

# Identifying robust response options to manage environmental change using an Ecosystem Approach I: developing a working typology

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## Abstract

A diverse range of response options is available for decision-makers to manage environmental change and meet sustainability objectives. These can include *inter alia*: top-down statutory regulation and levies; bottom-up initiatives including quality assurance networks or community-based partnerships; formal incentives; and voluntary market-based schemes such as 'payments for ecosystem services' or offsetting. Each type of response option has a distinct set of characteristics, which suggests that they may be best suited to different contexts rather than presumed to be effective in all circumstances. These attributes are used to develop a working typology to help understand the strengths and weaknesses of different response types, particularly regarding adaptation to long-term change and handling of uncertainty. To facilitate this, response types are referenced from a socio-ecological systems perspective using a refined version of the DPSIR integrated assessment framework to incorporate ecosystem functions and services. This shows that some responses are more clearly associated with maintaining resilience of natural functions whilst others are directed at human-defined services. Polluter-pays approaches can be contrasted with beneficiary-pays schemes through the role of stakeholder involvement and policy goals. The typology and framework can therefore provide a reference for recognising complementary rather than conflicting interventions, as guided by the holistic principles of the Ecosystem Approach.

*Keywords: response options, robust decision making, typology, environmental change, ecosystem services, integrated assessment, policy appraisal*

## 1. Introduction

Decision-makers have potential access to a varied suite of response options to manage environmental change, each option representing a distinct type of intervention or other form of influence on human-environment interactions (Sterner, 2003). The range of potential approaches has been stimulated by the diversity of issues to be addressed. Heterogeneity of socio-ecological systems would suggest that it is not realistic to identify particular response options as universal panaceas that can address all circumstances (Ostrom, 2007). However, it is common for institutions to continue to use a narrower set of familiar responses, despite their limitations, rather than strategically assessing the suitability of multiple options based upon their suitability for specific purposes as defined by decision objectives (Tonn *et al.*, 2000; Leach *et al.*, 2010).

The interconnectedness of socio-ecological systems also means that conflicts and inefficiencies are likely to result if decision objectives address only narrow outcomes, overlooking the potential for unintended consequences (i.e. externalities) elsewhere. For example, although the natural environment provides multiple societal benefits, over-emphasis on the provision of food and fibre at the expense of other less tangible services can have consequences for maintaining ecosystem resilience and net societal value. These trade-offs can be made explicit through an ecosystem services (ES) framework: in this example, food and fibre obtained through provisioning ES may have been prioritised to the detriment of other non-focal ES (supporting, regulating and cultural ES) (MA, 2005; Rodriguez *et al.*, 2006). Reducing such externalities implies the need for a more systemic basis to decision appraisals with objectives widened to identify response options that optimise outcomes across a broad array of ES rather than maximising only one or a few focal services (Everard and McInnes, 2013). Multiple benefits are also more likely to be realised if the decision criteria and their evaluation take better account of changes in external drivers that may modify the long-term sustainability of ES through both human behaviours and environmental processes (Janssen, 2002).

The complexity of socio-ecological systems has important implications for the knowledge and governance systems required to manage environmental change (Pahl-Wostl, 2007). The role of science in deciphering and communicating key issues to decision makers has often been constrained because of a frequent mismatch between broad holistic questions typically posed in policy formation and narrow reductionist questions that are susceptible to scientific method (Pullin *et al.*, 2009). This has highlighted a need to synthesise and communicate knowledge in formats accessible to decision-makers (Sarewitz and Pielke, 2007). A focus on the suite of response options available to decision makers can provide a conduit for this knowledge exchange, achieving synthesis by: (i) reviewing the role of different types of response options in the context of knowledge requirements and knowledge exchange; (ii) referencing response options against holistic frameworks that show cause-effect relationships and the role of different response options in influencing these relationships; and (iii) explicating which response options apparently work best in different contexts, including scale issues and the involvement of key stakeholders.

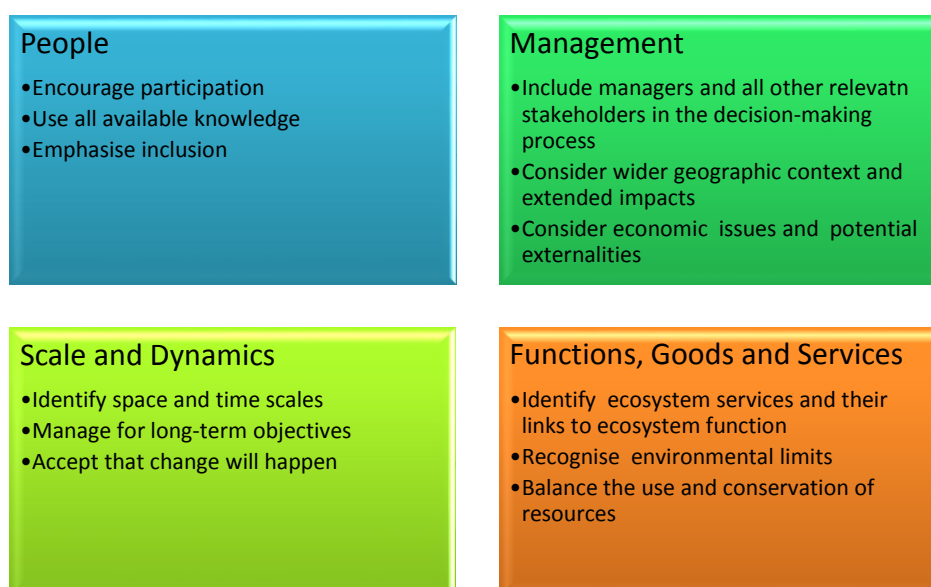
In this contribution, we develop a typology to describe, synthesise and facilitate comparison between response options based upon a set of common attributes. Then we evaluate the role of generic frameworks provided by the Ecosystem Approach and integrated assessment procedures in providing a holistic structure on which to evaluate response options with regard to sustainable long-term

delivery of ES. The role of different types of response options is then distinguished and compared with regard to sustainability goals.

