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A framework for assessing ecological quality based on ecosystem services

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ABSTRACT

Existing environmental legislation and ecological quality definitions such as ecosystem integrity tend to rely on measures that, either implicitly or explicitly, utilize naturalness as a key criterion. There are marked practical difficulties with employing the concept of naturalness in human dominated landscapes, and the management of such ecosystems is inevitably going to need to take account of human needs and expectations. We propose that ecological quality could be assessed by its ecosystem service profile (ESP): the overlap between societal expectations for, and the sustainable provision of, suites of ecosystem services. The status for each individual ecosystem service is defined by the ratio of its sustained provision to the expected level of provision for the service. The ESP measure is a multi-criterion, context-specific assessment of the match between expectation for and sustainable supply of ecosystem services. It provides a flexible measure of quality which takes into account that the ''ideal'' ecosystem state is largely dependent on the specific management context. The implementation of ESPs challenges us to develop indicators for the sustained provision of individual ecosystem services, much better understanding of the trade-offs among services, and practical tools for gauging societal demands. All of which are challenging problems. The proposed framework can help to strategically address research needs and monitoring requirements and foster a more integrative approach to ecosystem assessment and management in the future. The need for this follows from the fact that the undisturbed reference state represents only one aspect of an ecosystem and that ecological quality in human dominated landscapes will, ultimately, be determined by the value society places on the sustainable provision of multiple ecosystem services.

1. Introduction

Management of ecosystems for the conservation of biodiversity and the satisfaction of growing human needs for natural resources such as food and energy are crucial global issues (Foley et al., 2005; Lindenmayer et al., 2008). To aid decision making in ecosystem management and improve its transparency, we need measures and concepts to characterize ''desirable'' or ''undesirable'' conditions in an ecosystem, i.e. a definition of ecological quality (Freyfogle and Lutz Newton, 2001; Lackey, 2001; Smyth et al., 2007; Burkhard and Müller, 2008). This is increasingly important as wide ranging legislative changes, such as the *European Water Framework Directive* (2000/60/*EC*) or the *Rio Declaration on Environment and Development* (1992), begin to utilize notions of ecological quality as critical benchmarks for the improvement and harmonization of environmental standards and also encourage active societal involvement in the implementation process (Steyaert and Ollivier, 2007). Quality judgments of an ecosystem are, however, dependent on the aims of the people setting them (Freyfogle and Lutz Newton, 2001). For instance, a fisherman might evaluate a river by the abundance of certain game fish species, a farmer by its capacity to abstract water for irrigation, a conservationist by the presence of rare wildlife species, and a water supply company by the amount of treatment it requires to produce safe drinking water. Therefore, it is important that the quality concepts on which management decisions are based on are made explicit in order to enhance public support of ecosystem management (Smyth et al., 2007; Woolsey et al., 2007). Developing a transparent and consistent framework for ecosystem quality assessments in coupled human environmental systems is clearly important, but is hard to do in a robust and consistent way (Müller and Li, 2004; Hodgson et al., 2007).

Attempts to define ecological quality, either in general terms or with a view to developing management standards, have given rise to a number of related ideas, but pre-eminent among these are the concepts of *biological integrity* (Karr, 1999) and *ecosystem health* (Rapport et al., 1998; for a more comprehensive overview see Burkhard and Müller, 2008). While attractive in some respects, these ideas are subjects of considerable debate, both regarding their conceptual basis and their operational utility (Suter, 1993; De

