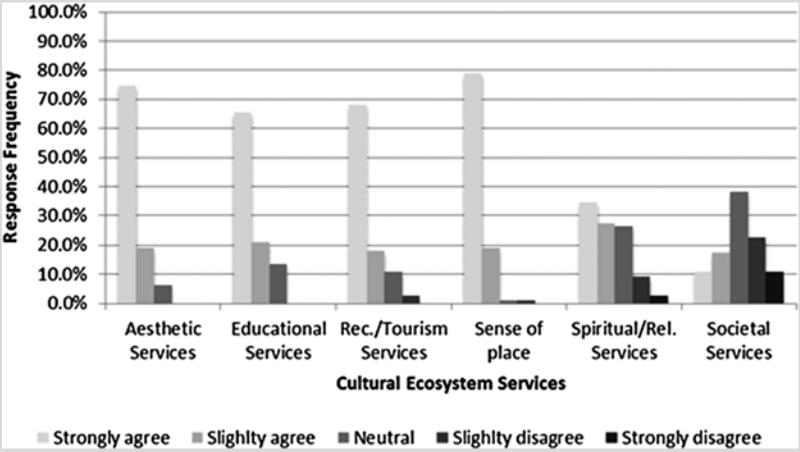

Fig. 3 Percent frequency distribution of respondent's cultural ecosystem services preferences. N = 110

respondents' answering options ranged from "strongly agree" to "strongly disagree", in a five-point Likert scale (Fig. 3).

Respondents visited Schleimünde mainly due to its "sense of place" (79.1 %) and its aesthetic services (74.5 %). Other stated reasons were the opportunities for recreation and tourism (68.2 %), and for education (65.5 %). Four, out of six cultural ecosystem services, had more than 65 % of response frequency, as the main reason for going to the reserve. These results confirm that people actively sought the benefits provided by the reserve's cultural services, as they were amongst the main reasons for visiting it. The importance of these ecosystem services to human wellbeing, i.e., their socio-cultural value, is here highlighted because each statement, from which respondents could choose, was coded as ecosystem's benefits, instead of the exact classification of a particular cultural service (e.g., "I visited the nature reserve because I like the scenic beauty of the landscape" instead of "aesthetic services"). This had the intention to ease the cognitive burden upon the respondents, so they could easily connect the obtained benefits with their own wellbeing.

To derive the economic value that respondents assigned to the reserve, I included in the survey a contingent valuation (Mitchell and Carson 1989). The valuation scenario proposed the implementation of an entrance fee to access the protected area. The respondents were reminded that these funds

(continued)

Box 1 (continued)

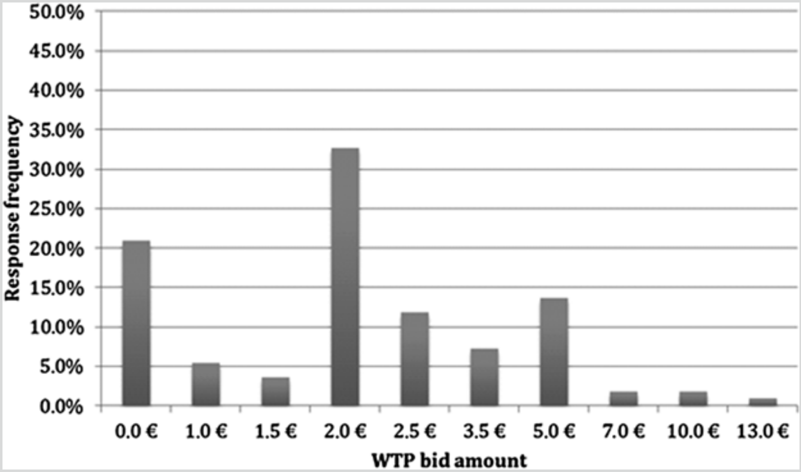

Fig. 4 Percent frequency distribution of respondents' WTP amount. N = 110

would be applied to improve the reserve's conservation efforts and the visiting conditions. Most of the respondents (79.1 %) chose a positive WTP amount for an entrance fee (Fig. 4).

Two groups of respondents did not choose a positive WTP amount – those who selected 0 € (9.1 %), therefore rejecting the valuation scenario, and those who declared "don't know" (11.8 %). I calculated a parametric WTP measure (Blaine et al. 2005) by using the statistically significant explanatory variables provided by a multiple linear regression model. The parametric WTP mean was 3.26 €, which reaches an aggregate value of 22,411 € for 2012, considering that 6,866 people visited the reserve during that year (Fischer and Burkhard 2012). This acknowledges the economic value of the reserve and its services. This amount is substantial and can be converted in potential revenues that can be used in the reserve's conservation projects in the future.

A follow-up was included after the WTP question to identify the reasons of the respondents' positive choices. Each reason corresponded to a different value type, i.e., use and non-use values (Fig. 5). Respondents could choose more than one option to justify their choices.

Most of the respondents (74.7 %) justified their choices with the reserve's bequest value, followed by the indirect-use value (65.5 %). About half of the respondents (50.6 %) highlighted the importance of the existence value and 46.0 % chose the option value. The direct-use value was the least stated choice (33.3 %).

(continued)

Box 1 (continued)

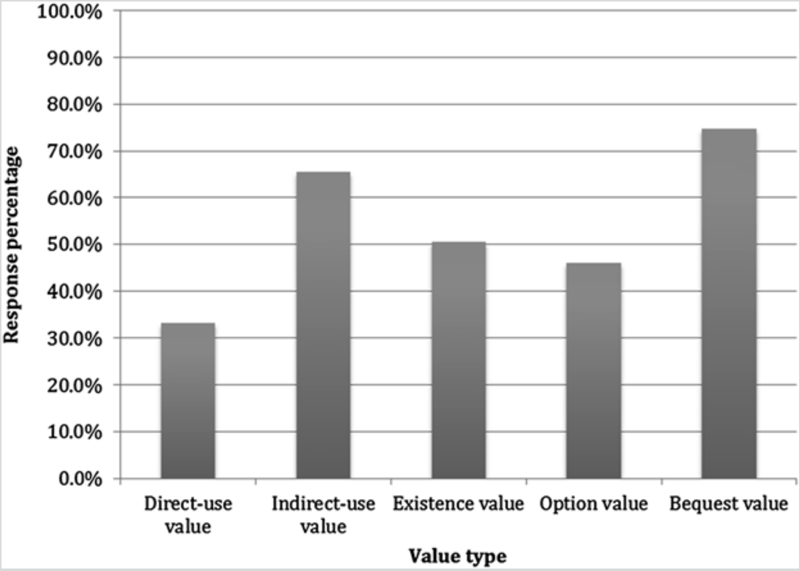

Fig. 5 Percent frequency distribution of the value types to justify WTP positive responses. N = 87

Non-use values, i.e., bequest and existence values, were the main reason for respondents' WTP. Knowing that the future generations will be able to experience Schleimünde (bequest value) was the most stated reason for WTP, which indicates that respondents recognised that conserving the ecosystem and its services are important to society as a whole, rather than to individuals, contradicting the theoretical fundamentals of economic valuation methodology based on individual utility maximization (Kumar and Kumar 2008). To support this assumption, direct-use values were the less stated reason for WTP, even though most of the respondents were involved in recreational and tourism activities. Indirect-use and option values were amongst the most stated reasons and indicate that respondents are well aware and recognise the ecosystems contribution to their own wellbeing.

The socio-cultural, ecological and economic dimensions of value were interlinked and all contributed to the total value of the ecosystem and its cultural services.

often undervalued as a consequence of being poorly understood, barely monitored, and many times undergoing rapid degradation and depletion (Daily et al. 2000). A valuation framework that encompasses the multi-dimensional levels of the ecosystem services value has the potential to be a useful management tool to protect ecosystems and their services. To illustrate these issues I present a case study where the multi-dimensions of cultural ecosystem services value were explored (Box 1).

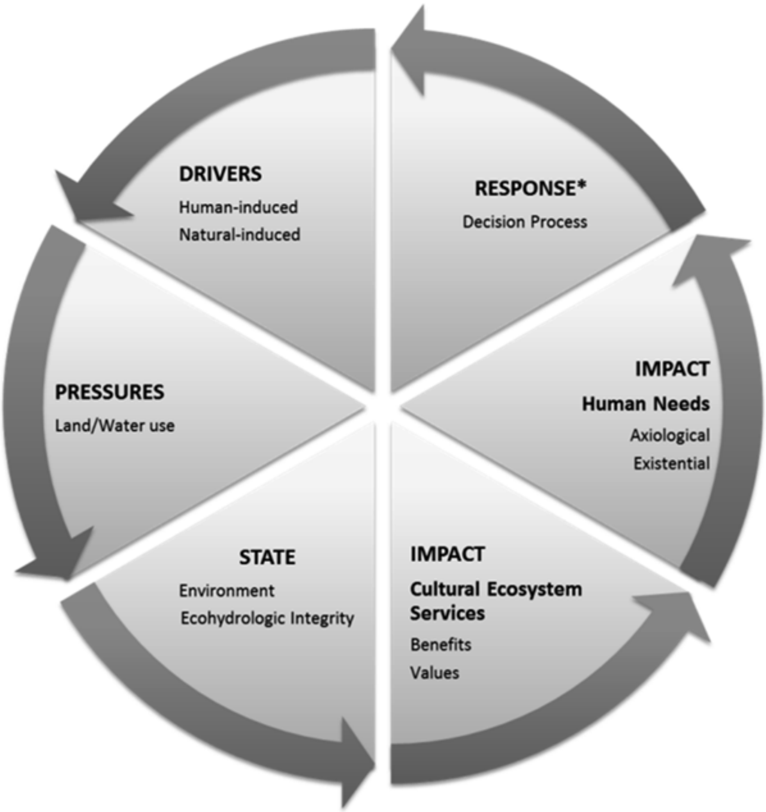

Fig. 6 Conceptual model of human-environmental relationships for cultural ecosystem services, based on the DPSIR framework (After Burkhard and Müller 2008). Human needs and cultural ecosystem service demand can both be drivers of change. *responses are not necessarily only targeted to drivers. They can be targeted to any other part of the system

5 The DPSIR Framework as a Management Tool for Decision-makers

The driver-pressure-state-impact-response (DPSIR) framework (EEA 1995) is a tool used to characterise the cause-effect relationships within human-environmental systems (Burkhard and Müller 2008). The approach allows having a holistic insight of the system and understanding how its components develop under change. Figure 6 provides an example of a DPSIR framework highlighting interactions of human needs, cultural ecosystem services, and ecohydrological integrity.

Drivers are the factors responsible for natural or human-induced changes in the system (Burkhard and Müller 2008). Satisfiers of human needs are drivers of change. Growing human demand for ecosystem services such as water supply, hydropower generation, or recreation and tourism activities, are all examples of typical human drivers of change in aquatic ecosystems. A particular serious global driver is population growth. It is associated with increasing agriculture and industrial production, growing natural resource consumption and urban sprawl, drivers that imperil many ecosystems. By affecting environmental systems, human-induced drivers also affect satisfiers of human needs, which are dependent on healthy ecosystems. Natural-induced drivers of change are all those processes not affected or controlled by humans, such as seismic or volcanic activity.

Pressures represent how drivers are expressed in the environment. All human activities affecting the environment result in pressures to the system and are often connected to specific causes (Burkhard and Müller 2008). They are many times expressed in diverse water uses, as is the case of water abstraction, flow regulation, or aquaculture discharges. The environmental conditions of the system i.e., its *state*, are changed and many times degraded by the pressures exerted on them. The state of aquatic ecosystems can be assessed by using the ecohydrologic integrity indicandum (Table 2).

Modifications in the state of healthy ecosystems lead to *impacts* in human livelihoods. Often it results in non-satisfaction or poor fulfilment of fundamental human needs. This is intimately related with impacts on the provision of cultural services that healthy ecosystems provide. It is thus essential, at this phase, to identify the key cultural services, associated benefits and their values, and devise strategies to attenuate or even reverse on-going impacts. As local beneficiaries are likely to have the most insightful knowledge about the importance of these services, their voice should be heard, so they can help identify the range of cultural services provided, the benefits obtained, and why these benefits are valuable to them (Chan et al. 2012b). Impact indicators are crucial for management and decision-making, as they depict directly the environmental and societal consequences of human actions (Burkhard and Müller 2008).

Once the impacts are acknowledged, *responses* are developed by decision-makers. Responses can be targeted to any other part of the system (Borja et al. 2006), but ideally should be directed to drivers and pressures, and as a result improve the state of ecosystems (Burkhard and Müller 2008). Common responses for ecosystem services conservation and protection are payment for ecosystem services schemes (Engel et al. 2008; Kinzig et al. 2011) and restoration initiatives (Trabucchi et al. 2012).

The DPSIR framework is intended to be iterative, promoting a progressive deepening understanding of human-environmental relationships. It has potential to be used as a guide for decision-makers and help them to better address cultural services in environmental policy.

6 Discussion

Cultural services are often neglected from ecosystem services studies due to difficulties in their operation such as uncertainties about their biophysical generation, unsuitable valuation methods, and absence of appropriate frameworks. This causes their undervaluation and underprotection, which coupled with the widespread degradation of aquatic ecosystems, may lead to irreparable losses. One approach to tackle these issues is to unveil the total value of cultural ecosystem services.

In Schleimünde Nature Reserve, I studied the cultural ecosystem services value by interviewing reserve's visitors. The case study indicated that people actively sought the benefits from cultural ecosystem services and that these services were the main reason for their visit. Undoubtedly, they were valuable to people and they had ecological, socio-cultural, and economic values.

The reserve's ecological value is inherent to the reserve itself. Its landscape and biodiversity are the fundamental contributors of the area's ecological value. They are part and contribute to the system's complexity, diversity, and rarity (de Groot et al. 2000). However, to fully capture the reserve's ecological value, it would have been useful to know the magnitude of these criteria. Particularly useful for this case study, would have been the quantification of ecohydrologic attributes such as endemic species, wilderness, remoteness, and openness, to know the system's cultural services supply by linking biophysical units with visitors' preferences. This approach is in the core of the "landscape" concept (or "waterscape" in this context), as it joins objective space and subjective place (Eisenhauer et al. 2000; Stephenson 2008; Gee and Burkhard 2010).

The socio-cultural value was highlighted by respondents in their reasons for visiting the reserve. Each statement represented one cultural ecosystem service and was "coded" in the form of benefits inherent to each service. Most of respondents "strongly agreed" with these statements as reasons for visiting the reserve, thus confirming that these cultural services provided benefits such as calmness, enjoyment, leisure, etc. Furthermore, the majority of respondents (74.7 %) justified their WTP choices for an entrance fee with the reserve's bequest value. This supports the notion that people are aware of the social value that these ecosystems and their services have to society, as they recognised their importance to the wellbeing of future generations. One interesting approach to understand the socio-cultural value and cultural ecosystem service demand would have been the engagement of visitors in an adapted human scale development matrix (Max-Neef et al. 1986, 1989).

To reveal the economic value that respondents assigned to the reserve, it was proposed the implementation of an entrance fee to access the area. The valuation scenario included monetary units because these are used by respondents on a daily basis and therefore, are easily understandable and recognisable. Most of respondents (79.1 %) were willing to pay for an entrance fee in a future visit, thus confirming their financial support towards the reserve's conservation efforts. This indicates that protected areas and their cultural ecosystem services have economic value and are able to generate revenues. These revenues can be greatly useful to support protected areas, converting them in independent institutions by allowing them to be financial self-sufficient or less dependent from external funds.

The implications of this case study are encouraging in what regards the protection of cultural ecosystem services, however it has limitations. It was targeted only to visitors. Other stakeholders may show different opinions and preferences. Furthermore, the survey had a local scope. More representative studies, with a comprehensive scale, have the potential to be more insightful.

The valuation approaches considered in this case study recognise the importance and demand of cultural ecosystem services. These approaches can be integrated in a DPSIR framework for a better understanding of human-environmental systems. The ecohydrologic integrity indicandum can also be included in the DPSIR framework to characterise the state of aquatic ecosystems and unveil how cultural services are generated. This framework acknowledges that human needs and the demand for cultural ecosystem services can be impacted but also be drivers of change, which contradicts the unidirectional causality between the framework's components, pointed out by Berger and Hodge (1998), and Rekolainen et al. (2003). Svarstad et al. (2008) stated that the DPSIR framework should not produce only a single researcher's "narrative", as frequently happens, but examine as well the different discourses among stakeholders. Discourse-base methods (Wilson and Howarth 2002) can contribute to broaden the DPSIR framework by considering all stakeholders perspectives. This deliberative assessments can help in designing responses in the form of institutions, judgements, management and restoration initiatives (de Groot et al. 2010), which together join the perspectives of social and natural sciences about environmental systems (Braat and de Groot 2012).

7 Conclusion

Cultural services are often neglected from ecosystem services due to difficulties in their operation. With an aquatic ecosystems context, I discussed some of these difficulties. I gave insights into the biophysical generation of cultural services and suggested the ecohydrologic integrity indicandum to assess their provision. I approached the demand side by considering that the benefits derived from cultural ecosystem services are satisfiers of fundamental human needs, and that deliberative approaches can help in unveiling key satisfiers.

I presented a case study to illustrate how cultural ecosystem service values are revealed. The multi-dimensions of value (ecologic, socio-cultural, and economic) were all significant in the formation of the total cultural ecosystem services value.

For decision-makers, the DPSIR framework is a useful management tool because it can integrate cultural ecosystem service demand and supply, human needs, valuation, and thus characterise human-environmental relationships. The framework can contribute in finding suitable responses to protect and conserve aquatic ecosystems and their services.

Many issues remain unsolved and future research needs to explore people's perceptions and preferences towards cultural ecosystem services and analyse how they affect the satisfaction of human needs, both individually and collectively. Moreover, it is rather important to identify which ecosystems structures, processes and functions are carriers and providers of cultural ecosystem services, in order to devise suitable strategies to protect their provision.

References

Alkire S (2002) Dimensions of human development. World Dev 30(2):181–205

Aylward B, Bandyopadhyay J, Belausteguigotia J, Börkey P, Cassar A, Meadors L, Saade L, Siebentritt M, Stein R, Tognetti S, Tortajada C, Tony A, Carl B, Carl B, Guimarães-Pereira A, Kendall M, Kiersch B, Landry C, Rodriguez EM, Meinzen-Dick R, Moellendorf S, Pagiola S, Porras I, Ratner B, Shea A, Swallow B, Thomich T, Voutchkov N (2005) Freshwater ecosystem services. In: Ecosystems and human well-being: policy responses, vol 3., pp 213–255

Barkmann J, Baumann R, Meyer U, Müller F, Windhorst W (2001) Ökologische Integrität: Risikovorsorge im Nachhaltigen Landschaftsmanagement. GAIA 10(2):97–108

Berger AR, Hodge RA (1998) Natural change in the environment: a challenge to the pressure–state–response concept. Soc Indic Res 44:255–265

Blaine TW, Lichtkoppler FR, Jones KR, Zondag RH (2005) An assessment of household willingness to pay for curbside recycling: a comparison of payment card and referendum approaches. J Environ Manage 76:15–22

Booth DJ, Figueira WF (2008) Resistance and buffer capacity. In: Jørgensen SE, Fath BD (eds) Population dynamics, vol 4 of Encyclopedia of ecology, 5 vols. Elsevier, Oxford, pp 3004–3009

Borja A, Galparsoro I, Solaun O, Muxika I, Tello EM, Uriarte A, Valencia V (2006) The European Water Framework Directive and the DPSIR, a methodological approach to assess the risk of failing to achieve good ecological status. Estuar Coast Shelf Sci 66(1):84–96

Braat LC, de Groot R (2012) The ecosystem services agenda: bridging the worlds of natural science and economics, conservation and development, and public and private policy. Ecosyst Serv 1(1):4–15

Brauman KA, Daily GC, Duarte TKE, Mooney HA (2007) The nature and value of ecosystem services: an overview highlighting hydrologic services. Annu Rev Environ Resour 32:67–98

Burkhard B, Müller F (2008) Drivers-pressure-state-impact-response. In: Jørgensen SE, Fath BD (eds) Ecological indicators, vol 2 of Encyclopedia of ecology, 5 vols. Elsevier, Oxford, pp 967–970

Chan KMA, Goldstein J, Satterfield T, Hannahs N, Kikiloi K, Naidoo R, Vadeboncoeur N, Woodside U (2011) Cultural services and non-use values. In: Kareiva P, Tallis H, Ricketts TH, Daily GC, Polasky S (eds) Natural capital: theory and practice of mapping ecosystem services. Oxford University Press, New York, pp 206–228

Chan KM, Guerry AD, Balvanera P, Klain S, Satterfield T, Basurto X, Bostrom A, Chuenpagdee R, Gould R, Halpern BS, Hannahs N, Levine J, Norton B, Ruckelshaus M, Russell R, Tam J, Woodside U (2012a) Where are cultural and social in ecosystem services? A framework for constructive engagement. Bioscience 62(8):744–756

Chan KMA, Satterfield T, Goldstein J (2012b) Rethinking ecosystem services to better address and navigate cultural values. Ecol Econ 74:8–18

Church A, Burgess J, Ravenscroft N, William B, Blackstock K, Brady E, Crang M, Fish R, Gruffudd P, Mourato S, Pretty J, Tolia-Kelly D, Turner K, Winter M (2011) Chapter 16: Cultural services. In: UK National Ecosystem Assessment: understanding nature's value to society, Technical report. UNEP-WCMC, Cambridge, pp 633–692

Cocks ML, Dold T, Vetter S, Herbarium B, Cocks M (2012) "God is my forest"–Xhosa cultural values provide untapped opportunities for conservation. S Afr J Sci 108:5/6

Costanza R, d'Arge R, de Groot R, Farbek S, Grasso M, Hannon B, Limburg K, Naeem S, O'Neill R, Paruelo J, Raskin R, Suttonkk P, van den Belt M (1997) The value of the world's ecosystem services and natural capital. Nature 387:253–260

Cruz I, Stahel A, Max-Neef M (2009) Towards a systemic development approach: building on the human-scale development paradigm. Ecol Econ 68(7):2021–2030

Daily GC (1997) Nature's services: societal dependence on natural ecosystems. Island Press, Washington, DC

Daily GC, Söderqvist T, Aniyar S, Arrow K, Dasgupta P, Ehrlich PR, Folke C, Jansson AM, Jansson BO, Kautsky N, Levin S, Lubchenco J, Mäler KG, Simpson D, Starrett D, Tilman D, Walker B (2000) The value of nature and the nature of value. Science 289(5478):395–396

Davis TJ (1994) The Ramsar Convention manual: a guide to the Convention on Wetlands of International Importance. International Union for Conservation of Nature and Natural Resources, Ramsar Convention Bureau, Gland, p 207

de Groot RS, van der Perk J, Chiesura A, Marguliew S (2000) Ecological functions and socio-economic values of critical natural capital as a measure for ecological integrity and environmental health. In: Crabbe P, Holland A, Ryszkowski L, Westra L (eds) Implementing eco-logical integrity: restoring regional and global environmental and human health, vol 1, NATO-science series, IV. Earth and environmental sciences. Kluwer Academic Publishers, Dordrecht/Boston/London, pp 191–214

de Groot RS, Wilson MA, Boumans RMJ (2002) A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecol Econ 41(3):393–408

de Groot RS, Fisher B, Christine M, Aronson J, Braat L, Gowdy J, Haines-Young R, Maltby E, Neuville A, Polasky S, Portela R, Ring I (2010) Integrating the ecological and economic dimension in biodiversity and ecosystem service valuation. In: Kumar P (ed) The economics of ecosystems and biodiversity: ecological and economic foundations. Earthscan, Washington, DC, pp 11–40

de Groot RS, Brander L, van der Ploeg S, Costanza R, Bernard F, Braat L, Christine M, Cross-man N, Ghermandi A, Hein L, Hussain S, Kumar P, McVittie A, Portela R, Rodriguez LC, Brink P, van Beukering P (2012) Global estimates of the value of ecosystems and their services in monetary units. Ecosyst Serv 1(1):50–61

Dodds WK (2002) Freshwater ecology: concepts and environmental applications. Academic, San Diego, pp 91–111

EEA (European Environment Agency) (1995) Europe's environment: the Dobris assessment. EEA, Copenhagen

Ehrlich PR, Mooney HA (1983) Extinction, substitution, and ecosystem services. BioScience 33(4):248–254

Eisenhauer BW, Krannich RS, Blahna DJ (2000) Attachments to special places on public lands: an analysis of activities, reason for attachments, and community connections. Soc Nat Resour 13(5):421–441

Engel S, Pagiola S, Wunder S (2008) Designing payments for environmental services in theory and practice: an overview of the issues. Ecol Econ 65(4):663–674

Fischer JF, Burkhard B (2012) NSG Schleimündung. Jahresber 2012

Gee K, Burkhard B (2010) Cultural ecosystem services in the context of offshore wind farming: a case study from the west coast of Schleswig-Holstein. Ecol Complex 7(3):349–358

Haines-Young R, Potschin M (2010) The links between biodiversity, ecosystem services and human well-being. In: Raffaelli DG, Frid CLJ (eds) Ecosystem ecology: a new synthesis. Cambridge University Press, Cambridge, pp 110–139

Halpern BS, Walbridge S, Selkoe KA, Kappel CV, Micheli F, D'Agrosa C, Bruno JF, Casey KS, Ebert C, Fox HE, Fujita R, Heinemann D, Lenihan HS, Madin EMP, Perry MT, Selig ER, Spalding M, Steneck R, Watson R (2008) A global map of human impact on marine eco-systems. Science 319(5865):948–952

Holling CS (1986) The resilience of terrestrial ecosystems: local surprise and global change. In: Clark WC, Munn RE (eds) Sustainable development of the biosphere. Cambridge University Press, Cambridge

Jørgensen SE (1992) Integration of ecosystem theories: a pattern. Kluwer Academic Publishers, Dordrecht

Kandziora M, Burkhard B, Müller F (2013) Interactions of ecosystem properties, ecosystem integrity and ecosystem service indicators—a theoretical matrix exercise. Ecol Indic 28:54–78

Karr JR (1991) Biological integrity: a long-neglected aspect of water resource management. Ecol Appl 1:66–84

Kinzig AP, Perrings C, Chapin FS, Polasky S, Smith VK, Tilman D, Turner BL (2011) Paying for ecosystem services-promise and peril. Science 334(6056):603–604

Kumar M, Kumar P (2008) Valuation of the ecosystem services: a psycho-cultural perspective. Ecol Econ 64:808–819

Limburg KE, O'Neil RV, Costanza R, Farber S (2002) Complex systems and valuation. Ecol Econ 41:409–420

Lotze HK, Lenihan HS, Bourque BJ, Bradbury RH, Cooke RG, Kay MC, Kidwell MS, Kirby MX, Peterson CH, Jackson JB (2006) Depletion, degradation, and recovery potential of estuaries and coastal seas. Science 312(5781):1806–1809

MA (Millennium Ecosystem Assessment) (2003) Ecosystems and human well-being: a framework for assessment. Island Press, Washington, DC

MA (Millennium Ecosystem Assessment) (2005) Ecosystems and human well-being: synthesis. Island Press/World Resources Institute, Washington, DC, p 40

Maltby E, Ormerod S, Acreman M, Dunbar M, Jenkins A, Maberly S, Newman J, Blackwell M, Ward R (2011) Chapter 9: Freshwaters: openwaters, wetlands and floodplain. In: UK National Ecosystem Assessment: understanding nature's value to society, Technical report. UNEP-WCMC, Cambridge, pp 295–360

Max-Neef M (1992) Development and human needs. In: Ekins P, Max-Neef M (eds) Real life economics. Routledge, London, pp 197–214

Max-Neef M, Elizalde A, Hopenhyan M (1986) Desarollo a Escala Humana: una opción para el futuro. Development dialogue. Cepaur-Dag Hammarskjöld Foundation, Uppsala

Max-Neef M, Elizalde A, Hopenhyan M (1989) (1) Human scale development. Development dialogue. Cepaur-Dag Hammarskjöld Foundation, Uppsala, Sweden

Mitchell RC, Carson RT (1989) Using surveys to value public goods: the contingent valuation method. Resources for the Future, Washington, DC

Müller F (2005) Indicating ecosystem and landscape organisation. Ecol Indic 5(4):280–294

Müller F, Hoffmann-Kroll R, Wiggering H (2000) Indicating ecosystem integrity—theoretical concepts and environmental requirements. Ecol Model 130(1):13–23

Poff NL, Ward JV (1989) Implications of stream-flow variability and predictability for lotic community structure: a regional analysis of stream-flow patterns. Can J Fish Aquat Sci 46:1805–1818
Postel SL, Carpenter S (1997) Freshwater ecosystem services. In: Daily GC (ed) Nature's services: societal dependence on natural ecosystems. Island Press, Washington, DC
Rekolainen S, Kämäri J, Hiltunen M (2003) A conceptual framework for identifying the need and role of models in the implementation of the water framework directive. Int J River Basin Manag 1(4):347–352
Richter BD, Baumgartner JV, Wigington R, Braun DP (1997) How much water does a river need? Freshw Biol 37:231–249
Rosenberg DM, McCully P, Pringle CM (2000) Global-scale environmental effects of hydrological alterations: introduction. Bioscience 50(9):746–751
Schaich H, Bieling C, Plieninger T (2010) Linking ecosystem services with cultural landscape research. GAIA-Ecol Perspect Sci Soc 19(4):269–277
Stephenson J (2008) The cultural values model: an integrated approach to values in landscapes. Landsc Urban Plan 84(2):127–139
Svarstad H, Petersen LK, Rothman D, Siepel H, Wätzold F (2008) Discursive biases of the environmental research framework DPSIR. Land Use Policy 25(1):116–125
TEEB (2010) The economics of ecosystems and biodiversity: mainstreaming the economics of nature: a synthesis of the approach, conclusions and recommendations of TEEB. TEEB
Tengberg A, Fredholm S, Eliasson I, Knez I, Saltzman K, Wetterberg O (2012) Cultural ecosystem services provided by landscapes: assessment of heritage values and identity. Ecosyst Serv 2:14–26
Tharme RE (2003) A global perspective on environmental flow assessment: emerging trends in the development and application of environmental flow methodologies for rivers. River Res Appl 19(5-6):397–441
Trabucchi M, Ntshotsho P, O'Farrell P, Comín FA (2012) Ecosystem service trends in basin-scale restoration initiatives: a review. J Environ Manage 111:18–23
UNEP (2006) Marine and coastal ecosystems and human wellbeing: a synthesis report based on the findings of the millennium ecosystem assessment. UNEP, 76pp. http://www.unep.org/pdf/Completev6_LR.pdf
Vogel M (2011) NSG Schleimündung. Jahresber 2011
Wallace KJ (2007) Classification of ecosystem services: problems and solutions. Biol Conserv 139:235–246
Walsh RG, Loomis JB, Gillman RA (1984) Valuing option, existence, and bequest demands for wilderness. Land Econ 60(1):14–29
Wilson MA, Howarth R (2002) Discourse-based valuation of ecosystem services: establishing fair outcomes through group deliberation. Ecol Econ 41:431–443
Worm B, Barbier EB, Beaumont N, Duffy JE, Folke C, Halpern BS, Jackson JBC, Lotze HK, Micheli F, Palumbi SR, Sala E, Selkoe KA, Stachowicz JJ, Watson R (2006) Impacts of biodiversity loss on ocean ecosystem services. Science 314(5800):787–790

The Importance of Hyporheic Zone Processes on Ecological Functioning and Solute Transport of Streams and Rivers

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Abstract The influence of hyporheic exchange on the transport and transformation of solutes occurs in the environments where hydrologic and biogeochemical processes are dynamic and highly heterogeneous. We present three examples to specify the importance of hyporheic zone processes on different spatial scales ranging from small reach scale to whole river sections. We investigate the (i) impact of physical and biological clogging on the functional significance of the hyporheic zone, use (ii) small scale numerical studies analysing factors controlling advective exchange and solute transport and transformation and reveal (iii) spatial variation of nitrogen removal in river networks. Using a river in a pristine environment in central Mongolia we demonstrate that biological clogging shows seasonal effects on the hydrologic connectivity whereas biogeochemical regulation and habitat seemed to be affected less. Physical clogging revealed to have long-term impacts on the hydrologic connectivity, biogeochemical regulation and habitat. The simulation study of a lower mountain range river in central Germany shows that the location of highest hydraulic gradient is the location of the highest water exchange and nitrogen transformation. In spite of uncertainties involved in the process-based models valuable conclusions can be made towards focused theoretical and experimental studies for new process understating. The large scale denitrification study indicates a decreasing denitrification rate with increasing river length, but river morphology may modulate this general trend considerably. Furthermore nitrate concentration affects the nitrogen removal significantly. The associated longitudinal pattern of nitrogen removal can be assumed to be typical in highly eutrophic low order rives of central Europe.

Keywords River • Nitrogen removal • Clogging • Advective exchange • River network

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1 Introduction

In addition to being of aesthetic value, streams provide many services including sources of drinking, irrigation, and cooling water; hydropower; commercial transportation; recreation; food; and waste disposal (Meyer et al. 1988). Detailed understanding of processes affecting the fate of solutes and water quality is essential for maintaining the multiple uses of streams. For example, nutrients may control stream productivity, thereby affecting food sources or recreational values. The ability of streams to assimilate or degrade pollutants from point or nonpoint sources is a highly important ecosystem service and depends on the maintenance of ecosystem integrity (Cairns 1977). Solute transport links terrestrial and aquatic systems and upstream and downstream aquatic systems (Meyer et al. 1988), and may be used as an indicator of effects of anthropogenic disturbance on catchments (Stream Solute Workshop 1990). Moreover, the functions of the hyporheic zone may be the bottleneck for ecosystem integrity of riverine ecosystems (Borchardt and Pusch 2009).

Down-stream transport processes physically deliver solutes to reactive sites. These transport processes can be conceptually distinguished from exchanges between various reactive zones. Exchanges between sites may include chemical transformations (i.e., changes in chemical species), changes in physical state such as phase changes, sorption and desorption, and biological processes such as algal or microbial nutrient uptake, microbial oxidation and reduction, and invertebrate consumption of algae. Benthic up-take occurs when materials are transferred from the water column to the streambed (Stream Solute Workshop 1990). The streambed or hyporheic zone is an interface between surface and subsurface water (Orghidan 1959), which constitutes a unique environment for bio-/geochemical reactions and ecological processes. The hyporheic zone may be loosely defined as the porous areas of the stream bed and stream bank in which stream water mixes with shallow ground water. Due to differences in chemical composition of the surface and groundwater, exchange of water and solute between stream and hyporheic zone has many biogeochemical implications (Runkel et al. 2003). Hyporheic zones influence the biogeochemistry of stream ecosystems by increasing solute residence times, and more specifically solute contact with substrates, in environments with spatial gradient in dissolved oxygen and pH (Bencala 2000). The influence of hyporheic exchange on the transport and transformation of solutes occurs in the environments where hydrologic and biogeochemical processes are dynamic and highly heterogeneous.

The hyporheic zone is the main site of nitrogen removal by denitrification. Duff and Triska (1990) found that all the denitrifying activity in their rather pristine studied streams was associated with the subsurface solids. Surface water-groundwater interaction in the hyporheic zone may therefore enhance biogeochemical cycling in streams, and it has been hypothesized that streams exchanging more water with the hyporheic zone should have more rapid nitrate utilization (Lautz and Siegel 2007). Thus, the efficiency of nitrogen removal in streams depend on (1) the ability of water column nitrate to reach the nitrate removal sites and of the ability of particulate organic matter to be trapped (2) the rates at which the removal sites can process nitrogen (3) the residence time of nitrogen laden water near removal sites and (4) the proportion of stream water volume to reach nitrogen removal sites (Birgand et al. 2007).

The hydrodynamics of hyporheic exchange can be complex owing to individual and collective effects of channel morphology, sediment heterogeneity (Cardenas et al. 2004; Salehin et al. 2004; Wondzell 2006), and regional groundwater flow (Cardenas and Wilson 2006). Fine-grained sediment input, e.g. caused by agricultural land use, are a key constraint for this ecotone. The so called "colmation of the riverbed" as described in Beyer and Banscher (1975) and Schälchli (1992) reduces the hydraulic conductivity and therefore, the exchange of surface and subsurface water. This process decreases the dynamics and alters the transformation of solute and particulate matter as well as the habitat suitability (Ibisch et al. 2009; Brunke and Gonser 1997; Wood and Armitage 1997; Greig et al. 2007). Thus, river systems are sensitive to impairments of aquatic functions from pressures on the rivers hyporheic zone. Documenting the biogeochemical function of the hyporheic exchange can be accomplished in detailed high-sampling-intensity research, and with numerical studies (Bencala 2011).

The objective of this contribution is to highlight the importance of the hyporheic zone for ecological functioning and the removal of solutes. We will present examples of process studies which quantify the exchange between the water body and the hyporheic zone. In addition we will analyse its impact on ecological functioning, solute transport and retention and hence on ecosystem services of streams and rivers. Using these examples we will specify the importance of hyporheic zone processes on different spatial scales ranging from small reach scale to whole river sections. The contribution will be structured as follows

1. The impact of physical and biological clogging on the functional significance of the hyporheic zone
2. Small scale numerical studies on factors controlling advective exchange and solute transport (nitrogen) and transformation
3. Spatial variation of nitrogen removal in river networks

2 Impact of Physical and Biological Clogging on the Functional Significance of the Hyporheic Zone

2.1 Introduction

The hyporheic zone is an important functional zone within the aquatic environment (Naiman et al. 1988; Gibert et al. 1990) acting as a hydrological connector of the surface and subsurface water compartments, a physical and biogeochemical filter and reactor, a place for secondary production as well as a habitat and refugium (e.g. Bencala 1993; Stanford and Ward 1988; Grimm and Fisher 1984; Triska et al. 1993;

Naegeli and Uehlinger 1997; Williams and Hynes 1974). These functions are affected by the blockage of interstices with fine sediment or biofilms: Firstly, the physical clogging reduces the hydrological exchange flux between the surface and subsurface water bodies (Schälchli 1992, 1995) that may lead to a disconnection of these two compartments and ecosystems (Packman and MacKay 2003). Due to decreased exchange and increased residence time of solutes the biogeochemical properties and processes may be altered (Nogaro et al. 2010). Infiltrated fine sediments may provide larger surface areas for physical reactions and microorganism and they may contain organic fractions resulting in a different kind of a reactor system (Boulton et al. 1998; Lefebvre et al. 2005). Furthermore, increased fine sediment inputs alter habitat conditions with severe impacts on the development, colonization and reproduction of biota like invertebrates and fish (Brunke and Gonser 1997; Wood and Armitage 1997).

Secondly, biological clogging caused by dense benthic algae or biofilm growth may also lead to a blockage effect (Beyer and Banscher 1975; Battin and Sengschmitt 1999; Ibisch et al. 2009). Consequences for hyporheic zone functions can be different from those described above. Biological processing may be less limited by solute input as nutrients can be delivered by the degradation of algae (Jones et al. 1995). Habitat conditions may be good in terms of food availability but may also be moderate due to enhanced oxygen demand.

Both the aforementioned clogging phenomena depend on spatial and temporal aspects acting on different scales: fine sediment infiltration can depend on the local hydromorphology, the discharge regime (Adams and Beschta 1980; Schälchli 1992; Arnon et al. 2010; Harvey et al. 2012) whereas benthic algae/biofilm growth is controlled by the shear stress, turbidity, temperature, nutrient supply and consumer populations control (Gibert et al. 1990; Graham 1990; Valett et al. 1994). These scale dependent spatial and temporal effects may be superimposed by climatic and anthropogenic pressures.

The aim of this study was to comprehensively investigate the effects of clogging on the hyporheic functions. Therefore, investigations were done in the pristine environment of the Kharaa catchment in Northern Mongolia. As this environment showed almost no stress through contaminants the effects of biological and physical clogging on the hyporheic functions could be considered separately and in a mono-factorial way. A slight tendency towards eutrophication was observed for the middle reaches where biological clogging could be identified at an early stage. A tributary acted as a point source for suspended sediment that affected increased turbidity and a decrease in benthic algae growth (Hartwig et al. 2012). Therefore, effects of physical clogging could be diagnosed by comparing sites up- and downstream of the confluence.

2.2 Sampling Sites and Methods

The Kharaa River belongs to the drainage system of Lake Baikal and has an area of 15.000 km^2. The climate is continental with an annual precipitation of 250–350 mm and a mean discharge of about 12 m^3 s^{-1} at the catchment outlet. The three sampling

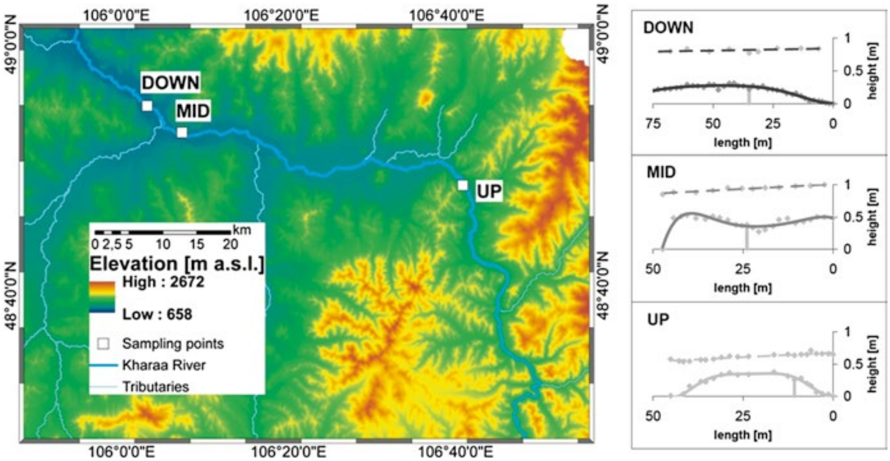

Fig. 1 Map of the sampling locations within the upper and middle reaches of the Kharaa catchment and longitudinal cross sections of the three specific riffle structures including the different water levels in June 2010 and location of the multilevel samplers at the infiltration zone

sites stretched from the mountain transition zone at "UP" to the floodplain zone where the sites "MID" and "DOWN" were located (see Fig. 1). The Kharaa River is a relatively pristine environment with nutrient surface water concentrations of 3.8 mg l^{-1} dissolved organic carbon (DOC), 0.3 mg l^{-1} nitrate (NO_3-N) and 0.015 mg l^{-1} soluble reactive phosphorus (SRP) (measured at DOWN, mean flow). Due to intensive livestock husbandry the riparian forest is almost entirely lost within the floodplain zone. Bank erosion leads to high sediment inputs (Theuring et al. 2013) and total suspended sediment concentrations drastically increased along the study reach (1.6, 4.0 and 15.4 mg l^{-1} at UP, MID and DOWN). Intensive monitoring campaigns were performed at the riffle scale every June and September of 2010 and 2011 after the discharge declined during snowmelt and summer rainfall events. Riverbed sediment was sampled twice at each site, sediment sampling campaigns using the freeze core method (Bretschko and Klemens 1986, modified). This method uses a metal pipe which is driven into the substratum and filled manually with liquid nitrogen. This leads to the freezing of the intra-gravel water and the included sediment to the pipe. The recovered core was split into three horizons (0–20, 20–40 and 40–60 cm depth) and analysed for the grain size distribution (see Hartwig et al. 2012). Temperature lances made from Tidbits with thin separators were installed at the riffle heads that recorded the temperatures at 15, 25 and 45 cm depth with an interval of 15 min. The temperature profiles were analysed with the help of the 'Exstream' software (Swanson and Cardenas 2011) in order to calculate the daily vertical fluxes according to Keery et al. (2007). Therefore, the suggested parameters for thermal conductivity, fluid density, specific heat of water and the system, and grain density were adopted. The porosity of each sediment layer was computed according to Vukovic and Soro (1992). Model outputs with a coefficient of determination smaller than 0.80 were discarded. Water quality parameters (electrical conductivity (EC), dissolved oxygen saturation (DO), DOC and NO_3-N) were measured in samples of the surface and subsurface water (see Hartwig et al. 2012). The subsurface water samples were collected with multilevel samplers (Lenk et al. 1999) at depth of 5, 15, 25 and 45 cm. The multilevel samplers were installed at the riffle heads and tail a half year prior to the campaigns. EC profiles were standardized using the following measured end-members: at the surface water and at depth of 45 cm at the riffle tail for UP assuming vertical and longitudinal subsurface transport, and at riffle heads for MID and DOWN assuming vertical transport only, respectively. The vertical profiles of the water quality parameters were normalized to the profiles taken in September 2011 when temperatures were only 4–8 °C expecting very low biogeochemical processing of these parameters. Data below the limit of detection were substituted with one-half of the distinct limit.

2.3 Results

The bed sediment at UP showed an increase of fine sediment within the matrix fraction with respect to depth (Fig. 2). The high spread within the data groups is indicative of the heterogeneity of the substrate. The fine sediment fraction was observed

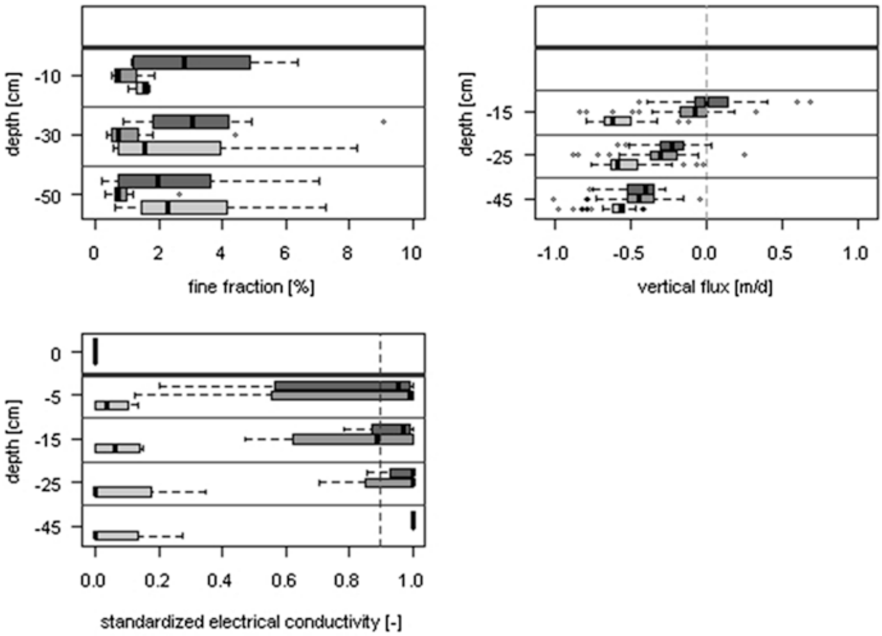

Fig. 2 Depth oriented overview of parameters at the three sampling sites UP (*light gray*), MID and DOWN (*dark gray*); parameters include the fine sediment fraction within the sediment matrix, the vertical flux component within the riffle infiltration zone over the summer period in 2011 (negative values implicate downwelling of surface water into the river bed sediment, positive stand for upwelling), the electrical conductivities standardized for the defined endmembers (‘0’ and ‘1’ refer to upper and lower boundary)

to be lowest at all layers of the MID site. Contrastingly, the DOWN site revealed the highest fine sediment fraction. Despite the use of the freeze coring technique for the extraction of the sediment samples, the upper 5 cm were either lost or not in the representative volume. Consequently statements on the uppermost benthic layer were rather weak.

For the characterization of the hydrological connectivity heat and EC were used as tracer. With the help of the temperature profiles, the vertical fluxes within the infiltration zone of the riffles were determined. The vertical fluxes were observed to be highest at UP across the whole observed sediment depth with a downwelling of about 0.6 m d^{-1}. At MID the downwelling was low until the depth of 15 cm at about 0.1 m d^{-1} but increased further down. A similar pattern was observed at DOWN as well, with downwelling of 0 m d^{-1} for the uppermost 15 cm. The penetration depth of the EC signal was used as an indicator for the active depth of the hyporheic zone. At UP the surface water EC signal penetrated the whole sediment depth with almost no interaction with subsurface water. Thus the active hyporheic zone was deeper than the monitored 45 cm. At MID and DOWN the pattern were quite distinct, the surface water signal was closely related to the subsurface water signal. The high spread in the data groups at MID was caused by a difference between the June and September samplings with EC values closely related to the surface water in June. At DOWN the sampling set taken in June 2010 corresponded strongly to the surface water EC signal whereas all other profiles were clearly related to the subsurface signal. Consequently, the penetration depth extended to a depth of less than 5–15 cm at MID and less than 5 cm at DOWN.

In order to characterize the biogeochemical potential, the profiles of EC, DO, DOC, and NO_3-N were compared for mean and low temperature conditions (Fig. 3). Similar hydraulic situations were assumed as the EC profiles showed a maximum discrepancy of ±22 % between the two conditions. Surface water DO differed little whereas subsurface DO showed high variation in dependency to the site. At UP and MID DO levels decreased by about of 15–70 % compared to the low temperature regime except for the depth of 25 cm at MID. Contrastingly, at DOWN DO values showed slightly elevated levels in comparison to the measurements during low temperature conditions. DOC levels were higher during the mean temperature conditions for both surface and subsurface waters at all sites. High subsurface DOC levels were found at UP and a substantial increase could be observed especially at the MID site with a peak at the topmost sediment layer. The two river sites within the floodplain zone, MID and DOWN, showed increased NO_3-N levels in surface water. With increasing distance from the spring subsurface vertical profiles revealed to be less heterogeneous by means of the range of concentrations. At UP a maximum in NO_3-N levels was observed for the uppermost centimetres whereas the other layers showed no distinct in- or decrease. At MID subsurface NO_3-N levels were lower for most of the time compared to the measurements during the low temperature regime and at DOWN no clear trend was found.

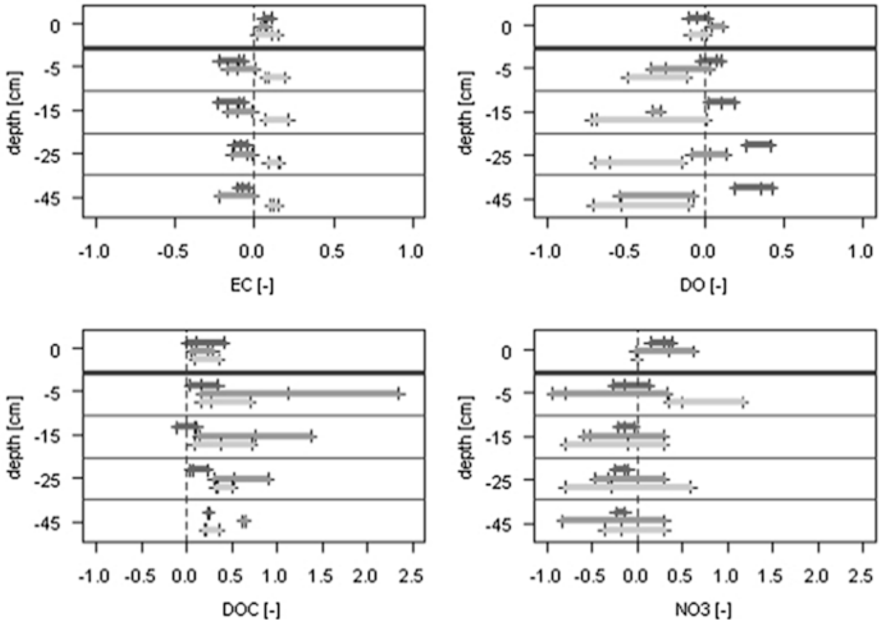

Fig. 3 Depth oriented overview of parameters normalized to the low temperature sampling at the three sampling sites UP (*light gray*), MID and DOWN (*dark gray*); parameters include the electrical conductivity (EC), oxygen saturation (DO), dissolved organic carbon (DOC) and nitrate-nitrogen (NO3)

2.4 Discussion

Along the longitudinal river gradient three conditions of the hyporheic zone exist: the upstream site seemed to consist of heterogeneous sediment with a slight tendency towards an inner clogging as bigger pores enable a deep infiltration of fine sediment. Clogging was observed at the two downstream sites. At MID no elevated fine sediment was found but substantial algae growth was observed (see Hartwig et al. 2012). Additionally, penetration depth were higher before the growing season and DOC, that was probably derived by algae lysis (Jones et al. 1995; Findlay 1995), showed a maximum in the topmost sediment layer. It can be concluded that this site was subject to biological clogging during growth periods. The very downstream site (DOWN) showed elevated fine sediments in the subsurface. This high sediment concentration led to high turbidity of the surface water and a muddy coating of the benthic layer. Hence, this site was subject to physical clogging.

According to the different conditions that may be named as “montane zone – natural”, “floodplain zone – biological clogging” and “floodplain zone – physical clogging” the functions revealed to be altered. Given the substantial decreases of vertical fluxes and penetration depth, the hydrologic connectivity was proven to be affected by the clogging and the advective infiltration became less in horizontal as well as vertical direction. When considering a threshold of 10 % of surface water content as a minimum for the delineation of the hyporheic zone (Triska et al. 1989) the extent of the hyporheic zone decreased in depth. Thereby, the physical clogging seemed to be more permanent. Only very high discharge events like in May 2010 resulted in the flushing of fine sediment as described by Schälchli (1992). The biological clogging seemed to be coupled to the algae succession that is dependent on the season, nutrient supply, hydrological regime as well as consumer population. Biological clogging was not as permanent as the physical clogging.

Consequences were also observed for hyporheic biogeochemical processes. Decreased advective exchange leads to lower input of solutes and surface water oxygen. Thus hyporheic aerobic turnover may be increasingly limited from up- to downstream. This is also supported by a decline of oxygen depletion from up to downstream. As fine particulate matter contained only 1.5 % organic carbon (data unpub.), microbial activity depended on another carbon source. Carbon could be easily proliferated from either the surface water at the upstream site or through the degradation of algae tissue at the site with observed biological clogging. Whereas these carbon sources were not accessible at the site with physical clogging. Denitrification seemed to take place where oxygen depletion exceeded advective input. Consequently, reduction of nitrate was highest for the "floodplain zone – biological clogging" site, followed by the "montane zone – natural" site. The biogeochemical active depth of the "floodplain zone – physical clogging" site was limited to the topmost 5 cm which could not be screened by the applied measurement techniques. As the system was rather limited by carbon sources only low activities may be assumed.

Findlay (1995) explained that the functional significance of the hyporheic zone for the whole stream ecosystem is dependent on the proportion of discharge through the subsurface and biogeochemical process rates. Process rates and biogeochemical activity of the hyporheic zone can only be described by profile analyses of DO, DOC and NO_3-N (like Claret et al. 1998). In the studied river reaches importance of hyporheic processes decreased with increasing flow length from high to moderate and low, whereas the loss of the hydrologic connectivity and biogeochemical regulation function was mediated by physical and biological clogging.

Additional to these two determinants the habitat function affects stream metabolism. Hofmann et al. (2011) assessed the ecological status using macroinvertebrate metrics. He observed a loss in biodiversity and shift in species composition within the middle reaches of the Khraa River, especially at the "floodplain zone – physical clogging" reach. The physical clogging had also an impact on habitat quality with long-term effects on the community structure (like Hellawell 1986; Bo et al. 2007). This would also reaffirm that the physical clogging was more permanent.

Further insights could be achieved through the application of the freeze panel technique for a better quantification of properties of the topmost sediment layer, the measurement of turbidity over time and the microbial activities. Models may help to analyse the systems behavior for different boundary conditions.

This study demonstrated the adverse effects of biological and physical clogging on hyporheic zone functions within a pristine environment. The biological clogging showed seasonal effects on the hydrologic connectivity between water column and pore water whereas biogeochemical regulation and habitat seemed to be affected less. Physical clogging revealed to have long-term impacts on the hydrologic connectivity, biogeochemical regulation and habitat as the fine sediment content within hyporheic interstices remained stable. Accordingly, the functional significance of the hyporheic zone for the stream ecosystem decreased to a moderate to low level. These clogging phenomena may be even more detrimental for ecosystem functions in regions with higher nutrient and suspended sediment loads.

3 Nitrogen Retention and Turnover at the Surface-Subsurface Interface of Riffle-Pool Sequences

3.1 Introduction

Excess nutrient loads have been recognized to be the major cause of various water quality problems in many estuaries and coastal waters of the world. Agriculture has been recognized in many regions as the main source of nitrogen emissions to the aquatic environment. As a result, there is a growing awareness that nutrient management must be handled at the catchment scale (Birgand et al. 2007).

Only very few numerical studies consider the influence of natural channel morphology and surface water elevations on flow and solute transport in the hyporheic zone. Saenger et al. (2005) conducted a numerical study on the exchange using a groundwater model for hyporheic flow (MODFLOW, MODPATH, MT3DMS) and a one dimensional surface water model (HECRAS) for surface water flow (see also Borchardt and Reichert 2001; Wawra et al. 2009; Vollmer et al. 2009). Lautz and Siegel (2006) used MODFLOW and MT3D to model surface groundwater mixing in the hyporheic zone around debris dams and meanders along a semi-arid stream. MT3D simulates both advective transport and sink/source mixing of solutes, in contrast to particle tracking (e.g. MODPATH), which only considers advection. Gooseff et al. (2006) used a two-dimensional groundwater flow and particle tracking models to simulate vertical and longitudinal hyporheic exchange along the longitudinal axis of streams of the second-, third, and fourth-order mountain reaches: Kasahara and Wondzell (2003) made an investigation on the hyporheic exchange flow and concluded that groundwater flow models allowed for effectively examining the morphologic features that controlled hyporheic exchange flow, and surface-visible channel morphologic features that controlled hyporheic zone in the mountain streams. Although several studies coupled longitudinal solute transport in stream with solute advection along a continues distribution of hyporheic flow paths (Wörman et al. 2002; Lautz and Siegel 2007) only very few studies use numerical modeling and reaction kinetics for solutes transport simulations (Gooseff et al. 2004). Most studies have used conceptual or empirical models to simulate denitrification at reach scale (Mulholland et al. 2004; Böhlke et al. 2004).

So far two-dimensional numerical models including reaction kinetics have not been used to assess the impact of river morphology on the nitrogen turnover in the hyporheic zone.

The objective of this study is to develop a coupled 2D hydraulic and hyporheic zone model including a first order nitrogen reaction module to analyze the hyporheic zone nitrogen turnover processes under varying morphological conditions. To accomplish these objectives the groundwater flow model MODFLOW and the reaction module RT3D was coupled with the hydrodynamic model TELEMAC. We compared model simulations with detailed measurements of tracer and nitrogen concentrations of the hyporheic flow in the third order stream Lahn (Germany).

3.2 Material and Methods

The River Lahn is a right-sided tributary to the middle reach of the River Rhine in Germany with a total length of 245 km. The study site is located 53 km downstream from the source and has a drainage area of 453 km^2 and a mean gradient of 2.36‰. The mean annual flow amounts to 7.3 m^3/s with a base flow of 0.567 m^3/s (annual precipitation: 810 mm). The river bed is 12–15 m wide and is characterized by a sequence of riffle-pool sections. The riffle under investigation (riffle B, Fig. 4) is situated 250 m downstream of the inlet of a wastewater treatment plant (WWTP) (Fischer et al. 2009).

Hydraulic head measurements were conducted at the Multi-Level Samplers in the middle of the stream with extraction depths of −0.5, −0.15, −0.25, and −0.45 m. At selected sites samples could be extracted from sediment depths of −0.55, −0.65 and −0.75 m. Methods and field measurements can be found in detail elsewhere (Saenger and Zanke 2009; Fischer et al. 2009).

Tracer experiments were carried out only at low and medium flow; at high flow the installation of the tracer extraction and head measurement equipment was not possible between 1997 and 1999. Fluoresceine was injected into the surface water via a Mariotte Vessel with a steady rate for 3–8 h. Samples were taken in the surface water above the multi level samplers (MLS) and in the subsurface via the MLS. We sampled all ports in the three cross sections (samplers V, VI, VII; see Fig. 4) every 30 min for the first 5 h of the experiments and then at longer intervals. The riverbed topography was surveyed in cross profiles about every 4 m in the summer of 1997, and its contour map is shown in Fig. 4 (Fischer et al. 2009; Ingendahl et al. 2009).

3.3 Model and Model Setup

To generate hydraulic heads the water surface elevation was generated using the TELEMAC hydrodynamic model. The processed bottom and water surface data were than exported to the groundwater flow model MODFLOW. MODFLOW was

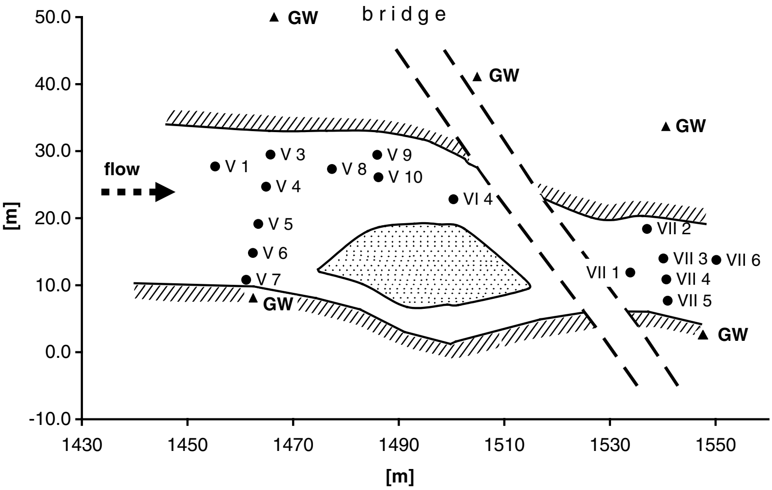

Fig. 4 Arrangement of multi-level samplers at riffle B (*V*, *VI*, *VII*: Transect *V* to *VII*; *GW* groundwater wells; *shaded area*: *gravel bar*)

used with its reaction module RT3D for the analysis. The first order reaction module was coded for NH4-NO3-N2 conversion process.

The flow and reactive transport model used solve the equations below using the FD method.

$$Flow \quad \frac{\partial}{\partial x_i}\left(K_{ij}\frac{\partial h}{\partial x_i}\right) = S_s\frac{\partial h}{\partial t} + W, \tag{1}$$

where K_{ij} [LT^{-1}] is the hydraulic conductivity of the porous media, h [L] is the hydraulic head, S_s [L^{-1}] the specific storage, t is the time [T], x_i is the spatial discretization tensor [L], and W [T^{-1}] is the volumetric flux per unit volume.

$$reactive\ \ Transport \quad \frac{\partial C_k}{\partial t} = \frac{\partial}{\partial}\left(D_{ij}\frac{\partial C_k}{\partial x_j}\right) - \frac{\partial}{\partial x_i}\left(v_i C_k\right) + \frac{q_s}{\phi}C_{s_k} + r_c, \tag{2}$$

where C_k is the aqueous phase concentration of kth species [ML^{-3}], D_{ij} is hydrodynamic dispersion coefficient [L^2T^{-1}], v_i is pore velocity [LT^{-1}], ϕ is the soil porosity, q_s is volumetric flux of water per unit volume of aquifer representing source and sinks [T^{-1}], C_s is concentration of source/sink [ML^{-3}], and r_c is the rate of all reactions that occur in the aqueous phase [ML^{-3} T^{-1}].

The first order reaction module integrated as r_c contains the nitrogen transformation;

$$r_{NH4} = -k_1 C_{NH4} \tag{3}$$

$$r_{NO3} = k_1 C_{NH4} - k_2 C_{NO3} \tag{4}$$

$$r_{N2} = k_2 C_{NO3} \tag{5}$$

Where C is the concentration of the constituents, and k_1 & k_2 are the reaction rates.

A river reach of 80 m length having various riffle-pool sequences and the same bottom bed volume was considered. A river package was used to model the source/sink term for the exchange between the surface and sub-surface (hyporheic-zone). Four layers were assumed each having the thickness of 4, 15, 15, and 15 cm. MODFLOW-RT3D solves flow and transport in the subsurface using a finite difference scheme for advection-dispersion-reaction equation. A conservative and a reactive tracer were simultaneously prescribed from upstream river cells. The necessary flow, transport and reactive parameters were defined within the range of literature values. Various initial and boundary conditions were analysed and numerical simulations were performed.

3.4 Results

Simulations results of the hydrodynamic TELEMAC model showed good results for water level calculations. The tracer's flow-through curves in the surface water and in the subsurface as well as the hydraulic heads were utilized to estimate the flow velocities in the subsurface. Figure 5 shows the through-flow curves for the different surface water flows. Although the tracer concentration decreased rapidly with sediment depth, tracer-laced surface water was detected up to a depth of 1 m (Saenger and Zanke 2009). It took the surface water longer to infiltrate deeper layers of sediment and the curves are wider. The measured breakthrough curves are shown for sampling point VI 4. Peak concentrations occurred after 10–14 h after tracer injection. Simulated tracer concentrations showed reasonable agreement with measured concentrations. The comparison of simulated and modeled NH_4-N and NO_3-N concentrations also showed a reasonable agreement for Transect V as well as for transect VII (Fig. 6). Regarding the fate of nitrogen compounds in the hyporheic zone a strong decrease in NH_4-N was observed in the upper sediment layers of both transects. Dilution by groundwater had only a minor influence on the decrease in solute concentrations in these sediment layers. This indicates that the measured vertical gradients in the River Lahn could be chiefly attributed to the conversion processes in the upper sediment layers (Fischer et al. 2009). Transformation was strongest for NH_4-N. The surface water concentration was reduced to 83.3 % in transect V and to 67.2 % in transect VII within the uppermost 0.15 m (Fischer et al. 2009). Because of the number of parameters involved and the limited available data calibrated parameter are associated with a considerable amount of uncertainty. Most

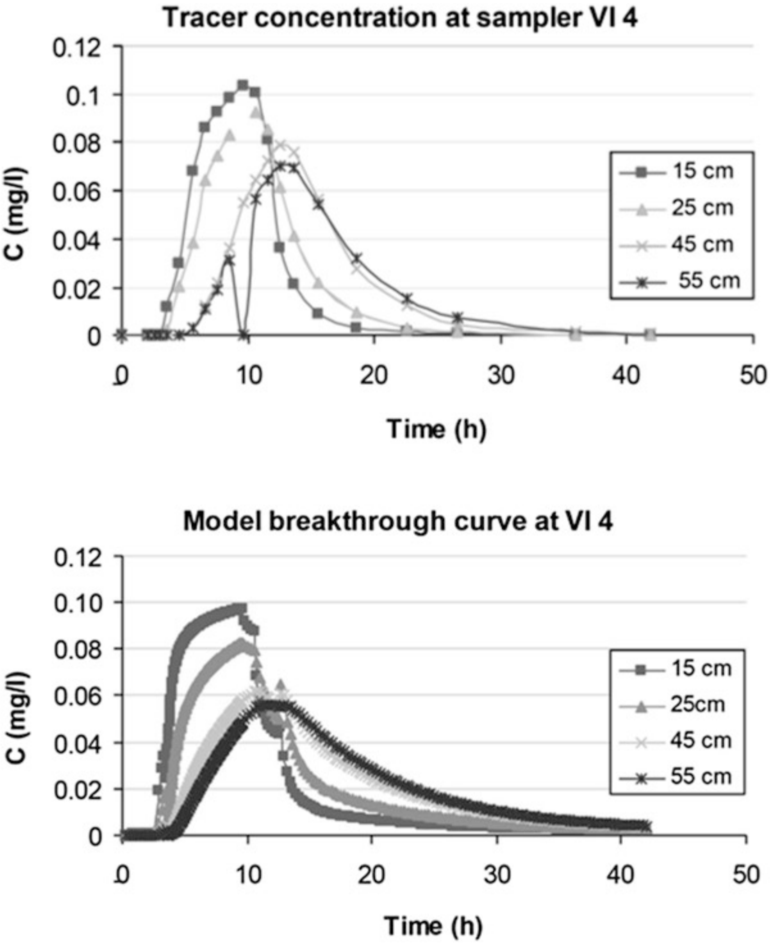

Fig. 5 Measured and modeled breakthrough curves

sensitive for nitrogen transport were the nitrogen reaction parameters. In general the flow parameters showed higher non-linearity than the transport and reaction parameters. The model is found to be highly sensitive to hydraulic conductivity, reaction parameters, specific storage, and longitudinal dispersivity, moderately sensitive to conductance, and slightly sensitive to specific yield and molecular dispersion coefficient. It was found that the beginning of the riffle is the location of the highest concentration gradient. The locations of highest hydraulic gradient as a result of morphological change (riffle) is the location of the highest water exchange and nitrogen transformation. In spite of uncertainties involved in the process-based models (data, parameters, and model structure) valuable conclusions can be made towards focused theoretical and experimental studies for new process understating.

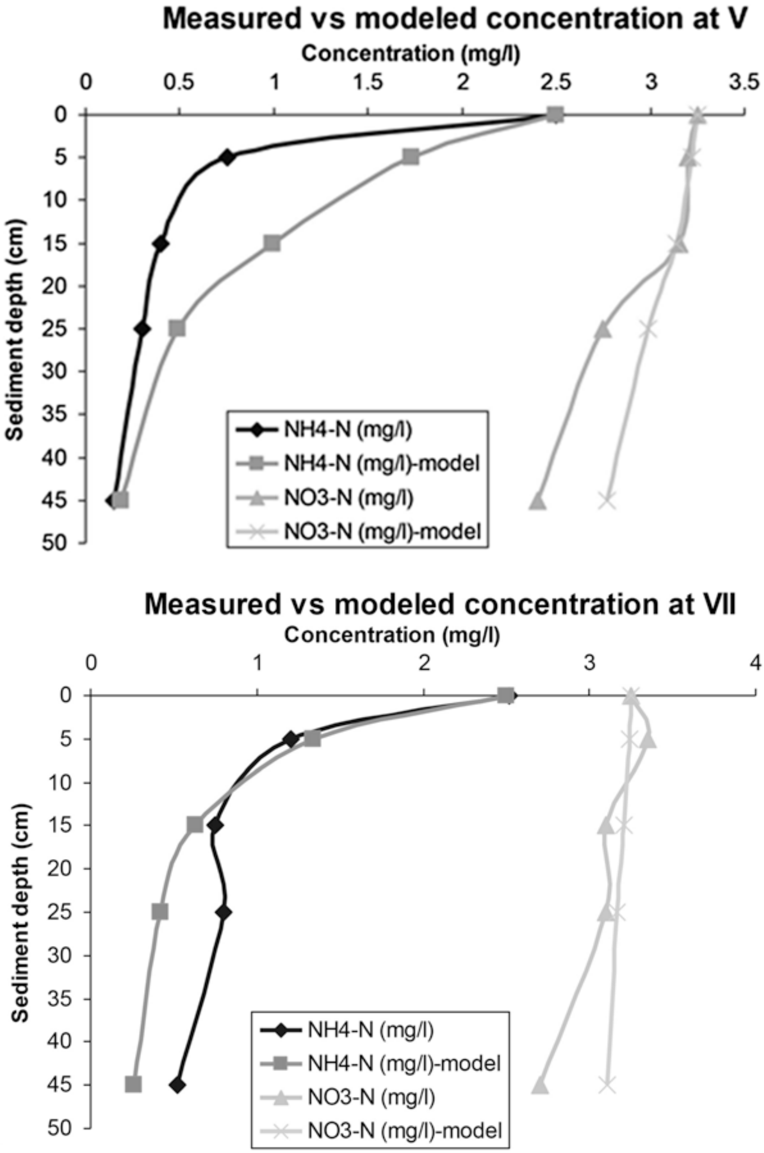

Fig. 6 Measured and modelled NH_4-N and NO_3-N concentrations in river sediments of sampling location V and VII at the Lahn study site

4 Spatial Variability of Nitrogen Removal in River Systems

4.1 Introduction

Nutrient retention processes and in-stream retention are mainly attributed to assimilation by suspended and benthic algae, uptake by macrophytes and denitrification. The load applied to catchments is substantially larger than the nitrogen losses to coastal areas, indicating an estimated river network N removal efficiency near 75 % (van Breemen et al. 2002). In small agricultural streams in-stream removal of nitrogen can range from 10 to 70 % of the total N load (Birgand et al. 2007). This important water purification process is distributed unevenly in space and time (McClain et al. 2003), with in-stream removal via assimilatory and dissimilatory pathways, and varying with environmental drivers (oxygen, discharge, light, temperature, organic matter concentrations) and stream order (Alexander et al. 2000; Peterson et al. 2001; Seitzinger et al. 2002). Only denitrification removes nitrogen permanently from the aquatic cycle, therefore it is one of the most important processes (Birgand et al. 2007; Wagenschein and Rode 2008; Whitehead et al. 2009; Mulholland et al. 2008).

Because nitrate transfer to the denitrifying and absorption sites is intrinsically linked to the mass transfer of water, it may vary at both spatial and temporal scales. The main factors controlling hyporheic exchange reported in the literature include hydraulic conductivity and porosity of the alluvium, velocity of the channel water, hydraulic gradient between riffle ends, height of bedforms of obstacles (Birgand et al. 2007).

Packman and Salehin (2003) and Packman et al. (2004) propose that an order-of-magnitude estimate of the hyporheic exchange rate can be obtained simply by considering the stream velocity and bed sediment size distribution. Therefore hydraulic conductivities, grain size distribution and stream bed geometries are together with stream velocities the most important factors controlling hyporheic exchange and hence nitrogen removal in streams. Because these factors together with nitrogen concentrations strongly change with flow length denitrification will also change with downstream flow length (Wagenschein and Rode 2008; Boyacioglu et al. 2012).

4.2 Material and Methods

4.2.1 Study Site

The Weisse Elster river basin is a subcatchment of the Saale River which is the second largest tributary of the Elbe River. The catchment area is about 5,300 km^2 and is mainly situated in the German States of Sachsen (Saxony), Thüringen (Thuringia) and Sachsen-Anhalt (Saxony-Anhalt), originating from Erzgebirge (Ore Mountains) in the Czech Republic (Fig. 7). The river is 250 km long and has a mean discharge

Fig. 7 Study catchment Weiße Elster, Germany

of 26 m^3/s (gauging station Oberthau). The river channel structure is very diverse with near-natural stretches as well as concrete-lined segments. Nutrient concentrations in the Weisse Elster River and its major tributaries given as 90-percentile of concentrations (2001) range between 6.1 and 13.0 mgN/l and 0.14 and 0.74 mgP/l. Diffuse sources have been estimated to have contributed to the overall nutrient load by 84 % (nitrogen) and 65 % (phosphorus). High NH_4 and PO_4 concentrations at the lowland river reaches are caused by high sewage inputs from urban areas (Rode et al. 2008).

4.2.2 Model and Model Set Up

The water quality model WASP5 (Water Quality Analysis Simulation Program) is a one- to three- dimensional numerical model and includes a deterministic approach to describe the hydrodynamics and the turnover of nutrients and chemicals in water column and sediments. It was developed at the U.S. Environmental Protection Agency (Ambrose et al. 1993). The WASP5 modelling system consists of three stand-alone computer programs, that can be run in conjunction or separately: DYNHYD is a hydrodynamic model, which is based on the Saint Venant equations; EUTRO can be used to model oxygen depletion, eutrophication, and nutrient enrichment in the river; and TOXI simulates the sediment transport and the fate of toxic inorganic and organic chemicals. In this study a modified version of DYNHYD (Warwick 1999) was used which allows the consideration of weirs. Also an extended version of EUTRO was applied (Shanahan and Alam 2001), which consists of nine model variables: biomass of phytoplankton (PHYT), biomass of periphyton (PERI), dissolved oxygen (DO), biochemical oxygen demand (BOD), ammonia nitrogen (NH4), nitrate nitrogen (NO3), organic nitrogen (ON), phosphate (PO4) and organic phosphorus (OP). The complex system of these variables is described by several processes, such as growth and decay of the autotrophic organisms, settling, reaeration, sediment oxygen demand, nitrification, denitrification and mineralization. In total up to 39 temperature coefficients and kinetic parameters are used in the EUTRO submodel.

The matter exchange between the water column and the hyporheic or benthic zone is considered in a simplified way. Denitrification of nitrate is calculated as a function of water depth, size of sediment area, a denitrification constant, temperature and the concentration of nitrate in the water column. Additional information about the latest model can be found in Wagenschein and Rode (2008). The main advantage of WASP5 compared to other water quality models is its flexibility as it offers with the possibility to build one-, two- or three-dimensional networks. Complex aquatic systems can be subdivided into lateral, vertical and longitudinal segments. Another advantage is the freely available source code of WASP5, which makes it possible to implement additional processes and components in the modelling system.

The Weiße Elster River water quality model (1D) was set up for the river section from Straßberg near Plauen to Großzschocher with a length of 145 km and model element size of 250 m. A detailed implementation of the model for investigating river morphological impacts on nutrient turnover was carried out for the river section between Gera and Großzschocher (70.6 km) using segment length of 100 m (872 river cross sections). Discharge and nutrient load input data were obtained from the water authorities and additional field campaigns. Point source data from sewage systems were directly used as inputs into the WASP5 model for the Weiße Elster river. Uncertainty analysis based on the Monte Carlo approach was carried out for the calibrated model. The WASP5 model was calibrated with the automatic parameter estimation tool PEST (Doherty 2004) using the eight most important parameters (Wagenschein and Rode 2008). The model was validated for the time period from May to October 2001 based on discharge and water quality data from environmental authorities (Rode et al. 2008). Based on modelled denitrification rates total nitrogen retention can be calculated for each single river section. Using yearly time series of discharge, temperature and nitrate concentrations the yearly nitrogen retention can be simulated with the model.

4.3 Results

Denitrification was calculated with the WASP5 model in the Weiße Elster between Straßberg and Großschochner. The results show that the denitrification rate is strongly related to morphological characteristics of the river and varies between 0.3 and 2.2 mg N/(L d). There is a decreasing trend of the per litre transformed nitrogen amount from upstream to downstream reaches (Fig. 8). This is caused by the decreasing relationship between wetted river cross section and wetted perimeter because the river depth is continuously increasing with increasing flow length. With regard to the optimization of possible management options, the removal of nitrogen per kilometre flow length is of importance. This removal depends not only on the denitrification rate, but also on nitrate concentration, which is slightly elevated in the downstream direction with highest levels of 7.5 mg N/L at river kilometre 100. Highest N-removal occurred at the lower part of the central river reach of the Weiße Elster. Denitrification is dominating total nitrogen removal and is considerably larger than nitrogen uptake by phytoplankton and periphyton. This is true also for downstream reaches (Wagenschein and Rode 2008). In the river section between Gera and Großzschochner (70.6 km) N-retention caused by denitrification was 20–32 % for summer low flow conditions (Wagenschein 2006). For the whole river section between Strassberg and Großzschocher yearly N-retention by denitrification was 19 %. Although the data of the validation period cover a wide range of concentration from 3.5 to 7.4 mg TN not the total range of nitrogen concentrations has been considered. Due to data constraints the model has not been tested for rare but extremely elevated concentrations that sometimes occur during intermediated flows, especially during the winter season when denitrification processes are likely to be less pronounced. This knowledge gap is not restricted to the present study.

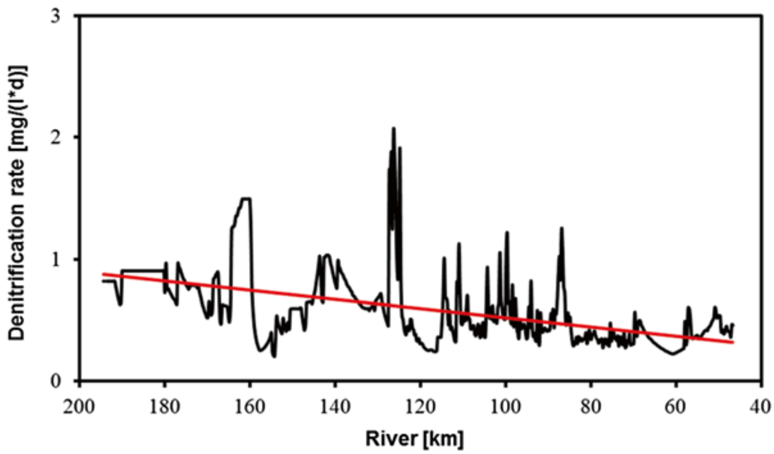

Fig. 8 Denitrification rate in the main stem of the Weiße Elster River between Strassberg and Großzschochner

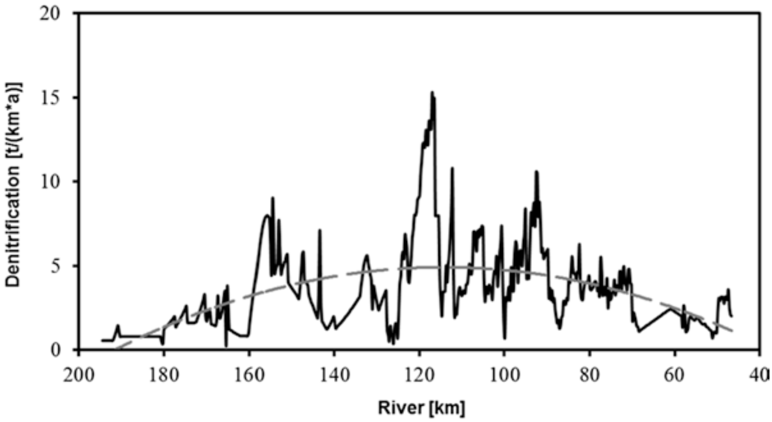

Fig. 9 Yearly denitrification in the main stem of the Weiße Elster River between Strassberg and Großzschochner based on WASP 5 calculations

In general this leads to considerable uncertainties which are particularly related to estimates of reaction rates covering the total variability of flows and site characteristics. Especially the substantial lack of nitrogen removal data for higher flow stages result in significant uncertainties in network scale total nitrogen loss estimates (Böhlke et al. 2009; Basu et al. 2011). Furthermore our model approach does not consider seasonally important phenomena such as litter fall or submerged aquatic vegetation phenology. However, given the high performance measures especially for nitrogen simulations (Wagenschein and Rode 2008) we believe that the results provide a reasonable indication of important controlling factors of N removal in the study river.

The Fig. 9 illustrates a general pattern of decreasing denitrification rates and changing nitrogen removal. The results show that anthropogenic impacts do not only determine the nitrogen loading of a given river but also impact indirectly the nitrogen removal. These alterations are typically not homogenously distributed within a catchment and are not restricted to agricultural inputs only but may also include atmospheric depositions. Also urban inputs may have impact on nitrogen concentration especially during low flow conditions and therefore can modulate nitrogen removal processes.

4.4 Discussion

Compared to empirical models of nutrient retention the WASP5 model is more complex and therefore does also allow to consider the impact of river morphology as well as the impact of primary production on nitrate uptake. Using the 1D hydraulic model WASP5 is able to consider flow velocities and hence to compute the residence time of water during different flow stages. Our experiences with 1D hydraulic modelling gives us high confidence in such modelling approach. Furthermore 1D hydraulic modelling is well accepted in science and engineering (Lindenschmidt et al. 2005; Buttner et al. 2006; Rode et al. 2007). The uncertainties associated with model results do not necessarily increase with increasing model complexity. The incorporation of additional mechanism may improve the confidence in predictions made for a variety of conditions if model parameters can be well identified (Rode et al. 2010). In another study (Wagenschein 2006) we conducted a detailed uncertainty analyses on selected model output variables and showed that nitrate calculations had the lowest uncertainties of all output variables. There exist also simpler model approaches to simulate nitrate transport in river systems (Mulholland et al. 2008; Wollheim et al. 2008; Alexander et al. 2009; Hall et al. 2009; Covina et al. 2012; Ye et al. 2012). These approaches use empirical equations derived from experimental studies. In recent years these empirical approaches for simulation of in-stream nitrogen retention tend to become more complex because they also try to include nitrogen uptake by primary production (using the Michaelis-Menten kinetic) and the impact of stream morphology on denitrification. Recently also temporally variable denitrification rates have been computed (Alexander et al. 2009). This development shows that there is a need to increase the number of factors controlling nitrogen retention in natural rivers. The uncertainties of "simpler" models may at least not deviate very much from uncertainties of so called "complex" or mechanistic models, but mechanistic models allow a sound consideration of river hydraulics. The main problem of both approaches is the lack of data (in our case for the cold season). Our on-going work therefore focuses on continues UV sensor measurements of nutrients (temporal resolution 10 min) which allow a more detailed temporal analysis including diurnal concentration changes.

4.5 Conclusions

Denitrification is the largest sink of inorganic nitrogen. In the present study we could show that it exceeds the assimilation by phytoplankton and periphyton more than three times. Nitrogen retention by denitrification varies significantly along the modelled river section (Wagenschein and Rode 2008). The sinuosity is the most sensitive morphological feature. The impact of river structure restoration on nitrogen retention was small. The impact of river structure restoration on inorganic nitrogen concentrations might be larger for smaller rivers, because of the more intensive contact to the interstitial sediments. Future work should quantify the effects of seasonal changes (temperature, light intensity) on nitrogen retention. There is a strong need to further improve the modelling of denitrification processes. We emphasize the need of variable denitrification rates which depend on the sediment characteristics and the hydraulic exchange. These site-specific characteristics depend mainly on variations of the hydraulic radius, i.e. on river width and depth, flow velocity and channel slope. The definition of empirical relationships between these known features and the denitrification rate would be a reasonable approach for future modelling of in-stream denitrification (Wagenschein and Rode 2008).

The model results indicate a decreasing denitrification rate with increasing river length, but river morphology may modulate this general trend considerably. Furthermore nitrate concentration affects the nitrogen removal significantly which in combination with morphological characteristics led to highest nitrogen removal in the central section of the 4th order river. Because nitrogen concentrations and river morphology often show comparable properties in intensely used meso-scale catchments this longitudinal pattern of nitrogen removal can be assumed to be typical in highly eutrophic low order rives of central Europe.

The impact of river morphology on nitrogen removal can also be accelerated by climate change. The study of Boyacioglu et al. (2012) suggests a future change in discharge may have a larger impact on denitrification rates than future temperature changes although the relative sensitivity of temperature was higher than of discharge. Changing future discharge modifies the relationship between wetted river cross section and wetted perimeter and therefore affects the denitrification rate. These findings are restricted to mid-sized rivers in temperate climate which may be highly affected by decreasing discharge. Thus, denitrification in rivers with less future discharge variations may be less affected by climate change.