Recently, the Ecosystem Approach has gained increased traction in decision-making (Fish, 2011) by framing societal and environmental issues within broader geographical and socio-economic contexts as guided by its 12 core principles (CBD, 2004). These complementary and interlinked principles may be summarised into four broad categories (Figure 1): (i) people and their inclusion; (ii) linking ecosystem functions and services; (iii) implementing ecosystem-based management; and (iv) recognising cross-scale linkages (spatial and temporal). Since publication of the Millennium Assessment (MA), implementation of the principles of the Ecosystem Approach has been particularly directed through use of the framework provided by ES to explicate the multiple societal benefits provided by ecosystems (MA, 2005), although attempts to value such services actually have a much longer timeline (Baveye *et al.*, 2013). A particular feature of the ES framework is the notion of a service 'flow' from natural ecosystem processes and functions to human beneficiaries in both space and time (Serna-Chavez *et al.*, 2014), which includes both material flows (e.g. crops for food) and non-material flows (e.g. cognitive and cultural benefits). Explicit recognition of these notional flows from a providing area to its beneficiaries another can then provide a focus for economic valuation and integrated management to address trade-offs (Bagstad *et al.*, 2013).

Current declines in many ES indicate that there is considerable scope to improve the design of response options to achieve balanced sustainable delivery of ES with enhanced societal benefits (Carpenter *et al.*, 2009; Everard *et al.*, 2014). These sustainability objectives appear particularly dependent on implementing Ecosystem Approach principles associated with scale and dynamics (Bastian *et al.*, 2012), including recognition that change is inevitable and that decision-making objectives are set for the longer term as well as meeting more immediate requirements (CBD, 2004; Figure 1).

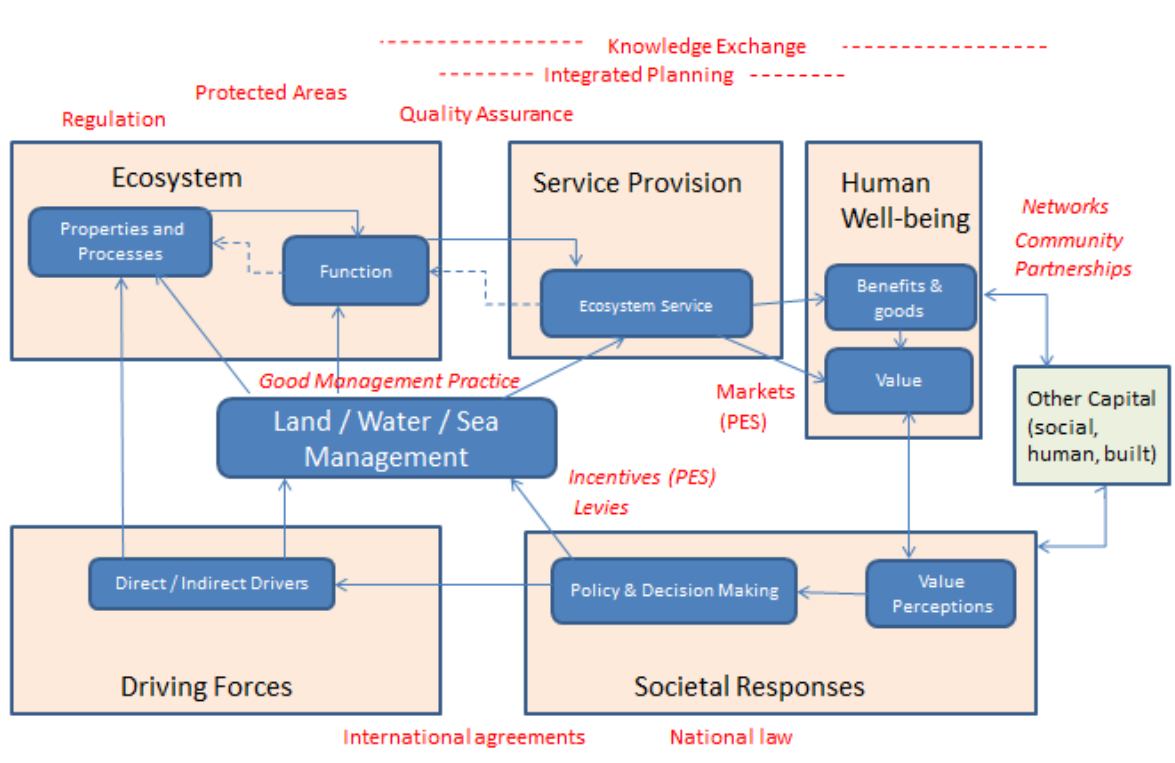
**Figure 1: The core principles of the Ecosystem Approach (CBD, 2004) summarised and grouped into 4 clusters**



For strategic environmental decision-making, integrated assessment (IA) frameworks have been developed to deal with multiple interactions across the whole cause-effect chain (Rotmans, 2006). Key requirements of IA can be identified as: (i) capacity to deal with complex issues that extend beyond an individual risk or impact assessment; (ii) evaluation of both positive and negative effects; (iii) provision of a synoptic and balanced assessment of impacts; and (iv) inclusion of a participatory process with key stakeholders. The most commonly used IA structure is the DPSIR framework (Drivers, Pressures, State, Impacts, Responses), in which the individual DPSIR components are linked to define a causal pathway that is used to frame responses to environmental change (e.g. Borja *et al.*, 2006; Binimelis *et al.*, 2009). However, DPSIR has been criticised for its association with a prescribed approach to managing environmental change that is sometimes accompanied by a preference for responses that do not address long-term challenges beyond the current reporting cycle (Tscherning *et al.*, 2012). Linear deterministic application of the DPSIR causal chain can therefore fail to recognise dynamic interactions between multiple drivers and pressures, including lagged responses, system feedback effects, natural variability, and cross-scale interactions in space and time (Rapport *et al.*, 1998).

