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ORIGINAL PAPER

A conceptual framework to assess the effects of environmental change on ecosystem services

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Abstract A new conceptual framework is presented for the assessment of the impacts of environmental change drivers on ecosystem service provision and the policy and management responses that would derive from the valuation of these impacts. The Framework for Ecosystem Service Provision (FESP), is based on an interpretation of the widely-used Drivers-Pressures-State-Impact-Response (DPSIR) framework. FESP differs from the DPSIR by offering clarity in the definitions of the various DPSIR components as well as introducing novel elements of relevance to the ecosystem service approach. The value of a common framework lies in making the comparison across competing services accessible and clear as well as highlighting the conflicts and trade-offs between not only multiple ecosystem services, but also multiple service beneficiaries. The framework is explicit, for example, in recognising as state variables not only the attributes of the Ecosystem Service Providers (ESPs), but also the attributes of the Ecosystem Service Beneficiaries (ESBs). That a service depends as much on the attributes of the people whose well-being benefits from the service as on the attributes of the biology providing the service is an important step in integrated social-ecological thinking. FESP also identifies the mechanisms of either mitigation or adaptation to the environmental change problem through the effect of these response strategies on specific pressure or state variables. In this way, FESP can contribute to the policies and strategies that are used to support conservation management. This paper describes the principles of FESP and presents some indicative examples of its practical implementation.

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Abbreviations

DPSIR	Drivers-Pressures-State-Impact-Response
EEA	European Environment Agency
ESB	Ecosystem Service Beneficiary
ESP	Ecosystem Service Provider
FESP	Framework for Ecosystem Service Provision
IPCC	Intergovernmental Panel on Climate Change
MA	Millennium Ecosystem Assessment
SES	Social Ecological System
SPU	Service Providing Unit
SRES	Special Report on Emissions Scenarios

Introduction

Predicting environmental change and its impacts on ecosystems and human well-being at local to global scales remains a significant challenge for the international scientific community (Carpenter et al. 2009; MA 2005). This is because the Earth is made up of complex, interactive and non-linear dynamic systems that are often unpredictable. For instance, although climate and natural resources have significantly shaped the development of civilizations throughout history, human society now impacts on, and in many cases dominates, the regulation of our climate, biogeochemical cycles and biodiversity, all of which are essential to human life itself. Uncertainty about the interactions and feedbacks between the natural and human drivers of environmental change that may operate at different spatial and temporal scales, make it difficult for societies to resolve an appropriate course of collective action to pursue sustainable livelihoods. Ehrlich and Ehrlich (1970) suggested that "the most subtle and dangerous threat to man's existence … is the potential destruction, by man's own activities, of those ecological systems upon which the very existence of the human species depends". More recent studies have continued to emphasise the importance of human–environment interactions through the relationships that exist between people, biodiversity and ecosystems (De Chazal and Rounsevell 2009; Luck et al. 2009). That is human society and the economic and legal institutions they develop, drive, both directly and indirectly, changes in biodiversity, changes in ecosystems and ultimately changes in the services ecosystems provide (MA 2005).

The provision of ecosystem services in the future will be influenced by multiple and interacting environmental change drivers. Assessing the complex effects of these drivers is not a trivial task, made all the more difficult by the lack of consistent conceptual frameworks that integrate across multiple ecosystem services. The Millennium Ecosystem Assessment (MA), for example, emphasized the lack of a theoretical framework to link ecological diversity with ecosystem dynamics and, in turn, to the ecosystem services underlying human well-being that is manageable at multiple scales (Carpenter et al. 2006). A framework that integrates across multiple services and multiple scales is desirable for a number of reasons as it would: (a) create a common basis to compare applications in different contexts; (b) standardize concepts and terminology; (c) make explicit the exogenous and endogenous components of the system and; (d) build on established approaches that are embedded in a number of policy and decision-making organizations and institutions. The MA (2005) proposed a framework to explore the broad relationships between ecosystem service provision and human well-being that addressed different types of direct and indirect drivers, but making such a framework operational within the context of conservation management remains a considerable challenge.

The Drivers-Pressures-State-Impact-Response (DPSIR) framework was originally derived from the social sciences (Rapport and Friend 1979) and later became more widely adopted as a general framework for organising information about the state of the environment (EEA 1995). The framework assumes cause–effect relationships between interacting components of social, economic and environmental systems, which are described below and exemplified through the issue of climate change:

- Driving forces of environmental change (e.g. increasing atmospheric greenhouse emissions)
- Pressures on the environment (e.g. temperature and precipitation changes)
- State of the environment (e.g. lowered crop production)
- Impacts on population, economy, ecosystems (e.g. food insecurity and malnutrition)
- Response of society (e.g. policy response, such as the Kyoto protocol for reducing greenhouse gas emissions).

In recent years, the DPSIR framework has evolved into an interdisciplinary tool for environmental analyses (EEA 1995, 1999). The framework is useful in that it provides a structure in which a number of physical, biological, chemical and societal indicators can be analysed to set and evaluate targets and give a clear picture of progress or lack of progress in a number of policy areas (EEA 1999).

More recently, criticisms have been raised about the linearity of cause–effect schemes such as the DPSIR (Svarstad et al. 2007; Fusco 2001). However, much of this criticism confuses linearity with sequential thinking. Whilst the components of the DPSIR are represented in sequence, there are potential feedbacks both within (e.g. state dynamics) and between its various components and this accounts for non-linear behaviour. For example, the feedback in the DPSIR schema is explicit in the action of the responses made by society (such as through policy). Other events are connected as a linear sequence of causes and effects and the connections between drivers, pressures and responses are much more complex as a result of (i) positive and negative feedback responses between different activities, (ii) economic and social mechanisms, and (iii) policy responses having multiple effects (Fusco 2001). Moreover, the DPSIR is a framework for environmental assessment; it is not a model. Its various components may in practice be represented by models of more complex processes and interactions, including feedbacks and thus the potential exists to develop within this framework a wealth of non-linear process interactions. This is consistent with the view expressed by Carpenter et al. (2009) who call for more research on dynamics, feedbacks and non-linear interactions between social and ecological systems.