References

Adams JN, Beschta RL (1980) Gravel bed composition in Oregon coastal streams. Can J Fish Aquat Sci 37(10):1514–1521

Alexander RB, Smith RA, Schwarz GE (2000) Effect of stream channel size on the delivery of nitrogen to the Gulf of Mexico. Nature 403:758–761

Alexander RB, Bohlke JK, Boyer EW, David MB, Harvey JW, Mulholland PJ, Seitzinger SP, Tobias CR, Tonitto C, Wollheim WM (2009) Dynamic modeling of nitrogen losses in river networks unravels the coupled effects of hydrological and biogeochemical processes. Biogeochemistry 93:91–116

Ambrose RB, Wool TA, Martin JL (1993) The Water Quality Simulation Program, WASP5: model theory, user's manual, and programmer's guide. U.S. Environmental Protection Agency, Athens

Arnon S, Marx LP, Searcy KE, Packman AI (2010) Effects of overlying velocity, particle size, and biofilm growth on stream–subsurface exchange of particles. Hydrol Process 24(1):108–114

Basu NB, Rao PSC, Thompson SE, Loukinova NV, Donner SD, Ye S, Sivapalan M (2011) Spatiotemporal averaging of in-stream solute removal dynamics. Water Resour Res 47

Battin TJ, Sengschmitt D (1999) Linking sediment biofilms, hydrodynamics, and river bed clogging: evidence from a large river. Microb Ecol 37:185–196

Bencala KE (1993) A perspective on stream-catchment connections. J North Am Benthol Soc 12:44–47

Bencala KE (2000) Hyporheic zone hydrological processes. Hydrol Process 14:2797–2798

Bencala KE (2011) 2.20 – Stream–Groundwater interactions. In: Editor-in-Chief: Peter W (ed) Treatise on water science. Elsevier, Oxford, pp 537–546

Beyer W, Banscher E (1975) Colmation of river beds when obtaining bank filtrate. Zeitschrift Fur Angewandte Geologie 21:565–570

Birgand F, Skaggs RW, Chescheir GM, Gilliam JW (2007) Nitrogen removal in streams of agricultural catchments – a literature review. Crit Rev Environ Sci Technol 37:381–487
Bo T, Fenoglio S, Malacarne G, Pessino M, Sgariboldi F (2007) Effects of clogging on stream macroinvertebrates: an experimental approach. Limnologica 37(2):186–192
Bohlke JK, Harvey JW, Voytek MA (2004) Reach-scale isotope tracer experiment to quantify denitrification and related processes in a nitrate-rich stream, midcontinent United States. Limnol Oceanogr 49:821–838
Bohlke JK, Antweiler RC, Harvey JW, Laursen AE, Smith LK, Smith RL, Voytek MA (2009) Multi-scale measurements and modeling of denitrification in streams with varying flow and nitrate concentration in the upper Mississippi River basin, USA. Biogeochemistry 93:117–141
Borchardt D, Pusch M (2009) The ecology of the hyporheic zone of running waters: patterns, processes and bottleneck functions. Adv Limnol 61:224
Borchardt D, Reichert P (2001) River water quality model No. 1: compartmentalisation approach applied to oxygen balances in the RIVER LAHN. Wat Sci Technol 43(5):41–50
Boulton AJ, Findlay S, Marmonier P, Stanley EH, Valett HM (1998) The functional significance of the hyporheic zone in streams and rivers. Annu Rev Ecol Syst 29:59–81
Boyacioglu H, Vetter T, Krysanova V, Rode M (2012) Modeling the impacts of climate change on nitrogen retention in a 4th order stream. Clim Change 113:981–999
Bretschko G, Klemens W (1986) Quantitative methods and aspects in the study of the interstitial fauna of running waters. Stygologia 2:279–316
Brunke M, Gonser T (1997) The ecological significance of exchange processes between rivers and groundwater. Freshw Biol 37:1–33
Buttner O, Otte-Witte K, Kruger F, Meon G, Rode M (2006) Numerical modelling of floodplain hydraulics and suspended sediment transport and deposition at the event scale in the middle river Elbe, Germany. Acta Hydrochimica Et Hydrobiologica 34:265–278
Cairns JR (1977) Quantification of biological integrity. In: Ballentine PK, Guarraia KJ (eds) The integrity of water. U.S. Government Printing Office, Washington, DC, pp 171–187
Cardenas MB, Wilson JL (2006) The influence of ambient groundwater discharge on exchange zones induced by current-bedform interactions. J Hydrol 331:103–109
Cardenas MB, Wilson JL, Zlotnik VA (2004) Impact of heterogeneity, bed forms, and stream curvature on subchannel hyporheic exchange. Water Resour Res 40
Claret C, Marmonier P, Bravard JP (1998) Seasonal dynamics of nutrient and biofilm in interstitial habitats of two contrasting riffles in a regulated large river. Aquat Sci 60:33–55
Covino T, McGlynn B, McNamara R (2012) Land use/land cover and scale influences on in-stream nitrogen uptake kinetics. J Geophys Res Biogeosci 117
Doherty J (2004) PEST – model independent parameter estimation, 5th edition of user manual. Watermark Numerical Computing, Brisbane
Duff JH, Triska FJ (1990) Denitrification in sediments from the hyporheic zone adjacent to a small forested stream. Can J Fish Aquat Sci 47:1140–1147
Findlay S (1995) Importance of surface-subsurface exchange in stream ecosystems: the hyporheic zone. Limnol Oceanogr 40(1):159–164
Fischer J, Borchardt D, Ingendahl D, Ibisch RB, Sänger N, Wawra B, Lenk M (2009) Vertical gradients of nutrients in the hyporheic zone of the River Lahn (Germany): relevance of surface versus hyporheic conversion processes. Adv Limnol 61:105–118
Gibert J, Dole-Olivier MJ, Marmonier P, Vervier P (1990) Surface water-groundwater ecotones. In: Naiman RJ, Décamps (eds) The ecology and management of aquatic and terrestrial ecotones. UNESCO MAB and the Parthenon Publishing Group, pp 199–216
Gooseff MN, McKnight DM, Runkel RL, Duff JH (2004) Denitrification and hydrologic transient storage in a glacial meltwater stream, McMurdo Dry Valleys, Antarctica. Limnol Oceanogr 49:1884–1895
Gooseff MN, Anderson JK, Wondzell SM, LaNier J, Haggerty R (2006) A modelling study of hyporheic exchange pattern and the sequence, size, and spacing of stream bedforms in mountain stream networks, Oregon, USA (Retracted article. See vol 20, p 2441, 2006). Hydrol Process 20:2443–2457

Graham A (1990) Siltation of stone-surface periphyton in rivers by clay-sized particles from low concentrations in suspension. Hydrobiologia 199:107–115

Greig S, Sear D, Carling P (2007) A field-based assessment of oxygen supply to incubating Atlantic salmon (Salmo salar) embryos. Hydrol Process 21:3087–3100

Grimm NB, Fisher SG (1984) Exchange between interstitial and surface water: implications for stream metabolism and nutrient cycling. Hydrobiologia 111:219–228

Hall RO, Baker MA, Arp CD, Koch BJ (2009) Hydrologic control of nitrogen removal, storage, and export in a mountain stream. Limnol Oceanogr 54(6):2128–2142

Hartwig M, Theuring P, Rode M, Borchardt D (2012) Suspended sediments in the Kharaa River catchment (Mongolia) and its impact on hyporheic zone functions. Environ Earth Sci 65:1535–1546

Harvey JW, Drummond JD, Martin RL, McPhillips LE, Packman AI, Jerolmack DJ, Stonedahl SH, Aubeneau AF, Sawyer AH, Larsen LG, Tobias CR (2012) Hydrogeomorphology of the hyporheic zone: stream solute and fine particle interactions with a dynamic streambed. J Geophys Res 117(G4):2156–2202

Hellawell JM (1986) Biological indicators of freshwater pollution and environmental management. Pollution monitoring series, Mellanby K (ed). Elsevier Applied Sciences Publishers, London, 546 pp

Hesser FB, Franko U, Rode M (2010) Spatially distributed lateral nitrate transport at the catchment scale. J Environ Qual 39(1):193–203

Hofmann J, Hürdler J, Ibisch R, Schaeffer M, Borchardt D (2011) Analysis of recent nutrient emission pathways, resulting surface water quality and ecological impacts under extreme continental climate: the Kharaa River basin (Mongolia). Int Rev Hydrobiol 96:484–519

Ibisch R, Seydell I, Borchardt D (2009) Influence of periphyton biomass dynamics on biological colmation processes in the hyporheic zone of a gravel bed river (River Lahn, Germany). Adv Limnol 61:87–104

Ingendahl D, Borchardt D, Saenger N, Reichert P (2009) Vertical hydraulic exchange and the contribution of hyporheic community respiration to whole ecosystem respiration in the River Lahn (Germany). Aquat Sci 71(4):399–410

Jones JB Jr, Fisher SG, Grimm NB (1995) Nitrification in the hyporheic zone of a desert stream ecosystem. J N Am Benthol Soc 14(2):249–258

Kasahara T, Wondzell SM (2003) Geomorphic controls on hyporheic exchange flow in mountain streams. Water Resour Res 39

Keery J, Binley A, Crook N, Smith JWN (2007) Temporal and spatial variability of groundwater-surface water fluxes: development and application of an analytical method using temperature time series. J Hydrol 336:1–16

Lautz LK, Siegel DI (2006) Modeling surface and groundwater mixing in the hyporheic zone using MODFLOW and MT3D. Adv Water Resour 26:1618–1633

Lautz LK, Siegel DI (2007) The effect of transient storage on nitrate uptake lengths in streams: an inter-site comparison. Hydrol Process 21:3533–3548

Lefebvre S, Marmonier P, Pinay G, Bour O, Aquilina L, Baudry J (2005) Nutrient dynamics in interstitial habitats of low-order rural streams with different bedrock geology. Arch Hydrobiol 164:169–191

Lenk M, Saenger N, Traebing K, Killich F, Fischer J, Borchardt D (1999) Chapter Bp09: Die Multi-Level-Interstitialsonde: Methode zur tiefenorientierten Untersuchung von Gewässersedimenten. In: DVWK Schriften, 128th edn. The German Water Federation : Dtsch. Verb. Wasserwirtsch. Kulturbau (DVWK), pp 150–157

Lindenschmidt KE, Poser K, Rode M (2005) Impact of morphological parameters on water quality variables of a regulated lowland river. Water Sci Technol 52:187–193

McClain ME, Boyer EW, Dent CL, Gergel SE, Grimm NB, Groffman PM, Hart SC, Harvey JW, Johnston CA, Mayorga E, McDowell WH, Pinay G (2003) Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems. Ecosystems 6:301–312

Meyer JL, McDowell WH, Bott TL, Elwood JW, Ishizaki C, Melack JM, Peckarsky BL, Peterson BJ, Rublee PA (1988) Elemental dynamics in streams. J N Am Benthol Soc 7:410–432

Mulholland PJ, Valett HM, Webster JR, Thomas SA, Cooper LW, Hamilton SK, Peterson BJ (2004) Stream denitrification and total nitrate uptake rates measured using a field N-15 tracer addition approach. Limnol Oceanogr 49:809–820

Mulholland PJ, Helton AM, Poole GC, Hall RO, Hamilton SK, Peterson BJ, Tank JL, Ashkenas LR, Cooper LW, Dahm CN, Dodds WK, Findlay SEG, Gregory SV, Grimm NB, Johnson SL, McDowell WH, Meyer JL, Valett HM, Webster JR, Arango CP, Beaulieu JJ, Bernot MJ, Burgin AJ, Crenshaw CL, Johnson LT, Niederlehner BR, O'Brien JM, Potter JD, Sheibley RW, Sobota DJ, Thomas SM (2008) Stream denitrification across biomes and its response to anthropogenic nitrate loading. Nature 452:202–205

Naegeli MW, Uehlinger U (1997) Contribution of the hyporheic zone to ecosystem metabolism in a prealpine gravelbed-river. J N Am Benthol Soc 16(4):794–804

Naiman RJ, Décamps H, Pastor J, Johnston CA (1988) The potential importance of boundaries of fluvial ecosystems. J N Am Benthol Soc 7(4):289–306

Nogaro G, Datry T, Mermillod-Blondin F, Descloux S, Montuelle B (2010) Influence of streambed sediment clogging on microbial processes in the hyporheic zone. Freshw Biol 5:1288–1302

Orghidan T (1959) Ein neuer Lebensraum des unterirdischen Wassers, de hyporheische Biotop. Arch Hydrobiol 55:392–414

Packman AI, MacKay JS (2003) Interplay of stream-subsurface exchange, clay particle deposition, and streambed evolution. Water Resour Res 39(4):1944–7973

Packman AI, Salehin M (2003) Relative roles of stream flow and sedimentary conditions in controlling hyporheic exchange. Hydrobiologia 494:291–297

Packman A, Salehin M, Zaramella M (2004) Hyporheic exchange with gravel beds: basic hydrodynamic interactions and bedform-induced advective flows. J Hydraul Eng-ASCE 130:647–656

Peterson BJ, Wollheim WM, Mulholland PJ, Webster JR, Meyer JL, Tank JL, Marti E, Bowden WB, Valett HM, Hershey AE, McDowell WH, Dodds WK, Hamilton SK, Gregory S, Morrall DD (2001) Control of nitrogen export from watersheds by headwater streams. Science 292:86–90

Rode M, Suhr U, Wriedt G (2007) Multi-objective calibration of a river water quality model – information content of calibration data. Ecol Model 204:129–142

Rode M, Klauer B, Petry D, Volk M, Wenk G, Wagenschein D (2008) Integrated nutrient transport modelling with respect to the implementation of the European WFD: the Weisse Elster Case Study, Germany. Water SA 34:490–496

Rode M, Arhonditsis G, Balin D, Kebede T, Krysanova V, van Griensven A, van der Zee S (2010) New challenges in integrated water quality modelling. Hydrol Process 24:3447–3461

Runkel RL, McKnight DM, Rajaram H (2003) Modeling hyporheic zone processes – preface. Adv Water Resour 26:901–905

Saenger N, Zanke UCE (2009) A depth – oriented view of hydraulic exchange patterns between surface water and the hyporheic zone: analysis of field experiments at the River Lahn (Germany). – Fundamental and applied limnology. Adv Limnol 61:9–27

Saenger N, Kitanidis PK, Street RL (2005) A numerical study of surface-subsurface exchange processes at a riffle-pool pair in the Lahn River, Germany. Water Resour Res 41

Salehin M, Packman AI, Paradis M (2004) Hyporheic exchange with heterogeneous streambeds: laboratory experiments and modeling. Water Resour Res 40

Schälchli U (1992) The clogging of coarse gravel river beds by fine sediment. Hydrobiologia 235:189–197

Schälchli U (1995) Basic equations for siltation of riverbeds. J Hydraul Eng 121(3):274–287

Seitzinger SP, Styles RV, Boyer EW, Alexander RB, Billen G, Howarth RW, Mayer B, Van Breemen N (2002) Nitrogen retention in rivers: model development and application to watersheds in the northeastern USA. Biogeochemistry 57:199–237

Shanahan P, Alam MM (2001) WASP Version 5.2. Maine Department of Environmental Protection, US

Stanford JA, Ward JV (1988) The hyporheic habitat of river ecosystems. Nature 335:64–66

Stream Solute Workshop (1990) Concepts and methods for assessing solute dynamics in stream ecosystems. J North Am Benthol Soc 9:95–119

Swanson TE, Cardenas MB (2011) Ex-Stream: a MATLAB program for calculating fluid flux through sediment water interfaces based on steady and transient temperature profiles. Comput Geosci 37:1664–1669

Theuring P, Rode M, Behrens S, Kirchner G, Jha A (2013) Identification of fluvial sediment sources in the Kharaa River catchment, Northern Mongolia. Hydrol Process 27:845–856

Triska FJ, Kennedy VC, Avanzino RJ, Zellweger GW, Bencala KE (1989) Retention and transport of nutrients in a third-order stream in Northwestern California: hyporheic processes. Ecology 70(6):1893–1905

Triska FJ, Duff JH, Avanzino (1993) The role of water exchange between a stream channel and its hyporheic zone in nitrogen cycling at the terrestrial-aquatic interface. Hydrobiologica 251:167–184

Valett HM, Fisher SG, Grimm NB, Camill P (1994) Vertical hydrologic exchange and ecological stability of a desert stream ecosystem. Ecology 75:548–560

Van Breemen N, Boyer EW, Goodale CL, Jaworski NA, Paustian K, Seitzinger SP, Lajtha K, Mayer B, Van Dam D, Howarth RW, Nadelhoffer KJ, Eve M, Billen G (2002) Where did all the nitrogen go? Fate of nitrogen inputs to large watersheds in the northeastern USA. Biogeochemistry 57:267–293

Vollmer S, Träbing K, Nestmann F (2009) Hydraulic exchange processes between surface and subsurface water: determination of spatial and temporal variability. Adv Limnol 61:45–65

Vukovic M, Soro A (1992) Determination of hydraulic conductivity of porous media from grain-size composition. Water Resources Publications, Littleton

Wagenschein D (2006) Einfluß der Gewässermorphologie auf die Näherstoffretention-Modellstudie am Beispiel der mittlere Weißen Elster. Dissertation an der Fakultät der Umweltwissenschaften und Verfahrenstechnik der Brandenburgischen Technischen Universität

Wagenschein D, Rode M (2008) Modelling the impact of river morphology on nitrogen retention – a case study of the Weisse Elster River (Germany). Ecol Model 211:224–232

Warwick JJ (1999) Enhancement of the hydrodynamic model DYNHYD. Commented source code

Wawra B, Saenger N, Montenegro H, Zanke UCE (2009) Driving forces for solute transport in the hyporheic zone of the River Lahn (Germany): numerical simulations. Adv Limnol 61:29–44

Whitehead PG, Wilby RL, Battarbee RW, Kernan M, Wade AJ (2009) A review of the potential impacts of climate change on surface water quality. Hydrol Sci J-Journal Des Sciences Hydrologiques 54:101–123

Williams DD, Hynes HBN (1974) The occurrence of benthos deep in the substratum of a stream. Freshw Biol 4:233–256

Wollheim WM, Vorosmarty CJ, Bouwman AF, Green P, Harrison J, Linder E, Peterson BJ, Seitzinger SP, Syvitski JPM (2008) Global N removal by freshwater aquatic systems using a spatially distributed, within-basin approach. Global Biogeochem Cycles 22, GB2026, 1–14

Wondzell SM (2006) Effect of morphology and discharge on hyporheic exchange flows in two small streams in the Cascade Mountains of Oregon, USA. Hydrol Process 20:267–287

Wood PJ, Armitage PD (1997) Biological effects of fine sediment in the lotic environment. Environ Manage 21:203–217

Wörman A, Packman AI, Johansson H, Jonsson K (2002) Effect of flow-induced exchange in hyporheic zones on longitudinal transport of solutes in streams and rivers. Water Resour Res 38(1) WR000769

Ye S et al (2012) Dissolved nutrient retention dynamics in river networks: A modeling investigation of transient flows and scale effects. Water Resour Res 48(6), W00J17

Marine and Coastal Ecosystems: Delivery of Goods and Services, Through Sustainable Use and Conservation

Angel Borja, Arantza Murillas-Maza, Marta Pascual, and María C. Uyarra

Abstract Despite the important role of ecosystem services, their study and associated monetary value is limited mostly to terrestrial, estuarine and coastal systems, with few studies undertaken in open marine waters and deep water systems. In addition, human activities are degrading marine ecosystems and the services they provide. To reverse this situation, various legislations have been implemented worldwide. Within this context, conservation activities (i.e. protection, prevention and restoration) are strongly encouraged. Hence, this Chapter: (i) reviews the marine goods and services provided by marine ecosystems; (ii) reviews conservation activities (as a means to maintain and improve goods and services), paying special attention to the effects of restoration on ecosystem services; and (iii) determines the gaps to be covered and the ways to move forward, in relation to conservation of marine goods and services.

Keywords Ecosystem services • Conservation • Protection • Restoration • Monetary assessment • Valuation

1 Introduction

Marine waters host some of the most valuable and biologically-diverse ecosystems on Earth (Costanza et al. 1997). Occupying 71 % of the Earth's surface and 97.5 % of the total volume of the Earth's waters, marine waters are home to a large number of species (between 0.7 and 1 million eukaryote species (Appeltans et al. 2012)),

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and provide numerous goods and services to millions of people (Pearce 1998; Barbier et al. 2012; de Groot et al. 2012). Some of these goods and services provided by marine waters and ecosystems include: biogeochemical services (e.g. carbon sequestration); nutrient cycling; coastal protection (e.g. as provided by coral reefs); food provision; sources of pharmaceutical products; recreational grounds; and species for the aquarium trade (Costanza et al. 1997; Moberg and Folke 1999; Beaumont et al. 2007). Despite the important role of such goods and services, their study and their associated monetary value is limited mostly to estuarine and coastal systems (Rönnbäck et al. 2007; Lange and Jiddawi 2009; van den Belt and Costanza 2012). Only a few recent studies have been published on goods and services, provided by open marine waters and deep water systems (Murillas-Maza et al. 2011; Armstrong et al. 2012; Salomidi et al. 2012; de Groot et al. 2012).

It is not surprising that most published studies on marine goods and services focus upon coastal areas. Forty-five percent of the global population live within 10 % of what it is defined as 'coastal land' (Mee 2012), whilst the average population density within the first 50 km from the coastline is 2.5 times the global average (Crossland et al. 2005). More than 500 million people depend directly upon fish for living (i.e. subsistence and livelihoods) (MEA 2005). Despite the high dependence of people on marine goods and services, degradation of coastal and marine ecosystems due to human activities (e.g. fishing, pollution discharges, tourism, shipping, aquaculture, marine renewables, etc.) continues to increase, which is leading to biodiversity loss and subsequent reduction in the provision of goods and services (MEA 2005; Halpern et al. 2008; Butchart et al. 2010; Lotze 2010; Bullock et al. 2011).

On a global scale, 50 % of salt marshes, 35 % of mangroves, 30 % of coral reefs and 29 % of seagrasses have been lost already, or degraded (Barbier et al. 2012). However, studies carried out over more specific ecosystems and/or at local scales, have shown degrading rates that are even more extreme (Wilkinson 2004). For example, 60 % of coral reefs world-wide are classified as being under some kind of local threat (e.g. overfishing, pollution). In the Caribbean and Indo-Pacific regions, more than 50 % of coral reefs have been lost (Gardner et al. 2003; Mora 2008). The first people to be affected by the declining conditions of marine waters and ecosystems are those that benefit most directly from them this happens often in developing countries. Therefore, the reverse of degradation trends of marine waters and ecosystems must be encouraged. This is to sustain and improve the goods and services that are provided.

Reversing degradation trends requires understanding, initially, the pressures acting throughout the system under investigation. However, both natural (e.g. volcanos, hurricanes) and anthropogenic (e.g. fishing, tourism, pollution) stressors and pressures often coexist, making it difficult to identify their associated impacts (Borja et al. 2012a). Since natural pressures are difficult to manage (although we can adapt to them or mitigate them, Heckbert et al. (2012)), it may be worthwhile more realistic to focus upon anthropogenic pressures, in order to maintain and improve the goods and services provided by marine waters and ecosystems. Indeed, the following initiatives have been arisen to: (i) protect (e.g. the United Nations Convention on

Law of the Sea (UNCLOS 1982), the Convention of Biological Diversity (CBD 1992)); (ii) understand (e.g. the Millennium Development Goals (MDG 2000), the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES 2008)); and (iii) know (e.g. Millennium Ecosystem Assessment (MEA 2005), the Economics of Ecosystems and Biodiversity (TEEB, ten Brink et al. 2009)) the terrestrial and marine biodiversity and the goods and services that ecosystems provide. In addition, world-wide legislation is seeking to develop, under an ecosystem-based approach strategies to conserve and manage the marine environment (for an overview, see Borja et al. 2008). These strategies incorporate three steps: (i) to protect what exists; (ii) to restore what has been damaged; and (iii) to prevent from future harm. The main Directives in Europe, dealing with these aspects, are the Habitats Directive (92/43/EEC), the Water Framework Directive (WFD, 2000/60/EC) and the Marine Strategy Framework Directive (MSFD, 2008/56/EC). Similarly, the Clean Water Act and the Oceans Policy, exist in the USA. In addition, several international conventions apply these aspects at regional sea level (e.g. the Oslo-Paris Convention, in the Atlantic; the Helsinki Convention, in the Baltic; the Barcelona Convention, in the Mediterranean; and the Bucharest Convention, in the Black Sea).

Overall, the objectives of these legislative initiatives are to protect and/or restore the corresponding seas, by ensuring that human activities are carried out in a sustainable manner, to provide safe, clean, healthy and productive marine waters. In summary, they try to promote the sustainable use of the seas, whilst conserving marine ecosystems. Hence, the main objective of these legislative measures and policies (e.g. in the MSFD) is to maintain the good environmental status of marine waters, habitats and resources, or restore them, when good status is not achieved (Borja et al. 2008).

The MSFD highlights the importance of developing an initial assessment of the present ecological status, towards achieving or maintaining the required status. This Directive requires also the inclusion of socio-economic analyses in the initial assessment, as well as analyses of pressures and impacts; the aim is the management of marine waters, following an ecosystem approach (Bertram and Rehdanz 2013). Although there are several methodologies to undertake economic analyses, the informal European Working Group on Economic and Social Assessment offers guidance, in relation to two possible approaches: (i) the ecosystem services approach; and (ii) the marine water accounts approach, which considers, from a problem-based approach, maritime sectors developing their activity within the marine environment.

Within this political context, conservation activities (i.e. protection, prevention and restoration) are recommended strongly (Elliott et al. 2007). Although protection of all levels of biodiversity (e.g. genetic, species, habitats, ecosystems, functional, etc.), adequate management of marine waters (i.e. ecological status) and preventive measures (e.g. to avoid, or reduce, the introduction of invasive species), are the first and the most important steps in maintaining and improving marine-related goods and services, restoration is gaining in importance as marine systems continue to degrade (Young 2000; Simenstad et al. 2006). Restoration measures may occur too late; therefore, there is a risk to rely too much on restoration capacities. Conversely, restoration activities in degraded marine systems can contribute potentially to the improvement of human livelihoods, by enhancing biodiversity and ecosystem services (Palmer and Filoso 2009). Hence, restoration of habitats and ecosystems is having an increasingly central influence on global environmental policy (Day et al. 2009; Nellemann and Corcoran 2010). Accordingly, ecological restoration programs are increasing world-wide, whilst restoration ecology (the science behind the programs) has gained recognition over the last decade (Hobbs 2007; Roberts et al. 2009; Aronson et al. 2010).

Within this context a three-fold objective has been established for this Chapter: (i) a review of the marine goods and services presented in the literature, as well as their economic valuations; (ii) a review of conservation activities (as a means to maintain and improve goods and services), paying special attention to the effects of restoration on ecosystem services; and (iii) the determination of gaps to be covered and the ways to progress, in relation to the conservation of marine goods and services. It should be noted that, whilst the terms "goods and services" and "ecosystem services" are often used in an interchangeable manner throughout the Chapter, in both cases we refer to the human benefits derived from both natural and managed ecosystems.

2 Classification and Monetary Assessment of Goods and Services

The criteria used to classify marine goods and services vary between authors, having changed over time. For example, Primack (1993) proposed classifying marine services into those that generate direct (e.g. production) and indirect values (e.g. climate regulation, existence value). For comparison, Pearce and Moran (1994) classified the value of environmental assets into use and non-use values (existence and bequest values, respectively). In the classification adopted here, in turn, use values were classified into direct use values (e.g. food provision) and indirect use values (e.g. the bioremediation of waste).

Most recent classifications have opted to classify marine services according to their function. For example, Daily (1997) identified in her classification the following marine ecosystem services or functions: the production of goods; regeneration processes; stabilizing processes; life-fulfilling functions; and the preservation of options. De Groot et al. (2002) classified ecosystem services according to the functions displayed by the system: production, regulatory, habitat, and information. Similarly, the Millennium Ecosystem Assessment (MEA 2005) and Armstrong et al. (2012) classified ecosystem services into: provisioning (e.g. food, fuel, freshwater); regulatory (e.g. climate regulation, water purification); support (e.g. primary production, soil formation); and cultural (e.g. cognitive development, recreation). The MEA classification provides a widely-acknowledged reference framework. Beaumont et al. (2007), although focused specifically on goods and services provided by marine biodiversity, followed the MEA classification but included an additional function: the option-use value for presently-unknown potential future uses of the marine environment. Murillas-Maza et al. (2011) introduce also an option-based value, to evaluate biodiversity conservation. Finally, Salomidi et al. (2012) compile a total of 56 types of European seabed biotopes and their related goods and services; they are classified on the basis of an adaptation of the categories proposed by MEA (2005) and Beaumont et al. (2007). A synthesis of the abovementioned classifications is presented in Table 1, where different ecosystem services functions, processes and ecosystem components are related to important marine habitats. The relations between habitats and ecosystem services provided were set using information from Armstrong et al. (2012) and Barbier et al. (2012), together with our own investigations and experience.

Murillas-Maza et al. (2011) have stated that the work carried out during the last decade, in the field of the valuation of ecosystem services, has focused upon terrestrial ecosystems; this is in sharp contrast with the fact that marine ecosystems constitute around two-thirds of the total economic value of world ecosystem services (Costanza et al. 1997). Since 1983, when the first paper using the term "ecosystem services" was published, 2,386 papers covering this topic have been published in journals included in the ISI database; this pattern is on an exponential trajectory (Costanza and Kubiszewski 2012). Furthermore, amongst the few valuation studies related to marine ecosystems, most of them focus mainly upon coastal ecosystems (e.g. beaches, seagrass beds, etc.), or fisheries. Costanza et al. (1997, 2006), Beaumont et al. (2006, 2007, 2008), Derous (2007), Murillas-Maza et al. (2011), de Groot et al. (2012), and van den Belt and Costanza (2012) are some of the most recent studies, which provide a framework for the valuation of marine biodiversity, obtaining a monetary value for some of the most important ecosystem services and functions (Table 2).

Costanza et al. (1997) synthesized information, together with original calculations from over 100 studies, to estimate the total value of the different ecosystem services per biome and service type; likewise, for the entire biosphere, by establishing a major division between marine and terrestrial systems. Beaumont et al. (2008) and DEFRA (2006) determined the economic value of marine goods and services in the UK, whilst Murillas-Maza et al. (2011) valued open ocean services for the Spanish Exclusive Economic Zone (EEZ). De Groot et al. (2012) have provided an overview analysis of the value of the ecosystem services of 10 biomes; this was obtained over 320 publications, covering more than 300 case study locations, with 665 value estimates.

Valuing ecosystem services, in market values or monetary terms, encounters difficulties. However, different techniques are used to assign a monetary value to such "intangibles" (Costanza et al. 1997). For production services (i.e. fisheries, aquaculture and raw materials) market monetary values can be estimated, using: the price approach (Costanza et al. 1997; Beaumont et al. 2008); or the net-added value (Murillas-Maza et al. 2011), which is the difference between the revenue (according to the market price) and the costs incurred in the production. Murillas-Maza et al. (2011) estimated the value of water supply, applying also the net-added value approach.

Table 1 Type of ecosystem services, functions, related ecosystem processes and components, together with the services provided and the type of uses of each of them, in relation to several important marine habitats

					Habitats													
	Type of services	Ecosystem processes and components	Ecosystem benefits	Type of uses	Salt-marshes	Mangroves	Sand-beaches and dunes	Seagrasses	Sublittoral sediments	Sublittoral vegetated substrata	Coral reefs	Cold water corals	Open slopes and basins	Canyons	Deep-sea	Sea-mounts	Water column	Sub-seabed
Supporting	Water circulation and exchange	Transport, dispersion, dilution	System ventilation	Indirect	++	++	0	++	+	+	+	+	+	++	0	+	++	0
	Nutrient cycling	Biogeochemical activity	Biological productivity	Indirect	++	++	+	++	+	++	++	+	+	++	++	+	++	+
	Primary production	Photosynthesis, chemosynthesis	Nutrient fixing, Food-webs	Indirect	++	++	+	++	+	++	++	++	0	0	0	0	++	0
	Biodiversity	System diversity	Ecosystem resilience	Indirect	++	++	+	++	+	++	++	++	+	++	++	++	++	0
	Habitat	Suitable space for wild organism, reproductive and nursery areas	Maintenance of biodiversity and beneficial species	Indirect	++	++	+	++	+	++	++	++	+	+	+	+	++	0
	Resilience	Stabilize environments	Reduce vulnerability	Indirect	++	++	++	++	+	++	++	++	+	+	+	+	+	0
	Maintenance of fisheries	Suitable habitats	Food	Direct	+	++	0	++	+	++	++	+	+	++	0	+	++	0
	Energy and minerals	Geochemical activity	Industry, energy sources	Direct	0	0	0	0	0	0	0	0	0	0	0	0	0	++
	Chemicals/pharmaceuticals		Industrial and health compounds	Direct	++	++	+	++	+	++	++	++	+	+	+	+	+	+

	Waste disposal	Transport, dispersion, dilution	Pollutants trapping and cycling	Direct	0	0	0	0	++	0	0	0	0	0	0	0	++	++
	Raw materials	Biogeochemical activity	Food inputs to the system, food-webs	Direct	++	++	+	++	+	++	+	+	0	0	0	0	+	0
Regulating	Carbon capture and storage (artificial)	CO_2 reduction	Climate change mitigation	Direct	0	0	0	0	0	0	0	0	0	0	0	0	+	++
	Gas regulation	Role of ecosystems in biogeochemical processes	UV protection, maintenance of air quality	Indirect	++	++	+	++	+	++	++	++	0	+	+	0	+	+
	Climate regulation	Influence on biologically mediated processes	Influence of climate, maintenance of T[a], precipitation	Indirect	++	++	+	++	+	++	++	+	+	+	+	+	++	0
	Disturbance prevention	Dampening environmental disturbance, wave attenuation	Storm protection, flood mitigation, coastal protection	Direct	++	++	+	++	+	++	++	+	0	0	0	0	0	0
	Erosion control	Sediment stabilization	Sea-level rise, subsidence	Indirect	++	++	++	++	+	++	++	+	0	0	0	0	0	0
	Water regulation	Runoff, river discharge, infiltration	Drainage, natural irrigation, flood mitigation	Direct	++	++	+	++	+	+	+	0	0	0	0	0	++	0

(continued)

Table 1 (continued)

	Type of services	Ecosystem processes and components	Ecosystem benefits	Type of uses	Habitats													
					Salt-marshes	Mangroves	Sand-beaches and dunes	Seagrasses	Sublittoral sediments	Sublittoral vegetated substrata	Coral reefs	Cold water corals	Open slopes and basins	Canyons	Deep-sea	Sea-mounts	Water column	Sub-seabed
	Nutrient regulation	Role of biota in storage and recycling	Maintenance of productive ecosystems	Direct	++	++	+	++	+	++	++	+	+	+	+	+	++	0
	Waste treatment	Removal of nutrients and pollutants	Pollution control, detoxification	Indirect	++	++	+	++	+	++	++	+	+	+	+	+	++	+
	Biological regulation	Trophodynamic regulation of populations	Control of pests, invasions	Indirect	++	++	+	++	+	++	++	++	+	+	+	+	+	0
Cultural	Educational		Training	Direct	++	++	++	++	+	++	++	++	+	+	+	++	+	+
	Scientific		Knowledge, blue technologies	Direct	++	++	++	++	+	++	++	++	++	++	++	++	++	+
	Aesthetic		Well-being	Nonuse	++	++	++	++	+	++	++	++	0	0	0	0	+	0
	Existence/ Bequest		Biophilia	Nonuse	++	++	+	++	+	++	++	0	0	0	0	0	+	0
	Tourism		Recreational grounds	Direct	++	++	++	++	+	++	++	0	0	0	0	0	++	0

Modified and expanded from Armstrong et al. (2012) and Barbier et al. (2012)
Key: 0 irrelevant; + some importance; ++ very important

Table 2 Monetary values of the ecosystem services and functions provided by marine environments, in different countries

Country/Zone	United Kingdom	Spain	New Jersey	Global	Global
Marine environment	**Shelf & coastal waters**	**Open ocean**	**Coastal systems**[a]	**Coastal & marine systems**[b]	**Coastal & marine systems**
Publication	**Beaumont et al.** (2006)	**Murillas-Maza et al.** (2011)	**Costanza et al.** (2006)	**Costanza et al.** (1997)	**De Groot et al.** (2012)
Year of price	2004	2005	2004	1994	2007
Currency	GBP	Euro	US$	US$	US$
Unit value	Million GBP (£)	Million €	US$ $acre^{-1}$ $year^{-1}$	US$ ha^{-1} $year^{-1}$	Int.$/ha/year
Provisioning services					
Food provision (fish)	513	593	NA	316	4.9 10^3
Raw materials	81.5	22.1	NA	27	22 10^3
Water supply	NA	1.6 10^3	570	NA	3.4 10^3
Other provisioning services	NA	NA	NA	NA	34 10^3
Regulating services					
Nutrient cycling	800–2,320 10^3	NA	734	[c]	1.7 10^3
Gas and climate regulation	0.4–8.47 10^3	3.8 10^3	NA	38	2.3 10^3
Bioremediation of wastes	NA	292	5.4 10^3	NA	165 10^3
Habitats, biodiversity, Biological control	NA	81.4	59	122	948
Disturbance prevention	0.3 10^3	NA	NA	NA	NA
Disturbance regulation	NA	NA	27.8 10^3	NA	NA
Erosion prevention	NA	NA	NA	NA	185 10^3
Other regulating services	NA	NA	NA	NA	30.9 10^3
Habitat services					
Nursery service	NA	NA	NA	NA	12.1 10^3
Genetic diversity	NA	NA	NA	NA	24 10^3
Cultural services					
Cultural heritage (cultural & spiritual)	NA	NA	332	105	721

(continued)

Table 2 (continued)

Country/Zone	United Kingdom	Spain	New Jersey	Global	Global
Aesthetic information	NA	NA	NA	NA	$12.7\ 10^3$
Leisure and recreation	$11.8\ 10^3$	NA	332	381	$103\ 10^3$
Cognitive values	317	NA	NA	NA	$1.2\ 10^3$
Non-use values (bequest and existence)	$0.5–1.1\ 10^3$	NA	NA	NA	NA

Key: *NA* not available

[a]Continental shelf, estuary, beach, saltwater wetland, freshwater wetland, open water

[b]Open oceans, estuaries, seagrass/algae beds, coral reefs, continental shelves

[c]\$62.1–174 ha^{-1} $year^{-1}$ for open oceans; \$11,100–31,100 ha^{-1} $year^{-1}$ for estuaries; \$10,000–28,000 ha^{-1} $year^{-1}$ for seagrass/algae beds and, \$752–2,110 ha^{-1} $year^{-1}$ for continental shelves

The value of regulating services, such as that of CO_2 regulation, can be calculated using the primary production valued at a fixed price (Costanza et al. 1997; Beaumont et al. 2008), or valued at the price of carbon which comes from future financial assets markets (Murillas-Maza et al. 2011). In particular, this work considers the CO_2 price from the European Climate Exchange Carbon Financial Futures Contracts that are listed and traded on the ICE (Intercontinental Exchange) Futures.[1]

The value of bioremediation of waste proposed by DEFRA (2006) is estimated, using a replacement cost methodology (services could be replaced with man-made systems); this is based upon the average biochemical oxygen demand (BOD) contained in wastewater discharged into the oceans. The avoidance cost method (services which allow for avoiding cost that would have been incurred) is used also to value regulatory services, such as disturbance prevention and alleviation. Finally, the replacement cost method is used for the valuation of the nutrient cycling capacity of the environment, which is considered as being a supporting service for the marine environment. Furthermore, Murillas-Maza et al. (2011) proposed the anticipated net-added value. Namely, the estimation of a payment to retain the option of fishing in the future is to be used as a proxy for the biomass conservation value. In addition, the net-added value based upon Government Financial Transfer is used as a proxy value of the marine biodiversity conservation. Cultural services, cognitive values, leisure and recreation services are estimated normally through their market value. However, monetary valuation of cultural services is highly debated, due to highly subjective perceptions, different value systems, etc. (Ghermandi et al. 2012; Hernández-Morcillo et al. 2013). Finally, Beaumont et al. (2008) estimated non-use values (bequest and existence), through contingent valuation. This is a non-market technique which assigns a direct value by asking concerned people their willingness

[1] ICE Futures is the global, electronic market place for trading futures contracts (Brent and West Texas Intermediate global benchmark crude, refined products, power contracts for NBP natural gas, UK electricity peak and base, coal and emissions, soft commodities and financial indices).

to pay for a service. However, other methodologies such as conjoint analysis, travel cost, multicriteria analysis and choice modeling can be used as alternative methods for the valuation of non-use values (Stagl 2007). In general, non-market use values and non-use values, which are non-market, are not naturally expressed in monetary terms. This is the reason why several methodologies have been developed to address this problem, such as, the above mentioned techniques. Monetary values per ecosystem service are collated and summarized in Table 2.

3 Conservation of Ecosystem Goods and Services

3.1 Overview of the Protection Measures for Ecosystem Services

Biodiversity conservation must be seen as a means to maintain and improve ecosystem goods and services, through the implementation of protection, prevention and restoration measures. In addition, the intrinsic values of nature and biodiversity must be taken also into account (Derous 2007). Hence, the first step towards conservation management includes the protection of biodiversity, habitats and marine resources (Cullis-Suzuki and Pauly 2010; Halpern et al. 2010; EEA 2012). In general, protection plans include areas which maintain ecological and evolutionary processes that are important for biodiversity persistence (e.g. interspecific interactions, faunal movements and migrations, disturbance regimes; Balmford et al. 1998; Roberts et al. 2003). However, after the CBD's Malawi principles for ecosystem management, established in 1998 (www.fao.org/docrep/006/y4773e/y4773e0e.htm), some authors have advocated the inclusion of ecosystem services in conservation and protection plans (Balvanera et al. 2001; Cognetti and Maltagliati 2010).

Egoh et al. (2007) reviewed the conservation assessments and the extent to which they include ecosystem services, on the basis of 476 references. Of the 100 references selected for the study, only 7 included ecosystem services; 13 referred to ecosystem services as a rationale for conservation, without including them in the assessment. The majority included cultural ecosystem services, followed by regulatory, provisioning and supporting services, respectively. The authors concluded that there was an urgent need for an appropriate framework for the planning of ecosystem services, in conservation and protection management.

3.2 Restoration of Marine Ecosystems: Restoring and Creating Goods and Services

Ecological restoration is becoming regarded as a major strategy for reversing biodiversity loss, as well as increasing the provision of ecosystem services (Bullock et al. 2011). Since 1st January 2000 to 31st December 2012, 757 articles dealing with "restoration" and "ecosystem services" terms have been published in the scientific

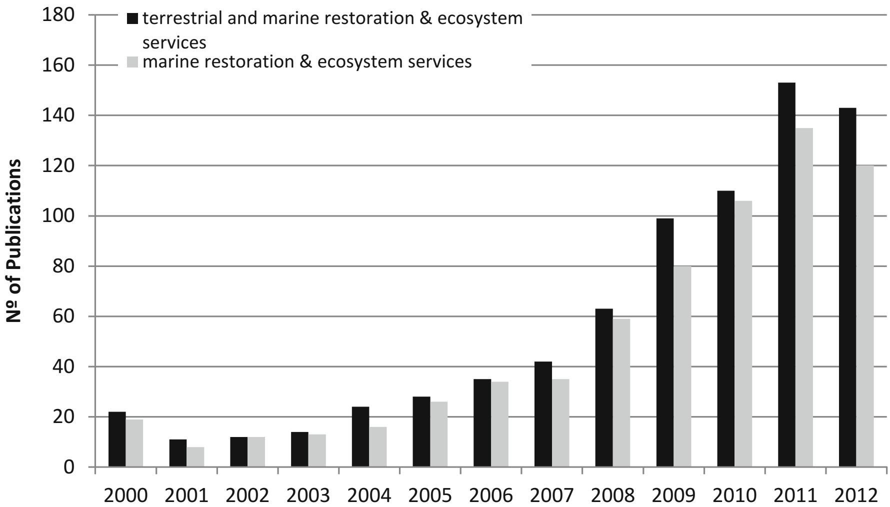

Fig. 1 Number of publications dealing with "restoration" and "ecosystem services" terms, only in marine systems, or in marine and terrestrial systems together. Information taken from the Web of Science (Accessed on 1st January 2013)

literature and are now available on the Web of Science (Fig. 1). However, of these publications, only 62 contained the term "marine", 89 "coastal" and 17 "ocean" (Fig. 2), with most of the publications focusing upon terrestrial ecosystems. Thus, despite the increasing number of publications in this particular field, more efforts are needed to investigate marine systems, as many uncertainties still remain about how marine biodiversity and ecosystem services recover in response to restoration efforts.

Consequently, Bullock et al. (2011) suggest that restoration projects can enhance effectively ecosystem services and decrease biodiversity loss rates. Since 1994, when restoration began gaining importance as means to provide goods and services (Pratt 1994), the context, including the political framework, have altered. For example, the Strategic Plan for Biodiversity 2011–2020, part of the CBD, includes the following two targets as a means to "enhance the benefits to all from biodiversity and ecosystem services" (Strategic Goal D):

- Target 14: "by 2020, ecosystems that provide essential services, including services related to water, and contribute to health, livelihoods and well-being, are restored and safeguarded, taking into account the needs of women, indigenous and local communities, and the poor and vulnerable".
- Target 15: "By 2020, ecosystem resilience and the contribution of biodiversity to carbon stocks has been enhanced, through conservation and restoration, including restoration of at least 15 per cent of degraded ecosystems, thereby contributing to climate change mitigation and adaptation and combating desertification".

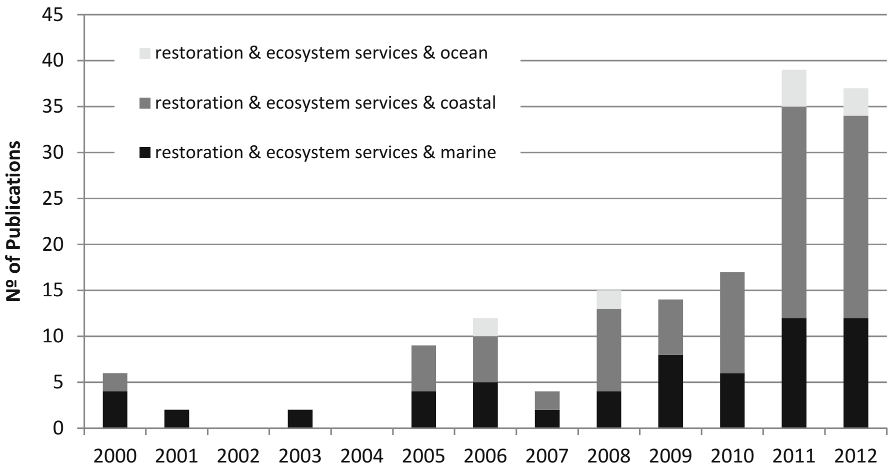

Fig. 2 Number of peer-reviewed publications in the field of "restoration and ecosystem services" of "ocean", "coastal" or "marine" ecosystems. Information taken from the Web of Science (Accessed on 1st January 2013)

The significant shift in the objectives of restoration, from a biodiversity to ecosystem service conservation perspective, could have both advantages and disadvantages as one could be attained at the expense of the other. Many examples have been acknowledged and published in the relationship between restoration and the possibility of altering the provisioning of goods and services, mostly for the better (Benayas et al. 2009; Bullock et al. 2011). However, only a few have actually measured the rate of change in this provisioning, together with the direct link between the application of a specific restoration measure and the actual benefit over the provisioning of goods and services. Investing in restoration of natural capital and ecosystem services can contribute to sustainable development. Pascual et al. (2012) have shown how investing in water treatment and restoration of a Spanish estuary can lead to important revenues, through biodiversity recovery and subsequent ecosystem services supplied. Restoration of mangroves and coral reefs has shown to be effective in increasing/improving ecosystem services, such as fisheries, coastal protection against cyclones, and water cycling (Moberg and Rönnbäck 2003). Other examples have shown how private business can even benefit from investing in the restoration of the natural environment (Kettunen and ten Brink 2006).

According to Bullock et al. (2011), ecosystem services need to be targeted as 'bundles' as, if they are targeted in isolation, conflicts may arise. However, once destroyed, natural ecosystems are often costly and sometimes impossible to restore. A policy shift towards restoration, focusing upon goods and services, examined at an appropriate scale, could provide a solid foundation upon which effective biodiversity conservation may be addressed (Bullock et al. 2011).

3.3 Prevention Measures, to Avoid Ecosystem Goods and Services Degradation

Biodiversity conservation is investigated usually through protection and restoration actions. Similarly, the conservation of marine goods and services relies primarily upon the protection of biodiversity and, only recently, on restoration (see previous Section). However, considering that achieving complete restoration is difficult and that the frequency of both natural extreme events (e.g. floods) and anthropogenic environmental disasters (e.g. oil-spills) are increasing (Goswami et al. 2006; Riegl 2007; Halpern et al. 2008; Ban et al. 2010), prevention actions are needed urgently if goods and services are to be maintained and/or improved (van den Belt and Costanza 2012). As stated by Benjamin Franklin "a stitch in time saves nine".

Anderson (1990) stated that *"the benefits obtained through disaster prevention, in all countries, are equivalent to the savings in losses a disaster would have brought"*. Whilst research undertaken into costs and benefits of preventions measures is commencing, publications reporting environmental and socio-economic losses caused by natural extreme or anthropogenic disastrous events are extensive. For example, the Prestige oil spill (along the northern waters of the Iberian Peninsula) caused a loss of 76 million euros for the Spanish fishing sector, within 1 year of the accident. Elsewhere, the State of Alaska and the US Government claimed from Exxon 1,000 million dollars, for damages caused to natural resources and compensation for injuries (Carson et al. 2003). Contingent valuation studies suggested that the loss of passive uses in this accident was equivalent to values that ranged between 4,870 and 7,190 million dollars (Carson et al. 2003). Within a different context, the invasive Indo-Pacific lionfish is regarded as having caused mainly negative impacts on the Caribbean fishery, tourism and aquarium industries. This is a result of a reduction in landings of economically-important species, the incidence of marine envenomation, and a reduction in the sales of lionfish for aquariums, respectively (Morris and Whitfield 2009). Uyarra et al. (2005) have suggested that coral bleaching, together with a reduction in beach extent induced by climate change, could potentially cause an 80 % reduction in tourism arrivals at two tourism-dependent islands of the Caribbean. Despite the fact that there more studies have explored the (environmental and socio-economic) impacts of disastrous events, than estimating the potential benefits and costs of implementing prevention measures, negative experiences have led to the signing of international agreements and the introduction of measures that can prevent future environmental harm, together with subsequent goods and services losses.

In addition to the UN Convention on the Law of the Sea (UNCLOS 1982), which provides the framework to *"prevent, reduce and control human caused pollution of the marine environment, including the intentional or accidental introduction of harmful or alien species to a particular part of the marine environment"*, more specific Conventions are already in place, or negotiations are ongoing. For example, the International Convention on the Control of Harmful Anti-fouling Systems in

Ships[2] (in force since 2008) and the International Convention for the Control and Management of Ships' Ballast Water and Sediments (BWM[3]) (adopted in 2004, but not ratified), complement scientific efforts to reduce and prevent the translocation of invasive species. The International Convention for the Prevention of Pollution from Ships (MARPOL[4]) has entered into force in 1983, with the aim of preventing/minimizing/managing pollution (i.e. noxious liquids, harmful substances, sewage, garbage and air pollution) originating from ships, either through accidents or routine activities. The Oil Pollution Act,[5] initiated in 1990 in the US, and Regulation (EC) No 417/2002 in the EU, to prevent oil spills by imposing or accelerating the change from single to double hull (or equivalent designs) on vessels, has reduced the number and volume of oil spills (Kim 2002). Cai et al. (2010) propose developing hard and soft structures (e.g. dykes and sand nourishment, respectively) and prohibiting unreasonable sand mining and land reclamation, as means to prevent coastal erosion from occurring in China.

The abovementioned, together with many other additional prevention measures, have cascading effects, including direct benefits to the marine environment and indirect prevention from harming the goods and services provided by such an environment. Such cascading effects have often been studied retrospectively, following an extreme natural event or when an accident has occurred and goods and services have already been damaged or lost. Efforts should be placed now in providing scientific evidence on how preventative measures benefit the marine environment, the goods and services and, consequently, society as a whole.

3.4 Conservation: Payments, Markets and Optimal Levels

Conservation of biodiversity is one of the most important policy responses to ecosystem degradation. However, cost/benefit analyses of conservation are scarce (Hancock 2010; Bullock et al. 2011) and the benefits to society are barely examined (Aronson et al. 2010). In addition, Balmford et al. (2002) have estimated that the overall benefit/cost ratio of an effective global program for the conservation of remaining wild nature is at least 100:1.

The evidence available suggests that conservation projects can be cost-effective when low-cost methods are utilized (Bullock et al. 2011). Some conservation projects are undertaken for research purposes only; thus, the associated costs are not accounted for usually. Further, conservation costs depend upon the ecosystem type

[2] www.imo.org/About/Conventions/listofconventions/pages/international-convention-on-the-control-of-harmful-anti-fouling-systems-on-ships-(afs).aspx

[3] www.imo.org/About/Conventions/ListOfConventions/Pages/International-Convention-for-the-Control-and-Management-of-Ships%27-Ballast-Water-and-Sediments-(BWM).aspx

[4] www.imo.org/About/Conventions/listofconventions/pages/international-convention-for-the-prevention-of-pollution-from-ships-(marpol).aspx

[5] www.epa.gov/osweroe1/content/lawsregs/opaover.htm

being examined, the state of the ecosystem requiring conservation, etc. (Bullock et al. 2011). Such variability may be a hindrance, comparing the costs of two conservation projects.

Conversely, conservation projects may also benefit people in various monetary and non-monetary ways. In this sense, the Ecosystem Service Valuation (ESV) is one of the main pathways that could be used to measure costs/benefits and outweigh the profitability of conservation projects (Bullock et al. 2011). Investing in conservation is seen increasingly as a "win-win situation", as it can potentially generate substantial ecological, social and economic benefits (de Groot et al. 2010; Montoya et al. 2012). Valuing the benefits of such investments and undertaking comparison amongst different projects or investment types, most definitively guide further decision-making processes.

Perrings and Halkos (2012) state that the optimal level of biodiversity conservation depends upon the relationship between the spatial and temporal distribution of the costs and benefits of conservation. Environmental changes derived from a conservation project can generate local costs, as well as affect people far removed, in space and time. These authors demonstrate also how the traditional 'Hotelling' conservation principle, which implies that conservation is an optimal strategy only if the growth rate of the value of the conserved asset is greater than the rate of return of the converted assets, can be used to obtain the optimal level of change in land conversion/conservation.

One critical question that remains unanswered in the field of biodiversity conservation is who should pay. In restoration projects, some argue that the "polluter pays" principle, which suggests that those causing ecosystem degradation are responsible for covering the costs of restoration, should apply (Kock and Hobbs 2007). However, others consider that if conservation (e.g. protection, restoration and prevention) efforts could benefit a wider population, then everybody should pay for it, either through environmental taxes, tradable permit systems and initiatives (e.g. biobanking), government funding, or other means.

Governments, other political institutions and individuals are more likely to cover costs produced by local environmental externalities, rather than the costs originating from externalities that occur in the future, or at distant locations. Other factors, such as the income or the institution's revenue are also likely to affect the 'willingness to pay' towards conservation projects. However, it is really people's dependence on the specific goods and services that determine their importance in society. Thus, it could be expected that there is an increase in the 'willingness to pay' as dependence increases, if a positive output is expected after conservation measures are implemented.

In addition to the "polluter pays" principle, other rather more complex systems are being developed to cover costs of conservation. For example, in response to the growing degradation of marine goods and services, different markets are emerging trying to protect them. The more relevant market types are: (i) those that aim at regulating some commercial fisheries through the use of Individual Transferable Quotas (ITQ) (the current Reform of the Common Fishery Policy seeks to set up ITQ for vessels up to 12 m in length); (ii) CO_2 emission i.e. which price could be used to estimate the value of the CO_2 regulation service; and (iii) eco-tourism, especially around Marine Protected Areas, which aims are to promote biodiversity while educating people. It is also within this context that the Payment for Ecosystem Services (PES) concept has appeared and started to gain in importance.

Wunder (2005) defines PES as "*a voluntary transaction, where a well-defined environmental service likely to secure that a service is being "bought" by a (minimum one) service buyer, from a (minimum one) service provider, if and only if the service provider secures service provision (conditionality)*". Therefore, the PES is intended to compensate individuals or communities with actions that maintain the provision of services (Farley et al. 2010). As with any market, the development of a PES system requires that there are willing buyers and sellers of a service, at an agreed-upon price, which is facilitated by a functioning institutional agreement. PES is considered to be an example of a market-based mechanism. It often involves a series of payments, which are made by the beneficiaries of the services in question, for example, individuals, communities, businesses or government acting on behalf of various parties. There are different PES types (Smith et al. 2013): (i) public payment schemes through which government pays land or resource managers to enhance ecosystem services on behalf of the wider public; (ii) private payment schemes, self-organised private deals, in which beneficiaries of ecosystem services contract directly with service providers; and (iii) public-private payment schemes that draw on both government and private funds to pay land or other resource managers for the delivery of ecosystem services.

Ecological conservation, more specifically restoration, programs and PES, while inherently linked, have been pursued largely separately; it has been suggested that they are coupled if full ecosystem management is to be achieved (Yin and Zhao 2012). Conservation temporal lags require careful analysis when implementing any kind of tradable permits, as they can lead to fluctuations in permits prices that reduce the efficiency of the permit market (Drechsler and Hartig 2011).

4 Conclusions

Human activities, based on socio-economic drivers (i.e. demand for food, resources, etc.), are creating pressures in marine waters, which change the state of natural systems (measured by comparing indicators, against quality standards) (Fig. 3). These systems have some resistance to change, following pressures; however, as pressure increases, changes in the system produce impacts on the ecosystem components. Such impacts can be assessed through the structure and functioning of the ecosystem, using methods that compare data with reference conditions and/or targets (Borja et al. 2012b). In addition, evaluating the loss of ecosystem services provided by such ecosystems can provide a new understanding of the impacts produced (Lester et al. 2010).

Despite the difficulties in identifying all services provided by marine ecosystems, the uncertainties in measuring the economic value of such services and their

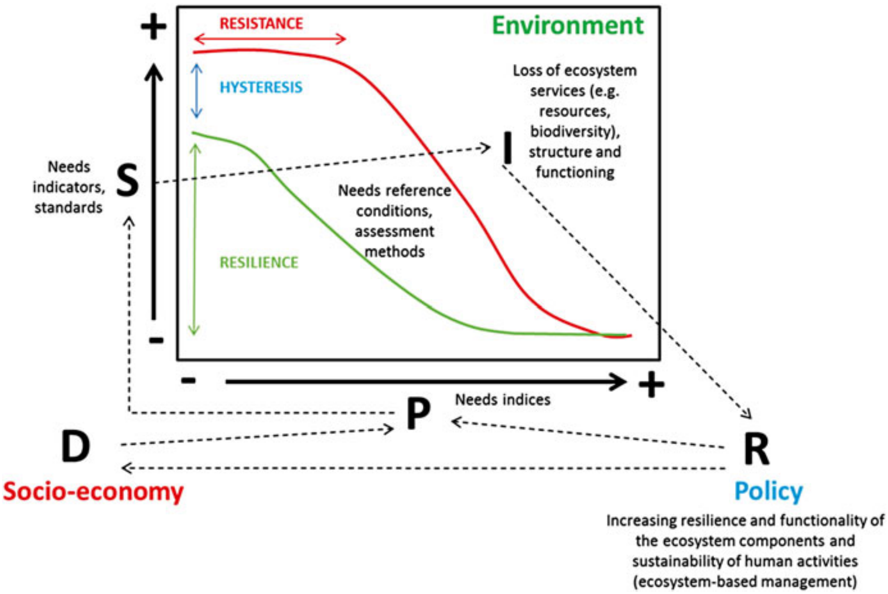

Fig. 3 Relationships between Drivers (D), Pressures (P), change of State (S), Impacts (I) and Responses (R), within the framework of socio-economic activities, the environmental effects and the policy actions taken to remove or minimize pressures

effects of conservation and restoration measures, there are clear pathways which could be improved, such that the provisioning of goods and services are enhanced. Policy responses, to remove or minimize pressures (by conservation, restoration or prevention), should focus upon enhancing the resilience and functionality of the ecosystem components, together with an effort to make human activities sustainable (Pine 2012). Resilience is the ability of an ecosystem to return to its original state after being disturbed, and can be evaluated when the ecosystem services provided by the original system are recovered. However, not always the systems can return to original levels. In this case, the difference is known as 'hysteresis', which can be shown or evaluated by the difference between the original ecosystem services provided and those provided in the new situation (Fig. 3).

Hence, the above framework is an ecosystem-based management, which requires: integration of multiple system components and uses; identifying and striving for sustainable outcomes; precaution in avoiding deleterious actions; and adaptation based on experience to achieve effective solutions (Boesch 2006). In this sense, the precautionary principle is very relevant in marine management and the contemporary environmental policy (O'Riordan and Jordan 1995).

In conclusion, research and management studies in coming years should focus upon a number of key objectives (see below).

- Underpinning decision-making, risk assessment and the management of marine ecosystems under complex multiple stress background, enhancing the understanding of multiple stressor interactions and accumulation (Crain et al. 2008; Ban et al. 2010).

- Understanding the species-stressor-relationships and impacts on the ecological functioning and resilience of the ecosystems. Actions to remove pressures, or restore marine systems, must be related to an increase of the ecosystems resilience; this permits its self-sustainability, when human activities are undertaken.
- Identifying, mapping and evaluating ecosystem services, to measure evolution (loss, gain) against an increase in pressures or actions taken to minimize them, including the effects of global change.
- Developing methods (including indicators, indices, metrics) to measure pressures, changes of state and impacts; these should include also changes in the functionality of the ecosystems and the goods and services that they provide. In this way, modelling is becoming a useful approach to investigate these issues and valuate ecosystem services, needing further development (Barbier 2012).
- Improving management of marine systems, including conservation and sustainability of human activities as paramount paradigms of such management.

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References

Anderson MB (1990) Which costs more: prevention or recovery? Managing natural disasters and the environment: selected materials from the Colloquium on the Environment and Natural Disaster Management, June 27–28, 1990. World Bank, Washington, DC

Appeltans W, Ahyong ST, Anderson G, Angel MV et al (2012) The magnitude of global marine species diversity. Curr Biol 22:2189–2202

Armstrong CW, Foley NS, Tinch R, van den Hove S (2012) Services from the deep: steps towards valuation of deep sea goods and services. Ecosyst Serv 2:2–13

Aronson J, Blignaut JN, Milton SJ, Maitre DL et al (2010) Are socioeconomic benefits of restoration adequately quantified? A meta-analysis of recent papers (2000–2008) in restoration ecology and 12 other scientific journals. Restor Ecol 18:143–154

Balmford A, Mace GM, Ginsberg JR (1998) The challenges to conservation in a changing world: putting processes on the map. In: Mace GM, Balmford A, Ginsberg JR (eds) Conservation in a changing world. Cambridge University Press, Cambridge

Balmford A, Bruner A, Cooper P, Costanza R, Farber S et al (2002) Economic reasons for conserving wild nature. Science 297:950–953

Balvanera P, Daily GC, Ehrlich PR, Ricketts TH et al (2001) Conserving biodiversity and ecosystem services. Science 291:2047

Ban NC, Alidina HM, Ardron JA (2010) Cumulative impact mapping: advances, relevance and limitations to marine management and conservation, using Canada's Pacific waters as a case study. Mar Policy 34:876–886

Barbier EB (2012) A spatial model of coastal ecosystem services. Ecol Econ 78:70–79

Barbier EB, Hacker SD, Koch EW, Stier A, Silliman B (2012) Estuarine and coastal ecosystems and their services. In: van den Belt M, Costanza R (eds) Ecological economics of Estuaries and Coasts. Treatise on Estuarine and Coastal Science (Series eds, Wolanski E, McLusky D). Academic, Waltham

Beaumont N, Townsend M, Mangi S, Austen MC (2006) Marine Biodiversity: an economic valuation. DEFRA, Plymouth, UK

Beaumont NJ, Austen MC, Atkins JP, Burdon D et al (2007) Identification, definition and quantification of goods and services provided by marine biodiversity: implications for the ecosystem approach. Mar Pollut Bull 54:253–265

Beaumont NJ, Austen MC, Mangi SC, Townsend M (2008) Economic valuation for the conservation of marine biodiversity. Mar Pollut Bull 56:386–396

Benayas JMR, Newton AC, Diaz A, Bullock JM (2009) Enhancement of biodiversity and ecosystem services by ecological restoration: a meta-analysis. Science 325:1121–1124

Bertram C, Rehdanz K (2013) On the environmental effectiveness of the EU Marine Strategy Framework Directive. Mar Policy 38:25–40

Boesch DF (2006) Scientific requirements for ecosystem-based management in the restoration of Chesapeake Bay and Coastal Louisiana. Ecol Eng 26:6–26

Borja A, Bricker SB, Dauer DM, Demetriades NT et al (2008) Overview of integrative tools and methods in assessing ecological integrity in estuarine and coastal systems worldwide. Mar Pollut Bull 56:1519–1537

Borja A, Basset A, Bricker S, Dauvin JC, Elliott M et al (2012a) Classifying ecological quality and integrity of Estuaries. In: Wolanski E, McLusky DS (eds) Treatise on Estuarine and Coastal Science, vol 1. Academic, Waltham, pp 125–162

Borja A, Dauer DM, Grémare A (2012b) The importance of setting targets and reference conditions in assessing marine ecosystem quality. Ecol Indic 12:1–7

Bullock JM, Aronson J, Newton AC, Pywell RF, Rey-Benayas JM (2011) Restoration of ecosystem services and biodiversity: conflicts and opportunities. Trends Ecol Evol 26:541–549

Butchart SHM, Walpole M, Collen B, van Strien A et al (2010) Global Biodiversity: indicators of recent declines. Science 328:1164–1168

Cai F, Xianze S, Jianhui L, Bing L, Gang L (2010) Coastal erosion in China under the condition of global climate change and measures for its prevention. Prog Nat Sci 19:415–426

Carson RT, Mitchell RC, Hanemann M, Raymond JK, Stanley P, Ruud PA (2003) Contingent valuation and lost passive use: damages from Exxon Valdez oil spill. Environ Resour Econ 25:257–286

CBD (1992) Convention on biological diversity United Nations www.cbdint/doc/legal/cbd-en.pdf

Cognetti G, Maltagliati F (2010) Ecosystem service provision: an operational way for marine biodiversity conservation and management. Mar Pollut Bull 60:1916–1923

Costanza R, Kubiszewski I (2012) The authorship structure of "ecosystem services" as a transdisciplinary field of scholarship. Ecosyst Serv 1:16–25

Costanza R, d'Arge R, de Groot R, Farber S, Grasso M et al (1997) The value of the world's ecosystem services and natural capital. Nature 387:253–260

Costanza R, Wilson M, Troy A, Voinov A, Liu S, D'Agostino J (2006) The value of New Jersey's ecosystem services and natural capital Part II. Gund Institute for Ecological Economics/New Jersey Department of Environmental Protection, pp 1–177

Crain CM, Kroeker K, Halpern BS (2008) Interactive and cumulative effects of multiple human stressors in marine systems. Ecol Lett 11:1304–1315

Crossland CJ, Kremer HH, Lindeboom HJ, Marshall Crossland JI, Le Tissier MDA (2005) Coastal fluxes in the Anthropocene; the Land-Ocean Interactions in the Coastal Zone Project of the International Geosphere-Biosphere Programme. Springer, Berlin

Cullis-Suzuki S, Pauly D (2010) Marine protected area costs as "Beneficial" fisheries subsidies: a global evaluation. Coast Manag 38:113–121
Daily GC (1997) Nature's services: societal dependence on natural ecosystems. Island Press, Washington, DC
Day JW, Hall CA, Yáñez-Arancibia A, Pimentel D, Ibáñez Martí C, Mitsch WJ (2009) Ecology in times of scarcity. Bioscience 59:321–331
De Groot RS, Wilson MA, Boumans RM (2002) A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecol Econ 41:393–408
de Groot RS, Alkemade R, Braat L, Hein L, Willemen L (2010) Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecol Complex 7:260–272
de Groot R, Brander L, van der Ploeg S, Costanza R et al (2012) Global estimates of the value of ecosystems and their services in monetary units. Ecosyst Serv 1:50–61
DEFRA (2006) Marine biodiversity. An economic valuation. Building the evidence base for the Marine bill, Technical Report Defra. Department of Environment, Food and Rural Affairs, London
Derous S (2007) Marine biological valuation as a decision support tool for marine management. PhD thesis, University of Ghent
Drechsler M, Hartig F (2011) Conserving biodiversity with tradable permits under changing conservation costs and habitat restoration time lags. Ecol Econ 70:533–541
EEA (2012) Protected areas in Europe – an overview. European Environment Agency (EEA) Report 5
Egoh B, Rouget M, Reyers B, Knight AT, Cowling RM, van Jaarsveld AS, Welz A (2007) Integrating ecosystem services into conservation assessments: a review. Ecol Econ 63:714–721
Elliott M, Burdon D, Hemingway KL, Apitz SE (2007) Estuarine, coastal and marine ecosystem restoration: confusing management and science – a revision of concepts. Estuar Coast Shelf Sci 74:349–366
Farley J, Aquino A, Daniels A, Moulaert A, Lee D, Krause A (2010) Global mechanisms for sustaining and enhancing PES schemes. Ecol Econ 69:2075–2084
Gardner TA, Côté IM, Gill A, Grant A, Watkinson AR (2003) Long-term region-wide declines in Caribbean corals. Science 301:958–960
Ghermandi A, Nunes PALD, Portela R, Rao N, Teelucksingh SS (2012) Chapter 11: Recreational, cultural and aesthetic services from estuarine and coastal ecosystems. In: van den Belt M, Costanza R (eds) Ecological economics of Estuaries and Coasts. Treatise on Estuarine and Coastal Science (Series eds, Wolanski E, McLusky D), vol 12. Academic, Waltham, pp 217–237
Goswami BN, Venugopal V, Sengupata D, Madhusoodanan MS, Xavier PK (2006) Increasing trend of extreme rain events over India in a warming environment. Science 314:1442–1445
Halpern BS, Walbridge S, Selkoe KA et al (2008) A global map of human impact on Marine ecosystems. Science 319:948–952
Halpern BS, Lester SE, McLeod KL (2010) Placing marine protected areas onto the ecosystem-based management seascape. Proc Natl Acad Sci 107:18312–18317
Hancock J (2010) The case for an ecosystem service approach to decision-making: an overview. Biosci Horiz 3(2):188–196
Heckbert S, Costanza R, Poloczanska ES, Richardson AJ (2012) Chapter 10: Climate regulation as a service from estuarine and coastal ecosystems. In: van den Belt M, Costanza R (eds) Ecological economics of Estuaries and Coasts. Treatise on Estuarine and Coastal Science (Series eds, Wolanski E, McLusky D), vol 12. Academic, Waltham, pp 200–216
Hernández-Morcillo M, Plieninger T, Bieling C (2013) An empirical review of cultural ecosystem service indicators. Ecol Indic 29:434–444
Hobbs RJ (2007) Restoration ecology: are we making an impact? Restor Ecol 15:597–600
IPBES (2008) Intergovernmental science-policy platform on biodiversity and ecosystem services. United Nations. www.ipbes.net/previous-ipbes-meetings/1st-meeting-on-ipbes.html

Kettunen M, ten Brink P (2006) Value of biodiversity – Documenting EU examples where biodiversity loss has led to the loss of ecosystem services. Final report for the European Commission. Institute for European Environmental Policy (IEEP), Brussels, 131 pp

Kim I (2002) Ten years after the enactment of the Oil Pollution Act of 1990: a success or a failure. Mar Policy 26:197–2007

Kock JM, Hobbs RJ (2007) Synthesis: is Alcoa successfully restoring a Jarrah forest ecosystem after bauxite mining in Western Australia? Restor Ecol 15:S137–S144

Lange G-M, Jiddawi N (2009) Economic value of marine ecosystem services in Zanzibar: implications for marine conservation and sustainable development. Ocean Coast Manag 52:521–532

Lester SE, McLeod KL, Tallis H, Ruckelshaus M et al (2010) Science in support of ecosystem-based management for the US West Coast and beyond. Biol Conserv 143:576–587

Lotze HK (2010) Historical reconstruction of human-induced changes in U.S. estuaries. Oceanogr Mar Biol Annu Rev 48:267–338

MDG (2000) Millennium Development Goals Millennium. United Nations Millennium Declaration. www.un.org/millenniumgoals. Accessed on 5 Jan 2013

MEA (Millennium Ecosystem Assessment) (2005) Ecosystems and human well-being: biodiversity synthesis. World Resources Institute, Washington, DC, 100 pp

Mee L (2012) Between the devil and the deep blue sea: the coastal zone in an era of globalisation. Estuar Coast Shelf Sci 96:1–8

Moberg F, Folke C (1999) Ecological goods and services of coral reef ecosystems. Ecol Econ 29:215–233

Moberg F, Rönnbäck P (2003) Ecosystem services of the tropical seascape: interactions, substitutions and restoration. Ocean Coast Manag 46:27–46

Montoya D, Rogers L, Memmott J (2012) Emerging perspectives in the restoration of biodiversity-based ecosystem services. Trends Ecol Evol 27:666–672

Mora C (2008) A clear human footprint in the coral reefs of the Caribbean. Proc R Soc 275:767–773

Morris JA Jr, Whitfield PE (2009) Biology, ecology, control and management of the invasive Indo-Pacific Lionfish: an updated integrated assessment. NOAA Technical Memorandum NOS NCCOS 97. 57 pp

Murillas-Maza A, Virto J, Gallastegui MC, González P, Fernández-Macho J (2011) The value of open ocean ecosystems: a case study for the Spanish exclusive economic zone. Nat Res Forum 35:122–133

Nellemann C, Corcoran E (2010) Dead planet, living planet-biodiversity and ecosystem restoration for sustainable development. United Nations Environment Programme Report

O'Riordan T, Jordan A (1995) The precautionary principle in the contemporary environmental policy. Environ Values 4:191–212

Palmer M, Filoso S (2009) Restoration of ecosystem services for environmental markets. Science 352:575–576

Pascual M, Borja A, Franco J, Burdon D, Atkins JP, Elliott M (2012) What are the costs and benefits of biodiversity recovery in a highly polluted estuary? Water Res 46:205–217

Pearce D (1998) Auditing the earth: the value of the world's ecosystem services and natural capital. Environ Sci Policy Sustain Dev 40:23–28

Pearce D, Moran M (1994) Economic value of biodiversity. World Conservation Union EARTHSCAN. Earthscan Publications Ltd, London

Perrings C, Halkos G (2012) Who cares about biodiversity?. Optimal conservation and trans-boundary biodiversity externalities. Environ Resour Econ 52:585–608

Pine JC (2012) Enhancing the resilience of coastal communities: dealing with immediate and long-term impacts of natural hazards. In: van den Belt M, Costanza R (eds) Ecological economics of estuaries and coasts. Treatise on estuarine and coastal science (Series eds, Wolanski E, McLusky D), vol 12. Academic, Waltham, pp 271–288

Pratt JR (1994) Artificial habitats and ecosystem restoration-managing for the future. Bull Mar Sci 55(1–2):268–275

Primack RB (1993) Essentials of conservation biology. Sinauer Associates, Sunderland

Riegl B (2007) Extreme climatic events and coral reefs: how much short-term threat from global change? Geol Approaches Coral Reef Ecol 192:315–341
Roberts CM, Andelman S, Branch G, Bustamante RH et al (2003) Ecological criteria for evaluating candidate sites for marine reserves. Ecol Appl 13:S199–S214
Roberts L, Stone R, Sugden A (2009) The rise of restoration ecology. Science 325:555
Rönnbäck P, Kautsky N, Pihl L, Troell M, Söderqvist T, Wennhage H (2007) Ecosystem goods and services from Swedish coastal habitats: identification, valuation, and implications of ecosystem shifts. Ambio 36:534–544
Salomidi M, Katsanevakis S, Borja A, Braeckman U et al (2012) Assessment of goods and services, vulnerability, and conservation status of European seabed biotopes: a stepping stone towards ecosystem-based marine spatial management. Mediterr Mar Sci 13:49–88
Simenstad C, Reed D, Ford M (2006) When is restoration not?: incorporating landscape-scale processes to restore self-sustaining ecosystems in coastal wetland restoration. Ecol Eng 26:27–39
Smith S, Rowcroft P, Everard M, Couldrick L, Reed M, Rogers H, Quick T, Eves C, White C (2013) Payments for ecosystem services: a best practice guide. Defra, London
Stagl S (2007) SDRN rapid research and evidence review on emerging methods for sustainability valuation and appraisal: final report to the sustainable development research network, 68 pp. Available at: http://www.pik-potsdam.de/news/public-events/archiv/alter-net/former-ss/2007/11-09.2007/stagl/literature/stagl_2007.pdf. Accessed 10 July 2013
ten Brink P, Berghöfer A, Schröter-Schlaack C, Sukhdev P, Vakrou A, White S, Wittmer H (2009) TEEB – the economics of ecosystems and biodiversity for national and international policy makers 2009. United Nations Environment Programme and the European Commission, 48 pp
UNCLOS (1982) United Nations Convention on the Law of the Sea, signed at Montego Bay, Jamaica, on 10 Dec 1982, 202 pp. www.un.org/Depts/los/index.htm
Uyarra MC, Côté IM, Gill JA, Tinch RRT, Viner D, Watkinson AR (2005) Island-specific preferences of tourists for environmental features: implications of climate change for tourism-dependent islands. Environ Conserv 32:11–19
Van den Belt M, Costanza R (2012) Ecological economics of estuaries and coasts. In: Wolanski E, McLusky DS (eds) Treatise on estuarine and coastal science, vol 12. Academic, Waltham, 525 pp
Wilkinson C (2004) The status of coral reefs of the world: 2004. Australian Institute for Marine Sciences, Townsville
Wunder S (2005) Payment for environmental services: some nuts and bolts. (Occasional Paper No, 42). Center for International Forestry Research, Bogor, Indonesia
Yin R, Zhao M (2012) Ecological restoration programs and payments for ecosystem services as integrated biophysical and socioeconomic processes: China's experience as an example. Ecol Econ 73:56–65
Young TP (2000) Restoration ecology and conservation biology. Biol Conserv 92:73–83

Terrestrial Ecosystem Services in River Basins: An Overview and an Assessment Framework

Olaf Bastian, Karsten Grunewald, and Ralf-Uwe Syrbe

Abstract The concept of ecosystem services is a promising approach to assess ecosystems also in river basins, particularly in view of their multifunctional use and sustainable development. The chapter mainly focuses on terrestrial ecosystems. Starting from the definition and classification of these ecosystems, relevant provisioning, regulating and socio-cultural services are listed, together with short descriptions, examples, their relations to water, and with suitable indicators. The EPPS assessment framework (Ecosystem Properties, Potentials and Services) is presented and then illustrated on the example of two very frequently occurring terrestrial ecosystems or land use types: (1) semi-natural grassland, and (2) farmland (with the specification "energy cropping").

Keywords Assessment framework • Energy crops • Indicators • Potentials • Semi-natural grassland

1 Introduction

The land surface, with its terrestrial ecosystems, is the primary human habitat. Humankind has influenced and shaped it actively for millennia and centuries, with increasing intensity. Global change, including demographic changes, climate change and economic globalization, constitute enormous challenges for the management of the land. This involves the provision of food and energy, housing, recreation, health, and many other issues, and also the maintenance of ecosystems and biodiversity in general.

Land use and the driving forces of land use change are first and foremost a socio-economic category. Humans depend absolutely on the utilization of land. The question

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is, however, how are we intervening in ecosystems? How can we ensure economic and social development, while sustainably maintaining the functioning and the potentials of ecosystems as the basis for human life? This is theoretically clear, but practically difficult to achieve, because of the growing world population and our striving for higher material well-being.

The implementation of the guiding principle of "sustainable development" should rely upon the concepts of ecosystem services and multifunctional landscapes, for only in this way can human demands be balanced with the real potentials of nature, in order to achieve rational goals (Grunewald and Bastian 2015).

Ecosystem services can be defined as contributions of ecosystems to human well-being, i.e. as services and goods, which provide direct or indirect economic, material, health related or psychological benefits to humans (e.g. Fisher et al. 2009; Naturkapital Deutschland – TEEB DE 2012). For about two decades the concept of ecosystem services has gained more and more attention among scientists and – increasingly – politicians worldwide. Important milestones have been e.g. the Millennium Ecosystem Assessment (MEA 2005), the international TEEB study (TEEB 2010), emerging national TEEB processes or the foundation of the network Ecosystem Services Partnership.

The knowledge transfer and propagation of ecosystem services can strengthen public awareness about the role and values of ecosystems and biodiversity in general. The growing popularity of the ecosystem services concept can be seen primarily as a reaction to the interplay of firstly the long-term neglect of biophysical and ecosystem functions – often considered gratis – in our economic cycles and the societal system as such and, secondly, the growing degradation of the ecosystems providing these services.

The concept of ecosystem services represents a multilayered approach to the interface between environmental and societal claims, with special consideration of economic aspects. However, these are closely interlinked with ecological and social ones, i.e. all three dimensions of sustainability are addressed. Thus, the concept can be used as a stimulus and tool to find appropriate solutions for land use and ecosystem management and to balance economic interests with ecological and social requirements in multifunctional landscapes.

The application of the ecosystem services concept to watersheds may contribute essentially to the socially acceptable, cost-effective and environmentally-friendly, sustainable management of freshwater resources, because there are many interrelationships between the hydrological cycle and ecosystems – including terrestrial ecosystems. One of its main goals is to price side effects and follow-up costs of particular land use forms. This is designed to help

- internalize, as much as possible, the costs and benefits of measures that affect ecosystems and landscapes,
- provide prices and incentives for land users in such a way that the protection of biodiversity and environment generally are ensured, and
- reduce market distortions, which affect biodiversity and ecosystem services negatively.

The early recognition of land use problems and conflicts is a precondition for appropriate human action, but not necessarily a guarantee for the "right" action. At the interface of ecosystem services and land use, an integrative management is necessary, with the goal of a balance between the conservation, sustainable use and fair allocation of benefits from the utilization of the land. For this, we need suitable concepts and methodological approaches, such as those shown below.

This chapter will (1) provide an introduction to terrestrial ecosystems; (2) give an overview of important water related terrestrial ecosystem services and relevant indicators for analyzing them; and (3) introduce a framework for the assessment of ecosystem services. This assessment framework consists of five pillars: ecosystem properties, potentials, services, benefits/values, and beneficiaries. The framework emphasizes especially their potentials as an intermediate step between biophysical processes and real services. Moreover, it focuses on human feedback to ecosystems through decision-making, utilization and management; it particularly addresses space and time issues. Finally (4), the chapter illustrates the given framework using the example of two terrestrial ecosystem types which frequently occur in river basins: semi-natural grasslands, and farmland. The first example involves a special focus on nature conservation issues with the complex implications, conflicts, problems, relationships and governance structures typical for ecosystems with restricted land use forms and intensities. The second example, farmland, is examined especially from the view-point of a rather new utilization demand: the production of energy crops, may imply a further intensification of agriculture with various impacts on bodies of water and watersheds, such as soil erosion, nutrient and pesticide spillovers, and water retention.

2 Defining Terrestrial Ecosystems

The term and concept of the *ecosystem* originates with Tansley, who introduced it to ecology as a basic principle (Tansley 1935). Since then, an international interdisciplinary and transdisciplinary ecosystem research community has emerged, and has attempted to develop and apply holistic and systemic concepts (e.g. Odum 1969). Ecosystem research is a conceptual approach with which particularly natural scientists identify themselves, since it enables analytical models of the structure and dynamics of spatial segments to be processed. This includes consideration of life environmental relationships, albeit not necessarily as the central factor (Fränzle 1998; Jørgensen 2006–2007).

The strong natural-scientific focus of the ecosystem concept expresses the definition of Odum (1971) in his classical book on the fundamentals of ecology: "Any unit that includes all of the organisms (i.e. the 'community') in a given area interacting with the physical environment, so that the flow of energy leads to clearly defined trophic structure, biotic diversity and material cycles (i.e. exchange of materials between living and nonliving parts) within the system, is an ecological system or an ecosystem." In one of his more recent monographs, (1993) Odum used a much simpler and more “down to earth” definition for ecosystems: “The community and the non-living environment function together as an ecological system or ecosystem.” Ellenberg (1973) defined the ecosystem, as “an interacting system, formed by living organisms and their abiotic environment”.

Although those authors claimed to have a holistic ecosystem conception, it is a weakness of the definitions presented above that the position and role of humankind remained unclear, although even Tansley regarded humans as a major driver of change, and as an integral part of the ecosystem concept. In fact, Ellenberg realized very well the dichotomic position of humans, acting as a “supernatural factor” both inside and outside of the ecosystem (Naveh 2010). For the concept of ecosystem services, it is crucial to have a strong focus on the human sphere, and not to be caught up in the “natural ecosystem” paradigm, by which humans are considered as unwanted and disturbing external ecosystem agents, distorting nature’s harmony by deflecting the achievement of that homeostatic climactic stage.

Ecosystems can be classified according to their essential properties, e.g. the characteristics of their matter and energy cycles, and required space and distribution. Usually, *classification systems* rely on

- structure, e.g. size, space, trophic level
- geographical situation, e.g. alpine and snow levels
- water balance: terrestrial or aquatic ecosystems
- dynamics: energy flow, matter flow, succession.

Similar ecosystems can be combined to ecosystem types, e.g. according to naturalness (natural, semi-natural, non-natural or artificial ecosystems), or dominant land use (forest, agricultural, or urban ecosystems). Ecosystems with similar structural appearance but different combinations of species occurring in different geographical regions can be combined into plant formations, vegetation formations, vegetation zones, zono-biomes or ecological zones. Large coherent ecosystems of a region are known as eco-regions or biomes (Schmitt et al. 2012). The word “biome” is used to describe a major vegetation type which extends over a large geographic area. The ten major biomes are tropical rain forests, tropical dry forests, tropical savannah, desert, temperate grassland, temperate woodland, temperate forest, northwest coniferous forest, boreal forest and tundra (Fig. 1).

Classifying ecosystems into ecologically homogeneous units is an important step towards effective ecosystem management. For instance, the American geographer Bailey defined a scale-based hierarchy of ecosystem units ranging from micro-ecosystems (individual homogeneous sites, on the order of 10 km^2), through meso-ecosystems (landscape mosaics, on the order of 1,000 km^2) to macro-ecosystems (eco-regions, on the order of 100,000 km^2, Bailey 2009).

Lugo et al. (1999) defined ten characteristics of an effective ecosystem classification system: It should

- be based on geo-referenced, quantitative data
- minimize subjectivity and explicitly identify criteria and assumptions
- be structured around the factors that drive ecosystem processes

Fig. 1 Examples for important terrestrial ecosystem types (biomes) of the earth. *Top left*: Mountain rain forest in Sri Lanka (Photo: O. Bastian). *Top right*: Kalahari Desert, Namibia (Photo: O. Bastian). *Bottom left*: Temperate forest of Central Europe (Lusatian mountains, Czech Republic) (Photo: O. Bastian). *Bottom right*: Taiga and tundra, Denali National Park, Alaska (Photo: K. Grunewald)

- reflect the hierarchical nature of ecosystems
- be flexible enough to conform to the various scales at which ecosystem management operates
- be tied to reliable measures of climate so that it can "anticipate global climate change"
- be applicable worldwide
- be validated against independent data
- take into account the sometimes complex relationship between climate, vegetation and ecosystem functioning
- be able to adapt and improve as new data become available.

As open systems, ecosystems and physical landscapes have no fixed boundaries; rather, they are – to some extent – constructs, and can be differentiated and classified according to defined rules. That delimitation is a pragmatic decision driven by the issue/research question and the existing personal resources. A biocoenosis, a community of plant and animal species, can be the only or the most important criterion of the spatial delimitation of an ecosystem.

A *terrestrial ecosystem* is by definition an ecosystem found only on a landform. Such ecosystems cover around 144,150,000 km^2 (28.2 %) of the earth's surface (Wessells and Hopson 1988). In relation with land use, the compartments soil and bios play very important roles. Terrestrial environments are segmented into a subterranean portion, from which most water and ions are obtained, and an atmospheric portion, from which gases are obtained and where the physical energy of light is transformed into the organic energy of carbon bonds through the process of photosynthesis (cp. Leser 1997).

Due to its clear features, the vegetation cover is a factor primarily used to distinguish between terrestrial ecosystems. In the global dimension, the classification of terrestrial ecosystems can be combined with climate classifications (climatic zones) and subdivided into ecological zones according to quantifiable characteristics.

The geographical location of ecosystem regions usually depends on the amount of precipitation. Natural ecosystems reflect upon the variation of precipitation and temperature across the earth's surface, e.g. tundras, deserts, grassland ecosystems, or tropical forests. For the classification of ecosystems, land cover data of the CORINE LANDCOVER system of the European Environmental Agency (EEA 2000) is suitable. Table 1 shows a classification of ecosystems based on land use and land cover data (CORINE). The four main categories are artificial surfaces (such as urban areas), agricultural areas, forests and semi-natural areas, and – as non-terrestrial ecosystems – wetlands (Fig. 2). There should be no doubt that an absolutely clear differentiation between terrestrial and "non-terrestrial" ecosystems is almost impossible, but rather there are many links and feedbacks. This is especially the case for river basins. For example, precipitation leaches to the groundwater, which feeds the river sources. Soil particles and chemicals from terrestrial ecosystems (esp. arable fields) may pollute the surface waters. Conversely, rivers may flood adjacent farmland, forests and settlements.

Table 1 Ecosystem types according to the CORINE LANDCOVER classification (EEA 2000), using a land use and land cover classification from a European perspective (without seas and lakes; terrestrial ecosystems in standard face, *semi- or non-terrestrial ecosystems in italics;* TEC = symbols s, a, g, f, n – classification of terrestrial ecosystems used in Table 2 – see the legend there)

TEC	Level 1	Level 2	Level 3
s	1. Artificial surfaces	1.1. Urban fabric	1.1.1. Continuous urban fabric
			1.1.2. Discontinuous urban fabric
		1.2. Industrial, commercial and transport units	1.2.1. Industrial or commercial units
			1.2.2. Road and rail networks and associated land
			1.2.3. Port areas
			1.2.4. Airports
		1.3. Mine, dump and construction sites	1.3.1. Mineral extraction sites
			1.3.2. Dump sites
			1.3.3. Construction sites
		1.4. Artificial non-agricultural vegetated areas	1.4.1. Green urban areas
			1.4.2. Sport and leisure facilities
a	2. Agricultural areas	2.1. Arable land	2.1.1. Non-irrigated arable land
			2.1.2. Permanently irrigated land
			2.1.3. Rice fields
		2.2. Permanent crops	2.2.1. Vineyards
			2.2.2. Fruit trees and berry plantations
			2.2.3. Olive groves
g		2.3. Pastures	2.3.1. Pastures
a		2.4. Heterogeneous agricultural areas	2.4.1. Annual crops associated with permanent crops
a, g			2.4.2. Complex cultivation
a, n			2.4.3. Land principally occupied by agriculture, with significant areas of natural vegetation
a, f			2.4.4. Agro-forestry areas
f	3. Forests and semi-natural areas	3.1. Forests	3.1.1. Broad-leaved forest
			3.1.2. Coniferous forest
			3.1.3. Mixed forest
n		3.2. Shrub and/or herbaceous vegetation	3.2.1. Natural grassland
			3.2.2. Moors and heathland
			3.2.3. Sclerophyllous vegetation
			3.2.4. Transitional woodland shrub
		3.3. Open spaces with little or no vegetation	3.3.1. Beaches, dunes, and sand plains
			3.3.2. Bare rock
			3.3.3. Sparsely vegetated areas
			3.3.4. Burnt areas
			3.3.5. Glaciers and perpetual snow
–	*4. Wetlands*	*4.1. Inland wetlands*	*4.1.1. Inland marshes*
			4.1.2. Peatbogs
		4.2. Coastal wetlands	*4.2.1. Salt marshes*
			4.2.2. Salines
			4.2.3. Intertidal flats

Fig. 2 Examples for ecosystem types according to the CORINE LANDCOVER classification (Level 1). *Top left* – Urban fabric: the city of Beijing, China. *Top right* – Arable land: Pre-Inca field terraces in the Colca Canyon, Peru. *Bottom left* – Scrubland vegetation (*tomillares*) on the Portuguese coast. *Bottom right* – Inland wetland: Peatbog (raised bog) in the Ore Mountains of Saxony, Germany (Photos: O. Bastian)

The connection and the interdependencies of the different ecosystems or an ecosystem mosaic may be reflected properly by the *landscape* concept. A physical or geographical landscape is typically composed of a number of different ecosystems, each of which produces a whole package of different ecosystem services (MEA 2005). There are also several other names for such entities, e.g. geo-complex, natural complex, natural sphere, land unit, land system, eco-region, physical region, geochore, or, in German, *Naturraum* (Bastian et al. 2006).

Neef (1967) defined landscape as "a segment of the earth's surface characterised by a uniform structure, and the same structure of effects (process structure), of which the full integration of all geo-factors (geological subsoil, relief, soil, climate, water balance, flora, fauna, humankind and its works) of a site or a space consists." For the provision of many services the landscape as a whole and its associated ecological patterns may be greater than the sum of its parts. The landscape matrix determines the effectiveness and importance of the individual components, rather than simply adding them up (Vandewalle et al. 2008), and the landscape's character can be very important for the provision of ecosystem services. For example, the water conditions of a river depend on the ecosystem pattern and the geographical situation or the landscape complex of its catchment area.

3 Ecosystem Services of Terrestrial Ecosystems – A Classification

In view of the diversity and complexity of ecosystems and the services they supply, it is difficult to develop a classification of ecosystem services which is clear, widely accepted and meets broad requirements. With respect to the classification of ecosystem and landscape functions, potentials and services, there are numerous proposals, classification systems and divergent opinions. According to Costanza et al. (1997) and the Millennium Ecosystem Assessment (MEA 2005), it is possible to define the categories of *provisioning*, *regulating*, *supporting* and *cultural* ecosystem services. We, on the other hand, recommend a trinomial classification of ecosystem services according to the economic, ecological and societal categories of sustainability and risk. The breakdown into productive (economic), regulatory (ecological) and societal functions or services (Bastian 1997; Bastian et al. 2012a) has the advantage that it can be linked to both fundamental concepts of sustainability and risk using the established ecological, economic and social development categories. This is in line with the OECD (2008), which distinguishes between provisioning services, regulating services and cultural services. We consider supporting services an intermediate (analytical) stage. They are a prerequisite for defining the other three groups of services, but they are more related to the first pillar of our EPPS framework, ecosystem properties. Other authors also suggest treating them differently from the other ecosystem services which provide their benefits directly to humans. Due to thematic overlaps with regulating ecosystem services there is a high risk of double-counting particular natural processes (Hein et al. 2006; Burkhard et al. 2010; TEEB 2010).

It is worth mentioning that the concept of sustainable development may be implemented in land use and landscape management by applying another very important concept, that of *multifunctionality*. This presupposes that all human demands (on landscapes) and the wide variety of functions/services are considered (Wiggering et al. 2003).

In the following, we present an overview of ecosystem services supplied by terrestrial ecosystems, based on current knowledge (e.g. Costanza et al. 1997; de Groot et al. 2002; Müller and Burkhard 2007; Vandewalle et al. 2008; Grunewald and Bastian 2013; Haines-Young and Potschin 2013) and on our own experiences and reflections (Tables 2, 3, and 4). We classify the 27 ecosystem services according to three main categories (*provisioning*, *regulating*, and *socio-cultural* services), each with subdivisions, and provide a short definition and description, with examples, and mention selected indicators for the analysis or the assessment of the ecosystem services, with no claim to completeness. The indicators represent quite different types: bio-physical characteristics, yields, benefits/costs, damages/threats but also proxies (e.g. land use). We estimate the relevance of the particular ecosystem services for big groups of terrestrial ecosystems. As water, too, plays a crucial role in terrestrial ecosystems, we indicate its impact, differentiating between strong, medium, slight and no relevance for the ecosystem services concerned. This classification gives only a rough orientation. For example, "Regulation of pests and diseases" (R.9) and "Pollination" (R.10) are less dependent on water (sl) than "Preservation of biodiversity" (R.11), particularly with regard to water related habitats.

3.1 Provisioning Services

Ecosystems may provide many goods and services, from oxygen and water, through food and energy, to medicinal and genetic resources, and materials for clothing and shelter. As a rule, these goods and services refer to renewable biotic resources, i.e.

Table 2 Provisioning services of terrestrial ecosystems

Code/Name of the ecosystem services	Definition/ Description	Examples	Selected indicators	Terrestrial ecosystems	Water related
I Food (provision of plant and animal materials)					
P.1 Food and forage plants	Cultivated plants as food/forage for humans and animals	Cereals, vegetables, fruits, edible oil, hay	Harvested yields (dt/ha), contribution margin (€/ha)	a, g	me
P.2 Livestock	Slaughter and productive livestock from pasturing	Cattle, pigs, horses, sheep, goats, poultry	Stock density (livestock units per ha), contribution margin (€/ha)	g	sl

(continued)

Table 2 (continued)

Code/Name of the ecosystem services	Definition/ Description	Examples	Selected indicators	Terrestrial ecosystems	Water related
P.3 Wild fruits and game	Edible plants and animals from the wilderness	Berries, mushrooms, game	Shooting quota (animals per ha), yields (€/ha)	f, n	sl
II Renewable raw materials					
P.4 Wood and tree products	Raw materials from trees in forests, plantations or agroforest systems	Timber, cellulose, resin, natural rubber	Stock, growth, yields (m^3/ha), revenues (€/ha)	f	me
P.5 Vegetable fibres	Fibres from herbaceous plants (from nature or cultivated)	Cotton, hemp, flax, sisal	Yields (t/ha), revenues (€/ha)	a, n	me
P.6 Bio-energy	Biomass from energy crops and wastes	Fire wood, charcoal, maize, rape, dung, liquid manure	Yields (t/ha), energy amount (MJ/ha)	a, g, f, s	me
P.7 Other natural materials	Materials for industry, crafts, decoration, arts, souvenirs	Leather, flavorings, pearls, feathers, ornamental fishes	Sold units (e.g. furs per year), revenues (€/ha)	a, g, f, n	me
III Other renewable natural resources					
P.9 Genetic resources	Genes und genetical information for breeding and biotechnology	Seeds, resistance genes	Number of species, proportion of natural areas	g, f, n	sl
P.10 Biochemicals, natural medicine	Raw materials for medicine, cosmetics and others to enhance health and well-being	Etheric oils, tees, *Echinacea*, garlic, food supplements	Yields, amounts of active substance (kg/ha), revenues (€/ha)	g, f, n, s	me

Relevance of the ecosystem service for terrestrial ecosystems: *a* farmland, *g* grassland, *f* forests (a, g, f – more or less artificial, intensively used ecosystems), *n* natural/semi-natural ecosystems (e.g. heaths, dry meadows), *s* human settlements. Relationship of the terrestrial ecosystem to water: *st* strong, *me* medium, *sl* slight or none

Table 3 Regulating services of terrestrial ecosystems (Legend see Table 2)

Code/Name of the ecosystem services	Definition/Description	Examples	Selected indicators	Terrestrial ecosystems	Water related
I Climatologic and air hygienic services					
R.1 Air quality regulation	Air cleaning, gas exchange	Filter effects (fine dust, aerosols), oxygen production	Proportion of forests (%), leaf area index	f, n	me
R.2 Climate regulation	Impacts on the maintenance of natural climatic processes and on reducing the risks of extreme weather events	Cold air production, humidification, reducing temperature by the vegetation, weakening of extreme temperatures and storms	Proportion of forests and open areas (%), slope (°), albedo	f, a, g, n	st
R.3 Carbon sequestration	Removing carbon dioxide from the atmosphere and relocation into sinks	Photosynthesis, fixation in the vegetation cover and in soils	Proportion of vegetation areas (%), C-content of soils (t/ha)	f, n, g	me
R.4 Noise protection	Reducing noise immissions by vegetation and surface forms	Noise protection effects of vegetation	Vitality, layering and density of vegetation	f, n, s	sl
II Hydrological services					
R.5 Water regulation	Balancing impacts on the water level of watercourses, reducing and avoiding floods, droughts and (forest) fires	Natural soil storage, leaching/ groundwater recharge	Slope (°), land use (land cover) (%), soil types	f, g, n, a, s	st
R.6 Water purification (terrestrial)	Filter effects of the soil, decomposition by soil organisms	Nitrogen retention, denitrification	Soil form, substrate layering, ecological moisture degree	g, f, n	st
III Pedological services					
R.7 Erosion control	Effects of vegetation on soil erosion, sedimentation, capping and silting	Reducing soil losses	Slope (°), land use, permanent land cover, slope protection forests, crop spectrum, soil types	f, g, n, s	st
R.8 Maintenance of soil fertility	Regeneration of soil quality by the edaphon, soil generation (pedogenesis) and nutrient cycles	Nitrogen fixation, waste decomposition, humus formation and accumulation	Crop diversity, soil types, removal of harvest remnants and wood	f, g, a	sl

IV Biological services (habitat functions)					
R.9 Regulation of pests and diseases	Mitigating influences on pests and the spread of epidemics	Pest control by songbirds, lacewings, ladybirds, parasitic wasps, tics (*Encephalitis*)	Biocide application, naturalness and vitality of the vegetation, proportion of (semi-)natural vegetation areas (%), species spectrum (parasites, predators, pests)	f, n, g	sl
R.10 Pollination	Spread of pollens and seeds of wild and domestic plants	Honey and wild bees, bumblebees, butterflies, syrphid flies	Proportion of (semi-)natural vegetation areas (%), biocide application, proportion of flowering plants, genetically modified organisms	a, g, f, n, s	sl
R.11 Preservation of biodiversity	Conservation of wild species and breeds of cultivated plants and livestock	Refuge and reproduction habitats of wild plants and animals, partial habitats of migrating species, nursery spaces, cattle breeds	Natural/semi-natural vegetation (proportion %), naturalness structural diversity, biotope compound, number of species, rarity, endangering	n, f, s, g, a	st

Table 4 Socio-cultural services of terrestrial ecosystems (Legend see Table 2)

Code/Name of the ecosystem services	Definition/Description	Examples	Selected indicators	Terrestrial ecosystems	Water related
I Psychological-social goods and services					
C.1 Ethical, spiritual, religious values	Possibility to live in harmony with nature, Integrity of Creation, freedom of choice, fairness, generational equity	Bioproducts, sacred places	Natural/semi-natural vegetation (proportion %), extinct/threatened, genetically modified organisms, biocide application	n, f, g, s, a	st
C.2 Aesthetic values	Diversity, beauty, singularity, naturalness of nature and landscape	Flowering mountain meadows, harmonious landscape	Land use, vegetation types, crop diversity, relief diversity/slopes	s, n, f, g, a	st
C.3 Identification	Possibility for personal bonds and sense of home in a landscape	Natural and cultural heritage, places of memory, traditional knowledge	Natural and cultural monuments, historical landscape elements, architectural styles, persistence/continuity of landscape	s, n, g, f, a	st
C.4 Opportunities for recreation and (eco-) tourism	Conditions for sports, recreation and leisure activities in nature and landscape	Accessibility, security, stimuli	Level of accessibility, carrying capacity, snow cover, attractive species	f, s, g, n	st
II Information services					
C.5 Education and training values	Opportunities to gain knowledge about natural interrelations, processes and genesis, scientific research and technological innovations	Natural soil profiles, functioning ecosystems, rare species	Natural and cultural monuments, naturalness, land use forms	s, n, f, g, a	st
C.6 Mental, literary and artistic inspiration	Stimulating fantasy and inventiveness, inspiration in architecture, painting, photography, musics, dance, fashion, folklore	Impressive landscapes, mounts, old trees	Natural and cultural monuments, diversity of the land	n, f, g, a, s	st
C.7 Environmental indication	Gaining knowledge of environmental conditions, changes and threats by visually perceptible structures, processes and species	Indication with lichens (air quality), indicator plants (site conditions)	Species spectrum (ecological groups), number of lichen species, indicator organisms, naturalness	n, f, g, a	me

the products of living plants and animals. Abiotic resources (raw materials the earth's crust), wind and solar energy cannot be assigned to particular ecosystems; hence, they are not, in our view, to be considered ecosystem goods and services. Especially in ecosystems strongly modified by humans (e.g. farmland) it is difficult to differentiate between the natural and human inputs in labour, material and energy to a service or a good.