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Leo and Levin, 1997; Calow, 2000; Lancaster, 2000; Lackey, 2001). A healthy ecosystem has been defined as *"being stable and sustainable, maintaining its organization and autonomy over time and its resilience to stress"* (Rapport et al., 1998). Thus, health is essentially defined by the lack of change in system organization and functioning. In order to make the concept of health operational either the direction of change needs to be determined, or an "*optimum state*" or target needs to be defined. Based on this fundamental concept several modifications and further, sometimes even contradictory, definitions of ecosystem health have been proposed, but often lack precise conceptual as well as operational elaborations (Lancaster, 2000; Lackey, 2001; Vugteveen et al., 2006). Similarly, also several definitions of the notion of *biological* or *ecosystem integrity* have been proposed (e.g. Karr and Dudley, 1981; Kay and Schneider, 1992; Westra and Lemons, 1995; Barkmann et al., 2001). Frequently used is the concept of biological integrity by Angermeier and Karr (1994), which is defined by the closeness of the diversity, species composition and functional organization of the communities of organisms to that of natural habitats in the region. Common definitions of "natural" are in essence as either being without human influence or without human technology (Hunter, 1996; Angermeier, 2000). Thus, ecological integrity indicates the divergence from natural reference conditions attributable to human activities (Karr, 1999). Closeness to a natural state, sometimes explicitly defined as integrity, is often an aim for nature conservation (Callicott et al., 1999) and has been adopted by major environmental legislation (Karr and Chu, 2000; Lackey, 2001), such as the US Clean Water Act and the European Water Framework Directive, which challenges river basin management with the central aim of achieving good "ecological status", defined as deviation from "undisturbed conditions", for all surface waters. However, in human dominated landscapes the return to the natural state is usually not feasible because society places a variety of other demands on ecosystems, such as food and energy production and water supply (Palmer et al., 2004; Foley et al., 2005). Alternative descriptions of ecosystem integrity and health are based on system theoretical considerations that originate from thermodynamics (Schneider and Kay, 1994; Kay, 2000), succession theory (Odum, 1969), gradient theory and the concept of ecological orientors (for a detailed review see Müller and Li, 2004; Müller, 2004). The theory assumes that an ecosystem has at least one optimum operation point at which it most effectively stores and degrades exergy (usable energy, typically solar radiation) (Schneider and Kay, 1994; Müller, 2004). As a consequence, certain characteristics of the ecosystem (e.g. complexity, food web connectedness, energy and nutrient storage, productivity) should increase during undisturbed development of the system (Müller et al., 2000). Such a system theory based definition of integrity provides a potentially important approach to understand and assess the possible responses of an ecosystem to environmental changes. However, to usefully integrate these theoretical considerations of health or ecosystem integrity in environmental decision making processes, anthropogenic values that indicate which changes in an ecosystem are acceptable or desirable need to be considered (Kay and Schneider, 1992; Burkhard and Müller, 2008). For instance, eutrophication that increases primary production is considered desirable where food or fiber production are involved, but could be undesirable for ecosystems valued for their aesthetic or recreational value (Odum et al., 1979); or society might prefer a semi-natural grassland with certain rare plant species to a woodland and a clear oligotrophic lake to a more species rich eutrophic lake even though the latter might store and degrade more exergy.

Increasing recognition of the validity of including human values in ecosystem assessments has arisen with the concept of ecosystem services, i.e. the benefits people obtain from ecosystems (de Groot et al., 2002; Millenium Ecosystem Assessment, 2003; Meyerson et al., 2005; Folke, 2006). This shift in emphasis towards ecosystem services provides an approach to the valuation of ecosystems in terms of what they do, particularly in relation to the support of human well-being (Millenium Ecosystem Assessment, 2003). So far, the ecosystem service concept has been predominantly applied in economic valuation of ecosystems or in global ecosystem assessments (Millenium Ecosystem Assessment, 2003). The Millennium Ecosystem Assessment (MEA), for instance, has focused on ecosystem services in assessing the conditions and trends of ecosystems at the global and national scale (Millenium Ecosystem Assessment, 2003). However, such large-scale assessments of ecosystem services do not lend themselves to site-specific evaluations of individual ecosystems, the scale at which much ecosystem management takes place (Chan et al., 2006). So far, a consistent approach for assessing ecological quality based on ecosystem services has not been developed (Egoh et al., 2007). A critical first step in contributing towards such an approach is the development of a clear and consistent conceptual framework. Here, we propose one idea for such a conceptual framework with the aim of stimulating examination of both the logic and robustness of the idea. If the framework seems to offer useful gains over alternatives, then the challenges of practical implementation need to be tackled. Our particular perspective comes predominantly from evaluation of rivers and streams and we take examples from river and catchment management to discuss the practical challenges of our proposed ecological quality concept. Freshwater systems provide a useful testing ground for ideas about ecological quality as they provide a range of pivotal ecosystem services that are highly valued by society (Meyer, 1997; Baron et al., 2002; Giller, 2005), ecological quality assessments have been widely applied for rivers (Karr, 1991; Woolsey et al., 2007), and river catchments have become a focal point in environmental management and legislation (Mostert, 2003).

2. Assessing the status of ecosystem services

Ecosystem services are benefits people obtain from ecosystems (Costanza et al., 1997; de Groot et al., 2002; Daily et al., 1997; Millenium Ecosystem Assessment, 2003). The MEA classifies ecosystem services as provisioning services (e.g., food and timber), regulating services (e.g., climate and flood regulation), cultural services (e.g., recreation and aesthetic enjoyment), and supporting services (e.g., soil formation and nutrient cycling), which are necessary for the production of all other services (Millenium Ecosystem Assessment, 2003).