An important strength of the DPSIR approach is that it emphasizes the role of humans-in-nature (Berkes and Folke 1998) by representing a system that includes societal (human) and ecological (biophysical) subsystems in mutual interaction, consistent with the use of the term Social-Ecological Systems (SES) (Gallopin 1991). Furthermore, the indicator system of the DPSIR is well established and embedded in a number of policy decision-making organizations and institutions. These indicators are useful for describing the impacts of environmental change, such as climate change or habitat loss and degradation, on the provision of ecosystem services measured over appropriate geographic scales (EEA 1999). The DPSIR is also specifically geared towards policy and management development, explicitly structuring statistics and indicators across the interactions between man and nature, which should ensure 'buy-in' from many stakeholder organisations involved with monitoring of indicators related to demographic, socio-economic and environmental conditions. A significant weakness of the approach, however, is the lack of consistency in its application to environmental problems. The various components can mean different things to different people and this inhibits universal and coherent applications. Thus, a vital aspect of the application of the DPSIR is in defining the focal question of any analysis. Furthermore, the DPSIR is not clear in identifying appropriate indicators for monitoring and assessing change, especially within the broader context of complex social-ecological systems rather than the narrower remit of physical environmental impacts (see Feld et al. 2010).

In this paper we discuss how the DPSIR framework may be applied to investigate the complex dynamics of environmental change drivers that influence ecosystem services and societal responses to better manage and protect them. In doing so, we propose a new formulation of the DPSIR known as the Framework for Ecosystem Service Provision (FESP) that is more explicit about its various components and the role of people within these components. The paper discusses the proposed modifications to the DPSIR framework that leads to FESP and gives examples of its application to ecosystem service assessment.

Towards a Framework for Ecosystem Service Provision

The concepts that underpin FESP have derived from an extended process of review and consultation with a number of experts in ecology, social sciences, economics and policy through workshops and e-conferencing that was coordinated by the RUBICODE project.[1] The starting point for FESP is the focal question, i.e. the question we want the framework to help us to address. This question is defined here as: how will environmental change affect ecosystem services and how will society adapt to maintain service provision? In addressing this question we need to define the various components of the modified DPSIR framework that are illustrated in Fig. 1. *Drivers* are the underlying causes of environmental change that are exogenous to the system or region in question, e.g. climate and socio-economic change, national and international policy. They reflect either the past, present or future conditions that cause changes to ecosystems. Future conditions are often described using qualitative, narrative storylines or through quantitative scenarios and projections such as those developed within the IPCC-SRES (Nakićenović et al. 2000). Drivers within the DPSIR framework are equivalent to the 'indirect drivers' of the MA. *Pressures* are the endogenous variables that quantify the effect of drivers within a system or region, e.g. temperature, precipitation, land cover, regional population, per capita water demand, crop prices or gross margins, and are usually assessed by developing regional, quantitative scenarios. Pressures in the DPSIR framework are equivalent to the 'direct drivers' of the MA, which are defined as physical, biological or chemical processes that tend to influence directly changes in ecosystem goods and services (Nelson et al. 2005). What is defined as

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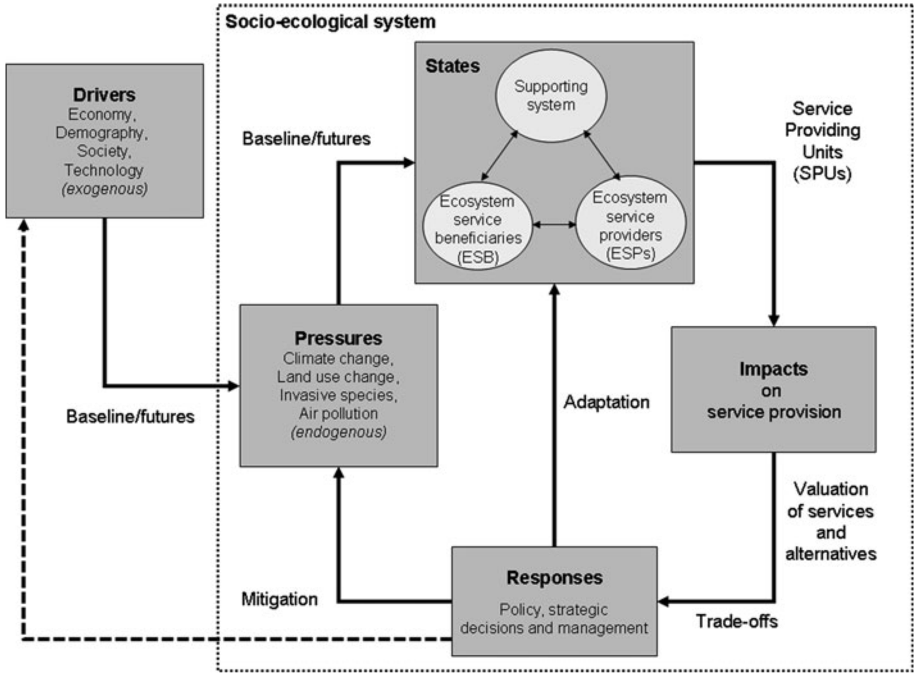

Fig. 1 A Framework for Ecosystem Service Provision (FESP) based on a modified Driver-Pressure-State-Impact-Response (DPSIR) framework

an exogenous driver or an endogenous pressure clearly depends on the location of the social-ecological system boundaries. Changing the location of this boundary by, for example, changing the spatial scale of observation may result in an exogenous driver becoming an endogenous pressure. The notion of a driver or a pressure is very much dependent, therefore, on the geographic extent of the system being considered.