3.2 Regulating Services

The biosphere and its ecosystems are the main preconditions for human life. Such processes as energy transformation, mainly from solar radiation into biomass, storage and transfer of mineral material and energy in food chains, bio-geochemical cycles, mineralization of organic matter in soils, and climate regulation are essential for life on earth. On the other hand, these processes are influenced and enabled by the interaction of abiotic factors with living organisms. The existence and functioning of – among them also natural and semi-natural – ecosystems must be ensured, so that people will, in future, still be able to benefit from these processes. Due to the "merely" indirect benefits of regulating services, they are often not well known, ignored and not sufficiently considered until they are damaged or lost, although they are the basis for human life on earth.

3.3 Socio-cultural Services

Especially natural and semi-natural ecosystems provide manifold opportunities for enjoyment, inspiration, intellectual enrichment, aesthetic delight, and recreation. Such "psychological-social" services are not less important for people than are regulating and provisioning services; however, they are often neglected or not fully appreciated. One reason is the difficulty of valuating them economically, especially in monetary terms, which is even more difficult than for many provisioning and regulating services. A second group includes information services, i.e. the contribution of ecosystems to knowledge and education.

4 Assessment of Ecosystem Services: The EPPS Framework

According to the recently elaborated EPPS framework (*E*cosystem *P*roperties, *P*otentials and *S*ervices – Fig. 3, Bastian et al. 2013a), which is basing on the cascade model of Haines-Young and Potschin (2008) and TEEB (2010) considering various scientific schools of landscape ecology and the international scientific discussion, the assessment of ecosystem services should include the following steps (or pillars):

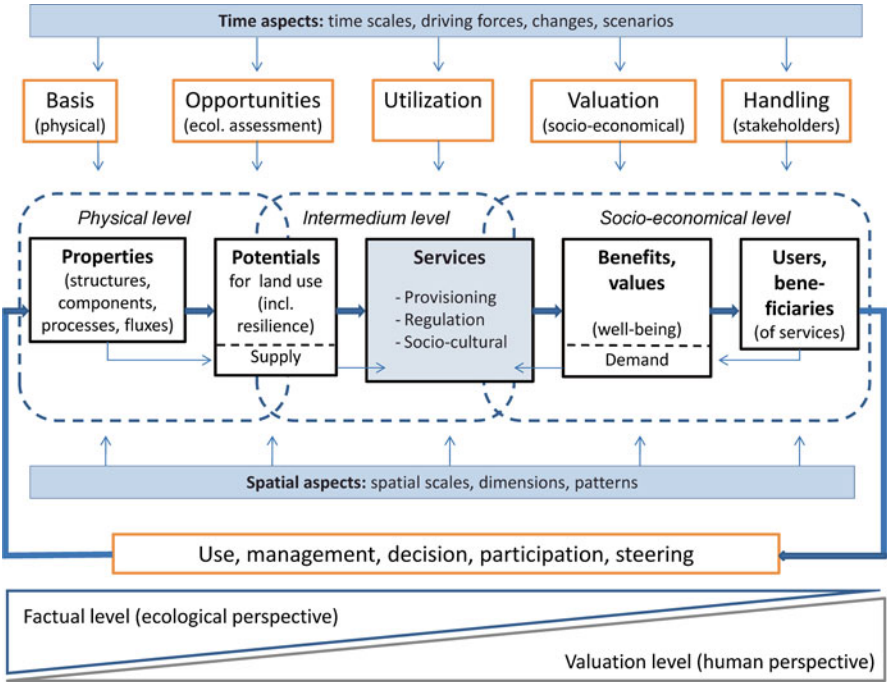

Fig. 3 Conceptual framework for the analysis of ecosystem services – the five pillar EPPS approach (Bastian et al. 2013a; Grunewald and Bastian 2015)

Pillar 1 – the *properties* of ecosystems, i.e. the set of ecological conditions, structures and processes (e.g. soil qualities, nutrient cycles, biological diversity) that determine whether an ecosystem service can be supplied. Such more or less value-free ecological categories as complexity, diversity, rarity, ecological integrity, or ecosystem health, also belong here. The analysis of ecosystem properties is predominantly driven by natural scientific methods, using analytical indicators.

Depending on their properties, ecosystems are able to supply services; they have particular *potentials* or capacities for that. Potentials have consciously been included as *pillar 2*, so as to distinguish between the possibility of use and an actual use or a demand for the use, which is the expression of the real service (Bastian et al. 2012a). Potentials can be regarded and quantified as stocks of ecosystem services, while the services themselves represent the actual flows (Haines-Young et al. 2012). In terms of the ecosystem potentials, various prerequisites need to be considered, e.g. the ecological bearing capacity. The assessment of ecosystem potentials also pursues the goal of ascertaining the potential use of particular services, and is more normative than a mere accounting of ecosystem properties. It constitutes an important basis for planning, e.g. for the implementation of sustainable land use systems: the suitability of an ecosystem to carry different forms of land use can be established, the available but still unused potentials can be put to actual use, and risks can be estimated.

Only human needs or demands actually convert a potential into a real service. *Ecosystem services*, the *pillar 3* of the framework, reflect an even stronger human perspective (value level), since the services (and goods) are in fact currently valued, demanded or used. In other words, the status of an ecosystem service is influenced not only by its provision of a certain service, but also by human needs and the desired level of provision for this service by society, which connects inseparably supply and demand of ecosystem services (Burkhard et al. 2012; Syrbe and Walz 2012).

The analysis of ecosystem services always involves a valuation step, i.e., scientific findings (facts) are transformed into human driven value categories. Through the link "ecosystem services", human beings benefit from ecosystems. That means, ecosystems yield *benefits and values (pillar 4)*, which contribute to human well-being. The benefit is the socio-cultural or economic welfare gain provided through the ecosystem service, such as health, employment and income. Moreover, the benefits of ecosystem services must have a direct relationship to human well-being (Fisher et al. 2009). Value is most commonly defined as the contribution of ecosystem services to goals, objectives or conditions that are specified by a user (van Oudenhoven et al. 2012). Actors in society can attach a (monetary or another) value to these benefits. Monetary value can help to internalize so-called externalities (impacts, side-effects) in economic valuation procedures so that they can be better taken into account in decision-making processes at all levels. It should be noted that not all dimensions of human well-being can be properly expressed in monetary terms, e.g. cultural and spiritual values (Spangenberg and Settele 2010).

An ecosystem service is only a service if there is a human benefit. Without human beneficiaries, there are no ecosystem services (Fisher et al. 2009). The stakeholders, providers, users or *beneficiaries (pillar 5)* of ecosystems and their services can be single persons, groups, or the society as a whole. They do not only depend or benefit from ecosystems, they in turn influence ecosystems. There may occur also losers, if a service is maximized on the expense of another service. An example is the down-stream living population damaged by floods, if the retention capacity of the upper river basin is reduced by the transformation of water-storing natural ecosystems into farmland or by surface sealing for housing.

The use and management of services (often regulated and controlled by decisions and legislation tools) can modify or change the properties and potentials of ecosystems. Appropriate *management* has to bridge the gap between the state and targets for ecosystem services.

All categories of the ecosystem services framework (ecosystem properties, potentials, services, values/benefits, providers/beneficiaries) can or should be analyzed and differentiated in terms of *space* (e.g. scale, dimension, pattern) *and time*, e.g. driving forces, changes, scenarios. Spatial aspects (e.g. minimum function areas, the arrangement of parts of the ecosystem or functional traits, ecosystem patterns and complexes, interactions between elements as well as fluxes of matter, energy and living-beings, service providing areas/service connecting areas and service benefiting areas) should always be taken into consideration. Spatial characteristics of ecosystem services are important not only for the assessments, but for the maintenance of ecosystem services as such (Bastian et al. 2012b; Syrbe and Walz

2012). Spatial aspects are crucial also for the concept of multifunctionality (Wiggering et al. 2003). For example, Brandt and Vejre (2004) defined three general types of multifunctionality from a spatial point of view: (1) spatial combination of separate land units with different functions, (2) different functions devoted to the same land unit, but separated in time, and (3) the integration of different functions on the same unit of land at the same time.

Finally, it should be taken into account that ecosystems, the potentials and services they deliver are subject to constant changes triggered by driving forces, e.g. climate change, demographic change, technological development. The state of ecosystems and ecosystem services can be predicted or assessed for different scenarios. Not only ecosystems and ecosystem services can change but also the value humans attribute to them.

5 The Example Semi-natural Grassland and Nature Conservation

Among the terrestrial ecosystems, grassland is a very important type, occurring in a wide range of naturalness, from natural grasslands (e.g. in the African or Central Asian steppes or as alpine grassland in the high mountains) through semi-natural meadows and pastures (e.g. mountain meadows) to the intensively used, non-natural grasslands which have arisen under human influence, and require regular utilization or management measures. In the context of river basins, too, grasslands play a crucial role, e.g. in terms of water and nutrient balance (groundwater recharge, decreasing water runoff, erosion and matter or nutrient outflows). Europe's grasslands covers some 90 million ha, and account for at least one third of its agriculturally used land. In Germany, the proportion of grassland is more than one quarter (5 million ha), although that share is declining.

Semi-natural grassland (Figs. 4 and 5) includes meadows and pastures with a near-natural species composition and low land use intensity. According to geographical location, nutrient content, water balance of soils, and management regimes (e.g. frequency of mowing, fertilization), there are quite different types of semi-natural grasslands. If they are especially rich in species and/or are ecologically important and valuable, they are classified as "High Nature Value Grassland" (in Germany: approx. 16.8 % of total grassland area – BfN 2009).

Certain properties or combinations of properties (pillar 1) of the semi-natural grassland are necessary or responsible for the potential or capacity of the ecosystems, or of its parts (functional traits – de Bello et al. 2010), to supply ecosystem services. Semi-natural grassland is potentially able to supply manifold ecosystem services in all three classes described (pillar 2):

- *Provisioning* (economic) *services*: Probably the most important role of grassland ecosystems is to provide forage for livestock. For some organisms, including medicinal plant and also, some fungi and animals, only natural and semi-natural grassland ecosystems are suitable habitats.

Fig. 4 Species-rich mountain meadow in the Ore Mountains (Saxony, Germany) (Photo: O. Bastian)

Fig. 5 Arnica (*Arnica montana*), a rare and threatened plant species of mountain meadows, and an indicator for nutrient-poor soil conditions (Photo: O. Bastian)

- *Regulating* (ecological) *services*: As semi-natural grasslands provide habitats for thousands of pollinator species, including bees, flies, moths, beetles, birds, and butterflies, they score highly in this respect. Moreover, their contribution to overall biodiversity is high. Due to their low nutrient levels, semi-natural grasslands provide protection for drinking water. All grassland types contribute to carbon sequestration and to balance the local climate. They also play a significant role in erosion control and flood mitigation, which is particularly important for river basins.
- *Socio-cultural services*: Depending on their wealth of flowers, some semi-natural grasslands, the mountain meadows, may provide exceptional aesthetic/scenery services. Due to their richness in plants and animals, natural and semi-natural grasslands are also suitable for environmental education and bio-indication. Semi-natural grassland also represents a type of historical landscape element which documents the cultural and economic life of former human generations in the landscape (Bastian et al. 2013b).

Not all the potentials (pillar 2) of semi-natural grasslands rich in species but low in yields are actually used (pillar 3). For example, their biomass is used little or not at all in highly intensified agricultural systems, because the nutritional value of the plants is too low for high-yield dairy cows. Until today, there is also no real market for their energetic use. As the amount of the plant biomass grown on such grasslands plays a limited role in terms of provisioning services compared to intensively used high-performance grassland, we can assign only reduced economic value or benefit to it (pillar 4). The situation with regard to socio-cultural services is quite different. The beauty of rich flowering meadows supports human well-being, and thus makes holiday regions more attractive, so that increasing numbers of tourists who come to see the meadows (like in the Ore Mountains in Saxony/Germany – Bastian et al. 2010), and pay money for food and accommodation, do represent an economic value. Both the tourists and the tourism providers are thus the beneficiaries of these meadows (pillar 5). With respect to biodiversity on the regional, national, European or global levels, the circle of beneficiaries covers the entire society – or even all of humankind. There are also significant benefits resulting from regulating services in river basins, e.g. due to the higher water quality and the reduced soil erosion.

Semi-natural grassland relies on appropriate human management measures, which are absolutely necessary to maintain the ecosystem and several services it provides. Labour power is needed for that purpose, e.g. by farmers, landscape management associations or environmental organizations. Those who ensure the maintenance of the grassland and its ecosystem services are not necessarily identical with the beneficiaries, e.g. the tourists or society as a whole (pillar 5), which has an interest in preserving biodiversity, as many laws, regulations, conventions and strategies at various levels show. Therefore, society often pays for those management measures, e.g. via nature conservation programmes, agro-environmental payments or other incentives. Moreover, the state ensures the designation of protected areas (e.g. nature reserves, the European network Natura 2000).

If farmers avoid over-intensive use or ploughing of grassland, they support the goals of the EU Water Framework Directive as well as those of climate protection. By analyzing abatement costs, damage costs and willingness-to-pay, Reutter and Matzdorf (2013) showed that significant monetary values can be generated, which may be understood as arguments for the maintenance of less intensively used, species-rich grasslands. The ecosystem services concept promotes the complex view on these valuable ecosystems. Thus, decision-makers could be encouraged to give incentives for management measures, e.g. through agro-environmental payments.

All these levels, beginning from the ecosystem "semi-natural grassland" (physical level), through the ecosystem services (intermediate level), to the benefits and beneficiaries (socio-economic level) are subject to various space and time conditions. For example, at the ecosystem level, the size of the meadow or its position in the biotope pattern may be important for meeting the requirements of particular species, e.g. the minimum required habitat areas of animals. A large meadow may provide more services than an identical, but much smaller one, e.g. with respect to aesthetic impression. River basins cover different hierarchical levels, from small rivulets to big streams. There are spatial aspects, too, also with regard to benefits and beneficiaries: The local landscape management association ensures the maintenance of valuable meadows in a rural district, but external visitors also benefit from the aesthetic values. The service "maintenance of biodiversity" should not be seen as spatially limited, but rather as having a national or even an international dimension, e.g. if a single meadow hosts the whole population of a threatened plant species.

The time-related conditions to be considered include changes in the ecosystem pattern of a river basin, or alterations in single ecosystems, which, in the case of semi-natural grassland, especially involve lacking or improper management. It is possible that, over time, people's attitudes toward nature conservation issues will change, as a result of such factors as globalization, the EU's Common Agricultural Policy (CAP), and technological progress, which could reduce the value of semi-natural grassland for farmers. Demographic change (rural population decline, especially in eastern Germany) could lead to a shortage staff of non-governmental conservationist groups, and hence to fewer actors to care for the meadows (Walz et al. 2013). Climate change, too, could affect these sensitive ecosystems.

6 The Example of Farmland and Energy Crops

According to various official statistics, the currently usable arable land of the earth is about 1.5 billion ha. In Germany, nearly 12 million ha are used as farmland. Although its task is primarily to provide food, fodder and raw materials, other ecosystem services should not be underestimated, e.g. groundwater recharging or the socio-cultural services which rely on typical agro-biodiversity. Ecosystem properties (pillar 1) important for several services provided by farmland, such as food provision or erosion control, include the soil form (grain size, nutrient content), the water balance and the local climate (Bastian and Schreiber 1999).

In recent times, an additional task has been assigned to farmland: the production of energy crops. European and German energy policies have moved towards the promotion of a significantly higher share of renewable energy resources, including biogas and fuel from energy crops. It is estimated that for 2012, energy crops were cultivated on 2,526,000 ha, or more than 21 % of Germany's agricultural land. The most important crops in 2012 were rapeseed for biodiesel and blending on 913,000 ha, and various other crops for biogas production on 962,000 ha, the largest share being maize (Fig. 6) for biogas production, which accounted for 800,000 ha of that (FNR 2012).

The extended cultivation of energy crops may lead to conflicts, e.g. severe impacts on various ecosystem services (or their potentials – pillar 2) due to its effects on groundwater, soils, biodiversity and the overall appearance of the landscape. Energy crops compete in space with food production, with a variety of ecological, economic and social impacts. The addition of another demand, bio-energy provision, upon agricultural land, results in an additional overemphasis on provisioning services over other ecosystem services, resulting in so-called trade-offs, potentially at the expense of such other stakeholders as water managers, nature lovers or tourists (pillar 5).

Fig. 6 Recently, maize has become one of the most important crops in Germany and abroad (Photo: O. Bastian)

Energy crop production can interfere with such goals of nature conservation as biodiversity maintenance (Rode et al. 2005; Lupp et al. 2011). Negative impacts are seen in shortened rotation periods on farmland and vast crop monocultures (the "self-rotation" of maize). Currently, crops such as maize, with a high demand for nitrogen inputs and nitrogen spill-overs, are favoured, leading to the increased application of mineral fertilizers. Erosion processes (especially in maize fields) may be accelerated due to the extraction of organic material on vulnerable sites, e.g. slopes. The use of residues competes with nutrient cycling and humus formation, as well as with such regulating services as carbon storage and water retention, in addition to the competition mentioned above, of energy crops with food and raw materials production for industrial needs. Many of these processes may influence water balance and water quality in river basins essentially.

If farmers reduce intensity in favour of biodiversity and several regulating services (e.g. soil and water conservation) in the framework of agro-environmental programmes, they can obtain financial compensation for usage difficulties from society (EU and national funds). This is an example of payments for ecosystem services, which are demanded by society (pillar 4).

Spatial aspects manifest themselves e.g. in the size of arable fields: large fields are often more efficient for the farmer, since they permit the use of large machines, but negative for biodiversity and several ecosystem services. There is also a spatial relationship between the production of energy crops and the utilization of the biomass in biogas plants. The consumers of the electrical power (not necessarily of the heat) produced by the biogas, however, generally live far away from the source. Crop rotation, and changes in land use intensity due to the enhanced cultivation of energy crops and the elimination of set-asides are examples of the aspect of time-related conditions. The drivers of such changes include primarily the present energy policy, the demand for biomass as a "renewable energy" source.

The increased cultivation of energy crops and the resulting net enhancement of importance of the provisioning services aspect has repercussions upon the properties and potentials of sites. Soils, the water balance and biodiversity may be threatened, and the natural potentials, or the capacity of ecosystems to supply services, e.g. soil fertility or recreation value, reduced (pillar 2).

The supply of various regulating and socio-cultural services should be possible in the framework of "good agricultural practice". Society can stimulate farmers with financial incentives through agro-environmental schemes, so that they receive compensation payments for reduced management intensities and yields, and contribute to maintain or recover the multifunctionality of the landscape.

7 Conclusion

In river basins, natural conditions, land use structures and processes are very important, particularly in view of such hydrological aspects as water runoff, water retention, groundwater recharge or water pollution. They are closely interconnected with running waters, terrestrial ecosystems play a crucial role as a source of impacts on the waters, as a carrier of land use, or in connection with the generation of manifold ecosystem services.

Most of the ecosystem services listed in the tables are not limited to terrestrial ecosystems, but they can – regardless of their peculiarities – be provided both by terrestrial and aquatic ecosystems, e.g. the preservation of biodiversity or environmental indication. Hence, the EPPS framework presented may be applied not only to terrestrial but also to aquatic ecosystems (because both show properties and potentials, provide services and benefits/values, and beneficiaries can be identified). This framework can be applied world-wide, in all biomes or vegetation zones. The two examples of very common terrestrial ecosystem and land use types – intensively used farmland on the one hand and semi-natural grassland on the other, show the mostly complex interrelations in the context of ecosystem services. These are between ecosystem properties and ecosystem potentials, services, benefits/values and beneficiaries, with their feedback to the ecosystems through land use and management. All these issues are influenced or interfered by manifold space and time-related aspects.

References

Bailey RG (2009) Ecosystem geography, 2nd edn. Springer, New York

Bastian O (1997) Gedanken zur Bewertung von Landschaftsfunktionen – unter besonderer Berücksichtigung der Habitatfunktion. NNA-Berichte, Schneverdingen 10:106–125

Bastian O, Schreiber K-F (1999) Analyse und ökologische Bewertung der Landschaft, 2nd edn. Spektrum Akadem. Verlag, Heidelberg/Berlin

Bastian O, Krönert R, Lipsky Z (2006) Landscape diagnosis in different space and time scales – A challenge for landscape planning. Landsc Ecol 21:359–374

Bastian O, Neruda M, Filipová L, Machová I, Leibenath M (2010) Natura 2000 sites as an asset for rural development: the German-Czech Ore Mountains Green Network Project. J Landsc Ecol 4(2):41–58

Bastian O, Haase D, Grunewald K (2012a) Ecosystem properties, potentials and services – the EPPS conceptual framework and an urban application example. Ecol Indic 21:7–16

Bastian O, Grunewald K, Syrbe R-U (2012b) Space and time aspects of ecosystem services, using the example of the EU Water Framework Directive. Int J Biodivers Sci Ecosyst Serv Manage 8(1–2):5–16. http://dx.doi.org/10.1080/21513732.2011.631941

Bastian O, Syrbe R-U, Rosenberg M, Rahe D, Grunewald K (2013a) The five pillar EPPS framework for quantifying, mapping and managing ecosystem services. Ecosyst Serv 4:15–24

Bastian O, Walz U, Decker A (2013b) Historical landscape elements: part of our cultural heritage – a methodological study from Saxony. In: Kozak J et al (eds) Integrating nature and society towards sustainability. Environmental science and engineering. http://dx.doi:10.1007/978-3-642-12725-0_31. Springer, Berlin/Heidelberg, pp 441–459

BfN (Bundesamt für Naturschutz) (2009) Erfassungsanleitung für die HNV-Farmland-Probeflächen. Bonn

Brandt H, Vejre H (2004) Multifunctional landscapes – motives, concepts and perspectives. In: Brandt J, Vejre H (eds) Multifunctional landscapes, vol 1. Theory, values and history. WIT Press, Southampton/Boston

Burkhard B, Petrosillo I, Costanza R (2010) Ecosystem services – bridging ecology, economy and social sciences. Ecol Complex 7:257–259
Burkhard B, Kroll F, Nedkov S, Müller F (2012) Mapping ecosystem service supply, demand and budgets. Ecol Indic 21:17–29
Costanza R, d'Arge R, de Groot R, Farber S, Grasso M, Hannon B, Limburg K, Naeem S, O'Neill R, Paruelo J et al (1997) The value of the world's ecosystem services and natural capital. Nature 387:253–260
de Bello F, Lavorel S, Diaz S, Harrington R, Cornelissen JHC, Bardgett RD, Berg MP, Cipriotti P, Feld CK, Hering D, Martins da Silva P, Potts SG, Sandin L, Sousa JP, Storkey J, Wardle DA, Harrison PA (2010) Towards an assessment of multiple ecosystem processes and services via functional traits. Biodivers Conserv 19:2873–2893
de Groot RS, Wilson M, Boumans R (2002) A typology for description, classification and valuation of ecosystem functions, goods and services. Environ Econ 41:393–408
EEA (2000) Corine land cover technical guide (Addendum 2000) European Environment Agency EEA technical report 40, 105 pp. Prepared by Bossard M, Ferance J, Otahel J
Ellenberg H (1973) Ökosystemforschung. Springer, Heidelberg/Berlin/New York
Fisher B, Turner RK, Morling P (2009) Defining and classifying ecosystem services for decision making. Ecol Econ 68(3):643–653
FNR (2012) Fachagentur Nachwachsende Rohstoffe e.V.: Biogas. www.nachwachsende-rohstoffe.de. Accessed 23 Aug 2012
Fränzle O (1998) Grundlagen und Entwicklung der Ökosystemforschung. Handbuch der Umweltwissenschaften, 3. Erg. Lfg. 12/98, pp 1–24
Grunewald K, Bastian O (2015) Ecosystem services - Concept, methods and case studies. Springer, Heidelberg
Haines-Young R, Potschin MB (2008) England's terrestrial ecosystem services and the rationale for an ecosystem approach. Overview Report, DEFRA 33
Haines-Young R, Potschin MB (2013) Common International Classification of Ecosystem Services 261 (CICES): Consultation on Version 4, August–December 2012. Report to the European Environment 262 Agency, Nottingham
Haines-Young R, Potschin M, Kienast F (2012) Indicators of ecosystem service potential at European scales: mapping marginal changes and trade-offs. Ecol Indic 21:39–53
Hein L, Koppen K, van de Groot RS, van Ierland EC (2006) Spatial scales, stakeholders and the valuation of ecosystem services. Ecol Econ 57:209–228
Jørgensen SE (2006–2007) An integrated ecosystem theory. Annals of the European Academy of Science. EAS Publishing House, Liège, pp 19–33
Leser H (1997) Landschaftsökologie, 4th edn. Eugen Ulmer, Stuttgart
Lugo AE, Brown SL, Dodson R, Smith TS, Shugart HH (1999) The Holdridge life zones of the conterminous United States in relation to ecosystem mapping. J Biogeogr 26:1025–1038
Lupp G, Albrecht J, Darbi M, Bastian O (2011) Ecosystem services in energy crop production – a concept for regulatory measures in spatial planning? J Landsc Ecol 4:49–66
MEA (Millennium Ecosystem Assessment) (2005) Ecosystem and human well-being: scenarios, vol 2. Island Press, Washington, DC
Müller F, Burkhard B (2007) An ecosystem based framework to link landscape structures, functions and services. In: Mander Ü, Wiggering H, Helming K (eds) Multifunctional land use – meeting future demands for landscape goods and services. Springer, Berlin/Heidelberg/New York
Naturkapital Deutschland – TEEB DE (2012) Der Wert der Natur für Wirtschaft und Gesellschaft – Eine Einführung.- München, ifuplan: Leipzig, Helmholtz-Zentrum für Umweltforschung – UFZ Bonn, Bundesamt für Naturschutz
Naveh Z (2010) Ecosystem and landscapes – a critical comparative appraisal. J Landsc Ecol 3(1):64–81
Neef E (1967) Die theoretischen Grundlagen der Landschaftslehre. Haack, Gotha
Odum EP (1969) The strategy of ecosystem development. Science 164:262–270
Odum EP (1971) Fundamentals of ecology. W.B. Sounders Company Ltd., Philadelphia

Odum EP (1993) Ecology and our endangered life-support systems. Saunders Associated, Inc. Publishers, Sunderland

OECD (2008) Strategic environmental assessment and ecosystem services. http://www.oecd.org/dataoecd/24/54/41882953.pdf

Reutter M, Matzdorf B (2013) Leistungen artenreichen Grünlandes. In: Grunewald K, Bastian O (eds) Ökosystemdienstleistungen. Springer/Spektrum, Heidelberg/Berlin, pp 216–224

Rode M, Schneider C, Ketelhake G, Reißhauer D (2005) Naturschutz-verträgliche Erzeugung und Nutzung von Biomasse zur Wärme- und Stromgewinnung. BfN-Scripten 136, Bonn

Schmitt E, Schmitt T, Glawion R, Klink H-J (2012) Biogeographie.- Braunschweig: Westermann – Das Geographische Seminar

Spangenberg JH, Settele J (2010) Precisely incorrect? Monetising the value of ecosystem services. Ecol Complex 7:327–337

Syrbe R-U, Walz U (2012) Spatial indicators for the assessment of ecosystem services: providing, benefiting and connecting areas and landscape metrics. Ecol Indic 21:80–88

Tansley AG (1935) The use and abuse of vegetational terms and concepts. Ecology 16(3):284–307

TEEB (2010) The economics of ecosystems and biodiversity: ecological and economic foundations. Edited by Kumar P. Earthscan, London/Washington, DC

van Oudenhoven APE, Petz K, Alkemade R, Hein L, de Groot RS (2012) Framework for systematic indicator selection to assess effects of land management on ecosystem services. Ecol Indic 21:110–122

Vandewalle M, Sykes MT, Harrison, PA, Luck GW, Berry P, Bugter R, Dawson TP, Feld CK, Harrington R, Haslett JR, Hering D, Jones KB, Jongman R, Lavorel S, Martins da Silva P, Moora M, Paterson J, Rounsevell MDA, Sandin L, Settele J, Sousa JP, Zobel M (2008) Review paper on concepts of dynamic ecosystems and their services. RUBICODE Deliverable D2.1. http://www.rubicode.net/rubicode/RUBICODEeconferencereport.pdf. Accessed 05 Feb 09

Walz U, Bastian O, Kästner A, Wende W, Schwarze H (2013) Situation des Ehrenamts im Naturschutz. Naturschutz und Landschaftsplanung (NuL) 45(8):233–240

Wessells NK, Hopson JL (1988) Biology. Random House, New York

Wiggering H, Müller K, Werner A, Helming K (2003) The concept of multifunctionality in sustainable land development. In: Helming K, Wiggering H (eds) Sustainable development of multifunctional landscapes. Springer, Berlin/Heidelberg/New York, pp 3–18

Quantifying, Modelling and Mapping Ecosystem Services in Watersheds

Stoyan Nedkov, Kremena Boyanova, and Benjamin Burkhard

Abstract Quantifying, modelling and mapping ecosystem services is an important step to the application of ecosystem services in practice and decision making. Watersheds are functional entities that provide an appropriate spatial scale for water flow analysis and integrate all the processes that occur within their boundaries. Multiple ecosystem functions occur within watersheds, providing water-related ecosystem services such as freshwater provision, groundwater recharge, water purification and flood regulation. A matrix approach was applied, linking different land cover types within watersheds to different ecosystem functions and services. Supply capacities of different land cover types and respective changes over time were assessed. By applying the watershed-based hydrologic model KINEROS and the GIS based AGWA tool, water retention functions of different land cover classes in the Bulgarian case study areas Malki Iskar, Vidima and Yantra were assessed. Based on the modelling results, flood regulating ecosystem service supply capacities were quantified and mapped in the three watersheds. A digital elevation model, land cover information and accessibility data were used to compile maps of demands for flood regulating ecosystem services. Supply-demand budgets were calculated and mapped for the study areas using the flood regulation supply and demand maps. The results quantify and illustrate complex ecosystem function–service–benefit relations in watersheds. Comparable procedures and calculation algorithms can be applied for other ecosystem functions and services relevant on the watershed scale. The approach is transferable to other regions and can provide important information for integrated watershed management.

Keywords Flood regulation • Supply • Demand • AGWA • KINEROS

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1 Introduction

Water is the basis for numerous ecosystem functions and supply of related ecosystem services. Water is a core component of human well-being and activities (Vigerstol and Aukema 2011) and demands for water have been rising steadily (MA 2005). Therefore, the appropriate assessment of available water resources and their sustainable management are obligatory for long-term human development.

1.1 Watershed Management

Traditionally, watersheds were managed following the principal of maximum exploitation required for maximum economic development (Dehnhardt and Petschow 2008). As a result of this heavy human utilization, natural resource degradation and the demand for ecosystem services increased. As a consequence, the need for different management regimes became obvious (Burkhard et al. 2012b). Traditional "hard" measures to reduce negative impacts of water flows (i.e. flood hazards) include construction of new or reinforcement of existing flood defense infrastructure such as dykes and dams in vulnerable areas.[1] The potential of natural landscapes to mitigate the negative effects of extreme water-related phenomena (caused for example by extreme precipitation events, snow melt, coastal floods, Tsunamis) has usually been neglected (Jonkman et al. 2004). Negative effects include flood hazards like human fatalities, destruction of property or infrastructure, harvest or other land use losses.

There potentially exists cost-effective ways of achieving flood protection by profiting from nature's capacity to absorb excess waters. Under this premise, the European Union (EU) Floods Directive[2] was accepted in 2007 for all EU member countries. The aim of the Floods Directive is to provide guidelines for the reduction and management of the risks that floods pose to human health, the environment, cultural heritage and economic activity. The Floods Directive was proposed in coordination with the EU Water Framework Directive[3] (WFD) – a European regulation framework for the management of water that came into force in 2000. The objectives of the Water Framework Directive are, by 2015, to improve or to conserve the good status for surface and groundwater and to prevent any further deterioration. According to the guidelines, management actions to improve water quality have to be based on the watershed scale and not, as hitherto practiced, on administrative or political boundaries.

[1] http://ec.europa.eu/environment/water/flood_risk/better_options.htm

[2] http://ec.europa.eu/environment/water/flood_risk/index.htm

[3] http://ec.europa.eu/environment/water/water-framework/objectives/index_en.htm

1.2 Quantifying Ecosystem Services in Watersheds

Appropriate indicators are needed to quantify the processes by which water flows are regulated in watersheds. Indicators are also needed to assess the capacities for water-related ecosystem service supply. In de Groot et al. (2010), the use of the following state indicators have been suggested:

- "total amount of water (m^3/ha)" for water supply,
- "water storage capacity (buffer) in m^3" for natural hazard mitigation,
- "water retention capacity in soils" for water regulation; and as performance indicators,
- "maximum sustainable water extraction (m^3/ha/year)",
- "reduction of flood danger and prevented damage to infrastructure", and
- "quantity of water retention and influence of hydrological regime".

Some state indicators could be derived from primary data sources such as river discharge or amount of water in the soil, while others could be modelled.

1.3 Modelling Water-Related Ecosystem Services

Watershed-scale hydrologic models calculate various water balance parameters that can be used as indicators for different water-related ecosystem services. The choice of indicators produced from hydrologic models is a key part of an ecosystem services' assessment. The indicators should have clear and relevant cause-effect relationships to the ecosystem services (Kandziora et al. 2013a).

An example for an estimation of water retention capacities of different land cover classes and soil types is given by Guo and Gan (2002). Water storage capacity as an indicator for the damage mitigation function of floodplains and wetlands was used by Ming et al. (2007). However, it is more difficult to derive indicators for some regulating ecosystem services such as flood prevention or erosion regulation. Water regulating services depend not only on storage capacities but also on a number of other factors and functional processes such as interception and infiltration, surface parameters like roughness and slope as well as external factors like rainfall quantity and intensity, seasonal state of the vegetation and initial soil saturation (Nedkov and Burkhard 2012). However it is not always practical to apply complicated and time consuming hydrologic modelling, in which case ecosystem service modeling can be applied.

Vigerstol and Aukema (2011) have compared hydrological and ecosystem service models for water-related ecosystem services. In their review, Soil and Water Assessment Tool (SWAT) and Variable Infiltration Capacity (VIC) are presented as examples of traditional hydrological tools that focus on ecosystem service drivers but they require post processing for ecosystem service assessments. Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) and Artificial Intelligence for Ecosystem Services (ARIES) represent a new breed of ecosystem services specific tools, focusing mainly on end-services and visualization of these services across a landscape (Vigerstol and Aukema 2011).

In the example of the flood regulating ecosystem service we present in this paper, results from different hydrologic models can be used and the differences in model characteristics and selection of indicators should be carefully considered in consultation with hydrologists. For example, we apply results from distributed, single-event model KINEROS (KINematic Runoff and EROSion model). If results from semi-distributed, continuous hydrologic model like VIC (Variable Infiltration Model) are applied for the same assessment method, the output variables should be extracted for the event of interest and normalized with precipitation and time. In any case the scale of the assessment is determined by the scale of the model.

1.4 Mapping Ecosystem Services in Watersheds

Ecosystem service mapping is a useful tool to identify links across the many domains of integrated water resource management (Vigerstol and Aukema 2011; Liu et al. 2013). Spatially explicit ecosystem service assessments provide information and data about linkages between geobiophysical structures and processes, ecosystem functions, services and human benefits needed for sustainable balancing of different human use interests (de Groot et al. 2010). Watersheds provide the appropriate spatial scale for ecosystem service assessments because several ecosystem functions related to water cycling are taking place within their limits. Water-related ecosystem services such as freshwater provision, groundwater recharge, water purification and flood regulation are based on ecosystem structures and processes that occur at the watershed scale.

Mapping is a good tool for the creation of comprehensive ecosystem function, service and benefit visualizations. Maps have the capacity to represent and generalize spatial data and to support decision making (Martínez-Harms and Balvanera 2012; Crossman et al. 2012; Burkhard et al. 2013). Mapping visualizes and explains patterns, shows potential conflicts and limits in environmental management and indicates spatial mismatches between supply of and demand for different ecosystem services (Syrbe and Walz 2012). Nevertheless it is important to remember that each ecosystem service map is a model of real conditions, trying to reduce the complexity of human-environmental systems in an appropriate, logical and reproducible manner (Burkhard et al. 2009, 2012a).

In this paper we present an approach to quantify ecosystem service supply, demand and resulting budgets in three watersheds in Bulgaria. We use the example of flood regulating ecosystem services which are quantified based on hydrologic modelling, geobiophysical data and spatially explicit GIS analyses. The method has successfully been applied before in one of the three watersheds (Nedkov and Burkhard 2012). Thus, the main objectives of this contribution are:

- To further develop the ecosystem service quantification, modelling and mapping approach;
- To test the transferability of the approach in different watersheds; and
- To demonstrate the application to integrated water resources management at watershed scale.

2 Material and Methods

We use hydrologic modelling to quantify flood regulation functions of different land cover classes and then assign ecosystem service supply capacities to each class. Our method is based on the assumption that land cover classes in areas with high water regulation capacities (as calculated by the hydrologic modelling and the soil type assessment) have high flood regulating capacities. Thus, the results of the capacity assessments performed in the case study areas can be used for ecosystem service mapping in all areas where respective land cover and soil data are available.

2.1 Case Study Areas

The flood regulating ecosystem service assessment was applied in three watersheds in northern Bulgaria: Malki Iskar, Yantra and Vidima (Fig. 1). The watersheds occupy the northern slopes of the Stara Planina Mountains and have temperate-continental climate characterized by relatively warm summers and cold winters. The annual precipitation varies from 750 to 800 mm in their lower parts to 1,100–1,200 mm in the higher mountains. Extreme precipitation events are typical for these areas. Therefore there is high flood hazard (Nedkov and Nikolova 2006; Nikolova et al. 2007, 2008). The Malki Iskar watershed upstream the village of Svode covers an area of 664 km^2 and has an elevation ranging from 240 to 1,800 m. The Yantra watershed upstream the town of Veliko Tarnovo covers an area of 1,286 km^2 and has an elevation ranging from 140 to 1,550 m and the Vidima watershed covers an area of 562 km^2 and has an elevation ranging from 250 to 2,200 m.

2.2 Watershed Model AGWA

The GIS based AGWA (Automated Geospatial Watershed Assessment; Miller et al. 2002) tool and its constituent model KINEROS (KINematic Runoff and EROSion model; Semmens et al. 2005) were used to model hydrological processes. These models generate appropriate parameters for quantifying water-related ecosystem service indicators. A more detailed description of the model application is given in Nedkov and Burkhard (2012). Here we present further development and application

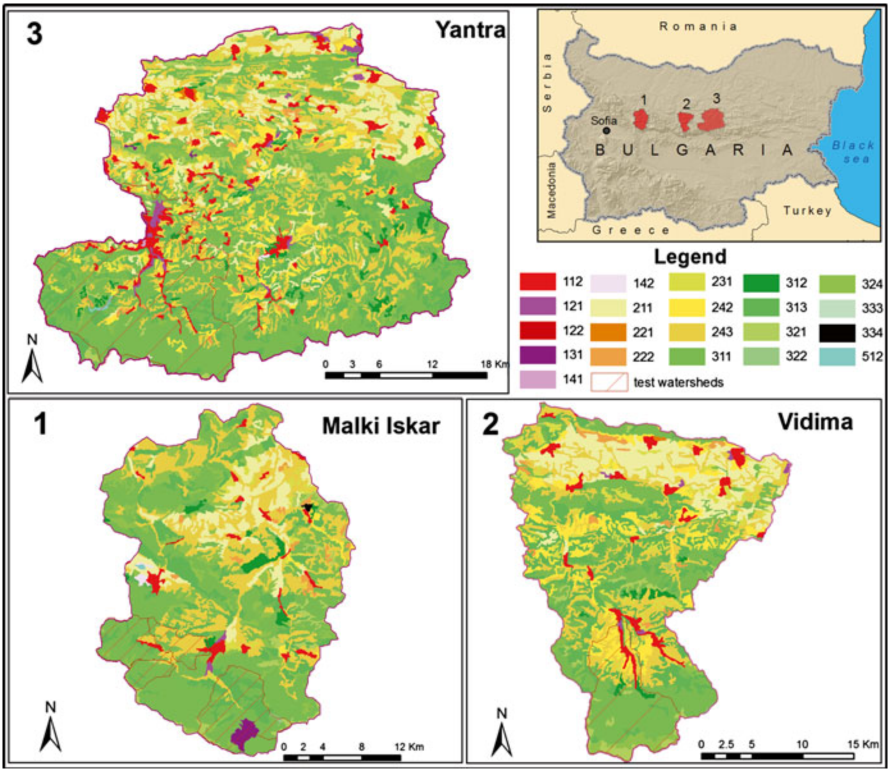

Fig. 1 CORINE Land Cover maps with test watersheds and the case study areas' locations (the names of CORINE classes are given in Table 3)

of the approach in three case study areas which enables us to verify the results of the capacity assessment and include more land cover classes which were not presented in the previous case study.

2.3 Database

The KINEROS model was run for the test sub-watersheds (see Sect. 2.4) using a 30 m Digital Elevation Model (DEM) derived from 25 m topographic maps by manual digitalization, digital soil map data (FAO/UNESCO 2003) and CORINE Land Cover (CLC; Bossard et al. 2000) data. As the spatial resolution of the standard CORINE data is too coarse (100 m geometric accuracy) for the model requirements, additional interpretations of Landsat ETM+satellite images with resolution 30 m taken in July 2006 and orthophoto maps with resolution 60 cm and 5 m

Table 1 Parameters of the watersheds where KINEROS modeling was applied

Watershed	Area (ha)	Number of model elements	Largest model element (ha)	Smallest model element (ha)	Average area of model elements (ha)	Average infiltration rate (mm)	Average peak flow (mm)	Average runoff (mm)
Yantra	28,614	67	1,522	0.1	427	42.9	6.4	4.1
Vidima	7,658	61	456.2	2.1	125.5	44.4	1.9	2.6
Ravna	2,557	65	126.7	0.5	39.6	11.1	1.5	3.7
Malki Iskar	5,459	58	380.1	4.9	94.1	12.5	3.1	2.3
Kobilya	1,004	35	107.6	0.2	28.9	8.4	1.1	2.4
Ablanitsa	1,537	55	121.5	0.5	28.1	9.9	1.3	5.2

geometric accuracy have been made for the test sub-watersheds. The interpretation was conducted manually. The resulting revised and amended Land Cover database, using the same CORINE nomenclature, contains more polygons of different land cover types and shows a more detailed representation of the landscapes. For instance, the Ravna test sub-watershed contains 14 land cover classes and 449 separate spatial units (individual vector polygons) while the standard CLC2000 has respectively 11 classes and 65 units for the same area. This ensures more precise hydrological modelling results as well as more representative data for the ecosystems service capacity assessment, as demonstrated by Kandziora et al. (2013b).

2.4 Hydrological Modelling in the Case Study Areas

The hydrologic modelling in the Malki Iskar case study area was performed in four test sub-watersheds: Ravna, Kobilya, Ablanitsa and Malki Iskar (the upstream part). The hydrologic modelling in the Yantra watershed was performed in the sub-watershed upstream the town of Gabrovo (Fig. 1) and the hydrologic modelling in the Vidima watershed was performed in the sub-watershed upstream the town of Apriltsi. The model was initially applied in the Ravna and Yantra watersheds because measured runoff data were available. The simulated data were calibrated against runoff and precipitation data for particular storm events with high discharge. The calibration for Ravna watershed was performed for an event that occurred on May 26, 2005 with precipitation of 59 mm and peak flow of 7.6 m^3/s. The data for Yantra were calibrated against an event on July 6, 1991 with precipitation of 48 mm and peak flow of 43 m^3/s. Then the model was applied for the other watersheds (Fig. 1), using the adjustments of Ravna event for Kobilya, Ablanitsa and Malki Iskar, and the adjustments of Yantra event for Vidima (Table 1).

Table 2 Value ranges of the model results for the indicators of flood regulation supply

Watershed/parameter		No relevant supply	Low relevant supply	Relevant supply	Medium relevant supply	High relevant supply	Very high relevant supply
Yantra	Infiltration	25.7–35.9	35.9–43.4	43.4–44.8	44.8–45.3	45.3–45.5	45.5–45.6
	Peak flow	20.3–12.4	12.4–7.5	7.5–5.4	5.4–2.6	2.6–1.2	1.2–0.1
	Surface Runoff	22.1–7.8	7.8–3.7	3.7–2.1	2.1–1.4	1.4–1.1	1.1–0.7
Vidima	Infiltration	34.1–43.8	43.8–44.8	44.8–44.9	44.9–45.0	45.0–45.2	45.2–46.6
	Peak flow	6.6–3.7	3.7–2.3	2.3–1.2	1.2–0.6	0.6–0.2	0.2–0.1
	Surface Runoff	13.7–3.4	3.4–2.1	2.1–1.8	1.8–1.7	1.7–1.5	1.5–0.5
Ravna	Infiltration	6.5–9.1	9.1–10.4	10.4–11.0	11.0–11.3	11.3–14.2	14.2–14.6
	Peak flow	5.1–3.1	3.1–2.1	2.1–1.1	1.1–0.6	0.6–0.2	0.2–0.1
	Surface Runoff	8.4–5.9	5.9–4.1	4.1–3.7	3.7–3.4	3.4–0.6	0.6–0.1
Malki Iskar	Infiltration	8.4–10.9	10.9–11.9	11.9–12.7	12.7–13.6	13.6–14.4	14.4–14.6
	Peak flow	11.7–6.1	6.1–3.6	3.6–1.9	1.9–1.2	1.2–0.5	0.5–0.1
	Surface Runoff	7.4–3.7	3.7–2.8	2.8–2.3	2.3–1.2	1.2–0.8	0.8–0.2
Kobilya	Infiltration	5.8–8.1	8.1–8.6	8.6–8.7	8.7–8.8	8.8–8.9	8.9–9.1
	Peak flow	6.3–4.5	4.5–2.1	2.1–1.4	1.4–0.9	0.9–0.5	0.5–0.2
	Surface Runoff	10.4–6.9	6.9–5.8	5.8–4.9	4.9–3.2	3.2–1.5	1.5–0.2
Ablanitsa	Infiltration	4.1–7.8	7.8–9.1	9.1–10.1	10.1–10.7	10.7–12.2	12.2–14.2
	Peak flow	5.1–2.1	2.1–1.6	1.6–0.8	0.8–0.4	0.4–0.2	0.2–0.1
	Surface Runoff	11.5–7.3	7.3–6.2	6.2–5.1	5.1–4.2	4.2–2.6	2.6–1.1

2.5 Capacity Assessment and Mapping of Flood Regulating Ecosystem Service

KINEROS is a semi-distributed model, therefore the test watersheds were divided into channel and plane model elements and AGWA calculates water parameters for each of them. The plane elements were used as basic spatial units for the assessment of flood regulation. The flood regulation supply capacities of these units were determined on the base of model results for infiltration, surface runoff and peak flow. The assumption is that high infiltration rates correspond to high capacities while the units of high surface runoff and peak flow have low capacities. The model result values for these parameters were analyzed and grouped into six categories using quantile statistical distribution. The ranges of these categories are given in Table 2. Then, the capacities of the spatial units were assessed on a relative scale ranging from 0 to 5 (after Burkhard et al. 2009). A 0-value indicates that there is no relevant capacity and a 5-value indicates the highest relevant capacity for the supply of flood regulation services in the study region. The capacities of the CORINE land cover classes were calculated using a GIS-based approach that includes spatial analyses and statistics directed to define the share of each land cover class within the modeling units with corresponding capacities. The supply capacities of the land cover classes in the study areas were assigned to every spatial unit in the GIS databases and they were used to generate flood regulation supply capacity maps. The approach is presented in more detail in Nedkov and Burkhard (2012).

A similar relative scale ranging from 0 to 5 was applied to assess the demands for flood regulation, where 0-values indicate that there is no relevant demand and 5 means very high demand. The calculations were based on the assumption that the most vulnerable areas would have the highest demand for flood regulation. The maps of the demand for flood regulating ecosystem services were prepared using topography data (30 m DEM and topographic maps), CORINE land cover data and statistical data for the areas which have been flooded during the recent flood events.

The maps of flood regulating ecosystem service budgets were created using spatial overlays of the supply and demand map layers. The measurement of flood regulation demand was transformed into negative values so that every polygon created from the overlay would have a value after the addition of the supply and demand values (Nedkov and Burkhard 2012).

3 Results

The overall relevant capacities of the different land cover classes were calculated as average values of the capacities obtained in the three watersheds (Table 3). The results for the Malki Iskar area were taken from the previous study which included modelling results of four test sub-watersheds (Nedkov and Burkhard 2012). Very small land cover classes within the modelled sub-watershed could not be accurately modelled. For example, smaller areas of bare rocks in forest-dominated landscapes would obtain high relevant capacities due to higher capacities of the forests but not of the bare rocks themselves. Therefore, land cover classes with areas less than 20 ha were excluded from further analysis.

3.1 Supply of Flood Regulating Ecosystem Services

Supply capacities were calculated for 19 of the 21 CORINE land cover classes present in the three watersheds. The CLC classes *Water bodies* and *Vineyards* occupy only small areas in the test sub-watersheds. Their capacities have been taken from the matrix based on expert evaluations published in Burkhard et al. (2009). The capacities of five land cover classes have the same scores in all three watersheds,

Table 3 Flood regulating ecosystem service supply capacities of the different land cover classes (ranging from 0 = no relevant capacity to 5 = maximum supply capacity in the study area; empty fields indicate that the land cover class was not present/has too small spatial extend in the respective watershed)

CORINE Land Cover class	Test watersheds			overall
	Malki Iskar	Yantra	Vidima	
112 Discontinuous urban fabric	0	0	0	0
121 Industrial or commercial units		0	0	0
122 Road and rail networks	0	0		0
131 Mineral extraction sites	0			0
141 Green urban areas		0		0
142 Sport and leisure facilities		4		4
211 Non-irrigated arable lands	1	0		1
221 Vineyards				0[a]
222 Fruit trees and berries	2	1	5	3
231 Pastures	2	2	5	3
242 Complex cultivation patterns		1	5	3
243 Agriculture & nat. vegetation	2	1	5	3
311 Broad-leaved forests	4	4	3	4
312 Coniferous forests	5	3	3	4
313 Mixed forests	5	5	5	5
321 Natural grasslands	3	4	0	2
322 Moors and heathland	3	2	2	2
324 Transitional woodland-shrub	3	3	1	2
332 Bare rocks			0	0
333 Sparsely vegetated areas		2	0	1
512 Water bodies				1[a]

[a]The capacities of *Vineyards* and *Water bodies* are taken from the matrix in Burkhard et al. (2009) as there are no modeling results

while the others are different (Table 3). The CLC classes *Green urban areas*, *Sport and leisure facilities* and *Bare rocks* occur in only one watershed, so their results need further verification. The largest variation between the values obtained in the three watersheds was observed in the agricultural classes (*Fruit trees and berries, Pastures, Complex cultivation patterns* and *Agriculture & natural vegetation*), ranging between 1 and 5.

The maps of flood regulation supply capacities (Fig. 2) show that all three watersheds contain large areas with relatively high capacities for flood regulation due to their high amount of forest cover. The areas of high and very high capacities cover about 50 % of the Malki Iskar watershed, 48 % of Yantra and 44 % of Vidima. They are located in the southern, more mountainous parts of the watersheds. The areas of no and low relevant capacities cover 11 %, 17 % and 16 % respectively. They are located in the middle and northern parts of the watersheds downstream of the more populated and cultivated areas. All three case study areas have similar distributions of the capacity classes within their watersheds. The standard deviation between the

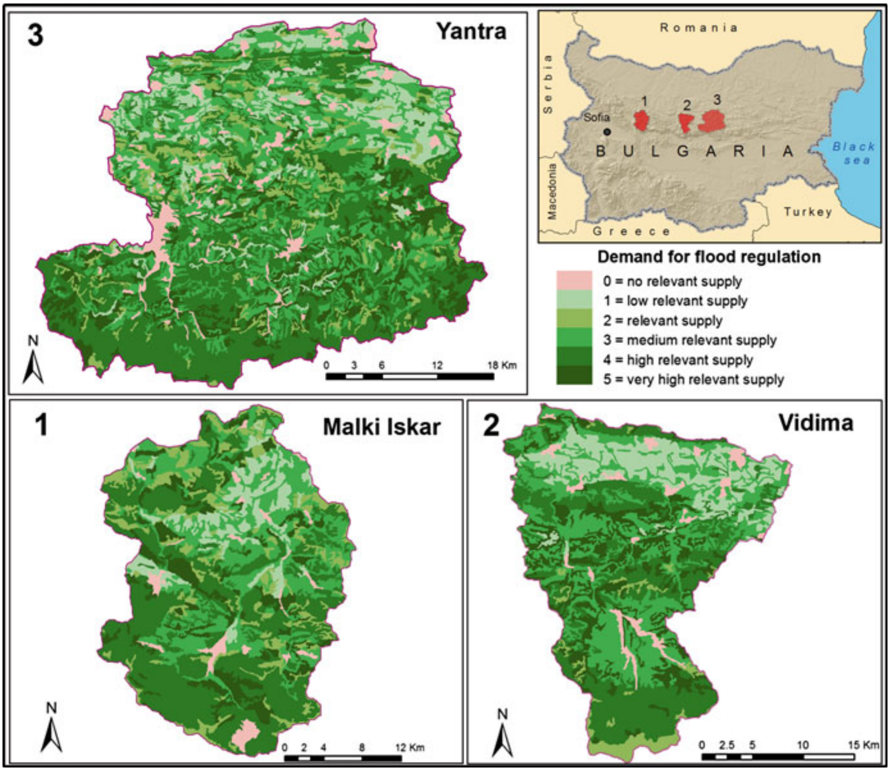

Fig. 2 Map of flood regulating ecosystem service supply capacities in the three study areas

percentages covered by each capacity class in the three watersheds varies from 1.1 % (areas of no relevant capacity) to 6 % (areas of high relevant capacity).

3.2 Demands for Flood Regulating Ecosystem Services

Unlike the supply capacities, which are spatially heterogeneous within each land cover class, the relevant demands for flood regulating ecosystem services can vary according to the specific location within the particular land cover classes' areas. For instance, the urban areas located in the most vulnerable parts of the floodplain have very high demand. The other urban areas within the floodplain have high relevant demands while those located outside floodplains have medium relevant demands (Table 4). The other artificial surface classes as well as the agricultural areas possess two different scores for the areas within and outside floodplains. The rest of the land cover classes have no relevant demands.

The maps of demand for flood regulating ecosystem services (Fig. 3) show that in all case studies the areas of low or no relevant demand far exceed the areas of

Table 4 Demands for flood regulating ecosystem services in the different land cover classes

CORINE Land Cover class	Watersheds								
	Malki Iskar			Yantra			Vidima		
112 Discontinuous urban fabric	2	4	5	2	4	5	2	4	5
121 Industrial or commerc. units	2	4		2	4		2	4	
122 Road and rail networks	2			2					
131 Mineral extraction sites	2			2					
141 Green urban areas				2					
142 Sport and leisure facilities	2	4		2	4				
211 Non-irrigated arable lands	1	3		1	3		1	3	
221 Vineyards							1		
222 Fruit trees and berries	1	3		1	3		1	3	
231 Pastures	1	3		1	3		1	3	
242 Complex cultiv. patterns	1	3		1	3		1	3	
243 Agriculture & nat. veget.	1	3		1	3		1	3	
311 Broad-leaved forests	0			0			0		
312 Coniferous forests	0			0			0		
313 Mixed forests	0			0			0		
321 Natural grasslands	0			0			0		
322 Moors and heathland	0						0		
324 Trans. woodland-shrub	0			0			0		
332 Bare rocks	0						0		
333 Sparsely vegetated areas	0			0			0		
512 Water bodies	0			0					

The figures represent demand classes as shown in Fig. 3 (from 0 = no relevant demand to 5 = very high relevant demand; empty fields indicate that the land cover class was not present/has too small spatial extend in the respective watershed; subdivisions in 2 or 3 demand classes within one land cover class refer to varying demands according to specific location and related vulnerabilities within this land cover class)

high and very high demand. The latter comprise 0.6 % in the Malki Iskar watershed, 0.7 % in Yantra and 1.6 % in Vidima which correspond to the results of our previous study (Nedkov and Burkhard 2012). The areas of low or no relevant demand cover 93 %, 90 % and 92 % respectively. The areas of relevant and medium relevant demand vary between 5.6 % (Malki Iskar) and 8.2 % (Vidima). Although the areas of high and very high demand have smaller spatial extends at watershed level, their share in the urban territories is much higher. It is about 39 % in Vidima, 24 % in Mali Iskar and 12 % in the Yantra watershed. This means that significant parts of the population live in areas with high and very high flood regulation demand although it varies between the different watersheds. Nevertheless, the overall distribution of the demand areas is quite similar in all three case study areas. The standard deviation between the percentages covered by each demand class in the three watersheds varies between 0.1 % (areas of no relevant demand) and 7.2 % (areas of high relevant capacity).

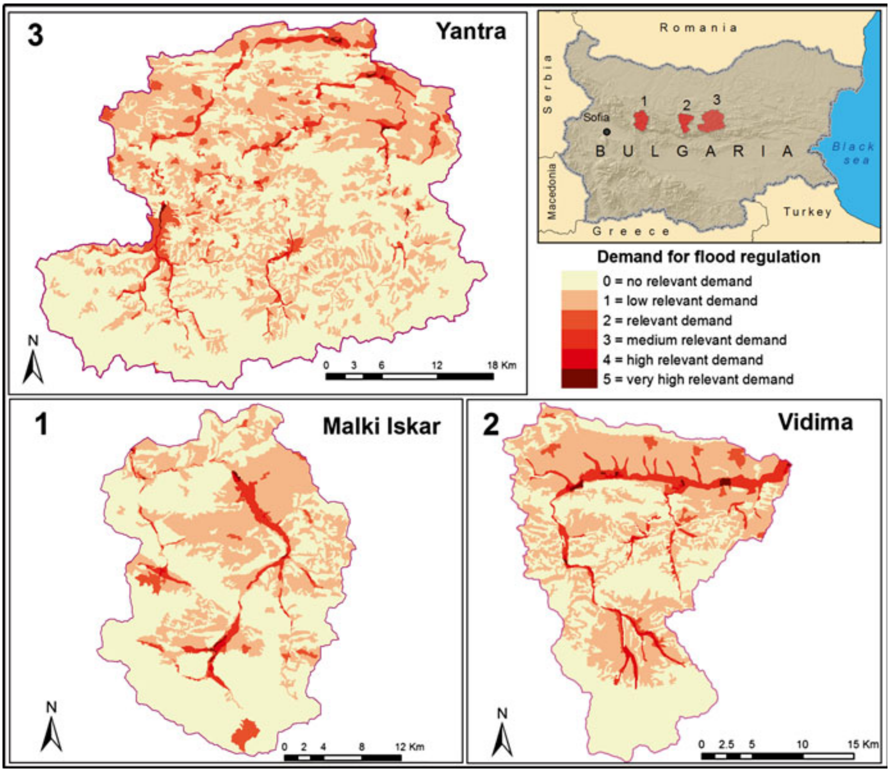

Fig. 3 Map of demands for flood regulating ecosystem services in the three study areas

3.3 Budgets between Flood Regulation Supply and Demand

The map of flood regulating ecosystem service supply and demand budgets (Fig. 4) shows that areas where supply exceeds demand predominate in all three case studies. They cover about 87 % in Malki Iskar, 82 % in Yantra and 78 % in Vidima. The supply demand budgets in most of these areas are 2 or 4. The opposite cases, where the demand exceeds the supply cover 5–6 % of the watersheds. This means that areas of high relevant demand are located mainly in places of low relevant supply capacities. Most of the areas of high and very high supply capacities preserve their relative share also in the map of supply/demand budgets, therefore they can effectively perform their flood regulation function. The areas of "0"-budgets (meaning the supply equals the demand) cover 15 % in Vidima, 11 % in Yantra and 8 % in Malki Iskar watershed. Most of them are located in agricultural areas which have low relative supply capacities and low demands.

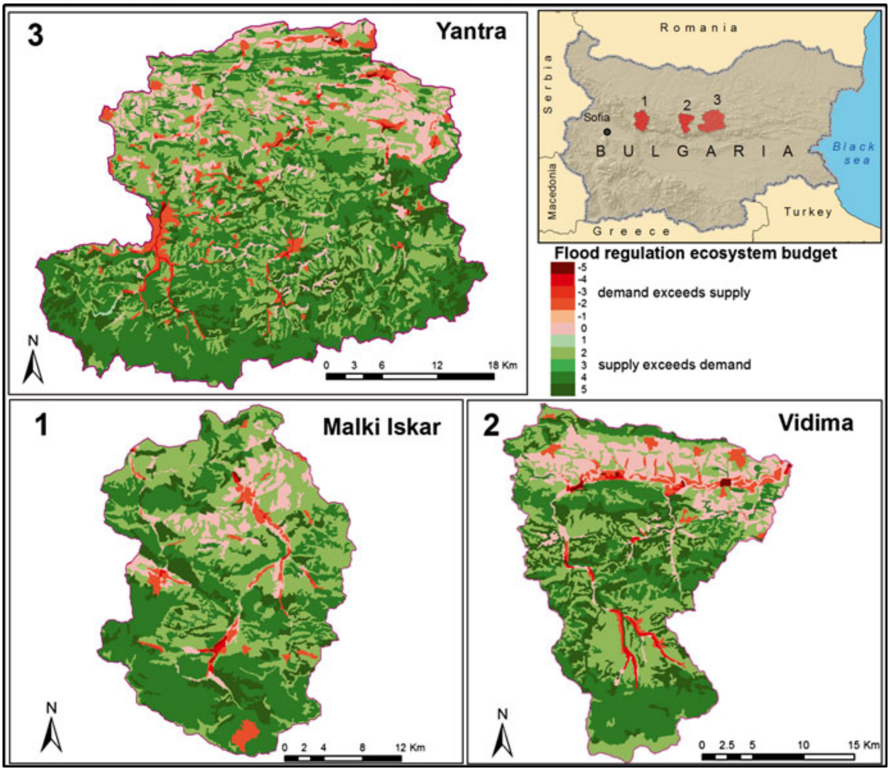

Fig. 4 Map of flood regulating ecosystem service supply-demand budgets

4 Discussion

Flood regulating ecosystem services can have preventing or mitigating functions. In the first case, the ecosystems (i.e. forests) redirect or absorb incoming water (from rainfall), reducing surface runoff and the amount of discharge from rivers. This ecosystem service takes effect before flood occurrence and in some cases it can prevent floods. KINEROS simulations of the peak flow under different scenarios of land cover change show significant differences in the river runoff and consequently the flood hazard (Nikolova et al. 2009; Nedkov 2010). However, the flood mitigation function takes effect when the flood is already formed. The ecosystems (i.e. flood plains and wetlands) provide retention space for the water surplus to spill, thus reducing the flood's destructive power (Nedkov and Burkhard 2012).

The interesting point with many regulating ecosystem services (including flood regulation) is that most of these ecosystem services cannot be transported or imported from other regions (like many of the provisioning ecosystem services) to the areas where the respective demands are located. Thus, the areas of ecosystem service supply (the Service Providing Areas – SPA; after Syrbe and Walz (2012)) must be physically connected (by Service Connection Areas – SCA) with the areas of demand (Service Benefitting Areas – SBA). Therefore, integrated water resource management has been discussed over that last few decades as a strategy for sustainable use of water. There is a clear relationship between the management of a river basin's land and water resources and the quality and quantity of the downstream water resource (Dehnhardt and Petschow 2008).

5 Conclusion

The flood regulation supply capacities modeled here in a land cover-based assessment emphasize the different land cover classes' functions and especially highlight high water regulation capacity of forests. Nevertheless there are still serious floods occurring regularly in all three study regions. This indicates that the flood regulating ecosystem service supply is not sufficient on the watershed scale, although large areas with high supply capacities for flood regulation have been modelled and mapped. The next step would be to calculate the areas of forest cover (the land cover class with the highest capacities for flood regulation) necessary to avoid/mitigate floods more efficiently, which would provide information for landscape planning and flood risk prevention. The research presented so far was focused mainly on the function of land cover (soils were assessed only in Malki Iskar watershed) in order to better understands regulation capacities. More precise results for flood regulating ecosystem services will be obtained in further works where soils and slopes are included in the assessment (Guo and Gan 2002).

This study together with the results presented in Nedkov and Burkhard (2012) can be considered as representative for the mountain watersheds in Bulgaria. These watersheds have high capacities for flood regulation due to the predominant forest land cover. The areas of high supply and demand are clearly distinguished and those of high supply far exceed the areas of high demand. The potential of our approach is its applicability on different spatial and temporal scales and with varying levels of detail, from rather easy land cover-based assessments to highly sophisticated model integration. Our methods allows for rapid as well as more comprehensive ecosystem service quantification, modelling and mapping in watersheds.

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References

Bossard M, Feranec J, Otahel J (2000) CORINE Land cover technical guide Addendum 2000. Technical report No 40. European Environment Agency, Copenhagen. http://www.dmu.dk/fileadmin/Resources/DMU/Udgivelser/CLC2000/technical_guide_addenum.pdf. [Online; Accessed 22 Mar 2013]

Burkhard B, Kroll F, Müller F, Windhorst W (2009) Landscapes capacities to provide ecosystem services–a concept for land-cover based assessments. Landsc Online 15:1–22

Burkhard B, Kroll F, Nedkov S, Müller F (2012a) Mapping ecosystem service supply, demand and budgets. Ecol Indic 21:17–29

Burkhard B, de Groot R, Costanza R, Seppelt R, Jørgensen SE, Potschin M (2012b) Solutions for sustaining natural capital and ecosystem services. Ecol Indic 21:1–6

Burkhard B, Crossman N, Nedkov S, Petz K, Alkemade R (2013) Mapping and modelling ecosystem services for science, policy and practice. Ecosyst Serv 4:1–3

Crossman ND, Burkhard B, Nedkov S (2012) Quantifying and mapping ecosystem services. Int J Biodivers Sci Ecosyst Serv Manage 8:1–4

de Groot RS, Alkemade R, Braat L, Hein L, Willemen L (2010) Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecol Complex 7:260–272

Dehnhardt A, Petschow U (2008) Governance in river basins – introduction and overview. In: Denhardt A, Petschow U (eds) Sustainability in river basins – a question of governance. Ökom Verlag, München

FAO/UNESCO (2003) Digital soil map of the World and derived soil properties, CD-ROM, Version 3.6. Information Division, FAO, Viale delle Terme di Caracalla, 00100 Rome, Italy

Guo ZW, Gan YL (2002) Ecosystem function for water retention and forest ecosystem conservation in a watershed of the Yangtze River. Biodivers Conserv 11(4):599–614

Jonkman SN, Brinkhuis-Jak M, Kok M (2004) Cost benefit analysis and flood damage mitigation in the Netherlands. HERON 49(1):95–111

Kandziora M, Burkhard B, Müller F (2013a) Interaction of ecosystem properties, ecosystem integrity and ecosystem service indicators – a theoretical matrix exercise. Ecol Indic 38:54–78

Kandziora M, Burkhard B, Müller F (2013b) Mapping provisioning ecosystem services at the local scale using data of varying spatial and temporal resolution. Ecosyst Serv 4:47–59

Liu S, Crossman ND, Nolan M, Ghirmay H (2013) Bringing ecosystem services into integrated water resources management. J Environ Manage 129:92–102

MA (Millennium Ecosystem Assessment) (2005) Ecosystems and human well-being: synthesis. Island Press, Washington, DC (World Resources Institute)

Martinez-Harms MJ, Balvanera P (2012) Methods for mapping ecosystem service supply: a review. Int J Biodivers Sci Ecosyst Serv Manage 8:17–25

Miller SN, Semmens DJ, Miller RC, Hernandez M, Goodrich DC, Miller WP, Kepner WG, Ebert DW (2002) GIS-based hydrologic modeling: the automated geospatial watershed assessment tool. In: Proceedings of the second federal interagency hydrologic modeling conference, Las Vegas, Nevada, July 28–August 1

Ming J, Xian-guo L, Lin-shu X, Li-juan C, Shouzheng T (2007) Flood mitigation benefit of wetland soil – a case study in Momoge National Nature Reserve in China. Ecol Econ 61:217–223

Nedkov S (2010) Modelling flood hazard due to climate change in small mountainous catchments. In: Car A, Griesebner D, Strobl J (eds) Geospatial crossroads @ GI_Forum'10 – Proceedings of the Geoinformatics Forum Salzburg, pp 172–176

Nedkov S, Burkhard B (2012) Flood regulating ecosystem services – mapping supply and demand, in the Etropole municipality, Bulgaria. Ecol Indic 21:67–79

Nedkov S, Nikolova M (2006) Modeling floods hazard in Yantra river basin. In: Proceedings from Balwois conference, Ohrid, May 23–26, 2006

Nikolova M, Nedkov S, Semmens D, Iankov S (2007) Environmental quality and landscape risk assessment in Yantra river basin. In: Petrosillo I, Müller F, Jones KB, Zurlini G, Krauze K, Victorov S, Li BL, Kepner WG (eds) Use of landscape sciences for the assessment of environmental security. Springer, Dordrecht

Nikolova M, Nedkov S, Nikolov V (2008) Modeling local dimensions of the climate change in Etropolska Stara Planina. In: Papers from the international conference global environmental change: challenges to science and society in the Southeastern Europe, Sofia 19–21 May

Nikolova M, Nedkov S, Nikolov V, Zuzdrov I, Genev M, Kotsev T, Vatseva R, Krumova J (2009) Implementation of the “KINEROS” for estimation of the flood prone territories of the Malki Iskar basin. Inf Secur Int J 24:76–88

Semmens D, Goodrich D, Uncrich C, Smith R, Woolliser D, Miller S (2005) KINEROS2 and the AGWA modeling framework. International G-WADI modeling workshop. http://gwadi.org/shortcourses/chapters/Semmens_L6.pdf

Syrbe R-U, Walz U (2012) Spatial indicators for the assessment of ecosystem services: providing, benefiting and connecting areas and landscape metrics. Ecol Indic 21:80–88

Vigerstol KL, Aukema JE (2011) A comparison of tools for modeling freshwater ecosystem services. J Environ Manage 92:2403–2409

A Methodology for Quantifying and Mapping Ecosystem Services Provided by Watersheds

Amy M. Villamagna and Paul L. Angermeier

Abstract Watershed processes – physical, chemical, and biological – are the foundation for many benefits that ecosystems provide for human societies. A crucial step toward accurately representing those benefits, so they can ultimately inform decisions about land and water management, is the development of a coherent methodology that can translate available data into the ecosystem services (ES) produced by watersheds. Ecosystem services (ES) provide an instinctive way to understand the tradeoffs associated with natural resource management. We provide a synthesis of common terminology and explain a rationale and framework for distinguishing among the components of ecosystem service delivery, including: an ecosystem's capacity to produce a service; societal demand for the service; ecological pressures on this service; and flow of the service to people. We discuss how interpretation and measurement of these components can differ among provisioning, regulating, and cultural services and describe selected methods for quantifying ES components as well as constraints on data availability. We also present several case studies to illustrate our methods, including mapping capacity of several water purification services and demand for two forms of wildlife-based recreation, and discuss future directions for ecosystem service assessments. Our flexible framework treats service capacity, demand, ecological pressure, and flow as separate but interactive entities to better evaluate the sustainability of service provision across space and time and to help guide management decisions.

Keywords Beneficiaries • Capacity • Demand • Flow • Pressures

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1 Introduction

Watershed processes – physical, chemical, and biological – are the foundation for many benefits that ecosystems provide for human societies. A crucial step toward accurately representing those benefits, so they can ultimately inform decisions about land and water management, is the development of a coherent methodology that can translate available data into the ecosystem services (ES) produced by watersheds. Such an approach needs to be founded on a conceptual framework of how ES are produced by ecosystems then delivered to society. Our main objectives in this chapter are to (1) establish links between water's flow through watersheds and the delivery of ES, (2) outline basic components of ES delivery and how they vary across types of ES, (3) describe selected methods for measuring ES components as well as constraints on data availability, (4) present several case studies to illustrate our methodology, and (5) discuss future directions for ES methodology.

1.1 Importance of a Watershed Perspective

A prominent feature of natural watershed hydrology is connectivity via flow of water, which also transports sediment, nutrients, pollutants, and organisms. Surface water flows longitudinally from uplands to floodplains and river channels, and from upstream channels to downstream channels; water flows laterally in both directions between floodplains and river channels; surface water flows to ground water and vice versa in multiple directions. Human alterations of these connections, via uses of land and water and built infrastructure, are pervasive and strongly affect delivery of the ES derived from connectivity. Accurate representation of the delivery of many water-related ES requires an understanding of natural hydrologic connectivity as well as how it has been modified by human actions. Further, new methods are needed to measure and monitor dynamics of aquatic ES, to assess their sustainability, and to guide their management (Arthington et al. 2010). Thus, a main goal of this chapter is to present methods to represent ES not as independent products but as manifestations of integrated physicochemical, biological, and social processes connected and occurring across entire watersheds.

Riparian corridors, encompassing floodplains and streambank communities along waterways, are especially important components of watersheds because of their roles in connecting upland and instream components. The type, extent, density, and vertical structure of riparian vegetation influences many ecological processes that contribute to ES, including water infiltration, instream production, nutrient cycling, channel and habitat formation, sediment transport, and groundwater storage. Thus, some of the methods we describe are focused on riparian corridors.

Human uses of land and water, such as deforestation, agriculture, impoundments, and urbanization, commonly alter land cover and soil properties in ways that affect the volume and temporal distribution of water and sediment movement across landscapes. Such changes in water and sediment movement can profoundly affect the ability of an ecosystem to provide certain ES. Human activities that directly affect flows of water and sediment in waterways include sand and gravel mining, channelization, levee construction, flow diversions, and woody debris removal. More common are extensive activities such as timber harvest, row-crop farming, livestock grazing, and wetland drainage, which indirectly affect flows of water and sediment via alteration of transport processes such as water infiltration, evapotranspiration, and soil erosion. An important objective underlying many ES analyses is to demonstrate the environmental tradeoffs associated with using land and water in different ways. Thus, our methods are designed to facilitate demonstration of such tradeoffs, especially in the context of sustainability. A watershed framework is well-suited to represent water and sediment movements, their consequences for ES delivery, and the spatiotemporal tradeoffs resulting from management decisions.

1.2 Relationships between Water Flow and ES Delivery

By definition, the term "ecosystem service" applies only to those ecological processes and features that confer clear benefits to people. Thus, many ecological and hydrological processes are not represented in ES and those that are may not be represented in every socio-cultural context. That is, what counts as a service strongly depends on social context. For example, nitrogen regulation may be a valuable service for people concerned about undesirable consequences of eutrophication, but, despite the fact that similar ecological and hydrological functions and processes would operate, it may not be recognized as a service in areas where the impacts of additional nitrogen in the system are not viewed as harmful.

The spatial extent over which ES are delivered varies greatly among specific services and environmental contexts. Some services flow locally (e.g. soil retention) and others globally (e.g. climate regulation), but in general the benefit zone, the area where a service is experienced, depends on the capacity of an ecosystem to provide a particular service as well as natural and anthropogenic connectivity across a landscape (Fig. 1; also Fisher et al. 2008; Bagstad et al. 2012). Potential beneficiaries are those people within the benefit zone (Hein et al. 2006; Boyd and Banzhaf 2007; Johnston and Russell 2011; Martin-López et al. 2012). For services like flood or water quality regulation, hydrologic principles apply: the capacity of an ecosystem to generate services upstream affects the potential receipt of benefits downstream. Therefore, a watershed approach is warranted. A watershed approach may also be appropriate for multi-ES assessments that include hydrology-dependent services. For services not tightly coupled to the direction of water flow, like timber production or most recreation, a watershed approach may be less necessary. The benefit zones for these services may be greatly extended by the transportation network (Fig. 1). We discuss the spatial independence between areas of ES generation and use in Sect. 4.

In addition to hydrologic connectivity, another feature of a watershed framework that makes it valuable in ES analyses is that watersheds are hierarchical. Smaller

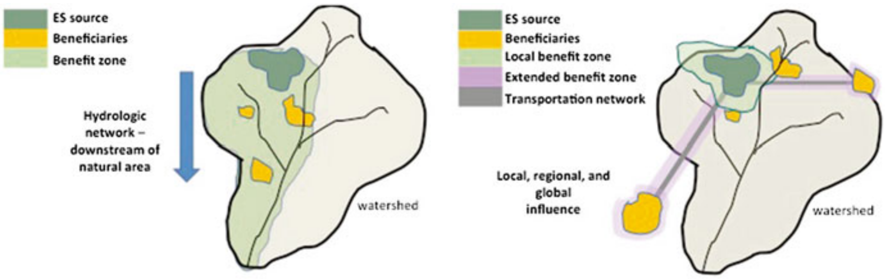

Fig. 1 The flow of ecosystem services can vary greatly depending on area of service production, its natural flow paths, as well as anthropogenic flow corridors. For many freshwater-related services the flow path is naturally hydrologic, where the capacity to produce a service upstream affects the flow of benefits downstream (*left*). Alternatively, the benefit zone can be extended by anthropogenic corridors like roads, canals, or exportation (*right*) (Figure from Villamagna et al. 2014a)

watersheds are spatially nested in larger watersheds; any level of that hierarchy can be used to examine a particular analytical question. For example, a regional-scale analysis of geographic variation in flood regulation might be better examined at a coarse spatial grain such as river basins, whereas a smaller-scale analysis of sediment regulation might be better examined at a finer spatial grain of intermittent stream watersheds. A watershed approach to ES analysis facilitates selection of the spatial scale most appropriate for a given problem or issue.

2 Distinctions among Provisioning, Regulating, Cultural ES

There are four distinct classes of ES that differ in how they are created, delivered, and experienced by humans (MA 2005; Fisher and Turner 2008). It is important to consider these differences when developing analytical frameworks and quantitative methods. Provisioning services, akin to ecosystem goods, are largely the tangible services that are feature-based and commonly accounted for in economic markets (Table 1). In contrast, cultural services are intangible and include the benefits derived from experiences in natural areas. Cultural services are especially challenging to fully quantify because they largely depend on individual experience, value sets, and preferences (Chan et al. 2012). We suggest that cultural services are founded on a mix of biophysical and societal features (Table 1). Unlike the tangible, market-tractable benefits of provisioning services, regulating services are the benefits derived from ecosystem processes that control valued environmental conditions, such as water quality and climate, or that are necessary for the production of other ES (MA 2005). In other words, regulating services collectively contribute to the resilience of an ecosystem. For most regulating and many cultural services, economic markets are not common and therefore our attention to measuring and monitoring these services has been limited. Finally, supporting service are the

Table 1 Ecosystem service (ES) delivery comprises four distinct components which differ among three ecosystem service categories. A general definition as well as categorical attributes and indicator examples are provided for each component

Components of ES delivery	Ecosystem service categories		
	Provisioning	Regulating	Cultural
Ecosystem service capacity: An ecosystem's potential to deliver services based on biophysical and social properties and functions			
ES categorical attributes	Biophysical capacity; feature-based	Biophysical capacity; process-based	Biophysical and social capacity; feature- and process-based
Indicator examples	Quantity of water, timber, fiber, crops	Infiltration, filtration, sequestration	Area of opportunity, site condition (e.g. accessibility, trails, biodiversity)
Ecosystem service demand: the amount of a service required or desired by society			
ES categorical attributes	Amount of service desired per unit space and time multiplied by the number of potential users (rival service)	Amount of regulation needed to meet pre-determined condition	Rival: measure of total use desired multiplied by number of potential users; non-rival: measure of individual use
Indicator examples	Liters of water required per person per year multiplied by number of users; kg/ha of crop production needed to feed beneficiaries (scale-dependent)	Percent reduction in [nutrient] needed to meet agreed-upon numeric criteria; volume of water retention needed to avoid flood	Rival: total visitor-days per year; non-rival: individual visitation rates; **estimates should be from earlier time or equivalent, nearby location*
Ecological pressures: anthropogenic and natural stressors that affect capacity or flow of benefits; often attributed to overuse or feedback from land management decision to enhance other service capacities			
ES categorical attributes	Events that reduce stock and/or regenerative capacity	Biological, physical, or chemical disturbances that increase the amount of ecological work required to meet societal demands	Events that reduce stock, regenerative, or assimilative capacity of a system; commonly related to overuse
Indicator examples	Overharvest; surface water impoundment; soil erosion	Biophysical or chemical pollutants (e.g. nitrogen fertilizer); land management (e.g. stream channelization); human disturbance (e.g. land conversion)	Erosion, soil compaction, run-off, decreased wildlife abundance caused by overuse
Ecosystem service flow: the actual production or use of the service; incorporates biophysical and beneficiary components			
ES categorical attributes	Quantity harvested, consumed, or used; number of people served; number of industries served	Ecological work conducted = ecological pressures minus ambient condition (same units)	Amount of service used measured in units of time and/or space
Indicator examples	Volume of timber cut or water consumed	Contaminant inputs *minus* in-stream contaminant load; Volume of water retained by ecosystem that avoids flood condition	Rival: total visitor-days per unit time and space; non-rival: individual visitation rates per unit time and space

ecological processes and features that contribute to the production and delivery of the other three service classes (e.g. habitat support may contribute to food production and wildlife-based recreation).

2.1 Capacity – Demand – Flow of Services

Few researchers to date have distinguished the capacity of an ecosystem to produce a service from the actual production or use of that service, the societal demand for that service, or the natural and anthropogenic pressures limiting the service (Burkhard et al. 2012; Nedkov and Burkhard 2012; van Oudenhoven et al. 2012; Tallis et al. 2012; but see Beier et al. 2008). However, such distinctions are crucial to ES analyses that can inform watershed managers and other decision-makers. The capacity of an ecosystem to generate services differs from the actual services delivered (i.e. service flow) to society (Villamagna et al. 2013). Delivered benefits depend not only on an ecosystem's capacity to provide services, but also on ecological pressures and societal demand for those services. Demand for an ES, which is driven by biophysical setting, population size, cultural preferences, and the perceived value of the service, can change independently of capacity, and vice versa. Thus, a snapshot of one component of ES delivery fails to capture the full ES dynamic from production to benefit (Fig. 2). To enhance our ability to quantify, map, and ultimately make ES information more accessible and useful to decision-makers, we must acknowledge the inherent differences among ES classes (Table 1), the dynamic processes by which ES are produced, and how ES benefit people (Carpenter et al. 2009; Bagstad et al. 2012; Chan et al. 2012). Notably, instructive knowledge of ES requires integrating expertise from both the natural and social sciences.

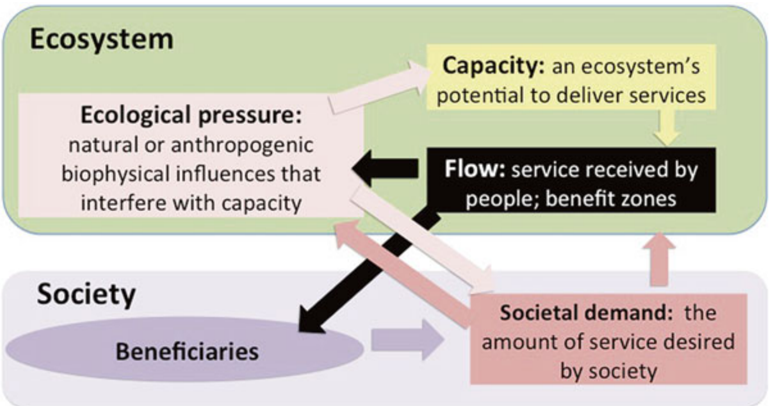

Fig. 2 Relationships among and definitions of key components of ecosystem service delivery. *Arrows* represent major effects

2.2 Definitions of ES Components

2.2.1 Capacity

Service capacity is an ecosystem's potential to deliver services based on biophysical features, social conditions, and ecological processes (Cairns 1997; Chan et al. 2006, 2012; Egoh et al. 2008; Daily et al. 2009; van Oudenhoven et al. 2012). ES capacity is site- and time-specific, but not constant; capacity responds to natural or anthropogenic changes over time and space (Villamagna et al. 2013, Fig. 2; also Burkhard et al. 2012; van Oudenhoven et al. 2012). For example, a change in land cover can alter the capacity of a landscape to regulate floods. Capacity ultimately limits long-term service provision; therefore, it can be considered the maximum potential service delivery under a given suite of conditions.

2.2.2 Demand

The demand for ES, a reflection of socio-cultural preferences, is the amount of service desired by society. Like ES in general, it can be measured at different scales (e.g. local versus global). Demand is largely independent of capacity, and may even exceed capacity. Human population density combined with average per capita consumption is an ideal indicator; however, precise data on consumption are often difficult to gather (Burkhard et al. 2012; Nedkov and Burkhard 2012). Not all services benefit all people within the benefit zone. The demand for some services, like cultural services, may come from a subset of the population (e.g. hunting and fishing are important services to a minority of the U.S. population; Villamagna et al. 2014a).

2.2.3 Flow

In contrast to capacity, which reflects the maximum potential service delivery, flow is the amount of a service received by people. Flow can be measured directly as the amount of a service delivered or indirectly as the number of beneficiaries served. Total service flow can be quantified as the service delivery per beneficiary multiplied by the number of beneficiaries. The economic assessment of a service is an extension of its flow that also requires knowledge of service value.

2.2.4 Ecological Pressures

Natural or anthropogenic influences, such as weather fluctuations or changes in land use, that make it more difficult for an ecosystem to meet societal demand for a service are called ecological pressures (MA 2005). Moreover, the source of a pressure can be related to overuse, like overfishing or crowding in recreation areas

(Rodiguez et al. 2006), or it can be a by-product of ES tradeoffs, like aquatic nutrient inputs from agricultural production (MA 2005). Sustained or extreme ecological pressures can alter the future capacity of an ecosystem to deliver services (Fig. 2; Carpenter et al. 2009).

2.3 Importance of Distinguishing ES Components When Assessing Sustainability

Separately measuring the components of ES delivery adds clarity to ES analyses and can enhance their utility in environmental planning and development. By distinguishing among ES capacity, demand, flow, and ecological pressures, we can (1) assess the current and future biophysical capacity of an area to produce ES, (2) evaluate the sustainability of ES use under different scenarios of ES demand, pressure, and capacity, and (3) examine how ES demand and ecological pressures influence biophysical capacity via feedback loops in which pressure may exceed ecological thresholds (Carpenter et al. 2009). Areas where service capacity is high and flow is low suggest additional use of the service can be sustained. Conversely, areas of medium to low capacity with relatively high service flow may not be sustainable. De Groot et al. (2010) provided a table of potential indicators for determining sustainability of ES; they distinguish between “state indicators”, similar to our conception of capacity, and “performance indicators”, measures of how much use a service can sustain. By comparing measures of current and future capacity, ecological pressures, and demand, planners can evaluate whether the needs of people can be met by existing ecosystem properties and processes or technological substitutes are needed to supplement service production. Analyses of these ES components also enable us to determine whether environmental costs and ES flows are distributed equitably (Tallis et al. 2012) or if the flow of services is sustainable over time (i.e. does not degrade capacity).

3 Conceptual and Quantitative Models

Constructing conceptual models that represent interactions among landscape features, processes influencing production of ES, and causal factors (Daily and Mattson 2008; Eigenbrod et al. 2010) is an important first step towards quantifying ES. Pictorial conceptual models provide a visualization of relationships among the ecosystem properties and processes that influence ES delivery (Fig. 3). For example, Keeler et al. (2012) use a simple pictorial model to illustrate complex relations among management actions, changes in water quality, changes in ES, and changes in social value. Well-designed conceptual models also help identify the key factors (properties or processes) that should be included in a quantitative and/or spatial assessment of ES. Models of cultural service capacity, like

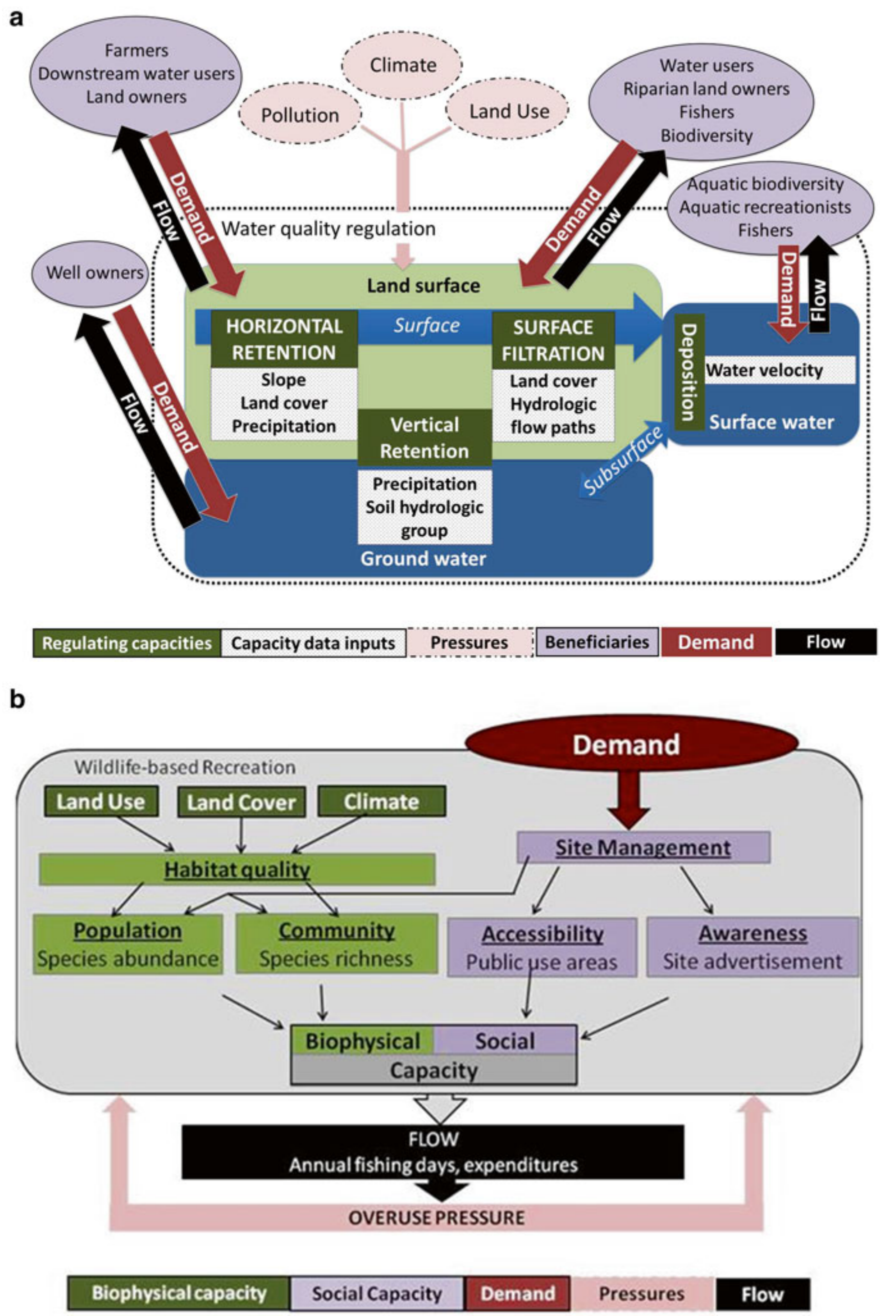

Fig. 3 Conceptual models of water quality regulation (**a**) and wildlife-based recreation (**b**). *Arrows* illustrate the relationships between capacity, demand, ecological pressures, and flow of services to beneficiaries (**a**). In each, ecological pressures are represented in *pink*, demand for the service in *red*, and the flow of the service in *black*. The water quality regulation model includes capacity features to illustrate the data inputs (*grey*) needed to map the processes (*green*); the wildlife-based recreation model distinguishes biophysical (*green*) and social (*purple*) capacity factors that should be included in assessments

wildlife-based recreation, are distinct from those of provisioning and regulating service capacities because they explicitly encompass social factors that influence experiential capacity (Fig. 3b). By integrating multiple factors that influence capacity, we can depict more of the real spatial and temporal heterogeneity involved in the production and delivery of ES. Furthermore, some conceptual models are transferable across landscapes and can facilitate comparable ES analyses based on widely available data.