An important issue with using ecosystem services to assess ecological quality is that measures of service provision (e.g., volume of timber production) do not lend themselves to a quality judgment (e.g., good status). The sub-global assessments of the MA adopted different approaches to relate measures of service provision to quality assessments of ecosystem services. The Norwegian assessment assessed current provision of services relative to their provision 100 years ago (Pereira et al., 2005). The Portuguese assessment resorted to a conceptual baseline, defined as the current capacity of an ecosystem to provide a service relative to a level at which the service could be maximized in a sustainable way (Pereira et al., 2005); while the Southern African Assessment mapped supply and demand for individual ecosystem services at nested spatial scales (van Jaarsveld et al., 2005). Such supply–demand surfaces are useful for identifying areas with supply–demand tensions for individual services across the landscape. The coexistence of the different assessment approaches has the practical advantage that individual assessments can account for variability in the availability of data and understanding but also the major disadvantage that it complicates/precludes comparisons across differently assessed systems. Comparisons of the status of ecosystem services in space and time are, however, of vital importance for ecosystem management, particularly at regional–local scales. For instance, a manager needs to able to compare the relative status of different wetlands in a given watershed in order to make informed decisions on individual management actions, such as prioritizing wetlands for conservation or restoration (Zedler, 2003; Hatfield et al., 2004). However, at present a consistent approach for the assessment of ecosystem service status is lacking.

3. Towards a quality assessment framework using ecosystem services

The ecosystem service approach takes us forward in terms of explicitly integrating human needs and expectations in the assessment of ecosystems. In order to take such an approach we need to develop a robust conceptual framework in applying the approach to assessing ecological quality. Such a framework needs to integrate a number of features, in particular: (1) the flexibility to accommodate different, and varying, needs and expectations for the provision of ecosystem services depending on context and time, (2) the facility to take account of the trade-offs between services, (3) recognition of the potential for the sustained rather than short-term provision of desired services, and (4) the ability to incorporate legislative requirements, and criteria of naturalness, as components of quality, but not as an exclusive goal. We propose that the status of an ecosystem in terms of its delivery of ecosystem services in relation to the expectation for those services could provide the basis of such a framework.

More formally, we define the Ecosystem Service Profile (ESP) as the match between the societal expectations (including potential future options) for a set of ecosystem services and the realized sustainable provision of those services, in terms of both quality and quantity. For each individual ecosystem service, its status is defined by the ratio (R) of its sustainable provision to the expected level of service provision (Fig. 1): $R < 1$ indicating expectation exceeding provision, $R > 1$ provision in excess of expectation. $R = 1$ is a critical ratio which indicates that the status or quality of the service is "good" because the provision and expectation for a service are matched. Higher ratios ($R > 1$) do not represent a higher