As a consequence, naive application of DPSIR may encourage an over-emphasis on negative aspects of human-environment interactions with an associated tendency to adopt a narrow range of response options that reactively address impacts after environmental damage has occurred, as encapsulated by the 'polluter-pays' principle. This approach may overlook wider options for enhancement of ES provision, and hence improvements in net societal value and resilience, through focusing on restoration of ecosystem structure and functioning, as for example in the case of river restoration, retrofit of green infrastructure in urban areas, and wider restoration ecology approaches (Cortina *et al.*, 2006). Naïve application of DPSIR may also be counter to the principles of the Ecosystem Approach by inadvertently reinforcing a 'preservationist' perspective and not recognising other potentially legitimate stakeholder or scientific positions that suggest maintaining the status quo is unviable and that managed change would be preferable (Svarstad *et al.*, 2008).

Nevertheless, adoption of DPSIR can offer a major advantage for comparative evaluation by providing a common framework to reference response options. Recent contributions suggest the general DPSIR framework can be further strengthened by integration with ES concepts within the broader context of socio-ecological systems to recognise the role of multiple systems interactions and adaptive responses (e.g. Bennett *et al.*, 2009; Bastian *et al.*, 2012; van Oudenhoven *et al.*, 2012). These developments have been used in the present study to explicitly define a common systems-based framework to investigate and characterise the role of different response options (Figure 2). As discussed below, this can include response options that adopt a 'beneficiary pays' strategy to maintain ES as well as those that follow a more conventional 'polluter pays' strategy that penalises those who degrade ES. It may also help distinguish response options most clearly associated with maintaining natural ecosystem functions as compared to those following an anthropocentric service-based rationale.



**Figure 2. Conceptual systems framework for identifying the role of external drivers on ecosystems and human well-being (modified from van Oudenhoven et al., 2012). Response option types (in red) are identified against the components of the system they seek to influence. PES can occur in multiple contexts (section 3.4)**

*Could add Potential / Actual either side of 'Service Provision' box?*

## 2. Developing a Typology of Response Options

A key requirement for evaluating efficacy of response options is to recognise the type of intervention and its anticipated effect in influencing environment-related behaviours. The MA developed a generic classification of response options based on the nature of the intervention, with an emphasis on institutional aspects (Bradnee Chambers *et al.*, 2005). A simple typology of ROs, as used by the MA 'Manual for Practitioners' and UK National Ecosystem Assessment, has classified them as foundational, enabling or instrumental types (Simpson and Vira, 2010; Vira *et al.*, 2011). This typology assumes a sequence of linked steps by which knowledge ('foundational' level) creates the context for government to develop policy frameworks ('enabling' level) through which stakeholders implement specific actions ('instrumental' level). However, deviations from this simple top-down model occur in practice, as responses can also originate from local bottom-up processes, such as through community partnerships or related initiatives and particularly where traditional knowledge or informal institutions shape practice (Ostrom, 1990; Berkes and Folke, 2002; Helmke and Levitsky, 2004). Many good examples of community-based natural resource management have followed this latter pathway, in which place becomes the key context for developing responses rather than general policy objectives (Brooks *et al.*, 2012). The role of policy in this bottom-up context may therefore be reframed as providing the room (intentionally or otherwise) for such initiatives to flourish, the role of the state

then shifting from a model of 'state as regulator' to 'state as facilitator' in effective decentralised resource management (Robinson, 2000). Enabling flexible alignment of policies with local institutional practices or social norms, rather than imposing rigid 'one size fits all' top-down rules, can allow for adaptive and community-based resource management that is responsive to local geographical and cultural characteristics, or differentiated priorities (Ostrom, 2000; Lejano and Fernandez, 2014). A broader perspective on response options should therefore consider the role of both top-down and bottom-up initiatives, and hence the potential for cross-scale integration. This is argued to be more consistent with the Ecosystem Approach, including also principles associated with 'people and inclusion' (Figure 1) namely: the need for local management; integration of scientific with local and traditional knowledge; and the participation of multiple stakeholders in decision-making.

The proposed new typology of response options (Table 1) is summarised through a set of attributes that differentiates each type. The first of these attributes describes the approach to governance: this may be either top-down or bottom-up decision-making, or possibly a hybrid approach that can occur at multiple levels. Secondly, responses may be characterised by whether or not they have statutory legal underpinning; if this is present it usually involves tightly-defined mandatory requirements to ensure compliance is explicit. Depending on the jurisdiction, statutory law can have complex linkages with constitutional or common law. In each case, legal frameworks provide a strong mechanism for compliance but may be time-consuming to adjust for changing circumstances, and narrow definitions may cause conflicts between different objectives. Linked to this is the method of compliance, distinguishing between response options that impose penalties and taxes to ensure compliance from those which operate through voluntary persuasion or 'peer pressure'. Again penalty-based enforcement may be designed as an effective response, but concern over whether such measures are actually cost-effective has led to greater interest in voluntary agreements (Brink, 2002).

The relationship between the decision-maker and those they seek to influence may be of a closed, direct form (as for example in the imposition of regulatory rules) or it may involve intermediaries (e.g. brokers, or buyers and sellers) and therefore be more open and indirect. Open relationships may potentially introduce more efficiency and accountability for managing supply and demand for ES, but may also be less predictable in terms of final outcomes as well as having higher transaction costs (Pagiola and Platais, 2007). The scale attribute distinguishes between interventions which aim to define universal standards over a large area compared to those which can be spatially variable and may be adjusted to local contexts. Universality may be an advantage in defining acceptable minimum standards but may be challenging to define for complex environmental phenomena that have considerable inherent variability, either natural or human-induced (Wätzold and Drechsler, 2005). Finally, some response options are also associated with monetary values ascribed to goods and services, whereas others are not. With monetisation, as with other attributes listed, the rationale is to internalise environmental costs (e.g. pollution prevention) into the prices of goods and services (Stranlund, 1995), relevant to both closed and open approaches, but full integration within both is challenged by difficulties in monetising less tangible services that have no direct market value.