We suggest several steps to developing useful conceptual models for ES analyses. First, data availability commonly limits ES analyses and can influence the analytical approach selected. Thus, a conceptual model should be prepared prior to a data hunt to ensure that the ecological foundation of the approach is sound and not unnecessarily influenced by data availability. In some cases, recognizing and acknowledging data gaps can lead to increased efforts to monitor important ecological and social processes. Second, a good conceptual model clearly illustrates relationships among features, distinguishes (perhaps symbolically) between features and processes, and includes exogenous and endogenous sources of variability that might affect model output. Third, conceptual models developed for multiple ES require comparable symbology across ES that clearly shows sources of ecological pressures (Fig. 3). In our work, we have developed separate conceptual models for each service, but a combined conceptual model also would be instructive for examining interactions among services and interpreting the potential tradeoffs associated with biophysical and socioeconomic changes to the landscape.

Conceptual models help identify the data needed to accurately depict ES delivery. These models can also clarify situations where a few landscape features adequately represent a service component versus the need for more sophisticated quantitative models of ecological processes. For example, water quality regulation (Fig. 3a) comprises several water purification services that are not captured by land cover or soil type alone. Instead, these complex processes have been represented by algorithms (Czymmek et al. 2003; Mayer et al. 2007); such vetted models of ecological and hydrological processes that regulate water quality can greatly enhance the transparency and consistency of ES assessments. For example, the U.S. Natural Resources Conservation Service's curve number method for estimating surface-water runoff (1972) incorporates land cover, land use, soil hydrologic grouping, and precipitation regime into its estimates. Similarly, the New York Nitrogen Leaching Index accounts for soil porosity, percolation, and seasonality of precipitation (Czymmek et al. 2003). For process-based models, site-specific primary data (e.g. precipitation) can enhance output accuracy. Based on the methodological comparisons of Eigenbrod et al. (2010), we expect estimates based on multiple factors of the contributing ecological processes and some primary data to provide more accurate results than those based on a single-feature proxy. Furthermore, these methods would be transferable across landscapes and mappable with widely available data.

4 Spatial Independence of ES Capacity, Demand, and Flow

By distinguishing capacity, demand, ecological pressure and flow from one another in conceptual models, we quickly realize that they are spatially independent. Although areas well endowed with ES have attracted people throughout history, modern technology has drastically increased our ability to live farther from ES-rich areas. Thus, we have effectively decoupled spatial links between ES capacity and flow. Nedkov and Burkhard (2012) discussed the spatial independence between capacity and demand for flood regulation, finding that downstream urban areas were hotspots of demand while flood regulation was generated in a less disturbed landscape upstream. Similarly, the benefits of water quality regulation upstream are independent from the demand for and use of clean water downstream. A more acute example may be for cultural services like recreation or aesthetic beauty in which beneficiaries travel to areas of high ES capacity to benefit from the service (Fig. 1). These examples illustrate how hydrologic processes or technology can promote spatial mis-matches among components of ES delivery. In a watershed context, the flow of services, such as flood regulation and water purification, is tightly linked to the directional flow of water. Thus, effects of changes in service capacity on service flow are detectable at downstream localities rather than at the source of capacity changes (Villamagna et al. 2013). Distinguishing capacity from demand, ecological pressure, and flow is an important step towards recognizing the full repercussions of landscape changes and understanding the tradeoffs associated with management choices (McShane et al. 2011).

5 Quantifying ES Components

5.1 Capacity

Capacity can be quantified and mapped by integrating the natural and anthropogenic factors that influence the ecological properties and functions that provide services (Egoh et al. 2008; Daily et al. 2009) regardless of how many people use or benefit from the services in question (Villamagna et al. 2013; Table 1). Provisioning service capacity is typically estimated from ecosystem properties (e.g. volume of water supply). Cultural service capacity is more difficult to quantify because it depends on a mix of biophysical and societal conditions (i.e., social capital; Chan et al. 2006, 2012; Villamagna et al. 2014a). Regulating service capacity is also challenging to quantify because it tends to comprise several interconnected processes that each rely on a distinctive suite of ecosystem properties (Fig. 3a). Thus measuring regulating service capacity requires extensive data that are often not available (Layke 2009).

5.2 Demand

Demand for provisioning services is, like capacity, fairly straightforward to quantify, although data availability varies greatly. Metrics of human population density and per capita consumption of services (e.g., drinking water) are typically used to estimate demand. For experience-based cultural services, the number of people wanting the experience (e.g. visitors to a park) can indicate demand. Likewise, the management priority assigned to a service may also reflect demand for that service (see Sect. 8.2). Since regulating services produce or maintain desirable environmental conditions, societal demand for them can be expressed as the amount of regulation needed to meet a desired end condition (e.g. the percentage reduction needed to meet numeric criteria for a pollutant), rather than the end condition itself. Estimating demand for regulating services is inherently challenging because it requires information about the ecological pressures needing regulation as well as the desired end conditions; the latter often are not well defined. To date, few assessments have quantified demand for regulating services in biophysical terms; instead it has been measured simply as human population (Nedkov and Burkhard 2012), which may be weakly related to the amount of regulation actually needed.

5.3 Flow

For provisioning services and some cultural services, measuring ES flow is straightforward (e.g. liters of water consumed or visitor-days enjoyed). For other cultural services, flow can be quantified by duration and quality of the experience with nature. These cultural services are inherently challenging to quantify and analyze because they are individualistic, difficult to aggregate, and sometimes influenced by social or moral factors (Chan et al. 2012).

Regulating services are process-driven and the data needed to directly quantify their flow are often unavailable at scales large enough to inform policy-making (Layke 2009). Ambient condition (e.g. water quality) is often used as an indicator of regulating service flow (Dale and Polasky 2007; Martinez et al. 2009; Jose 2009). However, ambient condition is the amalgamation of multiple service flows and pressures, expressed in the context of ecosystem capacity. Consider water quality in a real landscape. High ambient quality may be the result of high capacity or weak ecological pressures. Since a high-capacity system offers more resistance to ecological pressures, change in ambient condition is less and likely slower. In other words, regulation is occurring. A system with no (or very low) capacity (Fig. 4a, Villamagna et al. 2013) experiences quick decline in ambient condition (y axis) with increases in ecological pressure (x axis), while a high-capacity system can maintain acceptable ambient conditions under great ecological pressure (Fig. 4b). The regulating service flow is essentially the amount of "work" conducted by the ecosystem to regulate pressures.

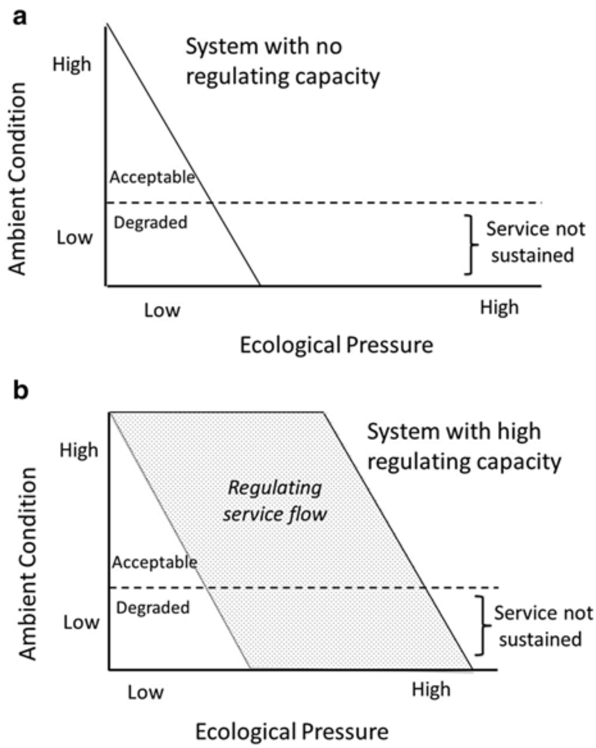

Fig. 4 Differences in service delivery and the effects of ecological pressure on ambient condition and ecological work within ecosystems with little to no capacity (**a**) compared to that of a system with high regulating capacity (**b**). Ambient condition is a function of regulating capacity and ecological pressure. In systems with little to no capacity (**a**), ambient condition is quickly degraded in response to increasing ecological pressure. Systems with higher capacity can maintain better ambient condition under greater ecological pressure (**b**). Ecological thresholds are determined by the ecosystem's capacity to provide a service. Once this threshold of ecological pressure is exceeded, ambient condition will degrade. The *shaded polygon* (**b**) illustrates the amount of ecological work performed (i.e. regulating service flow), which represents the difference between ambient condition and ecological pressure (Villamagna et al. 2013)

Systems with similar ambient condition and capacity may also differ in the flow of regulating services due to differences in ecological pressures. Consider two watersheds in which water quality is equal, meeting societal standards (i.e. demand), but that differ in ecological pressure, the amount of ecological work occurring is greater in the high-pressure system. One can also think analogously of an air conditioner working to cool a room. The same air conditioner set to the same temperature will work much harder on a hot day than on a cool day. Thus simply using ambient condition as a surrogate for the flow of regulating services ignores the potentially large effects of ES capacity and pressure.

6 Constraints on Data Availability

Assessments of ES are constrained by data availability (Eigenbrod et al. 2010) and explicit field measures of ES. Exact measures of ecological processes that produce services are not always a viable option, especially for large study areas. In the absence of data that directly reflect the service, we rely on proxies or indicators, which often require accepting assumptions (e.g. carbon sequestration rates based on tree species and volume); however, some indicators are more reliable than others (Chan et al. 2006; Eigenbrod et al. 2010; van Oudenhoven et al. 2012). Some proxy-based methods use land-cover data alone and borrow benefit-transfer values from other studies (Plummer 2009), while others use indicators based on a combination of causal factors. In general, proxies should be evaluated carefully and selected to minimize assumptions and biases while maximizing transferability (Eigenbrod et al. 2010).

Land cover is the most commonly used data proxy in ES assessments (Eigenbrod et al. 2010; Yapp et al. 2010; Chan et al. 2006). In many assessments, it represents a simple model of ES capacity and, at times, ecological pressure. Its use, however, is based on the assumption that all areas of a given cover type function similarly regardless of location (e.g. latitude), other biophysical features (e.g. soil type), or time (e.g. season). Land cover data range widely in spatial resolution, age, and classification schemes, largely based on the availability of satellite or aerial imagery, the time it takes to process and validate data, and the data's intended purpose. For example, the Cropland Data Layer, provided by the U.S. Department of Agriculture's National Agricultural Statistics Service (NASS), is available with 30-m resolution based on imagery from 2012, whereas the North American Land Change Monitoring System (NALCMS) provides land cover data for the U.S., Canada and Mexico at 250-m spatial resolution based on 2005 imagery. Further, the NALCMS dataset provides greater classification resolution with respect to forest type, while the NASS dataset provides more detail for agricultural systems.

Despite the wide variety of land cover data available, relying on such data alone to infer ecosystem capacity and pressure may ignore many important contributing factors (Fig. 3a). As the ecological processes involved become increasingly complex or the spatial extent of an assessment increases, our ability to accurately represent a service with a single landscape feature (e.g. land cover) decreases. For example, ES capacity may differ greatly between an oak-hickory forest in Virginia and a beech-maple-birch forest in New York, yet both are "deciduous forests". Land cover alone cannot capture the ecological heterogeneity and mechanistic complexity associated with many regulating and cultural services. Service capacity is rarely reducible to land cover alone so land-cover proxies do best when the relationship between landscape feature and service delivery is simple and tight. Along these lines, we expect snapshot assessments of provisioning services like crop, timber, and water supply based on land cover to be relatively accurate, given prior knowledge of expected yield per unit area. However, using single-feature data proxies may bias ES assessments toward tangible products (e.g., timber, crops, and other provisioning services) which could promote over-simplification of ecosystems and exacerbate the disparity among service-classes used in ES assessments.

Scientists struggle to assess, monitor, and predict the sustainability of ES because the ecological and social mechanisms that provide them are complex (Balvanera et al. 2006; Nedkov and Burkhard 2012) and the available methodologies are in flux. To date, most ES assessments have focused on services for which ecological or economic knowledge is strong and data are readily available. However, data gaps are pervasive and many important research questions can be approached only by using data collected for other purposes (Eigenbrod et al. 2010; Villamagna et al. 2014a) or by ignoring certain ES components. Inconsistencies in which components of ES delivery are measured and how they are represented have precluded meaningful comparisons of ES provision across landscapes and have hindered creation of accurate inventories and functional markets for ES. We concur with recent efforts to standardize measures of ES so that assessments become more useful (Layke 2009; de Groot et al. 2010).

7 Case Studies of Methods to Evaluate Capacity for Freshwater ES

To illustrate our methods to evaluate the capacity of freshwater ES, we focus on recent efforts conducted throughout Virginia and North Carolina (USA), with a focus on the Albemarle-Pamlico Basin (APB). The APB stretches across Virginia and North Carolina (30,000 mi^2) and includes the watersheds for the Chowan, Roanoke, Tar-Pamlico, and Neuse rivers. The APB is largely rural, with substantial forestry and agricultural activities interspersed with natural areas. The highest population density within this region is along the Atlantic coast and near Raleigh-Durham, North Carolina. We chose to focus on the APB because the livelihoods of people there are closely tied to ecosystems and natural resources. Fishing, timber harvest, and farming provide significant sources of income and interviews with APB stakeholders suggest strong ties to the environment (Villamagna and Giesecke 2014).

We present water purification services as an example of quantifying and mapping service capacity. Water purification (regulating) services play a critical role in providing clean water and supporting human and ecosystem health. We define water purification as the collective processes that constrain the biological availability of contaminants. Key metrics of the capacity of the resulting services are the amounts of contaminants precluded from entering or removed from water within a given area over a given time. Capacity is dependent on structural elements (physical, biological, and chemical) of the landscape that mediate the exclusion, removal, or conversion of contaminants (Correll 2005; Fennessy and Cronk 1997). Therefore, water quality regulation capacity is a function of geology, soil type, land cover, precipitation, and to the extent that it affects the structural composition of the landscape, land use. For example, in agricultural landscapes, capacity may change with cropping patterns that alter land cover but changes in soil porosity are not likely to change unless the area is mined or severely compacted. Contaminant inputs represent important ecological pressures since they make it more difficult to attain societal goals for water quality (Fig. 3a).

Water quality regulation comprises multiple water purification services that vary in importance depending on the location and extent of the study area. We describe our quantification and mapping approach for each service separately; however they could be integrated into a single water quality regulation service, for example by using relative ranks or standardized scores to reflect their overall contribution to purification within a given watershed.

For the services discussed, all data were publically available online. Analogous data with greater resolution or accuracy may be available for specific localities. Incorporating high-resolution, local data may reduce uncertainty of local analyses but may increase uncertainty for comparative analyses across landscapes. These tradeoffs should be considered when choosing data inputs.

7.1 Riparian Filtration (Surface Water Quality Protection)

Water filtration in riparian areas (Fig. 3a) provides an important service to local and downstream water users. Riparian filtration includes the uptake and denitrification of nitrogen and the physical filtration of sediment particles that may adsorb phosphorous. Mayer et al. (2007) suggests that vegetation type and the width of the riparian buffers are the strongest drivers of filtration effectiveness; therefore we used a fixed-distance approach in which 50-m buffers were drawn adjacent to all surface waters (e.g. lakes and rivers) and the land cover within those buffers was associated with published removal estimates (NASS Cropland data layer [30-m] 2009). We conducted the spatial analysis in ArcMap 10 (ESRI 2010) that includes geoprocessing tools such as *buffer, erase,* and *join*. Once we knew the land cover within the buffers, we assigned each land cover parcel a filtration rating (% removal) and estimated the potential filtration capacity by calculating the area-weighted mean removal within each 12-digit hydrologic unit (Fig. 5). Unlike many watershed approaches that calculate the expected nitrogen loading to surface waters, this approach keeps the ecological pressure (i.e. nitrogen loading) separate from the service capacity within riparian areas.

Flow-path estimates of nutrient removal capacity take into account the spatially-explicit hydrologic route of surface water from nutrient-loading sources (e.g. agricultural fields) to riparian buffers (Baker et al. 2006). This approach is more complicated than the fixed-distance approach and requires greater ArcGIS experience and hydrologic knowledge. Using this approach, patches of vegetation within the riparian zone that are not hydrologically connected to a known nitrogen source are excluded from the calculation of filtration capacity. The flow-path approach enables an evaluation of the effect of buffer width, which is known to affect nitrogen retention. This approach requires elevation (e.g. U.S. Geological Survey's National Elevation Data; Gesch 2007) and land cover data (e.g. National Land Cover Data [NLCD] 2006; Fry et al. 2006), that contaminant sources be identified, either from land cover classes or specific locations, and that buffers be predefined based on land cover or other mappable properties.

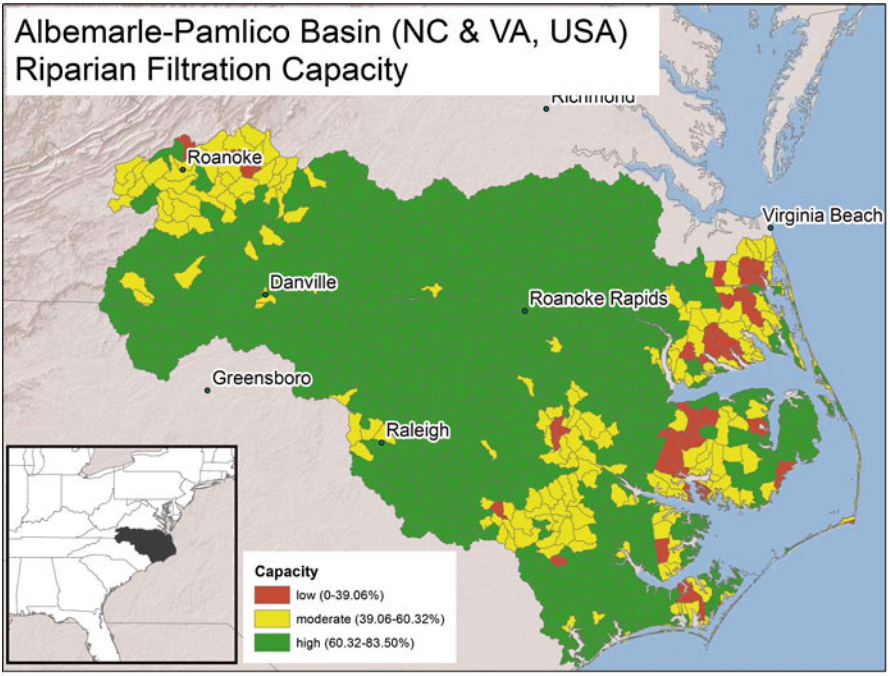

Fig. 5 Map of riparian filtration of nitrogen within the Albemarle-Pamlico Basin in Virginia and North Carolina. Estimated nitrogen (N) removal is based on an average of area-weighted removal values within 12-digit hydrologic units (outlined in grey). N removal values of each land-cover type were assigned according to estimates in Mayer et al. (2007)

The flow-path buffers can be further weighted by estimates of nitrogen removal to incorporate variability in buffer-type effectiveness (Mayer et al. 2007).

7.2 *Sediment Retention (Surface Water Quality Protection)*

We adopted the Revised Universal Soil Loss Equation (RUSLE) as the basis for our estimates of sediment retention capacity (USDA -NRCS 2003). The RUSLE incorporates information on the rainfall-runoff erosivity of soil, soil erodibility, cover practices, land slope, length of the slope, and erosion protection practices to estimate annual sediment loading.

At present, there is no single ArcGIS tool available for estimating soil loss in a spatially-explicit manner. Therefore, we created raster data layers for each of the aforementioned factors and used raster calculator to multiply the factors together. Soil erodibility (K) data are available from the U.S. Soil Survey Geographic Database (SSURGO), provided by the United States Department of Agriculture (USDA) for most of the U.S. Tables of rainfall-induced soil erosion (R) factors are also available through the USDA (Wischmeir and Smith 1978) among other sources. The cover factor (C) was derived by applying the C-factor look-up table associated with the Soil and Water Assessment Tool (SWAT) based on the NLCD for areas within Virginia and North Carolina. The slope-length factor (LS) is calculated using an equation that comprises the land slope for every raster cell and the flow length (i.e. the distance water travels before reaching stream). Slope is easily calculated using the Spatial Analyst extension from elevation data (10-m or 30-m resolution); however flow length is more complex and several geospatial approaches are published. We applied the most straightforward approach in which flow length is calculated from a flow accumulation layer calculated with Spatial Analyst (Lim et al. 2005). The raster cell value of flow length is multiplied by the resolution of the raster cell (30-m or 10-m resolution) to reflect the real flow length necessary for the LS factor equation. Given the complexity of this spatial analysis, we have mapped sediment retention in much smaller areas than those where we mapped riparian filtration, including specific military bases and conservation areas. Erosion protection practices (P) are difficult to determine over large spatial extents since they vary across space and time. While we recommend including the P factor wherever possible, we also believe that the assessment is informative without P, as it reflects the biophysical capacity of an area to retain sediment. With this design, the P factor can serve as a logical variable to include in scenario analysis. To estimate soil retention capacity (rather than the flow of the service), we multiplied the soil, land cover, and topographic factors together ($R \times K \times L \times S \times C$), and standardized the values on a scale of 0–1 to increase interpretability.

7.3 Vertical Nitrogen Retention (Ground Water Quality Protection)

Limiting the lateral movement of water-soluble pollutants is only part of the water purification equation; there is also the threat of pollutants, like nitrates, leaching through the soil into groundwater. Vertical retention of water-soluble pollutants reflects percolation capacity of the soil (i.e. hydrologic group) and precipitation and is inversely related to the probability of surface runoff (Fig. 3a). We adopted the New York Nitrate Leaching Index (Czymmek et al. 2003), which estimates leaching risk based on percolation potential of the soil and seasonality of precipitation, to estimate vertical retention of nitrogen. The percolation equation varies with soil hydrologic group, which characterize soil percolation capacity (Table 2). The State Soil Geographic Database (STATSGO; Soil Survey Staff 2012a) or SSURGO (Soil Survey Staff 2012b) data can be used estimate and map the percolation index using hydrologic group-specific equations (Table 2). We calculated mean winter precipitation (October–March) from the Parameter-elevation Regressions on Independent Slopes Model (PRISM; PRISM Climate Group 2004) monthly precipitation data (norms from 1971 to 2000; multiple stations) using the raster calculator provided by Spatial Analyst and calculated the seasonality index using annual precipitation data

Table 2 Soil properties, as defined by the U.S. Natural Resources Conservation Service SSURGO dataset (accessed 2012): soil hydrologic groups, soil characteristics, surface runoff and infiltration potential, surface and ground water supply potential, and soil hydrologic group-based percolation equations used to calculate the New York Nitrate Leaching Index

Soil hydrologic group[a]	Soil characteristics	Surface runoff & infiltration potential	Surface water supply potential	Ground water supply potential	Percolation Index (PI)
A	Well to excessively drained sands and gravel	Low surface runoff – high infiltration	Low	High	(PA – 10.28)2/(PA + 15.43)
B	Fine to moderately coarse texture	Low to moderate surface runoff – moderate infiltration	Low – moderate	Moderate	(PA – 15.05)2/(PA + 22.57)
C	Moderately fine to fine texture; impedes downward flow of water	Moderate surface runoff – low infiltration	Moderate	Low	(PA – 19.53)2/(PA + 29.29)
D	Clay soils with high swelling potential, clay layer near surface, and shallow soils over (nearly) impervious surfaces	High surface runoff – very low infiltration	High	Low – no potential	(PA – 22.67)2/(PA + 34.00)

[a]USDA SSURGO soil hydrologic group classification; PA: annual precipitation

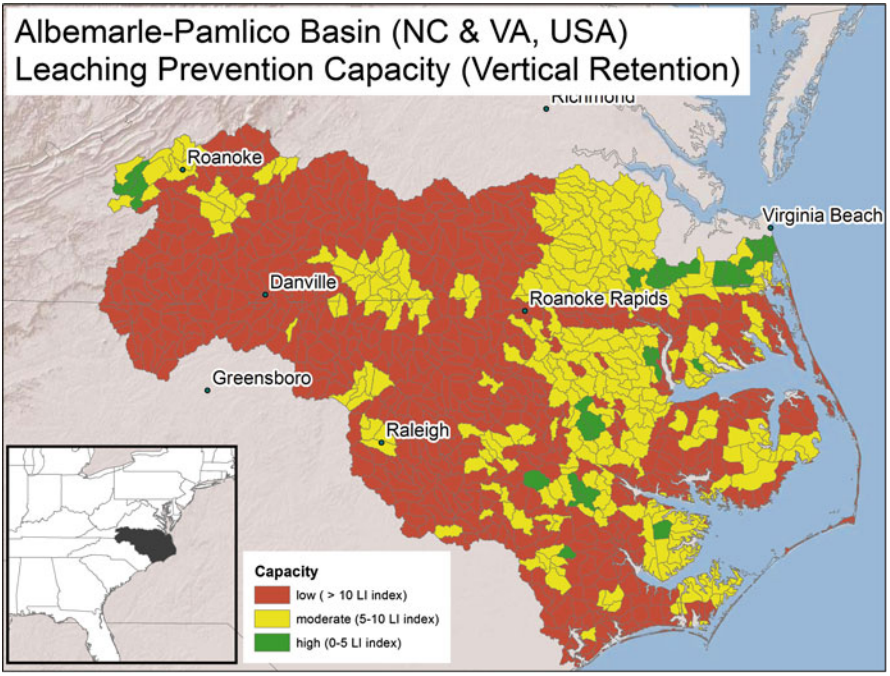

Fig. 6 Map of vertical nitrogen retention within the Albemarle-Pamlico Basin in Virginia and North Carolina. Estimates are based on average values within 12-digit hydrologic units (outlined in grey)

from PRISM climate group and mean winter precipitation data. Since the index represents the potential for nitrate to leach, we used inverse values to classify nitrogen retention capacity, also called groundwater quality protection. This produced a raster surface (30-m resolution) from which we calculated mean retention capacity for each hydrologic unit (Fig. 6).

8 Case Studies of Methods to Evaluate Demand for ES

Although it is largely accepted that ES are inherently linked to the needs and desires of humans, demand for ES has only recently been considered explicitly in non-economic ES assessments (Burkhard et al. 2012; Nedkov and Burkhard 2012). In an economic context, contingent valuation approaches have been used to assign value to ES and these values can be interpreted as measures of demand. However, these values are likely based on the perceived supply of a service, availability of alternatives, and preferences of the community in question, which means they may be highly biased measures of demand. Explicit measures of demand estimate the amount of a service desired, which ideally is measured by the number of people and the amount of desired service per capita. While this information is fairly straightforward to gather for provisioning services where desired amounts are easily quantified, demand for regulating and cultural services must often be based on surrogate measures. Since regulating services produce or maintain desirable environmental conditions, societal demand is best expressed as the amount of regulation needed to meet a desired end condition (e.g. the percentage reduction needed to meet numeric criteria for a pollutant). For cultural services there are several options for quantifying demand indirectly. We present two approaches for assessing demand for wildlife-based recreation (freshwater recreational fishing and bird-watching) that incorporate (1) recreation licenses and (2) public area management priorities.

8.1 Freshwater Recreational Fishing

Demand for freshwater recreational fishing can be quantified as the number of people wanting to fish or the number of days each angler wants to fish. We suggest that fishing license sales are convenient indicators of demand for freshwater recreational fishing, whereas data on how frequently licenses are used by anglers will typically require targeted surveys. Since most states do not limit fishing license sales, a key step in meeting demand for the service is to provide people legal access to the activity. Fishing licenses provide information about the number of anglers and where they live; therefore licenses illustrate areas where access to the service is potentially desired. We demonstrate this approach with an assessment of freshwater recreational fishing demand in Virginia and North Carolina during 2010. We collected tabular license data from the North Carolina Wildlife Resources Commission and the Virginia Department of Game and Inland Fisheries, selected appropriate license categories (all containing freshwater fishing except short-term licenses), and geocoded addresses in a mapping environment (ESRI 2010). Where home addresses were unmappable using the Environmental Systems Research Institute's (ESRI) address locator database, we mapped licenses to the nearest city. We summed and mapped licenses by 12-digit hydrologic unit and county (Villamagna et al. 2014a). Our assessment showed that a disproportionate number of anglers lived in four population centers while all other areas shared a lower, relatively homogenous level of demand (Fig. 7).

8.2 Bird Watching

Bird watching requires no license, so the data available to estimate demand are more limited than for freshwater fishing. In this example, we focus on the demand for bird watching in public use and management areas (PUAs) throughout Virginia and North Carolina, USA. Rather than trying to estimate the number of people who visited PUAs specifically for bird watching, we used the objectives outlined

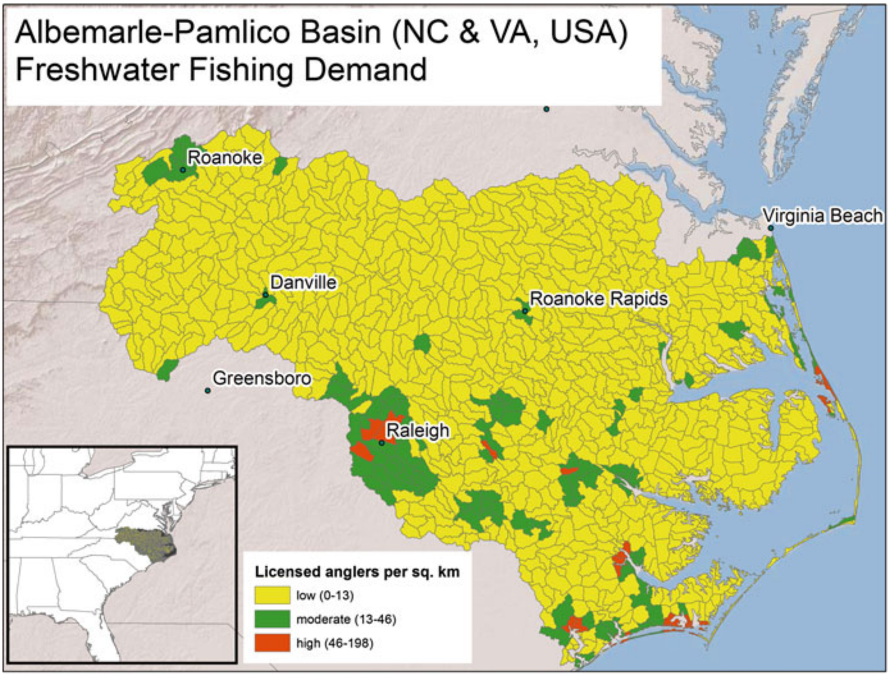

Fig. 7 Map of demand (licenses per square km within each 12-digit hydrologic unit outlined in grey) for freshwater recreational fishing based on 2010 sales of fishing licenses within the Albemarle-Pamlico Basin in Virginia and North Carolina

by the PUA management authority as indicators of conservation intent and ES demand. Given a finite budget, PUAs are forced to prioritize management efforts. We assume that such decisions reflect the demands of users and/or the managing agency. Thus, it follows that managing for bird watching or bird habitat is an indicator of high demand. To assess demand across all PUAs, we conducted an internet search for each PUA to identify and (ordinally) rank management objectives according to a pre-established rubric (see Villamagna et al. 2014b for methodological details). When ranked demand was compared to ranked capacity, we found that capacity was higher in PUAs that did not mention bird watching as a management objective (i.e. low demand) than in PUAs where birding was a higher priority (Welch's ANOVA: $p = 0.0006$, $F = 6.73$, $d.f. = 3$). Only 10.2 % of all PUAs surveyed in North Carolina noted bird watching as a primary management objective, suggesting low general demand for this cultural service. Overall, this methodology allowed us to quickly create a database of rankings for most PUAs, but the same approach would not work well for areas without accessible management plans. We suggest that assessments of smaller areas of interest and/or fewer PUAs be based on more detailed area-specific information using up-to-date management plans or surveys of PUA users.

9 Future Needs and Directions

Frameworks for ES analyses are evolving rapidly (Boyd and Banzhaf 2007; Wallace 2007; Costanza 2008; Daily et al. 2011; de Groot et al. 2010; Nedkov and Burkhard 2012; van Oudenhoven et al. 2012), yet it is not clear which frameworks are most helpful to environmental decision-makers. We suggest that this shortcoming can be rectified by more coherent conceptual models and terminology, more methodological transparency, and more integration of analyses across ES. There is little consensus on terminology and few frameworks address multiple components of ES delivery (Bagstad et al. 2012; Nedkov and Burkhard 2012; Tallis et al. 2012). We summarize the range of terminology used for the four primary components of ES delivery: capacity, demand, ecological pressure, and flow (Fig. 2). There is a clear tendency toward adopting ecological (Beier et al. 2008; Layke 2009; de Groot et al. 2010) and economic terms (Hein et al. 2006; Wallace 2007; Balmford et al. 2011), but few studies integrate the two lexicons (de Groot et al. 2010; Villamagna et al. 2013). To help support our choice of terminology, we explain the relationship of our terms to economic valuations and sustainability discourse (Table 3). Other unsettled issues within the literature include the proper level of focus on mechanics of ES delivery (Bagstad et al. 2012), which ecosystem properties and processes most influence service production (de Groot et al. 2010; van Oudenhoven et al. 2012), and the economic value of services (Grabowski et al. 2012). Despite clear progress in broadening ES research, the lack of cohesion on terminology and frameworks may limit the near-term utility of this work to managers and decision-makers. We encourage ES scientists and conservation practitioners to become comfortable with the variant terminology and develop or adopt analytical frameworks that are flexible and adaptive.

A major benefit of distinguishing among ES capacity, demand, ecological pressure, and flow is that we can more clearly evaluate ecological sustainability and key tradeoffs (McDonald 2009), which is central to making ES assessments useful to stakeholders and decision-makers. Spatially explicit ES budgets, based on demand and capacity (i.e. supply *ala* Burkhard et al. 2012), are helpful for identifying areas where additional inputs (e.g. transportation networks, flood control structures) will be needed to meet demand. Budgets may also reveal areas where development may be sustainable. When demand is similar to flow without decreasing service capacity, the service is being used sustainably. In contrast, ES flows are not sustainable when demand cannot be met by current capacity or when meeting demand causes undesirable declines in other services or in the future provision of the same service.

Geospatial tools like ArcGIS and R have catalyzed ES research but are often used without full disclosure of methods and assumptions. This reduces the potential for others to replicate the methods and interpret the findings. We encourage greater development and sharing of GIS tools created to map ES in diverse landscape. These tools need not be as sophisticated as a stand-alone program (e.g. InVest, Nelson et al. 2009), which can be extremely helpful for many assessments. Practical

Table 3 Summary of relations of ecosystem service (ES) terms to sustainability and economic valuation

Key ecosystem service (ES) delivery terms	Relationship to sustainability	Relationship to economic valuation
Capacity	The maximum potential service to be delivered to beneficiaries. Capacity constrains the service delivery and may be reduced by extensive ecological pressures. Service production (i.e. flow) at maximum capacity need not be sustainable over time. [consider maximum yield versus maximum sustainable yield versus optimal sustainable yield]	Analogous to the standing stock of goods. The amount potentially available for use. Capacity may be greater than the service delivered depending on demand
Demand	When demand exceeds capacity, technological substitutes or supplements may be required to meet demand	Together, demand and capacity estimates help establish the value of a service. Thus, changes in demand given constant capacity may influence the (economic) value of the service and vice versa. Economic valuation techniques include hedonic pricing and willingness to pay (De Groot et al. 2002)
Ecological pressures	Extensive ecological pressures can cause short- and long-term reductions in capacity (i.e. environmental degradation), decreases in services delivered, and ultimately reductions in human well-being. Pressures on regulating services can also increase the amount of ecological work needed to meet societal demand and affect the production of other services. Changes in pressures can flip system from sustainable to unsustainable or vice versa	Ecological pressures may be considered externalities of certain markets. They are rarely accounted for in economic valuation of ES; however, they may dramatically affect the stock, regenerative, or assimilative capacity of an ecosystem
Flow	The supply of services reflects the coupling of capacity and demand for the service. Service benefits are limited by capacity, regardless of demand, but overuse of ES may result in degradation of ES capacity and ultimately decrease ES flow	The quantity of service benefits delivered (i.e. ES flow) can be reflected in economic markets and payments for ecosystem services. The benefits delivered will increase with demand until capacity limits them

Benefit zone	The area where ES benefits are experienced may be local or far-reaching. Use of the ES will continue sustainably if the process of benefit delivery does not negatively impact the stock, regenerative, or assimilative capacity of ecosystems. If the benefit zone does not overlap with the ES production area, decisions made within the benefit zone may not impact provision or delivery of benefits	If the benefit zone overlaps with the service-producing area, then the ES delivery process is closed and decisions made within the benefit zone also affect the capacity to provide the service. If the benefit zone and service-producing area do not overlap, the benefits can be considered positive externalities from the decisions made within the service-producing area since this system is not closed (i.e. decisions in benefit zone do not affect capacity to supply ES)
Beneficiaries	Those who benefit from ES will likely experience greater well-being. The sustainability of ES relies in part on the equitable distribution of benefits to beneficiaries	Beneficiaries are the target audience for economic markets. The demand exerted by potential beneficiaries will influence the price of a service in formal markets
Benefit flow	The distance and direction between service-producing area and benefit zone relate to the overall sustainability of the system where decisions in the service-producing area are made without consideration of the benefit zone, and vice versa. The farther the distance, the more likely that decisions affecting capacity will be made in the absence of input from beneficiaries, resulting in substantial trade-offs	The relationship between service-producing areas and the benefit zone reflects the potential economic market based on supply and demand. The demand reflected by those in the benefit zone will dictate the flow of benefits and, in formal markets, the price of the service
Value	The value of a service, assigned through formal economic markets or through more integrative assessments of human well-being, will vary based on environmental and socioeconomic conditions. The value of the services derived will likely drive land management and conservation decisions that influence the long-term sustainability of service provision	The value of the benefits derived from ES will vary with demand (i.e. consumer preferences) and perceived supply. Value of services can be expressed in various terms related to human well-being; economic valuation is the most common approach

and transparent tools that are developed to be shared across landscapes and analytical problems will be most helpful to standardizing ES analyses and enhancing our ability to conduct scenario analysis. Furthermore, we expect such tools to be adopted by practitioners quicker than stand-alone programs that may need to be vetted by a security board prior to use (e.g. Department of Defense). In this context, we suggest it may be beneficial to develop GIS tools that can be installed and run in the ArcMap environment. To be broadly useful these tools need to be accompanied by conceptual models that illustrate the geoprocessing methods and can be revised as needed.

While it is important to consider the sustainability of a single service based on capacity, demand, and flow relationships, it may be of greater socio-economic importance to consider the sustainability of several interacting services at once. Prolonged periods of excess ecological pressure or overuse may shift ecosystem functions in ways that permanently alter ES capacity and delivery (Scheffer and Carpenter 2003; MA 2005; Rodríguez et al. 2006). To avoid environmental damage and decreases in long-term net human well-being that accompany over-stressed systems, society can choose to (1) invest in natural capital that fosters service production, (2) reduce demand, or (3) invest in technological substitutes (Villamagna et al. 2014a). Management choices can strongly influence capacity of other services (Bennett et al. 2009) and a change in the flow of one service can greatly influence ecological pressure on another (Rodríguez et al. 2006; Barbier 2009). Options 1 and 2 would essentially maintain existing relationships among services; however, option 3 may alter the balance. Some technological solutions targeting a single service (e.g. water treatment plant) fail to restore all potential benefits that would exist in a non-degraded system (e.g. habitat provision). In contrast, other technological responses may create novel ecosystems that enhance multiple services (e.g. a reservoir providing water supply, flood regulation, and recreation). A key role for ES researchers is to develop tools and methods that clearly illustrate to non-experts how service provision changes with each management decision. Given the complex interactions among services, visualizing and understanding ES tradeoffs – both among ES and among stakeholders – that examine the effects of services on one another can help assess landscape-level ES sustainability and contribute to environmental decision-making (Rodríguez et al. 2006; Daily et al. 2009; Raudsepp-Hearne et al. 2010).

9.1 Conclusion

Watershed-based analyses of ES are potentially powerful tools for understanding ecological processes and how spatiotemporal variations in those processes translate into societal benefits. Moreover, distinguishing service capacity from demand, ecological pressure, and flow open the door to broader evaluations of ES sustainability and tradeoffs stemming from management choices. Methods to conduct these analyses are developing rapidly; some of those we describe herein may be obsolete within a decade. Even so, we encourage others to adopt and adapt the methods and data now available so they can provide stakeholders and decision-makers with timely scientific knowledge of ES. We look forward to the development of more sophisticated, widely applicable methods that can provide even clearer insights into how and where ES are produced and used and how they influence each other.

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References

Arthington AH, Naiman RJ, McClain ME, Nilsson C (2010) Preserving the biodiversity and ecological services of rivers: new challenges and research opportunities. Freshw Biol 55:1–16

Bagstad KJ, Johnson GW, Voigt B, Villa F (2012) Spatial dynamics of ecosystem service flows: a comprehensive approach to quantifying actual services. Ecosyst Serv 4:117–125

Baker ME, Weller DE, Jordan TE (2006) Improved methods for quantifying potential nutrient interception by riparian buffers. Landsc Ecol 21:1327–1345

Balmford A, Fisher B, Green RE, Naidoo R, Strassburg B, Turner RK, Rodrigues ASL (2011) Bringing ecosystem services into the real world: an operational framework for assessing the economic consequences of losing wild nature. Environ Resour Econ 48:161–175

Balvanera P, Pfisterer AB, Buchmann N, He JS, Nakashizuka T, Raffaelli D, Schmid B (2006) Quantifying the evidence for biodiversity effects on ecosystem functioning and services. Ecol Lett 9:1146–1156

Barbier EB (2009) Ecosystem service trade-offs. In: Mcleod K, Leslie H (eds) Ecosystem-based management for the oceans. Island Press, New York

Beier CM, Patterson TM, Chapin FS (2008) Ecosystem services and emergent vulnerability in managed ecosystems, a geospatial decision-support tool. Ecosystems 1:923–938

Bennett EM, Peterson GD, Gordon LJ (2009) Understanding relationships among multiple ecosystem services. Ecol Lett 12:1394–1404

Boyd J, Banzhaf S (2007) What are ecosystem services? The need for standardized environmental accounting units. Ecol Econ 63:616–626

Burkhard B, Kroll F, Nedkov S, Muller F (2012) Mapping ecosystem service supply, demand and budgets. Ecol Indic 21:17–29

Cairns J (1997) Protecting the delivery of ecosystem services. Ecosyst Health 3:185–194

Carpenter SR, Mooney HA, Agard J, Capistrano D, DeFries RS, Diaz S, Dietz T, Duraiappah AK, Oteng-Yeboah A, Pereira HM, Perrings C, Reid WV, Sarukhan J, Scholes RJ, Whyte A (2009) Science for managing ecosystem services: beyond the Millennium Ecosystem Assessment. Proc Natl Acad Sci U S A 106:1305–1312

Chan K, Shaw MR, Cameron DR, Underwood EC, Daily GC (2006) Conservation planning for ecosystem services. PLoS Biol 4:2138–2152

Chan K, Satterfield T, Goldstein J (2012) Rethinking ecosystem services to better address and navigate cultural values. Ecol Econ 74:8–18

Correll DL (2005) Principles of planning and establishment of buffer zones. Ecol Eng 24:433–439

Costanza R (2008) Ecosystem services: multiple classification systems are needed. Biol Conserv 141:350–352
Czymmek K, Ketterings QM, van Es H DeGloria S (2003) The New York nitrate leaching index. CSS Extension Publication, 34 pp. Available online March 15 2015 at http://nmsp.cals.cornell.edu/publications/extension/nleachingindex.pdf
Daily GC, Matson PA (2008) Ecosystem services: from theory to implementation. Proc Natl Acad Sci U S A 105:9455–9456
Daily GC, Polasky S, Goldstein J, Kareiva PM, Mooney HA, Pejchar L, Ricketts TH, Salzman J, Shallenberger R (2009) Ecosystem services in decision making: time to deliver. Front Ecol Environ 7:21–28
Daily G, Kareiva P, Polasky S, Ricketts T, Tallis H (2011) Mainstreaming natural capital into decisions. In: Kareiva P, Tallis H, Ricketts T, Daily G, Polasky S (eds) Natural capital: theory and practice of mapping ecosystem services. University Press Oxford, Oxford
Dale VH, Polasky S (2007) Measures of the effects of agricultural practices on ecosystem services. Ecol Econ 64:286–296
De Groot RS, Wilson MA, Boumans RM (2002) A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecol Econ 41:393–408
de Groot RS, Alkemade R, Braat L, Hein L, Willemen L (2010) Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecol Complex 7:260–272
Egoh B, Reyers B, Rouget M, Richardson DM, Le Maitre DC, van Jaarsveld AS (2008) Mapping ecosystem services for planning and management. Agr Ecosyst Environ 127:135–140
Eigenbrod F, Armsworth PR, Anderson BJ, Heinemeyer A, Gillings S, Roy DB, Thomas CD, Gaston KJ (2010) The impact of proxy-based methods on mapping the distribution of ecosystem services. J Appl Ecol 47:377–385
ESRI (Environmental Systems Research Institute) (2010) ArcGIS Desktop (ArcInfo) Software v10.0 ESRI. Redlands, California
Fennessy MS, Cronk JK (1997) The effectiveness and restoration potential of riparian ecotones for the management of nonpoint source pollution, particularly nitrate. Crit Rev Environ Sci Technol 27:285–317
Fisher B, Turner K (2008) Ecosystem services, classification for valuation. Biol Conserv 141:1167–1169
Fisher B, Turner K, Zylstra M, Brouwer R, de Groot R, Farber S, Ferraro P, Green R, Hadley D, Harlow J, Jefferiss P, Kirkby C, Morling P, Mowatt S, Naidoo R, Paavola J, Strassburg B, Yu D, Balmford A (2008) Ecosystem services and economic theory: integration for policy-relevant research. Ecol Appl 18:2050–2067
Fry J, Xian G, Jin S, Dewitz J, Homer C, Yang L, Barnes C, Herold N, Wickham J (2011) Completion of the 2006 national land cover database for the conterminous United States. Photogramm Eng Remote Sens 77:858–864
Gesch DB (2007) The National Elevation Dataset. In: Maune D (ed) Digital elevation model technologies and applications: the DEM users manual, 2nd edn. American Society for Photogrammetry and Remote Sensing, Bethesda
Grabowski JH, Brumbaugh RD, Conrad RF, Keeler AG, Opaluch JJ, Peterson CH, Piehler MF, Powers SP, Smyth AR (2012) Economic valuation of ecosystem services provided by oyster reefs. Bioscience 62:900–909
Hein L, van Koppen K, de Groot RS, van Ierland EC (2006) Spatial scales, stakeholders and the valuation of ecosystem services. Ecol Econ 57:209–228
Johnston RJ, Russell M (2011) An operational structure for clarity in ecosystem service values. Ecol Econ 70:2243–2249
Jose S (2009) Agroforestry for ecosystem services and environmental benefits: an overview. Agrofor Syst 76:1–10
Keeler BL, Polaskya S, Braumana KA, Johnsond KA, Finlayc JC, O'Neille A, Kovacsf K, Dalzellg B (2012) Linking water quality and well-being for improved assessment and valuation of ecosystem services. Proc Natl Acad Sci 109:18619–18624

Layke C (2009) Measuring nature's benefits: a preliminary roadmap for improving ecosystem service indicators. World Resources Institute, Working Paper, 36 pp

Lim KJ, Sagong M, Engel BA, Tang Z, Choi J, Kim K (2005) GIS-based sediment assessment tool. CATENA 64:61–80

MA [Millennium Ecosystem Assessment] (2005) Ecosystems and human well-being. Synthesis Island Press, Washington, DC

Martin-Lopez B, Iniesta-Arandia I, Garcia-Llorente M, Palomo I, Casado-Arzuaga I, Del Amo DG, Gomez-Baggethun E, Oteros-Rozas E, Palacios-Agundez I, Willaarts B, Gonzalez JA, Santos-Martin F, Onaindia M, Lopez-Santiago C, Montes C (2012) Uncovering ecosystem service bundles through social preferences. PLos One 7(6):e38970

Martinez ML, Perez-Maqueo O, Vazquez G, Castillo-Campos G, Garcia-Franco J, Mehltreter K, Equihua M, Landgrave R (2009) Effects of land use change on biodiversity and ecosystem services in tropical montane cloud forests of Mexico. For Ecol Manage 258:1856–1863

Mayer PM, Reynolds SK, McCutchen MD, Canfield TJ (2007) Meta-analysis of nitrogen removal in riparian buffers. J Environ Qual 36:1172–1180

McDonald RI (2009) Ecosystem service demand and supply along the urban-to-rural gradient. J Conserv Plann 5:1–14

McShane TO, Hirsch PD, Trung TC, Songorwa AN, Kinzig A, Monteferri B, Mutekanga D, Thang HV, Dammert JL, Pulgar-Vidal M, Welch-Devine M, Brosius JP, Coppolillo P, O'Connor S (2011) Hard choices: making trade-offs between biodiversity conservation and human well-being. Biol Conserv 144:966–972

Nedkov S, Burkhard B (2012) Flood regulating ecosystem services-mapping supply and demand, in the Etropole municipality, Bulgaria. Ecol Indic 21:67–79

Nelson E, Mendoza G, Regetz J, Polasky S, Tallis H, Cameron DR, Chan KMA, Daily GC, Goldstein J, Kareiva PM, Lonsdorf E, Naidoo R, Ricketts TH, Shaw MR (2009) Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Front Ecol Environ 7:4–11

Plummer ML (2009) Assessing benefit transfer for the valuation of ecosystem services. Front Ecol Environ 7:38–45

PRISM Group (Parameter-elevation Regressions on Independent Slopes Model) (2004) Mean annual and monthly precipitation 1971–2000. Oregon State University. http://www.prismclimate.org. Accessed 10 Feb 2012

Raudsepp-Hearne C, Peterson GD, Bennett EM (2010) Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. Proc Natl Acad Sci U S A 107:5242–5247

Rodriguez JP, Beard TD, Bennett EM, Cumming GS, Cork SJ, Agard J, Dobson AP, Peterson GD (2006) Trade-offs across space, time, and ecosystem services. Ecol Soc 11(1):28. [online] URL: http://www.ecologyandsociety.org/vol11/iss1/art28/

Scheffer M, Carpenter S (2003) Catastrophic regime shifts in ecosystems: linking theory to observation. Trends Ecol Evol 18:648–656

Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture (2012a) Soil Survey Geographic (SSURGO). Database available online at http://websoilsurvey.sc.egov.usda.gov. Accessed 15 Mar 2015

Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture (2012b) US General Soil Map (STATSGO2). Database available online at http://websoilsurvey.sc.egov.usda.gov. Accessed 15 Mar 2015

Tallis H, Mooney H, Andelman S, Balvanera P, Cramer W, Karp D, Polasky S, Reyers B, Ricketts T, Running S, Thonicke K, Tietjen B, Walz A (2012) A global system for monitoring ecosystem service change. Bioscience 62:977–986

USDA -NRCS (Natural Resources Conservation Service, *formerly* Soil Conservation Service) (1972) National engineering handbook hydrology, Section 4 Chapters 4–10. USDA, Washington, DC

USDA -NRCS (2003) RUSLE2 (Revised Universal Soil Loss Equation: Version 2) Available online at 1 March 2010. http://fargo.nserl.purdue.edu/rusle2_dataweb/RUSLE2_Index.htm. Accessed 1 Mar 2010 van Oudenhoven A, Petz K, Alkemade R, Hein L, de Groot R (2012) Framework for systematic indicator selection to assess effects of land management on ecosystem services. Ecol Indic 21:110–122

Villamagna AM, Angermeier PL, Bennett EM (2013) Capacity, demand, pressure, and flow: a conceptual framework for analyzing ecosystem service provision and delivery. Ecol Complex 15:114–121

Villamagna AM, Giesecke C (2014) Adapting a human well-being framework for ecosystem service assessment. Ecol Soc 19(1):11

Villamagna AM, Mogollon B, Angermeier PL (2014a) A multi-indicator framework for mapping cultural services: the case of freshwater recreational fishing. Ecol Indic 45:255–265

Villamagna AM, Angermeier PL, Niazi N (2014b) Evaluating opportunities to enhance ecosystem services in public use areas. Ecosyst Serv 7:167–176

Wallace K (2007) Classification of ecosystem services: problems and solutions. Biol Conserv 139:235–246

Wischmeier WH, Smith DD (1978) Predicting rainfall erosion losses: a guide to conservation planning, USDA agriculture handbook. US Government Printing Office, Washington, DC

Yapp G, Walker J, Thackway R (2010) Linking vegetation type and condition to ecosystem goods and services. Ecol Complex 7:292–301

Assessing the Impact of Land-Use Changes on Providing Hydrological Ecosystem Functions (ESF) and Services (ESS) – A Case-Study Experience Based Conceptual Framework

Christine Fürst and Wolfgang-Albert Flügel

Abstract In this chapter, requirements for and lessons learnt from assessing the impact of land use and land cover change (LULCC) on the provision of hydrological ecosystem functions (ESF) and services (ESS) are demonstrated based on selected case studies. First, potentials, limits and transferability of a detailed land classification scheme developed for Germany are explored. Second, an approach how to make use of landscape metrics to correct the assessment of ESF and ESS provision in LULCC impact assessment is presented to better account for land-use pattern heterogeneity. Third, the potential of Hydrological Response Units (HRU) to bridge scale-related discrepancies between modeling, assessment and decision units is discused. Finally, a conceptual framework approach is suggested that builds on the HRU concept and merges the latter with a cellular automaton based LULCC impact assessment framework.

Keywords Ecosystem functions (ESF) • Ecosystem services (ESS) • Hydrological Response Units (HRU) • Integrated land and water resource management (ILWRM) • Distributed river basin modeling • Eco-hydrology

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1 Introduction

1.1 Objectives

This chapter presents a conceptual framework for better connecting different spatial scales in ESF and ESS assessment (definitions, see Sect. 1.2) and decision making. This framework is based on experiences taken from selected case studies. We start with an example that demonstrates assessment limits that arise from use land classification standards and how these could be overcome. A second case study demonstrates how landscape composition and configuration can be taken into account to improve the quality in ESF and ESS assessment. Subsequently, we discuss strengths and weaknesses of a hierarchical approach that aims at combining both, land-use and land-use pattern contribution to ESF and ESS. The transferability of this approach in the context of hydrological ESF and ESS is explored, making use of the concept of hydrological response units (HRU, Flügel 1996a, b).

1.2 Definitions

Ecosystem Functions (ESF) Based on the cascade approach of Haynes-Young and Potschin (2010) we understand by ecosystem functions the expression of how biophysical structures or processes impact the temporal and spatial disposability of natural resources that can be requested or consumed as services. Consequently, **hydrological ecosystem functions** are those functions, which drive the temporal and spatial availability of water resources (based on: de Groot et al. 2002).

Ecosystem Services (ESS) ecosystem services are based on a concretely assessable functional processes-pattern-structure chain and bring up consumable benefit to human-well-being that can, but not necessarily must be expressed in monetary terms (Haynes-Young and Potschin 2010). Consequently, **hydrological ecosystem services** are related to consumable water provision (e.g. drinking water), supporting water cycles (plant growth, waste water), mitigated risks (regulating services such as water erosion and flood control), and culturally-economically important water use (fishing, shipping, etc.) (Brauman et al. 2007).

Region is understood as a historically developed, culturally and environmentally defined socio-ecological land system (Laszlo and Krippner 1998).

1.3 Challenges for Assessing the Impact of Land-Use Changes on Hydrological ESF and ESS

Land-use and land-cover changes (LULCC) are acknowledged to be major processes in driving ecological processes and the related natural potential to provide societally requested services (Metzger et al. 2006). This is documented best for essential natural resources to which the provision of drinking water, cleaning of waste water, control of water erosion and flooding and using of water resources for nutrition and transport count among the most important ones (DeFries and Eshleman 2004). Though many studies have been performed to describe and model the relations between LULCC and hydrological functions and services, understanding and predicting the impact of changes at different scales is still not sufficiently solved to serve planning and management at the regional to local scale where most LULCC decisions are made (see e.g. Koschke et al. 2014).

A challenge consists in refining the scale where interrelations between hydrological functions and services and LULCC are modeled: the assessment of LULCC focuses mainly on land-cover classes and ignores the variability of land-use and management alternatives (Van Oudenhoven et al. 2012; Verburg et al. 2009; Dale and Polasky 2007). Examples can be found in agricultural and forest management systems. In agriculture, soil management strategies (conventional tillage, conservation-till, no-till) impact considerably the provision of regulating and supporting services (e.g. van Capelle et al. 2012; Lorenz et al. 2013; Koschke et al. 2013, 2014). Same applies when considering diversified crop rotations, cash crops, mixed cropping and intercropping (Zhang et al. 2007; Lorenz et al. 2013). Integrating management alternatives in modeling and impact assessment has also a higher potential to be accepted and implemented, and therefore contribute to improved ESS provision compared to extensification of agricultural sites or afforestation (Power 2010). In forestry, sustainable forest management using natural processes and working with mixed and multi-layered forests is sufficient for enhancing ESF and ESS (Fürst et al. 2011, 2012). Conversion from coniferous to deciduous forests that was often proposed as best practice (e.g. Spieker et al. 2004) failed to be broadly applied in non-governmental forests mainly due to economic reasons (e.g. Knoke et al. 2005).

Second, the provision of ESS as described in the Millennium Ecosystem Assessment (MEA 2005) is often restricted to the scale of ecosystems under study. It does not account in sufficient detail for parameters such as the spatial heterogeneity of ecosystems or of processes at regional scale that trigger the dynamics of particular ESF and ESS (Frank et al. 2012; Fürst et al. 2012). Results on the important role of such interactions are already available in biodiversity research, and regarding some supporting and regulating services (pollination and pest control) (e.g. Tscharntke et al. 2005), but are not available for numerous other ESS. An example for the impact of landscape heterogeneity on ESF and ESS provision are European cultural landscapes that developed over centuries with a highly heterogeneous structure of land-uses. Nowadays, this traditional land-use diversity is endangered to be lost due to economic development and migration processes (e.g. Rounsevell et al. 2012; Eigenbrod et al. 2011). Such structural changes impact greatly hydrological processes such as runoff generation, groundwater recharge, water erosion, sediment transport or nitrate leaching through the unsaturated zone into the groundwater aquifer (Fink et al. 2007; Flügel 2011c). Increasing homogenization of land-use accompanied by growing average sizes of management units amplifies such processes. In response, this generates negative impacts on ESF and ESS, even, if cross compliance regulations and good management practices are applied (Poggio et al. 2012).

Third, a scale conflict arises when assessing ESF and ESS based on stakeholder needs and perceptions: most ESF and ESS can only be analyzed in regional (socio-ecological system) context and encompass the catchment, watershed or river basin context that is used in hydrological modeling (Trabucchi et al. 2012). This applies, for instance, for provisioning services (wood and fiber, food and fodder), supporting services (primary production, soil formation) and cultural services (aesthetics, recreation), and only few regulating (flood control, water purification) or some provisioning (fresh water) services.

To better combine hydrological ESF and ESS with the concept of integrated land and water resource management (ILWRM), multi-scale regionalization concepts are required (Flügel et al. 2011a, b) that connect disciplinary modeling entities with those at which management, administrative or political decisions are made. Here, function and processes-based entities for nested modeling would be favorable as they are described, for instance, in hydrological modeling by the concepts of hydrogeomorphic units (HGMU, Brinson 2011; Maltby et al. 1994), Hydrological Response Units (HRU; Flügel 1996a, b), still applied in SWAT (Arnold et al. 1998) or most recently for quantitative ecosystem services modeling (Logsdon and Chaubey 2012). Referring to our intention to embed hydrological ESF and ESS better in a regional context, especially the concept of HRU provides a well-tested spatial reference. HRU were successfully applied in distributed river basin models such as JAMS/J2000 (Krause and Flügel 2005; Krause et al. 2006; Fink et al. 2007; Nepal et al. 2012) and support the integration of micro-scale (land-use and management) and meso-scale aspects (catchment, region, administrative districts).

Based on this, the discussion paper will raise the following questions for suggesting advances in modeling and assessment:

(a) How to better account for land-use and management practices when simulating and evaluating the impact of LULCC on ESF and ESS?
(b) How can land-use pattern heterogeneity be integrated more efficiently when assessing ESF and ESS?
(c) How to overcome scale differences in integrated land-use and ILWRM in ESF and ESS assessment?

2 Case Studies in ESF and ESS Assessment and Lessons Learnt

2.1 Case Study 1 – Land-Use Classification and Its Limits for ESF and ESS Assessment

The information quality in assessing regional potentials to maintain ESF and provide ESS is closely related to the modality how land-use or land-cover (LULC) are classified and how land-management is considered when applying LULC for ESS assessment (Dale and Polasky 2007). Land-cover classification as applied, for instance, in Corine Land Cover (CLC) 2006 (www.eea.europa.eu/data-and-maps/data/), is a generic and transferable system, but provides only crude details on spatio-temporal dynamics in land-use and land-management. ESF and ESS assessment could benefit greatly from replacing land-cover by a more functional classification concept (Erb 2012; Verburg et al. 2009) that better accounts for how land-use and land-management trigger eco-hydrological process and the resulting dynamics in ESF and ESS. Examples for more functional classification schemes in river basin modeling were, for instance, applied for assessing N and P loads in water systems (Bossa et al. 2012), for changes in hydrological regimes of catchments (Troy et al. 2007) or for flood regulation (Nedkov and Burkhard 2012). Furthermore, land systems cannot only be understood as sum of interacting ecosystems, but as socio-ecological systems that include a strong cultural component (Lambin and Meyfroidt 2010). In result, a "functional" classification has to consider the purpose of the assessment for which the classification is developed (Koschke et al. 2012).

In a case study "Upper Elbe Valley – Eastern Ore Mts. (UEEO)", a detailed forest and agricultural land-use classification was developed for assessing the impact of LULCC on ESF and ESS under climate change and opportunities for land based mitigation of undesirable trade-offs (Fürst et al. 2011). In cooperation with Euromap GmbH (now renamed into GAF AG), remote sensing and terrestrial data were combined to get the highest possible spatial and thematic resolution for land-use mapping (Euromap Land-cover Classification, EMLC). Problems were (a) the accessibility and spatial matching of the terrestrial data and (b) their association with specific land-management concepts.

2.1.1 Forest Land-Use Classification

For forest land-use classification, the accessibility of terrestrial data is dependent on the type of land ownership. Data from a terrestrial forest inventory were used for governmental forests; for non-governmental forests, biotope and land-use mapping (www.umwelt.sachsen.de/umwelt/natur/18615.htm) was the only accessible data source (Witt et al. 2013).

Considering the spatial matching of these data sets, forest inventory is done in the model region at the scale of forest management planning units (stands) as smallest ESF and ESS assessment entity (Anonymous 2005) with a maximum spatial resolution of $100 \times 100\ m^2$. In contrast, biotope and land-use mapping are available for all ecosystem types in a resolution of $10 \times 10\ m^2$. Both data sets were combined with other terrestrial data, namely forest site classification (Kopp and Schwanecke 1994), soil classification (Sponagel et al. 2005), and the digital elevation model (DEM, 1:25,000). Subsequently, contextual information on site quality dependent silvicultural concepts (Eisenhauer and Sonnemann 2009) was added to ensure compatibility with the Federal State level silvicultural regime (Witt et al. 2013; Fürst et al. 2011, 2012). The latter foresees a reorganization of current stand types to future forest classes with higher tree species diversity.

2.1.2 Agricultural Land-Use Classification

In agriculture, Federal State level statistical data on average percentages of different crops (e.g. Anonymous 2010) were available, but without spatially explicit reference. Furthermore, management information was available at level of so-called "field-blocks", which are the smallest spatial entity for reporting in the context of the European Agricultural Fund for Rural Development (EAFRD) program. These are delineated based on topographic, edaphic and infrastructural parameters and do not contain land tenure information (see e.g. Heinrich et al. 2009).

To harmonize the temporal dynamics of ESF and ESS in agriculture and forestry, data were aggregated to crop rotations to account for intra and inter-annual management effects (Lorenz et al. 2013). These crop rotations provide thematic reference to practices in arable and mixed farming including conventional and organic farming. Statistical data on cultivated crops from 2005 to 2010 were analyzed to identify regionally typical pre- and post-crops to key crops. For spatial transfer, the mesoscale agricultural soil mapping (1:25,000) and field-block data were used. Additionally, conventional, conservation till, and no-till farming were added as management attributes.

2.1.3 Overall Classification Result

In result, 85 land-use classes were differentiated (Fig. 1). They include 22 land-use classes derived directly from remote sensing such as urban fabric, water bodies, or urban green space, and 32 forest land-use classes that were added and linked for scenario simulation with 22 future classes (not displayed in Fig. 1; Fürst et al. 2012; Witt et al. 2013). For agriculture, 31crop rotations were added that represent most common management practices for diluvial sites ("D"), loess sites ("L"), and deeply weathered bedrock sites ("V") (Lorenz et al. 2013). The adapted classification had a maximum spatial resolution of 25×25 m^2 that supports LULCC down to the management planning unit level (micro-scale).

2.1.4 Applicability of the Approach for ESF and ESS Assessment

When using the UEEO data set for ESF and ESS assessment, it became obvious that models or monitoring data accounting for this high level of detail in LULC were not or only partially available.

Considering the forest land-use classes, it turned out that most forest models focus on few tree species and the behavior of pure and single layered stands (e.g. Pabst et al. 2008). Information on mixed and multi-layered stands could neither be obtained from forest yield tables (e.g. Schober 1995).

In agriculture, crop rotations are applied in bio-physical process models or economic models on farm level to derive different environmental impacts (e.g. Janssen and van Ittersum 2007; van Ittersum et al. 2008), while empirical data are rarely

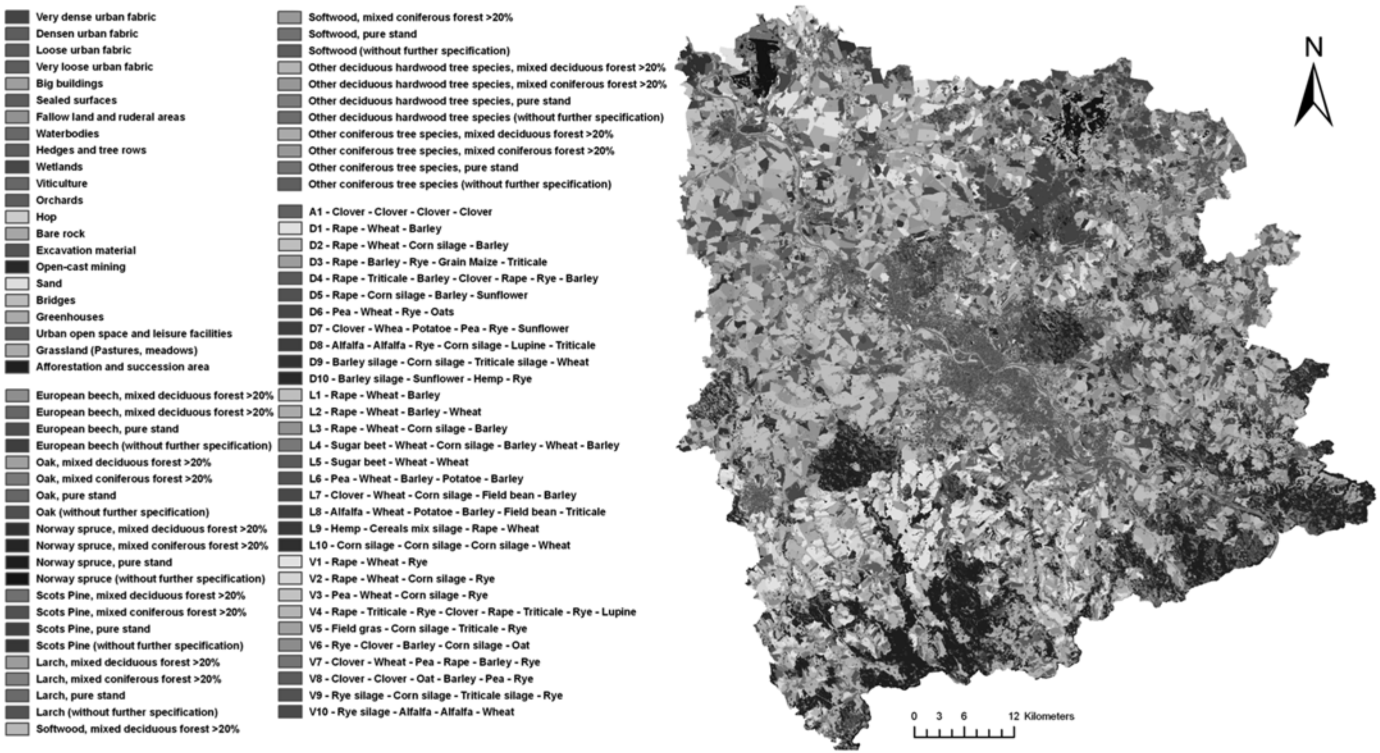

Fig. 1 Euromap Land Cover Classification (EMLC) result for the model region Upper Elbe Valley – Eastern Ore Mts (UEEO)

available (Schönhart et al. 2011a). The implementation of crop rotation-based modeling is often restricted to individual case studies, farm surveys or extracting expert knowledge (Belcher et al. 2004; van Ittersum et al. 2008; Rode et al. 2009). Crop rotations applied at regional scale remain unknown. Spatial allocation of crop rotations or cropping systems plays a fundamental role, but is also a source of uncertainty in deriving environmental impacts on regional scale (see also Bell and Irwin 2002).

For many of the other land-use classes, we experienced that ESF/ESS models are either missing, or data for model calibration and applicable model parameters are lacking. Expert knowledge and benefit transfer had to be used to close these gaps (Koschke et al. 2012). Critically spoken, the attempt to achieve an improved understanding of land-use impact ESF and ESS by increasing the degree of detail in land-use classification did not necessarily provide a better assessment basis.

The problem of data availability gains in importance if classification approaches should be transferred to other regions. Terrestrial information for training the interpretation of remote sensing (RS) data might not be accessible and only few RS techniques provide data that are spatially and temporally detailed enough to delineate functional entities such as crop rotations (e.g. Bach et al. 2006; Colditz et al. 2011; Gulinck and Wagendorp 2002).

A lesson learnt was that land-use classification that supports ESF and ESS assessment should go more in detail than land-cover classification can do, but must respect limits in detailedness given by the availability of modeling, monitoring, and remote sensing data. A suitable approach involves a hierarchic concept that bases on standardized RS information and should be kept compatible to existing land-cover classification sets such as CLC 2006. It should also provide interfaces to differentiate spatially or thematically the land-use information depending on the objective of the assessment and data accessibility (Fürst et al. 2012). Regional experts should be involved in participatory mapping campaigns, contributing their knowledge about typical land-use practices and available information on their impact on ESF and ESS (e.g. Hought et al. 2012; Kristjanson et al. 2005; Lebel and Daniel 2009). This demands for open technological solutions that allow for easy adjustment and modification of already interpreted land-use information.

2.2 Case Study 2 – Landscape Metrics for Improving ESF and ESS Assessment

ESS and ESF assessment concepts still do not intensively account for landscape features such as composition and spatial constellation of different land uses, though interest in this aspect is increasing (Bartel 2000; Frank et al. 2012; Hou and Walz 2013; Lautenbach et al. 2011; Syrbe and Walz 2012). Current research addresses issues such as species diversity or habitat connectivity (e.g. Dover and Settele 2009; Lomba et al. 2011; Verdú et al. 2011), aesthetical landscape value (e.g. Uuemaa et al. 2013), the assessment of ecological sustainability (Renetzeder et al. 2010), or on optimal structures to enhance the provision of one or several ESF and ESS (Laterra et al. 2012).

Frank et al. (2012) developed an approach based on the EMLC in the UEEO case study for a standardized set of landscape metrics to assess criteria such as landscape fragmentation, habitat connectivity and landscape diversity and their implication for basic ecological processes and the perception of landscape aesthetics. Objective was to derive recommendations for regional planning on how to improve the landscape structure by afforestation or alternatively by the establishment of short rotation coppices (Fürst et al. 2012, 2013).

To quantify landscape fragmentation, Frank et al. (2012) implemented the metrics core area index (von Haaren and Reich 2006) and effective mesh size (Jaeger, et al. 2008). For assessing habitat connectivity they used cost distance analysis (Zebisch et al. 2004). The latter was modified to include infrastructural elements (roads, highways, railways) which impact the maximum distance when moving from one potential habitat to the other. Considering landscape diversity, Frank et al. (2012) applied the shape index (Baessler and Klotz 2006; Renetzeder et al. 2010), the Shannon-Wiener diversity index (Yeh and Huang 2009; Kim and Pauleit 2007), and the patch density per km^2 (Hein et al. 2004).

A transfer of this approach to hydrological ESF and ESS could be possible for runoff generation, water erosion and sediment transport. In case of water erosion, small scale structural aspects found already entrance in water erosion risk modeling (Volk et al. 2010). Lowicki (2012) proposes a landscape metrics based approach for runoff and water pollution regulation in agricultural areas. Control of water quality (Amiri and Nakane 2009) and sediment delivery ratio (Vigiak et al. 2012) are already based on landscape metrics.

Transferability problems are related to the land-use classification and scale. Uuemaa et al. (2005) demonstrate the scale and spatial resolution dependency for assessing water quality. When testing our EMLC set (see Sect. 2.1) in comparison to CLC 2006, the same phenomenon was observed. Another problem is the consideration of the spatio-temporal variability of the land-use pattern in agricultural areas. Winter or summer aspect in temperate zones or rain period and dry season in the subtropics could lead to completely different results for different seasons. Here, Zhou et al. (2012) recommend the combination of a cellular automaton with landscape metrics analysis to better account for land-use change trajectories in regions threatened by salinization. Also, Seppelt et al. (2009) underline the necessity of combining landscape ecological approaches and process-based models for upscaling, requiring the identification of suitable modeling entities in a nested approach.

2.3 Scaling Approach – Using HRU in ESF and ESS Assessment

The challenge of scale in hydrological modeling has been addressed by Flügel (1996a, b) who proposed the concept of Hydrological Response Units (HRU) for Integrated Environment System Analysis (IESA). The concept is generating synergy by combining physiographic landscape features and land-use classification into a knowledge-based analysis of the dynamics in each HRU. Per se, the HRU

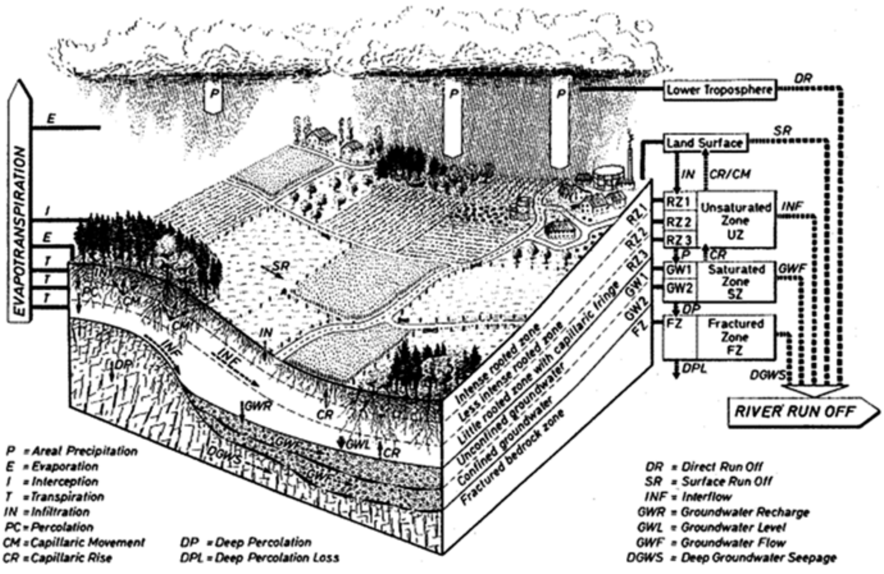

Fig. 2 HRU – schematic overview on their delineation

concept is not scale related and the size of each unit depends on the spatial and thematic resolution applied in the IEAS. HRU are distributed model entities for hydrological water balance modeling in multi-scale landscape drainage systems (Flügel 1996a, b).

As shown in Fig. 2, HRU are defined by an individual setup of land-use and associated topo-pedo-geological features. They control the transformation of precipitation input into evapotranspiration output, soil moisture storage, groundwater recharge, and surface and subsurface runoff components ultimately generating the river runoff response. Consequently, HRU have a priority ranked surface and subsurface water resources regeneration dynamics that relates to respective ESF and ESS.

HRU are delineated based on process understanding obtained from the IESA that is transferred into process-based logical selection criteria by means of GIS analysis including land-use and different landscape feature layers (Fig. 3). HRU as process-based landscape differentiate the landscape drainage system in individual polygons. Applying the digital elevation model (DEM), a topological model is generated that networks all HRU by defining the gradient driven topology between neighboring HRU for water, nutrient or sediment transport routing within the drainage system (Wolf et al. 2009a, b).

Hydrological process dynamics and related ESF in each HRU are quantified by means of a distributed rainfall-runoff model JAMS/J2000 that is calibrated and validated by time series (Kralisch et al. 2007; Krause et al. 2006, Fig. 4). The model represents all processes shown in Fig. 2 by physically based mathematical equations and empirical parameterization for each HRU. JAMS/J2000 quantifies the distributed

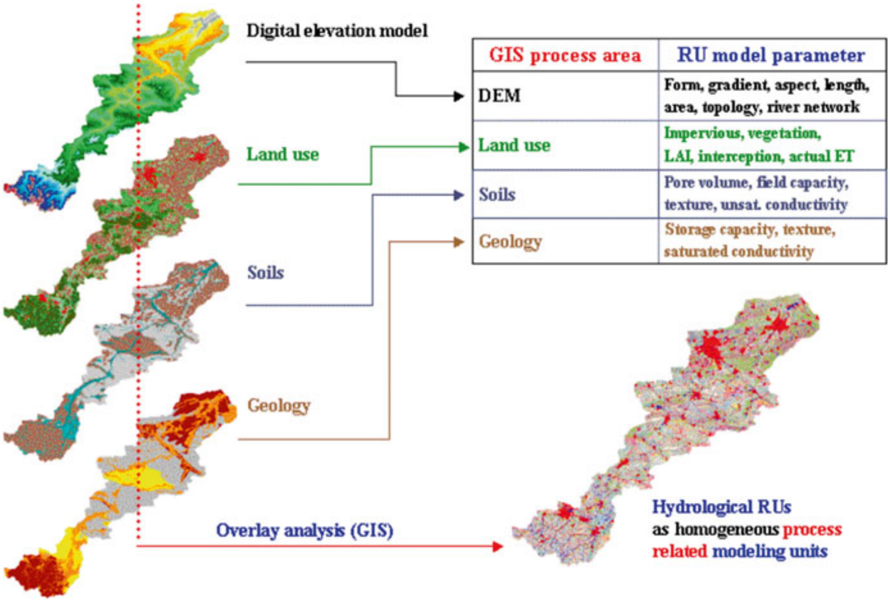

Fig. 3 Scheme for the GIS-based delineation of HRU

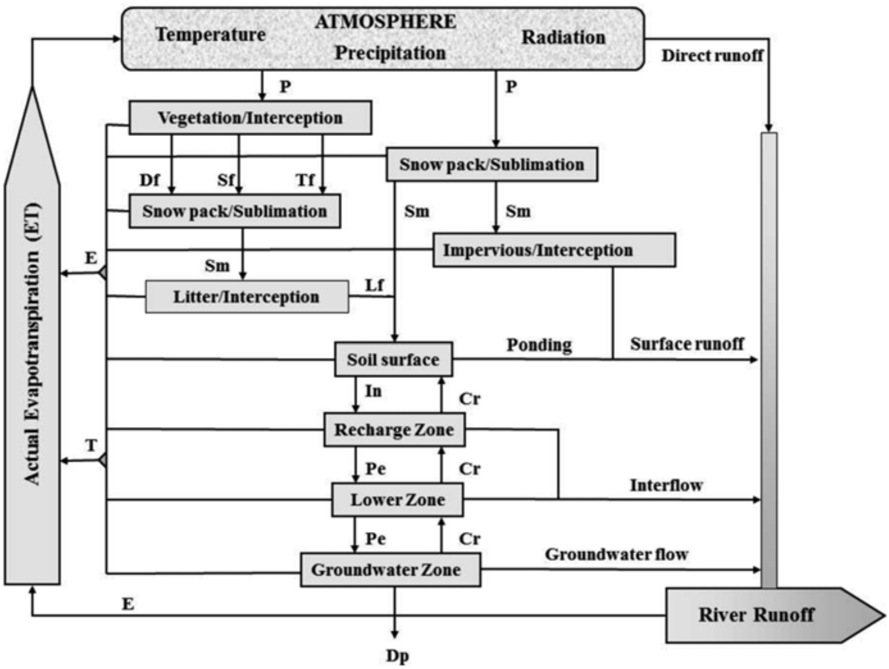

Fig. 4 JAMS/J2000 flow chart

regeneration dynamics of subsurface and surface water resources at catchment scale and supports a scenario based impact analysis for LULCC impact on ESF and ESS (Bende et al. 2007; Fink et al. 2007; Nepal et al. 2012). JAMS/J2000 is meanwhile a component of the Integrated Land-management System (ILMS) platform (Kralisch et al. 2012; Fig. 4).

A restriction of this approach is that only few land-use classes, namely agriculture, forestry, rangeland and impervious areas can be differentiated, whose historically developed spatial distribution is related to topographical aspects that drive soil formation. The use of this classification scheme reduces land-use diversity to a degree that allows for parameterization of the model entities. Land-use changes within a HRU, such as urban sprawl, afforestation or succession cannot be considered. A detailed land-use classification as described in 2.1 would probably lead to the delineation of too small and inefficient modeling entities.

This necessitates an approach that makes use of the process-based HRU, but likewise allows for a more detailed modeling of LULCC. Recent research addresses already an improved representation of LULC based on statistic cluster analysis of the landscape morphometry for multilateral flow simulation (Pfennig et al. 2009).

3 Conceptual Framework for ESF and ESS Provision in Catchment Scales

Based on the concept of HRU (Flügel 1996a, b), and the software platforms ILMS (Kralisch et al. 2012) and GISCAME (Fürst et al. 2010a, b), we propose the concept of a networked distributed modeling approach (Fig. 5). The original idea of the HRU to account for soil-vegetation-atmosphere interactions as drivers for processes shown in Fig. 2 could be enhanced by the consideration of more land-use classes including their spatial and temporal dynamics and landscape metrics assessment. In

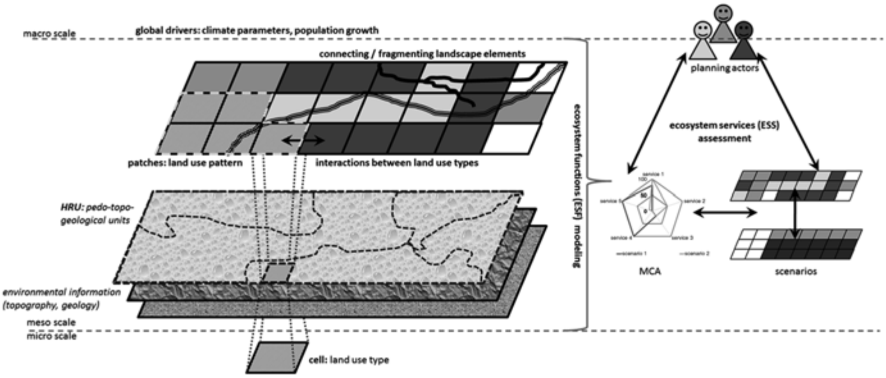

Fig. 5 Conceptual framework overview

this framework, ILMS is needed for the HRU delineation and GISCAME for the cellular-automaton (CA)-based (Cochinos 2000) LULCC simulation and multicriteria impact assessment. Main application area of the combined ILMS and GISCAME toolsets could be the assessment of impacts of LULCC for supporting regional planning (Fürst et al. 2013) and in ILWRM (Flügel 2011b; Kralisch et al. 2012).

The smallest functional entity in the proposed framework is the cell defined as bio-pedo-topological reference unit. Each cell has one specific land-use type as basic attribute, whose impact on ESF and ESS is locally driven by physiogeographic site factors. Associated with the cells, land-use type specific models and empirical data provide information on ESF and ESS (Koschke et al. 2012); at meso-scale, our cells can be integrated within the HRUs.

CA and HRU use different geometric forms for modeling. In CA, these are usually square shaped (Cochinos 2000), while HRU are irregular polygons (Flügel et al. 1996a, b). To achieve spatial compatibility, cell borders could be adapted to pedo-topo-geological conditions and get an irregular shape. Alternatively, physiographic HRU features could be adapted to regular shaped modeling entities. We suggest square shaped modeling entities that are standardized in size and invariable over time. This facilitates nesting, comparative simulation and spatially explicit assessment of LULCC impact on ESF and ESS provision (Norman et al. 2012). Also, approved concepts such as the Moore or v. Neumann neighborhood can be applied for the assessment of interactions between different land-uses, which are not available for irregularly shaped modeling entities (e.g. Itami 1994) and modify ecosystem processes (Verburg et al. 2004). An example related to hydrological processes is nitrate leaching in forests neighbored or surrounded by agricultural fields. Furthermore, HRU and landscape metrics can be merged based on CA as land-use types at cell level and their patches are an important analytical assessment entity for landscape metrics. Patches are defined as areas of spatially connected (neighbored) cells with homogeneous land-use type. Assessing their shape, size, spatial distribution and spatial constellation within HRUs or bridging several HRUs could contribute to better account for the impact of the land-use pattern on hydrological ESF and ESS (Frank et al. 2012) and enhance the gradient based topological model for routing water, nutrient or sediment transport. Connecting or fragmenting effects of linear elements such as roads, highways, railways or hedge rows could be better included taking them as process-relevant cell attributes.

The proposed framework could provide an improved interface to transform biophysically modeled ESF and ESS in ILWRM for regional land-use planning and help to better account for processes that bridge decision scales.

4 Discussion – Applicability of the Framework

A problem in making hydrological ESF and ESS compatible for regional land-use planning is the integration of hydrological processes that cross and link ecosystems and occur at different scales compared to other ESF and ESS. HRU provide a broadly validated methodology for involving process-based components in ESF and ESS assessment (e.g. Immerzeel et al. 2008; Quintero et al. 2009; Welderufael et al. 2013) with a great potential to be developed towards eco-hydrological response units (EHRU; see e.g. Hörmann et al. 2005) that integrate also ecological aspects such as vegetation and land-use dynamics.

Remaining is the question of how to harmonize the interaction of such different modeling approaches and validate the outcomes. Micro-scale models can be easily validated based on measured data (e.g. .g. Pretzsch et al. 2002 for forest models; Schönhart et al. 2011b for agriculture), but they cannot take into account lateral interactions and larger-scale processes. Benefit could be taken from river basin modeling, where this is already done by generating topological routing schemes between model entities (JAMS/J2000; Pfennig et al. 2009). When down-breaking lateral process on cell level, interactions between cells can only be represented empirically so far, as little research has been carried out that allows for detailed parameterization and validation (e.g. Schulp and Veldkamp 2008). Also, landscape metrics based assessments to correct the ESF and ESS assessment within or cross HRU can so far not be validated as comparative studies are missing (Frank et al. 2012).

Another problem is the model performance. Using land-use type specific models to feed each cell in our framework with spatially explicit quantitative or qualitative information including temporal land-use dynamics requires high computational capacities and processing of high data amounts, as each model must be parameterized and validated for various site conditions. Pre-processing for cell parameterization or stepwise modeling to generate larger spatial modeling entities could contribute to reduce this effort. This might provoke either less flexibility in simulating land-use change scenarios or lower precision in the considerations of micro-scale site variability, even more as data and models data are not available for all land-use types (Fürst et al. 2010a, b; Koschke et al. 2012).

Accumulated uncertainties and model errors considering ESF and ESS cannot easily be quantified (Grêt-Regamey et al. 2012). Test, comparison and validation of the outcomes requires harmonized environmental and hydrological monitoring approaches that are land-use and land-use pattern sensitive (Fürst et al. 2012; Fink et al. 2012). Outcomes of the suggested conceptual framework cannot be validated for the moment, but help to identify monitoring needs and improve iteratively the quality of ESF and ESS assessment.

5 Conclusions

The assessment of ESF and ESS and their dependence from LULCC is still faced to many obstacles that result from incompatibilities in land classification and available data for impact assessment, missing understanding of the role of landscape composition and configuration on ESF and ESS and finally from non-compatible modeling approaches for different kind of ESF and ESS. We propose a conceptual framework that brings together a well tested hydrological modeling approach based on process-oriented modeling units (HRU) with a CA based hierarchical accounting for the impact of cell-specific land-uses in their local and regional context. A benefit of this approach could be to provide more detailed consideration of land-use dynamics within HRUs and to further the understanding of ecological processes, their relation to functions and impact on services over different scales. This could provide a more integrated view on how to make use of different model types not only for scientific purposes, but for the concrete questions in regional planning. Ongoing research is focused on the question how to best delineate functional and process-based modeling units that bridge hydrological and ecological processes and serve for a holistic understanding of beneficial spatial planning alternatives for sustaining and enhancing ESF and ESS.