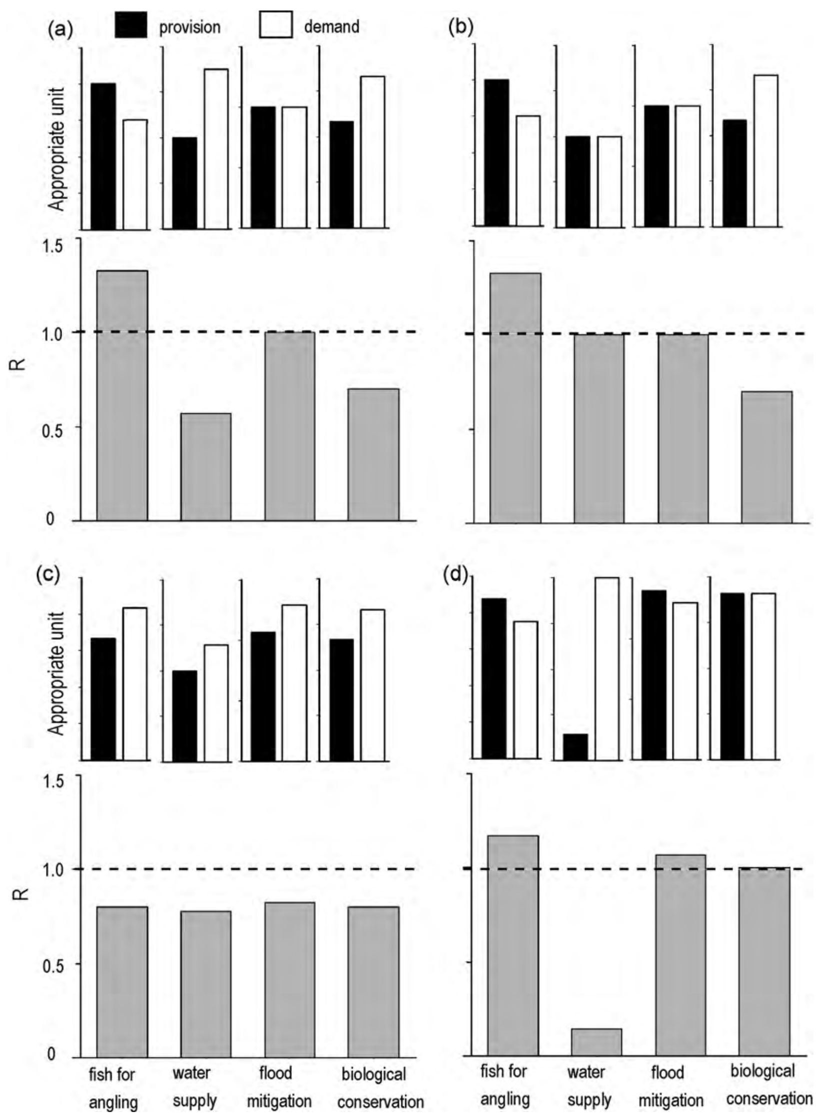

Fig. 1. ESP for a hypothetical example of selected ecosystem services; (a) upper panel shows levels of provision of and demand for ecosystem services, note that for different ecosystem services such levels need to be measured in different units (appropriate units for the different services could be for instance fish biomass per meter river length for angling or number of species for biodiversity), lower panel shows respective ESP resulting from the status of individual services expressed by the ratio of sustained provision to demand (R); dotted line indicates where provision equals demand. (b) Improved ESP resulting from reduction in demand for the ecosystem service of water supply; (c) balanced ESP with similar status of all services; (d) unbalanced ESP with high quality for all but one service for which demand is much higher than provision, note that average status across all ecosystem services is similar to ESP in (c).

quality of a services at any one time, but indicate further buffering capacity against potential fluctuations in the both the provision of (e.g. variation in water supply or crop yield resulting from climatic variability) and expectation for an ecosystem service. Variations in the expectations/demands for an ecosystem service can result from changing societal behavior. The status of an individual ecosystem service, therefore, is dependent not only on the level of provision but also on the desired level of provision for the service (Fig. 1a and b). This makes explicit that ecosystem management cannot simply be based on increasing service provision, but must also look at options to manage expectations and uses. If such a set of expectation/provision assessments of services can be made for a system, then we propose that this could form the basis of a metric for ecological quality assessments based on ecosystem services, which has the potential to satisfy the criteria above.

An important step in making the ratio *R* operable is to include both social and biophysical aspects in ecosystem service assessments. This is in line with the recently proposed operational model for implementing the safeguarding of ecosystem services by Cowling et al. (2008). Their model, which highlights the importance of responding to stakeholder needs from the outset by collaborating with and empowering stakeholders in strategy development and implementation (Cowling et al., 2008), could be potentially applied to guide the practical implementation of our ESP approach. The Southern African Millennium Ecosystem Assessment provides a good practical example for the implementation of a supply and demand based approach for assessing important ecosystem services, such as water supply, food and fuel wood production (van Jaarsveld et al., 2005). Based on such quantitative information the proposed ratio, *R*, could be readily calculated for the assessed services. However, the quantification of both societal demands for and the provision of ecosystems are major challenges that require further attention in future assessments of ecosystem services (see Section 4).

3.1. Context dependence and flexibility

If ecosystem services are defined by their contribution to human well-being then the approach is inherently adaptable to situations in which human pressures and expectations on the environment vary both in intensity and type. An ecosystem in one situation might provide a very different set of services to those provided by a similar system in another context, yet both might provide a good match to the societal expectations, which are generally context and situation dependent (Millenium Ecosystem Assessment, 2005). Similarly, changes in pressure and expectation over time can also be incorporated into such a scheme. It would be tempting to derive a composite index of quality across all services (e.g. the mean of the status of all services), but the potential differences in importance of the different services to people mean that, at a minimum, some sort of weighting would be required to account for this (Fig. 1c and d). However, we agree with Müller and Li (2004) that it is problematic and also not necessary to reduce the different components of an ecosystem into one dimension, which often tends to be money. Therefore, we think that the strength of the ESP is that it can provide a condensed, context-based, overview of the status of key services. This is also likely to be generally more helpful in targeting specific management actions and also in diagnosing potential pressures on the ecosystem than a single composite index (see Lopez-Ridaura et al., 2002).