The response option typology recognises the critical importance of trust and power relations in the uptake of response options. The influence of the response options can be encouraged through comparison with penalties for non-compliance (condign power), or it can be bought (compensatory power), or gained by persuasion (conditional power) (Galbraith, 1983).

**Table 1.A generic typology of response options for managing environmental change**

<i><b>Response Options</b></i>	<i><b>Characteristics</b></i>	<i><b>Examples</b></i>
<i><b>Regulation (statutory)</b></i>	Legally-enforced universal minimum quality standards	Drinking water, Bathing water, Air quality, Food Safety, Fishing quotas
<i><b>Levies</b></i>	Taxes to support environmental standards or improvements	Aggregate Levy Fund (UK), Landfill Tax (UK), Climate Change Levy (UK)
<i><b>Protected areas</b></i>	Defined zones that have restrictions on their use or conservation-based obligations	Natura2000 sites (EU), Marine conservation zones
<i><b>Common law, Civil law or Constitutional law</b></i>	Legal rights and responsibilities based upon precedent (common law); general rules (civil law); or constitution	Conservation covenants, Environmental impact assessment
<i><b>Direct payments and Incentives</b></i>	Payments to support a particular use or management practice based upon service provided	Agri-environment schemes (EU), Payments for Ecosystem Services (PES), Biodiversity offsetting
<i><b>Market-based schemes</b></i>	Trading of goods and services on an open market	Carbon trading, Biodiversity offsetting
<i><b>Voluntary quality assurance</b></i>	Independent schemes that provide accreditation for maintaining minimum standards via a quality marque	Forest Stewardship Council, Marine Stewardship Council
<i><b>Spatial &amp; Integrated planning</b></i>	Combined cross-sectoral planning instruments to maximise resource efficiencies and opportunities	Green infrastructure, Integrated catchment planning, Integrated coastal zone management
<i><b>Investment in Science &amp; Technology</b></i>	Investment in new science and technology with associated infrastructure to improve uptake.	Precision farming, Ecosystem services, Renewable energy, Water treatment, Waste reduction, Recycling
<i><b>Education &amp; Knowledge exchange</b></i>	Formal and informal schemes to communicate and share knowledge	Campaigns, Professional development, Demonstration projects, Citizen science, Eco-schools
<i><b>Networks &amp; Partnerships</b></i>	Formal and informal arrangements of multiple stakeholders based upon a common shared interest	Community woodlands, Coastal partnerships, Rivers trusts, Biodiversity partnerships
<i><b>Good management practice</b></i>	Guidelines to share and encourage adoption of best practices	Integrated farm management, Natural flood management

<b>Response Options</b>	<i>Governance</i>	<i>Legal underpinning</i>	<i>Compliance with good practice</i>	<i>Relationships</i>	<i>Scale</i>	<i>Monetisation</i>
<b>Regulation (statutory)</b>	Top-down	Yes	Penalties	Closed	Universal	No
<b>Levies</b>	Top-down	Yes	Penalties	Closed	Universal	Yes
<b>Protected areas</b>	Top-down	Yes	Penalties	Closed	Variable	No
<b>Common law, Civil law or Constitutional law</b>	Top down	Yes	Penalties	Closed	Universal and Variable	No
<b>Direct payments and Incentives</b>	Top-down	Yes (but voluntary opt-in)	Loss of payment	Closed	Variable	Yes
<b>Market-based schemes</b>	Hybrid	No	Persuasion (added value)	Open	Variable	Yes
<b>Voluntary quality assurance</b>	Hybrid or bottom-up	No	Audit or Self-certification	Closed	Variable	No
<b>Spatial &amp; Integrated planning</b>	Hybrid	No	Incentives and persuasion (added value)	Open	Variable	No
<b>Investment in Science &amp; Technology</b>	Hybrid	No	Persuasion (added value)	Open/Closed	Variable	No
<b>Education &amp; Knowledge exchange</b>	Hybrid or bottom-up	No	Persuasion	Open	Variable	No
<b>Networks &amp; Partnerships</b>	Hybrid or bottom-up	No	Persuasion	Open	Variable	No
<b>Good management practice</b>	Bottom-up	No	Persuasion	Closed	Variable	No



### 3. Types of Response Option

Each type of response option, either well-established or a recent innovation, has developed for particular reasons. Following the rationale of Table 1, they are described in order from top-down (or 'command or control') interventions to others that are more varied or better described as bottom-up. With regard to delivery of ES, responses may have either direct or indirect applicability, and may be more relevant to some categories of ES more than others.

#### 3.1 Statutory Regulation

The use of statutory regulation stipulates mandatory minimum standards for environmental quality (Percival *et al.*, 2003). The universality of regulatory standards can also be seen as providing 'a level playing for all' in terms of requirements for compliance. Statutory regulation is most commonly applied to control human activities associated with regulating ES (e.g. soil/air/water quality), although standards are defined through biological, physical or chemical thresholds that are assumed from evidence to represent safe limits for selected ecosystem processes and functions. Regulation represents a well-established response often implemented within a DPSIR framework to meet requirements for regular monitoring and reporting on compliance by a regulating authority. Compliance and impact monitoring requirements mean that this response option type can be expensive, and may be ineffective in addressing complex issues such as diffuse pollution (Collins and McGonigle, 2008). Effective implementation can therefore be further challenged by cost-efficiency goals (Hollins and Meffe, 1996) and the confounding effects of external drivers of change, notably climate change, which interact with local factors and natural variability to produce heterogeneous variations that challenge the definition and implementation of universal standards. Furthermore, regulation is only effective in achieving its selected goals when it is enforced uniformly, which is often far from the case where resources are limited (a common situation in the developing world: Everard *et al.*, 2009) or vested interests are strong and sanctions rare (as in the case of lax observance of minimum cross-compliance standards in farming practice: House of Lords, 2008).