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References

Amiri BJ, Nakane K (2009) Modeling the linkage between river water quality and landscape metrics in the Chugoku district of Japan. Water Resour Manag 23(5):931–956

Anonymous (2005) Richtlinie zu den Bestandeszieltypen (BZT) für den Staatswald des Freistaates Sachsen [Internet]. 2005. Sächsisches Staatsministerium für Umwelt und Landwirtschaft. www.forsten.sachsen.de/wald/249.htm

Anonymous (2010) Federal state office of statistics Saxony, 2010. Flächennutzung in Sachsen 2010.www.statistik.sachsen.de/html/506.htm

Arnold JG, Srinivasan R, Muttiah RS, Williams JR (1998) Large area hydrologic modeling and assessment—part I: model development. J Am Water Resour Assoc 34(1):7389

Bach M, Breuer L, Frede HG, Huisman JA, Otte A, Waldhardt R (2006) Accuracy and congruency of three different digital land-use maps. Landsc Urban Plan 78(4):289–299

Baessler C, Klotz S (2006) Effects of changes in agricultural land-use on landscape structure and arable weed vegetation over the last 50 years. Agric Ecosyst Environ 115:43–50

Bartel A (2000) Analysis of landscape pattern: towards a 'top down' indicator for evaluation of landuse. Ecol Model 130(1–3):87–94

Belcher KW, Boehm MM, Fulton ME (2004) Agroecosystem sustainability: a system simulation model approach. Agric Syst 79:225–241

Bell KP, Irwin EG (2002) Spatially explicit micro-level modelling of land-use change at the rural–urban interface. Agric Econ 27(3):217–232

Bende-Michl U, Kemnitz D, Helmschrot J, Krause P, Cresswell H, Kralisch S, Fink M, Flügel WA (2007) Supporting natural resources management in Tasmania through spatially distributed solute modeling with JAMS/J2000-S. In: Proceedings MODSIM 2007. mssanz.org.au/MODSIM07/papers/43_s47/SupportingNaturals47_Bende-Michl_.pdf

Bossa AY, Diekkrüger B, Giertz S, Steup G, Sintondji LO, Agbossou EK, Hiepe C (2012) Modeling the effects of crop patterns and management scenarios on N and P loads to surface water and groundwater in a semi-humid catchment (West Africa). Agric Water Manag 115:20–37

Brauman KA, Daily GC, Duarte TK, Mooney HA (2007) The nature and value of ecosystem services: an overview highlighting hydrologic services. Annu Rev Environ Resour 32:67–98
Brinson MM (2011) Classification of wetlands. In: LePage BA (ed) Wetlands – integrating multidisciplinary concepts. Springer, Dordrecht/New York
Cochinos R (2000) Introduction to the theory of cellular automata and one-dimensional traffic simulation. www.theory.org/complexity/traffic
Colditz RR, Schmidt M, Conrad C, Hansen MC, Dech S (2011) Land-cover classification with coarse spatial resolution data to derive continuous and discrete maps for complex regions. Remote Sens Environ 115(12):3264–3275
Dale VH, Polasky S (2007) Measures of the effects of agricultural practices on ecosystem services. Ecol Econ 64:286–296
DeFries R, Eshleman KN (2004) Land-use change and hydrologic processes: a major focus for the future. Hydrol Process 18(11):2183–2186
Dover J, Settele J (2009) The influences of landscape structure on butterfly distribution and movement: a review. J Insect Conserv 13(1):3–27
Eigenbrod F, Bell VA, Davies HN, Heinemeyer A, Armsworth PR, Gaston KJ (2011) The impact of projected increases in urbanization on ecosystem services. Proc R Soc B Biol Sci 278(1722):3201–3208
Eisenhauer DR, Sonnemann S (2009) Silvicultural strategies under changing environmental conditions – guiding principles, target system and forest development types. Waldökologie, Landschaftsforschung und Naturschutz 8:71–88
Erb KH (2012) How a socio-ecological metabolism approach can help to advance our understanding of changes in land-use intensity. Ecol Econ 76:8–14
Fink M, Krause P, Kralisch S, Bende-Michl U, Flügel W-A (2007) Development and application of the modelling system J2000-S for the EU-water framework directive. Adv Geosci 11:123–130
Fink M, Wetzel M, Kipka H, Varga D, Kralisch S, Flügel W-A (2012) Designing a measurement network in a meso-scale catchment to provide data for modelling. In: Seppelt R, Voinov AA, Lange S, Bankamp D (eds) Proceedings of the International Environmental Modelling and Software Society (iEMSs) 2012, sixth biennial meeting, Leipzig, Germany
Flügel WA (1996a) Hydrological Response Units (HRU) as modelling entities for hydrological river basin simulation and their methodological potential for modelling complex environmental process systems. – Results from the Sieg catchment. Die Erde 127:42–62
Flügel WA (1996b) Application of GIS to derive hydrological response units for hydrological modelling in the Bröl catchment. IAHS-Publ 235:413–420
Flügel WA (2011a) Twinning European and South Asian river basins to enhance capacity and implement adaptive integrated water resources management approaches – results from the EC-projects BRAHMATWINN. Adv Sci Res 7:1–9
Flügel WA (2011b) Development of adaptive IWRM options for climate change mitigation and adaption. Adv Sci Res 7:91–100
Flügel WA (2011c) Geoinformatics concepts, methods and toolsets for comprehensive impact assessment and analysis of climate change for IWRM. In: Joshi PK (ed) Geoinformatics for climate change studies. Springer/TERI Press, New Delhi
Frank S, Fürst C, Koschke L, Makeschin F (2012) Towards the transfer of the ecosystem service concept to landscape planning using landscape metrics. Ecol Indic 21:30–38
Fürst C, Volk M, Pietzsch K, Makeschin F (2010a) Pimp your landscape: a tool for qualitative evaluation of the effects of regional planning measures on ecosystem services. Environ Manag 46(6):953–968
Fürst C, König HJ, Pietzsch K, Ende HP, Makeschin F (2010b) Pimp your landscape – a generic approach for integrating regional stakeholder needs into land-use planning. Ecol Soc 15(3):34. www.ecologyandsociety.org/vol15/iss3/art34/
Fürst C, Lorz C, Makeschin F (2011) Integrating land-management and land-cover classes to assess impacts of land-use change on ecosystem services. Int J Biodivers Sci Ecosyst Serv Manag 7(3):168–181

Fürst C, Pietzsch K, Frank S, Koschke L, Witt A, Makeschin F (2012) How to better consider sectoral planning information in regional planning – example afforestation and forest conversion. J Environ Plan Manag 55(7):855–883
Fürst C, Frank S, Witt A, Koschke L, Makeschin F (2013) Assessment of the effects of forest land-use strategies on the provision of ecosystem services at regional scale. J Environ Manag 127:96–116
Grêt-Regamey A, Brunner SH, Altweg J, Bebi P (2012) Facing uncertainty in ecosystem services-based resource management. J Environ Manag. http://dx.doi.org/10.1016/j.jenvman.2012.07.028
Groot RS, Wilson MA, Boumans RMJ (2002) A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecol Econ 41(3):393–408
Gulinck H, Wagendorp T (2002) References for fragmentation analysis of the rural matrix in cultural landscapes. Landsc Urban Plan 58(2–4):137–146
Haynes-Young R, Potschin M (2010) The links between biodiversity, ecosystem services and human well-being. In: Raffaelli D, Frid C (eds) Ecosystem ecology: a new synthesis. BES ecological reviews series. CUP, Cambridge
Hein S, Pfenning B, Hovestadt T, Poethke H-J (2004) Patch density, movement pattern, and realised dispersal distances in a patch-matrix landscape—a simulation study. Ecol Model 174(4):411–420
Heinrich K, Ullrich F, Hofmann E (2009) Betriebs- und Landnutzungsformen 2008 in Sachsen. Schriftenreihe des Sächsischen Landesamtes für Umwelt, Landwirtschaft und Geologie 34:63 pp. http://nbn-resolving.de/urn:nbn:de:bsz:14-qucosa-25396
Hörmann G, Horn A, Fohrer N (2005) The evaluation of land-use options in mesoscale catchments: prospects and limitations of eco-hydrological models. Ecol Model 187(1):3–14
Hou W, Walz U (2013) Enhanced analysis of landscape structure: inclusion of transition zones and small-scale landscape elements. Ecol Indic 31:15–24
Hought J, Birch-Thomsen T, Petersen J, de Neergaard A, Oelofse M (2012) Biofuels, land-use change and smallholder livelihoods: a case study from BanteayChhmar, Cambodia. Appl Geogr 34:525–532
Immerzeel W, Stoorvogel J, Antle J (2008) Can payments for ecosystem services secure the water tower of Tibet? Agric Syst 96(1–3):52–63
Itami RM (1994) Simulating spatial dynamics: cellular automata theory. Landsc Urban Plan 30(1–2):27–47
Jaeger JAG, Bertiller R, Schwick C, Müller K, Steinmeier C, Ewald KC, Ghazoul J (2008) Implementing landscape fragmentation as an indicator in the Swiss Monitoring System of Sustainable Development (Monet). J Environ Manag 88:737–751
Janssen S, van Ittersum MK (2007) Assessing farm innovations and responses to policies: a review of bio-economic farm models. Agric Syst 94:622–636
Kim KH, Pauleit S (2007) Landscape character, biodiversity and land-use planning: the case of Kwangju City region, South Korea. Land Use Policy 24:264–274
Knoke T, Stimm B, Ammer C, Moog M (2005) Mixed forests reconsidered: a forest economics contribution on an ecological concept. For Ecol Manag 213(1–3):102–116
Kopp D, Schwanecke W (1994) Standörtlich-naturräumliche Grundlagen ökologiegerechter Forstwirtschaft. Deutscher Landwirtschaftsverlag, Berlin
Koschke L, Fürst C, Frank S, Makeschin F (2012) A multi-criteria approach for an integrated land-cover-based assessment of ecosystem services provision for planning support. Ecol Indic 21:54–66
Koschke L, Fürst C, Lorenz M, Witt A, Frank S, Makeschin F (2013) The integration of crop rotation and tillage practices in the assessment of ecosystem services provision at the regional scale. Ecol Indic 32:157–171
Koschke L, Lorz C, Fürst C, Lehmann T, Makeschin F (2014) Assessing hydrological and provisioning ecosystem services in a case study in Western Central Brazil. Ecol Process 3:2. doi:10.1186/2192-1709-3-2

Kralisch S, Krause P, Fink M, Fischer C, Flügel W-A (2007) Component based environmental modelling using the JAMS framework. In: Kulasiri D, Oxley L (eds) MODSIM 2007, December 2007. mssanz.org.au/MODSIM07/papers/14_s51/ComponentBasedEnvironmenta_s51_Kralisch_l.pdf
Kralisch S, Böhm B, Böhm C, Busch C, Fink M, Fischer C, Schwartze C, Selsam P, Zander F, Flügel WA (2012) ILMS – a software platform for integrated environmental management. In: Seppelt R, Voinov AA, Lange S, Bankamp D (eds) iEMSs proceedings, 2012. www.iemss.org/society/index.php/iemss-2012-proceedings
Krause P, Flügel W-A (2005) Model Integration and development of modular modelling systems. Adv Geosci 4:1–2
Krause P, Bende-Michl U, Bäse F, Fink M, Flügel WA, Pfennig B (2006) Multiscale investigations in a mesoscale catchment – hydrological modelling in the Gera catchment. Adv Geosci 9:53–61
Kristjanson P, Radeny M, Baltenweck I, Ogutu J, Notenbaert A (2005) Livelihood mapping and poverty correlates at a meso-level in Kenya. Food Policy 30(5–6):568–583
Lambin EF, Meyfroidt P (2010) Land-use transitions: socio-ecological feedback versus socio-economic change. Land Use Policy 27(2):108–118
Laszlo A, Krippner S (1998) Systems theories: their origins, foundations, and development. In: Jordan JS (ed) Systems theories and a priori aspects of perception. Elsevier Science, Amsterdam, pp 47–74
Laterra P, Orúe ME, Booman GC (2012) Spatial complexity and ecosystem services in rural landscapes. Agric Ecosyst Environ 154:56–67
Lautenbach S, Kugel C, Lausch A, Seppelt R (2011) Analysis of historic changes in regional ecosystem service provisioning using land-use data. Ecol Indic 11(2):676–687
Lebel L, Daniel R (2009) The governance of ecosystem services from tropical upland watersheds. Curr Opin Environ Sustain 1(1):61–68
Logsdon RA, Chaubey I (2012) A quantitative approach to evaluating ecosystem services. Ecol Model 257:57–65
Lomba A, Bunce RGH, Jongman RHG, Moreira F, Honrado J (2011) Interactions between abiotic filters, landscape structure and species traits as determinants of dairy farmland plant diversity. Landsc Urban Plan 99(3–4):248–258
Lorenz M, Fürst C, Thiel E (2013) A methodological approach for deriving regional crop rotations as basis for the assessment of the impact of agricultural strategies using soil erosion as example. J Environ Manag 127(Suppl):37–47
Łowicki D (2012) Prediction of flowing water pollution on the basis of landscape metrics as a tool supporting delimitation of Nitrate Vulnerable Zones. Ecol Indic 23:27–33
Maltby E, Hogan DV, Immirzi CP, Tellam JH, Van der Peijl M (1994) Building a new approach to the investigation and assessment of wetland ecosystem functioning. In: Mitsch WJ (ed) Global wetlands: old world and new. Elsevier, Amsterdam
MEA (Millenium Ecosystem Assessment) (2005) Ecosystems and human well-being: synthesis. Island Press, Washington, DC. www.millenniumassessment.org
Metzger MJ, Rounsevell MDA, Acosta-Michlik L, Leemans R, Schröter D (2006) The vulnerability of ecosystem services to land use change. Agric Ecosyst Environ 114(1):69–85
Nedkov S, Burkhard B (2012) Flood regulating ecosystem services—mapping supply and demand, in the Etropole municipality, Bulgaria. Ecol Indic 21:67–79
Nepal S, Krause P, Flügel W-A, Fink M, Fischer C (2012) Understanding the hydrological system dynamics of a glaciated alpine catchment in the Himalayan region using the J2000 hydrological model. Hydrol Process. doi:10.1002/hyp.9627. Wiley Online Library
Norman LM, Feller M, Villarreal ML (2012) Developing spatially explicit footprints of plausible land-use scenarios in the Santa Cruz Watershed, Arizona and Sonora. Landsc Urban Plan 107(3):225–235
Pabst RJ, Goslin MN, Garman SL, Spies TA (2008) Calibrating and testing a gap model for simulating forest management in the Oregon Coast range. For Ecol Manag 256(5):958–972

Pfennig B, Kipka H, Wolf M, Fink M, Krause P, Flügel W-A (2009) Development of an extended spatially distributed routing scheme and its impact on process oriented hydrological modelling results. IAHS Publ 333:37–43

Poggio SL, Chaneton EJ, Ghersa CM (2012) The arable plant diversity of intensively managed farmland: effects of field position and crop type at local and landscape scales. Agric Ecosyst Environ 166:55–64. doi:10.1016/j.agee.2012.01.013

Power A (2010) Ecosystem services and agriculture: tradeoffs and synergies. Philos Trans R Soc Lond B Biol Sci 365(1554):2959–2971

Pretzsch H, Biber P, Dursky J (2002) The single tree-based stand simulator SILVA: construction, application and evaluation. For Ecol Manag 162(1):3–21

Quintero M, Wunder S, Estrada RD (2009) For services rendered? Modeling hydrology and livelihoods in Andean payments for environmental services schemes. For Ecol Manag 258(9):1871–1880

Renetzeder C, Schindler S, Peterseil J, Prinz MA, Mücher S, Wrbka T (2010) Can we measure ecological sustainability? Landscape pattern as an indicator for naturalness and land-use intensity at regional, national and European level. Ecol Indic 10:39–48

Rode M, Thiel E, Franko U, Wenk G, Hesser F (2009) Impact of selected agricultural management options on the reduction of nitrogen loads in three representative meso-scale catchments in Central Germany. Sci Total Environ 407:3459–3472

Rounsevell MDA, Pedroli B, Erb KH, Gramberger M, Busck Gravsholt A, Haberl H, Kristensen S, Kuemmerle T, Lavorel S, Lindner M, Lotze-Campen H, Metzger MJ, Murray-Rust D, Popp A, Pérez-Soba M, Reenberg A, Vadineanu A, Verburg PH, Wolfslehner B (2012) Challenges for land system science. Land Use Policy 29(4):899–910

Schober R (1995) Ertragstafeln wichtiger Baumarten, JD-Sauerländer Verlag Frankfurt am Main, 4. Aufl., 166 pp

Schönhart M, Schauppenlehner T, Schmid E, Muhar A (2011a) Integration of bio-physical and economic models to analyze management intensity and landscape structure effects at farm and landscape level. Agric Syst 104:122–134

Schönhart M, Schmid E, Schneider UA (2011b) CropRota – a crop rotation model to support integrated land-use assessments. Eur J Agron 34(4):263–277

Schulp CJE, Veldkamp A (2008) Long-term landscape – land-use interactions as explaining factor for soil organic matter variability in Dutch agricultural landscapes. Geoderma 146(3–4):457–465

Seppelt R, Müller F, Schröder B, Volk M (2009) Challenges of simulating complex environmental systems at the landscape scale: a controversial dialogue between two cups of espresso. Ecol Model 220(24):3481–3489

Spieker H, Hansen J, Klimo E (2004) Norway spruce conversion: options and consequences, European forest institute research reports European forest I (Book 18). Brill Academic Publishers, Leiden

Sponagel H, Grottenthaler W, Hartmann KJ, Hartwich R, Jaentzko P, Joisten H, Kühn D, Sabel KJ, Traidel R (2005) Bodenkundliche Kartieranleitung. Ad-hoc-AG Boden, Schweizerbart'sche Verlagsbuchhandlung

Syrbe RU, Walz U (2012) Spatial indicators for the assessment of ecosystem services: providing, benefiting and connecting areas and landscape metrics. Ecol Indic 21:80–88

Trabucchi M, Ntshotsho P, O'Farrell P, Comín FA (2012) Ecosystem service trends in basin-scale restoration initiatives: a review. J Environ Manag 111:18–23

Troy B, Sarron C, Fritsch JM, Rollin D (2007) Assessment of the impacts of land-use changes on the hydrological regime of a small rural catchment in South Africa. Phys Chem Earth A/B/C 32(15–18):984–994

Tscharntke T, Klein AM, Kruess A, Steffan-Dewenter I, Thies C (2005) Landscape perspectives on agricultural intensification and biodiversity – ecosystem service management. Ecol Lett 8:857–874

Uuemaa E, Roosaare J, Mander Ü (2005) Scale dependence of landscape metrics and their indicatory value for nutrient and organic matter losses from catchments. Ecol Indic 5(4):350–369
Uuemaa E, Mander Ü, Marja R (2013) Trends in the use of landscape spatial metrics as landscape indicators: a review. Ecol Indic 28:100–106
Van Capelle C, Schrader S, Brunotte J (2012) Tillage-induced changes in the functional diversity of soil biota – a review with a focus on German data. Eur J Soil Biol 50:165–181
Van Ittersum MK, Ewert F, Heckelei T, Wery J, Alkan-Olsson J, Andersen E, Bezlepkina I, Brouwer B, Donatelli M, Flichman G, Olsson L, Rizzoli AE, van der Wal T, Wien JE, Wolf J (2008) Integrated assessment of agricultural systems – a component-based framework for the European Union (SEAMLESS). Agric Syst 96:150–165
Van Oudenhoven APE, Petz K, Alkemade R, Hein L, de Groot RS (2012) Framework for systematic indicator selection to assess effects of land-management on ecosystem services. Ecol Indic 21:110–122
Verburg PH, de Nijs TCM, Ritsema van Eck J, Visser H, de Jong K (2004) A method to analyse neighbourhood characteristics of land-use patterns. Comput Environ Urban Syst 28(6):667–690
Verburg PH, Van de Steeg J, Veldkamp A, Willemen L (2009) From land-cover change to land function dynamics: a major challenge to improve land characterization. J Environ Manag 90(3):1327–1335
Verdú JR, Numa C, Hernández-Cuba O (2011) The influence of landscape structure on ants and dung beetles diversity in a Mediterranean savanna—forest ecosystem. Ecol Indic 11(3):831–839
Vigiak O, Borselli L, Newham LTH, McInnes J, Roberts AM (2012) Comparison of conceptual landscape metrics to define hillslope-scale sediment delivery ratio. Geomorphology 138(1):74–88
Volk M, Möller M, Wurbs D (2010) A pragmatic approach for soil erosion risk assessment within policy hierarchies. Land Use Policy 27(4):997–1009
Von Haaren C, Reich M (2006) The German way to greenways and habitat networks. Landsc Urban Plan 76:7–22
Welderufael WA, Woyessa YE, Edossa DC (2013) Impact of rainwater harvesting on water resources of the model river basin, central region of South Africa. Agric Water Manag 116:218–227
Witt A, Fürst C, Makeschin F (2013) Regionalization of climate change sensitive forest ecosystem types for potential afforestation areas. J Environ Manag 127:48–55
Wolf M, Pfennig B, Krause P, Flügel W-A (2009a) Delineation of topographic process entities using SRTM for hydrological modelling. 18th World IMACS/MODSIM Congress, Cairns, Australia. http://mssanz.org.au/modsim09
Wolf M, Pfennig B, Krause P, Flügel W-A (2009b) Landscape dependent derivation of J2000 model parameters for hydrological modelling in ungauged basins. New approaches to hydrological prediction in data sparse regions. IAHS Publications, p 333
Yeh CT, Huang SL (2009) Investigating spatiotemporal patterns of landscape diversity in response to urbanization. Landsc Urban Plan 93:151–162
Zebisch M, Wechsung F, Kenneweg H (2004) Landscape response functions for biodiversity – assessing the impact of land-use changes at the county level. Landsc Urban Plan 67:157–172
Zhang W, Ricketts TH, Kremen C, Carney K, Swinton SM (2007) Ecosystem services and dis-services to agriculture. Ecol Econ 64(2):253–260
Zhou D, Lin Z, Liu L (2012) Regional land salinization assessment and simulation through cellular automaton-Markov modeling and spatial pattern analysis. Sci Total Environ 439:260–274

Valuation of Ecosystem Services Regarding the Water Framework Directive on the Example of the Jahna River Catchment in Saxony (Germany)

Karsten Grunewald and Sandra Naumann

Abstract Using the example of the catchment area of the Jahna River in the loess region of Saxony (Germany) status and deficit analyses were accomplished with regard to environmental objectives of the EU Water Framework Directive. Currently, there is no body of water in the study area in good ecological condition. In this intensively used agricultural area, the nutrients nitrogen and phosphorus as well as the acceptance and costs of measures against soil erosion and water pollution were determined as target variables. Significant water pollution through nutrient leaching was identified, and spatially differentiated risk and reduction potentials were demonstrated. The multi-criteria analysis in form of the utility value analysis was applied to support the objective decision process in the selection and prioritization of measures to reduce erosion and nutrient inputs to water. By equal weighting of the target variables the conservation tillage on the critical source areas and the cultivation of catch crops are representing the measures with the highest benefit in the catchment area of the Jahna River. A benefit-cost-ratio of 2 to 1 was estimated.

Keywords Jahna river basin • EU Water Framework Directive • Status and deficit analysis • Critical source areas • Utility value analysis

1 Introduction

The agricultural land use takes up a central place in the biodiversity and ecosystem services debate. Two fundamental aspects are at odds here: the ecological problems (poorly structured, drained, intensively cultivated agricultural landscapes; substances input into waters) on the one hand, and socio-economic needs (food and income securing, agricultural subsidies, labor stocking, food ethics, consumer

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protection, etc.) on the other. Consequently, environmental markets have emerged that take into account biodiversity and ecosystem services (ES), directly or indirectly. These include, inter alia:

– products from certified organic agriculture
– the remuneration of the landscape management and contractual nature conservation
– payments for watershed services (private and voluntary)
– payments for water-related ES (government-funded).

Soil and water are components in the landscape system, which are characterized by high complexity and are connected through a variety of interfaces with the other geo-components. The resulting ecosystem-based importance is taken into account on the one hand in the environmental objective of the soil and water conservation in the German Federal Nature Conservation Act (§ 1 BNatSchG 2009). On the other hand soil and water provide the requirement to fulfill a variety of utility functions (ES) and thus gain an existential meaning for the human society.

Hydrological ES (water regulation and water purification) as well as pedological ES (erosion and maintenance of soil fertility) form separated sub-groups within the group of regulatory ES. Moreover, freshwater presents one of the essential supply ES (direct market good). Less attention is mostly paid to indirect, usage-independent ES, for example, natural soil profile as an education ES. Soil and Water conservation form an inseparable unit (Fig. 1). For example, the soil type is an indicator for both the ES water regulation and water purification (chapter Bastian et al.).

Fig. 1 Land use induced soil erosion on arable land and matter input into the waters of the Kleine Jahna rivulet near the town Riesa. It is hard to define the on-site- und off-site damages and costs © Karsten Grunewald

Soil is a limited resource and the regeneration of soils, their functionality is affected by various forms of exploitation, is extremely difficult, if not impossible, and sometimes very costly. A European Soil Framework Directive with the aim of protecting soils is rejected among others by the government of Germany, in particular for reasons of subsidiarity (subordination of rules to others, e.g. WFD). In addition, it is also feared that the implementation of a Soil Framework Directive causes a disproportionate administrative burden and high consequential costs (Kluge et al. 2010).

However, the benefits of ES for water protection have become a high priority in policy-making. The EU Water Framework Directive (WFD 2000) is designed to harmonize the legal framework of water policy in the EU. It also aims at a stronger orientation of the water policy towards a sustainable and environmentally-friendly use of waters. The core of this directive is the establishment of environmental goals including sustainable land use (long-term sustainable water management basing on a high level of protection for the aquatic environment), and also the optimization of ES (e.g. human health protection, economic consequences).

The "translation" of the normative regulations in the WFD into numerical class limits of a "favourable state" is based on scientific methods. Socio-economic aspects are also taken into consideration by the WFD in form of "exceptions" from the goals, and of cost efficiency analyses.

The goals of the WFD mainly imply the following benefits, reflecting a whole bundle of ES:

– Human health protection by water-related utilizations, e.g. bathing water quality, drinking water quality
– Lower costs for water purification
– Maintenance of water supply
– Improvement of life quality by increasing the recreation value of surface waters
– Coping with conflicts and regional damages through the balance of interests among different social groups.

The precautionary principle, information and transparency shall be considered consequently. The WFD contains mechanisms to assure that socio-economic effects are considered in decision-making processes and that cost-effective options are preferred. The implementation of the environmental goals, however, can cause additional costs but it can be profitable for some beneficiaries too (e.g. landscape management companies), and – in the long run – for the whole society. According to the particular watershed the goals depend on the difference between the actual and the target state as well as on the choice of instruments and management measures. Space-time approaches play a decisive role in this context (Bastian et al. 2012).

The WFD requires water planning of whole, usually several 1,000 km^2 watersheds as units of management. For this purpose, the professional and inter-territorial cooperation of authorities, associations, actors and other stakeholders at the watershed scale is necessary, in the sense of a coherent and targeted sectoral planning and its subsequent implementation. The development of the best solution for the aquatic conserving management of a watershed, including the wishes of the various stakeholders is in general problematic. One solution provides the application of a decision support system, which supports the actors involved in the departmental planning technically and professionally. Integrated planning must be broken down from the level of river basin management to the level of small catchments (up to 300 km^2), since the implementation of concrete measures takes place here (Naumann and Kurzer 2010).

The catchment area of the Jahna River in Saxony (Germany), mainly used for agriculture, was selected as the study area for assessing of ES in order to achieve environmental objectives of the WFD, since it has already been studied extensively. The goal of the case study was to analyze the cost-effective combination of measures of agriculture to reduce water erosion and diffuse nutrient inputs in water and to assess selected, mainly not market-based ES.

The fundamental approach is based on the EPPS conceptual framework (Ecosystem Properties, potential and Services, see Bastian et al. 2012; Grunewald and Bastian 2015). The core is, that through the (natural) scientific bases the professional requirements of use/conservation of natural resources can be expressed (interdisciplinary, multiple political consideration). These characteristics and pressures on ecosystems are analyzed using ecological indicators and reduction potentials are simulated by models. Furthermore changes in agricultural land use and management form are also evaluated monetarily in regard of reduction in nutrient input into waters.

2 Structures, Processes and Selected Pollutions in the Catchment Area

The catchment area of the river Jahna is 244 km^2 in size, is part of the natural region Lößgefilde (loess-region) in Saxony and has a rural typical, relatively low population density. Due to the very fertile soils the river basin is primarily used for agricultural purposes since time immemorial. About 90 % of the land is occupied by agricultural land. The share of only 6 % grassland suggests that pure arable farms are predominantly located in the study area. Main crops grown are wheat, corn, rapeseed and root crops (sugar beet). About 14 % of the Jahna-catchment is designated as protected area which partially overlap (7 % drinking water protection areas, 6.1 % areas of protected landscape, 3.8 % bird protection areas (SPA), 2.4 % habitat protection areas (FFH), 0.2 % Nature Reserves).

Luvisols occur as soil types over a large area, Cambisols, Albeluvisols and Luvisol-Planosols/Stagnosols are also found in smaller plots. In the valleys and depressions partly mighty colluvisols indicate, that a high erosion deposition in the area are found. Accordingly, at the upper slope or slope shoulder locations extensively capped profiles and completely eroded areas are distributed. Water erosion is a problem in the study area since the beginning of the intensive land use of the landscapes. This is documented in the colluviums and highflood loams respectively in the altered sediment load of waters.

The river Jahna has a length of about 35 km and flows into the Elbe River in Riesa. Numerous interventions in the water system were undertaken in the catchment area, such as longitudinal and transverse profile barriers, run-straightening and relocations, melioration etc. About 40 dams/reservoirs respectively ponds characterize the surface water system currently, the reservoir Baderitz with 15.8 ha is the largest among them.

According to the WFD, the biological components fish, macroinvertebrates and macrophytes/phytobenthos are relevant for assessing the surface water bodies. These were assessed without exception as deficient in the catchment area of the Jahna River in the period 2005–2007. The nutrient pollution reflects the poor biological evaluation in the catchment. With the exception of one river water body, the guidance values for total-P and ortho-phosphate-P were exceeded two to threefold in all years. Currently there are no reference values for the WFD-relevant elements total-N and nitrate-N available. Compared to the nitrate-quality standard for the chemical status it will be clear, however, that all the surface water bodies are significantly affected by nitrogen. With a mean of 97 mg L^{-1} during the period 2007–2009, the quality standard for nitrate (50 mg L^{-1}) is significantly exceeded in the groundwater body Jahna. The monitoring results of the inventory lead to the conclusion that the objectives of the WFD cannot be achieved in the groundwater body Jahna and all eight river water bodies in the catchment area Jahna by 2015.

Cause analysis for the diffuse nutrient sources were calculated using the model STOFFBILANZ (Grunewald et al. 2007). The Web-GIS-based model STOFFBILANZ (www.stoffbilanz.de) is a method for quantification of sources and path-related nonpoint source pollution (nitrogen, phosphorus and sediment) from the surface (emission) in catchments of mesoscale size. In addition, the quantification of the immission, resulting from the matter inputs to surface waters, is possible using simple estimation methods.

Modelling results for phosphorus (P): in average 52 % of the total P-emissions of 14.5 t $year^{-1}$ of the Jahna catchment originate from agricultural land. The majority of the agricultural P-discharge is caused by particulate phosphorus (PP) input via water erosion (Haygarth et al. 1998, Fig. 1). Almost 80 % of PP inputs into surface waters are from the critical source areas. It means the majority of loss comes from a small part of the catchment where areas of high potential for supply (source) and transport (e.g., surface runoff) overlap. These areas termed critical source areas (CSAs, cf. Heathwaite et al. 2005; Halbfaß and Grunewald 2008; Qui 2009). The estimation of P concentrations from the total P loads (including upstream) resulted in a span from 0.33 to 0.73 mg L^{-1} for emissions. By an average P retention of about 70 % an immission load of 3.9 t $year^{-1}$, and P concentrations from 0.09 to 0.23 mg L^{-1} were determined for the surface waters.

Modelling results for nitrogen (N): According to the assessment by the STOFFBILANZ model in average 95 % of total N-emissions in the catchment area Jahna (574 t $year^{-1}$) originate from agricultural land. In contrast to phosphorus, nitrogen is discharged almost all dissolved on the different flow types. The underground drainage component baseflow dominates followed by intermediate and drain discharge. If a catch crop of 4 % (as in 2005/06 normal) of arable land in the catchment area Jahna is considered, STOFFBILANZ calculated a reduction of the total diffuse N-emissions (including from settlements) on watershed level of about 8–11 % (N-removal by the intercrops of 80 or 100 kg year^{-1}). Based on the N loads (including upstream), there were emissions-based N concentrations from 11 to 23 mg L^{-1}. Taking an average retention of 62 % into account a total load of 208 t year^{-1} (immission) and total N concentrations between 4 and 8 mg L^{-1} were calculated. With respect to the groundwater flow in the Jahna aquifer an N-input of 349 t year^{-1} in the surface waters was determined. This corresponds to an average load of 7.9 kg ha^{-1} year^{-1}. An average N concentration of approximately 69 mg t L^{-1} in groundwater discharge results from a modeled average base flow of 51 mm year^{-1}.

3 Determining of Endangering and Reduction Potentials

Maps of the potential risk of soil erosion by water in Saxony can be used to estimate and steer erosion control measures, such as conservation tillage farming. In addition to the approach, which is used currently in the context of cross-compliance and takes into account the slope inclination, the erodibility of the soil type and the long-standing average rain erosion factor, a comprehensive assessment of the potential risk of erosion was elaborated, which moreover includes the slope length (Bräunig 2009). The calculation was based on the Universal Soil Loss Equation (USLE) and resulted in the long-term average soil loss in tones per hectare and year for a bare soil (fallow land) only for the agriculturally used soils. The calculation results re-present only potential threats and not actual soil erosion because the soil cover and soil erosion prevention, such as conservation tillage, field subdivision and landscape elements (e.g. hedges) were not taken into account. Figure 2 shows the potential risk of erosion of the agricultural land in the catchment area Jahna. Erosion protection measures should be performed in particular in the highly erodible southern part of the Jahna catchment area so that the high risk of the potential input of sediment and PP in the waters will be reduced.

Surface runoff is often focused (bundled) on relief-based linear runoff pathways. A preventive measure against linear erosion is the greening of slope gutters and depth line by permanent grassland or forest. Since erosive runoff pathways mainly occur in areas with a stronger relief, the measure should always be implemented in combination with area-wide (planar) erosion protection measures (conservation tillage). Corresponding information are available in Saxony (e.g. Voß 2010).

Different agricultural measures and sets of activities were simulated with the model STOFFBILANZ to estimate the nutrient-reduction potential (Table 1, Fig. 3). The reduction of the P-input is shown on the basis of PP-emission, since a large proportion of the agricultural P discharge takes place via this path (Fraser et al. 2009; Halbfaß and Grunewald 2003; Pimentel et al. 1995). For nitrogen, however, the total diffuse N emissions are shown in Table 1.

The individual measures with the greatest reduction potential of P-input (40–70 %) are the conversion of arable land into grassland on the CSAs, conservation

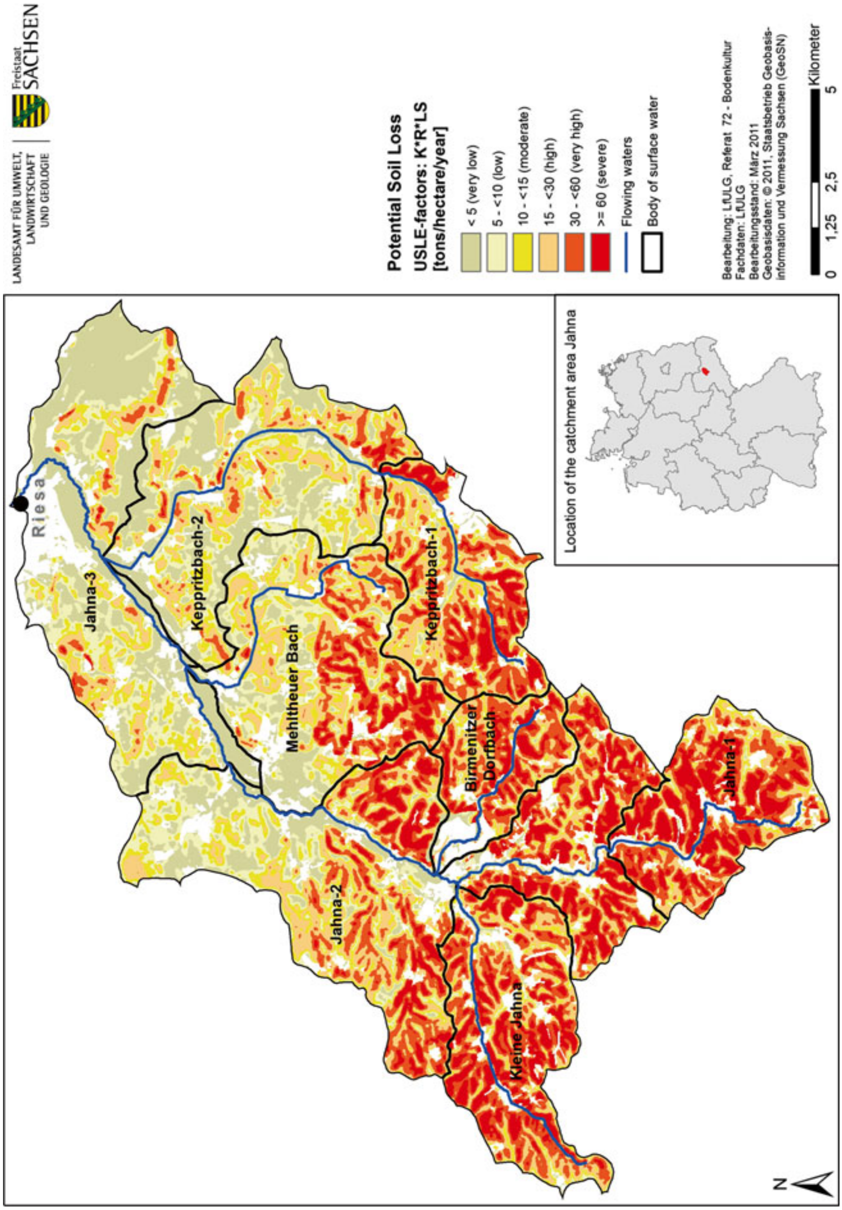

Fig. 2 Potential water erosion on arable lands in the river catchment Jahna

Table 1 Selected results of scenarios for the river catchment Jahna modelled with STOFFBILANZ

No.	Scenario/variant	Particulate P-input			Diffuse N-input		
		[t year^{-1}]	[kg ha^{-1} year^{-1}]	[%][a]	[t year^{-1}]	[kg ha^{-1} year^{-1}]	[%][a]
1	Increase of conservation tillage to 100 % on CSAs	3.5	0.14	−31	537	21.9	−3
2	Increase of conservation tillage to 100 % on the whole arable land	2.8	0.11	−45	451	18.4	−19
3	No maize and root crop cultivation on CSAs	3.7	0.15	−27	555	22.7	0
4	Greened runoff pathways on CSAs	4.6	0.19	−9	553	22.6	0
5	Water protection stripes (buffer zones)	2.7	0.11	−47	543	22.2	−2
6	Land use change on CSAs	1.6	0.07	−68	538	22.0	−3
7	Land use change on areas with the highest N-leaching	5.0	0.20	−2	489	20.0	−12
8	Catch crop cultivation – actual state + 5 % in the catchment	–	–	–	449	18.3	−19
9	Catch crop cultivation – actual state +16 % in the catchment (max.)	4.9	0.20	−3	324	13.2	−42
10	Reduced N-fertilizing if N-leaching is over 25 kg N ha^{-1} year^{-1}	–	–	–	385	15.7	−31
11	Reduced N-fertilizing on the whole arable land	–	–	–	291	11.9	−47
12	Combination of measures No. 1, 3, 5, 8, 10	1.8	0.07	−64	209	8.6	−62
13	Combination of measures No. 1, 5, 6, 8, 10	0.5	0.02	−90	240	9.8	−57
14	Combination of measures No. 2, 3, 7, 8, 11	0.6	0.03	−88	165	6.8	−70

[a]Decrease of P- or N-input compared to the actual state

Fig. 3 Mulch seeding as practise of conservation tillage (Photo *left*: Walter Schmidt) and cultivation of catch crops white mustard (*in front*) and phacelia (*at back*) (Photo: Anja Schmidt)

tillage and buffer zones (water protection stripes). If measures are combined, however, a reduction of PP-emission of 90 % is possible.

The measures intercropping and reduced fertilization have with 30–50 % the largest reduction potential for the N-input. The combination of measures can reduce the N-input up to 77 %. Looking at the reduction potential of the measures in each river water body, the amount depends on grown crops and the location of CSAs. Thus, the effect of conservation tillage on the P-leaching is stronger in the upper catchment area Jahna than in the lower one due to the higher number of CSAs. The same applies to the other measures on the CSAs: 'conversion of arable land into grassland', 'grassed drainage channels' and 'renunciation of maize and root crop cultivation'. The N-discharge shows greater differences on plots with highest N-leaching due to crop-specific intercropping systems and reduced nitrogen fertilization (Grunewald et al. 2007).

4 Services and Welfare Effects (Cost-Benefit-Analysis)

One way to evaluate and prioritize different measure scenarios provides the utility analysis (Zangemeister 1971). The different target variables can be better compared with each other through their transmission in a common value system. As targets for the WFD implementation, the reduction of N and P inputs into the waters and the costs and acceptability of measures plays an important role. Utility functions between 0 (no benefit) and 1 (highest benefit) are defined for these target variables, which in case of the nutrients are determined by environmental quality standards or guidance values (Naumann and Kurzer 2010). The part-worth utilities of the various scenarios were determined for the target variables with these utility functions. The total benefit of the different measure scenarios to be compared was the result of adding the part-worth values of the target variables, whereby a weighting of the target variables was still carried out by the agent (Table 2).

Table 2 Partly and total utility values of target variables of measures scenarios for the river catchment Jahna

Target variables/ Measure variables	Particulate P		Diffuse N		Costs		Acceptance		Total utility
Conservation tillage on CSAs (100 %)	0.80	0.2	0.29	0.07	0.90	0.23	0.5	0.13	**0.63**
Conservation tillage on the arable land (100 %)	1.00	0.25	0.57	0.14	0.21	0.05	0.5	0.13	**0.57**
Greened runoff pathways on CSAs	0.40	0.1	0.25	0.06	0.49	0.12	0	0	**0.28**
Water protection stripes (buffer zones)	1.00	0.25	0.27	0.07	0.92	0.23	0	0	**0.55**
Conversion of arable land into grassland on CSAs	1.00	0.25	0.29	0.07	0.72	0.18	0	0	**0.50**
Conversion of arable land into grassland on areas with the highest N-leaching	0.20	0.05	0.45	0.11	0.72	0.18	0	0	**0.34**
Catch crop cultivation 9 %	0.20	0.05	0.57	0.14	0.87	0.22	0.5	0.13	**0.54**
Catch crop cultivation 20 %	0.30	0.08	0.97	0.24	0.60	0.15	0.5	0.13	**0.60**

By an equal weighting of the target variables conservation tillage on the CSAs represents the measure with the highest total utility in the Jahna catchment area. The reasons for this are the low costs, the relatively low P-concentration by the modeled PP-input and the mean acceptance of the measure by the farmers. This is followed by the measure 20 % catch crops, whose high total utility is mainly due to the high part of the benefit in N-concentration, and the measure catchment-wide implementation of conservation tillage. If more emphasis is placed on the nutrient input, the extensively conservation tillage farming is the preferred option, followed by catch crop. The high part-worth utility of P-concentration of the conservation tillage and the high part-worth utility of N-concentration of intercropping contribute to this result. If the costs however have the highest priority despite the high relative cost water protection strips (buffer zones) gain in importance, as due to the small area the total costs for the watershed Jahna are low. The measures ‘greened runoff pathways’ and ‘conversion of arable land to grassland’ on areas with highest N-leaching occupy by all weights only lower ranking positions due to the low modeled contribution to matter input reduction and low acceptance.

According to the Saxon Funding Guidance (AuW 2007) agri-environment measures were performed on a total of 22 % of the agricultural land in the Jahna catchment area in 2010. The permanent conservation tillage/direct sowing method was funded on 20 % of the arable land, the use of catch crops on 5 %, and the investment of green strips or fallow strips/areas on 25 ha (0.13 % of agricultural land).

5 Monetisation and Discussion

The environmental costs of erosion and nutrients emission are not precisely known. Likewise, the social benefits of erosion protection and the reduction of nutrient translocation/leaching can only be estimated. In this case it is questioned whether the cost-benefit analysis is sufficiently precised to capture concrete effects of projects, measures and policies. The development of non-market-based valuation methods has grown fast recently (e.g. Holm-Müller and Muthke 2001; Grossman et al. 2010), but so far these are hardly addressed or even applied for erosion protection and nutrient retention.

In economic terms the level of protection is optimal, in which the avoided costs of damage (expected erosion and nutrient leaching risk) correspond with the cost of the erosion protection and nutrient retention in a marginal cost consideration. The costs of a further reduction of the residual risk are higher up from this level of protection than the recoverable minimizing risk (Grossmann et al. 2010).

Crop yields on eroded soils are lower than those on protected land as erosion reduces the ES soil fertility and water availability. Erosion affects soil quality and productivity adversely by reducing water infiltration rates, water-holding capacity, nutrients, organic matter, soil biota, and soil depth. Moderately eroded topsoils absorb from 10 to 300 mm less water per hectare per year than uneroded soils (correspond to 7–44 % of total rainfall, see Pimentel et al. 1995). A ton of fertile agricultural topsoil typically contains 1–6 kg N and 1–3 kg P, which can be lost through runoff. These are so-called on-site damages, which land owners and users wants to keep as low as possible. As shown in the previous sections the losses can be significantly reduced by erosion control measures.

The off-site costs of erosion must also be considered. The soil loss not only represents a loss for farmers but also can affect habitats on neighbouring areas adversely or blocks the public sewage system, which must then be cleaned with financial expense. The hydroecological damages were outlined (sediment and nutrient input, eutrophication, increased water treatment costs, etc.). The real costs of this are not exactly quantifiable and the causers are hard to make liable, even for small, localizable erosion events. Nevertheless, the entity shall presume that both the individual and the society are interested to keep the off-site damages (and therefore costs) as low as possible.

Pimentel et al. (1995) had estimated the on-site and off-site costs of erosion in the U.S. and came to about U.S. $ 100 per hectare per year in the mid-1990s. If one estimates the so-called replacement costs of soil and fertilizer (according to Internet research a ton of topsoil costs about 10 € and current fertilizer prices are to be set at about 600 € t^{-1} for N respectively 750 € t^{-1} for P; Source: AMI 2010) and damage costs (cleaning of roads, land, properties after erosion damages, desludging of reservoirs, ponds, canals, etc.), one arrives at a similar monetary magnitude of on-site and off-site damages for the Jahna catchment area.

Accordingly, replacement and damage costs of € 1.4 million per year would be calculated for the nearly 20,000 ha of agricultural land in the Jahna catchment area (with USD/EUR exchange rate of 1.4). A comparison of this order of magnitude with erosion reduction measures revealed a very positive benefit-cost ratio in areas with high erosion threats. Pimentel et al. 1995 give this example for the United States with 5 to about 1; thereby reducing soil erosion by water and wind from 17 to 1 t ha^{-1} $year^{-1}$. A benefit-cost ratio of about 2 to 1 would result for the society assuming the costs of most effective measures in the watershed Jahna with 760,000 € (100 % conservation tillage on CSAs and catch crops on 20 %, and current funding rates: 85 € ha^{-1} for intercropping, 68 € ha^{-1} for conservation tillage).

The assessment of the benefits is primarily oriented to the objectives of the WFD in the case study. Tangent goals concern soil protection, nature conservation, agricultural productivity and others. An integrated assessment and planning takes the impact of measures on all relevant target dimensions into account. The area under consideration – in this case, the catchment area of Jahna – therefore represents a common field of action for water management, agriculture and nature conservation.

The major criticism of the presented and for the catchment area of the Jahna exemplified ES-approach is that the data are required for the operationalization of quantitative models and the monetization (benefit transfer method in this case) are methodologically uncertain. Only few of the ecological services have an economic value, which land users can realize on markets. Numerous ES of catchment areas are in economic terms public goods. This means that markets do not adequately reflect the costs and benefits associated with a change in supply. Furthermore, it is unfortunate that social, aesthetic and health values are under-represented in the ES-evaluation and -planning.

A monetary assessment cannot capture all the values of an ecosystem (Spangenberg and Settele 2010; Grunewald and Bastian 2015). But by applying economically oriented planning methods and usage of a benchmark such as money exchange values of planning variants are more visible and more conscious.

References

AMI (2010) Marktinfo Düngemittel. Bericht Nr. 4 der Agrarmarkt Informations-Gesellschaft mbH (AMI), Bonn

AuW (2007) Richtlinie des Sächsischen Staatsministeriums für Umwelt und Landwirtschaft zur Förderung von flächenbezogenen Agrarumweltmaßnahmen und der ökologischen Waldmehrung im Freistaat Sachsen. SächsABl. Jg. 2007, Bl.-Nr. 49, S. 1694

Bastian O, Grunewald K, Syrbe RU (2012) Space and time aspects of ecosystem services, using the example of the European Water Framework Directive. Int J Biodivers Sci Ecosyst Serv Manag 8(1–2):5–16. doi:10.1080/21513732.2011.631941

BNatSchG – Bundesnaturschutzgesetz (2009) Gesetz über Naturschutz und Landschaftspflege. BGBl. I, S 2542

Bräunig A (2009) Erläuterung zu Kartendarstellungen und GIS-Daten der potenziellen Wassererosionsgefährdung. LfULG, Ref. 64, 11 S. (unveröff.), Freiberg

Fraser AI, Harrod TR, Haygarth PM (2009) The effect of rainfall intensity on soil erosion and particulate phosphorus transfer from arable soils. Water Sci Technol 39(12):41–45. doi:10.1016/S0273-1223(99)00316-9

Grossmann M, Hartje V, Meyerhoff J (2010) Ökonomische Bewertung naturverträglicher Hochwasservorsorge an der Elbe. Naturschutz und Biologische Vielfalt 89, Bonn

Grunewald K, Bastian O (ed) (2015) Ecosystem services – Concept, methods and case studies. Springer, Heidelberg

Grunewald K, Gebel M, Kaiser M, Halbfaß S, Bürger S (2007) Erstellung, Analyse und Bewertung ausgewählter Nutzungs- und Bewirtschaftungsszenarien im Flussgebiet Jahna im Hinblick auf die Erreichung von Umweltzielen der WRRL sowie Erarbeitung gebietsspezifischer Beiträge zum Maßnahmeprogramm und Bewirtschaftungsplan der Flussgebietseinheit Elbe. Entwurf Abschlussbericht (unveröff.), Dresden, 95 S

Halbfaß S, Grunewald K (2003) Räumliche Variabilität der Phosphorgehalte im Oberboden landwirtschaftlich genutzter Flächen in kleinen Einzugsgebieten. Plant Nutr Soil Sci 166:197–203

Halbfaß S, Grunewald K (2008) Ermittlung räumlich verteilter SDR-Faktoren zur Modellierung von Sedimenteinträgen in Fließgewässer im mittleren Maßstab. WasserWirtschaft 3:31–35

Haygarth PM, Hepworth L, Jarvis SC (1998) Forms of phosphorus transfer in hydrological pathways from soil under grazed grassland. Eur J Soil Sci 49:65–72

Heathwaite AL, Quinn PF, Hewett CJM (2005) Modelling and managing critical source areas of diffuse pollution from agricultural land using flow connectivity simulation. J Hydrol 304:446–461

Holm-Müller K, Muthke T (2001) Aktueller Einsatz und Perspektiven der Nutzen-Kosten-Untersuchung zur Vorbereitung von Investitionsentscheidungen in der Wasserwirtschaft. Zeitschrift für Umweltpolitik 23(4):455–473

Kluge HG, Ley F, Wittberg V (2010) Gutachten zur Abschätzung der Verwaltungskosten. www.unserboden.at/files/gutachten_zur_abschaetzung_der_verwaltungskosten_final.pdf. 28 Feb 2012

Naumann S, Kurzer HJ (2010) Etablierung eines Entscheidungshilfesystems zur Erstellung von Managementplänen auf Einzugsgebietsebene. Schriftenreihe des LfULG, Heft 7, 144 S

Pimentel D, Harvey C, Resosudarmo P et al (1995) Environmental and economic costs of soil erosion and conservation benefits. Science 267:1117–1123

Qui Z (2009) Assessing critical source areas in watersheds for conservation buffer planning and riparian restoration. Environ Manag 44:968–980. doi:10.1007/s00267-009-9380-y

Spangenberg JH, Settele J (2010) Precisely incorrect? Monetising the value of ecosystem services. Ecol Complex 7:327–337

Voß J (2010) Erosionsschutz in reliefbedingten Abflussbahnen. Schriftenreihe des Landesamtes für Umwelt, Landwirtschaft und Geologie, Heft 13, Dresden

WFD (Water Framework Directive) (2000) Directive 2000/60/EG of the European Parliament and of the Council establishing a framework for the Community action in the field of water policy. Published in the Official Journal (OJ L 327) on 22 December 2000

Zangemeister C (1971) Nutzwertanalyse in der Systemtechnik. Eine Methodik zur multidimensionalen Bewertung und Auswahl von Projektalternativen. 4. Aufl. Diss., Techn. Univ. Berlin, 1970, München, 370 S

Water-Related Ecosystem Services – The Case Study of Regulating Ecosystem Services in the Kielstau Basin, Germany

Britta Schmalz, Marion Kandziora, Nina Chetverikova, Felix Müller, and Nicola Fohrer

Abstract For planning and obtaining a sustainable environmental management of river basins, stakeholder information and participation is an important procedure for the decision-making process. Stakeholders need suitable information on spatio-temporal variations in ecosystem services which can be derived by different quantification methods such as modelling. This study shows an approach to combine ecohydrological modelling results with valuation methods for assessing ecosystem services. The rural lowland Kielstau river basin in Northern Germany serves as the study area. Based on the results of the ecohydrological model SWAT, simulated variables were used as indicators for regulating services and were directly translated into a 0–5 valuation scale for each different land use/land cover classes. One detailed example is given by providing and analysing the SWAT variable 'sediment yield' as an indicator for the regulating ecosystem service 'erosion regulation'. The SWAT model results reveal the temporal changes in erosion regulation due to crop rotation and different precipitation patterns over the years, which includes important information in the assessment of ecosystem services and the formulation of management actions.

Keywords Water-related ecosystem services • Erosion regulation • Sediment yield • SWAT model • Kielstau basin

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1 Introduction

Ecosystem services, i.e. the benefits people obtain from nature, are gaining more and more attention in scientific research and policy-making as an important concept for managing the diverse ecosystems globally (Daily 1997; Tognetti et al. 2004; MA 2005; Doyle and Yates 2010; Ojea et al. 2012). Important elements of the ecosystem services concept are the water-related ecosystem functions (Müller 2005), as water plays an outstanding role in the supply of various ecosystem services (Vigerstol and Aukema 2011). They are often defined as the ecosystem services that are provided by the ecosystems in a watershed (Smith et al. 2006). Here the term is perceived more closely to the connection of the direct influence of water on the specific regulating, provisioning and cultural services (Fig. 1). The need to investigate ecosystem services in a basin emerges from the awareness to manage water resources in an integrated and sustainable framework (Fohrer and Schmalz 2012).

Rural lowland basins provide a wide range of ecosystem services to human well-being, due to their various land uses, ranging from forests and pastures to intensive agriculture (Posthumus et al. 2010). The German Kielstau basin is an example of one of these agricultural-dominated ecosystem complexes which supply numerous provisioning, regulating and cultural ecosystem services and therefore several

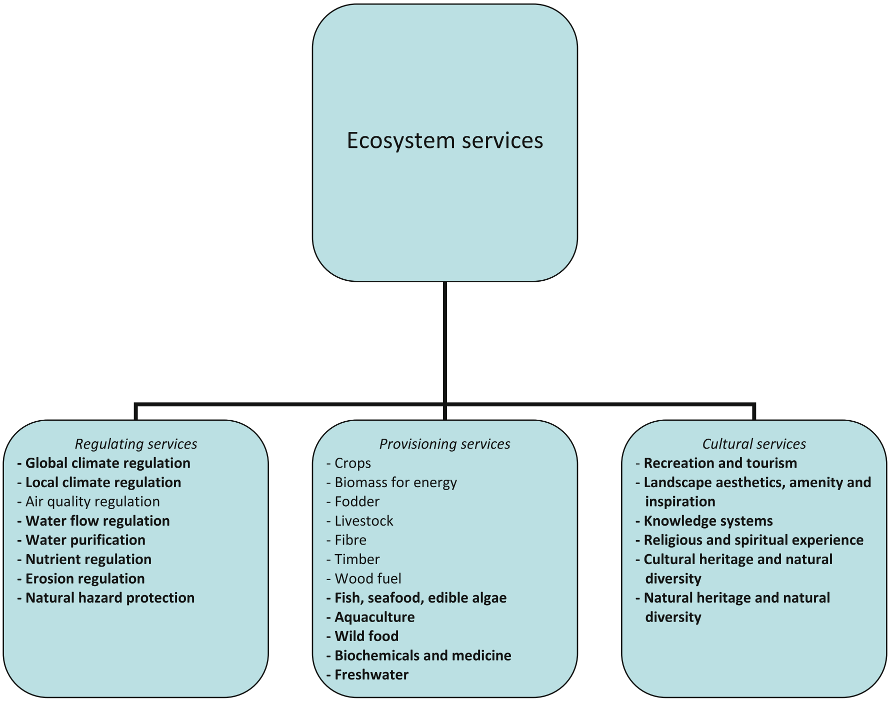

Fig. 1 Classification of ecosystem services and the directly water-related ecosystem services (*bold*) (Based on Kandziora et al. 2013)

benefits to human well-being. However, agricultural practices in the intensively used agroecosystems impact other ecosystem services, e.g. water purification, carbon sequestration and nutrient cycling. In return ecosystem services have high influences on the agricultural productivity and soil fertility (Dale and Polasky 2007). Consequently it is interesting to observe such trade-offs between different services and test whether they can be outcomes of model applications.

There are several methods for quantification, mapping and modelling ecosystem services, e.g. by the ecosystem services quantification models InVEST (Tallis and Polasky 2009) and ARIES (Villa et al. 2009). Therefore, models provide an important method to quantify selected ecosystem services and to visualise spatio-temporal variations as well as impacts of land use changes with maps based on management scenarios. Thus, the question arises how and in which quality model results can be used for ecosystem service assessments. Vigerstol and Aukeman (2011) derive a scheme to choose the appropriate hydrological or ecosystem service model based on data availability, expertise, scale and research question. Two focal questions have been defined based on the available expertise on the SWAT (Soil and Water Assessment Tool; Arnold et al. 1998) model in the rural lowland Kielstau basin (Schmalz and Fohrer 2010; Fohrer and Schmalz 2012) and the need to quantify water-related ecosystem services in the basin for decision-making and management.

The focal questions of this study are

- Is the ecohydrological SWAT model useful for ecosystem service assessments considering available output data and the use of this data to assess ecosystem services in rural basins?
- How can the SWAT output variables be translated into ecosystem services valuation schemes?

Therefore, the study basin Kielstau and its general supply of ecosystem services is introduced first, then the water-related ecosystem services with the focus on erosion regulation are analysed, and in the end the results are discussed and some conclusions for land use management and decision making are drawn.

2 The Kielstau Basin and Its Ecosystem Services

The Kielstau basin is located in northern Germany in the federal state of Schleswig-Holstein (Fig. 2). In 2010 the basin was included as a demonstration site in UNESCO's IHP Ecohydrology Programme (EHP) demonstration network (UNESCO 2011). In this framework it is used to demonstrate monitoring and modelling strategies for sustainable water resources management according to ecohydrology concepts (Fohrer and Schmalz 2012; Schmalz and Fohrer 2010).

The topography of this lowland area is flat but relatively uneven with elevations ranging from 78 m to 27 m a.m.s.l. (LVermA 2006). The prevailing soils are Haplic and Stagnic Luvisols (BGR 1999), while the river valleys are characterised by peat soils.

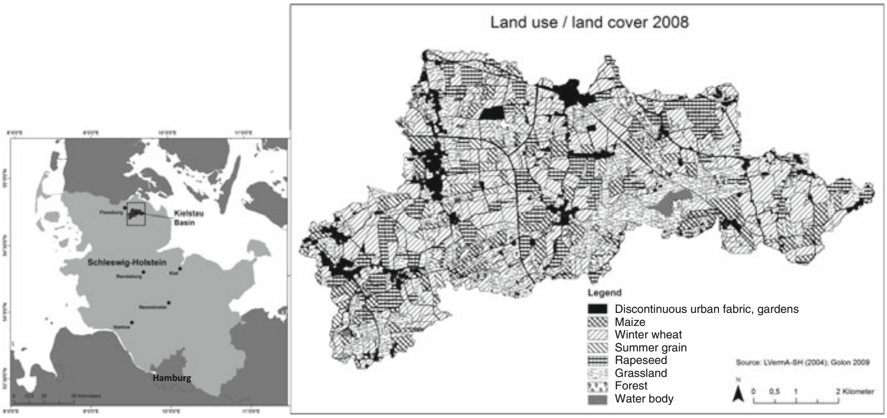

Fig. 2 Location of the Kielstau basin in Schleswig-Holstein (*left*) and land use/land cover map of the basin in the year 2008 (*right*, Golon 2009)

The mean annual precipitation and temperature are 919 mm and 8.2 °C, respectively (station Flensburg, period 1961–1990, DWD 2013). Land use in the rural basin is dominated by arable land and pasture (58 % and 17 %, respectively, in the year 2008; Golon 2009). There are only a few small villages and detached farms and no industrial facilities.

The Kielstau stream has a total length of 17 km (LAWAKÜ 1990) within its 50 km^2 drainage area. The stream is characterised as a gravel-dominated lowland stream. About 5 km downstream of its origin, the Kielstau flows through Lake Winderatt, which has a surface area of 0.24 km^2 and a mean depth of 1.2 m (LANU 1998). Downstream of Lake Winderatt two tributaries, the Moorau and the Hennebach, discharge into the Kielstau. At the gauging station Soltfeld, close to the catchment outlet, a mean discharge of 0.42 m^{3}s^{-1} and a mean water depth of 44 cm (1987–2005) were measured (LLUR 2013).

Many parts of the Kielstau river have been changed significantly and are now characterised by straightened and incised sections. According to the standard hydromorphological river survey method in Germany (LAWA 2000) the overall morphological state of the stream is assessed as ‘poor’ to ‘moderate’ (Olbert et al. 2006). The Kielstau water quality is not only influenced by the predominating agricultural land use in the basin (Schmalz et al. 2008a; Schmalz and Fohrer 2010), but also by six municipal wastewater treatment plants (Lam et al. 2010). The stream water quality is ranked as critically polluted (water quality class II-III; LLUR 2013). Using a phytoplankton index of biotic integrity, Wu et al. (2012) found that the ecological status varied seasonally. The general ecological status of the study region was ‘moderate’, which was not in accordance to the requirement (‘good’ status) of the European Water Framework Directive (EC 2000) by 2015.

The agriculturally dominated land use in the Kielstau basin influences the capacities and flows of all other ecosystem services. *Cultivated provisioning ecosystem services* in the case study area include crops, biomass for energy (e.g. maize), fodder and livestock (cp. Fig. 1 for all following ecosystem service description). Detailed land use mapping campaigns in 2008 and 2012 give information on the crop rotation and the changes in pasture and arable land shares (Golon 2009; INR 2012). The main crop rotation in this area is rapeseed – winter wheat – winter barley or silage maize in monoculture, sometimes interspersed with summer cereals (Kühling 2011). Climatic variations (e.g. wet summer periods) can influence the crop rotations significantly. The changes in shares of settlement, forest and grassland have been minimal between the years 2008 and 2012. Biomass cultivation for energy generation in biogas plants is a promoted renewable energy source by German law which leads to an increase in maize cultivation in Schleswig-Holstein (Statistikamt Nord 2011). Two biogas plants built in the years 2000 and 2010, respectively, in the basin provide heat and electricity for the local communities. A third one is planned. The cultivation of provisioning services influence *regulating ecosystem services* due to the application of fertilizers and pesticides (e.g. water purification and nutrient regulation) and the different phenologies of the individual crops (i.e. differences in erosion regulation during the year).

The basin and its surroundings are part of the touristic and recreational area of Schleswig-Holstein due to the position close to the Baltic Sea. The Kielstau is part of the Flora Fauna Habitat Protection Area (FFH-directive; EC 1992) and the Natura 2000 program. In the Eastern part of the study area, around Lake Winderatt, a nature protected area was established in 1989. Diverse terrestrial and aquatic habitats, high biodiversity and several hiking trails with information boards along with guided tours provide the opportunity for *cultural ecosystem services* like recreation, education and natural diversity.

The focal *regulating ecosystem services* include global climate regulation by carbon sequestration, local climate regulation, air quality regulation, water flow regulation, nutrient regulation, water purification and erosion regulation (Kandziora et al. 2013). Regulating services of the Kielstau area are primarily connected with the agricultural land use and the peculiarities of the local climate. Due to the shallow groundwater and high precipitation levels, high groundwater levels and saturated soils often disturb agricultural production. Even though 38 % of the agricultural area of the Kielstau basin is drained (Fohrer et al. 2007), flooding still occurs due to the elevation of the groundwater level (Schmalz et al. 2008b). Local and global climate regulation is based on the uptake of carbon in plants and soils, as well as on small-scale vegetation cover changes. On the global scale carbon sequestration by plants and soils may mitigate the constant temperature raise occurring during the last decades due to the accumulation of carbon in the atmosphere.

Being an intensely developed agricultural area, the Kielstau basin suffers from instream- and groundwater pollution by nutrients, mostly phosphorus and nitrogen. The main origin is based on the application of fertilizers on the agricultural fields, which after a chain of chemical reactions enter the aquifer or streams, deteriorating water quality and causing eutrophication (Lam et al. 2010; Schmalz et al. 2008a). However, some types of crops are characterised by the high leaching of nutrients from the fields, e.g. maize and rapeseed are characterised by high nutrient leaching potentials. Therefore, the ability of the different prevailing vegetation cover to regulate the nutrient balance can be considered as the regulating service nutrient regulation. Nutrient loss and nutrient regulation are directly connected to soil and sediment losses, as some nutrients, especially phosphorus, are adsorbed to soil particles. The abilities of the different vegetation covers to regulate soil erosion are varying by plant species and percentage of vegetation cover (Kandziora et al. 2013). The supply of this service is not only dependent on the properties of the vegetation cover but also on the characteristics of the growth period. A perennial vegetation cover supplies a higher erosion regulation than vegetation that is covering the soil only during special parts of the year (de Vries et al. 2010; Dale and Polasky 2007). Agricultural practices, e.g. undersown crops or catch crops, influence the erosion regulation capacity as well. Consequently, erosion regulation (i.e. year-round vegetation cover) is interacting directly with other ecosystem services positively or negatively (Table 1). Therefore, the focus of this SWAT modelling study is on the resulting sediment yields for mapping the temporal and spatial variations of the capacity of different land use/land cover types to provide erosion regulation. The supply capacity, which could also be defined as the ecosystem service flow, refers to the generation of the actually used ecosystem service and not to the potential of an ecosystem service (Burkhard et al. 2012).

Table 1 Direct positive (support) ↗ and negative (competition) ↘ interactions between erosion regulation and selected ecosystem services in the Kielstau basin (Based on Kandziora et al. 2013)

Erosion regulation	
	Regulating ecosystem services
↗	Global climate regulation
↗	Local climate regulation
	Air quality regulation
↗	Water flow regulation
↗	Water purification
↗	Nutrient regulation
	Erosion regulation
↗	Natural hazard regulation
	Provisioning ecosystem services
↘	Crops
↘	Biomass for energy
↘	Fodder
↘	Livestock
↘	Fibre
↗	Timber
↗	Wood fuel
↗	Wild food
↗	Freshwater
	Cultural ecosystem services
↗	Recreation and tourism
↗	Landscape aesthetics, amenity and inspiration
	Knowledge systems
↗	Cultural heritage and cultural diversity
↗	Natural heritage and natural diversity

3 Regulating Ecosystem Services: The Example of Erosion Regulation

3.1 Methods

3.1.1 SWAT Model and Outputs

The valuation of ecosystem services is a complex process, which requires holistic understanding of natural ecosystem processes and appropriate indicators for quantification. Models are suitable tools to understand and visualise these processes, especially in the framework of management scenarios incorporating different time steps for the assessment.

The high potential of the SWAT model (Arnold et al. 1998) for the valuation of ecosystem services in a river basin was emphasised in the recent literature (Vigerstol and Aukema 2011). The main advantage for ecosystem service estimation with the SWAT model is that it allows taking into account multiple characteristics of a specific basin and its streams over time. These characteristics are related to climate, geomorphology, soil and vegetation covers and management practices in the basin along with parameters governing water and sediment movement in the channel. The necessary input data for the model include a digital elevation model, soil and land use maps and their corresponding data bases, climate data, and loadings from wastewater treatment plants. The Kielstau basin was subdivided into 17 subbasins (Kühling 2011) and further into HRUs (HRU; hydrological response units) with the same soil, land use and slope combination. All necessary topographic, soil, land use information, agricultural management practices and climate data were implemented (Kühling 2011). The model can simulate the water balance, nutrients and pesticides, field erosion, plant growth cycles, management practices and water bodies on a daily time step for continuous simulations over long time periods (Neitsch et al. 2011).

The simulation of the hydrological processes of a watershed is separated into the land phase and into the water or routing phase (Neitsch et al. 2011). The simulation of the land phase of the hydrologic cycle is based on the water balance equation, which is calculated separately for each HRU. The hydrologic cycle is driven by climatic variables. Processes taken into account include interception, evapotranspiration, infiltration, percolation, recharge as well as surface runoff, lateral and return flow. The land phase of the hydrologic cycle controls the amount of water, sediment, nutrient and pesticide loadings to the main channel in each subbasin. Runoff generated in the HRUs is summed up to constitute the amount of water reaching the main channel in each subbasin. The water or routing phase of the hydrologic cycle is defined as the movement of water, sediments, etc. through the channel network of the watershed to the outlet (Neitsch et al. 2011). The routing of runoff in the channel is specified in the SWAT model by using either the variable storage coefficient method (Williams 1969) or the Muskingum routing method (Overton 1966).

Sediment yield is estimated for each HRU using the empirical Modified Universal Soil Loss Equation (MUSLE) (Williams 1975) and refers to the field erosion.

Erosion types that can be depicted with the MUSLE are sheet and rill erosion. Sediment routing in the channel is controlled either by deposition or by degradation. These channel erodibility processes are considered in the model, but the concept of SWAT using a spatial representation through subbasins is not sufficient for obtaining differentiated instream results along a stream channel. Furthermore, SWAT currently is not able to depict sediment input from artificial tile drains.

Erosion processes modelled in SWAT include detachment, transport and deposition of soil particles by the erosive forces of raindrops and surface runoff. In this study we focus on the influence of the land cover on the erosion processes which is included in the "cover and management factor" (USLE_C). This factor is defined as the ratio of soil loss from land cropped under specified conditions to the corresponding loss from clean-tilled, continuous fallow. The plant canopy reduces the erosive power of rain drops by their interception, because the velocity of the drop falling from the plant is smaller than the rain drop falling directly to the soil. Canopy's height and density will determine reduction of the rainfall energy to the soil surface. The residue cover on the soil is more effective than a plant cover, because residue intercepts falling rain drops so near to the surface that they regain no fall velocity. Residues also obstruct surface runoff, decreasing its velocity and transport capacity. SWAT updates the USLE cover and management parameter daily, because the plant cover varies during the growth cycle of the plants. The USLE_C factor is based on the minimum value for the cover and management factor for the land cover and the amount of residue on the soil surface. USLE_C is specified for each type of crop in the SWAT crop database (Neitsch et al. 2011).

In this case study, indicators for the quantification of the regulating ecosystem service 'erosion regulation' were selected from available SWAT output data sets (Table 2). The SWAT variable "SYLD" is the sediment yield which is transported from the HRU into the reach during the individual time steps in metric tons/ha (Neitsch et al. 2010).

Table 2 Indicators based on SWAT parameters used for the quantification of different regulating services

Regulating service	Potential indicators for quantification
Global climate regulation	BIOM (biomass) and YLD (harvested yield) calculated to the final biomass in the watershed (BIOM_end)
Local climate regulation	SW_INIT (initial soil water content) and SW_END (end soil water content)
Water flow regulation	GW_Q (groundwater discharge into reach) and GW_RCHG (amount of water entering both aquifers)
Erosion regulation	USLE (soil loss) and SYLD (sediment yield entering the river stream)
Nutrient regulation	SOL_P (soluble mineral forms of phosphorus transported by surface runoff), P_GW (soluble phosphorus transported by groundwater flow), ORG_P (organic phosphorus transported with sediment into the reach)

3.1.2 Valuation and Mapping of Erosion Regulation in the Kielstau Basin

The valuation method which was applied here, is based on land use/land cover data and the contribution of each of those classes to the provision of a specific ecosystem service (here erosion regulation) which was first introduced by Burkhard et al. (2009). This approach was applied to the eight land use/land cover classes in the Kielstau basin. Indicators for the quantification of erosion regulations are based on the SWAT outputs for several modelled vegetation types.

The average annual values of the SWAT output variable "SYLD" were calculated for the following eight land use/land cover classes: discontinuous urban fabric and gardens, maize, forest, rapeseed, summer cereals, winter barley, winter wheat, pastures and water bodies. The erosion regulation, which highly depends on vegetation cover, was calculated annually for the model simulation period 2003–2009. The logarithmic valuation scale from 0 to 5 was applied for the modelled results, with 5 representing the land use/land cover with the maximum capacity to provide erosion regulation and 0 the land use/land cover with the least capacity. The value of 2.1959 t/ha was the maximum of the modelled results, therefore it receives the lowest capacity class of 0 (Table 3) (cp. Burkhard et al. 2009, 2012). The water bodies and discontinuous urban fabric areas were assumed to have a value of 5 as no erosion takes place, meaning a very high erosion regulation capacity. The logarithmic scale was applied due to the wide range of values, which were then translated into semantic descriptions for aggregation (see Table 3).

For the visualisation of the model results, maps for the years 2007 and 2009 were created illustrating the respective changes of the supply capacities of erosion regulation (Figs. 3 and 4). The assigned 0–5 values to the modelled SYLD outputs were given to each respective land use/land cover in a Geographic Information System (GIS; ArcGIS 10). The assumed crop rotation of rapeseed – winter wheat – winter barley and maize – maize/summer cereals – maize was applied based on the mapped crops in 2008 (Fig. 2). The two maps in Figs. 3 and 4 show the spatial and temporal variations in crops, their shares and the supply capacity of the regulating ecosystem service erosion regulation for the years 2007 and 2009.

Table 3 Relative valuation scale and biophysical model results for the quantification of erosion regulation based on SWAT model results (2003–2009)

Erosion regulation capacity	SYLD (t/ha)
0 = no relevant capacity	>= 2.1959
1 = low relevant capacity	0.9478–2.1958
2 = relevant capacity	0.4091–0.9477
3 = medium relevant capacity	0.1766–0.4090
4 = high relevant capacity	0.0762–0.1765
5 = very high relevant capacity	<0.0761

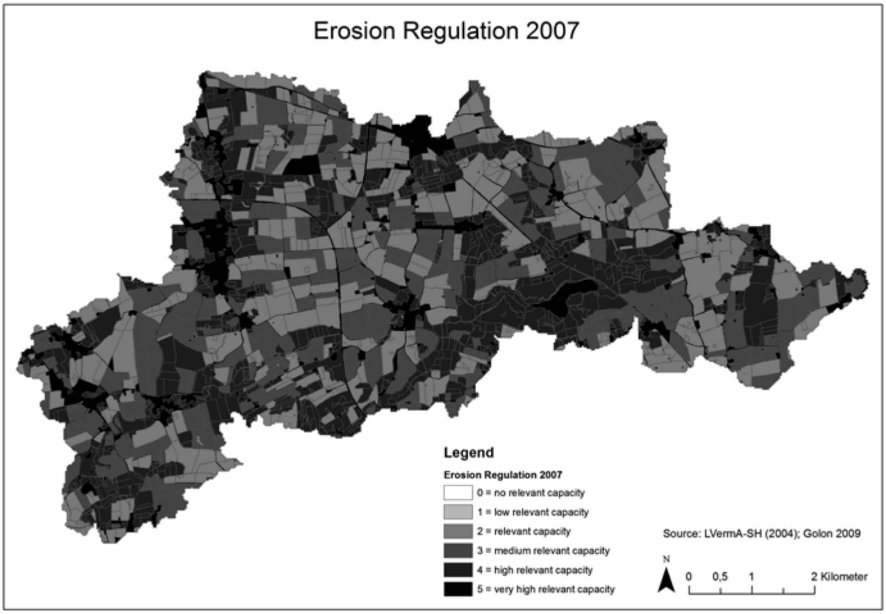

Fig. 3 Erosion regulation capacity in the Kielstau basin in 2007

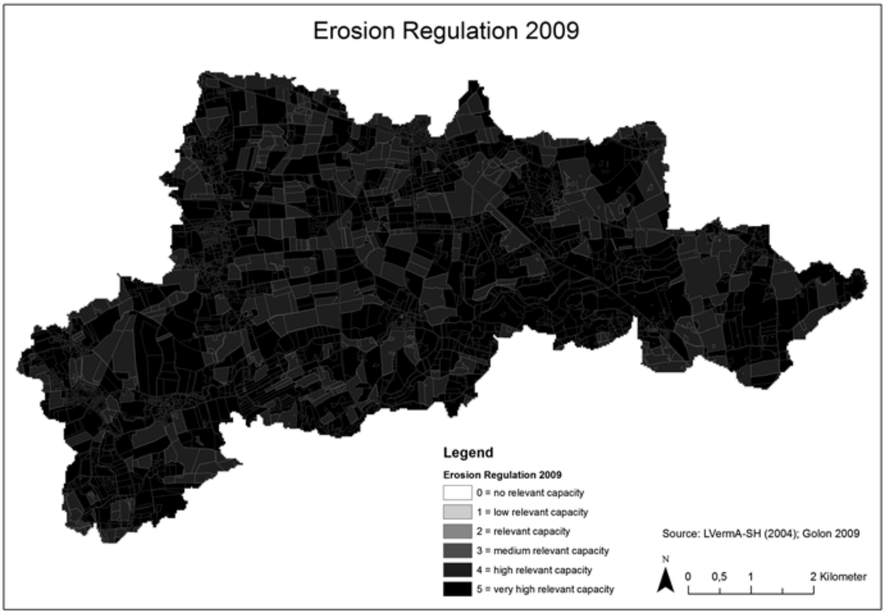

Fig. 4 Erosion regulation capacity in the Kielstau basin in 2009

3.2 Results

Based on the results of the SWAT model, it can be seen that there is a clear dependence between land use and soil erosion in the Kielstau basin. All SWAT model outputs are available as daily data. However, the daily data was aggregated to annual values per land use/land cover because of the practicability to generate maps for communicating the information to decision makers and land use managers. For this study, 2 years were selected to visualise the variability in the model results. The modelled sediment loss varies between 0.934 t ha^{-1} in the year 2007 for rapeseed and 0.0404 t ha^{-1} for forest areas in 2009. It is noticeable that the modelled sediment yields are lower in 2009 than in 2007 and therefore the corresponding valuation scale is showing higher modelled capacities for erosion regulation in 2009 than in 2007 (Table 4, Figs. 3 and 4). Water bodies and discontinuous urban fabric, gardens are understood to have the highest capacity (value = 5) to prevent soil loss due to the sealing of soils in settlements and the sediment trap function of lakes. The modelled results assume that all prevailing land use/land cover classes contribute to the prevention of erosion, therefore no land use/land cover class receives the value 0 (= no relevant capacity). Pastures and forest areas feature the second best capacities in both years due to the permanent covering of the soils. In 2007 winter wheat, winter barley and maize show higher capacities for erosion regulation than rapeseed.

Figure 3 illustrates the more heterogeneous spatial distribution of the different capacities for erosion regulation, whereas Fig. 4 clearly demonstrates the high modelled results for the erosion regulation capacity in 2009. Due to the assumed crop rotation, the share of maize is higher in 2007 and 2009 than in 2008 as no summer cereals are modelled.

Table 4 Logarithmic valuation scale for the erosion regulation ecosystem service relating the quantification data (output data from SWAT model and expert valuation) to the relative 0–5 valuation scale

Land use/land cover 2008 (based on Golon 2009)	SWAT model results (t/ha) 2007	Valuation scale	SWAT model results (t/ha) 2009	Valuation scale
Maize	0.3212	3	0.0624	5
Forest	0.1451	4	0.0404	5
Rapeseed	0.934	2	0.0897	4
Winter barely	0.2091	3	0.0906	4
Winter pasture	0.1306	4	0.0587	5
Winter wheat	0.3186	3	0.0684	5
Lake/water bodies	–	5	–	5
Urban fabric, gardens	–	5	–	5

4 Discussion

The SWAT model results are a reasonable base for the assessment of water-related regulating ecosystem services in the Kielstau basin in Northern Germany. In this study only erosion regulation was depicted in more detail but the future incorporation of SWAT models to the assessment of further water-related ecosystem services will highlight the interactions of the individual ecosystem services. One advantage of this kind of valuation derives from the distinction of agricultural land use types into different crops or crop rotations.

The validity of the analyses certainly is dependent on the accuracy of the model. The performance of the base model (Kühling 2011; daily time step) for the flow can be classified (Moriasi et al. 2007; based on monthly values) as very good both during the calibration (hydrological years 2004–2008) and the validation (hydrological year 2009) period, expressed by a Nash-Sutcliffe efficiency (NSE) of 0.81. The achieved NSE of 0.45 during the sediment calibration (hydrological year 2008) is classified to be unsatisfactory after Moriasi et al. (2007) – in terms of monthly data. Green and van Griensven (2008) assume, however, an acceptable agreement already at a NSE > 0.4. For the sediment validation period (hydrological year 2009) only a NSE of 0.27 was achieved. The basic dynamics is represented well by the model, however, the peaks are significantly underestimated (Kühling 2011).

River banks, drainage systems and agricultural fields are the three main sources of sediment in lowland regions. Results of Kiesel et al. (2009) from the Kielstau basin show that 15 % of the sediment input into the river comes from agricultural drains, 71 % from river banks and 14 % from adjacent fields. Because SWAT currently is not able to depict sediment input from tile drains or representing the bank erosion in an appropriate resolution, the simulated daily sediment input has to be assessed carefully. Particularly in a lowland basin as represented by the Kielstau basin, underestimations are caused by the drain depiction.

Another problem results from the low simulated sediment yield values and the unsatisfactory model performance concerning sediment calibration and validation. Low sediment entries are typical for lowland areas but they cause an uncertainty to depict low sediment yields for a larger subbasin or HRU area.

Referring to phosphorus, the SWAT model used in this study has a relatively poor model performance (NSE = 0.24; Kühling 2011). For nitrogen, the model was not calibrated. Therefore, nutrient regulation as another regulating service can indeed be calculated, but will not be discussed due to the unknown accuracy. The derivation of the global and local climate regulation ecosystem services is very complex. There are not enough SWAT variables available to derive enough data for quantification of these regulating services. In addition, the carbon cycle is implemented simplified as a one-pool SOM (soil organic matter) sub-model and there are no field data for validation. The water flow regulation corresponds to the very complex relationships between the different indicators (Table 2). It is very difficult to translate them into the 0–5 valuation scale because of the needed weighting of the different indicators.

The limited available time series do not allow any other choice for the calibration and validation periods. As the two hydrological years 2008 and 2009 on the one hand differ significantly from each other and on the other hand differ significantly from the average of all the simulated years, the validation of the data should be regarded critically. Another uncertainty is based on the 1-year periods, respectively. Since the land use management is based on 3-year crop rotations, minimal deviations are possible due to a slightly different distribution of area shares (Kühling 2011).

The highest ability to regulate erosion by vegetation cover and the respective modelled management practices during the whole simulation period was of the forests followed by winter pastures, winter wheat, winter barley and maize and terminated with rapeseed. De Vries et al. (2010) also rank an increase in erosion from winter rapeseed, winter wheat, sugar beet to maize due to phenological differences. Maize covers the soil late in the year and due to its late harvest, the fields remain fallow until the next cultivation period in the following spring (Kühling 2011). Previous studies confirm that maize is considered to have a higher erosion potential than grain crops which are mostly considered being less prone to erosion (Fohrer and Fiener 2013; Fiener and Auerswald 2007). However, slopes are an issue to be considered explicitly in assessing erosion. Results achieved in the Swiss Midlands show that winter wheat has the highest erosion rate and is then followed by maize and rapeseed (Prasuhn 2012). In contrast, the modelled erosion regulation capacities are higher for maize than for rapeseed. It must be taken into consideration that the annual SWAT model time step underestimates the soil loss and sediment transport from the maize fields due to the annual aggregation. This is due to the specific characteristics of the maize vegetation period, which as it was described above, reaches maturity in August and is not able to provide the ecosystem service at full capacity before that. The created erosion regulation maps did not distinguish these periods. Nevertheless SWAT has the opportunity to give daily output, for further studies seasonal maps should be discussed.

The year 2007 with its high precipitation values showed a significant change in the soil loss from the investigated land uses. Normally in this area, erosion is not high due to the flat topography that causes low surface runoff values (Lam et al. 2010; Kiesel et al. 2010). A study conducted by Erhard et al. (2003) in the assessment of the actual soil erosion risk in Germany, concluded an average value of soil loss of 0–0.5 t ha^{-1} in Schleswig-Flensburg, where the Kielstau basin is located. The results of this study are in accordance to this data, showing deviations among the different land use classes. However, the year 2007 can be characterised by higher precipitation values (DWD 2010, Table 5) and thus showed extremely high sediment loss compared with the other years. During the years with the lower precipitation values all the land use/land cover classes have similar capacities to provide the erosion regulation ecosystem service, but they differ significantly during the years with higher rainfall values. Achieved results are in good correlation with the data in international literature. The fact, that the forest provides the highest erosion regulation was confirmed by other studies (e.g. Burkhard et al. 2009; Koschke et al. 2012), which are based on expert evaluations.

Table 5 Precipitation (annual sums) in the years 2003–2009 (Station Glücksburg-Meierwik, DWD 2010)

Year	Sum precipitation [mm]
2003	587.5
2004	884.3
2005	765.2
2006	765.8
2007	1043.1
2008	828.1
2009	845.4

The results of the modelled erosion regulation ecosystem service can be further improved by the calibration of the vegetation parameters such as growth development phase and biomass, density of the vegetation on the fields and characteristics of the root system distribution. Using the SWAT model for this study, some parameters in the crop database were already changed to adapt the plant growth to the conditions of Northern Germany.

Furthermore, it must be considered that only selected crops were modelled. Other crops, such as sugar beets and potatoes, which are assumed to be prone to erosion (de Vries et al. 2010), are found in the Kielstau basin as well. Additionally, the implemented crop rotations can be seen as representatives for the study area but are not exactly representing each occurring crop rotation in the respective year. Only two of these crop rotations were selected for this study within the catchment.

Furthermore, the results of soil loss must be coupled with further investigations on the impacts of other ecosystem services and the ecological integrity of the basin, i.e. because soil loss and sediment input into streams can have severe impacts on habitats and biodiversity (Kiesel et al. 2009, 2013).

5 Conclusion

Combining modelling results with valuation methods and stakeholder participation can obtain sophisticated information for the decision-making process, which in the end can lead to a more sustainable environmental management of basins globally (Quintero et al. 2009). This study has demonstrated that also the results of hydrological models, which were not developed for the purpose of assessing ecosystem services, can be extremely helpful to quantify and assess the provision of selected ecosystem services.

The ecohydrological SWAT model is used for several research and management questions as it is a continuous model for long-term modelling which is based on annual, monthly and daily time steps. Discharge along with sediment, nutrients and pesticides can be modelled in a basin. In this study, SWAT results for the studied Northern German Kielstau basin were calculated and cover a multitude of variables on a daily time step. Some of these variables can be selected and be used as indicators for the assessment of regulating services. One detailed example was given by providing and analysing the sediment yield as an indicator for the regulating ecosystem service ‘erosion regulation’. The SWAT output was directly translated into a 0–5 valuation scale for the different land use/land cover classes. The SWAT model results reveal the temporal changes in erosion regulation due to crop rotation and different precipitation patterns over the years, which includes important information in the assessment of ecosystem services and the formulation of management actions.

Although the SWAT model calculates daily output, the final user, such as environmental managers and decision makers ask for aggregated, mainly annual scaled data for a simplified analysis. This is essential for valuations based on several ecosystem services. Therefore, the temporal demand of SWAT users and decision makers might be different. However, this study has demonstrated in a first attempt for the Kielstau basin that SWAT results can be used to quantify and assess regulating ecosystem services.

Nevertheless, there is further potential to improve this valuation procedure and the quantification and modelling of all ecosystem services within a basin. The need for a more detailed investigation on annual variations in erosion regulation is emphasised by the results from this study. To find a compromise between daily output and annual maps could be the provision of seasonal maps. There seasonal changes in land use as well as climatic characteristics and the temporal sediment variability can be considered even better.

Furthermore it is highlighted by this modelling exercise that there is additional need to integrate stakeholders for the full assessment of all ecosystem services and for land use change scenarios.

References

Arnold JG, Srinivasan R, Muttiah RS, Williams JR (1998) Large area hydrologic modelling and assessment part I: model development. J Am Water Resour Assoc 34(1):73–89

BGR Bundesanstalt für Geowissenschaften und Rohstoffe (1999) Bodenübersichtskarte 1:200000 CC.1518. Flensburg, Hannover, Germany

Burkhard B, Kroll F, Müller F, Windhorst W (2009) Landscapes’ capacities to provide ecosystem services – a concept for land-cover based assessments. Landscape Online 15:1–22

Burkhard B, Kroll F, Nedkov S, Müller F (2012) Mapping supply, demand and budgets of ecosystem services. Ecol Indic 21:17–29

Daily GC (1997) Nature’s services: societal dependence on natural ecosystems. Island Press, Washington, DC, 392 p

Dale VH, Polasky S (2007) Measures of the effects of agricultural practices on ecosystem services. Ecol Econ 64:286–296

De Vries SC, van de Ven GWJ, van Ittersum MK, Giller KE (2010) Resource use efficiency and environmental performance of nine major biofuel crops, processed by first-generation conversion techniques. Biomass Bioenergy 34:588–601

Doyle MW, Yates AJ (2010) Stream ecosystem service markets under no-net-loss regulation. Ecol Econ 69:820–827

DWD Deutscher Wetterdienst (2010) Precipitation, station Glücksburg-Meierwik, period 2003–2009. German Weather Service, Offenbach a.M

DWD Deutscher Wetterdienst (2013) Mean annual precipitation and temperature, station Flensburg, period 1961–1990. German Weather Service, Offenbach a.M., Germany. http://www.dwd.de/mittelwerte. Last access 6 Feb 2013

EC (1992) Council directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora. Off J Eur Communities L206/7

EC (2000) Directive 2000/60/EC of the European Parliament and of the Council. Water Framework Directive, European Council

Erhard M, Boeken H, Glante F (2003) The assessment of the actual erosion risk in Germany, based on CORINE Land-cover and statistical data from the main representative survey of land use. OECD expert meeting on soil erosion and soil biodiversity indicators, Rome, 25–28 Mar 2003

Fiener P, Auerswald K (2007) Rotation effects of potato, maize, and winter wheat on soil erosion by water. Soil Sci Soc Am J 71:1919–1925

Fohrer N, Fiener P (2013) Schutz vor Bodenabtrag durch Wasser und Überflutung. In: Blume HP, Felix-Henningsen P, Fischer WR, Frede HG, Horn R, Stahr K (eds) Handbuch der Bodenkunde. Ecomed, Landsberg (in German)

Fohrer N, Schmalz B (2012) Das UNESCO Ökohydrologie-Referenzprojekt Kielstau-Einzugsgebiet – nachhaltiges Wasserressourcenmanagement und Ausbildung im ländlichen Raum. Hydrol Wasserbewirtsch 56(4):160–168 (in German)

Fohrer N, Schmalz B, Tavares F, Golon J (2007) Modelling the landscape water balance of meso-scale lowland catchments considering agricultural drainage systems. Hydrol Wasserbewirtsch 51:164–169 (in German)

Golon J (2009) Environmental effects of varied energy crop cultivation scenarios on a lowland catchment in Northern Germany – a SWAT approach. Master thesis Kiel University, Germany, 92 p

Green C, van Griensven A (2008) Autocalibration in hydrologic modeling: using SWAT2005 in small-scale watersheds. Environ Model Softw 23:422–434

INR (2012) Land use mapping 2012 of the Kielstau catchment. Department of Hydrology and Water Resources Management, Institute for Natural Resource Conservation, Christian-Albrechts-University Kiel

Kandziora M, Burkhard B, Müller F (2013) Interactions of ecosystem properties, ecosystem integrity and ecosystem service indicators – a theoretical matrix exercise. Ecol Indic 28:54–78

Kiesel J, Schmalz B, Fohrer N (2009) SEPAL – a simple GIS-based tool to estimate sediment pathways in lowland catchments. Adv Geosci 21:25–32

Kiesel J, Fohrer N, Schmalz B, White MJ (2010) Incorporating landscape depressions and tile drainages of lowland catchments into spatially distributed hydrologic modelling. Hydrol Process 24:1472–1486

Kiesel J, Schmalz B, Brown GL, Fohrer N (2013) Application of a hydrological-hydraulic modelling cascade in lowlands for investigating water and sediment fluxes in the catchment, channel and reach. J Hydrol Hydromech 61(4):334–346

Koschke L, Fürst C, Frank S, Makeschin F (2012) A multi-criteria approach for an integrated land-cover-based assessment of ecosystem services provision to support landscape planning. Ecol Indic 21:54–66

Kühling I (2011) Modellierung und räumliche Analyse der Phosphateintragspfade im Einzugsgebiet eines norddeutschen Tieflandbaches. Master thesis Kiel University, Germany, 106 p. (in German)

Lam QD, Schmalz B, Fohrer N (2010) Modelling point and diffuse source pollution of nitrate in a rural lowland catchment using the SWAT model. Agric Water Manag 97:317–325

LANU Landesamt für Natur und Umwelt des Landes Schleswig-Holstein (1998) Seenkurzprogramm, 165 p. (in German)

LAWA Länderarbeitsgemeinschaft Wasser (2000) Gewässerstrukturgütekartierung in der Bundesrepublik Deutschland – Verfahren für kleine und mittelgroße Fließgewässer. Kulturbuch Verlag. ISBN-13: 978–3889612335, pp 1–164 (in German)

LAWAKÜ Landesamt für Wasserhaushalt und Küsten Schleswig-Holstein (1990) Gütelängsschnitt Kielstau. A 91. Kiel (in German)

LLUR Landesamt für Landwirtschaft, Umwelt und ländliche Räume des Landes Schleswig-Holstein (2013) Information about gauge Soltfeld/Kielstau and water quality classes 2002. Agricultural and environmental atlas. http://www.umweltdaten.landsh.de/atlas/script/index.php. Last Access 6 Feb 2013

LVermA Landesvermessungsamt Schleswig-Holstein (2006) DEM 25 m grid size. Land survey office Schleswig-Holstein, Kiel

MA Millennium Ecosystem Assessment (2005) Ecosystems and human well-being: synthesis. Island Press, Washington, DC

Moriasi DN, Arnold JG, Van Liew MW, Bingner RL, Harmel RD, Veith TL (2007) Model evaluation guidelines for systematic quantification of accuracy in watershed simulations. Trans ASABE 50:885–900

Müller F (2005) Indicating ecosystem and landscape organisation. Ecol Indic 5:280–294

Neitsch SL, Arnold JG, Kiniry JR, Srinivasan R, Williams JR (2010) Soil and water assessment tool – Input/Output file documentation – Version 2009. – Grassland, Soil and Water Research Laboratory – Agricultural Research Service. Blackland Research Center – Texas AgriLife Research. Texas Water Resources Institute TR-365

Neitsch SL, Arnold JG, Kiniry JR, Williams JR (2011) Soil and water assessment tool – theoretical documentation – Version 2009. – Grassland, Soil and Water Research Laboratory – Agricultural Research Service. Blackland Research Center – Texas AgriLife Research. Texas Water Resources Institute TR-406

Ojea E, Martin-Ortega J, Chiabai A (2012) Defining and classifying ecosystem services for economic evaluation: the case of forest water services. Environ Sci Pol 19–20:1–15

Olbert C, Fell F, Ahrens H, Beurton D, Wetzel H, Hnida D (2006) Structural river mapping 2005 in the Eider watershed. Informus GmbH, Water- and Soil Association, Schleswig-Holstein

Overton DE (1966) Muskingum flood routing of upland streamflow. J Hydrol 4:185–200

Posthumus H, Rouquette JR, Morris J, Gowing DJ (2010) A framework for the assessment of ecosystem goods and services; a case study on lowland floodlplains in England. Ecol Econ 69:1510–1523

Prasuhn V (2012) On-farm effects of tillage and crops on soil erosion measured over 10 years in Switzerland. Soil Tillage Res 120:137–146

Quintero M, Wunder S, Estrada RD (2009) For services rendered? Modeling hydrology and livelihoods in Andean payments for environmental services schemes. For Ecol Manag 258:1871–1880

Schmalz B, Fohrer N (2010) Ecohydrological research in the German lowland catchment Kielstau. IAHS Publ 336:115–120

Schmalz B, Bieger K, Fohrer N (2008a) A method to assess instream water quality – the role of nitrogen entries in a North German rural lowland catchment. Adv Geosci 18:37–41

Schmalz B, Springer P, Fohrer N (2008b) Interactions between near-surface groundwater and surface water in a drained riparian wetland. IAHS Publ 321:21–29

Smith M, de Groot D, Bergkamp G (2006) Pay – establishing payments for watershed services. IUCN, Gland

Statistikamt Nord (2011) Die Bodennutzung in Schleswig-Holstein 2011. Statistisches Amt für Hamburg und Schleswig-Holstein

Tallis H, Polasky S (2009) Mapping and valuing ecosystem services as an approach for conservation and natural-resource management. Ann N Y Acad Sci 1162:265–283

Tognetti SS, Mendoza G, Aylward B, Southgate D, Garcia L (2004) A knowledge and assessment guide to support the development of payment arrangements for watershed ecosystem services. Working paper. Environment Department. World Bank, Washington, DC

UNESCO (2011) Ecohydrology for sustainability. International Hydrological Programme, Division of Water Sciences. http://unesdoc.unesco.org/images/0021/002108/210826e.pdf. Last access 6 Feb 2013

Vigerstol KL, Aukema JE (2011) A comparison of tools for modeling freshwater ecosystem services. J Environ Manag 92:2403–2409

Villa F, Ceroni M, Bagstad K, Johnson G, Krivov S (2009) ARIES (Artificial Intelligence for Ecosystem Services): a new tool for ecosystem services assessment, planning and evaluation. In: 11th annual BIOECON conference on economic instruments to enhance the conservation and sustainable use of biodiversity

Williams JR (1969) Flood routing with variable travel time or variable storage coefficients. Trans ASAE 12(1):100–103

Williams JR (1975) Sediment routing for agricultural watersheds. Water Resour Bull 11:965–974

Wu N, Schmalz B, Fohrer N (2012) Development and testing of a phytoplankton index of biotic integrity (P-IBI) for a German lowland river. Ecol Indic 13:158–167

Aquatic Ecosystem Services and Management in East Africa: The Tanzania Case

Lulu T. Kaaya and George V. Lugomela

Abstract Diverse aquatic ecosystems in Tanzania provide economically important ecosystem services. The rich supply of these services is under threat. Projections show critical water scarcity in the country by the year 2050. Demography, excessive withdrawals, land use changes, exotic species invasions and climate change that result in loss of perennial flows, eutrophication, sedimentation, and algal blooms are among the major drivers of aquatic ecosystem changes in Tanzania. Water resources uses and their management in Tanzania are mainly determined by the national macroeconomics and policies. In this review, Great Ruaha River (GRR) and Lake Victoria Basin (LVB) are used as case examples for demonstrating status, trends and drivers of ecosystem changes, and their management options in Tanzania through government and donor efforts. As a way forward, in the new Tanzania National Water Policy (NAWAPO) of 2002, Integrated Water Resource Management (IWRM) approaches as tools to ensure ecosystem protection and stakeholder's participation have been adopted. Water for environment is given a second priority in water allocation after basic human needs. The Integrated Water Resources management and Development (IWRMD) plans currently being developed will form legal basis in management of water in an environmentally and ecosystem responsible manner. Through the IWRMD approved plans, drastic actions can be legally taken to protect and/or restore important ecosystem services in hotspot areas like the GRR.