3.2. Trade-offs among services

Because ecosystems provide multiple services, that are likely to be interdependent (Rodriguez et al., 2006), it is important to recognize the trade-offs among different services. Understanding the interdependence among services will rarely be straightforward because our recent understanding of the ecology of many ecosystem services is limited (Kremen and Ostfeld, 2005) and the concordance among different services varies widely and the nature of the relationships appear be location-specific (Naidoo et al., 2008; Anderson et al., 2009). However, in representing the ecosystem in terms of its service profile, we are forced to consider the state of multiple services, and by extension to ask whether particular management scenarios, which alter a particular service also leave others unchanged, or whether the gain in match between provision and expectation in one area is offset by an increasing mismatch elsewhere in the profile.

3.3. Sustainability of provision

Critical to the assessment of the match between expectation and provision is the way in which provision is assessed. Actual level of service provision might not indicate the level at which the service can be provided in the long-term. For instance, actual fish catch data do not provide information whether the catch is sustainable. Here we refer to the more narrow ecological sense of sustainability as the capacity for the long-term provision of services in line with the definition by Burkhard and Müller (2008) "keep available ecosystem services on a long-term, intergenerational and broad scale, intragenerational level". Currently, however, the sustainability aspect of service provision, particularly for the provisioning services, is often ignored (McMichael et al., 2005; Mooney et al., 2005). Therefore it is important to select, or develop, indicators that reflect the potential of the system to sustain the yield of each service (McMichael et al., 2005). To assess the sustainability aspect of a service, all relevant supporting ecosystem functions and components that are needed for the provision of this service need to be considered (Fig. 2). For instance, to estimate whether current levels of crop production can be sustained (e.g. without long-term losses in soil fertility) the underlying supporting services including soil formation, erosion, and nutrient cycling need to be taken into account. However, it may not be that all such underlying services require explicit representation in the ESP because they are predominantly means to provide sustained levels of those services which are directly utilized (Boyd and Banzhaf, 2007). The desired levels of the supporting services are also to some extent dependent on the context-specific expectations for the directly utilized services (provision, regulating, and cultural services). For instance, high levels of primary production (supporting service) in a wetland are desired if the main demand is to retain nutrients from agricultural runoff whereas if conservation of biodiversity would be of highest concern for the wetland, much lower levels of productivity would be desired (Zedler, 2003). Further, the direct inclusion of the underlying supporting services in ecosystem service assessments could result in the potential problem of double counting of services (de Groot et al., 2002; Boyd and Banzhaf, 2007).

3.4. Thresholds and naturalness

Minimum quality standards and legislative requirements can be included in the service profile by simply including them as an expectation. This expectation, as with others, can be context dependent and so can account for specific regional or local expectations, which may result in much higher quality requirements (e.g., biodiversity) than average minimum standards that need to be achievable across a wide range of environmental conditions. Closeness to the natural state, the key driver for quality definitions based on ecological integrity, can also be incorporated, but in this context it is defined by society's desire for such a state. A high expectation for conservation of particular species native to