The portfolio of regulation comprises a spectrum from instruments with a narrow focus, for example, clauses that address a particular metric or outcome (such as reaching a target chemical standard), towards more progressive regulations adopting a whole-systems approach (such as the EU Water Framework Directive for which 'good ecological status' is a key requirement). As a consequence, inflexible compliance with individual clauses can blunt the intent of more systemically focused regulations. Conversely, reinterpretation and implementation of legacy regulations in the light of emerging knowledge and strategic policy goals can represent a significant 'enabling' modification that be used to deliver a more systemic approach (Everard, 2010; Everard and McInnes, 2013). Within the context of decision appraisals, Strategic Environment Assessment (SEA) provides a flexible regulatory tool to implement integrated assessment at programme level. In conjunction with spatial planning approaches (section 3.10), SEA may therefore provide a common protocol to integrate ES concepts in at multiple scales (Geneletti *et al.*, 2011).

#### 3.2 Levy schemes

A related statutory response is to design levy schemes that require licensees to contribute financially (depending on scale of operation) to a central fund that supports good practice, pays for remedial work and environmental improvements, or acts to mitigate damages (such as offsetting schemes,

planning agreements, or pollution taxes: Varma, 2003). The link with the ES framework is currently indirect as these schemes support restoration or rehabilitation that could ultimately benefit an array of ES, though these multiple benefits are usually poorly defined. Levies may also be used to send signals aimed at achieving widespread changes in business behaviour, as for example in the case of the Landfill Tax or Aggregates Levy in the UK. They can be designed to be revenue-neutral for government; for example, revenues can be hypothecated to support specific environmental objectives, such as investment in cleaner technologies (e.g. Clean Development Initiative) or to reward better performers. A disadvantage is that transaction costs may be significant, especially with requirements to demonstrate transparency and fairness (as taxes are generally unpopular). They may also produce distortions which actually have negative impacts for environmental quality because of complex interactions between different tax regimes (e.g. Metcalfe, 2003), and practical outcomes may be considerably lower than claimed for compensatory actions and benefits (Erwin, 1991).

### *3.3 Protected Areas*

A related statutory response is to define and designate protected areas which have high importance because of their biodiversity or landscape quality, or other features. Although not usually explicitly linked with delivery of the ES framework at present, protection can act to maintain a range of services, with biodiversity acting as a key indicator of potential delivery across different ES classes (Mace *et al.*, 2012; Paloma *et al.*, 2014). Some designations explicitly address geodiversity, which may also indicate likely production of a range of services. Protected status generally imposes restrictions on the use of areas of land or sea (either implicitly or explicitly) to meet specific conservation objectives. These restrictions may cause conflict with local people (Mora and Sale, 2011), although this may be addressed through payments or incentives to influence management practices consistent with provision of favoured ES outcomes (section 3.5). As with statutory regulation, this response type may represent a translation from international obligations into national law (e.g. Ramsar Convention; EU Natura 2000 areas), together with additional national and/or local designations. Although the network of protected areas should in principle be dynamic to adjust to ecological change (especially from climate change), in practice it has been criticised for being less flexible and slow to adapt to change; this is attributed to the rigid structures inherent in national or regional planning systems (Wilson and Piper, 2008). As a consequence, protected areas are likely to exist as fragmented islands in a wider landscape with significantly less protection, which may be too restrictive to allow genetic diversity or to enable species dispersion (Paloma *et al.*, 2014). A special focus protected areas may also inadvertently sanction loss of biodiversity and ES in the wider landscape (Lawton *et al.*, 2010).

### *3.4 Common, civil and constitutional law*

In addition to international agreements, statutes and regulations (including mandatory environmental impact assessment in some countries), environmental law is covered by common and customary laws that reflect social norms. However, as with statutory regulation, the legal framework has developed piecemeal and therefore rights associated with different types of ES can be inconsistently defined in law, especially with regard to less material benefits. The relationship between environmental law and the ES framework therefore exists presently only as an emerging concept (Ruhl *et al.*, 2007; Everard and Appleby, 2009).

### *3.5 Direct payments and incentives*

An alternative approach is to encourage voluntary uptake of subsidies and incentives. Direct payments to managers or resource users may be used to reward modified management practices covering a specific area of land, water or sea. Payments are conventionally defined as compensation for 'income foregone' (to satisfy World Trade Organisation rules) resulting from a prescribed change, and may be arbitrated centrally or subject to local negotiation; in the latter case, this may introduce more local adaptability and flexibility. This type of response options includes land use subsidies (e.g. EU Common Agricultural Policy), agri-environment schemes (e.g. Environmental Stewardship initiatives in the UK) and forestry grants, including incentives to take land out of production (set-aside). Although potentially applicable to all types of ES, they are most frequently linked with enhancement of regulating ES, notably for carbon storage and water purification, in addition to biodiversity-related benefits (Whittingham, 2011). A potential disadvantage of such responses is that they may lead to over-reliance of communities on direct payments which may undermine long-term moves towards local sustainability (Ferraro and Kiss, 2002; Swart et al. 2003).