Keywords Ecosystem services • Water resources • IWRM • Lake Victoria • Rufiji basin

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1 Introduction

Ecosystem services represent the benefits that humans obtain from ecosystems and human well-being is fundamentally dependent upon these services (Millennium Ecosystem Assessment 2005). Of all the resources required for sustaining ecosystems and the services they provide for human health and wellbeing, water is arguably the most important. Water plays a crucial role in the delivery of many ecosystem services, including provisioning services such as biomass and crop production, as well as cultural, regulatory and supporting services. The earth's ecosystems would not function without adequate supplies of water of suitable quality. However, every time people access, develop, transport or utilize water resources, they leave an impact that may degrade the service provided by the river, lake, wetland or groundwater aquifer that supplied the water in the first place. Water security, therefore, depends on how well we can address disturbances to these water systems, which can, in turn, affect their services.

Because the notion of an ecosystem represents a useful framework to consider the many linkages between humans and their environment, a so-called 'ecosystem approach' and/or the concept of Integrated Water Resources Management (IWRM) have been advocated by many organizations and individuals as the means of addressing the interrelations between water, land, air, and all living organisms, and encompassing ecosystems and their services. Water management therefore translates into managing ecosystem services. Many environmental management options exist to tackle sustainable ecosystem functioning and services. The major ecosystem service management options and goals include: maintaining environmental flows, instituting pollution control measures, utilizing ecohydrological measures, habitat rehabilitation, conjunctive use of surface and groundwater, watershed management, water demand management, and valuation and payment for ecosystem goods and services.

Like many governments and agencies around the world, Tanzanian government is struggling to effectively implement IWRM for improvement of its aquatic ecosystems. This difficulty is attributable to the many complex scientific, socioeconomic and financial elements to be simultaneously considered in this chapter. This chapter provides an overview of the common ecosystem services provided at various watershed scales in Tanzania; local pressures on and drivers of ecosystem services in watersheds, and the management options employed to achieve sustainable watershed ecohydrology and delivery of ecosystem services. Case examples will be considered from the Lake Victoria Basin which contains the largest lake in Africa and the Great Ruaha River which forms a major tributary of the Rufiji River (largest river basin in the country) (Fig. 1).

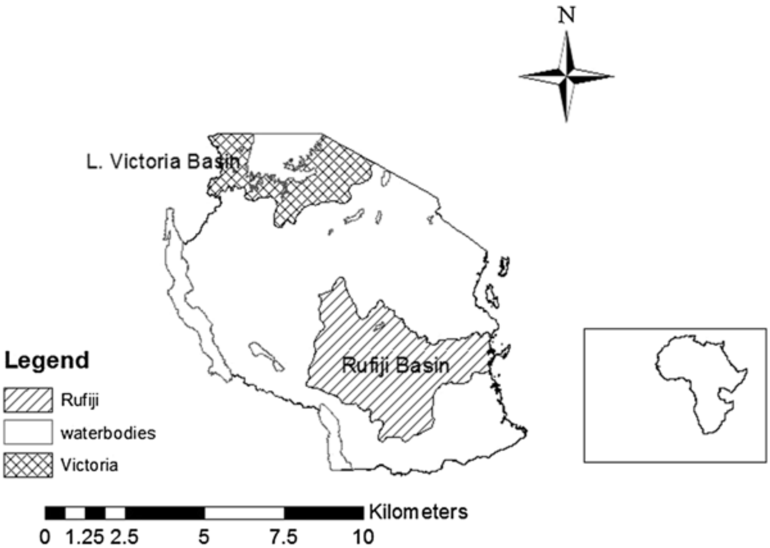

Fig. 1 Map of Tanzania showing location of Lake Victoria and Rufiji River basins

2 Status and Trends of Aquatic Ecosystem Services in Tanzania

Tanzania receives about 82 km^3/year internal renewable water resource (Water Resources Institute 2003) which is the major source of fresh water in the country. Tanzania inland water systems cover 61,495 sq km (URT 2002) which includes four lake basins (Lake Victoria, Lake Tanganyika, Lake Rukwa and Lake Nyasa), four river basins (Pangani, Wami-Ruvu, Ruvuma and Rufiji) and one internal drainage basin. Water bodies and wetlands form 7 % of the major ecosystem types in Tanzania (Water Resources Institute 2003). These provide a variety of aquatic ecosystem services across the country. Aquatic ecosystem services documented in Tanzania can be grouped following the Millennium Ecosystem Assessment categories; provisional, regulating, supporting and cultural services (Table 1). These ecosystems provide directly and indirectly to human welfare, livestock and wildlife and support key national economic sectors such as agriculture, energy production, industries and tourism. Nevertheless water resources in Tanzania have been projected to decline within a decade time. In the Water Sector Status Report (MOWI 2010), the total renewable water resource is projected to decline from 2,300 m^3/capital/ year in 2002 to 1,500 m^3/capital/ year by 2025 primarily due to population growth. According to the United Nations standard, water availability of less than 1,700 m^3/capital/year indicates a water stress status.

Table 1 Classified aquatic ecosystem services

Provisioning	Regulating	Supporting	Cultural
Building materials (Fibre, poles, timber, grass)	Support to nursery and reproduction cycles	Maintain and support aquatic and terrestrial biodiversity	Aesthetic
Freshwater as a resource	Flood attenuation	Carbon sequestration	Spiritual
Food (plants and animals)	Ground water recharge	Primary production	Tourism
Fuel (Charcoal and firewood)	Sediment retention	Hydrological cycle	
Handicraft materials	Water purification	Pollination	
Inland water transport	Nutrient cycling		
Medicinal materials	Nutrient input to agriculture		
Hydro-electric power	Micro-climate regulation		

The current status of aquatic ecosystem services and value in Tanzania is little known and where known is only in certain watersheds of the country. There is lack of national designed plan for monitoring and assessing changes in ecosystem health and services resulting in lack of alerts for threatened and deteriorated aquatic ecosystem functions and services. For most ecosystem services there is no readily available prices and ecosystem economic value. Under these circumstances the values of ecosystem services are only recognized after the loss of services as a result of degradation of natural processes supporting these services. There is a general gap between understanding ecosystem services and their relative value and it is just recent that ecologists and economists have begun analyzing ecosystem services and values simultaneously (Daily et al. 2000). In Tanzania valuation of aquatic ecosystem services is also a recent concept investigated in certain watersheds only. In this review we are focusing on Lake Victoria and the Great Ruaha River of Rufiji basins for case examples of ecosystem status, trends and change drivers in the country. The two basins are among important and critical ecosystems in Tanzania with relevance to economic aspects and provisioning of human well-being in the country thus will give a good insight of trends, status and influence of change drivers in Tanzania's aquatic ecosystems.

Lake and river ecosystems covering 6 % of Tanzania land cover are the major providers of water resource as an ecosystem services in the country for both *in situ* and withdrawal purposes. Major water supply consumptions are agriculture and livestock, domestic and industrial, energy generation and maintenance of aquatic and terrestrial ecosystems. Agriculture and livestock have higher water withdrawals (89 %) followed by domestic (9 %) and minimal for industrial purposes (2 %) (MOWI 2010). Water resources through *in situ* supply contribute up to 60 % of Tanzania electric supply. The LVB provides fisheries for multi million exports and millions local exports, support biodiversity containing about 400 endemic fish species, inland water transport across east Africa, support several industries, tourism and wildlife. In addition, LVB wetlands play an important role in filtering sediment and nutrients from watersheds before entering the lake, provide breeding habitat for aquatic life, building materials and fuel wood from vegetation cover (Swallow et al. 2009). The LVB ecosystem deterioration have happened over the past 60 years (Swallow et al. 2009) and a series of negative effects to the lake ecosystem have been noted and documented. Of significant effects include fluctuating lake water levels (Awange et al. 2007; Kiwango and Wolanski 2008), lake eutrophication and low oxygen levels (Scheren et al. 2000), high sediment loads due to collapsed wetlands, recurring of water hyacinth invasions, algal bloom proliferations, extinction of certain fish species, and increase in spreading of diseases (Malaria and HIV/AIDS) (Swallow et al. 2009).

Wetland ecosystems are important aquatic ecosystems within basins playing tremendous roles in provisioning aquatic ecosystem services in Tanzania. Wetland ecosystems occupy 10 % of the country and they support biodiversity maintenance by harboring about 654 animal species (SWMP 2010). Wetlands make backbone provision for direct consumers of natural resources including fishermen, farmers, pastoralists and local food vendors who form the majority of Tanzanian communities. Wetland services include water and food provision, source of energy and utensils and construction materials. They also support biodiversity by providing feeding, breeding and nursery areas for aquatic life and habitats and corridors for wildlife. A summary of highlights for services provided by wetlands in Tanzania is given in Box 1 as obtained from (SWMP 2010).

Box 1: Highlighting Figures of Wetlands Services to Tanzania Economy and Livelihoods (SWMP 2010)

- **95 %** hydro-electric power
- **95 %** of domestic, irrigation, industrial and livestock water
- **95 %** of rice production
- **95 %** of vegetables are grown in wetlands
- **95 %** of grazing on and drinking by wildlife
- **95 %** of game corridors/wildlife migration routes
- **95 %** of the 25 million livestock in dry season
- **95 %** of table salt production
- **95 %** of tourism (coastal and wildlife)
- **80 %** of traditional schemes irrigation
- **66 %** rural animal protein for food (livestock, fish and bush meat)
- **50 %** of non-forest products, medicines, clays, coral lime, salt and sand mining, house building materials, fibers for mats, etc is from wetlands
- 850,000 ha of wetlands have potential for future rice irrigation conversion.

3 Local Pressures and Drivers of Aquatic Ecosystem Change

The millennium assessment describes global direct and indirect drivers of ecosystem changes. The unequivocal drivers include climate change, pollution, land use changes and habitat conversion, natural resources exploitation, diseases and introduction or spread of invasive species. On the other side, the indirect drivers which cause or alter one or more direct drivers include demographic, micro and macro economic, social and political, scientific and technological and cultural and religious (Nelson et al. 2006). Most of these drivers transpire in different aquatic ecosystems of Tanzania.

One major national global driver of aquatic ecosystem change is increasing demographic trends. Population increase in the country has resulted in an increase in demand for water supply for domestic purposes, agriculture, livestock and hydroelectric power production. These water demands have contributed to low flows of surface waters and flow regulation which have tremendous contribution in ecosystem change. Future projections show that between 2025 and 2050 Tanzania will experience significant water scarcity based on population growth and water availability trends. Projections shows that the decreasing and increasing trends of water availability and population growth in the country from 1960 to 2125 (SWMP 2010).

The impact of increased population growth has been escalated by mismanagement of aquatic ecosystems, poor development planning, lack and poor enforcement of policies, lack of public awareness and national economic drive. Water resources and their management in Tanzania are determined by macroeconomics and policies (Turpie 2000). Currently the country's set up does not support payments or rewarding for ecosystem services use or protection. This has made water resource consumers in many areas of the country to act as if water is an infinite resource. If existing trends in population and resource exploitation persist, Tanzania is more likely to face critical water scarcity situation by 2050 (SWMP 2010).

3.1 Lake Victoria Basin

3.1.1 Demography

Lake Victoria with 69,000 km^2 surface area is shared by Kenya, Tanzania and Uganda. The lake has a total catchment area of about 194,000 km^2 estimated to provide services for over 30 million people. Annual fish catches of over one million tons in the LVB (Kolding et al. 2008); provide enormous economic opportunities which in turn perpetuate into fundamental demographic changes. Population growth rates in LVB have been alarming for the past decade (Odada et al. 2004, 2009). The population growth in the basin is reported to be higher than the absolute African continent by 2.5–11.2 % per decade from 1960 to 2000 years (Fig. 2) which

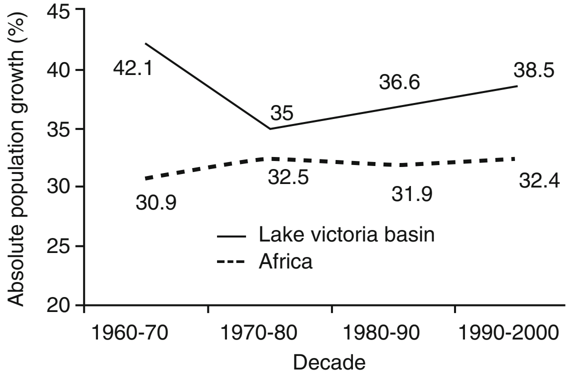

Fig. 2 Population growth in Lake Victoria in comparison with Africa continent (Source: Odada et al. 2009)

translates to 3 % annual growth rate per annum and average population density of about 165 pers/km^2 (Odada et al. 2009). This population depends on the LVB for resources and survival which is a threatening alarm to the functioning and provisioning of the ecosystem in the future.

3.1.2 Excessive Water Withdrawal

Construction of the Kiira dam for HEP production in 2000 in addition to the Nalubaale dam whose discharge forms the White Nile River was done against the set international agreement between Uganda and Egypt of the "Agreed Curve". The Agreed Curve was based on the natural discharge of the White Nile River. With concurrent operation of the Nalubaale and Kiira dams, adherence to the "Agreement Curve" was no longer possible resulting in overdrawing of water from Lake Victoria. This led to decrease in water level by 2.5 m between 2000 and 2006 (Kiwango and Wolanski 2008). Kiwango and Wolanski (2008) suggested that the water level in Lake Victoria would have not changed and rather remain constant during the 6 years period if the "Agreed Curve" was followed (Fig. 3). The decrease in water level caused severe ecological impacts in the LVB. The capacity of wetlands to filter sediments and nutrients and the role of papyrus in supporting tilapia artisanal fisheries were severely compromised. This further resulted into wetland loss, eutrophication, collapse of tilapia fisheries (Kiwango and Wolanski 2008) impacting food security and microeconomic within the shores of LVB.

3.1.3 Climate Change

Climate change and climate variability are significant direct drivers of ecosystem change within the LVB contributing up to 45 % of lake water level changes (Miller 2009). In the LVB, climate change has been noted since the Holocene era (Johnson

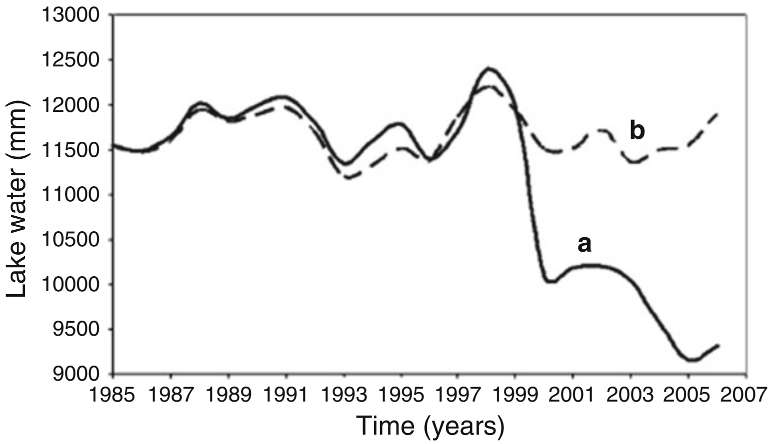

Fig. 3 Time series plots for the water level in Lake Victoria showing the effect of the Kiira dam in Uganda (**a**) observed and (**b**) predicted trends in the absence of the dam (Source: Kiwango and Wolanski 2008)

et al. 2000) and continues to vary over time and space as a result of anthropogenic activities (Odada et al. 2004). Trends in LVB climatic features showed a decrease of 10–40 % precipitation since 1960 (UNEP 2006) and there is a potential for further decrease in precipitation and increase in air temperature (Hulme et al. 2001). These climatic pressures on the LVB pose impacts on aquatic ecosystems and call for social and ecological impacts alerts. Severe weather stresses from climate change are high evaporation rates, increasing flood and draught events.

3.1.4 Exotic Species

LVB is an important example for elaborating ecosystem change and transformation as a consequence of exotic species introduction. Introduction of two fish species, Nile perch (*Lates niloticus*) and Nile tilapia (*Oreochromis niloticus*) in 1950s has led to development of highly valued fishery industry in the basin however with the consequence of resources over exploitation, lake eutrophication, land use and ecosystem changes which bring a concern on the sustainability of this valuable fishery (Kolding et al. 2008). Introduction of these fish species resulted in a decline in abundance and loss of the endemic haplochromines (Kaufman 1992; Goldsmidt et al. 1993; Gophen et al. 1993). Large scale and recurring invasion of water hyacinth (*Eichornia crassipes*) was severe in the 1990s affecting fisheries, changing food webs and clogging municipal water supply and transport systems (Scheren et al. 2000). The water hyacinth has financially cost the country and directly affected sectors.

3.2 Great Ruaha River

3.2.1 Land Use Change

Increasing population growth rates and overexploitation of ecosystem resources lead to further changes in land uses for survival. Land uses change is a growing driver in Tanzania for ecosystem changes. A good example is the GRR in Rufiji Basin which has undergone notable transformations in land uses leading to loss of the dry season perennial flows. Perennial rivers draining the Usangu highlands have contributed to the increasing growth of human activities over the past decade. The increased human activities are highly associated with land use changes through clearance of natural vegetation for irrigation and pastoralism. Since 2006, the Usangu plain of the GRR is considered a national rice producing centre (Mtahiko et al. 2006). The resulting ecosystem change have caused the loss of the GRR perennial flows which has a significant impact on ecosystem functioning and services.

Drying of the GRR has impacts on biological, ecological, social and economical functioning of GRR associated ecosystems. Some of the ecological effects of GRR drying up include loss of water provision to wild life and livestock, breaking of reproductive and life cycles of aquatic organisms, loss and extinction of some freshwater fish and oyster species, river bank erosion because of animals crowding in isolated pools for drinking purposes, eutrophication and algal blooms in over utilized isolated water pools and loss of dry season habitats. The major economic loss due to drying of the GRR is the decrease in the hydro electric power (HEP) supply in the country. GRR contributes 56 % of runoff to Mtera reservoir which provides water for Mtera and Kidatu HEP plants with combined installed power generating capacity of 280 MW which generates about 50 % of Tanzania's electricity. Figure 4 shows by percentage how human activities contribute to significant water losses in

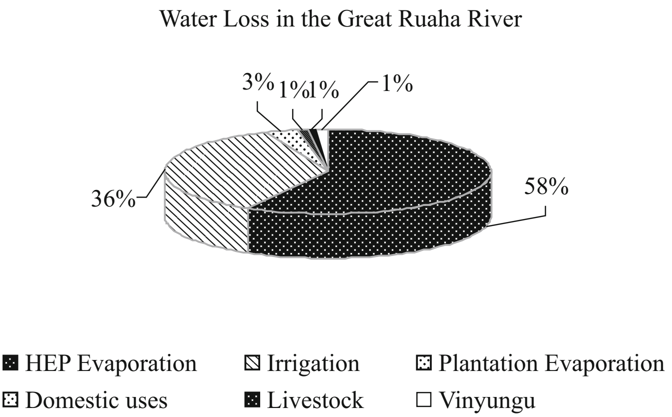

Fig. 4 Water loss by human activities in the Great Ruaha River (Source: adopted from Devisscher 2010)

the GRR. HEP evaporation contributes to 58 % water loss because of the temperature and altitude differences between Mtera dam and Usangu wetlands. The difference creates a 0.8 m/year higher evaporation rate in the dam than the wetland (Mtahiko et al. 2006). The resulting water scarcity builds to problems with food production, human health and economic development which further build up into sectoral conflicts among hydro-electric power production, irrigation and livestock.

4 Management Options for Lake Victoria Ecosystem

The management options for the LVB are aimed at reducing sediment, nutrients and agrochemical loading into the lake as well as strengthening institutional capacity for managing the shared water and fisheries resources of the lake. There does not seem to be any tangible effort to address the rapid population growth in the basin. The decline in water levels of Lake Victoria that reached crisis proportions in 2006 largely due to implementation of the second parallel hydropower plant (Kiira) at Jinja by Uganda have somehow recovered in recent years thanks to the pressure put by other riparian countries to follow the "Agreed Curve". Uganda has argued against the use of the "Agreed Curve" that it does not allow the use of the lake to store water in times of high rainfall and flows. This argument necessitated the East African countries that share the LVB to embark on the development of the New Water Release and Abstraction Policy (EAC 2011). The new policy has basically been adopted although its implementation is yet to start pending the completion of a Strategic Environment and Social Assessment (SESA) for the policy.

4.1 Government Efforts to Restore the Lake Victoria Ecosystem

The proliferation of water hyacinth in the Lake Victoria in the 1990s was arguably the most challenging ecological disaster the lake and community dependent on it ever faced. At the peak of proliferation the hyacinth occupied about 90 % of the littoral zone (EAC 2006). The total direct cost due to economic losses attributed to the water hyacinth proliferation in the Lake Victoria were estimated to be USD 6–7 million per annum in 1996 (EAC 2006). Although water hyacinth is a migratory plant introduced into Lake Victoria from the Kagera River, it is normally sustained and multiplies quickly due to high nutrients in the lake. The first phase of the Lake Victoria Management Project (LVEMP I) from 1997 to 2005 apart from supporting initiatives to remove and control water hyacinth, was also aimed at collecting and developing information base for improved management of the lake basin. Most of the water hyacinth and other weeds had been brought under control by the end of the 1990s through a number of methods namely; mechanical, manual and biological control methods.

LVEMP II that began in 2009 and is expected end in 2018 was developed based on the information and knowledge generated during the first phase. LVEMP II is expected to improve the ability of the riparian countries to embark on long term program for the basin resources in order to exploit and manage them in an environmentally sustainable manner (EAC 2006). Activities such as control of point source pollution have been undertaken and prevention of raw waste water discharges is done through rehabilitation and improvement of waste water treatment facilities. Strengthening of institutions capacity for managing shared water and fisheries resources that involves harmonization of policies and regulatory standards across all the riparian countries of East Africa is being undertaken through LVEMP II regional level programs (EAC 2006). The Simiyu catchment on the eastern side of Lake Victoria in Tanzania was identified during LVEMP I to be the most degraded and producing the highest sediment load and nutrients. LVEMP II has embarked on an intensive participatory process aimed at restoring the various ecosystems of the catchment through a number of watershed management initiatives that include reforestation, and up scaling of successful soil and water management piloted under the previous phase. Other initiative focus on natural resources conservation and livelihood improvement which is being conducted on selected areas based on the degree of soil erosion and natural resources degradation as well as the willingness of the community to participate as partners and stakeholders.

Destruction of the monotypic papyrus wetlands fringing the Lake Victoria significantly impact the micro and macroeconomic of the LVB and the country as a whole. Alerting increasing demographic rates along the lake shore increased the harvesting of papyrus and clearing of the welands for agriculture, livestock and settlements purposes leading to failure in wetlands fishery and wetlands ecological functioning (Van der Weghe 1981; Mafabi 2000). Excessive water withdraws as a result of Kiira dam construction in Uganda propagated the papyrus wetlands collapse along the shore of Lake Victoria (Kiwango and Wolanski 2008). As part of government initiative to mitigate the effects of papyrus wetlands loss, an experiment to restore the nutrients and sediments filtration role and tilapia refuge provision by creation of papyrus wetlands around the Lake Victoria was supported by the Rubondo Nationa Park (RNP). The creation of the wetlands was community based involving local fishery stakeholders for the purpose of valuing and ownership sense by the local communities (Kiwango et al. 2013). Kiwango et al. (2013) showed that the creation of the wetlands increased fishery around the area. Furthermore the study by Kiwango et al. (2013) showed the possibility of returning the degraded papyrus wetlands around Lake Victoria to ecological healthy and productive status. Thus papyrus wetlands creation could be one of the solutions to improve food security in the LVB.

5 Management Options for the GRR

The beginning of drying up of the GRR in Usangu catchment of Rufiji Basin for a prolonged period of time in the early 1990s served as wake up call for the Government of Tanzania and other interested agencies to take concrete measures to reverse the deteriorating ecosystem services that were causing enormous economic losses and environmental impacts. The start of the prolonged drying up of the GRR coincided with extreme low levels ever seen of the Mtera reservoir in 1992 that necessitated significant scaling down of hydropower generation at Mtera (80 MW) and the downstream plant of Kidatu (200 MW) for a couple of months resulting in pervasive black outs and huge economic losses.

A number of management options aimed at restoring the dry season flows in the GRR and subsequently re-establish the ecosystem services of the vital section of the GRR through the Ruaha National Park (RNP) into the Mtera were put forward by the SMUWC Project (SMUWC 2001). The proposed options ranged from supplementing irrigation with groundwater (conjunctive use of surface water and groundwater), restricting abstraction of water for irrigation in the dry season, construction of a major canal to convey wet season flows in the GRR from the western to the outlet of the eastern wetland at N'Giriama thus bypassing the Ihefu Swamp and construction of a reservoir to store water in the rain season and release it steadily during the dry season, among others. However, the option to restrict abstraction of water of irrigation in the dry season has always been conducted by the Rufiji Basin Water Office (RBWO) in collaboration with the Tanzania Electric Supply Company (TANESCO) without much success.

Currently, the detailed design for the construction of a dam on Ndembera River (one of the perennial tributaries feeding into the Ihefu Swamp) is underway. The primary purpose of the reservoir to be created is to regulate dry season flows such that the GRR section leading to Mtera through the RNP discharges water throughout the year. The proposed reservoir is projected to have a capacity of 351.7 million cubic metres. Studies have shown that release of water from the proposed reservoir is not likely to be able to meet the requirement of restoring flows for all the 6 months of the dry season without constructing a diversion short-circuiting Ndembera and GRR i.e. by passing the Ihefu Swamp (Lugomela 2012). Although the diversion has long been proposed by WWF, the current detailed design of the dam on Ndembera being overseen by the Government does not include the diversion.

5.1 Government Efforts to Restore the GRR Ecosystem

Expansion of irrigated land for rice cultivation and increase in the number of livestock are the two main social economic activities that are directly linked to the degradation of Usangu wetlands and its ecosystem which is the source of water for the GRR (Kihwele et al. 2012). In 2001, it was estimated that the maximum irrigated land under rice grown in a normal to wet year was 42,000 ha and the number of livestock were 300,767, 81,339 and 2,554 for cattle, sheep/goat and donkeys respectively in the late 1990s (SMUWC 2001). Several management decision and initiatives have been undertaken in order to reverse the deteriorating situation and regain lost ecosystem services.

In 1993 the Government of Tanzania established the Rufiji Basin Water Office (RBWO) with the core responsibility of managing the water resources of the basin particularly allocation of water and control of water abstraction for irrigation and other uses (SMUWC 2001). The River Basin Management and Smallholder Irrigation Improvement Project (RBMSIIP) implemented from 1996 to 2003 was aimed at ensuring that interlinkage between the needs of different users such as irrigators, livestock, domestic supply, hydropower and environmental use is taken into consideration when deciding on water allocation through promotion of participatory approach to planning and decision making involving all stakeholders (MoW 1999). The project also intended to deal with water management problems and improve the irrigation efficiency of small holders in order to save water that is badly needed downstream in the GRR and Mtera reservoir.

However, the efforts of the RBMSIIP did not produce tangible results as far as restoration of dry season flows in the GRR is concerned. The project simply did not address the core issue of presence of livestock and irrigation abstractions in Usangu that are considered as the main cause of drying up of the GRR. The amount of water saved through improved irrigation efficiency is too little to overcome the huge evaporative losses of the wetlands and trickle downstream. It is worthwhile to note that the average evaporation loss from the Ihefu swamp of the Eastern Usangu wetland is 5.71 m^3/s (Lugomela 2012). This means that a much bigger inflow is required to satisfy evaporation requirements of the wetland and swamp for water to eventually flow into the GRR downstream particularly during the dry season. Figure 5 shows the evaporation from Ihefu swamp for the simulated 2000–2009 period.

Although it has been demonstrated that the percentage of water consumed by livestock is only 0.6–1.3 % of the amount of water used for irrigation in Usangu (SMUWC 2001), the Government of Tanzania decided to forcibly removal of all livestock from the eastern wetland in 2006 in order to restore the wetland ecosystem and ultimately the year round flow in the GRR section leading to Mtera Reservoir

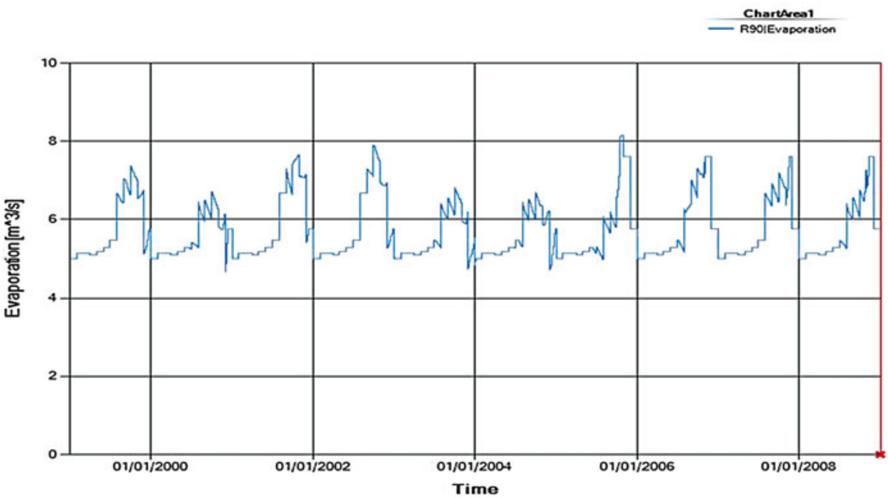

Fig. 5 Simulated Evaporation from Ihefu Swamp of Usangu Wetlands (Source: Lugomela 2012)

through the RNP. The move is reported to have re-stored 95 % of the Ihefu wetland (Kihwele et al. 2012). Its impact to discharges in GRR was to halve the duration of zero flows in the river in the dry season but its impact on inflows into Mtera reservoir was smaller but still readily measurable (Kihwele et al. 2012). Therefore the ecosystem services of the GRR section through the vital RNP remain fragile despite removal of livestock.

5.2 Donors Efforts to Restore the GRR Ecosystem

Several donor funded projects with the support of the Government of Tanzania have been conducted in the Rufiji Basin particularly Usangu catchment aimed at addressing the environmental challenges and water use management. The projects include the Rufiji Environmental Project (REMP) that operated from 1998 to 2003 with support from the International Union for Conservation of Nature (IUCN), Raising Irrigation Productivity and Releasing Water for Intersectoral Needs (RIPARWIN) that was supported by the Department for International Development (DFID) of United Kingdom and operated from 2002 to 2005 and the Ruaha Water Program (RWP) which has been operating from 2003 to date with the support of the World Wildlife Fund (WWF). The RWP has been the longest running and intensive one aiming at protecting water sources through participatory approaches involving Water User Associations (WUAs), providing financial and technical support to the RBWO on a number of areas, capacity building to farmers on proper irrigation methods through "field farm schools", carrying environmental flow assessment study of the GRR section though the RNP to Mtera reservoir and others. However, despite all these intervention from RWP and other previous undertakings the river condition has not changed (WWF – Tanzania Country Office 2012).

6 Aquatic Ecosystem Protection as a Key Driver to Adoption of IWRM Approaches in Tanzania and Way Forward

The drive for policy change in water resources management in Tanzania was contributed by a number of factors related to water resources conflicts and deteriorating riverine and wetlands ecosystem due to diminishing water flows. The decrease in water available for HEP plant as the Pangani Falls Redevelopment Project (66 MW) was being designed necessitated the establishment of the first basin water office in 1991, namely Pangani Basin Water Office (PBWO), in order to establish close and holistic control and regulation of water abstractions and waste water discharges upstream. Sectoral conflicts for water use were on the increase in the early 1990s particularly in Pangani and Rufiji Basins, the classic example being the conflicting requirements for the Lower Moshi Rice Irrigation Project against the Pangani Fall

HEP Project downstream whereby it was finally decided to abandon the second phase of the irrigation project due to water scarcity.

The beginning of prolonged drying up of the ecologically important GRR in the Usangu Catchment coupled with the extreme low levels of the Mtera reservoir in 1992 that resulted in the first extended power rationing in the country was another driver for policy change in Tanzania. This prolonged drying up of the GRR has been causing significant damage to wildlife dependent of river water in the RNP (WWF – Tanzania Country Office 2012). As a result the RBWO was established in 1993 for the same purpose as its PBWO counterpart.

Pollution was also on the increase resulting in riverine ecosystem degradation in Pangani Basin due to discharge of sisal wastes, mercury discharge in streams due to small scale miners in LVB, improper disposal of untreated sewage particularly in Mwanza municipal and improper use of fertilizers and pesticides (URT 1995). Other point sources pollution particularly industries and municipals rarely comply with required effluent standards. One example from the point source pollution surveillance monitoring program conducted by the PBWO shows poor compliance over a long period by large industrial companies and municipals (PBWO 2009). The proliferation of water hyacinth, weeds and algae bloom in the early 1990s in the Lake Victoria and the concomitant loss of ecosystem services and associated costs was arguably the most serious alarm ever sounded with regard to the need for ecosystem protection in Tanzania.

In 1994 the then Ministry of Water Energy and Minerals conducted a countrywide Rapid Water Resources Assessment in order to quantify the challenges, problems and set out the next steps for water resources management. The assessment revealed that water resources management and planning was fragmented and sector oriented resulting in water use conflicts. Review of the 1991 water policy showed that water resources management and aquatic ecosystem monitoring and protection were not given due attention (URT 1995). The assessment indicated that the then Water Utilisation and Control Act No. 42 of 1974, its amendment No. 10 of 1981 and the 1991 water policy were not adequate to deal with emerging water resources management issues. The River Basin Management and Smallholder Irrigation Improvement Project were implemented in Pangani and Rufiji basins (1996–2003) as a follow up of the rapid water resources assessment study. The project culminated in production of the New Water Policy (NAWAPO) that was approved by the government in July, 2002. The NAWAPO – 2002 advocates IWRM approaches as tools to ensure ecosystem protection and the environment is assigned second priority in water allocation after basic human needs (URT 2002). The new water policy also recommended that the Water Utilisation and Control Act No 42 of 1974 and its amendments be reviewed in order to have legislations that are more responsive to water resources management issues particularly aquatic ecosystem protection and monitoring.

The Water Resources Management Act No 11 of 2009 is a product of the NAWAPO – 2002 and it requires each basin in Tanzania to prepare an IWRM plan (URT 2009). The cornerstone of an IWRM plan is ecosystem protection as a way of achieving sustainable socioeconomic, environmental and water resources development and the new water act requires that the IWRM plan must include water reserves for each water source and environmental flows provided. Currently, all basins in Tanzania are developing the IWRM plans through the support of World Bank funds and other donors. The interim report for Rufiji Basin IWRM plan recognizes the environmental problems and ecosystem degradation caused by the drying up of the GRR (URT 2012). Once the IWRM Plans are finalized and implemented, the new era and approach to water resources management in Tanzania will have dawned.

7 Way Forward

In Tanzania almost all the necessary ingredients for an IWRM oriented water resources management are in place. The water policy that spearheads IWRM principles and a piece of legislation to enforce the policy desire are already in use. The institutional framework that reflects the participatory principle and holistic approach of IWRM has been set up. Water resources management is carried out through the water basin offices as basic units supported by Catchment/Sub catchment committees and Water Users Associations at the lower levels. The participation of different sectors in water resources planning is achieved through the Basin Water Boards (BWBs) and the National Water Board (NWB) at the local and national level respectively. Both the BWBs and the NWB are composed of members from different sectors. The IWRM plans will form legal basis to manage water in an environmentally and ecosystem responsible manner. It will be through the approved plans that drastic action can be legally taken to protect and/or restore important ecosystem services in hotspot areas like the GRR.

References

Awange JL, Sharifi MA, Ogonda G, Wickert J, Grafarend EW, Omulo MA (2007) The falling Lake Victoria water level: GRACE, TRIMM and CHAMP satellite analysis of the lake basin. Water Resour Manag. doi:10.1007/s11269-007-9191-y

Daily GC, Söderqvist T, Aniyar S (2000) The value of nature and the nature of value. Science 289:395–396

Devisscher T (2010) Ecosystem-based adaptation in Tanzania: the role of ecosystem services for human well-being and climate adaptation. Development Partners Group. UK aid from the department for International Development, UK, 45 pp

East African Community (EAC) (2006) Transboundary diagnostic analysis of the Lake Victoria Basin, LVBC, Arusha, Tanzania

East African Community (EAC) (2011) The development of a new water release and abstraction policy for the Lake Victoria Basin. Draft Final Report. Centre for Ecology and Hydrology (CEH)

Goldsmidt T, Witte F, Wanink J (1993) Cascading effects of the introduced Nile Perch on the detrivorous/phytoplanktivorous species in the sublittoral areas of Lake Victoria. Conserv Biol 7:686–700

Gophen M, Ochumba PBO, Pollingher U, Kaufman LS (1993) Nile Perch (Lates niloticus) invasion in Lake Victoria (East Africa). Verhandlungen der Internationale Vereinigung fur Limnologie 25:856–859

Hulme M, Doherty R, Ngara T, New M, Lister D (2001) African climate change 1900–2100. Clim Res 17:145–168

Johnson JC, Kelts K, Odada EO (2000) The Holocene history of Lake Victoria. Ambio 29:2–11

Kaufman L (1992) Catastrophic change in species-rich freshwater ecosystems. Bioscience 42:846–858

Kihwele E, Mnaya B, Meng'ataki G, Birkett C, Wolanski E (2012) The role of vegetation in the water budget of the Usangu wetlands – Tanzania. Wetl Ecol Manag 20:389–398

Kiwango Y, Wolanski E (2008) Papyrus wetlands, nutrients balance, fisheries collapse, food security and Lake level decline in 2000–2006. Wetl Ecol Manag 16:89–96

Kiwango Y, Gadiel M, Kibasa W, Mnaya B (2013) Papyrus wetlands creation, a solution to improve food security and save Lake Victoria. Wetl Ecol Manag 21:147–154

Kolding J, van Zwieten P, Mkumbo O, Silsbe G, Hecky RE (2008) Are the Lake Victoria fisheries threatened by exploitation or eutrophication? Towards an ecosystem based approach to management. In: Bianchi G, Skjodal HR (eds) The ecosystem approach to fisheries. CAB International, Rome

Lugomela GV (2012) Evaluation of the proposed reservoir on Ndembera River of Usangu catchment – Rufiji Basin.In: Proceedings of the 4th IHP Africa Water Science, Policy and Management Conference; Water Science for South Africa Water Policies and Governance in a Changing Environment, April, 2012 – Dar es Salaam

Mafabi P (2000) The role of wetland policies in the conservation of waterbirds: the case of Uganda. Ostrich 71:96–98

Millennium Ecosystem Assessment (2005) Ecosystems and human well-being: current state and trends. Island Press, Washington, DC. http://www.millenniumassessment.org/en/Reports.aspx

Miller NL (2009) Climate vulnerability assessments of water utilities in the Lake Victoria Basin. United Nations – Habitat Project, World Water Week Stockholm, Sweden. http://www.worldwaterweek.org/documents/WWW_PDF/2009/monday/K21/NLMiller-WWW17Aug09.pdf. Accessed 5 Nov 2013

Ministry of Water (MOW) (1999) River basin management and smallholder irrigation improvement project, vol 2. Dar es Salaam, Tanzania

Ministry of Water and Irrigation (MOWI) (2010) Water sector status report

Mtahiko GMM, Gereta E, Kajuni AR, Chiombola EAT, Ng'umbi GZ, Coppolillo P, Wolanski E (2006) Towards an ecohydrology based restoration of the Usangu wetlands and the Great Ruaha River, Tanzania. Wetl Ecol Manag 14:489–503

Nelson GC, Bennett E, Berhe AA, Cassman K, DeFries R, Dietz T, Dobermann A, Dobson A, Janetos A, Levy M, Marco D, Nakicenovic N, O'Neill B, Norgaard R, Petschel-Held G, Ojima D, Pingali P, Watson R, Zurek M (2006) Anthropogenic drivers of ecosystem change: an overview. Ecol Soc 11(2):29

Odada EO, Olago DO, Kulindwa K, Ntiba M, Wandiga S (2004) Mitigation of environmental problems in Lake Victoria, East Africa: casual chain and policy option analyses. Ambio 33:617–627

Odada EO, Ochola WO, Olago DO (2009) Drivers of ecosystem change and their impacts on human well-being in Lake Victoria Basin. Afr J Ecol 47(suppl 1):46–54

Pangani Basin Water Office (PBWO) (2009) Water quality and pollution control summary. Pangani Basin Water Office

Scheren PAGM, Zanting HA, Lemmens AMC (2000) Estimation of water pollution sources in Lake Victoria, East Africa: application and elaboration of the rapid assessment methodology. J Environ Manag 58:235–248. doi:10.1006/jema.2000.0322

Sustainable Management of the Usangu Wetland and its Catchment (SMUWC) (2001) Usangu water resources, Technical report – Final draft report, Dar es Salaam, Tanzania

Sustainable Wetlands Management Programme (SWMP) (2010) Informative policy brief for the establishment of community based natural resource management of Wetlands. MNRT, Sustainable Wetlands Management Program, Dar es Salaam

Swallow B, Sang JK, Nyabenge M, Bundotich DK, Duraiappah AK, Yatich TB (2009) Tradeoffs, synergies and traps among ecosystem services in the Lake Victoria Basin of East Africa. Environ Sci Pol 12(4):504–519

Turpie JK (2000) Environmental management and biodiversity conservation of forests, woodlands, and wetlands of the Rufiji Delta and floodplain. The Use and Value of Natural Resources of the Rufiji Floodplain and Delta, Tanzania. Rufiji Environment Management Project1. Technical report No. 17

United Nations Environmental Programme (UNEP) (2006) Lake Victoria environmental outlook: environment and development. UNEP, Nairobi

United Republic of Tanzania (URT) (1995) Rapid water resources assessment for Tanzania, United Republic of Tanzania, World Bank and DANIDA, Part I

United Republic of Tanzania (URT) (2002) National water policy. Dar es Salaam, Tanzania

United Republic of Tanzania (URT) (2009) Water resources management act. Government Printer, Dar es Salaam

United Republic of Tanzania (URT) (2012) Rufiji IWRMD plan: interim report volume 1, Rufiji River basin physical, socio-economic and management profile. WREM International Inc, Atlanta, USA

Van der Weghe JP (1981) Avifauna papyrus in Rwanda and Burundi. Gerfaut 71:489–536

Water Resources Institute (2003) Earth trends country profiles: water resources and freshwater ecosystems, Tanzania

World Wide Fund for Nature (WWF) – Tanzania Country Office (2012) Freshwater news, Volume 1; Ruaha Water Programme, Issue No. 1

Coastal Watershed Ecosystem Services Management in West Africa: Case of Ghana and Nigeria

Julius Ibukun Agboola and Shakirudeen Odunuga

Abstract Demands for ecosystem services such as food and clean water are growing with concurrent human actions diminishing the capability of many ecosystems to meet these demands. In West Africa, coastal watershed ecosystems are subject to many pressures (e.g., land-use change, resource demands, and population changes); their extent and pattern of distribution is changing, and landscapes are becoming more fragmented. In this context, we review the current state and analyze the drivers of change in coastal watershed ecosystem goods and services in West Africa (Ghana and Nigeria). Based on identified critical drivers of change- "climate change" and "socio-economic", we present possible scenario for effective management of coastal watershed ecosystem services. Whilst there is an urgent need to safeguard ecosystem services, policy goals leading to sustainable management of coastal watershed ecosystems for delivery of ecosystem services need to be established and implemented at both international and national levels.

Keywords Ecosystem goods and services • Coastal watershed • West Africa • Management

1 Introduction

Globally, aquatic ecosystems are rich and diversified sanctuaries for biodiversity, performing many important environmental functions: providing habitat for plant and animal species in the watershed, helping to absorb and slow floodwaters when rivers overflow, recharge ground water, absorb and recycle nutrients, sediments and other pollutants (Loeb 1994; Odada et al. 2008). Aquatic ecosystems are also used for human recreation and are very important to the tourism industry, especially in

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coastal regions (Daily 1997). The value of aquatic systems has been more understood in recent decades and the need for proper management approaches has been widely acknowledged (MEA 2005).

More importantly, coastal watersheds play an important role in maintaining biological diversity by providing a habitat for many plant and animal species, some of which are endemic or endangered. Drainages and coastal watersheds/basins in West Africa are presented in Figs. 1 and 2. Considering the numerous benefits of coastal watersheds, exploring trends and state of the coastal watershed ecosystem services in West Africa, especially in Ghana and Nigeria, and documenting implications of changes on the long-term sustainability of West African coastal ecosystems will help to inform future research priority and policy response for decision making. The Science Plan and Implementation Strategy on Global Environmental Change Research in Africa (Odada et al. 2008) indicates four top-level issues as the focus of concern with respect to global environmental change and its impacts in Africa: (1) Food and nutritional security, including crops, wild-gathered resources, livestock resources and fisheries; (2) Water resources, particularly in the water-limited sub-humid, semi-arid and arid regions; (3) Health, especially in relation to the biodiversity-linked, environmentally-mediated and vector-borne diseases that are responsible for the high disease burden in Africa; and (4) Ecosystem integrity, on which the persistence of biodiversity and the delivery of ecosystem services

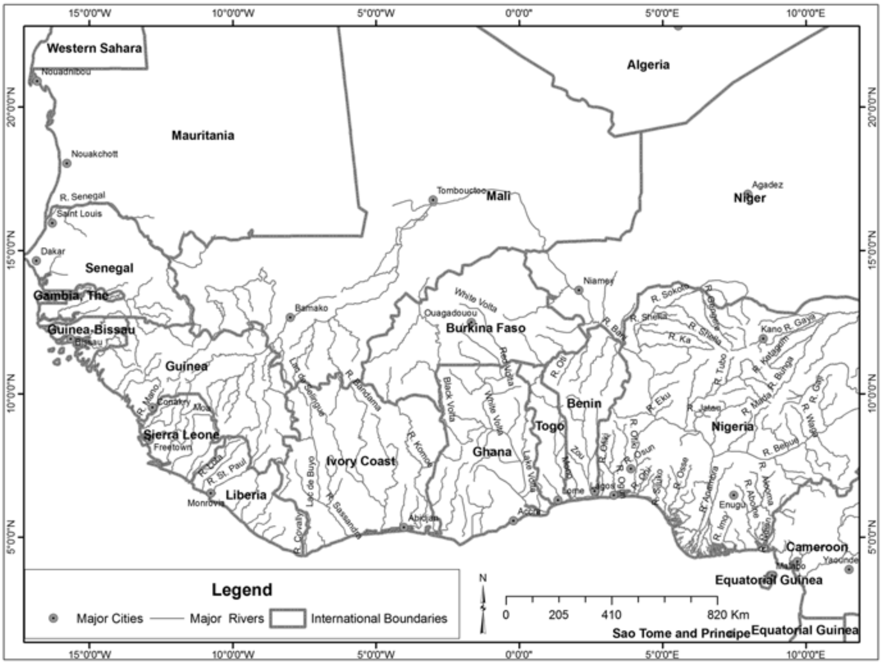

Fig. 1 Drainages in West Africa

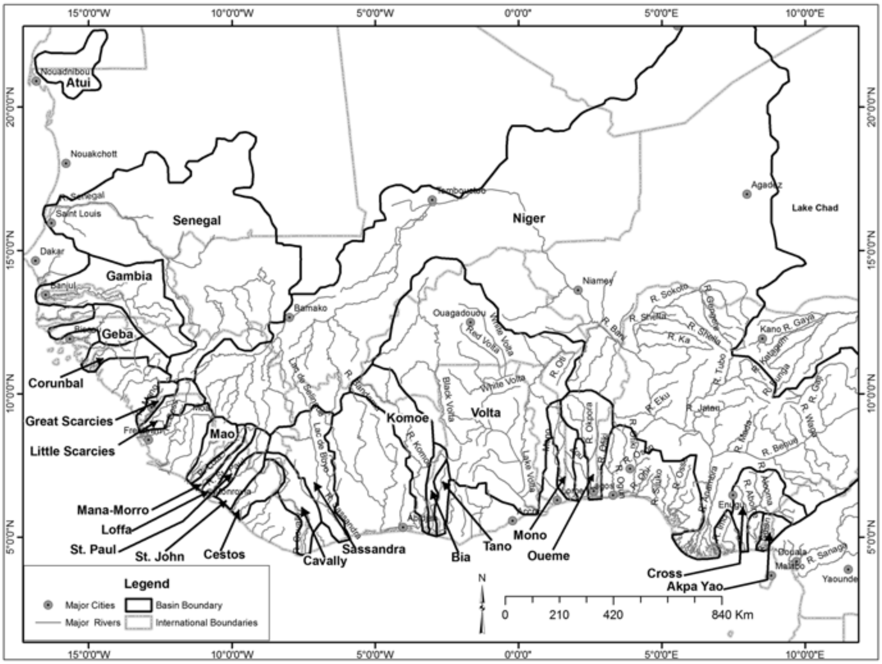

Fig. 2 West Africa coastal watersheds/basins

depends. These focal issues find expression, for instance, in the Millennium Development Goals and form main thrust of this chapter.

Ecosystem services are defined as “the benefits people obtain from ecosystems” and can be generally categorized as: provisioning (e.g. food, water, and energy), regulating (e.g. carbon sequestration and climate regulation, water purification, waste decomposition), supporting (e.g. nutrient dispersal and cycling, seed dispersal, primary production) and cultural (e.g. cultural and spiritual inspiration, recreational experience, scientific discovery) (Millennium Ecosystem Assessment 2005). According to Constanza (2008), ecosystem goods and services, whether intermediate (or “supporting” in the Millennium Assessment typology) or final services are all contributors to human well-being. Scientific work on ecosystem services has been growing globally as well as in Africa (Layke 2009). Human dependence on provisioning ecosystem services in particular is mostly acknowledged in developing countries like those in Africa, where many people are poor and reliant on natural resources (Egoh et al. 2012).

Ecosystem services could play an important role in helping policy makers understand local welfare impacts that they may not have considered otherwise, especially in the West African region where a majority of the population depends on coastal ecosystem services for their sustainable livelihood. Pressures of climate change on habitat and biodiversity will be largely indirect including carbon sink on most coastal wetland types, operating through changes in water level. In the same manner, over exploitation food and fiber production ecosystem services simultaneously form a pressure (land-use change) on other services (habitat, biodiversity, carbon sink). Thus, effective management of trade-offs arising from ecosystem services is non-negotiable in the face of current global environmental change uncertainties.

1.1 Description of the Coastline

The descriptive attributes of coasts provide baseline information and reference points for assessing the condition of the ecosystem's goods and services. They also are a major factor in the vulnerability and resilience of an area to a particular pressure. The extent and loss of these natural habitat types serve as a proxy condition indicator for many of the ecosystem services and values that are otherwise difficult to quantify.

Western African countries are homogenous, first, in relation to geology and physiography, secondly in relation to populations, culture and history, and finally in relation to economy and social conditions characterizing as developing countries. Several characteristics recur from one country to another: a strong demographic growth, a young population, fast and uncontrolled urbanization, a national economy dominated by the agricultural sector, slow human resources development, poor access to drinking water and insufficient or non-existing sanitation systems. There are 16 countries within the region; twelve of them are coastal countries. The coastal countries include: Benin, Côte d'Ivoire, Gambia, Ghana, Guinea, Guinea Bissau, Liberia, Mauritania, Nigeria, Senegal, Sierra Leone and Togo. In addition to these coastal states, there is also one island state (Cape Verde). Within the region there are several clusters of coastal river systems and morphological units that provide ecosystem services. The cluster of coastal river systems in the region can be divided into three: (1) Senegal to the Little Scarcies cluster: The major basins here are Senegal, Gambia, Geba, Corubal, Great Scarcies and Little Scarcies. (2) Mao to Sassandra cluster: The major rivers here include; Mao, Mana-Morro, Loffa, St. Paul, St. John, Cestos, Cavally and Sassandra and (3) The Niger, Volta and adjacent smaller basins cluster: The major river systems are Niger, Volta, Komoe, Cross River, Bia, Benin, Mono, Oueme, Ogun and Tano basins (Oyebande et al. 2002; UNEP WRC 2008). The Nigeria and Ghana coastline under investigation falls into the third cluster (The Niger, Volta and adjacent smaller basins).

Morphologically, the Nigerian Coastline can be classified into four zones (Fig. 3) based on districts morphology, ecology, beach type and ecosystem services; these are:

1. **The barrier-lagoon coast complex**: The barrier-lagoon coast complex extends for 250 km from the Benin/Nigeria border eastwards to the western limit of the transgressive mud beach;
2. **The Mahin transgressive mud coast and beach**: This extends for 75 km and terminates at the Benin river mouth on the Northwest flank of the Niger delta;

Fig. 3 Morphological zones of Nigeria coastline

3. **The Niger delta coast:** The Niger delta coast extends from the mouth of the Benin river for about 500 km to the mouth of the Imo river in the East. It contained one of the largest mangrove ecosystems in the world.
4. **The strand coast:** This extends from the Imo River to the Cross river estuary at the Nigeria/Cameroun border (Sexton and Murday 1994)

The Ghana coastline can be classified into three zones (Fig. 4) also based on districts morphology, ecology, beach type and ecosystem services these are: the eastern coast, the central coast and the western coast (Boateng 2006);

1. **Eastern coast**: this is about 149 km and stretches from Aflao (Togo Border) in the East to the Laloi lagoon west of Prampram. It is a high-energy beach with wave heights often exceeding 1 m in the surf zone (Ly 1980). It consists of an eroding sandy shoreline and is characterized by barrier beaches and bars confining a lagoon.
2. **Central coast** represents a medium energy environment. It is an embayment coast of rocky headlands and sand bars with spits enclosing coastal lagoons. It consists of 321 km of shoreline extending from Laloi Lagoon west of Prampram to the estuary of River Ankobra near Axim.
3. **Western coast:** This covers 95 km of shoreline and it is composed of relatively low energy beaches. It consists of a flat and wide beach backed by a coastal lagoon. The coast extends from the estuary of the Ankobra River to the border with La Cote d'Ivoire.

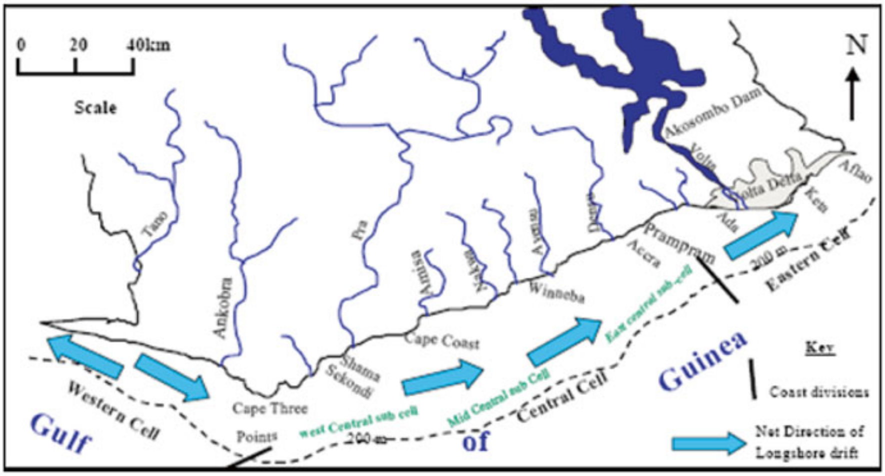

Fig. 4 Coast of Ghana showing the major rivers (After Benneh and Dickson 1988)

2 Coastal Watershed Ecosystem Goods and Services in West Africa

Africa's many aquatic systems display the characteristics of richness in diversity with regards to both the number of species present, as well as the localized endemism of species. In West Africa, there are several coastal communities whose livelihood revolves around the exploitation of biological resources in their environment. Besides, as in other parts of the world, the coastal area is preferred for urbanization and industrialization as well as amenities for recreation and tourism. Products from coastal watersheds include fisheries (fish, amphibians, crustaceans, and mollusks) and aquaculture; services include biodiversity, recreation, aesthetics, and biogeochemical cycling. Table 1 presents an overview of country profiles for Ghana and Nigeria. Recent developments show that the coastal and marine domain of Ghana boasts huge living and non – living resources that have not yet been fully exploited. Like many coastal states, Ghana has demarcated a 200 nautical mile Exclusive Economic Zone (EEZ) within the framework of the UN Convention on the Law of the Sea (UNCLOS). In 2010, the country also applied for an extension of its EEZ to the 350 nautical miles limit of the continental shelf. This would in effect bring a lot more resources under its jurisdiction. However, the nation lacks the requisite capacity to carry out the enormous task of monitoring, control and surveillance of the marine environment. In like manner, one would assume that Nigeria has greater potential coastal resources than Ghana; however, considering the increasing coastal population growth in Nigeria, pressure on these resources are far greater.

Table 1 Statistics for West Africa, covering Ghana and Nigeria (data partly extracted from UNESCO-IOC 2009)

Country's statistics/profile	Ghana	Nigeria
Capital city	Accra	Abuja
Population (CIA 2012 est)	24,652,402	170,123,740
Population growth rate (%)	1.9	2.2
Total area (km^2)	239,460	923,768
	Land-230,940	Land 910,768
	Water-11,000	Water-13,000
Length of coastline (km)	539	853
Continental shelf (km^2)	23,700	46,300
Exclusive Economic Zone (EEZ) (km^2)	218,100	210,900
GDP per capita (USD 2010 est)	$2,480	$2,381
Coral reef area (2001 est)	–	0
Marine protected areas (2007 est) (km^2) (% of total territorial waters)	None recorded	0 (0 %)
Mangrove area (2005 est) (ha)	12,400	997,000
Capture fisheries prod. (2006 est) (metric tons)	366,919	552,323
Aquaculture fisheries prod. (2006 est) (metric tons)	1,150	84,578

2.1 Current Status of Key Coastal Ecosystem Goods and Services

Both natural and managed ecosystems deliver important ecological services such as the production of food and fibre, the capacity to store carbon and to recycle nitrogen, and the ability to mitigate effects of climate and other disturbances. However, human activities are profoundly influencing the Earth with the consequence that survival pressure is now intense on both terrestrial and aquatic ecosystems (Agboola and Braimoh 2009). As a consequence, changes in the structure and function of ecosystems resulting from biodiversity alterations and loss have reduced the availability of vital services and affect the aesthetic, ethical and cultural values of human societies (Naiman et al. 2006). The reliance of communities on natural resources in Africa varies from place to place as aridity, vegetation and socio-economic conditions change (Rebelo et al. 2010). In the humid and forested areas in Western Africa, food and raw materials coupled with agriculture are important ecosystem services (Egoh et al. 2012).

"Ecosystem services on the coast are often disrupted by human activities. For example, the West African tropical mangrove forests and salt marshes provide goods and services (they accumulate and transform nutrients, attenuate waves and storms, bind sediments and support rich ecological communities), which are reduced by large-scale ecosystem conversion for agriculture, industrial and urban development, and aquaculture" (Nicholls et al. 2007). According to an OECD (1999) report, close to 50 % of wetland ecosystems have already been lost in the developing world and African freshwater bodies are more degraded than terrestrial and marine habitats (UNEP 2006). The link between wetland ecosystem functions and values, development and poverty alleviation is a challenge that has not been met in the efforts towards alleviating poverty. The multiple-usage of the coast and its socio-economic benefits has led to degradation in ecosystem goods and services.

In general, West Africa local communities rely mostly on food and raw materials such as non-timber forest products coupled with agriculture. For example, the hunting and trading of bush-meat in West Africa has developed into a large industry (Bowen-Jones et al. 2003). Timber extraction for export is also an important ecosystem service in these regions. Livelihoods are supported from a combination of these products as well as small to medium scale agriculture.

More importantly, West African fisheries are under tremendous pressure, some of this due to the increasing local demand for fish, and the growth of locally based industrial and artisanal fisheries (Palomares and Pauly 2004). However, the main reason for the much depleted state of West African fisheries resources lies in the presence, along the West African coast, of a huge array of Distant Water fleets from Western and Eastern Europe, and from East Asia. And every few years, new 'access agreements' are signed that increase this external pressure, not to mention numerous cases of illegal fishing by a variety of countries.

Ghana, located in the western Gulf of Guinea sub-region, between the Côte d'Ivoire and Benin, has, or rather had, very rich fishery resources, and a long tradition of artisanal and distant-water water fishing, the latter a unique feature amongst West African countries (Alder and Sumaila 2004). As in most other parts of the world (Pauly et al. 2002), Ghana's fisheries resources suffer from excessive fisheries pressure resulting in changes in ecosystem structure, reflected in declining catches of targeted species and, in combination with environmental changes, in short-lived outbursts of normally uncommon species (Koranteng 1998, 2002; Koranteng and Pauly 2004).

Nigeria alike is bordered by Benin, Chad, Cameroon, and Niger and has a coastline of 853 km which borders the Atlantic Ocean in the Gulf of Guinea. The limits of Nigeria's territorial waters and Exclusive Economic Zone (EEZ) are 12 nautical miles (nm) and 200 nm respectively. The total area of the continental shelf in the EEZ is approximately 37 900 km^2 (FAO 2007) but the flats are interrupted coast-wide by unburied fossil corals at 40–120 m depth. There is over-capitalization in the industrial fleet; over fishing of the coastal resources; declining catch, both in quantity and especially in quality; environmental degradation seriously impeding the productivity of the artisanal sector; and declining efficiency due to lack of technical innovation.

For the estuarine and brackish-water fisheries, the main issue is pollution (industrial, human) and geophysical (such as natural hazards). Many fishing households in this environment can only just subsist, having lost their income generating capacity. The waters around are becoming less and less productive (FAO fishery country profile).

3 Pressures on Key Coastal Ecosystem Goods and Services

As earlier mentioned, coastal watershed ecosystems in West Africa are subject to many pressures (e.g., land-use change, resource demands, and population changes); their extent and pattern of distribution is changing, and landscapes are becoming more fragmented. Generally, the main coastal issues peculiar along the West African coasts are: urban sprawl, coastal erosion caused by natural and anthropogenic factors and high pollution levels. Other issues are sea-level rise, precipitation and floods and depletion of mangrove and fisheries resources. Table 2 shows coastal threats, hotspot locations and the sensitive areas on the coasts of Ghana and Nigeria in West Africa. Climate change constitutes an additional pressure that could change or endanger ecosystems and the many goods and services they provide. Here, we highlight on four major pressures below.

3.1 Coastal Population and Livelihoods

Increasing coastal population growth is becoming a pressure on the West African coasts. As shown in Table 1, Ghana has a coastline of 565 km. The population in 2000 was estimated over 18 million people with a growth rate of 2.6 % (ACOPS 2002b). 42.5 % of the people live within 100 km from the coast while 25 % of the population lives below the 30 m contour along the coastal zone (ACOPS 2002b). The coastal zone of Ghana covers about 7 % of the total land area with a population of 4.5 million people, which constitutes 43.1 % of the population (ACOPS 2002b).

In Nigeria, about 26 % of the population lives within 100 km of the coast while the population living in the coastal zone is 25 % (ACOPS 2002c). For instance, Lagos State, with a current population of 17 million, is the commercial and industrial hub of Nigeria. Estimated to account for over 60 % of Nigeria‘s industrial and commercial establishments, with over 2,000 manufacturing industries and about 200 financial institutions, provides 60 % of Nigeria‘s Gross Domestic Product, 65 % of national investments (Web and Lagos State 2010). Current demographic trend analysis in terms of urban population revealed that the population in Lagos is growing ten times faster than New York and Los Angeles with grave implications for urban sustainability (Web and Lagos State 2011). Also, the implication of this on coastal watershed ecosystems goods and services cannot be over-emphasized.

The Niger Delta area in Nigeria is projected to lose over 15 000 km^2 of land by the year 2100 with a 1 m sea-level rise. A sea-level rise of 0.5 m is projected to result in 9,000 km^2 of land loss, displacing about 1.9 million people (ACOPS 2002c). Much of the land loss from sea-level rise will be due to inundation. A 1 m sea-level rise, however, could see over 15,000 km^2 or 2 % of the Nigerian coastal zone inundated, about 3.7 million people put at risk and a projected 812 villages along the Nigerian coast impacted. Of all the coastal zones, the Niger delta will be the most affected with up to 350 villages impacted and 2 to 3 million people displaced (ACOPS 2002c; Abuodha 2009).

3.2 Fisheries, Resource Management and Biodiversity

The wealth of fisheries has been intensely exploited by foreign and local fishing fleets since the 1960s with Ghana and Nigeria claiming the largest share of the harvest on the West African coasts (Fig. 5). In Ghana, marine fisheries resources especially small pelagic fisheries as well as lagoon and estuary fisheries have suffered depletion due to

Table 2 Coastal threats/problems, hotspot and sensitive locations in West Africa: Ghana and Nigeria

Coastal threats/problems	Hotspot locations	Sensitive locations
Ghana		
Soli solid wastes	Keta Lagoon Complex, Korle Lagoon, Sakumo I Wetlands	Son Songor Lagoon
		Ea East Central Sandy Beach
		Ad Ada/Volta Estuary/Anyanui Mangrove Wetlands
Modification of ecosystems	Sakumo I Wetlands, Keta Lagoon Complex, Korle Lagoon	Songor Lagoon, Ada/Volta Estuary/Anyanui Mangrove Wetlands
Reduction of stream flow	Keta Lagoon Complex, Sakumo I Wetlands	Songor Lagoon, Ada/Volta Estuary/Anyanui Mangrove Wetlands
Illegal coastal sand mining	Ahanta West District	Sekondi-Tekorado, Ahanta west district
Sea piracy in the Gulf of Guinea	Ghanian coastal area	
Illegal unreported unregulated (IUU) fishing	Ghanian coastal area	
Nigeria		
Solid wastes	Lagos Island Area	Ibeju-Lekki West
Spills	Eket Area, Ogoni Land, Bonny Area	Opobo Area, Barrier Island between Dodo and Nun Rivers, Brass area
Modification of the ecosystems	Lagos Island Area including Lekki	Opobo Area, Barrier Island between Dodo and Nun Rivers, Brass area
	Peninsular and Ibeju-Lekki Area, Eket Area, Ogoni Land, Bonny Area	
Overexploitation	Eket Area, Ogoni Land, Bonny Area	Opobo Area, Barrier Island between Dodo and Nun Rivers, Brass area
Illegal coastal sand mining	Lagos Island and Badagry area	Ibeju-Lekki, Maroko, Ajido, Badagry.
Environmental changes caused by floods, sea-level etc.	Lagos Island area	Bar-Beach, Victoria Island, Ikoyi
Illegal unreported unregulated (IUU) fishing	Nigerian coastal area	
Coastal erosion	Victoria beach, Awoye/Molume, Escravos/Ugborodo, Forcados, Brass, Bonny, Ibeno-Eket, Ikot-Abasi	
	Ondo coastline (Aiyetoro)	
Sea piracy in the Gulf of Guinea	Nigerian coastal area	Lagos, Warri, Calabar, Port Harcourt area

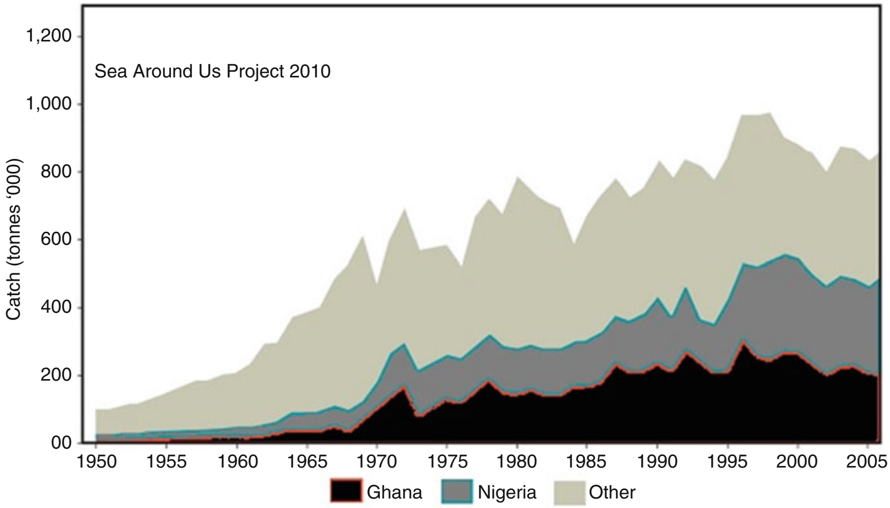

Fig. 5 Fish landings from the Guinea current large marine ecosystem (West Africa) by Ghana, Nigeria and others (Source: http://www.seaaroundus.org/lme/28/4.aspx)

over-fishing (ACOPS 2002b). Biodiversity has also suffered great loss as a result of anthropogenic impacts through overexploitation, habitat damage and pollution while coastal mangroves and wetlands have been destroyed to make way for development and settlement expansion. These coastal issues in Ghana are enhanced by the lack of research vessels and basic research facilities and equipment to enable quality coastal research for sustainable management (ACOPS 2002b; Aboudha 2009).

The overexploitation of mangroves in Ghana, as for example in the Ada-Volta Delta Anyanui Estuary Mangrove Complex (AVDEAMC) and the damming on the Volta River have caused sediment transport changes and thereby erosion and destruction of key species habitat (ACOPS 2002a; Aboudha 2009). The mangrove cover was estimated to be 0.2 km^2 based on 1986 aerial photo cover. Recent estimates, however, put the mangrove cover at 0.16 km^2. This trend suggests that the mangrove stands have diminished over the past two decades (ACOPS 2002a). Most of the mangroves have been lost through exploitation for fuel-wood and conversion of the habitats for solar salt production. In many instances, former mangrove habitats have been reduced to saline grasslands. Similar threats to mangroves exist in several countries in the western coast of Africa such as Gambia and Senegal (ACOPS 2002a).

In Nigeria, the mangroves of the Niger delta, estimated to cover approximately 7,000 km^2 comprise a significant regional resource, with fishing being a major activity. The pressure of a subsistence population has adversely affected the mangroves which have increased since the discovery of hydrocarbon reserves in the mid-1950s in and around the Niger delta. Nigeria currently produces around 1.6 million barrels per day from more than 4,000 oil wells spread within the Niger delta and adjacent coastal areas. 23 out of 62 oil fields are within the mangrove ecosystem (ACOPS 2002a; Abuodha 2009). Oil terminals are spread throughout the delta while 8,000 km of seismic lines (20–30 m wide) and oil pipelines criss-cross the mangrove ecosystem. Oil spills are common; between 1970 and 1982 alone, there were 1,581 oil spills involving a total two million barrels (Abuodha 2009). While most of the oil spills have been small, they have tended to occur within the mangrove waterways. As a result, many of the surface waters are contaminated and undrinkable, localised fisheries production has declined and in many instances, inhabitants have been forced to immigrate to other areas (ACOPS 2002a). In addition, bottom trawling, use of explosives and chemicals, and use of wrong mesh sizes have been recognized as major causes of destruction to the fishing environment.

3.3 Coastal Tourism

Several countries in Sub-Saharan Africa are turning towards tourism as a viable option for economic growth. West Africa accounts for a very small share of Africa's coastal tourism market compared to Eastern and Southern Africa. Nevertheless, the industry makes a significant contribution to the economies of the West African region. However, forecast sea-level rise and intensified storm surges will seriously threaten this growth. Over the course of the twenty-first century, Ghana may lose more than 50 % of its coastal GDP, while GDP loss in absolute terms would be highest in Nigeria (US$ 408 million). About 100 % proportion of coastal agriculture, in terms of extent of croplands, will be affected in Nigeria and 67 % proportion in Ghana (Dasgupta et al. 2009a, b; Abuodha 2009).

In Nigeria, the total coastal zone land loss due to a 0.2 m sea-level rise is estimated to be over 3,000 km^2 resulting in 800,000 people being displaced (Fig. 6). Such adverse impacts will affect residential, commercial and tourist facilities on the Victoria, Ikoyi and Lagos Islands costing over US $ 12 billion in land loss (ACOPS 2002c).

3.4 Fresh Water, Food Security, Pollution and Sanitation

In Ghana, sources of pollution include municipal and industrial waste, chemical runoff from agriculture activities, and oil spillage. Groundwater abstraction for irrigation has resulted in significant saltwater intrusion into aquifers (ACOPS 2002b). In most of the coastal urban centres only a very small part of the population is connected to sewage. In Ghana, untreated sewage is discharged into Korle lagoon which has rendered the lagoon unfit for any economic use (Abuodha 2009). Also, municipal or domestic input is the most common source of solid waste into the coastal environment. Fundamental causes include poverty and population pressure. Other concerns include low private sector participation in the provision of sanitation facilities. The waste, composed of 70–80 % organic matter, originates from households, markets, transport termini, restaurants schools and hospitals and contains, among others, plastics, food leftovers, paper, metals, glass, textiles, excreta, grass and wood cuttings, batteries and construction waste (Abuodha 2009). In Accra, the environmental impacts on the Korle Lagoon and its catchment are gross pollution and changes and losses in biodiversity, including fish

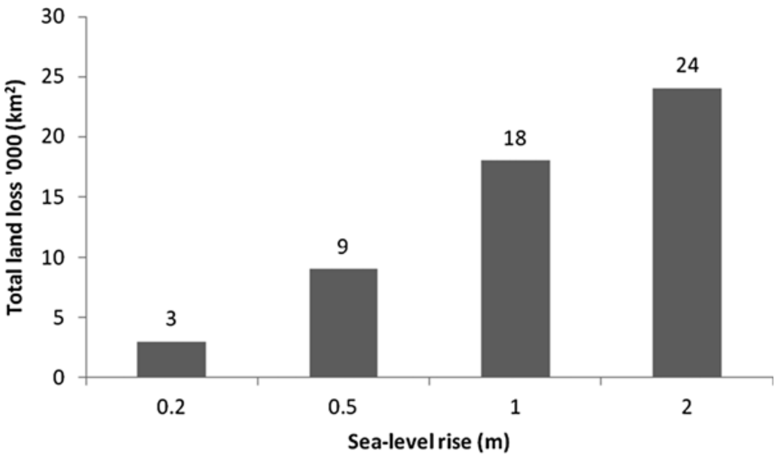

Fig. 6 Total land loss due to erosion and inundation at different sea-level rise scenarios in Nigeria (Data source ACOPS 2002c)

species and invertebrates. The main socio-economic impacts are increased diseases and loss of property and deaths arising from flooding events. There are also negative impacts on tourism (ACOPS 2002a). In addition, the Densu Delta, which is a Ramsar Site because of its important bird population and other biodiversity characteristics, is also undergoing rapid degradation because of improper land use and water pollution activities in the river basin (Abuodha 2009). As a result of the dumping of waste and other pollutants into the lagoon and its riverine system the Government of Ghana has been compelled to commit over US$ 40 million for restoration of the Korle Lagoon.

While 80 % of households in Nigeria's coastal cities are connected to sewage (ACOPS 2002a), major coastal towns and cities in Nigeria such as Lagos, Warri and Port Harcourt have large human populations that invariably lack sewage treatment plants except in a few relatively new and isolated residential or industrial estates. Most residents use septic tanks whose contents when dislodged are discharged into coastal rivers, lagoons and near shore waters without further treatment. The associated problems include increases in BOD and the introduction of pathogenic micro-organisms and intestinal parasites which pose risks to swimmers and fishermen as well as the general public (ACOPS 2002a). Solid waste constitutes a major environmental problem in the coastal areas especially in major coastal cities like Lagos, Warri and Port Harcourt. Due to the rapid increase in the coastal population, the volumes of solid waste generated by residents have quadrupled in recent years. In the Lagos Islands and other areas, human excrement is sometimes associated with solid waste dumps hence introducing health problems normally associated with human wastes (ACOPS 2002a). Poor waste management policies and practices, inefficient collection and disposal as well as insufficient awareness and negative attitudes to the environment are some of the causes. The environmental impacts include contamination of ground water due to leachates from solid waste dumps which reduces availability of fresh water. The fiscal implication directly related to solid waste clearing for example in the Lagos runs to about US$ 10,000 per day (ACOPS 2002a; Abuodha 2009).

4 Impacts of Pressures on Coastal Ecosystem Goods and Services

Impacts of pressure on coastal zones in Ghana are presented in Box 1 and Fig. 7. In Ghana, there are six key issues critical to the coastal zone and these are: erosion, pollution, and impacts of crop production, impact of fisheries, biodiversity loss and habitat loss (ACOPS 2002b). The key issues identified in the hot spot and sensitive areas in Nigeria (Fig. 8) were (1) modification of ecosystems from coastal erosion, flooding (Box 2), deforestation (2) pollution from oil spills, solid wastes, sewage and industrial effluents and (3) global climate change and sea-level rise (UNESCO-IOC 2009; ACOPS 2002c). The Nigerian coastal environment has a variety of both living and non-living resources, which account for almost 90 % of its economic growth.

Box 1: Coastal Erosion Along the Coasts of Ghana

In Ghana, 25 locations along the coast have been identified to undergo critical erosion (ACOPS 2002a). Coastal erosion in Ghana is due mainly to the destruction of coconut trees at Cape St Paul as well as sand and pebble mining, wave action and construction of dams. Coastal erosion in Ghana was identified as a major feature of the shoreline especially on the eastern shores in the Ada-Volta Delta Anyanui Estuary Mangrove Complex (AVDEAMC) (ACOPS 2002a). The erosion is enhanced by mangrove overexploitation, causing erosion in the delta area of the Volta River. Erosion is also destroying turtle nesting sites and also exposes the eggs to predators such as dogs, pigs and humans (ACOPS 2002a). Following the damming of the Volta River, the result of which cut off substantial amounts of sediments that reach the littoral zone, erosion has become of critical concern averaging about 2–3 m/year in recent times (ACOPS 2002a). It is estimated that the recession in the Keta area has increased from 4 m/year before the construction of the dam on the Volta River in 1965 to 8 m/year after the dam construction. This is among the highest rates of coastal erosion in Ghana. The Loggerhead turtle, *Caretta caretta*, could be described as highly endangered in Ghanaian waters in the Keta Lagoon Complex hot spot and the East Central Sandy Coast sensitive area where the rate of coastal retreat is estimated at 3 m per year (ACOPS 2002a).

Past and existing management interventions along Ghana's shorelines and rivers were based on site-specific and ad-hoc interventions without proper analysis and assessment of impacts on other sections of the shoreline. This has resulted in increased coastal erosion and other coastal management problems in Ghana. Although some of the management techniques have been successful, they have resulted in initiating erosion along their down-drift side.

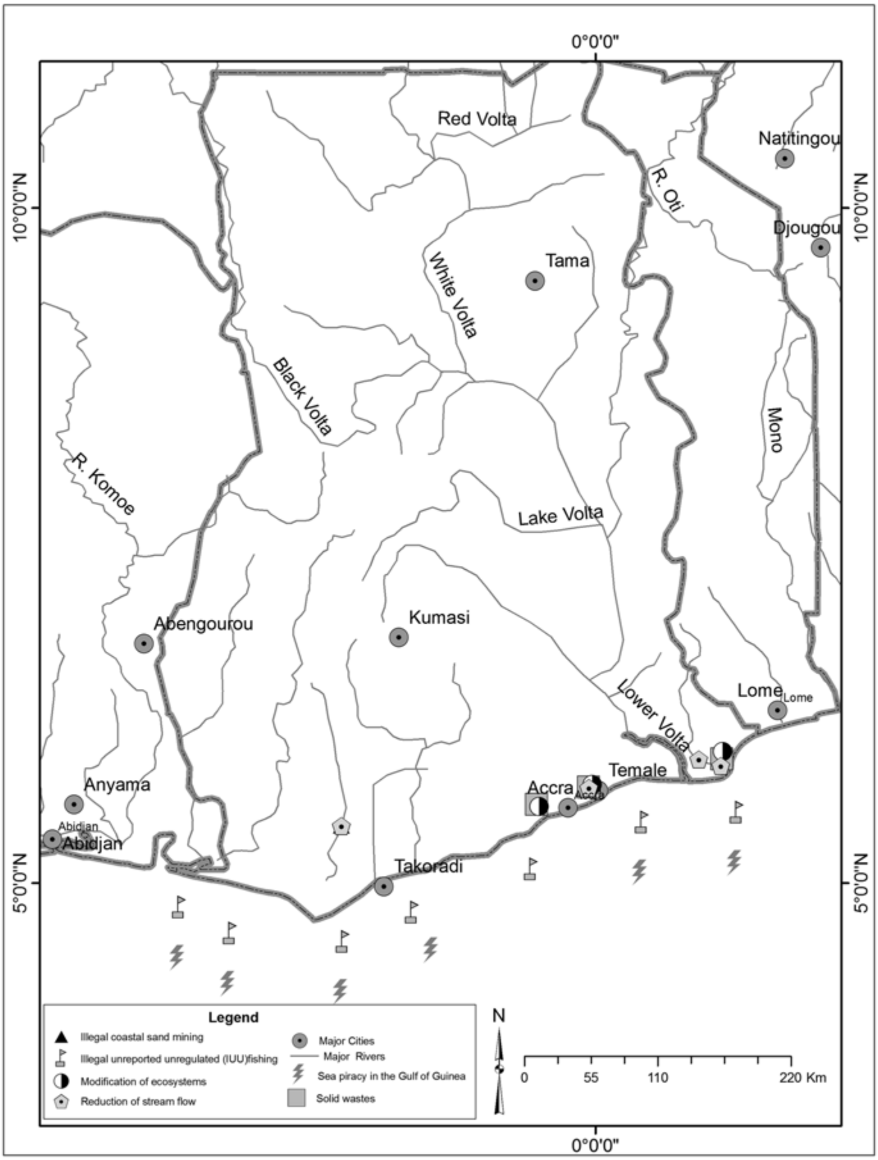

Fig. 7 Ghana sensitive locations

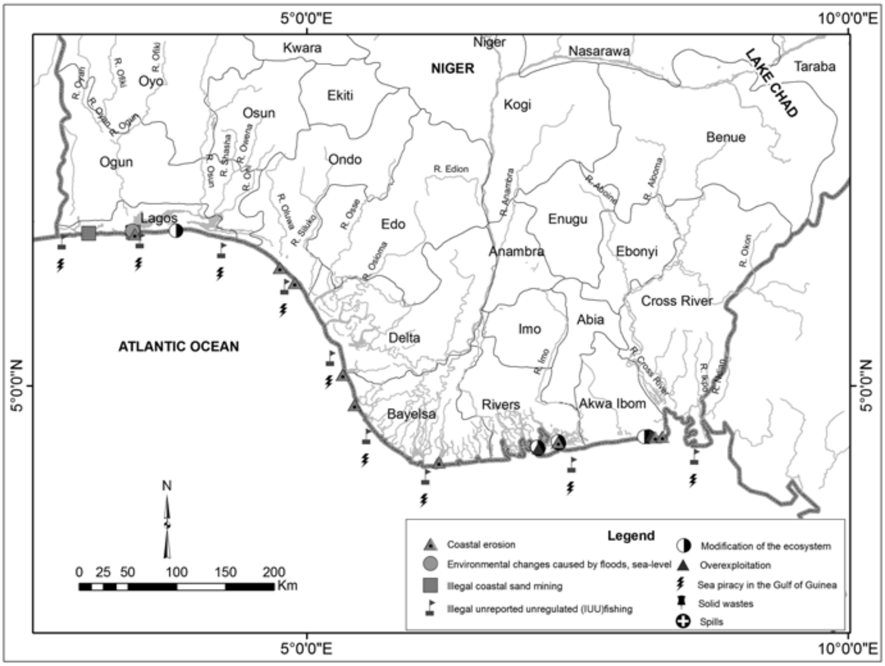

Fig. 8 Nigeria hotspot locations

4.1 Direct and Indirect Drivers of Change in Coastal Ecosystems and Services

As earlier stated, sea-level rise poses a particular threat to deltaic environments, especially with the synergistic effects of other climate and human pressures.

4.1.1 Sea-Level Rise and Coastal Erosion

Coastal settlements in the Gulf of Guinea are particularly prone to inundation. Sea-level rise resulting from global climate change threatens different coastal and marine ecosystems e.g. lagoons and mangrove forests of both Nigeria and Ghana in Western Africa (Cugusi and Piccarozzi 2009).

Box 2: Coastal Erosion Along the Lagos Coast, Nigeria- Amadu Bello Way Along Victoria Beach Prior to Eko Atlantic City Intervention

In Nigeria, coastal erosion results from the modification of ecosystems. Affected sites include Eket, Lagos, Forcados and Ondo (ACOPS 2002a). The Victoria Beach is the fastest eroding beach in Nigeria with average erosion rates of 20–30 m/year. From 1900 to 1959, Victoria Beach retreated by over 1 km (ACOPS 2002c). Annual erosion rates of 25–30 m had been reported between 1981 and 1985 for Victoria Beach. This high rate of erosion has been linked to the construction of the moles built to stop the silting up of the entrance to Lagos harbour (ACOPS 2002c). However, the Lighthouse Beach near the western breakwater accreted by over 500 m within the same period. The average mean sea level for Nigeria between 1960 and 1970 was estimated to be 0.46 m above the zero of the tide gauge (ACOPS 2002c).

The emerging of the Eko Atlantic City project, starting with the reclamation work of the Victoria Beach in 2008 has totally changed the situation. The totality of the Eko Atlantic city project with its Great Wall of Lagos designed to withstand the worst imaginable Atlantic storm is already helping to reverse the environmental and physical damages at Victoria Island caused by century of coastal erosion. "The great wall of Lagos is an offshore sea wall that is built to protect the city of Lagos and the new city of Eko Atlantic. The sea wall is designed to deflect the threat of flooding from the Atlantic Ocean".

4.1.2 Precipitation and Floods

The beaches along the Nigerian coastline are very susceptible to flooding due to their very low topography. Low-lying beaches like the Bar Beach and Mahin Mud Beach are easily flooded during high tides as they are almost at sea level. Whenever storm surges coincide with spring tides most beaches up to a maximum elevation of 3 m above sea level are usually topped by waves resulting in floods (ACOPS 2002c; Cugusi and Piccarozzi 2009). The low drainage heads of existing storm drainage channels increase the severity of flooding.

The flooding of the Victoria Island in August 1992, July 1995, April 1996 and May 1996 show that the height of the highest swells reaching the coast average about 2 m above normal high tide levels (ACOPS 2002c). The August 1984 and May 1990 storm surges resulted in the topping of the beach ridge along Victoria Island and flooding along most parts of the low-lying Nigerian coastline. Thus flooding which has characterised the Mahin mud coast has exacerbated the erosion problem along the Awoye Molume areas (ACOPS 2002c).

Results of the sea-level rise video mapping vulnerability analysis survey of the Nigerian coastal zone show that the barrier lagoon coastline in Lagos State could lose well over 284–584 km^2 of land from erosion and inundation arising from sea-level rises of 0.5 m and 1 m, respectively by the end of the twenty-first century (ACOPS 2002c; Abuodha 2009). On the Mahin Mud coast and the Niger Delta, native vegetation has died due to increase in saltwater levels and has been replaced by more salt tolerant vegetation like grasses (ACOPS 2002c).

4.2 Analysis of Drivers

Drivers of change in coastal watershed ecosystems goods and services (especially aquatic living resources) in West Africa (Ghana and Nigeria) were identified through critical review of literature and interviews of expert opinions (Table 3). Plausible descriptions of how the future may develop based on a coherent and internally consistent set of assumptions about key relationships and driving forces is presented in Table 3. The critical drivers identified are: "climate change" and "socio-economic factors". Climate change (including climate variability) refers mainly to changes in upwelling in the Gulf of Guinea resulting in changes in species migratory patterns and survival of certain aquatic species (Badjeck et al. 2011). Socio-economic refers to expanding or recessing economies, commercial and industrial development, market development and infrastructure, changes in social, human development, cultural (values, consumer preferences) and economic systems (markets) (Badjeck et al. 2011). Inevitable drivers are drivers that have an important impact and require management attention but are less uncertain while critical drivers have a high uncertainty and high importance.

Table 3 Current drivers of change in key ecosystem good and service

Country	Ecosystem good and service	Drivers of change: Inevitable	Critical	Others	Remark
Ghana	Fishery resources	Management	Climate change	Population growth	Uncertainty and importance that warrant strategic planning
		Technology	Socio-economic	Regional integration	
		Research and development		Poverty	
		Environmental change			
	Coastal tourism	Environmental change	Climate change	Coastal erosion	Requires renewed drive for investments
		Management	Socio-economic		
		Pollution			
Nigeria	Fishery resources	Management	Climate change	Poverty	Uncertainty and importance that warrant strategic planning
		Technology	Socio-economic	Loss of habitat	
		Research and development			
		Environmental change			
		Population growth			
	Coastal tourism	Environmental change	Climate change	Coastal erosion	Requires renewed drive for investments
		Management	Socio-economic		
		Pollution			

5 Measures to Reduce or Mitigate Threats/Risks to Ecosystem Goods and Services

From the analysis of drivers presented in Sect. 4.2, critical drivers have a high uncertainty and high importance, and are those used to create the scenario cross in Fig. 9 (Badjeck et al. 2011). Here, it is the combination between uncertainty and importance that warrants strategic planning. Based on “climate change” and “socio-economic” critical drivers, four scenario logics peculiar to Ghana and Nigeria are presented (Fig. 9) for effective management of coastal ecosystem goods and services.

In Fig. 9, the vertical axis signifies that coastal upwelling could in the future evolve in two directions: a “steady upwelling” defined as an environment where upwelling patterns in the gulf of Guinea are similar to the present or easily predictable with models. “Erratic upwelling” refers to unstable upwelling patterns leading to unpredictable changes in abundance, composition and distribution of species. Coastal upwellings are wind–driven masses of cold nutrient rich waters replacing nutrient poor surface warm waters; and if reduced or altered may seriously affect the fisheries sector (Badjeck et al. 2011).

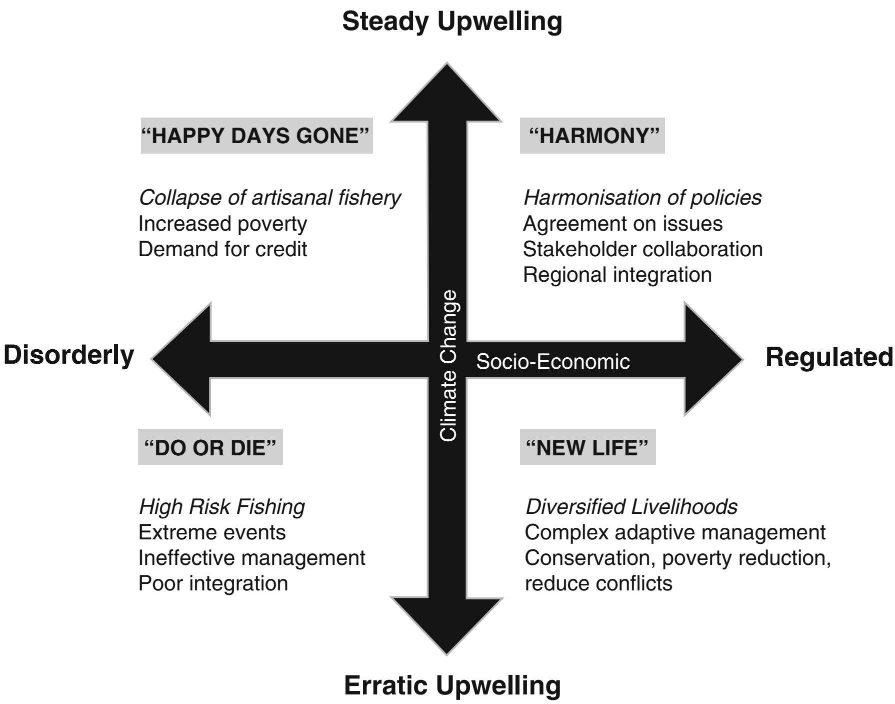

Fig. 9 Scenarios logic for management of coastal watershed living resources in Ghana and Nigeria (Adapted from Badjeck et al. 2011)

The horizontal axis is used to consider two extremes in socio-economic development: one where formal and informal institutions are not regulating society, more specifically the fisheries sector, and one where rules and norms are in place, enforced and effective.