State variables represent the sensitivity of the system/sector to the pressure variables. This involves the definition and quantification of all those elements relevant to the supply of the ecosystem service by biological organisms and the demand for ecosystem services from people. Thus, states are made up of variables that describe the whole of the social-ecological system, including the attributes of the Ecosystem Service Beneficiaries (ESBs) and the attributes of the Ecosystem Service Providers (ESPs). Formal recognition of ESBs is a departure from the traditional use of the DPSIR framework, but an important part of ecosystem assessments that integrate the role of people in shaping and benefiting from the natural world. As the attributes of the ESBs change then so do the ecosystem services being demanded and consequently the attributes of the ESPs supplying those services. ESBs may have very different attributes and demand very different ecosystem services. Wealthy individuals may have the luxury of demanding recreational and aesthetic services from ecosystems whereas less wealthy people may demand the supply of food and fibre services. The provision of any service can only, therefore, be considered in light of information about the attributes and demands of ESBs. Consequently, it is important to appraise conflicts between ESBs and those stakeholders who may be affected negatively by a service. Balancing negative and positive effects, which is the result of a societal process, enables the quantification of net service requirements.

ESPs represent the biology that is able to provide a service at a level demanded by ESBs. It is important to recognise that several different ESPs could provide the same service, and the actual in situ ESPs may depend on the location characteristics (i.e. pressures variables) and previous conservation management history. The quantity of the biology needed to supply a service is encapsulated in the term Service Providing Unit (SPU) (see Luck et al. 2003, 2009). SPUs are concerned with the identification and quantification of the organisms and their characteristics that provide services and how changes in these organisms impact on service provision. The relevant SPU characteristics which need quantifying will depend on the services in question and the organism(s) that supply them. Knowledge of population size may be important because an SPU may become functionally extinct below a critical threshold size. Population phenology may be important because the timing of service provision, for example pollination, is often critical, while population distribution can affect the functioning of an SPU. For example, if a population of a given size is highly aggregated, it may not perform its function as well as a more dispersed population of the same size.

The original SPU concept (Luck et al. 2003) focused primarily on populations of individual species. It was later combined with the concept of ESPs introduced by Kremen (2005) to form the SPU–ESP continuum (Luck et al. 2009), which we refer to here simply as the service-provider (SP) concept. This concept encompasses population, functional group and community levels. If the SP is a functional group, important characteristics include the intra- and inter-specific dynamics of the members of that group, the functional importance of each member (defined by factors such as abundance and relative contribution to service delivery) and the functional compatibility among members (i.e. if a member species is lost from the group will other species compensate completely to ensure no disruption to service delivery?) (Luck et al. 2009). Intra- and inter-specific dynamics are also important at the community level, but basic measures may be used to gauge relationships between communities and service delivery such as the area of a forest needed to provide a water filtration service. In the case of services that can be provided by more than one genotype or species, the traits that are important to service provision (‘effect traits’, see Harrington et al. 2010 for a definition) must be known, as different genotypes or species may contribute to service provision to a different degree at different times or in different places (de Bello et al. 2010). This is likely to be the case for most services. For example, in the case of conservation biocontrol of aphids, insects such as hoverflies, ladybirds and lacewings can be valuable. Quantification of the traits present such as voracity, prey handling time and intrinsic rate of population increase is therefore more informative than quantification of the species themselves.

The relationship between SPs and service provision may be complex, particularly for multi-species SPs, and it can be useful to define associations with organisms or systems that support them. Furthermore, it must be recognized that service-providing species are embedded in an ecosystem, and to separate out some species from the rest is generally biologically unrealistic as there are many interactions taking place that are not observed and certainly not measured (Haslett et al. 2010). Therefore, the sensitivity of the entire ecosystem or habitat should be considered in terms of its supporting role for the SPs.

The *Impact* is a measure of whether the changes in the state variables have a negative or positive effect on individuals, society and/or environmental resources. In the framework presented here the negative or positive effects are measured in terms of the capacity to provide a given service. Not supplying a service implies a negative impact. An important part of this assessment is the notion that an SPU could be interpreted as the minimum (threshold) quantity of biology required to supply a service at the required level (and we use the term 'SPU' here to emphasise this point and differentiate from the related SP and ESP concepts). The state variables of the ESPs change through time in response to the changing pressure variables. This in itself is not a problem until the quantity of available ESPs reaches it threshold value and an impact ensues. Changes in the attributes of the ESBs and the ESPs may reflect autonomous adaptation processes. Examples of this include changes in the physiology or morphology of ESPs in response to environmental conditions, such as leaf stomatal changes that occur when plants grow in elevated atmospheric CO_2 concentrations, or the capacity of ESBs to learn new ways of behaving in order to better manage their environment as that environment changes.

Finally, Responses through planned policy and management aim to minimise negative impacts (or maximise positive impacts/benefits) by acting on the socio-economic pressure variables or directly on the state variables. The different routes to minimising impacts reflect different generic types of response strategies. The route through the pressure variables (and potentially through the drivers) is based on 'mitigation' or reducing the severity of the problem. Examples of this include policy measures to change water consumption, restrict urban development and other land use planning regulations where land use change is a pressure. The consequences of different mitigation options can be tested using models by perturbing the pressure variables and assessing the consequences of this for the impacts. Iteration of this approach may be needed to explore the effects of multiple response options and to optimise management strategies. The direct route to the state variables is based on 'adaptation' or enhancing the capacity of the system to cope with pressures. Adaptation is relevant to both the ESPs and ESBs within a SES. Conservation management may, for example, be used to change the intrinsic properties of ecosystems and so enhance the capacity of ESPs to cope. Specific management examples include the improved mobility of species through corridors or their physical translocation from one location to another. Likewise, the capacity of people (ESBs) to cope can be enhanced by raising awareness by providing information, education and training.