Payment for Ecosystem Services (PES) can be included in this type of response option, although the term has been applied to a wide variety of different responses that cover a spectrum of market-based instruments (Pirard, 2012). The key attributes of PES are that they are voluntary schemes which apply monetary values to reward those who provide ES to defined beneficiaries, rather than using notions of 'income foregone' which overlooks the wider societal value of ES (Wunder, 2005). Privately-funded ('Coasean') arrangements made directly between service provider and beneficiary may be possible, typically when there is a relatively small number of beneficiaries; however, in practice the presence of complicating issues, such as transaction costs, free-rider effects and property rights, mean that third-party intermediaries are usually required ('Pigouvian' arrangements) (Engel *et al.*, 2008). Third parties may be government, wherein the relationship may become similar to top-down regulation because the paying agency does not have direct information on actual service delivery at the local level. Alternatively, third parties may be provided by independent brokers to maintain a distance from government policies and a closer link to monitoring service delivery. Schemes can be either directly related to the conditional delivery of specific ES outcomes (i.e. output-based), or more commonly, because of difficulties in monitoring ES, based upon management measures agreed to be likely to protect or enhance service delivery (i.e. input-based schemes: e.g. wetland restoration to enhance flood alleviation, sequester carbon, support biodiversity etc.). PES may be based upon single ES or the 'bundling' of multiple ES within the same agreement (Raudsepp-Hearne *et al.*, 2010). Although concerns have been raised regarding their permanence (Pagiola and Platais, 2007), flexibility of scheme design may provide efficiency compared to top-down regulation (Wünscher *et al.*, 2008), including adjustment to changing external conditions and the capacity for incorporating additional services in future (Smith *et al.*, 2013).

### *3.6 Market-based schemes*

More open buyer-seller type relationships may be developed through full market-based schemes with environmental benefits defined as tradable goods (Pirard, 2012). The intention of open market schemes is to create more efficient brokering of supply and demand for particular ES through their monetary value (Hepburn, 2006). At the simplest level, these include long-established markets for food, energy and water-based services, with recent extension to carbon-based markets. The rationale is that the use of markets as a 'regulator' is more efficient than through statutory regulation as it can act as a stimulus for innovation in ecological restoration and enhanced service provision rather than

just minimum compliance. One example is tradable emissions permits, as employed for carbon trading, which create an incentive for businesses to reduce or offset emissions and profit from the sale of permits (Aldy *et al.*, 2010). Similarly, offsetting schemes have been applied to biodiversity, based upon a rationale of 'no net loss', to compensate for the adverse impacts from land development (McKenney and Keisecker, 2010). Market-based solutions may also be more effective in influencing resource use and management decisions in situations where regulation is ineffective through lack of observance or inadequate enforcement (Everard *et al.*, 2009).

Counting against market-based schemes are previous market failures regarding provision of public goods; the introduction of new markets for a formerly overlooked service may result in similar kinds of externalities that occur with existing markets (Everard and McInnes, 2013). Furthermore, there are risks involved in market-based schemes because processes of ecological restoration (including their outcomes such as carbon sequestration) inevitably include elements of outcome uncertainty which are not factored into the market price. This uncertainty is likely to be further exacerbated by future ecosystem dynamics, as particularly affected by climate change.

### *3.7 Voluntary Quality Assurance*

Voluntary quality assurance initiatives provide another alternative to statutory approaches by emphasising the use of a certificated brand or marque to assure customers that quality standards are maintained (Boström, 2003; Auld *et al.*, 2008). Businesses subscribe to a general set of standards that are enforced by an audit of practice, or which are self-certifying. In return, they receive the reputational benefits of being associated with a brand which can also add value by securing potentially vulnerable supply chains of goods and services, contributing to staff morale and retention through greater coherence with employee values, and reinforcing shareholder confidence through better risk management (Everard, 2009). Standards are therefore designed to provide a demonstrable and auditable link between provisioning ES (notably for food or fibre production) and healthy ecosystem functioning, typically including measures to maintain soil, water or air quality. There may also be a shared benefit through association with a location or other 'brand' which has an association with quality assurance or provides a market premium. This association may also benefit from statutory protection (such as EU-registered Protected Domain of Origin designations).

A further use of voluntary standards, where transparently reported, is for self-certification of compliance with regulatory obligations, an approach favoured by the US Environment Protection Agency that is finding favour elsewhere, although there may be trade-offs between expected cost-efficiencies and effectiveness in terms of environmental benefits (Darnall and Sides, 2008).

### *3.8 Integrated/Spatial Planning*

Integrated or spatial planning combines multiple sectoral policies into a unified local or regional implementation plan that can enhance synergies between delivery of multiple ES based on common priorities associated with places and landscapes (Hurliman and March, 2012; Wilson and Piper, 2008, 2010). Through public participation, plans have the potential to be flexible, dynamic and forward looking tools, although flexibility may be constrained if current governance structures are fragmented (Scott *et al.*, 2013) or if decision-making is not inclusive or captured by strong vested interests (Johnston *et al.*, 2007).

Integrated planning is also manifest in cross-sectoral initiatives based upon strategic management units (e.g. river basins; littoral zones; green infrastructure). If existing barriers can be resolved, it offers the potential to design and manage targeted zones for the supply of multiple ES outcomes (e.g. Niemelä *et al.*, 2010), which, by harnessing natural processes, may also be low-input solutions that maximise public value across services (cf. 'systemic solutions': Everard and McInnes, 2013).

### *3.9 Investment in Science and Technology*

Targeted investment in science and technology can be used to address key gaps in 'foundational' knowledge and its application. As already highlighted, science for managing ES is rapidly advancing, though major knowledge gaps remain (Carpenter *et al.*, 2009). Technology can lead to rapid advances in the efficient use and management of natural resources, but can also cause significant negative impacts when used without awareness of environmental limits or when outcomes are not assessed in terms of the full suite of ES (Jaffe *et al.*, 2002). Rapidly-advancing technologies include the use of biotechnology (e.g. GMOs), nanotechnology, satellite-based remote sensing and automated monitoring. These are currently most strongly associated with provisioning ES, but also have clear potential to enhance regulating ES (e.g. precision farming). More broadly, ICT developments (e.g. social media) may facilitate improved awareness and knowledge exchange across all types of ES through networking and partnership-building.