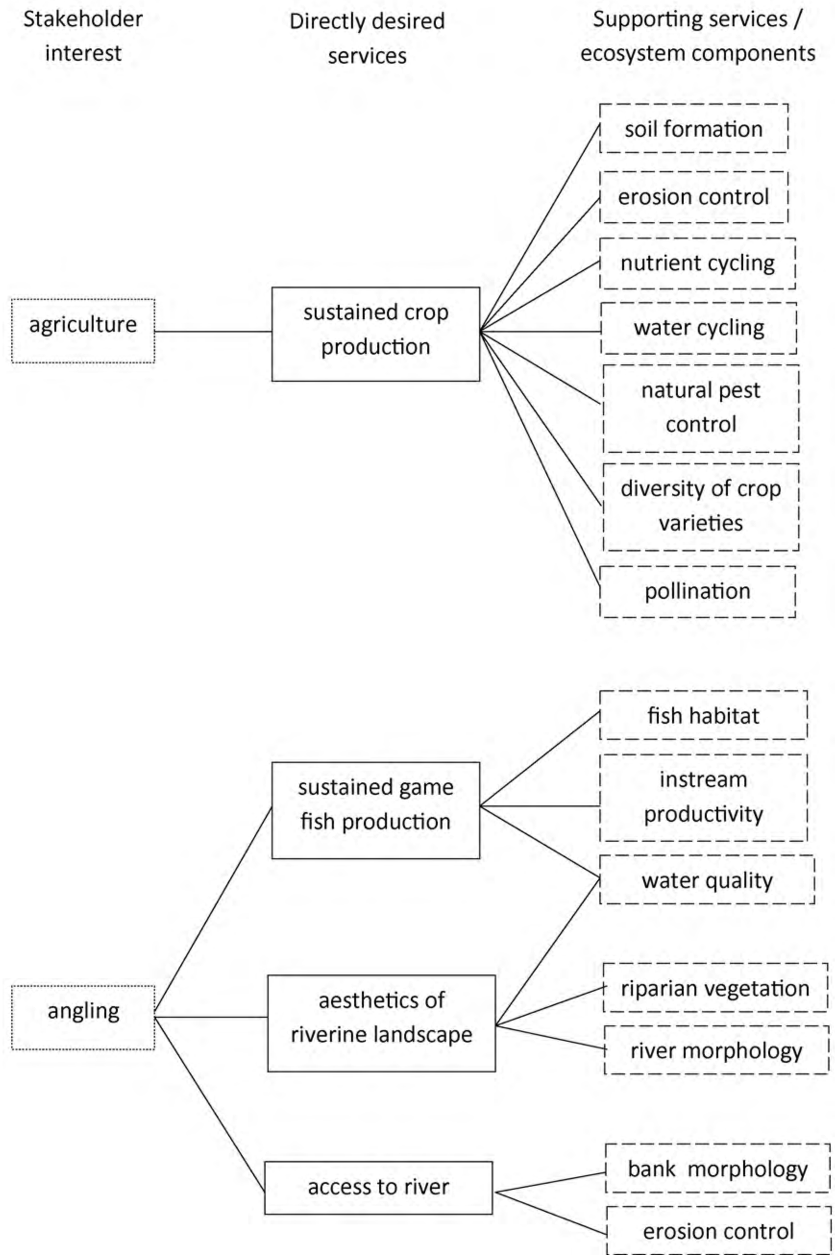

Fig. 2. Links between broad stakeholder interests and associated directly desired ecosystem services (provisioning and cultural ecosystem services), and major supporting services for the examples of agriculture (upper panel) and angling (lower panel). Assessment of the associated supporting services and ecosystem components are particularly important to address the sustainability aspect (i.e. long-term provision) for the directly utilized/demanded services.

the region, for recreational use, or for characteristic types of landscape could all provide demand for naturalness. The difference here is that naturalness, either in its own right or as a consequence of something else, becomes one of the criteria used in the profile, and not *de facto* the only quality target.

In principle, therefore, we suggest that the combination of expectation for and provision of multiple ecosystem services has the potential to provide a tool for assessing the status of ecosystems under varying degrees of anthropogenic pressure. It is a small, though distinct, step from there to viewing the extent to which a system is meeting the expectation for ecosystem services: the match, or mismatch, of the provision and expectation profiles as a measure of ecological quality. Systems that fulfill the expectations, in a sustainable fashion, can be regarded, in a non-absolute sense, as being of better quality than those that do not. There is no single target for what constitutes good ecological quality in this approach, but existing definitions of quality can be incorporated within it through their realization as components of particular ecosystem services.

4. Challenges

The flow in Fig. 3 provides an outline of the steps involved in assessing ESP. The first step would be to identify all important ecosystem services that can be provided in the catchment. This would be most efficiently done from master lists of services for major habitat types, which could then be adapted to specific situations. Then a manageable subset of ecosystem services that are of particular interest in a specific management context needs to be identified (e.g. angling, flood mitigation, water supply, and biological conservation). This requires a close collaboration between a diverse range of experts and stakeholders to ensure

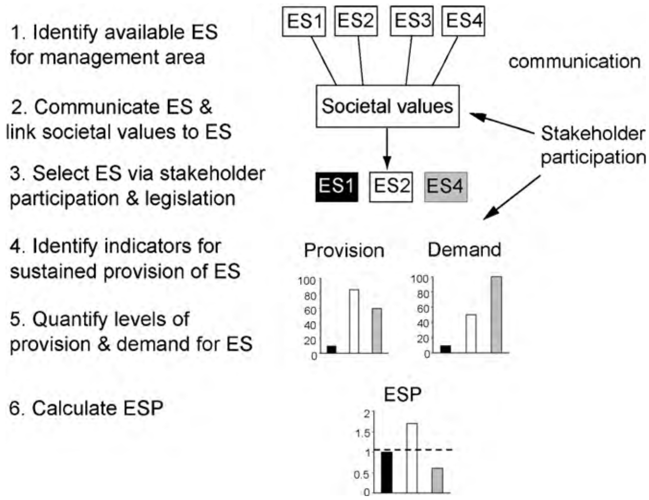

Fig. 3. Major steps in the development of an ESP.

the inclusion of both a good understanding of the potential provision of ecosystem services and a wide range of broad societal interests (Fig. 2). For the selected ecosystem services the level of provision and respective demand need to be identified. Demands need to be assessed in the same units as the provision indicators. Finally, the ESP can be derived by calculating the provision/demand ratio for all ecosystem services.