In Fig. 1 the solid boxes reflect variables or indicators, and the arrows between the boxes reflect the influences between variables usually representing processes (e.g. valuation or policy decisions) or thresholds (e.g. SPUs). The boundary of the social-ecological system itself is represented in Fig. 1 by the large dashed-line box. Everything within the box is endogenous to the system. So, for example, the pressures (or direct drivers) represent the variables that act upon the ecosystem state. Conversely the drivers (or indirect drivers) are exogenous to the system. This means they are influenced primarily by factors, processes and interactions that occur outside of the social-ecological system under consideration. The states change in response to the pressures in different ways, the dynamics of which (temporality) are characterized by concepts such as resilience and robustness that are discussed in more detail elsewhere (see Dawson et al. 2010). As the state changes it may reach a certain threshold (the SPU, see Luck et al. 2003, 2009) that has negative (or positive) consequences for the service provision demanded by the service beneficiary. Thus, the SPU is a function of both the attributes of the biology (ESP) and the attributes of the service beneficiary (ESB). If either of the ESP or ESB attributes were to be different there would be a different SPU. Supporting systems do not have a direct influence on the service being provided, but are important in supporting the ESPs supplying that service. An example of this might be a particular habitat type that is needed by an organism as part of its life cycle.

The nature of the impact is assessed using valuation techniques, including for example, trade-offs with alternative (non-biological) approaches to service provision. Responses, such as policy and/or conservation management, are then implemented in response to the measured costs of the impact. Policies and management act on the pressures or states, as these are endogenous to the system. Policy cannot act on the drivers in any meaningful way as these are exogenous to the system and, therefore, are beyond the influence of the human actors operating within that system. It could be argued that, for example, C sequestration at a regional scale feeds back to the (global) climate system as a climate change mitigation strategy, but at the scale of an individual ecosystem this feedback would be trivial (hence the dashed line used in Fig. 1).

Figure 1 is a simple representation of the conceptual framework implemented for a single ecosystem service. In practice, however, conflicts and trade-offs exist between multiple services. Schematically this is represented in Fig. 2. In Fig. 2 the various components of the DPSIR, as shown in Fig. 1, are reduced to the interaction between people and ecosystems, but there is a presentation of multiple and overlapping social-ecological systems, each with potentially multiple ecosystem services and multiple drivers. In the case of multiple SESs it is important to note that the pressures within one SES may represent the states of another SES. An example of this is land use change, which may be a state variable in a human dominated ecosystem, but a pressure on less intensively managed ecosystems. Again, the importance of defining the focal question in applying the DPSIR is paramount in this respect. An example of multiple services and their potential conflicts is discussed later.

Implementation of the framework

The conceptual framework provides a means of structuring thinking about real world examples of ecosystem service provision. In implementing the concept to real examples, we propose a seven stepped approach, which seeks to describe and detail each of the framework's components in a logical and sequential way. These steps are presented in Fig. 3 and are based on defining the various components of the DPSIR. In the following section we discuss three examples of how the proposed conceptual framework might be applied in practice. The first example is based on the acorn dispersal service provided by jays in the National Urban Park in central Stockholm and is taken from Hougner et al. (2006). This is a simple example of the application of the framework based on a single ESP

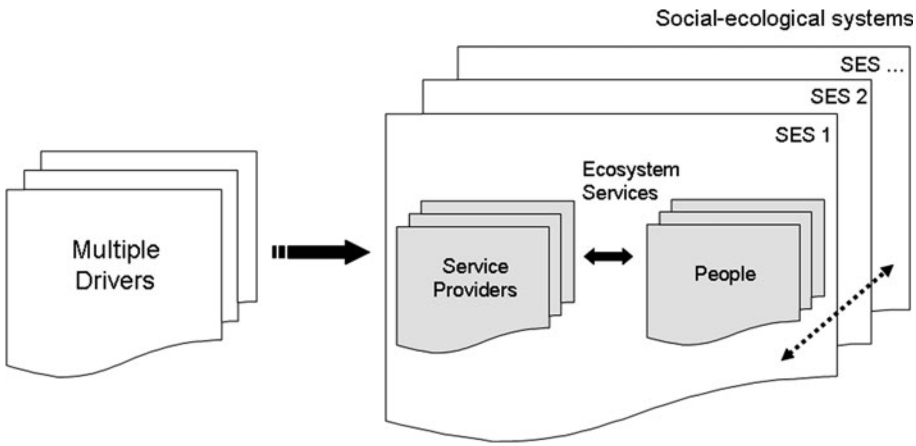

Fig. 2 Schematic representation of the case of multiple systems and multiple services