Scenario 1 "Harmony": Harmonisation of Policies This requires multi-sectorial and participatory approach to aquatic living resources management. An inventory of national policies governing natural resources is created to inform planning and management thereby increasing synergies and avoiding conflict between sectors. This will equally entail strategic "Outreach and In-Reach Plans discussed in Agboola and Braimoh (2009)". In terms of research and development, data are readily available to implement ex-ante strategies to adapt to changes. Fisheries resources are spatially mobile, not bounded by national jurisdiction. Bilateral and regional structures and processes are promoted, especially in terms of monitoring, transboundary data collection and sharing, and technological transfer.

Scenario 2 "New Life": Diversified Livelihoods Due to an erratic upwelling, capture fisheries can no longer provide sustainable livelihoods. Diversification is needed, and immediate strategies are required to build communities' adaptive capacity to climate change. Policies should focus on the needs of people most affected by climate change impacts, and aim to improve their resilience in the long term. Because the livelihoods of people from Ghana and Nigeria who live along the coastal areas is tied to the fisheries sector, which may experience significant declines during erratic upwelling and changes in rainfall patterns, people opt to switch to a new life that will involve non-fishing activities. Diversified livelihoods, for instance the development of aquaculture, have positive impacts such as reduced conflicts among resource users, conservation benefits and improved fisheries management practices. If well addressed, diversification may boost poverty reduction strategies, for example through activities which can be accessed by women.

Scenario 3 "Do or Die": High Risk Fishing and Tourism Climate change and increased climate variability will bring higher sea levels, more intensive extreme events and is likely to increase the strength of winds. The number of risks faced by the fisheries and tourism industry in Ghana and Nigeria will increase; resulting in fewer fishing activities and many losses such as capsising of fishing vessels, gear damage and even deaths caused by accidents. Fishing and tourism activities will be impacted severely. Under such conditions, it will be difficult to enforce the regulations and rules governing fisheries; therefore investment will decrease and the speed of environmental degradation will increase. Some fisherfolk with modern gears will "weather the storm" while the majority will quit fishing.

Scenario 4 "Happy Days Are Gone": Collapse of Artisanal Fisheries The fishery of Ghana is seasonal in nature and it is closely associated with upwelling. During the period of steady upwelling, spawning and recruitment of fish stocks will be enhanced resulting in higher abundance and availability of fish stocks, especially pelagic ones such as sardinella. Fishing effort increases, leading to excessive fishing pressure that may cause fisheries to collapse. The livelihoods of dependent communities will therefore be severely impacted, as a result poverty will increase; and there will also be an increased demand for credits to venture into other fields, or even for daily sustenance. In this scenario, fish consumption will decrease and price of fish will likely increase, leading to food insecurity, fish being an inexpensive source of protein.

Overall, careful consideration of the applicability of these scenario logics to the dynamic changes in ecosystem structures and functions is very crucial to effectively manage coastal watershed ecosystem services in West Africa.

6 Conclusion

This chapter has traversed a number of issues on coastal watershed ecosystem services and management in West Africa with special focus on Ghana and Nigeria. Amongst others, it gave an overview of the current state of coastal ecosystem goods and services with particular reference to fishery resources and tourism, identifying major pressures and impacts on ecosystem goods and services. Numerous factors have accounted for change in coastal ecosystem goods and services and threaten sustainability of coastal ecosystems. Amongst others are: (1) lack of coordinated action; (2) lack of political commitment to conservation and management action; (3) high cost of monitoring and enforcing fisheries management regimes; (4) persistent illegal, unreported and unregulated (IUU) fishing activities; and (5) failure of states to comply with international and regional agreements. Consequent upon these driver factors of change, this paper further present four scenario logics peculiar to Ghana and Nigeria based on "climate change" and "socio-economic" critical drivers, for effective management of coastal ecosystem goods and services.

Although, there are some efforts to safeguard ecosystem services in Africa, realizing benefits to livelihoods still faces serious challenges due to climate change, recent land grabbing and urbanization. Unsustainable use of ecosystem services coupled with other challenges discussed in this paper will be exacerbated in the future if projected increases in population are realized (UNFPA 2007). Thus, the need for a more proactive approach in management cannot be overemphasized. It will be relevant to set up a Commission within the cover of Natural Resources Conservation Council (NARESCON) similar to the Committee of the Eastern Central Atlantic Fisheries (CECAF) and Fishery Committee of the West Central Gulf of Guinea (FCWC) in West African region, to proffer solution for the conservation and management of ecosystems goods and services and obviate a total collapse of our coastal ecosystems.

References

Abuodha PA (2009) The African science-base for coastal adaptation: a continental approach. A report to the African Union Commission AUC at the UN Climate Change Conference in Copenhagen (7–18 Dec 2009). UNESCO-IOC, November, 2009

ACOPS – Advisory Committee on Protection of the Sea (2002a) Portfolio of project proposals for the global environment facility project on development and protection of the coastal and marine environment in Sub-Saharan Africa (GF/6010- 0016). Jointly undertaken with UNESCO-IOC, UNEP and GPA, 336 pp

ACOPS – Advisory Committee on Protection of the Sea (2002b) Ghana national report: phase 1: integrated problem analysis. Authored by Wellens-Mensah J, Armah AK, Amlalo DS, Tetteh K, for the global environment facility project development and protection of the coastal and marine environment in Sub-Saharan Africa (GF/6010-0016), jointly undertaken with UNESCO-IOC, UNEP, and GPA, 94 pp

ACOPS – Advisory Committee on Protection of the Sea (2002c) Nigeria national report: phase 1: integrated problem analysis. Authored by Awosika L, Osuntogun N, Oyewo E, Awobamise A, for the global environment facility project development and protection of the coastal and marine environment in Sub-Saharan Africa (GF/6010-0016), jointly undertaken with UNESCO-IOC, UNEP, and GPA, 80 pp

Agboola JI, Braimoh AK (2009) Strategic partnership for sustainable management of aquatic resources. Water Resour Manag 23(13):2761–2775

Alder J, Sumaila UR (2004) Western Africa: a fish basket of Europe past and present. J Environ Dev 13(2):156–178

Badjeck M-C, Katikiro RE, Flitner M, Diop N, Schwerdtner Máñez K (2011) Envisioning 2050: climate change, aquaculture and fisheries in west Africa. Dakar, Senegal 14-16th April 2010, Workshop report No. 2011–09. WorldFish/ZMT, Penang/Bremen

Benneh G, Dickson BK (1988) A new geography of Ghana. Longman Group UK Ltd., London

Boateng I (2006) Shoreline management planning: can it benefit Ghana? A case study of UK SMPs and their potential relevance in Ghana. In: Marine and coastal zone management-environmental planning issues, Proceeding, Promoting Land Administration and Good Governance Regional Conference, Accra, Ghana

Bowen-Jones E, Brown D, Robinson EJ (2003) Economic commodity or environ- mental crisis? An interdisciplinary approach to analyzing the bush-meat trade in central and West Africa. Area 35:390–402

Costanza R (2008) Ecosystem services: multiple classification systems are needed. Biol Conserv 141:350–352

Cugusi B, Piccarozzi M (2009) Climate change and human mobility in Africa. A thematic literature and organisational review. Prepared for the international conference towards the G8 summit. CeSPI working papers 59/2009, 57 pp

Daily GC (1997) Nature's services: societal dependence on natural ecosystems. Island Press, Washington, DC

Dasgupta S, Laplante B, Murray S, Wheeler D (2009a) Sea-level rise and storm surges: a comparative analysis of impacts in developing countries, Policy research working paper series 4901, The World Bank, 41 pp

Dasgupta S, Laplante B, Murray S, Wheeler D (2009b) Climate change and the future impacts of storm-surge disasters in developing countries. Working paper 182. Centre for Global Development, 26 pp

Egoh BN, O'Farrell PJ, Charef A, Gurney LJ, Koellner T, Abi HN, Egoh M, Willemen L (2012) An African account of ecosystem service provision: use, threats and policy options for sustainable livelihoods. Ecosyst Serv 2:71–81

FAO (2007) The state of the world fisheries and aquaculture. FAO, Rome

FAO fishery country profile, viewed 14 Oct 2012. http://www.fao.org/countryprofiles/default/en/

Koranteng KA (1998) The impacts of environmental forcing on the dynamics of demersal fishery resources of Ghana. PhD thesis, University of Warwick (U.K.), 376 p

Koranteng KA (2002) Status of demersal fishery on the inner continental shelf of Ghana, pp 261–274. In: McGlade JM, Cury P, Koranteng KA, Hardman-Mountford NJ (eds) The Gulf of Guinea large marine ecosystem: environmental forcing and sustainable development of marine resources. Elsevier, 392 p

Koranteng K, Pauly D (2004) Long-term trends in demersal fishery resources of Ghana in response to fishing pressure. In: Palomares MLD, Pauly D (eds) West African marine ecosystems: models and fisheries impacts, Fisheries centre research reports 12(7). Fisheries Centre, UBC, Vancouver, pp 75–80

Layke C (2009) Measuring nature's benefits: a preliminary roadmap for improving ecosystem service indicators, WRI working paper. World Resources Institute, Washington, DC. Available online at http://www.wri.org/project/ ecosystem-service-indicators

Loeb SL (1994) An ecological context for biological monitoring. In: Loeb SL, Spacie A (eds) Biological monitoring of aquatic systems. Lewis Publishers, Ann Abor, pp 3–7

Ly CK (1980) The role of the Akosombo dam on the Volta River in causing coastal erosion in central and eastern Ghana (West Africa). Mar Geol 37:323–332

Millennium Ecosystem Assessment (MEA) (2005) Ecosystems and human well-being: synthesis. Island Press, Washington, p 155

Naiman RJ, Pieur-Richard A-H, Arthington A, Dudgeon D, Gessner MO, Kawabata Z, Knowler D, O'Keefe J, Loque C, Soto D, Stiassny M, Sullivan C (2006) Freshwater biodiversity: challenges for freshwater biodiversity research, DIVERSITAS report No. 5. 48 pages

Nicholls RJ, Wong PP, Burkett VR, Codignotto JO, Hay JE, McLean RF, Ragoonaden S, Woodroffe CD (2007) Coastal systems and low-lying areas. In: Parry ML, Canziani OF, Palutikof JP, Van Der Linden PJ, Hanson CE (eds) Climate change 2007: impacts, adaptation and vulnerability, contribution of working group II to the fourth assessment report of the intergovernmental panel on climate change. Cambridge University Press, Cambridge, pp 315–356

Odada EO, Scholes RJ, Noone KJ, Mbow C, Ochola WO (2008) A strategy for global environmental change research in Africa: science plan and implementation strategy

OECD (1999) OECD environmental data compendium. Organization for Economic Cooperation and Development, Paris

Oyebande L, Amani A, Mahé G, Diop IN (2002) Climate change, water and wetlands in West Africa: building linkages for their integrated management, Special commissioned paper. IUCN-BRAO, Ouagadougou

Palomares MLD, Pauly D (2004) West African marine ecosystems: models and fisheries impacts, Fisheries centre research reports 12(7). Fisheries Centre, University of British Columbia, Canada

Pauly D, Christensen V, Guénette S, Pitcher TJ, Sumaila UR, Walters CJ, Watson R, Zeller D (2002) Toward sustainability in world fisheries. Nature 418:689–695

Rebelo LM, McCartneyand MP, Finlayson CM (2010) Wetlands of Sub-Saharan Africa: distribution and contribution of agriculture to livelihoods. Wetl Ecol Manag 8:557–572

Sexton WJ, Murday M (1994) The morphology and sediment characteristic of the coastline of Nigeria- the Niger delta. J Coast Res 10(4):959–977

UNEP (2006) Second Africa environment outlook: our wealth our future. UNEP, Earthscan, London

UNEP, WRC (2008) Freshwater threat: vulnerability assessment of freshwater resources to environmental change-Africa. WRC, Gezina, South Africa

UNESCO-IOC (2009) African oceans and coasts. In: Odido M, Mazzilli S (eds) IOC Information Document, 1255, UNESCO Regional Bureau for Science and Technology in Africa, Kenya, 163 pp

UNFPA (2007) State of the world population 2007. Unleashing the potential of urban growth. United Nations Population Fund, New York

Web, Lagos State (2010) Viewed 14 Oct 2010. http://www.lagosstate.gov.ng/

Web, Lagos State (2011) Viewed 12 Nov 2011. http://www.lagosstate.gov.ng/

Management of Agriculture to Preserve Environmental Values of the Great Barrier Reef, Australia

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Abstract Coral reef and seagrass ecosystems of the Great Barrier Reef (GBR) are in severe decline. Water quality associated with pollutant discharge from the rivers discharging into the GBR is a major issue for these GBR ecosystems and associated species such as dugongs, turtles and fish. The main source of river pollution is agriculture with sugarcane cultivation, beef grazing, grain cropping and horticulture the principal industries. Discharge to the GBR is of poor quality in many rivers, contaminants are present in the GBR lagoon at concentrations likely to cause environmental harm and the causal relationship between poor water quality and declining GBR ecosystem health is well understood. Action to improve management practices to reduce sediment, fertiliser and pesticide losses from farms is being taken and the pollutant loading of river discharge reduced. Improved practices are funded through the combined efforts of Australian Governments (Federal, State and local) and farmers. Whether these improved practices and the pollution reductions achieved are sufficient to improve GBR ecosystem health is not certain in the face of other threats to the GBR such as climate change and large scale coastal development associated with urban and port expansion.

Keywords Watershed management • Coral reefs • Agricultural pollution • Nutrients • Pesticides

1 Introduction

Globally most coral reefs are threatened by human activities (Burke et al. 2011) and show signs of some level of degradation (Pandolfi et al. 2003). Reefs are exposed to a combination of stresses including destructive fishing practices; overfishing;

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land-sourced pollution of sediment, nutrients, pesticides, toxic metals and toxic synthetic organic chemicals; coral predator outbreaks linked to trophic changes in the system, particularly the crown of thorns starfish; bleaching resulting from global climate change; ocean acidification; and increased incidence of and severity of coral diseases (Halpern et al. 2008). These anthropogenic stresses have led to severe declines in coral cover on most global coral reef areas from values near 60 % more than 50 years ago to 5–20 % recently, and led to persistent shifts from coral dominance to non-coral and algal dominance (Bruno and Selig 2007; Hughes et al. 2010). Seagrass meadows are in an equally threatened state globally (Unsworth et al. 2014) including the Western Pacific (Short et al. 2014). Many of the stresses affecting tropical seagrasses are the same as for coral reefs – e.g. land-sourced pollution and climate change.

A large range of approaches to manage the coastal zone have been developed including Integrated Coastal Zone Management (ICZM), Ecosystem Based Management (EBM), Marine Protected Areas (MPAs) and Integrated Marine (and Spatial) Planning (IMP). In addition there are many approaches to managing catchments (watersheds) which can be grouped into 'Integrated Catchment Management'. A few approaches also address the catchment – marine waters continuum and managing the land-sea boundary (e.g. Alvarez-Romero et al. 2011). In many cases, there are deficiencies in Marine Protected Area (MPA) management around the world in terms of size, spatial planning, representativeness, focus on limited impacts and lack of enforcement (Mora et al. 2006; Christie and White 2007; Osmond et al. 2010) and for ignoring the need for complementary terrestrial pollutant management. The Great Barrier Reef (GBR) on the other hand is generally seen as the best example of Ecosystem Based Management (EBM) (e.g. Ruckelshaus et al. 2008), MPA design and implementation (Fernandes et al. 2005; Agardy et al. 2011), Integrated Marine Planning (Dickinson et al. 2010) and to some extent combined ICZM and MPA design (Douvere 2008; Nobre 2011) with well-designed governance structures (Dale et al. 2013). However recently, with the severe declines in coral cover, seagrass health and dugong populations, it is more evident that management of the GBR has also failed (Brodie and Waterhouse 2012).

The GBR is an extensive coral reef system lying off the north east Australian coast (Fig. 1) which also contains high value areas of seagrass and mangroves, and a range of iconic megafauna including whales, dugongs, turtles, sharks, dolphins and large fish. The area is 344,000 km^2 with around 25,000 km^2 of coral reefs (Day and Dobbs 2013) and an adjacent catchment area of 400,000 km^2 (Brodie et al. 2012). The GBR has been managed as a national Marine Park since 1975 (*Great Barrier Reef Marine Park Act 1975*), and listed as a World Heritage Area (WHA) in 1981 (Lawrence et al. 2002). The GBR has been subject to an intensive management regime involving both the Australian and Queensland State Governments for 40 years focussing on managed use and ecosystem protection. The actual Marine Park falls under Australian Government jurisdiction but the adjacent catchments are within the jurisdiction of the Queensland State Government. These jurisdictional factors lead to political issues when adopting an ecosystem based approach to management and in addressing land based impacts.

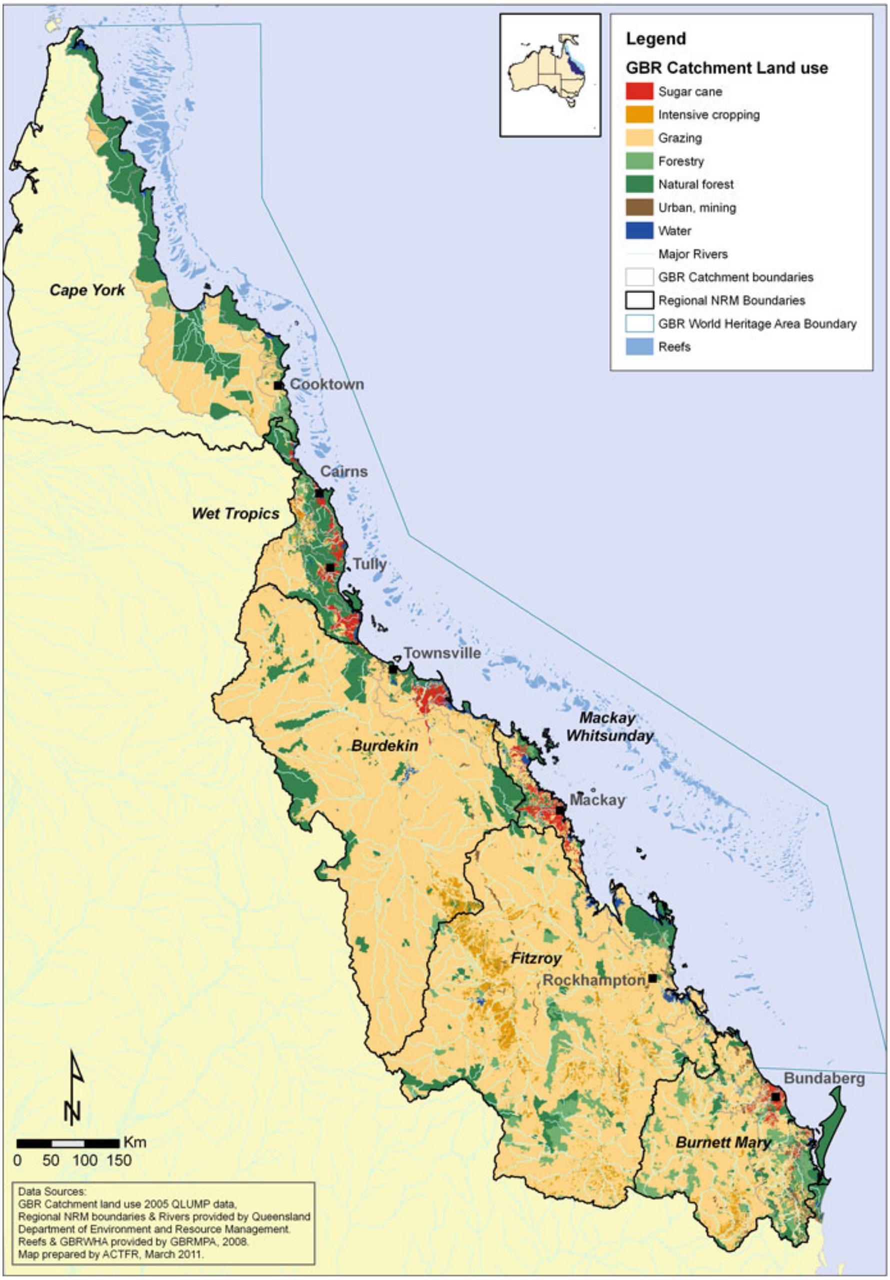

Fig. 1 The Great Barrier Reef showing the reefs, catchments and major rivers, land uses on the catchments and major cities and towns

Live coral cover on the mid shelf of the GBR has sharply declined from levels near 50 to 60 % in the 1960s to less than 14 % currently (Hughes et al. 2011; De'ath et al. 2012). The causes of the decline include: land-sourced pollution of sediment, nutrients (with the associated crown-of-thorns starfish outbreaks), and pesticides (Brodie et al. 2005, 2011; De'ath and Fabricius 2010; Fabricius et al. 2005, 2010; Lewis et al. 2009); coral bleaching/mortality and physical damage associated with climate change including increasing incidence of severe storms (cyclones) (Osborne et al. 2011; De'ath et al. 2012); ocean acidification (Cooper et al. 2008; De'ath et al. 2009); and coral diseases (Haapkylä et al. 2011). Similar declines have been observed on inshore reefs of the GBR, although the monitoring record is much shorter (Thompson et al. 2011). A recent series of large river discharge events have caused acute mortality to coastal reefs associated with the low salinity (Jones and Berkelmans 2014) and polluted water (Devlin et al. 2012a).

Seagrass health and abundance is also under anthropogenic threat (Grech et al. 2011), with recent declines associated with large river discharge events of polluted water (Petus et al. 2014) and severe cyclones (Rasheed et al. 2014). The 2010/11 major river discharge events from GBR rivers associated with the strong La Nina and Tropical Cyclone Tasha intense rainfall and the physical effects of Category 5 Tropical Cyclone Yasi have had devastating effects on large areas of GBR seagrass (Devlin et al. 2012a) and subsequent increased mortality of dugongs and turtles which depend on the seagrass for food. Populations of dugongs in the southern two thirds of the GBR have declined over recent decades with numbers reducing at a rate of 8.7 % a year between 1962 and 1999 from about 72,000 in the early 1960s to 4,000 in the mid-1990s (Marsh et al. 2005). Increased dugong mortality coincided with severe weather events of 2011 and 2013 and, in turn is linked to reduction in seagrass health and biomass from poor water quality.

Agriculture is the major land use on GBR catchments, with more than 90 % of the total area used for beef grazing and sugarcane, grains, horticultural and cotton cropping. Much of the cropping activity occurs on the coastal floodplains (Brodie et al. 2012). The proximity of the high conservation value GBR to this agriculture dominated catchment has raised concern as to the risk posed to the GBR from agricultural-sourced pollution.

The GBR, however, receives not only pressures from adjoining catchments, but is also ecologically dependent upon many of the upstream environments and processes they support (Stoeckl et al. 2011). For example, coastal freshwater riverine systems, wetlands, and mangrove ecosystems along the Queensland coast provide habitat, nutrient and sediment cycling and trophic linkages that are vital to the GBR. Degradation of these linkages erodes the ongoing functional integrity of the GBR, with spillover effects upon ecosystem benefits such as tourism and fisheries, which propagate through to the social, ecological and economic systems that are interdependent with them (Thomas et al. 2012).

Even from a purely economic perspective, the GBR is enormously valuable. For example, the monetary value of ecosystem benefits provided by coral reefs and coastal systems globally has been estimated to be worth over 2 billion (international) dollars per hectare per year (de Groot et al. 2012). Estimates of economic value for the GBR reveal the significance of this asset at the national level. For example, an early study assessed the present value of the GBR at approximately 4.7 % of Australia's annual gross domestic product (Oxford Economics 2009). The direct economic contribution of the GBR to commercial expenditure was recently estimated at just over AU$7 billion, of which tourism contributed AU$6.4b, recreation AU$330 m, and commercial fishing AU$190 m (Deloitte Access Economics 2013).

Tourism is a substantial industry in GBR catchments, most notably in the Cairns/Wet Tropics region. Snapshot studies of the tourism industry reveal the breadth and diversity of the ecological structures and processes that support it. For example, Stoeckl et al. (2010) report that each year, live-aboard dive boats are directly responsible for generating at least AU$16 million worth of income in the Cairns/Port Douglas region. Similarly, the annual value of tourism expenditure exclusively attributable to whale-watching in Hervey Bay is over AU$7 m, and seasonally this industry contributes approximately AU$30 m to the region each year (Knowles and Campbell 2011; Wilson and Tisdell 2003).

The total recreational value of Australian coral reefs, including fishing, is approximately US$120/visitor (Brander et al. 2007). The fishing component of recreational reef trips can be significant. For example, Prayaga et al. (2010) calculated the consumer surplus per trip on the Capricorn Coast at AU$385.34 per (group) trip, or approximately AU$5.53 m for this region of the GBR alone. Similarly, earlier work by Fenton and Marshall (2001a) reveals the total annual gross value of production (GVP) for GBR charter fishing businesses at approximately AU$23 m. By contrast, the same study showed that annual GVP for commercial fishing businesses at that time was AU$224 m (Fenton and Marshall 2001b).

Like tourism, commercial fishing in the GBR is diverse, and many species are dependent on seagrass meadows for substantial parts of their life cycle. Although few studies have examined the economic contribution of GBR seagrasses to fishery values, the loss in 1995/96 of 12 700 ha of seagrasses in Australia has been associated with lost fishery production of AU$235 000 (McArthur and Boland 2006). In contrast, international estimates have valued the provision of mangrove wood and fish nursery areas by mangroves and seagrasses at US$215,000 per hectare (Thorhaug 1990).

These figures underrepresent the value of the GBR because they do not consider indirect economic value or many non-use values. It has been suggested that the indirect benefits to coastal protection afforded by GBR ecosystems are worth at least $10 billion, and Australian non-use values may be in the order of $15.2 billion (Oxford Economics 2009). Many non-use values remain unquantified (Brander et al. 2007). For example, ecotourists visiting Hervey Bay sea turtles and whales are willing to pay AU$2 – 8 m per year to safeguard the survival of the species (Wilson and Tisdell 2003). It is unclear whether these and similar benefits are being realised.

The functional integrity of GBR ecosystems strongly depends on the decisions that are made by the agricultural sector. The social, economic, and ecological risks and benefits of agricultural activities can be difficult to formulate and implement because they arise externally to the GBR (i.e., "upstream"), and as such are often resolved externally. A systematic ecosystem-based approach will help clarify and quantify important interdependencies that affect system functionality, and can underpin effective policy formulation.

2 Agricultural Pollution and the GBR

Important land uses in the GBR catchment (Fig. 1) that cover a total area of 424,000 km^2 include rangeland beef grazing (314,000 km^2), sugarcane cultivation (5,700 km^2), horticulture (630 km^2), other cropping including grain and cotton cultivation (11,600 km^2), urban areas (2,600 km^2) and native forest (55,900 km^2) (Waterhouse et al. 2009). Discharges of suspended solids, nutrients and pesticides to the GBR have increased greatly over the last 150 years (Kroon et al. 2012) due to this extensive development (Waterhouse et al. 2012).

Mean annual suspended sediment (SS) loads to the GBR has increased by 5.5 times to 17,000 ktonnes/year (Kroon et al. 2012) since European settlement in the GBR catchment area (GBRCA) (c. 1850). The large beef grazing dominated catchments of the Fitzroy and Burdekin contribute over 50 % (7,400 ktonnes/year) to the mean annual anthropogenic (human caused) SS load of 14,000 ktonnes/year to the GBR lagoon (Kroon et al. 2012). Hillslope, streambank and gully erosion all contribute to the SS discharge with major variations between individual catchments (Bartley et al. 2010a, b; Wilkinson et al. 2013). Erosion may be severe in areas of cropping and urban development on high slope lands but such areas are of smaller extent than agriculture but may be a local threat to coastal reefs and seagrass areas. Port dredging and spoil dumping is an existing large source of remobilized sediment in the GBR and likely to increase greatly in the next decade (Brodie 2014) with currently poor management and governance arrangements (Grech et al. 2013).

Mean annual total nitrogen (TN) load to the GBR lagoon has increased by 5.7 times to 80,000 tonnes/year (Kroon et al. 2012). The anthropogenic nitrogen load comprises 11,000 tonnes/year dissolved inorganic nitrogen (DIN), 6,900 tonnes/year of dissolved organic nitrogen (DON), and 52,000 tonnes/year of particulate nitrogen (PN). Similarly, the mean annual total phosphorus (TP) load has increased by 8.9 times to 16,000 tonnes/year, with the anthropogenic phosphorus loads comprising 800 tonnes/year of dissolved inorganic phosphorus (DIP), 470 tonnes/year of dissolved organic phosphorus (DOP), and 13,000 tonnes/year of particulate phosphorus (PP). These nutrient increases are driven by the application of fertiliser on sugar cane, horticulture and other cropping areas in the GBRCA (Waterhouse et al. 2012), and losses of particulate bound nutrients from agricultural and urban lands due to soil erosion (Waterhouse et al. 2012). Pesticides would have been absent in runoff to the GBR prior to European settlement but now at least 30,000 kg/year of herbicides is discharged to the GBR (Kroon et al. 2012). This estimate comprises photosystem-II (PSII) inhibiting herbicides only (atrazine, ametryn, hexazinone, diuron, simazine and tebuthiuron), for which monitoring information exists.

This is an underestimate of the total pesticide load to the GBR as many pesticides known to be used in the GBRCA and hence have the potential to be discharged to the GBR are not monitored. Atrazine, ametryn, hexazinone, and diuron originate predominantly from the sugarcane industry (Bainbridge et al. 2009; Davis et al. 2012, 2013), with atrazine also being used in grains cropping, and tebuthiuron and simazine originating from the beef grazing industry and forestry plantations, respectively (Lewis et al. 2009; Shaw et al. 2010; Waterhouse et al. 2012).

In 2013 the current state of knowledge regarding the degradation of Great Barrier Reef ecosystems due to terrestrial pollutant runoff was reviewed and a 'Scientific Consensus Statement' was prepared for the Queensland Government (Brodie et al. 2013). The conclusions were:

1. The decline of marine water quality associated with terrestrial runoff from the adjacent catchments is a major cause of the current poor state of many of the key marine ecosystems of the Great Barrier Reef.
2. The greatest water quality risks to the Great Barrier Reef are from nitrogen discharge, associated with crown-of-thorns starfish outbreaks and their destructive effects on coral reefs, and fine sediment discharge which reduces the light available to seagrass ecosystems and inshore coral reefs. Pesticides pose a risk to freshwater and some inshore and coastal habitats.
3. Recent extreme weather – heavy rainfall, floods and tropical cyclones – have severely impacted marine water quality and Great Barrier Reef ecosystems. Climate change is predicted to increase the intensity of extreme weather events.
4. The main source of excess nutrients, fine sediments and pesticides from Great Barrier Reef catchments is diffuse source pollution from agriculture.
5. Improved land and agricultural management practices are proven to reduce the runoff of suspended sediment, nutrients and pesticides at the paddock scale.

3 Management of Agricultural Pollution for the GBR

3.1 Background

During the 1980s and 1990s, research and monitoring in the GBRCA and GBR identified land runoff of pollutants as a threat to the health of the GBR and provided an understanding of (i) pollutant generation in the GBRCA and the land uses/ agricultural industries and landscape processes contributing to the pollution, (ii) transport of pollutants from the GBRCA into the GBR, (iii) dispersion of pollutants in the GBR waters, (iv) effects of pollutants on specific GBR organisms and ecosystems, (v) management options to reduce pollution, and (vi) socio-economic and political issues in implementing improved management (Brodie et al. 2001; Furnas 2003). A major assumption was that point source pollution from sewage and industrial waste discharge were already well managed and in most cases, regulated.

3.2 Reef Plan

In 2003, the Australian and Queensland Governments jointly released the Reef Plan (Queensland Department of the Premier and Cabinet 2003). The plan aimed to halt and reverse the decline in water quality entering the Reef within 10 years (i.e. by 2013) by reducing diffuse pollution from agriculture. The Plan has objectives: (i) Reduce the load of pollutants from diffuse sources in the water entering the Reef, and (ii) Rehabilitate and conserve areas of the Reef catchment that have a role in removing water borne pollutants. In 2009, Reef Plan 2003 was revised and updated (Queensland Department of the Premier and Cabinet 2009) with better defined targets and actions and then a revised version was released in 2013 (Queensland Department of the Premier and Cabinet 2013). In addition to Reef Plan's 2003 aims and objectives, Reef Plan 2013 also has the somewhat visionary objective to ensure that "by 2020 the quality of water entering the GBR from adjacent catchments has no detrimental impact on the health and resilience of the GBR". Reef Plan 2009 had load reduction targets of (i) a minimum 50 % reduction in nitrogen and phosphorus loads at the end of catchments by 2013, (ii) a minimum 50 % reduction in pesticides at the end of catchments by 2013, (iii) a minimum of 50 % late dry season groundcover on dry tropical grazing land by 2013, and (iv) a minimum 20 % reduction in sediment load at the end of catchments by 2020. Given by 2013 the Reef Plan targets were known not to have been met Reef Plan 2013 set new targets with lengthened implementation timelines and generally reduced standards (see below). The Reef Plan 2013 targets:

Water quality targets (by 2018)

- At least a 50 % reduction in anthropogenic end-of-catchment dissolved inorganic nitrogen loads in priority areas.
- At least a 20 % reduction in anthropogenic end-of-catchment loads of sediment and particulate nutrients in priority areas.
- At least a 60 % reduction in end-of-catchment pesticide loads in priority areas.

Land and catchment management targets (by 2018)

- 90 % of sugarcane, horticulture, cropping and grazing lands are managed using best management practice systems (soil, nutrient and pesticides) in priority areas.
- Minimum 70 % late dry season groundcover on grazing lands.
- The extent of riparian vegetation is increased.
- There is no net loss of the extent, and an improvement in the ecological processes and environmental values, of natural wetlands.

Large changes in many of the targets were made in Reef Plan 2013 including lowering targets for nitrogen and phosphorus loads, an unchanged sediment load target and a small tightening of the pesticide load target (from 50 % reduction in loads to 60 % reduction). In 2009, a 50 % reduction in TN load was required by 2013 whereas in 2013 we estimate a 36 % reduction is all that is required by 2018. Similarly for TP, in 2009 a 50 % reduction was required whereas in 2013 only a 16 % reduction is required.

3.3 Reef Rescue

In 2007, the Federal Government implemented Reef Rescue, an AU $200 million investment for on-ground works, monitoring, research and partnerships over 5 years (Brodie et al. 2012). This voluntary program's objective is to improve the water quality of the GBR lagoon by increasing the adoption of land management practices that reduce the run-off of nutrients, pesticides and sediments from agricultural land. Whilst forming an integral component of Reef Plan 2009, Reef Rescue has its own 5-year outcome targets (i.e. by 2013). Both initiatives specify management action, catchment condition and end-of-catchment pollutant load targets for 2013 reported by catchment, regional and GBR-wide scales (Brodie et al. 2012). In 2013, Reef Rescue 2 was announced with a further A$200 million funding over the period 2014–2018. From 2008 Reef Rescue funded on-ground land management projects across the GBRCA with financial contributions from farmers, mainly in the sugarcane and grazing industries but also in dairy farming and horticulture. Projects include the introduction of new farming practices; fencing along streams for cattle management with off-stream watering points; pasture management through grazing pressure management; reduced fertiliser use through more efficient application techniques; machinery modifications including harvesters, fertiliser and pesticide application equipment; and cultivation and tillage equipment and practices.

3.4 Great Barrier Reef Protection Amendment Act 2009

The Queensland Government introduced the *Great Barrier Reef Protection Amendment Act 2009* (Reef Protection Package) in 2009. This Act introduces regulations to improve the quality of water entering the GBR in sugarcane growing and cattle grazing properties in the Burdekin Dry Tropics, Wet Tropics and Mackay Whitsunday Regions in North Queensland. The Act requires (i) Farm Environmental Risk Management Plans in sugarcane cultivation and beef grazing, (ii) Fertiliser management in sugarcane, (iii) Erosion management in grazing through managing pasture cover, and (iv) Pesticide management through application techniques and buffer strips.

3.5 Management Effectiveness

The effectiveness of currently recommended practices as well as newer Best Management Practices has been assessed (Thorburn et al. 2013; Thorburn and Wilkinson 2013) including using nitrogen fertiliser management systems such as ‘Six Easy Steps’ (Schroeder et al. 2010) and the ‘nitrogen replacement’ technique (Thorburn et al. 2011a, b; Webster et al. 2012) to reduce the current considerable losses of nitrogen from sugarcane cultivation. For herbicide management Masters et al. (2013) showed that Best Management Practices in sugarcane including controlled traffic resulted in load reductions of 60 %, 55 %, 47 %, and 48 % for ametryn, atrazine, diuron and hexazinone respectively. Herbicide losses in runoff were also reduced by 32–42 % when applications were banded rather than broadcast (Masters et al. 2013), a similar result to that shown in other sugarcane studies (Oliver et al. 2014) and cotton cropping systems in the Fitzroy catchment (Silburn et al. 2013).

In rangeland beef grazing lands, research into the effectiveness of pasture cover as an erosion prevention management technique have shown that grazing in semi-arid pastures should be managed to maintain >50 % ground cover to avoid excessive runoff and soil erosion, degradation of soil productivity and to maintain good off-site water quality (Silburn 2011a, b; Silburn et al. 2011). Reducing erosion in grazing lands is principally implemented through maintaining ground cover and biomass of pastures, especially during the dry season and droughts. Gully networks caused by livestock grazing are also important sources of sediment and targeted vegetation management will be important for reducing gully erosion (Thorburn et al. 2013). Other research has investigated the role and effectiveness of riparian forests and wetlands (constructed and natural) on trapping catchment pollutants (e.g. McJannet et al. 2011a, b; Connor et al. 2013). In general small wetlands (either natural or constructed) trap little sediment, phosphorus or nitrogen in north Queensland tropical climatic conditions.

The possibility of reaching the overall goal of Reef Plan (see above) of ‘no detrimental impact’ is in question given that current ‘Best Management Practices’ may not be enough to achieve this outcome (Kroon 2012; Thorburn and Wilkinson 2013). Modeling of land-use adoption scenarios across the entire GBR has shown that complete adoption of current best management practices in grazing and sugarcane would be sufficient to meet the Reef Plan targets for photosystem II herbicides, but are uncertain for suspended sediment, nitrogen and phosphorus (Thorburn and Wilkinson 2013; Waters et al. 2013) and unlikely to meet the desired ecological outcomes (Kroon 2012). If Reef Plan targets and goals are not met in the identified time frame (2018 and 2020), the conditions of inshore GBR ecosystems are unlikely to improve in the medium-term future given other major issues such as climate change are not being managed. The definition of specific ecological conditions to support a healthy and resilient ecosystem and hence ‘ecologically relevant’ load targets (Brodie et al. 2009) would enable an informed debate on the management actions and policy instruments required to achieve these ecological conditions. The social and economic costs of meeting the targets are not well understood, nor are the trade-offs required to meet long term Reef Plan goals (Dale et al. 2013).

3.6 Governance

Partnerships for effective governance for the GBR have been relatively well studied (e.g. Robinson et al. 2011) and evaluated using a SMART (Specific, Measurable, Achievable, Relevant and Timed) assessment (Robinson et al. 2009). By incorporating the range of local needs, values, aspirations and priorities into water quality objectives and values, water quality improvement plans are more likely to be supported by local communities (Bohnet et al. 2011; Tsatsaros et al. 2013).

At the GBR scale governance and its influence on effective environmental management has been reviewed by Dale et al. (2013) using a risk based approach. They note that in the analysis of governance systems designed to lead to better governance the following key points are relevant:

- *Best effect in participatory rather than expert-assessment contexts.* To establish better foundations for lasting reform, risk analysis of governance systems is best applied in participatory decision-making, enabling all participants to jointly analyse the health of their governance and to negotiate and monitor appropriate reforms in a structured way;
- *Best applied within reform-oriented approaches.* While risk analysis of governance systems can be a tool for dispassionate analysis by experts, its greatest strength lies in providing the evidence required for more participatory approaches to governance reform;
- *A foundation for benchmarking and monitoring governance systems.* Data outputs from risk analysis of governance systems create the ideal foundation, if applied periodically and consistently, for establishing long-standing benchmarks for governance systems, providing the foundation for monitoring progressive improvements;
- *Potential application in education and capacity building.* Operated in a strongly participatory way, or even within formal training, risk analysis of governance systems provides a clear framework for the delivery of education about governance, currently in short supply;
- *Determines risks and areas of strategic governance research.* Within any context, a useful outcome from the application of risk analysis of governance systems is the identification of strategic research themes required to address problems; and
- *Need for leadership in, and responsibility for, facilitating continuous improvement in governance.* To be effective, all governance systems need leadership in monitoring and driving continuous improvement. While government agencies are often best placed to lead and resource such attention, leadership can and should come from any key participants in the system. For best effect, the ongoing

process of risk analysis needs dedicated resourcing, and all system participants should be confident in those leading and managing analysis and reform.

3.7 Monitoring and Reporting

The success (or otherwise) of Reef Plan is assessed using an integrated monitoring, assessment and reporting program – the Paddock to Reef Program (Carroll et al. 2012). The program commenced in 2009 and report cards are released regularly e.g. the most recent in 2013 (The State of Queensland 2013). The program is built around a number of components including (a) management practice adoption monitoring and auditing; (b) paddock monitoring and modelling involving collecting runoff during actual rainfall events and rainfall simulation. Modelling is used to extend results from one situation to another not part of the monitoring scheme; (c) catchment monitoring and modelling to assess the water quality entering the GBR lagoon and to determine trends in water quality over time; identify potential source areas of contaminants; link plot to paddock to river scales; and validate and calibrate the existing catchment models; (d) marine monitoring including inshore biological monitoring of inshore coral reefs and intertidal seagrass meadows; and inshore water quality and flood plume monitoring focussing on TSS, nutrients, Chl *a*, salinity, pesticides, temperature, turbidity and light conditions; and (e) reporting on progress through an annual 'Report Card' supported by detailed technical reports. The latest report card released in 2013 shows that considerable progress has been made with small but significant reductions in suspended sediment, nutrient and pesticide loads discharged to the GBR compared to earlier years (The State of Queensland 2013).

4 Conclusions

Water quality associated with pollutant discharge from the GBRCA is still a major issue for GBR ecosystems. Recent research confirms that water discharge to the GBR is of poor quality in many rivers and in the associated flood plumes (Devlin et al. 2012a; Kroon et al. 2012; Kennedy et al. 2012); contaminants are present in the GBR lagoon at concentrations likely to cause environmental harm (Devlin et al. 2012b; Lewis et al. 2009, 2012; Schaffelke et al. 2012; Shaw et al. 2010); and evidence of the causal relationship between water quality and GBR ecosystem health is clear (Brodie et al. 2011; Fabricius et al. 2010; Lewis et al. 2012; Petus et al. 2014).

Currently, while management action is being taken and improvements in the pollutant loading of river discharge improved, whether this is enough to achieve the Reef Plan targets or the most appropriate form of management is not yet known (Thorburn and Wilkinson 2013; Kroon 2012). However, it is expected that measurable improvements in river and coastal marine water quality, or ecosystem health, may not be detected for up to several decades (Darnell et al. 2012).

References

Agardy T, di Sciara GN, Christie P (2011) Mind the gap: addressing the shortcomings of marine protected areas through large scale marine planning. Mar Policy 35:226–232

Alvarez-Romero JG, Pressey RL, Ban NC, Vance-Borland K, Willer C, Klein CJ, Gaines SD (2011) Integrated landesea conservation planning: the missing links. Annu Rev Ecol Evol Syst 42:381–409

Bainbridge ZT, Brodie JE, Faithful JW, Sydes DA, Lewis SE (2009) Identifying the land-based sources of suspended sediments, nutrients and pesticides discharged to the Great Barrier Reef from the Tully-Murray Basin, Queensland, Australia. Mar Freshw Res 60:1081–1090

Bartley R, Wilkinson SN, Hawdon AA, Abbott BN, Post DA (2010a) Impacts of improved grazing land management on sediment yields, Part 1: hillslope processes. J Hydrol 389:237–248

Bartley R, Wilkinson SN, Hawdon AA, Abbott BN, Post DA (2010b) Impacts of improved grazing land management on sediment yields, Part 2: catchment response. J Hydrol 389:249–259

Bohnet IC, Roebeling PC, Williams KJ, Holzworth D, van Grieken ME, Pert PL, Kroon FJ, Westcott DA, Brodie JE (2011) Landscapes toolkit – an integrated modeling framework to assist stakeholders in exploring options for sustainable landscape development. Landsc Ecol 26:1179–1198

Brander LM, van Beukering P, Cesar HSJ (2007) The recreational value of coral reefs: a meta-analysis. Ecol Econ 63:209–218

Brodie J (2014) Dredging the Great Barrier Reef: use and misuse of science. Estuar Coast Shelf Sci 142:1–3. doi:10.1016/j.ecss.2014.01.010

Brodie J, Waterhouse J (2012) A critical assessment of environmental management of the 'not so Great' Barrier Reef. Estuar Coast Shelf Sci 104–105:1–22. doi:10.1016/j.ecss.2012.03.012

Brodie J, Christie C, Devlin M, Haynes D, Morris S, Ramsay M, Waterhouse J, Yorkston H (2001) Catchment management and the Great Barrier Reef. Water Sci Technol 43:203–211

Brodie J, Fabricius K, De'ath G, Okaji K (2005) Are increased nutrient inputs responsible for more outbreaks of crown-of-thorns starfish? An appraisal of the evidence. Mar Pollut Bull 51:266–278

Brodie J, Lewis S, Bainbridge Z, Mitchell A, Waterhouse J, Kroon F (2009) Pollutant discharge management target setting for rivers in the Great Barrier Reef catchment area. Mar Freshw Res 60:1141–1149

Brodie JE, Devlin MJ, Haynes D, Waterhouse J (2011) Assessment of the eutrophication status of the Great Barrier Reef lagoon (Australia). Biogeochemistry 106:281–302. doi:10.1007/s10533-010-9542-2

Brodie J, Kroon F, Schaffelke B, Wolanski E, Lewis S, Devlin M, Bohnet IC, Bainbridge Z, Waterhouse J, Davis A (2012) Terrestrial pollutant runoff to the Great Barrier Reef: an update on issues, priorities and management responses. Mar Pollut Bull 65:81–100

Brodie J, Waterhouse J, Schaffelke B, Johnson J, Kroon F, Thorburn P, Rolfe J, Lewis S, Warne M, Fabricius K, McKenzie L, Devlin M (2013) Reef water quality scientific consensus statement 2013. Department of the Premier and Cabinet, Queensland Government, Brisbane

Bruno JF, Selig ER (2007) Regional decline of coral cover in the Indo-Pacific: timing, extent, and subregional comparisons. PLoS One 28:e711. doi:10.1371/journal.pone.0000711

Burke L, Reytar K, Spalding M, Perry A (2011) Reefs at risk revisited. World Resources Institute, Washington, DC, p 114. Available at www.wri.org

Carroll C, Waters D, Vardy S, Silburn DM, Attard S, Thorburn PJ, Davis AM, Schmidt M, Wilson B, Clark A (2012) A paddock to reef monitoring and modelling framework for the Great Barrier Reef: paddock and catchment component. Mar Pollut Bull 65:136–149

Christie P, White AT (2007) Best practices for improved governance of coral reef marine protected areas. Coral Reefs 26:1047–1056

Connor S, Nelson PN, Armour JD, Henault C (2013) Hydrology of a forested riparian zone in an agricultural landscape of the humid tropics. Agric Ecosys.Environ. http://dx.doi.org/10.1016/j.agee.2011.12.006

Cooper TF, De'ath G, Fabricius KE, Lough JM (2008) Declining coral calcification in massive porites in two nearshore regions of the northern Great Barrier Reef. Glob Chang Biol 14:529–538

Dale A, Vella K, Pressey RL, Brodie J, Yorkston H, Potts R (2013) Risk analysis across governance systems: a Great Barrier Reef case study. Environ Lett 8. doi:10.1088/1748-9326/8/1/015037

Darnell R, Henderson B, Kroon FJ, Kuhnert P (2012) Statistical power of detecting trends in total suspended sediment loads to the Great Barrier Reef. Mar Pollut Bull 65:203–209

Davis AM, Lewis SE, Bainbridge ZT, Brodie JE, Glendenning L, Turner R (2012) Dynamics of herbicide transport and partitioning under event flow conditions in the lower Burdekin region, Australia. Mar Pollut Bull 65:182–193

Davis AM, Thorburn PJ, Lewis SE, Bainbridge ZT, Attard SJ, Milla R, Brodie JE (2013) Environmental impacts of irrigated sugarcane production: herbicide run-off dynamics from farms and associated drainage systems. Agric Ecosyst Environ 180:123–135

Day JC, Dobbs K (2013) Effective governance of a large and complex cross-jurisdictional marine protected area: Australia's Great Barrier Reef. Mar Policy 41:14–24

de Groot R, Brander L, van der Ploeg S, Costanza R, Bernard F, Braat L, Christie M, Crossman N, Ghermandi A, Hein L, Hussain S, Kumar P, McVittie A, Portela R, ten Rodriguez LC, Brink P, van Beukering P (2012) Global estimates of the value of ecosystems and their services in monetary units. Ecosystem Serv 1:50–61

De'ath G, Fabricius KE (2010) Water quality as a regional driver of coral biodiversity and macroalgae on the Great Barrier Reef. Ecol Appl 20:840–850

De'ath G, Lough JM, Fabricius KE (2009) Declining coral calcification on the Great Barrier Reef. Science 323:116–119

De'ath G, Fabricius K, Sweatman H, Puotinen M (2012) The 27-year decline of coral cover on the Great Barrier Reef and its causes. Proc Natl Acad Sci. doi:10.1073/pnas.1208909109

Deloitte Access Economics (2013) Economic contribution of the Great Barrier Reef. Great Barrier Reef Marine Park Authority, Townsville

Devlin M, Brodie J, Wenger A, da Silva E, Alvarez- Romero JG, Waterhouse J, McKenzie L (2012a) Extreme weather conditions in the Great Barrier Reef: drivers of change? Proceedings of the 12th international coral reef symposium, Cairns, Australia, 9–13 July 2012

Devlin MJ, McKinna LW, Álvarez-Romero JG, Petus C, Abott B, Harkness P, Brodie J (2012b) Mapping the pollutants in surface riverine flood plume waters in the Great Barrier Reef, Australia. Mar Pollut Bull 65(4–9):224–235

Dickinson M, Rutherford M, Gunton T (2010) Principals for integrated marine planning: a review of international experience. Environ J 37:21–46

Douvere F (2008) The importance of marine spatial planning in advancing ecosystem-based sea use management. Mar Policy 32:762–771

Economics O (2009) Valuing the effects of Great Barrier Reef bleaching. Great Barrier Reef Foundation, Brisbane

Fabricius K, De'ath G, McCook L, Turak E, Williams D (2005) Changes in algal, coral and fish assemblages along water quality gradients on the inshore Great Barrier Reef. Mar Pollut Bull 51:384–398

Fabricius K, Okaji K, De'ath G (2010) Three lines of evidence to link outbreaks of the crown-of-thorns seastar *Acanthaster planci* to the release of larval food limitation. Coral Reefs 29:593–605

Fenton DM, Marshall NA (2001a) A guide to the fishers of Queensland. Part A. TRC-analysis and social profiles of Queensland's commercial fishing industry. CRC Reef Research Centre, Townsville, Australia

Fenton DM, Marshall NA (2001b) A guide to the fishers of Queensland. Part C. TRC-analysis and social profiles of Queensland's charter fishing industry. CRC Reef Research Centre, Townsville
Fernandes L, Day J, Lewis A, Slegers S, Kerrigan B, Breen D, Cameron D, Jago B, Hall J, Lowe D, Innes J, Tanzer J, Chadwich V, Thompson L, Gorman K, Simmons M, Barnett B, Sampson K, De'ath G, Mapstone B, Marsh H, Possingham H, Ball I, Ward T, Dobbs K, Aumend J, Slater D, Stapleton K (2005) Establishing representative no-take areas in the Great Barrier Reef: largescale implementation of theory on marine protected areas. Conserv Biol 19:1733–1744
Furnas M (2003) Catchments and corals: terrestrial runoff to the Great Barrier Reef. Australian Institute of Marine Science, Townsville, 334 pp
Grech A, Coles R, Marsh H (2011) A broad-scale assessment of the risk to coastal seagrasses from cumulative threats. Mar Policy 35(5):560–567
Grech A, Bos M, Brodie J, Coles R, Dale A, Gilbert R, Hamann M, Marsh H, Neil K, Pressey B, Rasheed M, Sheaves M, Smith A (2013) Guiding principles for the improved governance of port and shipping impacts in the Great Barrier Reef. Mar Pollut Bull 75:8–20. doi:10.1016/j.marpolbul.2013.07.013
Haapkyla J, Unsworth RKF, Flavell M, Bourne DG, Schaffelke B, Willis BL (2011) Seasonal rainfall and runoff promote coral disease on an inshore reef. PLoS One 6:e16893. doi:10.1371/journal.,pone.0016893
Halpern BB, Walbridge S, Selkoe KA, Kappel CV, Micheli F, D'Agrosa C, Bruno JF, Casey KS, Ebert C, Fox HE, Fujita R, Heinemann D, Lenihan HS, Madin EMP, Perry MT, Selig ER, Spalding M, Steneck R, Watson R (2008) A global map of human impact on marine ecosystems. Science 319:948e952
Hughes TP, Graham NAJ, Jackson JBC, Mumby PJ, Steneck RS (2010) Rising to the challenge of sustaining coral reef resilience. Trends Ecol Evol 25:633–642
Hughes TP, Bellwood DR, Baird AH, Brodie J, Bruno JF, Pandolfi JM (2011) Shifting base-lines, declining coral cover, and the erosion of reef resilience. Comment on Sweatman et al. (2011). Coral Reefs 30:653–660
Jones AM, Berkelmans R (2014) Flood Impacts in Keppel Bay, Southern Great Barrier Reef in the aftermath of cyclonic rainfall. PLoS One 9(1):e84739. doi:10.1371/journal.pone.0084739
Kennedy K, Schroeder T, Shaw M, Haynes D, Lewis S, Bentley C, Paxman C, Carter S, Brando V, Bartkow M, Mueller J (2012) Long term monitoring of photosystem-II herbicides – correlation with remotely sensed freshwater extent to monitor changes in the quality of water entering the Great Barrier Reef, Australia. Mar Pollut Bull 65:292–305
Knowles T, Campbell R (2011) What's a whale worth? Valuing whales for National Whale Day, a report for the International Fund for Animal Welfare (IFAW), prepared by Economists at Large, Melbourne, Australia
Kroon FJ (2012) Towards ecologically relevant targets for river pollutant loads to the Great Barrier Reef. Mar Pollut Bull 65:261–266
Kroon FJ, Kuhnert KM, Henderson BL, Wilkinson SN, Kinsey-Henderson A, Brodie JE, Turner RDR (2012) River loads of suspended solids, nitrogen, phosphorus and herbicides delivered to the Great Barrier Reef lagoon. Mar Pollut Bull 65:167–181
Lawrence D, Kenchington R, Woodley S (2002) The Great Barrier Reef. Finding the right balance. Melbourne University Press, Melbourne, p 296
Lewis SE, Brodie JE, Bainbridge ZT, Rohde KW, Davis AM, Masters BL, Maughan M, Devlin MJ, Mueller JF, Schaffelke B (2009) Herbicides: a new threat to the Great Barrier Reef. Environ Pollut 157:2470–2484
Lewis SE, Schaffelke B, Shaw M, Bainbridge ZT, Rohde KW, Kennedy KE, Davis AM, Masters BL, Devlin MJ, Mueller JF, Brodie JE (2012) Assessing the risks of PS-II herbicide exposure to the Great Barrier Reef. Mar Pollut Bull 65:280–291
Marsh H, De'ath G, Gribble N, Lane B (2005) Historical marine population estimates: triggers or targets for conservation? The dugong case study. Ecol Appl 15:481–492. doi:10.1890/04-0673
Masters B, Rohde K, Gurner N, Reid D (2013) Reducing the risk of herbicide runoff in sugarcane farming through controlled traffic and early-banded application. Agric Ecosyst Environ 180:29–39

McArthur LC, Boland JW (2006) The economic contribution of seagrass to secondary production in South Australia. Ecol Model 196:163–172

McJannet D, Wallace J, Keen R, Hawdon A, Kemei J (2011a) The filtering capacity of a tropical riverine wetland: I. Water balance. Hydrol Process. doi:10.1002/hyp.8108

McJannet D, Wallace J, Keen R, Hawdon A, Kemei J (2011b) The filtering capacity of a tropical riverine wetland: II. Sediment and nutrient balances. Hydrol Process. doi:10.1002/hyp.811

Mora C, Andrèfouët S, Costello MJ, Kranenburg C, Rollo A, Veron J, Gaston KJ, Myers RA (2006) Coral reefs and the global network of marine protected areas. Science 312(5781):1750–1751

Nobre AM (2011) Scientific approaches to address challenges in coastal management. Mar Ecol Prog Ser 434:279–289

Oliver DP, Anderson JS, Davis A, Lewis S, Brodie J, Kookana R (2014) Banded applications are highly effective in minimizing herbicide migration from furrow-irrigated sugar cane. Sci Total Environ 466–467:841–848

Osborne K, Dolman AM, Burgess SC, Johns KA (2011) Disturbance and the dynamics of coral cover on the Great Barrier Reef (1995–2009). PLoS One 6(3):e17516. doi:10.1371/journal.pone.0017516

Osmond M, Airame S, Day JC, Caldwell M (2010) Lessons for marine spatial planning: a comparison of three marine protected area processes. Ocean Coast Manag 53:41–51

Pandolfi JM, Bradbury RH, Sala E, Hughes TP, Bjorndal KA, Cooke RG, McArdle D, McClenachan L, Newman MJH, Paredes G, Warner RR, Jackson JBC (2003) Global trajectories of the long-term decline of coral reef ecosystems. Science 301:955–958

Petus C, Collier C, Devlin M, Rasheed M, McKenna S (2014) Using MODIS data for understanding changes in seagrass meadow health: a case study in the Great Barrier Reef (Australia). Mar Environ Res 98:68–85

Prayaga P, Rolfe J, Stoeckl N (2010) The value of recreational fishing in the Great Barrier Reef, Australia: a pooled revealed preference and contingent behaviour model. Mar Policy 34:244–251

Queensland Department of the Premier and Cabinet (2003) Reef water quality protection plan; for catchments adjacent to the Great Barrier Reef world heritage area. Queensland Department of Premier and Cabinet, Brisbane. http://www.reefplan.qld.gov.au/about/assets/reefplan-2003.pdf

Queensland Department of the Premier and Cabinet (2009) Reef water quality protection plan 2009: for the Great Barrier Reef world heritage area and adjacent catchments. Queensland Department of Premier and Cabinet, Brisbane. http://www.reefplan.qld.gov.au/resources/assets/reef-plan-2009.pdf

Queensland Department of the Premier and Cabinet (2013) Reef water quality protection plan 2013: for the Great Barrier Reef world heritage area and adjacent catchments. Queensland Department of Premier and Cabinet, Brisbane

Rasheed MA, McKenna SA, Carter AB, Coles RG (2014) Contrasting recovery of shallow and deep water seagrass communities following climate associated losses in tropical north Queensland, Australia. Mar Pollut Bull 83:491–499

Robinson CJ, Taylor BM, Pearson L, O'Donohue M, Harman B (2009) A SMART assessment of water quality partnerships in Great Barrier Reef catchments. Aust J Environ Manag 16:84–93

Robinson CJ, Margerum RD, Koontz TM, Moseley C, Lurie S (2011) Policylevel collaboratives for environmental management at the regional scale: lessons and challenges from Australia and the United States. Soc Nat Resour 24:849–859

Ruckelshaus M, Klinger T, Knowlton N, DeMaster DP (2008) Marine ecosystem based management in practice: scientific and governance challenges. Bioscience 58:53–63

Schaffelke B, Carleton J, Skuza M, Zagorskis I, Furnas MJ (2012) Water quality in the inshore Great Barrier Reef lagoon: implications for long-term monitoring and management. Mar Pollut Bull 65:249–260

Schroeder BL, Hurney AP, Wood AW, Moody PW (2010) Concepts and value of the nitrogen guidelines contained in the Australian sugar industry's 'six easy steps' nutrient management program. In: Proceedings of the international society of sugar cane technologists, vol 27, 11 pp

Shaw M, Furnas MJ, Fabricius K, Haynes D, Carter S, Eaglesham G, Mueller JF (2010) Monitoring pesticides in the Great Barrier Reef. Mar Pollut Bull 60:113–122

Short F, Coles RG, Fortes MD, Victor S, Isnain I, Andrew J, Seno A, Salik M (2014) Monitoring in the Western Pacific region shows evidence of seagrass decline in line with global trends. Mar Pollut Bull. http://dx.doi.org/10.1016/j.marpolbul.2014.03.036

Silburn DM (2011a) Hillslope runoff and erosion on duplex soils in grazing lands in semi-arid central Queensland. II. Simple models for suspended and bedload sediment. Soil Res 49:118–126

Silburn DM (2011b) Hillslope runoff and erosion on duplex soils in grazing lands in semi-arid central Queensland III. USLE erodibility (K factors) and cover-soil loss relationships. Soil Res 49:127–134

Silburn DM, Carroll C, Ciesiolka CAA, deVoil RC, Burger P (2011) Hillslope runoff and erosion on duplex soils in grazing lands in semi-arid central Queensland. Soil Res 49:105–117

Silburn DM, Foley JL, deVoil RC (2013) Managing runoff of herbicides under rainfall and furrow irrigation with wheel traffic and banded spraying. Agric Ecosyst Environ 180:40–53

Stoeckl N, Birtles A, Farr M, Mangott A, Curnock M, Valentine P (2010) Live-aboard dive boats in the Great Barrier Reef: regional economic impact and the relative values of their target marine species. Tour Econ 16(4):995–1018. doi:10.5367/te.2010.0005

Stoeckl N, Hicks CC, Mills M, Fabricius K, Esparon M, Kroon F, Kaur K, Costanza R (2011) The economic value of ecosystem services in the Great Barrier Reef: our state of knowledge. Ecol Econ Rev 1219:113–133

The State of Queensland (2013) Great Barrier Reef 2011 report card. Reef water quality protection plan. Published by the Reef Water Quality Protection Plan Secretariat. Queensland Government, Brisbane, Australia

Thomas CR, Gordon IJ, Wooldridge S, Marshall P (2012) Balancing the tradeoffs between ecological and economic risks for the Great Barrier Reef: a pragmatic conceptual framework. Hum Ecol Risk Assess 18:69–91

Thompson A, Costello P, Davidson J, Logan M, Schaffelke B, Uthicke S, Takahashi M (2011) Reef rescue marine monitoring program. Report of AIMS Activities – Inshore coral reef monitoring 2011. Report for Great Barrier Reef Marine Park Authority. Australian Institute of Marine Science, Townsville, 128 pp

Thorburn PJ, Wilkinson SN (2013) Conceptual frameworks for estimating the water quality benefits of improved agricultural management practices in large catchments. Agr Ecosyst Environ 180:192–209

Thorburn PJ, Biggs JS, Webster AJ, Biggs IM (2011a) An improved way to determine nitrogen fertiliser requirements of sugarcane crops to meet global environmental challenges. Plant Soil 339:51–67. doi:10.1007/s11104-010-0406-2

Thorburn PJ, Biggs JS, Attard SJ, Kemei J (2011b) Environmental impacts of irrigated sugarcane production: nitrogen lost through runoff and leaching. Agric Ecosyst Environ 144:1–12. doi:10.1016/j.agee.2011.08.003

Thorburn P, Wilkinson S, Silburn M (2013) Water quality in agricultural lands draining to the Great Barrier Reef: a review of causes, management and priorities. Agric Ecosyst Environ 180:4–20

Thorhaug A (1990) Restoration of mangroves and seagrasses – economic benefits for fisheries and mariculture. In: Berger JJ (ed) Environmental restoration. Science and strategies for restoring the earth. Island Press, Washington, pp 265–281

Tsatsaros JH, Brodie JE, Bohnet IC, Valentine P (2013) Water quality degradation of coastal waterways in the wet tropics, Australia. Water Air Soil Pollut 224:1443

Unsworth RKF, van Keulen M, Coles RG (2014) Seagrass meadows in a globally changing environment. Mar Pollut Bull 83:383–386

Waterhouse J, Grundy M, Brodie J, Gordon I, Yorkston H, Eberhard R (2009) Managing the catchment of the Great Barrier Reef, Chapter 15. In: Ferrier R, Jenkins J (eds) Handbook of catchment management. Blackwell Publishing, United Kingdom, pp 351–375

Waterhouse J, Brodie J, Lewis S, Mitchell A (2012) Quantifying the sources of pollutants to the Great Barrier Reef. Mar Pollut Bull 65:394–406

Waters D, Carroll C, Ellis R, McCloskey G, Hateley L, Packett R, Dougall C, Fentie B (2013) Catchment modelling scenarios to inform GBR water quality targets.In: MODSIM2013, 20th International Congress on Modelling and Simulation Society of Australia and New Zealand. (Adelaide, Australia, 2013), pp 3204–3210

Webster AJ, Bartley R, Armour JD, Brodie JE, Thorburn PJ (2012) Reducing dissolved inorganic nitrogen in surface runoff water from sugarcane production systems. Mar Pollut Bull 65:128–135

Wilkinson SN, Hancock GJ, Bartley R, Hawdon AA, Keen R (2013) Using sediment tracing to identify sources of fine sediment in grazed rangelands draining to the Great Barrier Reef. Agric Ecosyst Environ 180:90–102

Wilson C, Tisdell C (2003) Conservation and economic benefits of wildlife-based marine tourism: sea turtles and whales as case studies. Hum Dimens Wildl Int J 8(1):49–58. doi:10.1080/10871200390180145

Ecohydrology: A New Approach to Old Problems for Sustainable Management of Aquatic Ecosystem of Bangladesh for Ecosystem Service Provision

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Abstract Aquatic ecosystems of Bangladesh have crucial importance as majority of the people of Bangladesh are dependent on this sector. Recently this ecosystem is under constant threat due to increase of population, intensive agriculture, overexploitation, pollution, poorly planned infrastructures, climate change, water diversion by India, lack of proper policy and directives. These threats results degradation of this aquatic ecosystem which have deleterious impact on human livelihood and agro-environmental practice. Taking this into consideration, this review article describes the importance of aquatic ecosystem; causes and effects of degradation of aquatic ecosystem as well as the existing management practices are highlighted and based on this discussion probable solution are proposed for aquatic ecosystem management by focusing ecohydrological approach.

Keywords Water resources • Overexploitation • Pollution • Water availability • Ecohydrology

1 Introduction

Ecosystem services, a collective term for the goods and services produced by ecosystems that benefit humankind, have traditionally been undervalued as they often fall outside conventional markets (NRC 2005). They contribute to social and cultural well being (Fischer et al. 2009) and have high economic value (Turner et al. 2008). They have been broadly classified as provisioning, regulating, cultural and supporting. This concept has much earlier origins in the context of aquatic ecosystem research where it has been embedded in the ideas of ecosystem functioning and the consequent human values (Maltby 1986). Aquatic ecosystems are associated with a diverse and complex array of direct and indirect uses. Direct uses include the

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use of the aquatic ecosystem for water supply and the harvesting of aquatic products such as fish and plant resources. Indirect benefits are derived from environmental functions such as flood water retention, groundwater recharge, nutrient, abatement, etc.; depending on the type of wetland, also on soil and water characteristics and associated biotic influences (Mitsch and Gosselink 1993). Flood plain wetlands are particularly associated with groundwater recharge and individual floodplains may exhibit either or both of these functions (Thompson and Hollis 1995). However, the degradation and loss of aquatic ecosystems all over the world, together with the subsequent recognition of the ecological value of the services they provide, has made the restoration of aquatic ecosystems a top priority (Tong et al. 2007).

In Bangladesh, more than two-thirds of the country may be classified as aquatic ecosystem according to the Ramsar Convention's definition (FAO 1988). These consist of a wide variety of types, ranging from lakes, ox-bow lakes, rivers, flood plains, coastal wetlands, paddy fields and ponds (Craig et al. 2004). Ninety percent of the aquatic ecosystems of Bangladesh are dependent on the flow from three major rivers, but are now threatened by diversion of water from the Ganga–Padma River in India (Gopal 1995). All of these aquatic ecosystems form a unique mosaic of habitats with an extremely rich diversity of flora and fauna. They support the livelihood of millions of people in activities as diverse as fishing and collecting honey. They also support the provision of materials for thatching, in addition to wood for fuel and also have many uses in agriculture. Unfortunately these aquatic ecosystems are vanishing or have become degraded as a result of overexploitation of both ground and surface water, population pressure, ill-planned flood control and irrigation infrastructures that cause habitat destruction of flood plains (Fig. 1). These cause the loss and destruction of fish breeding grounds and nurseries. It also causes siltation of river basins and flood plains and pollution of aquatic ecosystems (Hussain 2010). In

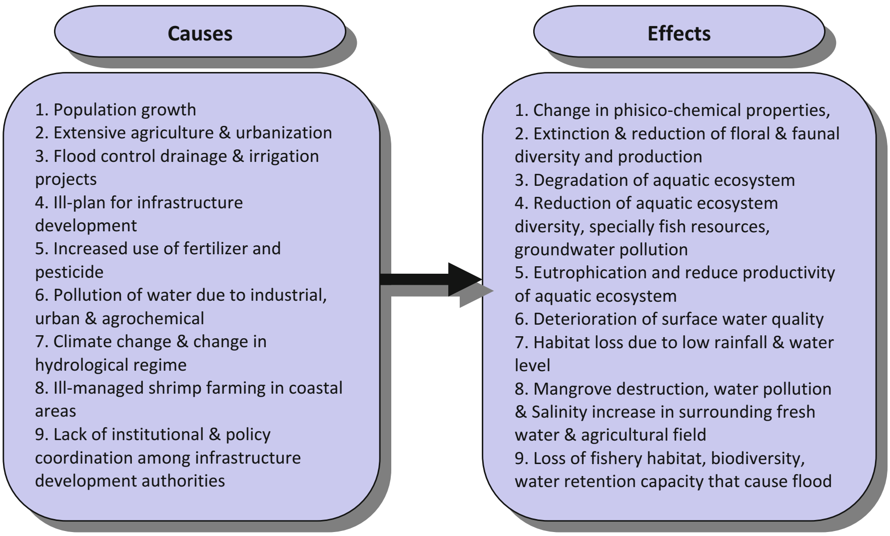

Fig. 1 Major ecological problem faced by the aquatic ecosystem and its effect (After Nishat 1993)

addition, present government policies with regard to management practices are somehow responsible for the degradation of the aquatic ecosystem of Bangladesh.

The dynamics of aquatic ecosystem depends on climate, geomorphology, plant cover and nutrient flow. In contrast to this its modification and degradation depends on the harmonization of population density, agriculture, urbanization, industrial development and hydrotechnical infrastructure with ecosystem potential. In the Anthropocene, the environment is dominated and transformed by socio-ecological processes and, for this reason, the existing traditional management strategy in Bangladesh is not sufficient to reverse the degradation of the aquatic ecosystem. Therefore, reversing the degradation of the biosphere requires solutions based on an integrative science such as ecohydrology, which increases the carrying capacity of the ecosystem in the current situation of human pressure and climate change (Zalewski 2010). To achieve these goals, ecohydrology uses hydrology to shape biota and vice versa. It also uses ecosystem properties as a management tool to increase the carrying capacity of an ecosystem (Zalewski 2000). Therefore, taking this into consideration the objectives of this paper are to (1) to explore the detailed ecosystem service values of aquatic ecosystem for Bangladesh, (2) to discover the specific causes and effects of aquatic ecosystem degradation, (3) to examine the present management and conservation approach and (4) to find a possible solution based on an ecohydrological approach.

2 An Overview of Aquatic Ecosystem Resources in Bangladesh

The abundance of water and wetlands has always been the geographical and historical destiny of Bangladesh. About 6.7 % of Bangladesh is permanently under water, 21 % is deeply flooded (more than 90 cm) and 35 % experiences shallow inundation (FAO 1988). The average discharge of water in the Bangladesh delta in the flood season is more than 5 million cusec. The wetlands in Bangladesh encompass a wide variety of dynamic ecosystems, ranging from rivers (7,497 km^2), estuaries and mangrove swamps (6,102 km^2), flood plains (45,866 km^2), Kaptai Lake (man-made reservoir, 688 km^2), ponds (1,469 km^2), ox-bow lakes, (1,197 km^2), and brackish-water farms (72,899 km^2) (Akonda 1989).

3 Values of Aquatic Ecosystem for Ecosystem Services

3.1 *Flood Control*

The flood plains of the major rivers of Bangladesh act as natural storage reservoirs. Embankments and polders have reduced flood plain storage capacity when flooding occurs, leading to an increase in water levels and discharges in many rivers (Chowdhury 1998).

3.2 Groundwater Replenishment

Many wetlands are directly connected to groundwater and play a vital role in regulating the quantity and quality of groundwater, which is often an important source of water for drinking and irrigation of crops. Unsustainable abstraction of groundwater for human use threatens the very existence of some wetlands. Irrational groundwater use is increasing because of the huge population pressure.

3.3 Shoreline Stabilization and Storm Protection

Wetland vegetation makes an important contribution to erosion control. Coastal wetland, particularly mangrove forest, contributes to shoreline stabilization and storm protection, at a much lower cost than engineered structures, by helping to dissipate the force and lessen the damage of wind and wave action in many low-lying areas. They can therefore play an important role in the natural management of coasts. In the Ganges-Brahmaputra delta of Bangladesh, for example, historically the extent of cyclone damage behind the mangrove swamps has been less than that behind the non-mangrove coast (Shine and Klemm 1999).

3.4 Sediment and Nutrient Retention and Export

Wetlands act as 'storehouses' for sediments and nutrients carried in rainwater runoff, streams and rivers. Disolved nutrients, such as nitrates and phosphates from fertilizers and sewage effluent are taken up by wetland plants and stored in their leaves, stems and roots, so helping to improve water quality. The continual supply of nutrients makes flood plains and deltas naturally fertile. The coastal waters are enriched with nutrients from the land and nutrient rich silt, which enable them to support a wide biological diversity (Hossain 2001).

3.5 Water Purification

Wetlands play an important role in purifying water by 'locking up' pollutants in their sediments, soils and vegetation. Some floating plants, e.g. Eichhornia crassipes (water hyacinth) can absorb and 'store' heavy metals. The natural ability of wetlands to 'filter' and clean water has been used to treat waste water, sewage, as well as tannery waste water. In Bangladesh, an uncontrolled discharge of tannery wastewater is causing serious deterioration of the water quality and ecological health of many waterways. Approximately, 18 m^3 d1 of liquid waste is produced from 300 small and medium scaled tannery industries, which discharge directly into local rivers, without any treatment (Mohanta et al. 2010). Saeed et al. (2012) found that application of the constructed wetland would be low-cost, energy-efficient, wastewater treatment technology for Bangladesh because of their natural ability to uptake pollutants.

3.6 Reservoirs of Biodiversity

Wetlands contain very rich components of biodiversity of local, national, and regional significance. Among the estimated 5,000 species of flowering plants and 1,500 species of vertebrates in the country, up to 300 plant species and some 400 vertebrate species are judged to be dependent on wetlands for all or part of their lifespan. Wetlands also provide habitat for a variety of resident and migratory waterfowl, a significant number of endangered species of international interest, and a large number of commercially important species. The inland capture fishery is based on the vast freshwater resources, with some 260 species of fin fish and 25 shellfishes (Khan et al. 1994).

3.7 Wetland Product and Livelihood

Wetlands of Bangladesh are one of the major sources of livelihoods particularly for cultivating food crops, vegetables, fishing, and pasture lands. Cultivation of rice is the major livelihood activity in and around the wetlands of Ganges- Brahmaputra flood plain and Haor basin. The second largest livelihood activity is fishing. A large section of the rural population depends on fishing in these aquatic ecosystems for their livelihoods. The available information indicates that over 60 % of all categories of farmers have had some participation in fishing (ODA 1997). Country boats account for nearly 60 % of employment in transport, and a section of the rural poor earn their livelihood by plying country boats. This is nearly three times greater than the number employed in all mechanised modes in total (Jansen et al. 1989). Bangladesh has extensive flood plain wetlands that harbour and support a wide range of aquatic plants and biodiversity. Wetland plants are harvested by the rural poor as a source of supplementary food, and for firewood, thatching, mat making, livestock fodder and medicinal use.

3.8 Cultural Values

Cultural ecosystem services refer to the intangible benefits people receive from ecosystems in the form of non-material, spiritual, religious, inspirational and educational experience. By supporting recreational activities, delivering spiritual and religious values, and providing aesthetic beauty, aquatic ecosystems are believed to substantially contribute to the well being of both coastal and inland inhabitants. Aquatic ecosystem attracts diverse recreational uses, generating significant income that benefits local communities. In Bangladesh, revenue earned through ecotourism of the Sundarban reserve forest (Mangrove wetland) is very high (Iqbal et al. 2010). Spiritual value might be gained from landscape beauty. Religious value might be gained from the provision of a place for conducting ceremonies. Educational and scientific value might be gained from the presence of unimpacted environments which provide an opportunity to understand natural biological processes.

3.9 Climate Change Pressure

Climate change is recognized as a major threat to the survival of species and integrity of ecosystems worldwide (Hulme 2005). It will most likely impact aquatic ecosystem services differently on a regional and mega-watershed level (Erwin 2009). Aquatic ecosystems themselves are part of the fight against climate change. They can help reduce both the level of future greenhouse gas emissions and the adverse effects of global warming. Some wetlands, especially mangroves, are large stores of carbon. In Bangladesh, conservation of 412,000 ha of natural mangrove forests is expected to generate over 210,000 tCO_2e annually in the course of a 30-year project period with a total emission reduction of about 6.4 million (GOB 2012). Adverse climate change can result in freshwater floods. Aquatic ecosystem reduces peak flood flows by delaying and storing waters which further detain polluted floodwaters and improve their quality. Mangrove ecosystems can increase the resilience of coastal areas through alluvial plain accumulation and create a freshwater buffer, preventing saline intrusion. Rising temperatures and less rainfall in arid areas will make droughts and water shortage more extreme. Various aquatic ecosystems can attenuate these impacts by releasing wet season flows slowly during drought periods and recharging groundwater aquifers during water-rich periods. Besides all these ecosystem services, aquatic ecosystems provide various sources of livelihood during extreme climate periods, floating garden for example. So, managing wetlands wisely must be part of an overall response to climate change.

4 Major Ecological Problems Faced by the Aquatic Ecosystem and Their Effect

4.1 Overexploitation of Water Resources

With its growing population, Bangladesh has more and more difficulty managing its limited water resources. An average Bangladeshi uses approximately 40 litres of water per day for household use, and the demand for irrigation water is gradually increasing (Rashid and Kabir 1998). The growing population causes an increase in demand for food. This has led Bangladesh to irrigate more crops which in turn generates a need for more water. Rice production for every individual requires over 800 m^3 of water (Rashid and Kabir 1998). When the total population of Bangladesh is considered, water demands can become enormous. Moreover, to accommodate and to fulfill the demand of human population, unplanned urban areas and industries are growing rapidly. As a result, the conversion of aquatic habitat into urban and industrial areas accelerates the degradation of the ecosystem. Another case of degradation is untreated effluent.

4.2 Degradation of Aquatic Ecosystem and Its Fishery Resources

In the last two decades, an accelerated expansion of physical infrastructure (i.e. flood control drainage and irrigation infrastructure (FCDIs), road and building structure) occurred in the flood plains and ox-bow lake areas. These infrastructures were often implemented without proper planning or without proper attention to natural water flows. The impacts of FCDI on aquatic ecosystem are the connectivity losses of water, biota and materials. There are two aspects to connectivity. Lateral connectivity is the ability of biota, water and materials to move from one distinct system, such as a flood plain lake, to another such as a river and/or tree swamp. Longitudinal connectivity is the 'upstream-downstream', or within system connectivity that is important for the movement of species within the wetland. A loss of connectivity can result in decreasing water quality. The duration and timing of periods of connection can be very important in order to allow opportunities for spawning, dispersal and migration. Species that migrate between wetland systems as part of their life cycles, such as diadromous fish, are particularly susceptible to the loss of lateral connectivity (Wetlandinfo 2012). In Bangladesh, these poorly planned infrastructures reduced valuable aquatic ecosystem resources, specially fish that cover almost 63 % of the animal protein intake of the country's population (Hussain 2010). Halls et al. (1998, 1999) found that in Bangladesh, fish yields were 50 % lower inside FCDIs compared with outside where 25 species of fish are absent or less abundant. The area under flood control and irrigation is expected to be 5.74×10^6 ha in 2010 causing a loss of 151,300 t of fish in the flood plain areas of Bangladesh (Craig et al. 2004).

Recent climate change in the form of higher temperatures, low and high rainfall, accelerate this degradation process. Alteration of aquatic ecosystems due to low rainfall and high temperature has direct and indirect adverse effects on fish through their reproduction, migration and survival (Hussain 2010). This adverse climatic effect creates a water deficiency in agriculture which ultimately causes more water extraction from the aquatic ecosystem during the dry season (Halls et al. 2001; Shankar et al. 2004, 2005).

4.3 Pollution Problems Faced by the Aquatic Ecosystem

Pollution problems originate from the extensive use of pesticides and fertilizers in agricultural landscape which results in severe eutrophication in the aquatic ecosystem. In Bangladesh, the use of fertilizers increased rapidly from the mid 1960s with the introduction of modern varieties and development of irrigation facilities. The annual urea (nitrogen-releasing fertilizer) consumption in Bangladesh was 2 million tons in the 1980s. From 1989/1990 to 1996/1997, urea consumption grew rapidly, registering an average growth rate of 7 % per year (MoA 2003). Excessive N-fertilizer application is therefore very common, especially in intensive rice, wheat, bean and vegetable producing areas. So the occurrence of nitrate pollution might be expected in groundwater and surface water. In the rivers of Bangladesh, the amount of dissolved nitrate is 13.26 mg/l (Subramanian 2008). According to WHO recommendations, the maximum allowable concentration of nitrate nitrogen should not exceed 10 mg/l (WHO 1971). The aquatic ecosystem of the whole country is the dumping ground for contaminated sediments and pollutants. However, flushing out of materials to the sea is quite slow. The result is serious deterioration of the aquatic resources, eutrophication for example (Fig. 2).

Most of the industries and factories are situated on the banks of the rivers or very close to a river system and the effluents and waste are mostly thrown directly into the river water without any treatment. The industrial effluent containing acids, heavy metals, ammonia, toxic substances, etc., is thrown directly and untreated into the water together with agrochemical substances (insecticides, pesticides, fertilizers etc.) with the huge quantity of domestic waste making the situation worse. Moreover, change in the resource use pattern is also responsible for aquatic ecosystem degradation.

4.4 Unsustainable Shrimp Cultivation in Coastal Zone

Bangladesh is blessed by goods and services provided by the coastal zone. Among all of these, shrimp aquaculture is the fastest growing economic activity in coastal areas and Bangladesh was the fifth largest producer in the world (FAO 2002). There are 1.2 million people employed in prawn and shrimp production and a further 4.8 million household members are associated with the sector (DoF 2009). However, the rapid expansion of shrimp farm development during the last decade, along with poor production technology, has caused growing concern. Its adverse effect on the coastal environment and socio-economic conditions are now responsible for the unsustainability of this sector. This farming system, requiring large land areas, has contributed most to the encroachment of agricultural land and mangrove clearance with the increased intrusion of salinity, degradation of land and destabilization of coastal ecosystems. Many scholars have already addressed these environmental issues (Islam 1999; Datta 2001) and therefore, a question is being raised about the sustainability

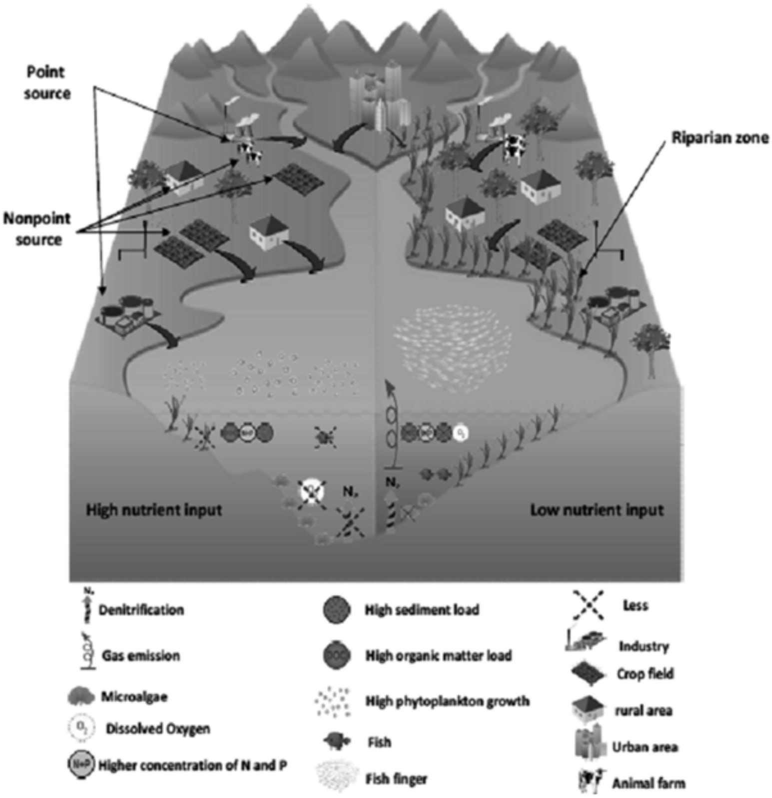

Fig. 2 Conceptual model of eutrophication problem (Source: Author)

of coastal shrimp aquaculture (Chowdhury et al. 2006; Primavera 1997). Figure 3 summarizes the overall impact of coastal shrimp farming in Bangladesh.

4.5 Impact of Water Diversion from Upstream

Bangladesh's topography is formed by three of the largest river systems in the world. It occupies the greater part of the Bengal Basin which was slowly built up by alluvial deposits carried from the adjoining mountains of the Himalayas by the Ganges–Brahmaputra river system. It is a riverine country with 230 tributaries and distributaries. The Ganges–Brahmaputra–Meghna river systems drain a total area of about 1.72 million km^2 (Ahmad et al. 2001) in India, China, Nepal, Bhutan and Bangladesh, hence the name Ganges–Brahmaputra–Meghna (GBM) river basin. A lower riparian located at the lowermost reaches of the three large rivers, Bangladesh itself, makes up only 7–8 % of the watershed (Ahmad et al. 2001). The

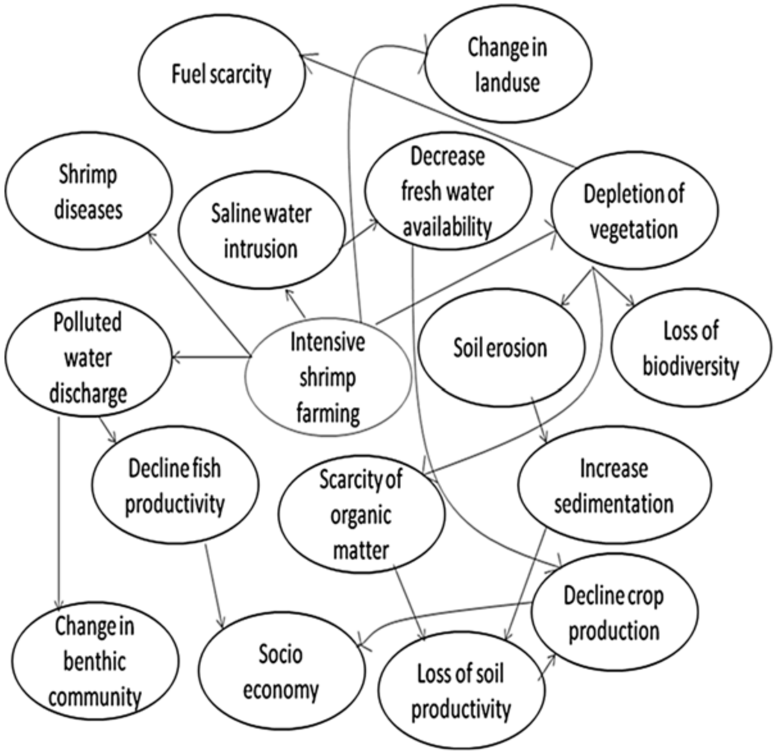

Fig. 3 Impact of coastal shrimp farming (Source: Sohel and Ullah 2012)

construction of the Farakka dam in India in 1974 has drastically reduced the natural flow of the Ganges water downstream in Padma, Bangladesh. This reduced water flow causes drought and the drying of ponds, a condition which leads to a drop in groundwater levels. It is therefore a matter of great concern. Diversion of low flows at Farakka has increased the inland penetration of salinity. Salinity levels increase rapidly and curve northwards in the area affected by the withdrawal of Ganges water in the low-flow season. This condition would lead to a massive loss of agricultural production, which would trigger the migration of at least 20 million people (Islam 2008). Reduction of the Ganges flow through the Farakka dam has severely affected the downstream river regime of the Ganges-Padma. For the Ganges-Padma River at Hardinge Bridge, the ratio of maximum and minimum discharge during pre-Farakka days and post-Farakka days are roughly 70 % and 27 % respectively, which is far greater than the ratio of 10 % of the maximum discharge required for maintaining a stable river regime (Rashid and Kabir 1998). The reduced flow of river water has reduced the major carp habitats in the Ganges River Basin in Bangladesh (Tsai and Ali 1985). The overall impact of Farakka dam of India is shown in Fig. 4.

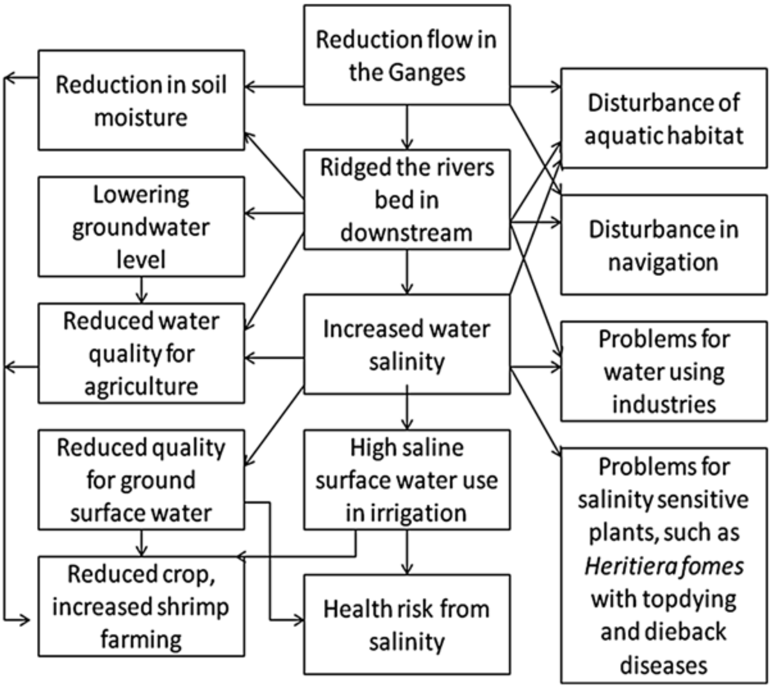

Fig. 4 Impact of Farakka dam of India (Islam 2008)

4.6 Global Climate Change Impact

Climate change is expected to increase the average temperature and spatio-temporal variability in precipitation, as well as cause a rise in sea level (Ellison 1994). The increase in temperature and variability in rainfall will put further pressure on freshwater resources and hence alter the freshwater inflows to the mangroves. Climate change also presents an increased frequency of tropical cyclones and storm surges, which may cause further changes in freshwater-seawater interactions, thereby affecting the mangroves (Ali 1995). Changed hydrological extremes due to climate change will have important implications for the design of future hydraulic structures, flood plain development, and water resource management (Cunderlik and Simonovic 2005). Maintaining hydrology, reducing pollution, controlling exotic vegetation, and protecting wetland biological diversity and integrity are important activities to maintain and improve the resiliency of wetland ecosystems so that they continue to provide important services under changed climatic conditions (Ferrati et al. 2005).

5 Present Aquatic Ecosystem Management Approach in Bangladesh

The Government of Bangladesh has taken several initiatives to tackle the water problems, such as flood control, drainage and irrigation (FCDI) projects at various scales. However, flood-control infrastructure that has been initiated by government is often ineffective because of improper planning. Bangladesh developed in 1999 the National Water Policy and an Integrated Water Management Plan (IWMP) for 2000–2025. The 1999 Water Policy assigns water-allocation decisions to local administrative authorities. The IWMP addresses three major issues: (1) efficient use of water in the face of increasing scarcity; (2) providing all people with access to sufficient, good-quality water; and (3) ecologically sustainable use of the resource (Gupta et al. 2005). The draft national water code of Bangladesh has been formulated since 2010, but not yet been enacted into law. However, there is no linkage between national development plans with aquatic ecosystem management which creates conflict among various policies. Moreover, up to now, there is no integration of forest policy with water policy, essential for watersheds. Although, in practice there is no river basin planning in Bangladesh, a major obstacle for IWRM process during the flood season is that it does not have control over the catchment areas of the major rivers, and during the dry season, no ensured minimum flow is available.

6 Towards Better Management Based on Ecohydrological Approach

6.1 Minimizing Environmental Effects of Intensive Agriculture

6.1.1 Creation of Ecotone Zone for Controlling Diffuse Pollution

Loss of nutrients from agricultural land to surface waters can cause environmental harm to fish and other aquatic organisms. Vegetated buffer zones (Fig. 5) between agricultural land and surface waters have proved to be effective filters for trapping diffuse pollutants (Syversen 2002).