The implementation of each of the stages in this process will generate very significant challenges. However, solving these challenges is not unique to this particular use of ecosystem services, but is likely to be an issue in a wide range of applications of the ecosystem service concept. One major challenge is the scarcity of appropriate context-specific data on the provision of and expectations for many ecosystem services (Chan et al., 2006). This is not surprising, given that quantification and monitoring of ecosystem services is a reasonably novel enterprise. Even though data tend to be available for many provisioning services in terms of realized current or historical yields (e.g., fish landing) the potential for their sustained production is likely to be much less readily available. For instance, for angling, indicators that reflect sustained fish production based on reproductive surplus and availability of fish habitat should be assessed rather than simply current fish catch, or for agricultural and timber production the underlying supporting services, including soil formation and nutrient cycling, need to be considered (Hilborn et al., 1995; McMichael et al., 2005; Mooney et al., 2005; Boyd and Banzhaf, 2007; Fig. 2). Only sufficient data on the trends of supporting and regulating services will enable experts to make reasonable predictions about long-term trends in the production of many provisioning services. Consequently, an ecosystem service orientated assessment would require a stronger emphasis on relevant ecosystem processes, not just ecosystem stocks, in monitoring. More recently, a number of promising indicators of ecosystem functions, such as respiration, nutrient cycling and storage, and transpiration, based on theoretical considerations of ecosystem development have been proposed and applied in different ecosystems (Müller, 2005; Burkhard and Müller, 2008). However, further development and assessments of these indicators in the context of the provision of specific sets of ecosystem services are needed.

A range of novel indicators and methods for assessment would need to be developed to identify not only the provision of, but also the societal expectations for, many ecosystem services. For certain services, particularly the provisioning services, information on societal expectations for service provision might be more readily available (e.g., drinking water and other natural goods) while for others (e.g., biological conservation) such expectations need to be derived from policies and legislation or via direct stakeholder participation. The stakeholder participation may involve different tools and processes ranging from focus group workshops, stakeholder panels, interviews and surveys, to social learning (Pereira et al., 2005; Pahl-Wostl, 2006; Smyth et al., 2007). The participatory process should also involve experts who can inform the stakeholders about the potential choice of services. Because many stakeholders might not be familiar with the ecosystem service concept it is important to translate the potential choice of ecosystem services into objectives that are valued by society and to link broad stakeholder interests and management objectives to the underlying ecosystem services (Fig. 2). For instance, the regulating service of water retention in a catchment needs to be explicitly linked and converted to the societal objective of reducing flood risk. In this context, we think that the explicit representation of the status of different ecosystem services in the ESP could provide a useful tool to improve the dialog between experts, managers, and other stakeholders. Further, expectations for service provision are likely to vary among stakeholders and consensus views may be difficult to achieve. However, Smyth et al. (2007) have demonstrated that clear societal preferences for a range of environmental objectives, such as desired level of fish catch, can be determined from stakeholder surveys by using aggregated social norm curves. For those services for which expectations vary considerably among stakeholders the range in status based on the highest and lowest level of expectation could be shown. Additionally, an ESP could be determined for different stakeholder groups. Such information could provide useful information related to important issues of human well-being including equity in the distribution of ecosystem services and societal acceptance (Chan et al., 2007; Smyth et al., 2007).

Scale is another aspect that requires further attention in the implementation of the ESP. Different ecosystem services are provided and demanded at different spatial and temporal scales (van Jaarsveld et al., 2005; Rodriguez et al., 2006). The development of an ESP for a given area of assessment or planning unit would require to identify which services are produced and demanded locally within the assessment area, which services are demanded or provided at larger scales (nationally or globally), and which services are exported or imported. For instance, flood protection might be demanded at the scale of individual river reaches, drinking water at catchment scale, and carbon sequestration at the global scale. Expectations for ecosystem services that are derived from larger scales would need to be allocated to scales relevant for assessment and planning. One way of dealing with spatial transfers of ecosystem services in the ESP could be to add exported services to and deduct imported services from the internal expectation for service provision. The expectation for or the use of a service and the service providing function can be spatially separated, particularly in highly connected systems such as rivers and their catchments. For instance, natural flood plains can serve as flood retention areas and thus provide flood protection for downstream located communities, or the provision of clean water for urban areas can be largely controlled by the land use in the upper catchment (Kremen and Ostfeld, 2005). Consequently, it is crucial to first define the spatial and temporal context for the assessment of ecosystem services but then also to consider the spatial linkages among service provisions and expectations within and outside the defined area in order to identify spatially disconnected trade-offs (van Jaarsveld et al., 2005).