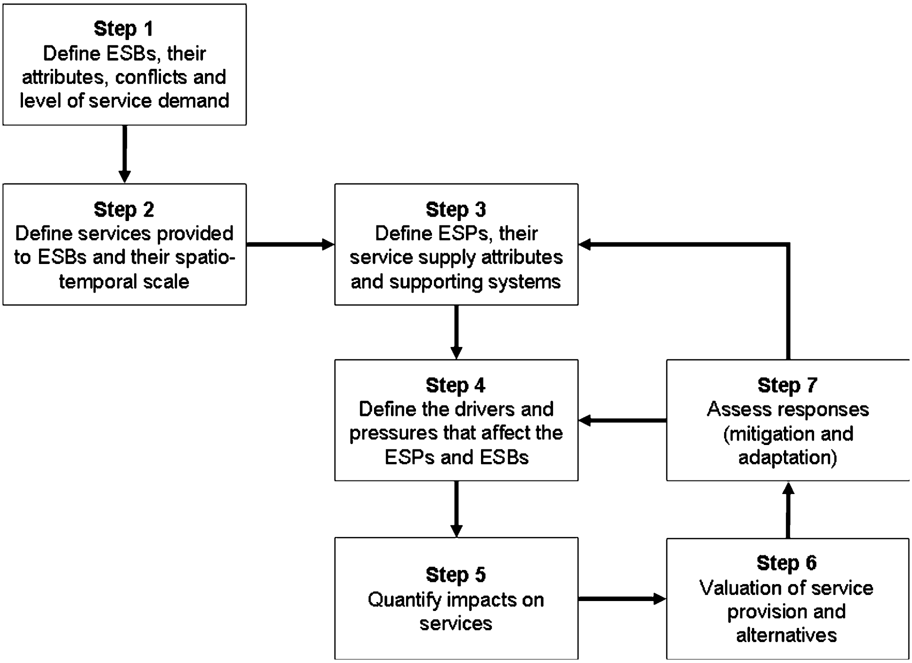

Fig. 3 Stepwise implementation strategy for the conceptual framework

and a single ESB. The second example discusses in a generic way the impacts of climate change on agro-ecosystems. The example is used to highlight the role of different mitigation and adaptation responses to environmental change. The third example is based on the relationships between hen harriers and grouse management in Scotland as an example of multiple services and potential conflicts between ESBs and how these conflicts might be resolved.

A simple example of a single service

Stockholm has the world's first National Urban Park, which with a total area of 26 km^2 is one of the largest oak forests in Sweden and one of Europe's largest populations of giant oaks. The Park is known for its unique and rich biodiversity with many rare species and oak trees that are up to 500 years old. It is important as a recreational area, and with more than 15 million visits per annum is the most frequented urban park in Sweden. The oak trees are an important structural component of the park and so, their regeneration is central to the provision of the cultural and aesthetic services enjoyed by the park visitors.

The ESBs in this case are the recreational visitors to the park (*Step 1*). Visitors are drawn mostly from the inhabitants of Stockholm, but also include tourists from other parts of the world. In this example there are no conflicts reported with other potential beneficiaries. The services provided to the ESBs are cultural and aesthetic in the use of the park for recreation (*Step 2*). One could envisage a very different set of ecosystem services for a different set of ESBs. The land occupied by the park might, for example, be used for food or fibre production. So, an important attribute of the ESBs here is that they have sufficient wealth to enjoy aesthetic services and do not need to use the land to produce food. The oak forest provides the direct service to humanity (*Step 3*). The Eurasian Jay (*Garrulus glandarius*) provides a seed dispersal service for the oaks. It collects and hides acorns during the autumn for later winter consumption at the ideal depth for germination and reduced predation. Such dispersal also enhances the gene pool of the oaks where 85% of the oaks are estimated to regenerate naturally. The jays, in turn, depend on the availability of coniferous forest to provide nesting sites, and this is defined as a supporting system. This example demonstrates the hierarchy of biology and the services they provide.

The oaks and the jays may be affected by drivers such as global climate change mediated through the pressure variables of temperature and precipitation, an epidemic oak disease or timber demand leading to land use change (*Step 4*). Alternatively the attributes of the ESBs depend strongly on drivers such as macro-economics which may affect their standard of living and thus demand for cultural and aesthetic services. The SPUs in this case are the number of oak trees that provide the aesthetic service and the minimum number of pairs of breeding jays that provide the seeding service to maintain this number of oaks (*Step 5*). This has been established as a minimum of 12 pairs of jays for the 2,700 ha park resulting in the establishment of 33,148 oak saplings per year (over a 14-year period). The number of oaks will, with time, reduce if the jays no longer provide the seeding service and so, the minimum number of jays defines the impact threshold (see Dawson et al. 2010). An alternative to the natural seeding of oak trees is for people to either seed acorns or plant oak saplings. Hougner et al. (2006) estimate the overall cost over 14 years of using human labour for this task to be 1.53 million SEK for seeding and 6.7 million SEK for planting (*Step 6*). The true value to the ESBs, however, may be considerably higher than these amounts if one considers the contingent valuation (willingness to pay) of park users. There is widespread public support for the maintenance of the park, which received formal status in 1995 and is now classified in the Swedish Environmental code as an area of national interest. New developments in the area are allowed, but only if they can be carried out without intruding on the park landscape and without affecting negatively the natural and cultural values of the area. Thus, the simple conclusion arising from this example is that it is much more cost effective to maintain jay populations in the National Urban Park rather than relying on human intervention to provide the service. Policy responses (*Step 7*) should seek, therefore, to continue investment in management that safeguards the jay population at a level suitable for the continued and successful regeneration of the oak forest. Appropriate jay conservation strategies include maintaining the coniferous forest upon which the jays depend for nest sites.

An example of mitigation and adaptation response strategies

The example of climate change impacts on agro-ecosystems exemplifies the role of mitigation and adaptation as alternative, and potentially complementary or conflicting (e.g. see Paterson et al. 2008) response strategies. The numerous impacts of climate change on agriculture have been widely reported in the literature (see Easterling et al. 2007 for a review). These impacts have important implications for the provision of services from agro-ecosystems such as food and fibre production and the cultural services engendered to rural communities. Within the framework proposed here the drivers of change derive from industrialisation, GHG emissions and subsequent changes in the global climate system. These drivers translate into pressure variables that describe the climate of a particular ecosystem (e.g. temperature, precipitation, atmospheric CO_2 concentrations, windspeed, variability, etc.). These pressure variables will affect agriculture in many ways. If we consider crop-based agro-ecosystems the pressures will be mediated through either direct effects (e.g. CO_2 fertilisation and higher temperature) or indirect effects (e.g. pests and diseases, pollination and soil workability) on plant growth with consequential changes in state variables such as crop yields (Olesen and Bindi 2002; Chakraborty and Datta 2003). The characteristics of the ESBs (in this case farmers) are affected by the socio-cultural context, political environment and availability of capital and these characteristics play a role in determining the level of management applied to the ecosystem and on the expectations of the ESBs for ecosystem services. These expectations would differ greatly, for example, between subsistence or cash crop farmers and the level of impact is compared against these expectations. Thus, an impact might be measured in terms of the complete failure of a crop in a subsistence agricultural system or the reduction of income to below a certain threshold for an agri-business. Appropriate valuation techniques (see Skourtos et al. 2010) can be used to express the magnitude of these impacts and explore trade-offs between conflicting impacts.