### *3.10 Education and Knowledge Exchange*

Knowledge of environmental change can exist in both formal and informal contexts. An important principle of the Ecosystem Approach is to combine and exchange this knowledge as part of the decision-making process, as may be encouraged by investment in demonstration projects, citizen science initiatives or awareness-raising campaigns. Uptake and application of science and technology may also require investment in associated knowledge schemes, such as education, skills training, and professional development (e.g. Clark and Lowe, 1992).

### *3.11 Networks and Partnerships*

Formal or informal networks or partnerships are usually bottom-up volunteer-based initiatives (e.g. communities, trusts or co-operatives) stimulated to take action by local or wider awareness of the value of the environment (Davies, 2002). If the issue or area is high profile, the outreach of the network can be national or international (such as India's 'Save Ganga Movement') led by prominent NGOs (Bäckstrand, 2006). Conversely, a local community woodland, urban garden area or local nature reserve may be maintained by a few dedicated individuals. Some networks and partnerships have a role in connecting diverse stakeholder groups around a common interest which can harness a range of other response options (grants, subsidies, legislation, good management practices, etc.) to achieve strategic goals (Ostrom, 2010). These initiatives are particularly linked with cultural ES through common recognition of the less material and collective benefits from the natural environment at local level (Pleininger *et al.*, 2013).

### *3.12 Good management practice*

Development and wider adoption of practical solutions can be encouraged at local level by improved awareness, promotion and incentivisation of those practices that deliver multiple ES (e.g. Broadmeadow and Nisbet, 2004). Published good practice standards can also discourage practices known to be damaging when formulating compliance criteria (e.g. agricultural subsidies) or as a basis for supporting enforcement actions (e.g. justifying pollution control enforcement). In these two examples, as with other cases, guidance on good management practices may reinforce other response

option types (e.g. incentives; regulatory compliance). Management objectives may also require co-operation between land managers (e.g. in river basins) or marine resource users. Particular benefits may be gained from schemes that increase efficiency and reduce waste (e.g. by recycling), or those that build flexibility and adaptability, including 'softer' schemes as an alternative to hard engineered schemes, including natural flood management, coastal realignment, and sustainable drainage systems (e.g. Iacob *et al.*, 2014; Hanson *et al.*, 2010).

#### **4. Response Options as Systemic Interventions**

Response option types may now be referenced against the general socio-ecological systems framework to further develop the scope for complementary interventions or influences (Figure 2). Complementarity may occur through combined schemes to integrate responses at both similar and different positions in a system structure together with matching of the primary attributes used to define response option types (governance, legal underpinning, compliance, relationships, scale, and monetisation). Key distinctions occur between top-down as opposed to bottom-up approaches, and in terms of whether the focus is on natural ecosystem components (structure, processes and functions) or on issues that are more relevant to humans (services and benefits).

Top-down approaches represented by regulation and protected areas are primarily focussed on ecological or physio-chemical structures, processes and functioning of an ecosystem. Hence, regulatory standards define obligations to maintain environmental quality based upon indicative measures of ecosystem health that provide a link (often indirectly) with ecosystem processes (e.g. water chemistry). Protected areas target key locations to maintain ecosystem structure (particularly its biodiversity) and hence indirectly protect ecosystem functions and the potential to provide services (Bastian, 2013). For both regulation and protected areas, a top-down emphasis on measures to maintain natural ecosystem components means that the broader range of benefits resulting from the intervention are often not directly apparent to most people, though a focus on cultural benefits such as the presence of charismatic species can serve as an indicator for likely production of these broader services (as for example the inclusion of fish in quality requirements for river ecosystems that provide a range of often under-appreciated societal benefits).

Levies and incentive payments represent other top-down interventions that either penalise detrimental practices or reward good practice respectively. For these responses, there is a direct link with people (e.g. land managers) and therefore they can have a powerful role in signalling preferable behaviours. Top-down incentives may be associated with some ES through payments made to managers for providing assumed services (i.e. input-based measures: section 3.6) but the top-down approach is remote from beneficiaries. In addition, levies and incentives only indirectly link with protection of ecosystem processes or functions. The design and effectiveness of some incentive-based approaches, notably PES, have been criticised due to their weak scientific foundations including failure to account for ecosystem dynamics (Naeem *et al.*, 2015).

Top-down interventions therefore often have the disadvantage of lack of specificity or flexibility as required to manage the spatial and temporal variability of ES, and are remote from local provider-beneficiary relationships. They usually have strong legal support for compliance, but environmental law is still in the early stages of incorporating the collective principles of the Ecosystem Approach.

Nevertheless, some advantages can be gained from a more systemic integration of top-down interventions. The monitoring and enforcement required to ensure effective regulation may be paid for by levies or the use of incentives to reduce the need for expensive enforcement procedures, following the general rationale that compensatory and conditional (persuasive) instruments are more progressive than condign instruments such as penalties (Galbraith, 1983).

Some of the response types referenced on Figure 2 may be defined as more service-orientated interventions with a more direct link to beneficiaries. These include the use of targeted incentives, PES, and market-based schemes based upon monetary values. An emphasis on services may be particularly useful to communicate the added value of benefits from the natural environment in a familiar language to the general public (Mander, 2011). It may also act to raise investment to sustain service delivery, including market efficiencies. However, it may also be argued that too much emphasis on a utilitarian service role may obscure broader long-term benefits from healthy functioning ecosystems (Petersen *et al.*, 2009).

A systemic approach should therefore seek to integrate response options based on service delivery with those that maintain ecosystem processes and function. In this context, the role of service-related schemes such as PES is not as a replacement for more conventional and established approaches such as regulation but to act as complements. Hence, regulation or related statutory instruments provide a 'safety net' with minimum standards for ecosystem health but these can be enhanced through service-related schemes that define added value for stakeholder beneficiaries. In this case, a shift is made from a 'polluter pays' principle to a 'beneficiary pays' principle but with the backup of legal enforcement to ensure compliance if necessary. Protected areas may then be used to safeguard particularly important 'hotspot' zones for ES provision.