Another potential use of the ESP could be to illustrate possible outcomes of alternative management scenarios (Fig. 4). The development of ESP scenarios would need to account for the fact that the provision of one service can impair or enhance the provision of other services (Millenium Ecosystem Assessment,

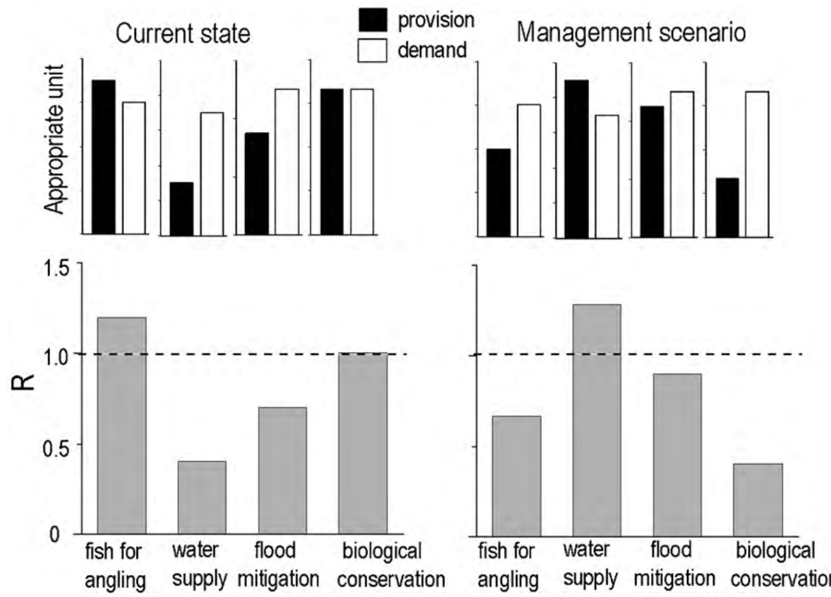

Fig. 4. Hypothetical example for using ESPs to demonstrate the possible outcome of a management scenario (e.g. construction of a reservoir): (a) current ESP and (b) scenario of ESP with reservoir.

2005; Chan et al., 2006; Rodriguez et al., 2006). For instance, the construction of a new reservoir can impair the conservation of natural aquatic and riparian biota and reduce downstream fish production through alterations of the natural flow regime, but can enhance water supply and moderate extreme water conditions such as floods and droughts (Petts, 1984; Richter et al., 2003; Paetzold et al., 2008). Exploration of alternative scenarios given our understanding of these interdependencies will be required to find the best achievable ESP. However, the processes that can cause interdependencies among services are likely to be extremely complex and our recent understanding is primarily qualitative and limited to a few specific ecosystems and services (Kremen, 2005; Olson and Wäckers, 2007). The drive to predict the outcome of possible management actions means that ways of tackling this problem will need to be developed, integrating more traditional process-based models and understanding with expert opinion using for instance Bayesian networks, multi-criteria analysis or phenomenological correlations (Lopez-Ridaura et al., 2002; Alcamo et al., 2005; Chan et al., 2006; Castelletti and Soncini-Sessa, 2007). Explicit representation of changes in the ESP resulting from future management actions will provide valuable information on these interdependencies, which can be fed back into the development of ESP scenarios. Such a refinement of ESP scenarios over time in combination with cyclic assessment and management frameworks, such as the MESMIS framework for assessing the sustainability of natural resource management systems (Lopez-Ridaura et al., 2002), can provide the basis for an adaptive management approach.

5. Conclusions

The proposed framework for ecological quality assessments based on ecosystem services, unlike traditional approaches (e.g., general water quality standards), accounts for the fact that the value put on the state of a system will depend on the specific management context (landuse, population density, etc.) and associated societal expectations. For instance, in catchments of low population density and land use intensity, conservation of biodiversity and aesthetics might be predominant valued services, whereas in urban dominated catchments the supply of drinking water, flood mitigation, and attenuation of pollutants tend to be highly desired ecosystem services. If the expectation–supply balance is periodically reviewed, it also allows for changes over time, either through changes in societal expectation, or changes in external drivers of function such as climate. By incorporating a suite of services, and their interdependencies, demand for one service has to take into account not just the provision of that service, but the effect of provision at that level on others—where the services may include supporting, provisioning, regulating and cultural services.

We suggest that the ESP framework could provide an adaptable and robust approach, which has the potential to foster a more integrative approach to ecosystem assessment and management in the future and to meet the broad challenges laid down by new and forthcoming environmental legislation. There are significant research challenges ahead before the concept could actually be applied in practice, but there is only merit in detailed discussion about the practicalities if the basic concept has merit. Our aim here is to stimulate debate about the idea.

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