Individual farmers will respond to changing circumstances (pressure variables) autonomously by changing land use and management regimes. The planned responses to the impacts are enacted through policies of either mitigation or adaptation or a combination of both of these. The mitigation route could involve feedback to the climate system itself (e.g. through C sequestration, or GHG emissions abatement), but in practice, as discussed earlier, these effects would be trivial in terms of the magnitude of the feedback from the global climate system to the pressure variables. This is driver mitigation. Other mitigation options would seek to reduce the severity of the effect of the climatic pressure variables (pressure mitigation). A good example of this being the use of irrigation to offset water losses arising from reduced precipitation or greater evapo-transpiration (e.g. Döll 2002; Fischer et al. 2007). Adaptation strategies would address the capacity of the system to cope with changing climatic pressures. This might include using pesticides to cope with pests and diseases, changing crop planting dates to account for changing growing seasons or planting new crop varieties (e.g. Howden et al. 2003; Challinor et al. 2007). Importantly these management actions would be informed by other planned adaptation options that target the capacity and attitudes of the ESBs such as the transfer of knowledge and information about new management practices, or providing financial incentives for certain management strategies (e.g. Aggarwal et al. 2004; Antle et al. 2004). The distinction made here between mitigation and adaptation responses attempts to clarify the goals of a particular response option, and how this option acts on the various components of the coupled social-ecological system.

The differences between the exogenous and endogenous parts of the system framework are extremely important in understanding how people through policy or management maintain the provision of services. The system boundary defines what is exogenous and what is endogenous to the system and thus, the variables that the response measures are able to act on. When dealing with multiple overlapping systems that are operating at different scales, the definition of the system boundary is critical. It is also important to recognise that people can influence the location of the system boundary by endogenising exogenous variables. Irrigation in agro-ecosystems is a good example of this point. If we consider food production at the regional scale, precipitation is an endogenous pressure variable that affects crop production, but precipitation is controlled by the drivers of the global climate system: the climate system being exogenous to regional food production. A farmer may, however, seek to endogenise the climate system by supplementing precipitation with irrigation water with the aim of maintaining the food provision service, i.e. mitigating the effect of precipitation. Thus, a system of rainwater catchment and storage coupled to an irrigation system can provide water on demand through regulation (Barron

2009). This approach replaces the uncertainties associated with rain-fed agriculture and to some degree provides a buffer against unpredictable shocks such as drought events. In this example, crop production is more resilient to precipitation events (Dawson et al. 2010).

In a similar way, agricultural policy can be used to endogenise food production by removing the vagaries of exogenous economic factors. The European Common Agricultural Policy (CAP) controls the import of food into the Europe Union (EU) from other countries. It achieves this through import tariffs and other barriers to trade that maintain food prices within a European market with the aim of securing the supply of food to consumers. Without these trade barriers, European food prices would fluctuate at the world price level. The CAP draws the system boundaries as the geographic limits of the EU, and effectively endogenises food prices and production within the internal European market. Under free market conditions food production would be dependent on world markets and so, exogenous to the EU and out of the control therefore of European policy intervention. Thus, consideration of the categorisation of drivers and pressures on ecosystem services into exogenous and endogenous processes is constructive. This supports the policy-maker in management decisions regarding whether there can be an influence on those drivers and pressures, through mitigation for example, or that the pressure and drivers affecting the system are externalities with policy decision-making having no significant influence. In the case of the latter, policy will attempt to control for those exogenous drivers through adaptation. The disadvantage of endogenising external or exogenous drivers, however, is that the management of the social-ecological system becomes more complex through the addition of processes and interconnections that are internally organised.

Conflict on UK grouse moors

The examples presented above reflect relatively simple situations involving single services provided by single sets of organisms. An ecosystem with multiple services, multiple providers and multiple beneficiaries becomes more complex, but can also be conceptualised within the same basic framework. To demonstrate this, we describe here the example of UK moorland as an ecologically important ecosystem that demonstrates conflicts between ESBs for multiple ecosystem service provision. The example is taken from the work reported by Thirgood and Redpath (2008). Moorland supports, amongst others, two important bird species: red grouse *Lagopus lagopus scoticus* (Latham) and hen harriers *Circus cyaneaus* (L.). Many UK moorlands are managed for grouse hunting, but grouse are predated by hen harriers with significant effects on their populations. There are two groups of ESBs in this example: conservationists who aim to conserve hen harriers, and hunters (as well as grouse moor managers) who aim to maximise grouse numbers. Clearly the different aims of the two ESBs are in direct conflict.

We can apply FESP to this problem by considering in turn the ecosystem service that is provided to each ESB. For hunters, the ESPs are the grouse themselves, who provide the amenity service of hunting and for which the state variable is the grouse population. The impact for hunters is defined as the size of the grouse population (the SPU) below which hunting is no longer viable either in terms of the number of birds taken or the financial return they provide (shooting rights are usually sold). Redpath and Thirgood (1997) estimated the cost of loosing the hunting service for a commercial shoot in the UK to be about £100,000 per annum. There are many potential pressures on grouse populations (e.g. climate variability, disease and habitat loss), but the population of hen harriers is the single most important pressure. The response of hunters (through grouse managers) is to mitigate the pressure of hen harriers on grouse populations by killing the raptors. For conservationists, the ecosystem service is the biodiversity benefits offered by the population of hen harriers with the impact being defined in terms of minimum viable breeding populations. The pressure on hen harrier populations comes primarily from being killed by grouse moor managers, so the response of conservationists to mitigate this pressure has been to lobby for the implementation of policy to protect the raptors. Such policy has been in place in the UK since 1952. A conflict clearly arises therefore between the provision of grouse and the provision of hen harrier services. Moreover, hen harriers continue to be killed (illegally), but also continue to affect grouse numbers so that neither ESB is satisfied with the level of service provided to them, at present.