6.1.2 Implementation of "Denitriphication Wall" to Protect Groundwater in Agricultural Area

Already, maximum allowable concentration of nitrate nitrogen (10 mg/l) in potable water of Bangladesh exceeds 13.26 mg/l (Subramanian 2008). Until today no initiatives have been taken to control this pollution. To solve the problem of water pollution, ecohydrology, a transdisciplinary science gives different ecological

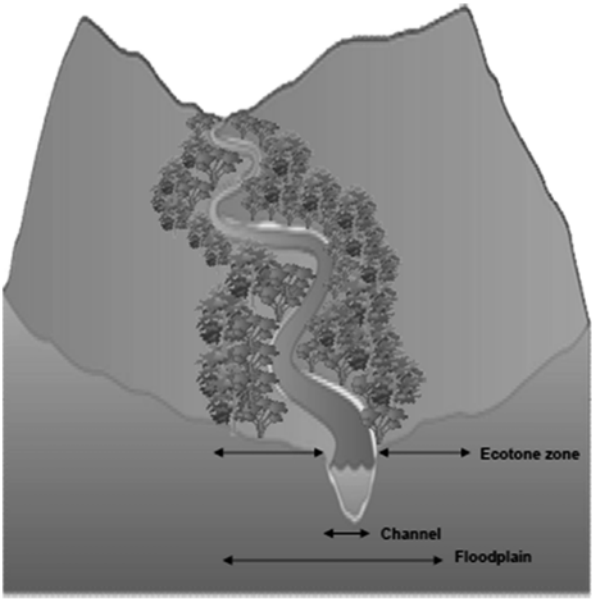

Fig. 5 Ecotone zone (Source: Author)

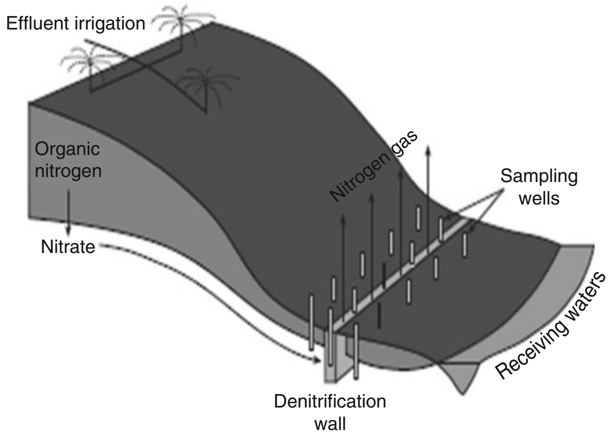

Fig. 6 Denitrification wall (Schipper et al. 2005)

biotechnology (Zalewski 2009) and "Denitrification wall" (Fig. 6) is one of them. This is a low-cost and effective tool for diminishing the nitrate inputs into groundwater and surfacewater (Schipper et al. 2005). Therefore, "Denitrification wall" could be an excellent ecological biotechnology to reduce nitrate pollution in Bangladesh.

6.1.3 Water Deficiency Control in Agricultural Landscape

Water shortage is observed in many regions of the world where low precipitation and high evapotranspiration occur and global climate change makes this condition more severe. To overcome water deficiency in the agricultural field, people put pressure on dry season surface water, resulting in the degradation of aquatic ecosystem services such as reduction of fish diversity. Proper water management in a landscape can improve these unfavorable conditions. This can be attained mainly through the creation of small water retention which increases the water retention capacity of the surrounding soil and also, particularly increases groundwater retention in the adjoining area (Jain 2006). Soil water holding capacity can also be increased by applying organic matter (Zalewski et al. 2004). Besides, to reduce evaporation in the agricultural landscape during the dry season, which becomes worse due to climate change, strip plantation or shelterbelt can be very effective because of its ability to break the wind force which helps to reduce evapotranspiration. Through this mechanism, strip plantation improves water availability in the agricultural landscape (Ryszkowski and Kędziora 2007; FAO 1989). Moreover, to reduce the irrigation water demand an improved irrigation system can be adopted for agri-crop cultivation. Irrigation systems like sprinklers can save water by about 35–40 % when compared with flood irrigation method. The systems are suitable for almost all field crops such as wheat, gram, pulses as well as vegetables, cotton, soybean, tea, coffee, tobacco, sugar cane, and other fodder crops and can be installed in residential and industrial units, hotels, resorts, public and government enterprises, playgrounds, and racecourses. While in the bed and furrow method, water is applied only in furrows. Another kind of irrigation system is drip irrigation which also uses water rationally as its target area of watering is rootzone (IUCN nd). For rice, rice intensification system (i.e. preparing high-quality land, developing nutrient-rich and unflooded nurseries, using young seedlings for early transplantation, transplanting the seedlings singly, ensuring wider spacing between seedlings, preferring compost or farmyard manure to synthetic fertilizers, weeding frequently) can be adopted as it is focused on rice cultivation by maintaining soil moisture rather than the flooded irrigation method (Fig. 7). In this method 25–50 % less water is needed than in conventional rice cultivation methods (WWF 2007).

6.2 Reducing the Impact of Flood Control Drainage and Irrigation (FCDI) Infrastructure

To reduce the impact of FCDI on the aquatic ecosystem specially on fish, adaptive management of sluice gates is essential to improve fish access to flood plain areas. Taking this into consideration, Halls et al. (1998, 1999) suggests to maximize the flow of water during the flooding period, which aids passive inward migration of fish in the floodplain areas, to open sluice gates more frequently in order to reduce the turbulence of water outside the gates during the flooding period which

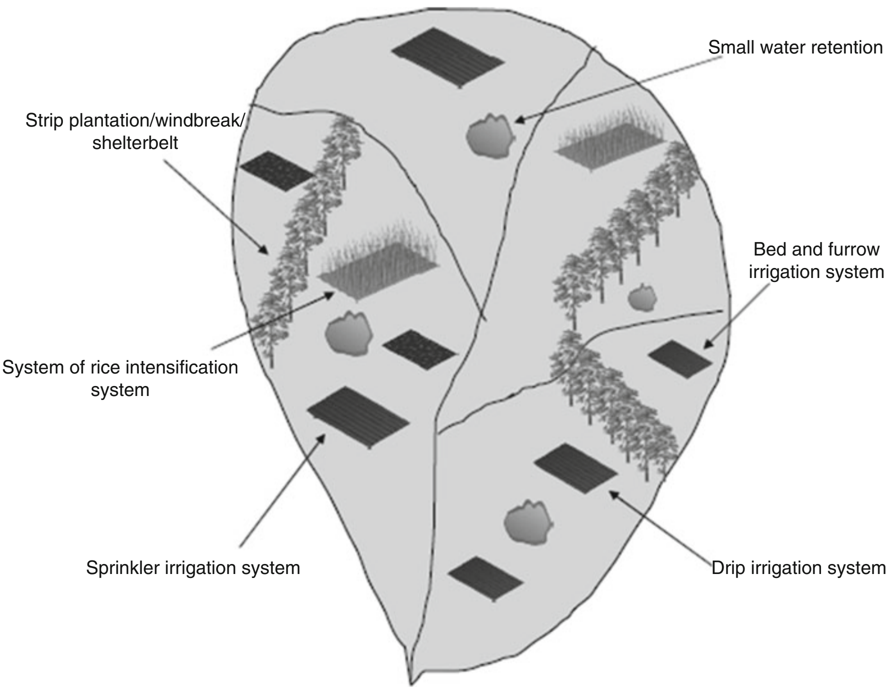

Fig. 7 Conceptual model of water deficiency control in agricultural landscape at catchment scale (Source: Author)

accelerates the smooth passage of fish and controls ebb flow from the sluice gate to attract more fish. It suggests that the best attraction velocity is about 0.1 ms^{-1}. Halls et al. (2001) and Shankar et al. (2004, 2005) have predicted that raising water levels during the dry season by as little as 0.25 m, could increase fish production by about 9 % at a loss of only 8 ha of rice production, mainly from marginal, low-lying land. Another potential strategy is to make changes to land use practices. In the dry season Boro rice production is highly dependent upon irrigation water from dry season water bodies. Switching to other dry season crops such as wheat and vegetables that are harvested several weeks before boro rice, and greater emphasis on more flood-tolerant Aman rice would also allow for earlier, more frequent opening of sluice gates for longer periods during the rising flood (Shankar et al. 2004, 2005). Apart from this, in order to reduce irrigation structure impact, various kinds of effective irrigation systems (sprinkler irrigation system, drip irrigation system) could be adopted. This strategy would help to reduce the pressure on the large demand for irrigation water. Application of such irrigation strategies is likely to become increasingly necessary in the face of climate change (Halls 2005). Such adaptive strategies are increasingly necessary where precipitation is predicted to increase during the flood season, but to decrease during the dry season in response to climate change (Halls 2005).

6.3 Overcoming the Impact of Coastal Shrimp Aquaculture

In many situations socio-economic and political constraints prevent the usage of certain technologies due to prohibitively high costs. In these cases we argue that the protection of coastal ecosystems can be better provided by applying ecohydrological principles since they provide a framework to develop low-cost technology. For example, the expansion of aquaculture ponds leads to the destruction of mangroves. This results in the loss of essential ecosystem services generated by mangroves, including fish nurseries, wildlife habitat, coastal protection, flood control, sediment trapping, water treatment, salinity intrusion into the nearby agricultural field and freshwater sources. Since mangrove vegetation plays a significant role as a buffer, it may be possible to utilize it to protect adjacent freshwater and agricultural fields (Fig. 8). This would involve the deliberate planting of salt accumulator mangrove species between production ponds and adjacent freshwater and agricultural fields (Rabhi et al. 2009; Ravindran et al. 2007). This is only one aspect of shrimp farm management using the understanding of the mangrove halophytes function as ecosystem properties is illustrative of the implementation of the second principle of ecohydrology. In addition, aquaculture pollution caused by excessive use of antibiotics and chemicals, may manifest itself as nutrients which precipitate to benthic lays at the pond bottom. This can also come from an adjacent inlet as well and later act as a source of disease. For this problem, a different solution would be required. It would be possible to install a sequential pond system with phytotechnologies to help trap sediments and nutrients which improve water quality. These nutrients can be used as fertilizers in the agricultural field. Hence, implementation of this measure focuses on the importance of the third principle of ecohydrology. The effective implementation of this approach required finding out the suitable salt accumulator halophytes that are able to absorb a good amount of nutrient, pollutant and salt.

6.4 Waste Water Treatment by Creating Constructed Wetland

Constructed wetlands (CWs) are among the recently proven efficient technologies for wastewater treatment. Compared with conventional treatment systems, constructed wetlands are low cost, easily operated and maintained, and have a strong potential for application in developing countries. In most developing countries, there are very few wastewater treatment facilities. This is mainly due to high costs of treatment processes and lack of effective environmental pollution control laws or law enforcement. CWs (Fig. 9) for wastewater treatment involve the use of engineered systems that are designed and constructed to utilize natural processes. These systems are designed to mimic natural wetland systems, utilizing wetland plants, soil, and associated shellfish, micro-organisms to remove contaminants from wastewater effluents (EPA 1993). In developed countries, CWs are used for treating

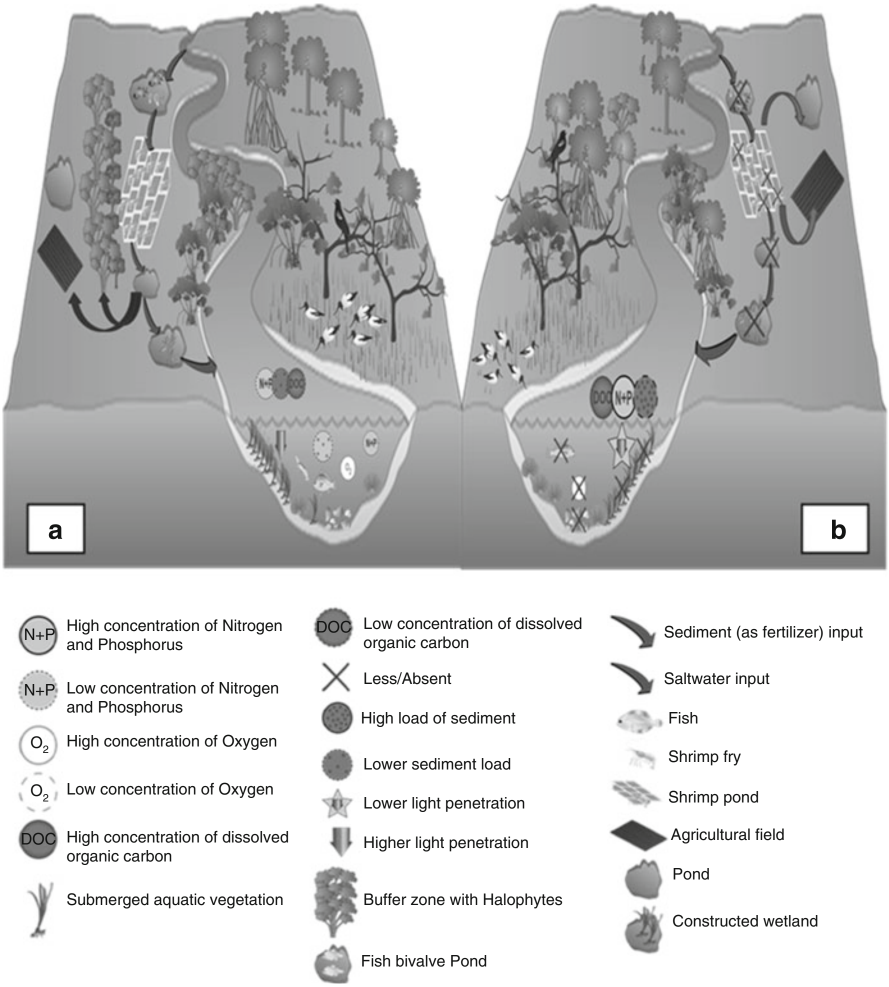

Fig. 8 Conceptual model of ESF (Ecohydrology based shrimp farming) system (**a**) and traditional (**b**) (e.g. intensive and semi-intensive) farming system (Source: Modified from Sohel and Ullah 2012)

various wastewater types e.g. domestic wastewater (Cooper et al. 1997; Schreijer et al. 1997), agricultural wastewaters (Rivera et al. 1997), landfill leachate (Trautmann et al. 1989), urban storm water (EPA 1993). CWs are also used for treating eutrophic lake waters (D'Angelo and Reddy 1994), and for the conservation of nature (Worrall et al. 1997). CWs can be an alternative for treating nitrate contaminated aquifers, denitrification of nitrified sewage effluents and irrigation return flow (Baker 1998).

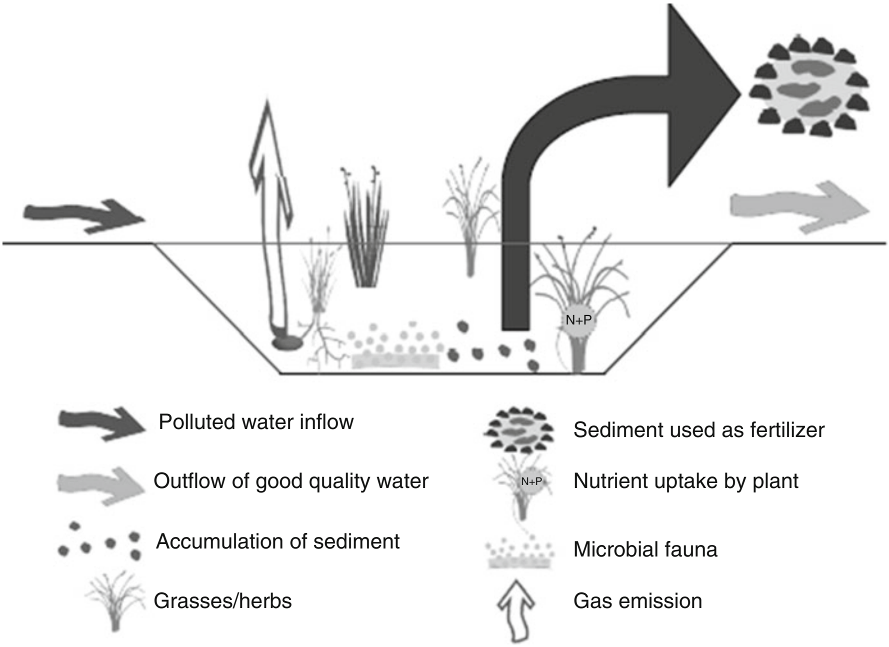

Fig. 9 Constructed wetland (Source: Author)

6.5 Probable Solution (Water Retention) to Reduce the Impact of Farakka Dam of India

Bangladesh-Nepal (Ministry of Water Resources) joint investigation report on environmental impact assessment studies in 1989, where water supply problems were recognized and suggestions given to build water storages in the upstream of the Ganges, in order to mitigate and solve the water problems in the Ganges catchment area, especially in the Bangladesh water catchment area. The study also strongly suggests to build seven water storages (Fig. 10) in Nepal with multilateral agreement. Actually those seven rivers of Nepal are carrying 71 % of fresh water annually at the Farakka Barrage in the dry season. It has been estimated that after construction of these proposed water storages in Nepal, Bangladesh can achieve extra 45,000 m^3/s water from upstream in the dry season (Islam 2008).

6.6 Facing Urbanization Impact

A broad range of direct and underlying effects of increasing urban pressures threaten the ability of aquatic habitats to provide various ecosystem services (Millennium Ecosystem Assessment 2005). These services depend to a great extent on the functioning of aquatic ecosystems and their ability to cope with high impacts,

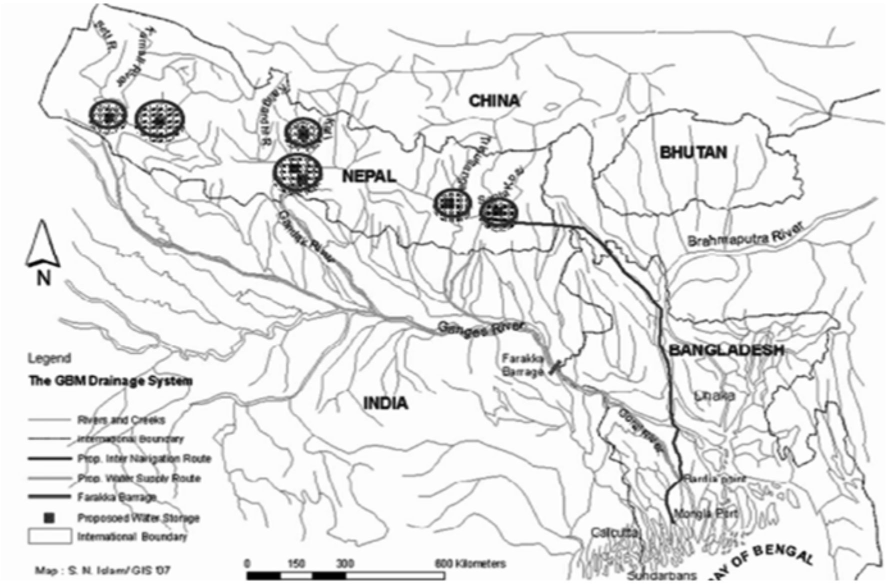

Fig. 10 Proposed upstream water supply plan (Source: Islam 2008)

determined among others by the size and distribution of available "green areas". Therefore introduction of water sensitive urban design (WSUD) which harmonizes the urban built environment and the urban water cycle, combining the functionality of water management with principles of urban design. This approach embraces an interdisciplinary cooperation of water management, urban design, and landscape planning in order to achieve integrated water resource management goals (SWITCH 2010). Adoption of ecohydrological solution for urban storm water management is such an example (Fig. 11).

7 The Possibility of Adopting Ecohydrology Based Technology in Bangladesh

Although existing management and policy guidelines have gone through extensive review and become more comprehensive than before, environmental degradation is still continuing. This may be due to the absence of proper policies and techniques on waste treatment, pollution abatement and irrigation techniques. The traditional water resource management approach is much more mechanistic and unsustainable because of financial and energy constraints. In many situations this mechanistic approach seriously reduces the role of ecological processes in moderating the water cycle. Hence intervention of the ecohydrology approach will enhance the carrying capacity of an aquatic ecosystem which will be helpful in better water resources management. In this context, it is expected that this new management approach

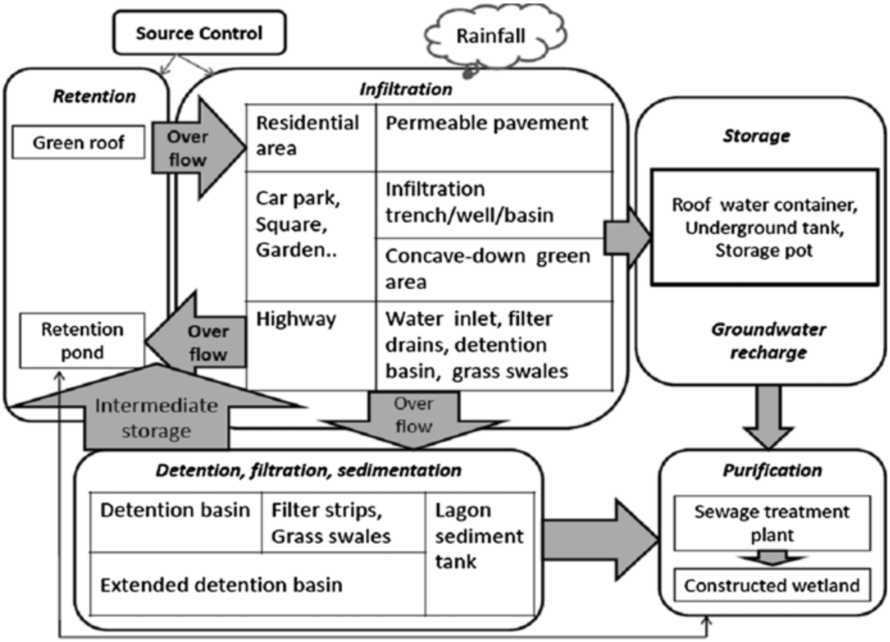

Fig. 11 An example of different methods of urban stormwater management integrated into the systematic ecohydrological solution with good urban design (Source: Li 2012)

would be cordially encouraged by the Government from its policy point of view considering environmental issues. In Bangladesh, most of the industries and shrimp farm land is operated by national and multinational investors who have access to the technology and the necessary capital to adopt new technology to gain benefits. On the other hand, most of the owners of the agricultural land are large farmers who can easily minimize the environmental effects of intensive agriculture by adopting low-cost ecohydrology technologies. If industries, farm owners and infrastructure development authorities adopt a new approach where a small portion of land is devoted to ecohydrological compensatory measures, it is expected that they will be accepted by them for two reasons. Firstly, adopting such technology will ensure the sustainability of aquatic ecosystem functions and reduce the environmental degradation which are key concerns of the relevant department of the State. This means large-scale owners should benefit from the Government in terms of greater subsidies and greater availability of state financial services. Secondly, because of environmental degradation from industry, agricultural land, shrimp farm, unplanned urban development and irrigation infrastructure, aquatic ecosystem has faced huge financial loss over the years where a cost-effective management approach is demanded. Here government can formulate policies to adopt such technologies. Therefore, adoption of an ecohydrology based management approach (Table 1) will be economically, socially and environmentally sustainable.

Table 1 Potential benefits and perceived constraints in the application of ecohydrology principles

Factor	Traditional approach	Ecohydrology based approach
Influent treatment	Absent	Present
Effluent treatment	Absent	Present
Salinity intrusion in freshwater & agricultural field	High	Might be lower
Diseases because of low water quality	Moderate to high	Might be low
Sediment trap pond	Absent	Alternatively can be used as fishery production & sediment can be used as fertilizer in agri crop production
Constructed wetland	Absent	Source of bioenergy and biodiversity
Buffer zone of halophytes (salt accumulator halophytes)	Absent	Source of bioenergy, biodiversity, protect from cyclone and climate change effect
Fish-bivalve pond	Absent	Source of fishery production & poultry feed production
Small water retention pond in agricultural filed	Absent	Source of fishery production
Shelterbelt/strip plantation	Absent	Source of fuelwood, biodiversity. Could be a source of fruits if planted with fruit bearing trees
Irrigation system (sprinkler, bed and furrow, drip)	Absent	Can consume less water for irrigation
Denitrification wall	Absent	Can reduce water pollution from agriculture filed
Social Implications	Moderate to high	Can be Less
Sustainability issue	Moderate to low	Can be high
Policy issue	Based on existing policy	Need modification of policy like waste treatment, pollution abatement, land use plan for ecohydrology based approach
Environmental impact	Moderate to high	Less
Development costs	Moderate to high	Probably high but once installed return can be more from fishery production from treatment pond
Water sensitive urban design	Absent	Improve urban water cycle

8 Conclusion

The fundamental aspect of any ecosystem management depends upon proper understanding of how a system works, how it is organized or structured, what the damaging factors are, and what the impact of those damaging factors are. Ecohydrology gives a good understanding here on interplay between biota and hydrology, and therefore provides a framework how to use ecosystem properties as a management tool for integrated water resource management. In this article, the concept of ecohydrology is introduced for the proper management of aquatic ecosystem. So potential mitigation measures in aquatic ecosystem include control of pollution, degradation of aquatic habitat, water shortage in the agricultural field, sedimentation, salinity intrusion, nutrient loading, mangrove destruction and maintaining river flow from upstream. Therefore ecohydrology based water resources management sustains the health of aquatic ecosystem services provision.

References

Ahmad QK, Biswas AK, Rangachari R, Sainju MM (2001) Ganges–Brahmaputra–Meghna region: a framework for sustainable development. University Press Limited, Dhaka

Akonda AW (1989) Bangladesh. In: Scott DA (ed) A directory of Asian wetlands. IUCN, Gland

Ali A (1995) Numerical investigation into retardation of floodwater outflow through the Meghna rice in Bangladesh due to SW monsoon wind. Estuar Coast Shelf Sci 41:689–704

Baker LA (1998) Design considerations and applications for wetland treatment of high nitrate waters. Water Sci Technol 38:389–395

Chowdhury J (1998) Some hydraulic aspects of floods in Bangladesh and their implications in planning. In: Ali MA, Hoque MM, Rahman R, Rashid S (eds) Bangladesh floods – views from home and abroad. United Press Limited, Dhaka, pp 209–217

Chowdhury MA, Shivakoti GP, Salequzzaman M (2006) A conceptual framework for the sustainability assessment procedures of the shrimp aquaculture industry in coastal Bangladesh. Int J Agric Resour Gov Ecol 5:162–184

Cooper P, Smith M, Maynard H (1997) The design and performance of a nitrifying vertical-low reed bed treatment system. Water Sci Technol 35:215–221

Craig JF, Halls AS, Barr JJF, Beand CW (2004) The Bangladesh floodplain fisheries. Fish Res 66:271–286

Cunderlik JM, Simonovic SP (2005) Hydrological extremes in a southwestern Ontario river basin under future climate conditions. Hydrol Sci J 50(4):631–654

D'Angelo EM, Reddy KR (1994) Diagnosis of organic matter in a wetland receiving hypereutrophic lake water: role of inorganic electron acceptors in nutrient release. J Environ Qual 23:937–943

Datta A (2001) Who benefits and at what costs? Expanded shrimp culture in Bangladesh, in grassroots voice. J Indig Knowl Dev 3:12–18

DoF (2009) Fishery Statistical Yearbook of Bangladesh 2007-08. Department of Fisheries, Ministry of Fisheries and Livestock, 25th edn. Matshya Bhaban, Dhaka. 42 pp

Ellison JC (1994) Climate change and seal level rise impacts on mangrove ecosystems. In: Pernetta J, Leemans R, Elder D, Humphrey S (eds) Impacts of climate change on ecosystems and species: marine and costal ecosystems. IUCN, Gland, pp 11–30

EPA (1993) Constructed wetlands for wastewater treatment and wild life habitat: 17 case studies. EPA832-R-93-005

Erwin KL (2009) Wetlands and global climate change: the role of wetland restoration in a changing world. Wetl Ecol Manag 17:71–84

FAO (1988) Agro-ecological regions of Bangladesh. FAO, Rome

FAO (1989) Arid zone forestry: a guide for field technicians. ISBN 92-5-102809-5. Food and Agriculture Organization of the United Nations, Rome

FAO (2002) FAO yearbook of fishery statistics, aquaculture production, vol 90/2. Food and Agricultural Organization of the United Nations, Rome

Ferrati R, Canziani GA, Moreno DR (2005) Estero del Ibera: hydrometeorological and hydrological characterization. Ecol Model 186:3–15

Fischer B, Turner RK, Morling P (2009) Defining and classifying ecosystem services for decision making. Ecol Econ 68:643–653

GOB (2012) Bangladesh REDD+ Readiness Roadmap. UN-REDD programme

Gopal B (1995) Limnology in developing countries, vol 1. SIL International Association for Limnology, New Delhi

Gupta AD, Babel MS, Albert X, Mark O (2005) Water sector of Bangladesh in the context of integrated water resources management: a review. Water Res Dev 21(2):385–398

Halls AS (2005) Promoting new knowledge of climate change impacts on fisheries (R8457): a review of impacts and interactions in the context of other renewable natural resource sectors. ASL, April, 2005, 46 pp

Halls AS, Hoggarth DD, Debnath D (1998) Impact of flood control schemes on river fish migrations and species assemblages in Bangladesh. J Fish Biol 53:358–380

Halls AS, Hoggarth DD, Debnath D (1999) Impacts of hydraulic engineering on the dynamics and production potential of floodplain fish populations in Bangladesh. Fish Manag Ecol 6:261–285

Halls AS, Kirkwood GP, Payne AI (2001) A dynamic pool model for floodplain-river fisheries. Ecohydrol Hydrobiol 1(3):323–339

Hossain SM (2001) Biological aspects of the coastal and marine environment of Bangladesh. Ocean Coast Manag 44:261–282

Hulme PE (2005) Adapting to climate change: is there scope for ecological management in the face of a global threat? J Appl Ecol 42:784–794

Hussain MG (2010) Freshwater fishes of Bangladesh: fisheries, biodiversity and habitat. Aquat Ecosyst Health Manag 13(1):85–93

Iqbal MS, Parvin M, Salequzzaman M, Haque E, Islam MR, Ahmed MS (2010) Ecotourism in Sundarbans and its surrounding- a possible sustainable option for alternative livelihood development. Bangladesh Res Pub J 4(3):244–253

Islam A (1999) Effects of shrimp farming on the physico-chemical and biological qualities of water. Bangladesh Agriculture University, Mymensingh

Islam MSN (2008) Cultural landscape changing due to anthropogenic influences on surface water and threats to Mangrove wetland ecosystems: a case study on the Sundarbans, Bangladesh. PhD thesis. Environmental Sciences and Process Engineering, the Brandenburg University of Technology

IUCN (nd) Efficient irrigation system. IUCN Pakistan country office, Karachi

Jain AK (2006) The concept of bio-drainage in flood prevention and the avoidance of water scarcity. In: Juszczak R, Kędziora A, Olejnik J (eds) Assessment of water retention capacity of small ponds in Wyskoć agricultural- forest catchment in western Poland. Polish J Environ Stud 16(5):685–695

Jansen EG, Dolman AJ, Jerve AM, Rahman RN (1989) The country boats of Bangladesh. University Press Ltd, Dhaka. ISBN 984 05 1108 4

Khan SM, Haq E, Huq S, Rahman AA, Rashid SMA, Ahmed H (1994) Wetlands of Bangladesh. Bangladesh Centre for Advanced Studies (BCAS), Dhaka

Li C (2012) Ecohydrology and good urban design for urban storm water-logging in Beijing, China. Ecohydrol Hydrobiol 12(4):287–300
Maltby E (1986) Waterlogged wealth: why waste the worlds wet places. Earthscan, London
Millennium Ecosystem Assessment (2005) Ecosystems and human well-being: wetlands and water synthesis. World Resources Institute, Washington, DC
Mitsch WJ, Gosselink JG (1993) Wetlands, 2nd edn. Van Nostrand Reinhold, New York
MoA (Ministry of Agriculture) (2003) Gateway to fertilizer information in Bangladesh, Ministry of Agriculture, Bangladesh Secretariat, Dhaka
Mohanta MK, Salam MA, Saha AK, Hasan A, Roy AK (2010) Effects of tannery effluents on survival and histopathological changes in different organs of Channa punctatus. Asian J Exp Biol Sci 1:294–302
National Research Council (NRC) (2005) Valuing ecosystem services: toward better environmental decision-making. National Academies Press, Washington, DC
Nishat A (1993) Freshwater wetlands in Bangladesh: status and issues. IUCN-The World Conservation Union, Dhaka
ODA (Overseas Development Administration) (1997) Impacts of flood control and drainage with or without irrigation projects on fish resources and fishing community in Bangladesh. In: Tsai CF, Ali MY (eds) Open water fisheries of Bangladesh. University Press Ltd., Dhaka
Primavera JH (1997) Socio-economic impacts of shrimp culture. Aquac Res 28:815–827
Rabhi M, Hafsi C, Lakhdar A, Hajji S, Barhoumi Z, Hamrouni MH, Abdelly C, Smaoui A (2009) Evaluation of the capacity of three halophytes to desalinize their rhizosphere as grown on saline soils under nonleaching conditions. Afr J Ecol 47:463–468
Rashid H, Kabir B (1998) Water Resources and Population Pressure in the Ganges River Basin. Case Study: Bangladesh. In: Sherbinin A, Dompka V (eds) Water and population dynamics: case studies and policy implications. American Association for the Advancement of Science and Others, Washington, DC, pp 171–194
Ravindran KC, Venkatesan K, Balakrishnan V, Chellappan KP, Balasubramanian T (2007) Restoration of saline land by halophytes for Indian soils. Soil Biol Biochem 39:26–61
Rivera R, Warren A, Curds CR, Robles E, Gutierrez A, Gallegos E, Caldeffin A (1997) The application of the root zone method for the treatment and reuse of highstrength abattoir waste in Mexico. Water Sci Technol 35:271–278
Ryszkowski L, Kędziora A (2007) Modification of water flows and nitrogen fluxes by shelterbelts. Ecol Eng 29(4):388–400
Saeed T, Afrin R, Muyeed AA, Sun G (2012) Treatment of tannery wastewater in a pilot-scale hybrid constructed wetland system in Bangladesh. Chemosphere 88:1065–1073
Schipper LA, Barkle GF, Vojvodic-Vukovic M (2005) Maximum rates of nitrate removal in a denitrification wall. J Environ Qual 34:1270–1276
Schreijer M, Xampf R, Toet S, Verhoeven J (1997) The use of constructed wetlands to upgrade treated sewage effluents before discharge to natural surface water in Texel Island, The Netherlands: pilot study. Water Sci Technol 35:231–237
Shankar B, Halls AS, Barr J (2004) Rice versus fish revisited: on the integrated management of floodplain resources in Bangladesh. Nat Res Forum 28:91–101
Shankar B, Halls AS, Barr J (2005) The effects of surface water abstraction for rice irrigation on floodplain fish production in Bangladesh. Int J Water 3(1):61–83
Shine C, Klemm CD (1999) Wetlands, water and the law: using law to advance wetland conservation and wise use. IUCN, Gland/Cambridge/Bonn, XVI +330 pp
Sohel MSI, Ullah MH (2012) Ecohydrology: a framework for overcoming the environmental impacts of shrimp aquaculture on the coastal zone of Bangladesh. Ocean Coast Manag 63:67–78
Subramanian V (2008) Nitrogen transport by rivers of south Asia. Currt Sci 94(11):1413–1418
SWITCH (2010) Sustainable water management improves tomorrow's cities' health: achievements and way forward. In: 5th SWITCH scientific conference held at Lodz, 13–16 October
Syversen N (2002) Cold-climate vegetative buffer zones as filters for surface agricultural runoff – retention of soil particles, phosphorus and nitrogen. Doctor thesis, Agricultural University of Norway

Thompson JR, Hollis G (1995) Hydrological modelling and the sustainable development of the Hadejia-Nguru wetlands, Nigeria. Hydrol Sci J 40:97–116

Tong C, Feagin RA, Lu J, Zhang X, Zhu X, Wanga W, He W (2007) Ecosystem service values and restoration in the urban Sanyang wetland of Wenzhou, China. Ecol Eng 29:249–258

Trautmann NM, Jr M, Porter KS, Hawk KC (1989) Use of artificial wetlands for treatment of municipal solid waste landfill leachate. In: Hammer DA (ed) Constructed wetlands for wastewater treatment. Lewis, Chelsea, pp 245–251

Tsai C, Ali L (1985) Open water fisheries carp management programme in Bangladesh. Fisheries Information Bulletin, 24: 51 pp. BFRSS, DoF, Dhaka

Turner RK, Georgiou S, Fisher B (2008) Valuing ecosystem services: the case of multi-functional wetlands. Earthscan, London

Wetland*info* (2012) Condition – connectivity. The Department of Environment and Resource Management, The Australian Government. http://wetlandinfo.derm.qld.gov.au/wetlands/SupportTools/MonitoringExtentAndCondition/Stressormodeloverview/Connectivity/Condition.html. Accessed 25 Dec 2012

WHO (World Health Organization) (1971) International standards for drinking-water, 3rd edn. WHO, Geneva

Worrall P, Peberdy KJ, Millet MC (1997) Constructed wetlands and nature conservation. Water Sci Technol 35:205–213

WWF (2007) More rice with less water: SRI – System of Rice Intensification. WWF International – ICRISAT Project, ICRISAT, Patancheru – 502 324, Andhra Pradesh

Zalewski M (2000) Ecohydrology – the scientific background to use ecosystem properties as management tools toward sustainability of water resources. Ecol Eng 16:1–8

Zalewski M (2009) Ecohydrology for engineering harmony between environment and society. Danube News 11(19):1–4

Zalewski M (2010) Ecohydrology for compensation of global change. Braz J Biol 70(3):689–695

Zalewski M, Wagner-Lotkowska I, Robarts RD (2004) Integrated watershed management-ecohydrology and phytotechnology – manual. UNESCO – IHP, UNESCO – ROSTE, UNEP – IETC, International Centre for Ecology PAS, Department of Applied Ecology University of Lodz, Venice Osaka, Shiga, Warsaw, Lodz, 210 p

Ecosystem Services in Estuarine Systems: Implications for Management

Rute Pinto and João Carlos Marques

Abstract Estuaries can be considered strategic locations for human settlement, supporting many anthropogenic activities. These pressures are then added to the naturally occurring ones, resulting in many cases in eutrophication processes and water pollution. The integrity of ecosystems functioning, especially concerning the benefits that attain human well-being, can come easily under pressure if not properly managed. Assuming that human well-being relies on the services provided by well-functioning ecosystems, changes in the ecological functioning of a system can have direct and indirect effects on human welfare. However, many of the interrelations between ecosystem functioning and the provision of services still require quantification in estuarine ecosystems. Therefore, it becomes fundamental to understand the complex and intricate relations in estuarine ecosystems, among ecological, social and economic factors, which are fundamental in designing and implementing management policies. Hence, this chapter tries to explore the interrelations between ecosystem functioning and services provision in estuaries, highlighting that linear relationships between biodiversity and services provision are unlikely to occur. A general overview of several pressures influencing biodiversity, functioning and integrity of estuarine systems, as well as their associated services, is also provided. Furthermore, an illustrative example is provided based on the evaluation of the trade-offs among the services provided by the Mondego Estuary (Portugal). Limitations of the methodologies used to assess estuarine services are discussed. The use of this knowledge on natural resources governance is assumed to be the key to attain the sustainable use of these systems.

Keywords Water quality improvements • Human well-being • Biodiversity-ecosystem functioning • Governance and management

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1 Introduction

The need to understand the benefits of estuarine ecosystems in both ecological and economic terms attains higher importance as pressures acting on these systems are increasing. Estuarine ecosystems provide several benefits to human societies (e.g. Costanza et al. 1997; Pinto et al. 2010; Barbier et al. 2011) through the provision of marketable goods (e.g. food provision) and non-marketable services (e.g. support of cultural uses). Nevertheless, most of these services are subject to degradation, ranging from global anthropogenic pressures (e.g. climate change) to more local/regional issues (e.g. pollution and habitat degradation) (Marques et al. 1993; Stein and Cadien 2009).

This chapter seeks to explore the connections between functions of estuaries and the corresponding benefits provided to society, which ultimately contribute to human well-being. This first section will deal with the internal functioning of estuarine systems and its relations with human well-being. Section 2 will cover the trade-offs and synergies among ecosystem services (hereafter ES) provided by estuaries, using the Mondego Estuary as an illustrative example. Finally, Sect. 3, relying on the ES approach, will focus on the governance aspects of estuarine areas.

1.1 Ecosystem Structure, Functioning and Services: How Applicable Are These Concepts to Estuarine Systems?

Transitional systems, such as estuaries, are characterised by a highly variable physical and chemical conditions, at both temporal and spatial scales. Additionally to the natural pressures acting on the system (e.g. tidal movements), anthropogenic pressures are also present, mainly due to increasing socio-economic demand. As consequence, these systems have been experiencing significant impacts, including physical and chemical alterations, habitat destruction and changes in biodiversity (Halpern et al. 2008; Borja et al. 2012), with further consequences to its resilience and stability.

Because of the complexity and integration of concepts and methodologies, it is essential to clearly define the terms used in present work:

(i) Ecosystem: '*dynamic complex of plants, animals and micro-organisms communities and their non-living environment interacting as a functional unit*' (MEA 2005);
(ii) Biodiversity: variability among living organisms and their habitats from all sources, including diversity within species, between species and within entire ecosystems (Heywood 1995);
(iii) Ecosystem functioning: refers to all biogeochemical processes occurring within a system, like the cycling of nutrients, matter or energy (Naeem 1998);
(iv) Ecological condition: refers to the ecosystems' integrity (Jorgensen et al. 2010);
(v) Stability: collective notion defined by three properties: constancy, resilience and persistence (*sensu* Grimm and Wissel 1997);

(vi) Ecosystem services: can be defined as the functions of ecosystems having value for human welfare (Fisher et al. 2009). According to the MEA (2005), ES can be classified into four categories: regulating, supporting, provisioning, and cultural;
(vii) Human well-being: human experiences that include the basic materials for human lives, freedom of choice, health, good social relations, a sense of 'cultural identity' and security (MEA 2005; Díaz et al. 2006).

The structure and functioning of an ecosystem is sustained by synergistic feedbacks between organisms and their environment (Costanza et al. 2001), determining its properties and setting limits to the types of processes occurring there (Mace and Bateman 2011). Most of this discussion regards the links between biodiversity assets, ecosystem functioning and their relation to ES provision. Several works have been conducted addressing this issue (e.g. Pimm 1984; Schwartz et al. 2000; Loreau et al. 2001; Tilman et al. 2005; Balvanera et al. 2006), although controversy still remains. Some studies claim a positive relation between biodiversity and ecosystem functioning (e.g. Tilman et al. 2005; Balvanera et al. 2006), while other works state that it is difficult to establish a direct mechanism and quantification of this linkage (e.g. Schwartz et al. 2000; Nunes and Bergh 2001; Worm et al. 2006). Nevertheless, common to all, biodiversity importance is '*argued to lie in its role in preserving ecosystem resilience, by underwriting the provision of key ecosystem functions over a range of environmental conditions*' (Perrings et al. 1995).

According to Mace and Bateman (2011), the biodiversity of a system plays a key role on ES provision, since:

(a) Biological composition of ecosystems, measured by its biodiversity, is fundamental for the ecosystem processes that underpin ES delivery (Díaz et al. 2006), acting as an insurance value of the system (more diversity buffers systems against change; Hooper et al. 2005) and offering more options for the future (Yachi and Loreau 1999);
(b) Genetic and species biological diversity may directly supply some goods;
(c) Many components of biodiversity are valued by people for altruistic reasons (e.g. appreciation of wildlife).

However, it is important to highlight that, biodiversity, *per se*, is not a service (Haines-Young and Potschin 2010), although biodiversity conservation might be. In fact, the loss of biodiversity and changes in ecosystem functioning may be an explanation to declining water quality, decreased coastal protection from flooding and storm events (Barbier et al. 2011).

An illustrative example of this complexity and of how physical, chemical and biological assets underpin estuarine functioning, is the reduction, or even disappearance, of macrophytes or seagrasses caused possibly by competition with green macroalgae. These situations are often caused by nutrient enrichment of the water column, caused by anthropogenic activities, combined with high water residence times and good light conditions. The changes in macrophytes or seagrasses habitats and communities can lead to changes in ecosystem functions and trophic structure (e.g. uncoupling of biogeochemical sedimentary cycles or even changes in the abundance of benthic fauna) (Marques et al. 1993; Valiela et al. 1997; Baeta et al. 2009).

1.2 Estuarine Services: Which Relations to Human Well-Being?

Estuaries are considered as one of the most valuable and productive ecosystems around the world. Costanza et al. (1997) estimated that these wetlands had an overall value of 22,832$ $ha^{-1}year^{-1}$, although more recent studies conclude that this value may be much higher (Jørgensen 2010; Barbier et al. 2011). Many of the world's largest urban areas (22 of the 32 largest cities) are located around estuaries (Ross 1995) and globally around 71 % of the world's coastal population is concentrated within 50 km of an estuary (Agardy et al. 2005). In Portugal, coastal areas account for only 8 % of the continental area, although 76 % of the population is concentrated in these areas (OECD 2011), with a density which is twice as high as the average for continental Portugal (244.2–112.4 inhabitants'/km^2; Pinho 2007; OECD 2011). Situations like the ones described may create massive pressures on natural resources (due to activities' expansion, development, nutrients inputs, among others), impacting around 90 % of previously important species and destroying roughly 65 % of seagrass and wetland habitats, while degrading water quality and accelerating species invasion in estuaries (Lotze et al. 2006; Koch et al. 2009; O'Higgins et al. 2010). These trends, following Worm et al. (2006), are known to affect services valued by societies, like fisheries (33 % decline), nursery habitats (69 % decline), and purification services (63 % decline).

Thus, when addressing natural resources management, the overarching question is 'how to deal with the relation between environmental quality/biodiversity assets and services provision?'

To address these complex relations, the Convention for Biological Diversity (2004) asks for a clear integration, under the Ecosystem Approach (EA), of all services provided to people, by biodiversity and ecosystems, into a holistic framework. Assuming the EA holistic perspective (Maltby 2006; de Jonge 2007), ES can act as the link between natural assets and human benefits. This approach defends an integration of the ecological, economic and socio-cultural perspectives when valuing a system (de Groot et al. 2002; Farber et al. 2002; MEA 2005), providing a methodological framework for the integration of ecosystems management (de Jonge 2007). ES clearly have an ecological and a socio-economic aspect and it is their interdependencies that need to be clarified (Mace and Bateman 2011).

Once the concepts and approaches are well-established, some questions still remain: 'How can the importance of these services be measured, to get a basis for decision-making processes?' or even 'How robust and reliable are the estimated values of ES?' In this regard, the Millennium Ecosystem Assessment (MEA 2003, 2005) proposed a simple conceptual guiding principle:

$$\text{biodiversity} \rightleftarrows \text{ecosystem functioning} \rightleftarrows \text{ecosystem services} \rightleftarrows \text{human well-being}$$

where each arrow represents a causal relationship (Naeem et al. 2009; Pinto et al. 2014) and where ES may be seen as functions that ultimately benefit humans (Costanza et al. 1997; Daily 1997; Naeem et al. 2009). This framework relies on the assumption that increased biodiversity augments, at least to a certain extent, ecosystem functioning, which improves ES and may eventually improve human well-being. A number of studies have attempted to link explicitly or implicitly the biological composition of ecosystems, given by biodiversity proxies, to the stability of ecosystem functioning (e.g. MacArthur 1955; Odum 1959; May 1972; Pimm 1984; Loreau et al. 2001; Balvanera et al. 2006; Godbold et al. 2011). These studies assumed that such links may have a determinant role in ES delivery (e.g. Costanza et al. 1997; Tilman et al. 2006; Díaz et al. 2007; Haines-Young and Potschin 2010). Typically, researchers have considered that if ecosystems' biodiversity was linked with functioning, it would imply that ES could be related to human well-being (Naeem et al. 2009). Relying on previous categorizations (Costanza et al. 1997; Daily 1997; de Groot et al. 2002; MEA 2003), 20 ES can be highlighted for estuarine systems (Table 1), grouped into three categories: provision, regulation, and cultural.

Table 1 Inventory of estuarine ecosystem ES following the MEA classification (MEA 2003) and the mostly used valuation method for each service (After Pinto et al. 2010)

Ecosystem service category	Ecosystem service	Valuation method
Provision services	Food production	MP*; PL
	Raw materials	MP
	Renewable energy	CVM
	Ornamental resources	MP
Cultural services	Aesthetic resources	CVM; BT; HP
	Tourism	MP*; CVM; TC
	Recreation activities	CVM; MP*; TC
	Cognitive values	CVM
	Cultural heritage	CVM; HP
	Non-use values	CVM
Regulation services	Gas and climate control	PF
	Disturbance regulation	PF; AC; RRC; DC
	Carbon sequestration	PF
	Bioremediation	RRC; PF
	Soil erosion prevention	AC; RRC; DC
	Nurseries	PF; PL; AC; RRC; DC
	Habitat provision	PF; AC; RRC; DC
	Nutrient cycling	PF
	Water supply	MP
	Water quality	CVM; AC; RRC; DC

Note: *MP* market prices method, *PL* productivity loss, *AC* avoided cost, *TC* travel cost, *RRC* replacement and restoration costs, *HP* hedonic pricing, *CVM* contingent valuation method, *DC* damage costs, *PF* production functions

Provision services include all the tangible goods that can be obtained from ecosystems, like food or fibre. Cultural services are the non-material benefits people obtain from ecosystems, such as aesthetic values or recreation activities. Regulation services encompass the benefits obtained from the regulation of ecosystem functioning, e.g. water or climate regulation.

The supply of the provision goods relies on several supporting and regulating services. Besides, there are substantial connections between the provisioning and cultural services, due to the historical and social aspects of food production. A close relation among the several categories of services is verified when performing this kind of assessments (Pinto et al. 2010). Among these categories, the provision and cultural services are simple to recognize and relatively easy to attribute a value. However, regulation services are more difficult to understand and quantify, often becoming undervalued (Wattage 2002).

2 Synergies and Trade-Offs Among ES: The Mondego Estuary Case-Study

2.1 Study-Site Description

2.1.1 Mondego Basin

The Mondego River, located in central Portugal (39° 46′ and 40° 48′N; 7° 14′ and 8° 52′W; Lima and Lima 2002), has a catchment area of approximately 6,670 km^2 and its basin is highly diverse in topography, hydrology and land use. Its functional structure ranges from mountainous areas to a large alluvial plain discharging into the Atlantic Ocean (Marques et al. 1997; Graça et al. 2002). The Lower Mondego region (with a total area of 250 km^2) connects the mountain river with the ocean and consists of open valleys and plains. This region also includes the Mondego Estuary (Fig. 1). The Mondego is the largest Portuguese river which entire watershed is contained in national territory.

The Mondego Basin has a highly natural variability in environmental and social conditions. The downstream portions of the catchment area are densely populated while the upper and middle basin regions experience low to moderate human impacts (Feio et al. 2009). The basin, supporting over half a million inhabitants (Marques et al. 2003; Pinto et al. 2013), covers a wide range of uses, ranging from intensive agriculture to industry. The whole system is characterised by activities belonging to the secondary and tertiary economic sectors, however, in its terminal part (Lower Mondego region) a strong pressure from activities belonging to the primary sector are also felt (Pinto et al. 2010). Other impacts are dams and barrages built along the river course that may influence the environmental quality of running waters (Feio et al. 2009). Among these water uses, some can have a direct effect on water quantity and quality, such as water extraction for agricultural fields or

Fig. 1 Mondego watershed and geographic sampling levels considered: (1) Mondego Estuary; (2) Lower Mondego; (3) Mondego Basin; (4) Portugal

wastewater discharges from the industry; while other uses, such as tourism/recreation activities (e.g. sport fishing or canoeing), have an indirect impact on the system quality. Overall, the water quality of the Mondego watershed has been classified as Moderate (INAG 2009 data). Data review from several studies conducted on the system, allow saying that water quality has been improving over the years, allied to biodiversity indicators enhancement (e.g. Marques et al. 1997; Graça et al. 2002; Feio et al. 2009).

2.1.2 Mondego Estuary

The terminal part of the basin comprehends the Mondego Estuary, with a total area of 7.2 km^2 (Fig. 1). In this region, activities belonging to the primary sector (e.g. agriculture and fisheries) and touristic activities play a major role, while supporting higher population densities (around 167 inhabitants/km^2) than the rest of the basin (circa 90 inhabitants/km^2). Moreover, the water flowing into the estuary has been loaded for decades with nutrients, mainly due to the upstream-downstream effects caused by activities taking place along the river watershed (Pinto et al. 2013). In the

Lower Mondego, strong pressures are caused by the primary economic sector (15,000 ha of highly productive agriculture of mainly rice) and by harbour-related activities in Figueira da Foz.

2.2 The Ecosystem Services Approach (ESA) Application to the Mondego Catchment Area

ESA has been applied to natural resources management to ensure that the value of ecosystems' assets is fully taken into consideration in decision-making processes. Nevertheless, this approach faces several criticisms mainly related to its anthropocentric approach and due to the perception that it puts a price-tag on nature (e.g. Polasky and Segerson 2009; Blancher et al. 2011). Nevertheless, according to Polasky and Segerson (2009) '*a key motivation for conducting valuation of ES is to improve public policy decisions*'. This requires the (e)valuation of ecosystems and of their benefits. A key point is the scale at which the evaluation is being conducted, both spatially and temporally, that should be incorporated in the initial steps of these assessments (Hein et al. 2006).

Five steps are considered essential to perform an accurate evaluation of estuarine services: (1) Ecological evaluation of the system, (2) Inventory, (3) Prioritization, (4) Assessment, (5) Valuation.

2.2.1 Ecological Evaluation of Estuarine Integrity

An accurate evaluation of a systems' integrity is fundamental for ES assessments, since it allows dealing with the functional and structural ecosystems' indicators. The aim of this kind of analysis is to allow applying a holistic indicator to represent the different ecosystem states and conditions (Mueller 2005), estimating the environmental effects of altered use patterns on biota. A key aspect is to guarantee that an accurate description of the system under study and current conditions is done, to guarantee a comprehensive knowledge of the trade-offs and synergies taking place.

A large amount of information regarding the Mondego Basin's physical structure and functioning is available in the literature (e.g. Marques et al. 1997, 2003; Graça et al. 2002; Feio et al. 2009). Most studies have focused on the macroinvertebrate communities' biotic integrity as well as on the water quality status, mainly in the scope of the WFD implementation. To estimate biodiversity and water quality in the Mondego Estuary (local scale), the chosen dataset was provided by a programme monitoring the estuarine subtidal soft bottom communities. These data characterised the local system with regard to species composition/abundance and water and sediment physicochemical parameters. Samplings were carried out at 14 stations along the two estuarine arms during spring in 1990, 1992, 1998, 2000 and every year from 2002 to 2006: Euhaline estuarine sand, North arm polyhaline sand, South arm polyhaline sand, South arm polyhaline muddy sand (Teixeira et al. 2009)

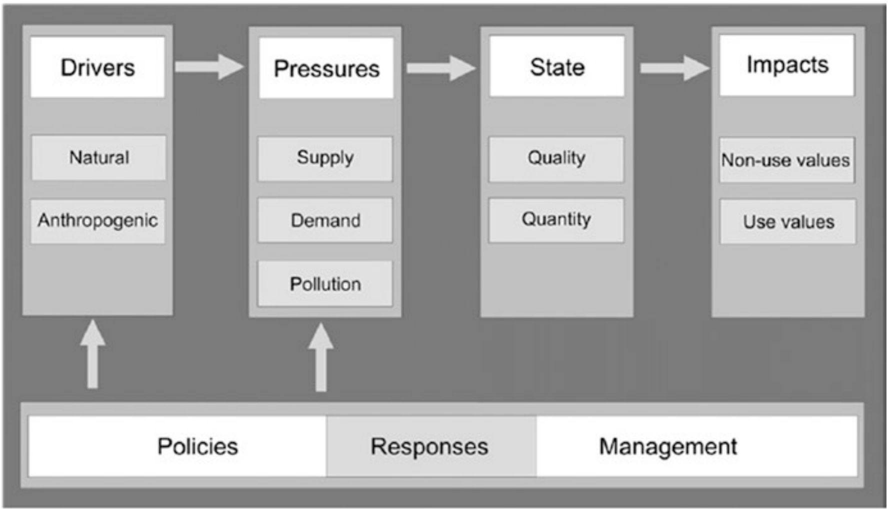

Fig. 2 DPSIR approach applied for the Mondego Estuary: identification of (natural and anthropogenic) Drivers and of the main Pressures occurring on the supply, demand and pollution of aquatic resources. This allowed for a qualitative and quantitative Status evaluation and for measuring the Impacts on the use and non-use values of the system. The societal Responses meant to improve the system should take into account both implemented policies and management actions taken (past and future) (After Pinto et al. 2013)

(Fig. 2). For the biodiversity analysis, a 1-mm mesh screen was used to sieve the samples and the collected organisms were identified and counted. To estimate the ecological condition, the Benthic Assessment Tool (BAT) was applied (Teixeira et al. 2008, 2009) based on the ecological quality of benthic macroinvertebrates and following the reference conditions proposed for Portuguese transitional water bodies (Teixeira et al. 2009) (Table 2A). The water quality in the estuary was characterised by the concentrations of dissolved nutrients (nitrate-nitrogen, nitrite-nitrogen and phosphorus) in surface and bottom water samples (Strickland and Parsons 1972; APHA 1980). The assessment of nitrite + nitrate (mmol/l^{1}) and phosphate (mmol/l^{1}) levels followed the EEA proposal (EEA 1999) (Table 2B) prepared by the European Topic Centre on Inland Waters (ETC/IW) for transition, coastal and marine waters.

2.2.2 Inventory

Several services provided by wetlands have been identified (Costanza et al. 1997; Acharya 2000; Atkins and Burdon 2006). From these available sets of services we considered two main factors that determine the Mondego Estuary services: the importance of its natural resources stock to local populations (i.e. estimation of their dependency upon the system) and the ecological importance of the system to the

Table 2 Reference conditions for benthic quality assessment (A) and water quality status (B)

	Euhaline	Polyhaline sand	Polyhaline muddy
(A) High statuses for the Margalef, Shannon-Wiener and AMBI indices used in BAT to assess the different estuarine stretches of Portuguese transitional water bodies (After Teixeira et al. 2009)			
Margalef	5.0	4.0	3.0
Shannon-Wiener (bits/ind)	4.1	4.0	3.8
AMBI	0.8	1–1.5	2.4
Quality status	$NO_2^- + NO_3^- \left(\propto mol L^{-1}\right)$		$PO_4^- \left(\propto mol L^{-1}\right)$
(B) European Environmental Agency criteria for assessing nutrient levels in transition coastal and marine waters (EEA 1999)			
Good	<6.5		<0.5
Fair	6.5–9		0.5–0.7
Poor	9–16		0.7–1.1
Bad	>16		>1.1

intrinsic biodiversity. For the inventory assessment, several methods were considered to evaluate ES. Using the total economic value (TEV) method, ES can be divided into use values and non-use values (Turner 2000; de Groot et al. 2002; Atkins and Burdon 2006). Generally, wetlands' non-use values (i.e. existence or bequest values) may be estimated through the use of contingent valuation methods (CVM). Within the wetlands' use values, three main categories can be identified: direct use values, which include services such as food production; indirect use values, e.g. aesthetic values; and option values, ensuring that a resource will be available for future use. Usually, the direct use values can be calculated through methods such as market analysis prices (MP), productivity loss (PL), hedonic pricing (HP), travel cost (TC), replacement/restoration costs (RRC), or even CVM. The indirect use values can be estimated through such methods as the damage costs (DC), production function (PF), HP, RRC, or CVM. The option values can also be assessed through the use of the CVM technique.

2.2.3 Prioritization

Although controversial, undertaking a prioritization process of ES can play an important role in these assessments. Based on the inventory, the first step is to identify the services more relevant to (local or global) communities, depending on the scale we are working at. Two scales were assessed: at the regional scale (Lower Mondego) the interaction and overlap between agricultural activities and water quality supply was considered; while at the local scale (Mondego Estuary) the interdependence between the four main assets (food production, recreation, water quality and biodiversity) was analysed. Although this selection may seem limited, it was done for two main reasons: (i) these services have a greater economic or social importance for the region and (ii) data availability. For more details, please see Pinto et al. (2010).

2.2.4 Assessment

The Driver-Pressure-State-Impact-Response (DPSIR; Fig. 2) framework has proved to be a useful tool to assess the sustainable development issues in coastal zones (e.g. de Jonge et al. 2012). The DPSIR framework can be used as an analytical tool to trace changes in the transitional wetlands structure and function over time in relation to human uses. The main driving forces have to be identified and their impacts on the system functioning evaluated. The scale issue should also be considered in the drivers and pressures trend analysis. Considering the Mondego example, three successively higher geographic scales were considered: Mondego Estuary, Lower Mondego and Mondego Basin (Fig. 1). This approach was used to assess water condition and status in the most seaward part of the Mondego River and to make inferences about the effects of upstream activities on the estuarine region.

2.2.5 Valuation

Clearly defining functions, services and benefits is fundamental to make ecosystem assessments more explicit to economic valuation (de Groot et al. 2010). Regular markets do not exist or fail to accomplish some service's full value, especially when dealing with public goods. Stated preference techniques frequently used to estimate both use and non-use values (Mitchell and Carson 1989; Spurgeon 1992; Carson 2000), being especially useful to estimate public goods values (Wattage 2002). CVM, for instance, have been widely used to examine water quality improvements in several contexts (e.g. Mitchell and Carson 1989; Söderqvist 1996; Atkins and Burdon 2006). CVM allow to estimate respondents' preferences by directly asking how much they would be willing to pay (WTP) or willing to accept (WTA) for changes in the provision of a service (Mitchell and Carson 1989). This method relies on the construction of a hypothetical market for the surveyed good. Thus, the elicited WTP amount is contingent upon the hypothetical market presented to the respondent (Mitchell and Carson 1989; Wattage 2002). Hence, through CVM surveys implementation is possible to examine irreversible changes of a system provision capacity and to evaluate the direct (e.g. recreational fishing) and indirect use values (e.g. improved water quality), while also promoting the measurement of the associated option use and non-use values within a system (Birol et al. 2006). The investigation of society' preferences is important since the development of EU water legislation, like the WFD, is imposing significant costs (Elliott and de Jonge 2002). Therefore, it becomes crucial to estimate the social awareness for water-related environmental problems and the economic importance of water-quality improvements to human well-being. In the specific chapter the economic valuation of estuarine services was, nevertheless, not carried out.

2.3 Results and Discussion

2.3.1 Ecological Evaluation

Due to the lack of data regarding the larger scales, only the Mondego Estuary was analysed in the biodiversity assessment (Table 3). The North arm (Euhaline estuarine, North arm polyhaline sand) presented a strong biodiversity decline in 1992 followed by some recovery. From 1998 onwards, the estuarine mouth and North arm showed significant improvement from moderate to good ecological status. The South arm also presented a significant decline in biodiversity until 1998. After 1998, following the implementation of several experimental mitigation measures (Teixeira et al. 2009), the system's biodiversity began to show signs of improvement. As a whole, a gradual enhancement of the system's ecological condition has been observed.

2.3.2 Inventory

A preliminary assessment of the services provided by the Mondego Basin was carried out, having as bottom-line the system's ecological quality. An inclusive set of ES supplied by the Mondego's catchment area was assessed (Table 1), based on the system knowledge and literature review. Within the provisioning category, services such as food production, raw materials and renewable energy could be identified. Such services as aesthetic resources, tourism/recreation activities and cognitive values were found within the cultural category. Within the regulating category, was possible to identify services as gas and climate regulation, disturbance regulation, carbon sequestration, habitat provision for certain species, and water supply. This

Table 3 EEA classification with respect to the nitrate + nitrite and phosphate water concentrations (*surface and bottom*), as well as BAT assessment of Ecological Quality Status (EQS) based on macrofaunal communities during spring months (April–June) from 1990 to 2006 in four estuarine areas (*E* Euhaline estuarine, *PNA* Polyhaline North Arm, *PSSA* Polyhaline Sand South Arm, *PMSA* Polyhaline Muddy South Arm). EEA classifications – *Red* Bad, *Yellow* Poor, *Green* Fair, *Blue* Good. EQS classifications: *Orange* Poor, *Yellow* Moderate, *Green* Good, *Blue* Excellent

	NO_3+NO_2								PO_4									Benthic Macrofauna (EQS)			
	Surface water				Bottom water				Surface water				Bottom water								
	E	PNA	PSSA	PMSA	E	PNA	PSSA	PMSA	E	PNA	PSSA	PMSA	E	PNA	PSSA	PMSA		E	PNA	PSSA	PMSA
90					1.5	5.3	5.9	4.9					0.96	0.59	0.92	1.19	90	M	G	G	M
91																	91				
92					5.9	15.1	16.0	11.0					7.21	12.47	15.52	11.75	92	M	M	G	G
93																	93				
94																	94				
95																	95				
96																	96				
97																	97				
98					15.9	31.1	16.6	22.6					0.55	0.57	0.24	0.66	98	G	G	P	M
99																	99				
00					1.5	23.1	18.4	16.5					1.02	2.25	2.08	2.15	00	G	M	G	G
01																	01				
02					12.7	13.8	8.9	8.2					3.20	3.70	3.71	3.46	02	G	G	M	G
03	10.2	18.2	5.7	21.6	4.5	8.2	4.4	13.6	0.96	1.39	0.71	1.80	0.79	1.03	0.67	1.70	03	M	M	M	G
04	5.8	11.6	3.8	14.6	1.8	7.1	2.9	13.2	0.68	0.96	0.64	2.13	0.39	1.11	0.51	1.85	04	G	G	M	G
05	5.9	8.1	5.9	11.5	6.6	7.3	5.7	12.2	0.55	0.88	0.72	2.00	0.49	0.80	0.71	1.89	05	G	G	G	G
06	17.3	20.4	12.1	20.0	12.8	10.5	13.7	19.9	1.18	1.18	0.87	1.99	0.58	0.78	0.78	1.94	06	G	M	M	H

kind of approach will enable decision-makers to consider several alternatives for management based on the available parameters (e.g. uncertainty of results).

In the Mondego Basin, population pressure has triggered changes in water uses. Shipping, fishing, agriculture and recreation were the most important uses reported. Across the entire basin, there was a positive trend among all the economic sectors considered; nevertheless, through a scale refinement approach, was possible to recognise a negative trend for the secondary sector at the Lower Mondego and Mondego Estuary scales. In the estuary, there was a decrease in the activity of the primary sector (reflecting the abandonment of activities like agriculture or fisheries) combined with an increase of services provision, mainly tourism/recreational activities. Regarding the water resources, the variables showed an increasing trend across the three assessed scales. Not surprisingly, these variables followed the population data trend. Industrial water use and water extraction for domestic usage and irrigation also appear to play an important role at each of the three scales analysed. Land use have a substantial impact on water quantity or quality. Currently, the water quality enhancement seems to be crucial as it influences, to a large extent, the trends of all of the other variables. For instance, the decline in fish farming production appears to be mainly related to decreasing water quality. As population increases, so does the related activities promoting higher water uses and effluents production. Moreover, the higher levels of nitrates and nitrites on surface waters, compared to bottom conditions, suggest that the main source of these nutrients lies upstream of the study-area. The systems' nutrient enrichment, and the subsequent eutrophication effects, is one of the possible factors affecting the estuarine' productivity. Overall, human activities cause a sequence of environmental damages and stresses that may alter the ecosystem's natural processes and thus alter its equilibrium. Based on this specific assessment, several factors were identified as promoters of changes, such as high nutrient concentrations, land occupation rates and habitat maintenance.

2.3.3 Prioritization

Three services of the previous inventory were considered has having a prominent role in the ecosystem: food production, recreation and water quality maintenance, as well as for their relation with the biodiversity assets. For this trade-offs evaluation, only the Mondego Estuary scale was taken into account.

On top of the environmental challenges, social, cultural and economic problems overlap. Activities are never isolated or result from cause-effect linear relations; they interact and compete for area and resources. They sum up effects and produce a complex network of interrelations, which becomes even more difficult to analyse than each ES alone (Fig. 3). A typical example of such interactions between different activities can be seen in the Mondego Estuary. The selection of only three services for this approach may reduce the validity of valuations for decision-making; nevertheless, the intent was to compile an overview of some of the region's services and examine the interrelations among them.

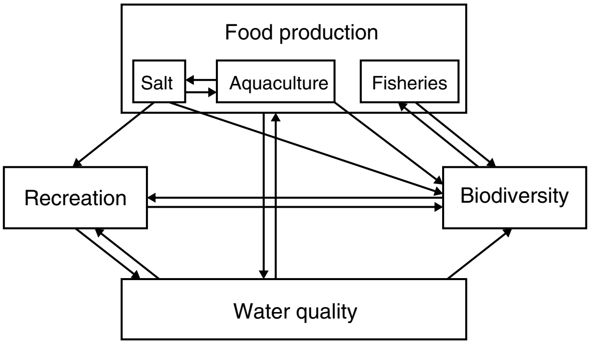

Fig. 3 Interrelations between the different services in the Mondego Estuary (after Pinto et al. 2010)

The increasing nutrient concentration in the water, essentially due to agricultural runoff, influences aquaculture production and affects the aquatic communities' diversity. Impoverished benthic communities, which serve as food for many fish species, might eventually cause a decrease in fish production. In general, due to this intrinsic and complex network of interrelations and interdependencies, any measure undertaken to improve one ES in isolation will directly or indirectly have repercussions on the others, as also demonstrated by, for example, Acharya (2000) and Atkins and Burdon (2006).

Water management plays a crucial role in the provision and delivery of all services considered for the Mondego Estuary. Therefore, it becomes essential to simultaneously achieve economic efficiency, environmental protection and sustainability within a system. Along with water management, an accurate biodiversity asset evaluation is needed to better understand which ES are essential for human well-being.

2.3.4 Assessment

Relying on the competing uses of estuarine resources, an integration of ecological value, water uses and ES into the DPSIR framework was used as an added-value for policy-making and management. Supporting and regulatory services (e.g. water supply) are essential to sustain crucial ecosystem processes and functions; whereas the water required for human activities (water demand) is an essential ES. Through DPSIR, the main changes in the Mondego Estuary were outlined and causes and effects described. Within the Mondego Basin the main water consumers are agriculture, industry and households. Baseline scenarios predict an increase in water use, mainly by the tourism sector. This analysis illustrates that pressures caused by human population growth and related activities gradually increased over the studied period. Land-use patterns, diversion of freshwater flows, water pollution and morphological interventions directly caused physical, chemical and biological modification and degradation. Consequently, this led to negative ecological and socio-economic impacts, like the eutrophication symptoms.

A progressive increase in social drivers occurred during the studied years, concomitantly with a decrease in some economic drivers (e.g. primary sector proxies). The available data reflected the estimated changes in land-use patterns for ES (baseline scenario). The selected indicators showed that agricultural area occupies the largest portion surrounding the estuary. This area has, however, decreased at an annual average rate of 5.2 %. In contrast, the urban area has increased at an average rate of almost 8 %/year (Table 4). Assuming these trends for 2015, it is possible to see that special attention has to be given to activities and pressures coming from the social drivers, water uses and activities as tourism. These findings suggest that to implement effective management policies, becomes fundamental to have a clear understanding of the complex and intricate trade-offs among ecological, social and economic needs.

3 Governance of Estuarine Areas: Assessing the Role of ES

EU environmental policies introduced into member countries' national policies the concept of ES as a tool for mainstreaming the prevention of biodiversity losses (e.g. EC 2011) and water quality improvement (e.g. EC 2000). That is for instance the case of the Portuguese Water Law (Law 58/2005). Undoubtedly in an attempt to make explicit the value of natural resources to society, the term 'ecosystem services' was more recently also integrated in the Portuguese Government management policies (DL 142/2008) as a key factor for the assessment and preservation of natural assets. The assumption is that biodiversity and water quality play a fundamental role in determining both ecosystem functioning and the provisioning of ES, which underpins human well-being.

Despite the usefulness and wide application of the ES approach, the fundamental issue is to ensure its accurate integration into policies' design and management actions.

Ecosystems influence and are influenced by human activities. Biodiversity assets are essential for the self-organization capacity of ecosystems (Levin 1999), both to absorb disturbances and to re-organize after them (Folke et al. 2004). The functional characteristics of the species composing a system are considered a key factor for the maintenance of ecological stability/integrity (e.g. Díaz et al. 2006; Naeem et al. 2009) and for the provision of services (e.g. Luck et al. 2003). Due to this complex and multi-causal dynamics inherent to ecosystems, uncertainty is an intrinsic characteristic of ES assessments (Martín-López et al. 2009; Pinto et al. 2013). The complexity of the links between biodiversity and ecosystem functioning makes more complicated to trace the impact of changes in biodiversity assets through the variations in ES outputs. In fact, while demands for certain services increase, like food provision, human actions can determine the inherent ecosystem capacity to continue providing those services at the same levels. Several studies have demonstrated the relation between biodiversity and several types of ES, showing that the maximization of some services (usually provisioning services) came at the expense of

Table 4 Baseline 2015 scenario for the Mondego Estuary region, following the 2006 data and posterior trends considering selected indicators of natural and anthropogenic drivers

Drivers		Selected indicator	1994 data	2006 data	Trends (%/year)	2015 scenario
Natural	Natural					
	Invasive species					
	Extreme events					
	Floods					
	Drought					
Anthropogenic	Social population	Area (ha)	661.7[c]	1773.9[d]	7.9	6110.4
		Total population (n°)	61,830	63,372	0.16	64,386
		Population density (hab/Km^2)	163	167	0.17	170.04
		Households number (n°)	22,976	42,685	0.48	44,734
		GDP per capita	95.2	99.3	1.10	110.2
	Economic agriculture	Area (ha)[a]	272,107.7[c]	124,917.2[d]	−5.2	12,491.7
		Employment (n°)[b]	330,955	114,528	−7.27	31,266
		Explorations (n°)[b]	128,119	96,253	−2.77	69,591
		Output (t)[b]	925,227	875,781	−0.59	824,110
	Industry	Enterprises (n°)	7,625	6,714	−1.32	5,828
		Employment (n°)	11,500	19,895	8.1	36,009.9
		Output (1,000 €)	1,029,000	2,041,247	10.9	4,266,206.2
	Salt-works	Area (ha)	137	33	−8.4	5.28
		Units (n°)	83	15	−9.1	1.35
		Production (t/year)	1,511	870	−4.7	461.1
	Tourism	Lodging capacity (n°)	2,339	1,499	−3.6	959
		Output (1,000 €)	3,271	6,849,000	10.9	14,314,410

	Fisheries	Employment (n°)	737	560	−1.7	465
		Fishing boats (n°)	319	211	−2.4	160
	Commercial harbour	Traffic (entrance n°)	269	363	7.0	617
	Ecological water extraction	Water volume (Hm^3)	4,910	4,877	−0.1	4,828
	Wastewater	Drainage (Hm^3)	1,586	2,620	7.2	4,506
		Treated (Hm^3)	0.539	2,474	39.9	12,345

GDP gross domestic product
[a]For the Mondego Estuary only
[b]Trends for the Centre region
[c]1990 data
[d]2000 data

others services (usually regulating services). For more detail among services trade-offs please see Braat and ten Brink (2008).

Most of the times, these conditioning relationships are multi-layered and cumulative. Services are not independent of one another and there are often inherent trade-offs in implementing management actions designed to enhance a single service (Bennett and Balvanera 2007; Koch et al. 2009). Natural resources managers are moving from approaches considering a single ES, to approaches integrating several ecosystems' compartments ('Integral System', proposed by de Jonge 2007). To ensure an accurate investigation of ES, researchers must recognise two aspects (Martín-López et al. 2009):

1. When performing integrated assessments, becomes essential to consider not only the effects of biodiversity and ecosystem functioning on services provision but also the effects that human well-being and ES provision may have on biodiversity and ecosystem functioning (Carpenter and Folke 2006). In fact, according to Martín-López et al. (2009): '*Ecologists need to know the essence of ES trade-offs, competing uses in ES and conflicting choices over temporal and spatial scales*'.
2. The inherent ecosystem functioning is a key-piece to determine the ecosystem condition responsible for the flow of ES (Kumar and Kumar 2008; Martín-López et al. 2009).

Despite possible conservation policies that might be implemented to protect ecosystems, it must be taken into account that these policies adequacy depends on the impacts induced by the surrounding human activities.

Applying ESA to estuarine systems may provide a common language to researchers and decision-makers and consequently promote communication. Synergies and trade-offs evaluations between ecological quality status (as a proxy for biodiversity and ecosystem functioning) and ES (as a proxy for human well-being) are fundamental to ensure a sustainable management of these ecosystems. However, the relations between a desired level of a service and the minimum required biodiversity to achieve it, or the effects that biodiversity changes may have on services provision, are the key-issues that are less known in estuarine ecosystems. The estimation of social values are fundamental to offer a structured framework that can (or should) be used to explore social and ecosystem responses to different managing approaches. The combination of ES approach with other techniques, e.g. spatial planning and multi-criteria analysis, can provide an added-value to ensure a balanced management of estuarine systems. These frameworks would allow incorporating ES into policies, making the implicit value of estuarine ecosystems explicit to society.

4 Conclusions

ES are faced as a promising tool to improve the understanding on the links between estuarine biodiversity/functioning and human well-being, ultimately advising for policies design and implementation. Nonetheless, to understand the underlying complex systems supporting ES provision is an essential step to successfully implement this approach. One of the main pitfalls is to look at complex ecosystems, like estuaries, as temporally and spatially static resources rather than as dynamic socio-ecological systems (Villa et al. 2014).

The way existing drivers and pressures determine changes in ecosystems' ecological quality status and consequently how ecosystem internal functioning is affected, was investigated focusing on ES provision on which human well-being relies. Using the Mondego watershed as a case-study, was possible to improve our knowledge on the integrated functioning of estuarine ecosystems, namely regarding responses to environmental pressures induced by human activities. The Mondego Estuary case study represented an opportunity to study and understand the complex and intricate relations in estuarine ecosystems, among ecological, social and economic factors, which were partially used in designing and implementing restoration measures. Results illustrated that the interrelations between estuarine functioning and ES provision are very complex and that methodologies used to assess estuarine services are into a certain extent limited. Also, the relationships between biodiversity, ecosystem functioning and services provision in estuarine systems are often cumulative and nonlinear, requiring for an in-depth knowledge on these relations to ensure effective governance of these systems.

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References

Acharya G (2000) Approaches to valuing the hidden hydrological services of wetland ecosystems. Ecol Econ 35:63–74

Agardy T, Alder J, Ash N, DeFries R, Nelson G (2005) Synthesis: condition and trends in systems and services, trade-offs for human well-being, and implications for the future. In: Hassan R, Scholes R, Ash N (eds) Ecosystems and human well-being: current state and trends: findings of the Condition and Trends Working Group. Island Press, Washington, DC, pp 823–834

APHA (1980) In: American Water Works Association & Water Environment Federation (ed) Standard methods for the examination of water and wastewater, 15th edn. American Public Health Association, Washington, DC, 1134 pp

Atkins JP, Burdon D (2006) An initial economic evaluation of water quality improvements in the Randers Fjord, Denmark. Mar Pollut Bull 53:195–204

Baeta A, Valiela I, Rossi F, Pinto R, Richard P, Niquil N, Marques JC (2009) Eutrophication and trophic structure in response to the presence of the eelgrass *Zostera noltii*. Mar Biol 156:2107–2120

Balvanera P, Pfisterer AB, Buchmann N, He JS, Nakashizuka T, Raffaelli D, Schmid B (2006) Quantifying the evidence for biodiversity effects on ecosystem functioning and services. Ecol Lett 9:1146–1156

Barbier EB, Hacker SD, Kennedy C, Silliman BR, Koch EW, Stier AC (2011) The value of estuarine and coastal ecosystem services. Ecol Monogr 81(2):169–193

Bennett EM, Balvanera P (2007) The future of production systems in a globalized world. Front Ecol Environ 5:191–198

Birol E, Karousakis K, Koundouri P (2006) Using economic valuation techniques to inform water resources management: a survey and critical appraisal of available techniques and application. Sci Total Environ 365(1–3):105–122

Blancher P, Vignon C, Catalon E, Maresca B et al (2011) Ecosystem services approach for water framework directive implementation. Management of Natural Resources, Sustainable Development and Ecological Hazards III. Ecol Environ 148:1743

Borja A, Basset A, Bricker S, Dauvin J et al (2012) Classifying ecological quality and integrity of estuaries. In: Wolanski E, McLusky D (eds) Treatise on estuarine and coastal science. Academic, Waltham, pp 125–162

Braat LC, Brink P ten (2008) The cost of policy inaction: the case of not meeting the 2010 biodiversity target. Report to the EC under contract: ENV.G.1./ETU/2007/0044, Alterra report 1718, Wageningen, Brussels

Carpenter SR, Folke C (2006) Ecology for transformation. Trends Ecol Evol 21:309–315

Carson RT (2000) Contingent valuation: a user's guide. Environ Sci Technol 34:1413–1418

Convention for Biological Diversity (2004) How the Convention on Biological Diversity promotes nature and human well-being? Secretariat of the Convention on Biological Diversity with the support of the United Nations Environment Program (UNEP) and the Government of the United Kingdom

Costanza R, d'Arge R, de Groot R, Farber S et al (1997) The value of the world's ecosystem services and natural capital. Nature 387:253–260

Costanza R, Low BS, Ostrom E, Wilson J (2001) Institutions, ecosystems, and sustainability. CRC Press, Boca Raton, 270 p

Daily GC (1997) Nature's services: societal dependence on natural ecosystems. Island Press, Washington, DC

de Groot RS, Wilson MA, Boumans RMJ (2002) A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecol Econ 41:393–408

de Groot RS, Alkemade R, Braat L, Hein L, Willemen L (2010) Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecol Complex 7:260–272

de Jonge VN (2007) Toward the application of ecological concepts in EU coastal water management. Mar Pollut Bull 55:407–414

de Jonge VN, Pinto R, Turner K (2012) Integrating ecological, economic and social aspects to generate useful management information under the EU Directives' 'ecosystem approach'. Ocean Coast Manag 68:169–188

Díaz S, Fargione J, Chapin FSIII, Tilman D (2006) Biodiversity loss threatens human well-being. PLoS Biol 4(8):e277

Díaz S, Lavorel S, de Bello F, Quétier F, Grigulis K, Robson TM (2007) Incorporating plant functional diversity effects in ecosystem service assessments. Proc Natl Acad Sci U S A 104:20684–20689

EC (2000) Directive 2000/60/EC of European Parliament and of the Council of 23-October-2000 establishing a framework for community action in the field of water policy. Off J Eur Comm L327 22/12/200:0001–0073

EC (2011) Communication from the Commission to the European Parliament, the Council, the Economic and Social Committee and the Committee of the Regions. Our life Insurance, Our Natural Capital: An EU Biodiversity Strategy to 2020. COM(2011)244final

EEA (1999) Nutrients in European ecosystems. Environmental assessment report no. 4. European Environmental Agency, Copenhagen, 155 pp

Elliott M, de Jonge VN (2002) The management of nutrients and potential eutrophication in estuaries and other restricted water bodies. Hydrobiologia 475(476):513–524

Farber SC, Costanza R, Wilson MA (2002) Economic and ecological concepts for valuing ecosystem services. Ecol Econ 41:375–392

Feio MJ, Almeida SFP, Craveiro SC, Calado AJ (2009) A comparison between biotic indices and predictive models in stream water quality assessment based on benthic diatoms communities. Ecol Indic 9:497–507

Fisher B, Turner RK, Morling P (2009) Defining and classifying ecosystem services for decision making. Ecol Econ 68(3):643–653

Folke C, Carpenter SR, Walker BH, Scheffer M, Elmqvist T, Gunderson LH, Holling CS (2004) Regime shifts, resilience and biodiversity in ecosystem management. Ann Rev Ecol Evol Syst 35:557–581

Godbold JA, Bulling MT, Solan M (2011) Habitat structure mediates biodiversity effects on ecosystemproperties. Proc R Soc B 278(1717):2510–2518

Graça MAS, Coimbra CN, Carvalho MJ, Oliveira R, Abelho M (2002) Freshwater macroinvertebrates in the Mondego river basin. In: Pardal MA, Marques JC, Graça MAS (eds) Aquatic ecology of the Mondego river basin global importance of local experience. Imprensa da Universidade de, Coimbra, 576 pp

Grimm V, Wissel C (1997) Babel, or the ecological stability discussion: an inventory and analysis of terminology and a guide for avoiding confusion. Oecologia 109:323–334

Haines-Young RH, Potschin M (2010) The links between biodiversity, ecosystem services and human well-being. In: Raffaelli D, Frid C (eds) Ecosystem ecology: a new synthesis. BES Ecological Reviews Series CUP, Cambridge

Halpern BS, Walbridge S, Selkoe KA, Kappel CV, Micheli F, D'Agrosa C (2008) A global map of human impact on marine ecosystems. Science 319:948–952

Hasler B, Brodersen SI, Christensen LP, Christensen T et al (2009) Denmark-assessing economic benefits of good ecological status under the EU WFD. Testing practical guidelines in Odense river Basin. AQUAMONEY, 73 pp

Hein L, van Koppen K, de Groot RS, van Ierland EC (2006) Spatial scales, stakeholders and the valuation of ecosystem services. Ecol Econ 57:209–228

Heywood VH (1995) The global biodiversity assessment. United Nations Environment Programme. Cambridge University Press, Cambridge, 1140 pp

Hooper DV, Chapin FS, Ewel JJ, Hector A et al (2005) Effects of biodiversity on ecosystem functioning: a consensus of current knowledge. Ecol Monogr 75:3–35

Jørgensen SE (2010) Ecosystem services, sustainability and thermodynamic indicators. Ecol Complex 7:311–313

Jorgensen SE, Costanza R, Xiu FL (2010) Handbook of ecological indicators for assessment of ecosystem health, 2nd edn. Taylor & Francis, Boca Raton, 482 pp

Koch EW, Barbier EB, Silliman BR, Reed DJ et al (2009) Non-linearity in ecosystem services: temporal and spatial variability in coastal protection. Front Ecol Environ 7(1):29–37

Kumar M, Kumar P (2008) Valuation of the ecosystem services: a psycho-cultural perspective. Ecol Econ 64:808–819

Levin S (1999) Fragile domination: complexity and the commons. Perseus Book, Reading

Lima MIP, Lima JLMP (2002) Precipitation and the hydrology of the Mondego catchment: a scale-invariant study. In: Pardal MA, Marques JC, Graça MA (eds) Aquatic ecology of the Mondego river basin: global importance of local experience. Imprensa da Universidade de Coimbra, Coimbra, 13–28 pp

Loreau M, Naeem S, Inchausti P, Bengtsson J, Grime JP, Hector A, Hooper PU (2001) Biodiversity and ecosystem functioning: current knowledge and future challenges. Science 294:804–808

Lotze HK, Lenihan HS, Bourque BJ, Bradbury RH et al (2006) Depletion, degradation, and recovery potential of estuaries and coastal seas. Science 312:1806–1809

Luck GW, Daily GC, Ehrlich PR (2003) Population diversity and ecosystem services. TRENDS Ecol Evol 18(7):331–336

MacArthur RH (1955) Fluctuations of animal populations and a measure of community stability. Ecology 36:533–536

Mace GM, Bateman I (2011) Conceptual framework and methodology (Chp2). In: The UK national ecosystem assessment: synthesis of the key findings. UNEP-WCMC, Cambridge

Maltby E (2006) Wetland conservation and management: questions for science and society in applying the ecosystem approach. Ecol Stud 191:93–116

Marques JC, Maranhão P, Pardal MA (1993) Human impact assessment on the subtidal macrobenthic community structure in the Mondego estuary (Western Portugal). Estuar Coast Shelf Sci 37:403–419

Marques JC, Pardal MA, Nielsen S, Jorgensen SE (1997) Analysis of the properties of exergy and biodiversity along an estuarine gradient of eutrophication. Ecol Model 102:155–167

Marques JC, Nielsen SN, Pardal MA, Jørgensen SE (2003) Impact of eutrophication and river management within a framework of ecosystem theories. Ecol Model 166(1–2):147–168

Martín-López B, Gómez-Baggethun E, González JA, Lomas PL, Montes C (2009) The assessment of ecosystem services provided by biodiversity: re-thinking concepts and research needs. In: Aronoff JB (ed) Handbook of nature conservation. Nova, New York. ISBN 978-1-60692-993-3

May RM (1972) Stability and complexity in model ecosystems. Princeton University Press, Princeton

MEA (2003) Ecosystems and human well-being: a framework for assessment. Island Press, Washington, DC

MEA (2005) Ecosystems and human well-being: wetlands and water synthesis. World Resources Institute, Washington, DC, 80 pp

Mitchell RC, Carson RT (1989) Using surveys to value public goods: the contingent valuation method. Resources for the Future, Washington, DC

Mueller F (2005) Indicating ecosystem and landscape organization. Ecol Indic 5:280–294

Naeem S (1998) Species redundancy and ecosystem reliability. Conserv Biol 12:39–45

Naeem S, Bunker DE, Hector A, Loreau M, Perrings C (2009) Introduction: the ecological and social implications of changing biodiversity: an overview of a decade of biodiversity and ecosystem functioning research. In: Naeem S, Bunker DE, Hector A, Loreau M, Perrings C (eds) Biodiversity, ecosystem functioning & human well-being: an ecological and economic perspective. Oxford University Press, New York, 3–13 pp

Nunes PALD, Bergh JCJM (2001) Economic valuation of biodiversity: sense or nonsense? Ecol Econ 39:203–222

O'Higgins TG, Ferraro SP, Dantin DD, Jordan SJ, Chintala MM (2010) Habitat scale mapping of fisheries ecosystem service values in estuaries. Ecol Soc 15(4):7

Odum EP (1959) Fundamentals of ecology, 2nd edn. Saunders, Philadelphia

OECD (2011) OECD environmental performance reviews: Portugal 2011. OECD Publishing, Washington

Perrings C, Maler KG, Folke C, Holling CS, Jansson BO (1995) Introduction: framing the problem of biodiversity loss. Cambridge University Press, Cambridge, 364 pp

Pimm S (1984) The complexity and stability of ecosystems. Nature 307:321–326

Pinho L (2007) The role of maritime public domain in the Portuguese coastal management. J Coast Conserv 11:3–12

Pinto R, Patrício J, Neto JM, Salas F, Marques JC (2010) Assessing estuarine quality under the ecosystem services scope: ecological and socio-economic aspects. Ecol Complex 7:389–402

Pinto R, de Jonge VN, Neto JM, Domingos T, Marques JC, Patrício J (2013) Towards a DPSIR driven integration of ecological value, water uses and ecosystem services for estuarine systems. Ocean Coast Manag 72:64–79

Pinto R, de Jonge VN, Marques JC (2014) Linking biodiversity indicators, ecosystem functioning, services provision and human wellbeing in estuarine ecosystems: application of a conceptual framework. Ecol Indic 36:644–655

Polasky S, Segerson K (2009) Integrating ecology and economics in the study of ecosystem services: some lessons learned. Ann Rev Resour Econ 1:409–434

Ross DA (1995) Introduction to oceanography. Harper Collins College Publishers, New York

Schwartz MW, Bringham C, Hoeksema JD, Lyons KG, Mills MH, van Mantgem PJ (2000) Linking biodiversity to ecosystem function: implications for conservation ecology. Oecologia 122:297–305

Söderqvist T (1996) Contingent valuation of a less eutrophied Baltic Sea. Beijer discussion paper series, no 88. Beijer International Institute of Ecological Economics, Stockholm

Spurgeon J (1992) Economic valuation of coral reefs. Mar Pollut Bull 24(11):529–536

Stein ED, Cadien DB (2009) Ecosystem response to regulatory and management actions: the southern California experience in long-term monitoring. Mar Pollut Bull 59:91–100

Strickland JDH, Parsons TR (1972) A practical handbook of seawater analysis. Bulletin of fisheries research board of Canada, vol 167, 2nd edn. Fisheries Research Board of Canada, Ottawa, 311 pp
Teixeira H, Salas F, Neto JM, Patrício J et al (2008) Ecological indices tracking distinct impacts along disturbance-recovery gradients in a temperate NE Atlantic Estuary e guidance on reference values. Estuar Coast Shelf Sci 80:130–140
Teixeira H, Neto JM, Patrício J, Veríssimo H, Pinto R, Salas F, Marques JC (2009) Quality assessment of benthic macroinvertebrates under the scope of WFD, the Benthic Assessment Tool. Mar Pollut Bull 58:1477–1486
Tilman D, Polasky S, Lehman C (2005) Diversity, productivity and temporal stability in the economies of humans and nature. J Environ Econ Manag 49:405–426
Tilman D, Reich PB, Knops JMH (2006) Biodiversity and ecosystem stability in a decade-long grassland experiment. Nature 441:629–632
Turner RK (2000) Integrating natural and social science in coastal management. J Mar Syst 25:447–460
Valiela I, McClelland J, Hauxwell J, Behr PJ, Hersh D, Foreman K (1997) Macroalgal blooms in shallow estuaries: controls, ecophysiological and ecosystem consequences. Limnol Oceanogr 42(5/2):1105–1118
Villa F, Voigt B, Erickson JD (2014) New perspectives in ecosystem services science as instruments to understand environmental securities. Philos Trans R Soc B 369:1471–2970
Wattage P (2002) Literature review: CVM. Effective management for biodiversity conservation in Sri Lankan coastal wetlands–final report, 56 p
Worm B, Barbier EB, Beaumont N, Duffy JE, Folke C et al (2006) Impacts of biodiversity loss on ocean ecosystem services. Science 314(3):787–790
Yachi S, Loreau M (1999) Biodiversity and ecosystem productivity in a fluctuating environment: the insurance hypothesis. Proc Natl Acad Sci 96:1463–1468