Science has an important role in facilitating this integration by further developing principles and guidelines from the ecosystem approach to be consistently applied across the chain of human-environment connections represented by Figure 2 (e.g. Naeem *et al.*, 2015). For example, scientific guidelines may be translated into the audit procedures used by quality assurance schemes to enhance function-service linkages (which through reputation-based branding may provide monetary and other benefits). At a local level, practical knowledge to maintain function-service relationships can also be codified as good management practice that combines scientific with informal knowledge systems. Knowledge exchange activities between key actors, such as land managers, regulatory agencies and planners, are increasingly associated with actions to increase awareness and positively change attitudes and behaviours, with peer-to-peer learning from trusted sources recognised as particularly important. Hence, partnerships and networks have a crucial knowledge exchange role in communicating together the shared benefits from natural, human and social capital, including through education and training, and access to science and technology. In this context, stakeholder analysis approaches can provide valuable tools to enhance participation and inclusive decision-making (Prager *et al.*, 2012).

Systemic interventions also need to maximise multiple benefits and the plural values of ES (including non-monetary values) across the different scales over which both process-function-service-benefit components and their inter-relationships occur. Spatial and integrated planning can therefore be a key enabler in linking other response options: for example, within the framework of national statutory requirements, river basin planning can provide a context for local schemes which have closer link with

ES and their beneficiaries in smaller sub-basin units. It is also important to recognise that effective responses also evolve over time and may morph into each other. This is particularly apparent at local level where context can strongly influence the shaping of combined responses, often associated with partnerships between different institutions and stakeholder groups (Brooks *et al.*, 2012).

An additional appraisal tool for delivering systemic interventions with both statutory support and flexibility to adjust to decision context is provided by SEA which, when framed in terms of ES benefits, can potentially provide a standard decision-making protocol for linking function-service relationships (Helming *et al.*, 2013).

A further issue to consider is that different types of response options may be more or less suited to different ES. Consequently, over-reliance on one type can inadvertently drive implicit or explicit trade-offs between ES outcomes, potentially undermining net societal value. Hence, response options that are based upon notional safe limits (e.g. regulation) or on monetary values (e.g. PES) which may be less suitable for maintaining less tangible benefits, as particularly exemplified by cultural ES. In PES schemes, the notional bundling of multiple ES either to the same buyer (under a precise definition), or more broadly bundling to multiple buyers (also known as 'layering') represents a potentially attractive solution (Bennett *et al.*, 2009; Raudsepp-Hearne *et al.*, 2010). However, operational issues for effective bundling are generally still at an early stage (Engel *et al.*, 2008; Asquith *et al.*, 2008; Wunder and Wertz-Kanounnikoff, 2009). Most commonly, a dominant market for an individual ES (an 'anchor service', *sensu* Schomers and Matzdorf, 2013) may also be used to produce 'piggy-back' co-benefits that are not directly paid for (e.g. payments for water quality may provide benefits for biodiversity and landscape amenity 'for free'), rather than parallel markets being developed for different ES. There is a risk therefore that bundling protocols may overlook the more immaterial benefits from cultural ES that are inherently more difficult to define in the landscape (Chan *et al.*, 2012a,b). Tackling this issue provides one of the major remaining challenges for implementation of an ES framework and may be addressed through a more integrated recognition of the sociocultural component of landscapes in integrated planning frameworks (Setten *et al.*, 2012; Albert *et al.*, 2014).

Attributes of different response options types will also affect how they handle changing risk and uncertainty through their influence on the adaptability of ESs against changing external conditions, including context-dependent relationships between ESs and their relationship to ecosystem function and biodiversity (Mace *et al.*, 2012; Harrison *et al.*, 2014). Those response options that are primarily based upon ecosystem processes and function may be more appropriate to manage the slower and less readily understood and valued variables that maintain resilience within a socio-ecological system (e.g. organic matter), whereas service-based response options may be better utilised to modify the key fast variables that operate over shorter time scales (e.g. food production) (Bennett *et al.*, 2009). This may also allow the characterisation of a framework of indicators that can be used to monitor and fine-tune different components of the system based upon scientific knowledge of ecosystem functioning and its most important features to sustain human well-being (Egoh *et al.*, 2012).



## 5. Conclusion

A rationale for a working typology of responses (policy and practice) is presented based upon their key attributes. Review of different response types, using refinement of DPSIR within a socio-ecological systems framework, shows that they operate at varying positions in the system. This means that they have different roles in securing the sustainable delivery of ES. These roles are evident in relation to their inclusion of scale (spatial and temporal), the influencing of ecosystem function-service relationships, and in the participatory role of actors and stakeholders.

Typing of response options highlights how individual responses are suited to addressing the diverse array of issues associated with environmental change, though also recognising that each option only address part of the system and therefore may produce unwanted trade-offs (inadvertent or deliberate) if used as isolated interventions. These trade-offs may be exacerbated as the profile of different risks change in the future, and uncertainty of outcomes is not factored into the decision appraisal.

As current responses are often shaped by legacy issues associated with institutional rules and constraints in which they operate (Young, 2002), there appears considerable scope to develop strategic responses comprising linked 'toolkits' of response options that can provide a more integrated, forward-looking, and adaptable approach to sustainable management recognising the challenges of long-term environmental change. This may be enabled by extension of integrated assessment concepts as guided by the Ecosystem Approach to design and evaluate complementary strategies.

The twelve principles of the Ecosystem Approach, as set out by the Convention on Biological Diversity (CBD, 2004, as summarised in Figure 1) establish a framework against which it is possible to test the strengths of response options, and to determine how they are best combined to achieve systemic interventions optimising benefits across the spectrum of ecosystem services and their beneficiaries. Our approach to developing a response option typology aids that process of tool selection, justification and integration to strengthen function-service relationships, incorporate scale issues (across space and time), and to engage relevant actors/stakeholders and management issues.

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