The responses currently implemented by each ESB are based on mitigating the pressures. However, there is also the potential to implement adaptation strategies that affect the state variables directly. Thirgood and Redpath (2008), for example, suggest that stakeholder (ESB) views are often entrenched, which suggests considerable scope to moderate the behaviour and/or expectations of the ESBs as a response strategy that could resolve the conflict. Indeed Thirgood and Redpath (2008) propose different ways in which this could be achieved. Notably, for example, by imposing a quota on hen harrier numbers on all grouse moors (through culling), but permitting the introduction of hen harriers (also within a quota) to moors where they currently do not exist. This would allow grouse numbers to be maintained on all moors, but would increase the total hen harrier population by increasing their geographic distribution. The acceptability of such an approach requires a change in the attitudes of both hunters and conservationists in order to accept a trade-off. Thus, the example demonstrates how the FESP may contribute to the resolution of ecological conflicts by first establishing the role and attributes of the ESBs. Appropriate adaptation strategies may then be devised that satisfy the ESB requirements for ecosystem services that avoid conflicts.

It is interesting to note that Thirgood and Redpath (2008) strongly criticise much of the past ecological research on grouse moors for its lack of attention to the human dimension of the problem. Again, making ESBs an explicit part of the ecosystem approach is an important contribution to conservation and is consistent with the anthropocentric nature of the ecosystem service concept.

Discussion and conclusions

Table 1 provides further examples of the application of FESP to a range of ecosystem service issues, which are organised according to the step-wise approach presented in Fig. 3. What is striking about these examples is their diversity in terms of ESBs, ESPs, SPUs, valuation methods and response options, but that in spite of this diversity each example fits into the FESP approach. Consequently comparison across the different examples becomes more amenable through the use of a common framework. The examples in Table 1 also demonstrate how FESP may be used to identify a range of environmental indicators that are appropriate in the assessment of ecosystem service provision. Indicators are apparent primarily as pressure and state variables and so reflect the underlying causes of change and the magnitude and direction of the change itself.

Ecosystem service management, policy and governance involve many different spatial and temporal scales, even for a single service. Equally, the different states (habitats, species, human behaviour and their inter-relationships) also involve a wide range of scales. Thus, it is important to state that scales of SES will vary and that there are many nested scales. Consequently, it is necessary to clearly define scale or the range of scales for any

Table 1 Selected examples from the literature for the application of the FESP stepwise implementation strategy

Step 1 (ESBs)	Step 2 (service)	Step 3 (ESPs)	Step 4 (drivers and pressures)	Step 5 (SPU and impacts)	Step 6 (valuation)	Step 7 (responses)
Regional farmers (central French Alps)	Fodder production[a]	*Festuca paniculata* functional group (fibrous and N-poor leaves) and *Dactylis glomerata* functional group (tender and N-rich leaves)	Management intensity, land use change, nitrogen deposition, climate change	Fodder quantity related to plant stature and abundance of the 2 ESPs. Fodder quality related to the community's average leaf nitrogen content	Perceived value by local stakeholders (farmers)	Promote traditional fertilisation and mowing practices with market and/or direct financial support to farmers. Improved product branding
Local watermelon farmers in California	Pollination of watermelons[b]	Native bees that pollinate watermelon crops	Deforestation, fragmentation, agricultural intensification, climate change, disease, competition with honeybee/Africanised bee, market demand	Functional groups with sufficient diversity (20–30) and abundance of bee species to cope with spatial and temporal environmental variation and visitation rate and pollination effectiveness (pollen deposition rate)	No economic valuation, but this could be based on trade-offs between rental costs, protection of native bee habitat, opportunity costs (loss of farmland to native bee habitat), crop yield and crop price	A balance between protecting native habitat for bees and losing land area for crops. Gaining wider acceptance of the contribution of native bees to pollination
Coffee growers in Costa Rica	Pest regulation in coffee plantations[c]	Shade trees, especially poró (*Erythrina poeppigiana*)	Intensification, land use change, climate change	The optimum density and distribution of shade trees has not been calculated	The value of increased coffee yield due to reduced herbivory achieved by shade plants versus the costs of coffee as a monoculture with chemical control	Promoting use of shade trees as part of a more sustainable pest control strategy
General public	Water purification[d]	Multi-species-multi-zone riparian plant community (trees, shrubs, herbs and grasses)	Agricultural land use (food and biofuel demand), intensification	Nutrient and sediment filtration related to the number, density and width of buffers	Calculation of replacement costs if the service were provided by conventional waste water treatment plants	Restoration of riparian buffers to meet the demands of the Water Framework Directive

Table 1 continued

Step 1 (ESBs)	Step 2 (service)	Step 3 (ESPs)	Step 4 (drivers and pressures)	Step 5 (SPU and impacts)	Step 6 (valuation)	Step 7 (responses)
Global population	Climate regulation[e]	Multi-species forest populations in Changbaishan Reserve, China, comprising Korean pine, broad-leaved species and spruce	Forestry and timber demand, deforestation, fragmentation, climate change	The area and biomass of each species and thus, the amount of pure carbon stored per year has been calculated	The value of carbon stored and other ecosystem services estimated at 510 million yuan per year—10 times higher than the opportunity cost of timber production	Reducing CO_2 emissions as an obligation under the UN Climate Change Framework Convention
Farmers, residents, road users, hunters, horse riders, hikers and bird watchers of south-central Washington State	Erosion regulation[f]	Plant communities of the shrub-steppe habitat	Land use change, overgrazing, climate change	The protective ability of the plants was related to their canopy cover, root growth, abundance and distribution	Opportunity costs and benefits were estimated for different aspects of soil stabilization in dollars/acre/year	Appropriate land management and conservation policy to maintain shrub-steppe vegetation cover
Bird and Game NGOs, and the general bird-loving and bird-shooting public	Aesthetic value[g]	Farmland birds	Land use change, agricultural intensification	Abundance and distribution of 19 species of farmland birds with a range of desired traits and trait values (colour, size, song, taste, etc.)	Willingness to pay for membership of relevant NGOs. The value of reduced healthcare costs arising from the health benefits of visiting the countryside	Conservation management to maintain abundance above the current index level in compliance with the 2010 CBD target

Table 1 continued

Step 1 (ESBs)	Step 2 (service)	Step 3 (ESPs)	Step 4 (drivers and pressures)	Step 5 (SPU and impacts)	Step 6 (valuation)	Step 7 (responses)
Local farmers, who benefit financially from crop production, to the global population, who benefit from the availability of food and fibre	Nutrient cycling[h]	Soil invertebrates with an emphasis on saprophagous organisms (e.g. detritivores and microbivores), in particular earthworms (e.g. *Lumbricus rubellus*, *Lumbricus terrestris*, *Aporrectodea caliginosa*)	Land use change (e.g. land abandonment, afforestation and deforestation), agricultural practices (e.g. intensive management, intensive grazing and soil mobilization and compaction), pollution	Earthworm individual species densities were related to decomposition and nutrient cycling	Indirect economic value of food supply	Habitat management strategies to enhance nutrient cycling and organic matter decomposition by promoting earthworm abundance and diversity
The global population, particularly in arid and semi-arid regions with eroded soils at risk of desertification	Water cycling[i]	Soil macrofauna, in particular bioturbators (e.g. termites, ants and earthworms)	Land abandonment, urban expansion, deforestation, agricultural intensification, climate change, overgrazing (and trampling by cattle)	Species density: at least 30 termite macropores per square meter are necessary to assure water retention by significantly decreasing runoff	The economic and social values of soil quality and crop production: bioturbation has positive effects on crop productivity	Reduced tillage to limit soil erosion and improve soil physical properties. Management to enhance termite activity. Sustainable grazing management

References: [a] Quétier et al. (2007), [b] Kremen et al. (2002, 2004), [c] Varon et al. (2007), [d] Correll (2005); Dosskey (2001), [e] Xue and Tisdell (2001), [f] Scott et al. (1998), [g] Butler et al. (2007), [h] Postma-Blaaw et al. (2006), [i] Leonard and Rajot (2001) given situation and frameworks such as FESP can assist in doing this. The boundaries of the system need to be flexible as these can be different in different situations. Furthermore, there is often a mismatch between the scale of the system and the scale at which policy-makers operate. A larger system boundary may be needed if we want to assess feedback processes.

Temporal dynamics are an important component of the ecosystem service approach since responses are often subject to time lags (Dawson et al. 2010). Decision-making needs to be re-evaluated through time (as the system is dynamic) and adapted to the evolution of drivers and state changes. The time scale of relevance to policy-makers tends to be short whilst ecosystem responses occur over much longer time periods. Three possible time scales can be identified: political, biogeochemical cycling and ecological, each with their own sub-components that operate at different scales. It is difficult to generalise about the temporal scale that is the most important and this needs to be defined for each situation.

The conceptual framework represented here can promote insight into the properties of social-ecological systems and their responses to a variety of drivers and pressures, and thus aid understanding of sustainable conservation strategies. It has a number of advantages over previous realisations of the DPSIR framework. First amongst these is that it is explicit in identifying the role and attributes of service beneficiaries in the ecosystem service concept. Previous use of the ecosystem service concept has focused on the attributes of the biology providing the service, i.e. the supply-side of service provision, rather than the attributes of the beneficiaries or the demand-side of service provision (Luck et al. 2009; Harrison et al. 2010). The recognition that a service depends as much on the attributes of the people whose well-being benefits from the service as on the attributes of the biology providing the service is an important step in integrated social-ecological thinking. An ESB focus also highlights the potential for conflict between beneficiaries. Different groups within society may benefit from different ecosystem services, and it is often the role of policy to make judgements about which benefits are realised in practice. A balanced approach to these policy decisions becomes increasingly difficult to achieve when one considers the trade-offs between not only multiple beneficiaries, but also multiple ecosystem services. Thus, the value of a common framework lies in making the comparison across competing services accessible and clear. The proposed framework also makes explicit the potential for different ESPs, the thresholds of biology (SPUs) that is needed to supply a given service, and the role of valuation in assessing trade-offs between service provision strategies as part of the response decision process. Furthermore the framework is able to identify the mechanisms of either mitigation or adaptation to the environmental change problem and how these response strategies affect specific pressure or state variables.

This paper has set-out the principal elements of FESP as a means of pulling together the different elements of the ecosystem service concept. It has not attempted to describe in detail the various components of FESP, for example, the valuation process or the assessment of the biodiversity that underpins ecosystem service provision, which are covered elsewhere in this issue. Moreover, it is important to note that FESP does not substitute for other frameworks for conservation such as protected area strategies (IUCN 1994) or adaptive management schemes (Salafky and Wollenberg 2000), but complements them through acknowledgement and identification of potential conflicts between beneficiaries and management opportunities to maximise the *comparative advantage* of individual species, SPs and ecosystem processes within the system as a whole. FESP has, however, its limitations. It certainly needs more comprehensive testing against a wider range of real world examples, which would help in refining the framework and its practical application. Furthermore, FESP does not describe in detail the ecological or human processes that make up complex social-ecological systems; it is afterall a framework and not a model. It also does not help in making the value judgements that are needed to translate state variables into impacts through the definition of thresholds. Value judgements are normally beyond the aims of frameworks that seek instead to support such judgements by providing structure to the organisation of information and relationships. However as a starting point in applying the concept of coupled social-ecological systems thinking, FESP has many positive aspects that merit its further development and application in ecosystem service assessment and conservation management.

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