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A step-wise process of decision-making under uncertainty when implementing environmental policy

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ABSTRACT

An ecosystem approach forms the basis of many recent environmental policies. The underlying concept states that decision-makers must consider the environmental, social and economic costs and benefits in the course of deciding whether to implement a management action. Decision-making can be undermined by uncertainty. Here, we discuss potential sources of uncertainty and their effect on an ecosystem approach-driven environmental policy, the factors affecting the choice and potential for management actions to achieve their objectives, the challenges associated with setting realistic and achievable targets, and how we can prioritise management of detrimental activities. We also consider how human challenges such as the availability of infrastructure and political will and ways of measuring costs and benefits and Member State interactions could also undermine environmental management. Potential limitations along with areas where further effort may be required to support ecosystem-based management objectives are highlighted and the advantages of a structured step-wise interdisciplinary approach to ecosystem management is shown.

1. Introduction

There has been a proliferation of environmental management policies in Europe and worldwide, many of which specify an ecosystem approach (Hassan et al., 2005). Environmental managers are obliged to consider the impact of a management action – an action primarily designed to improve ecosystem health – on existing social and economic systems (Samways et al., 2010). Moving from the aspirational objectives of an environmental policy to the implementation of management actions to effect ecosystem change requires decisions to be

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made with input (often independently) from environmental, social, economy and governance stakeholders, who make considerable effort to provide best available evidence. However, there is often uncertainty surrounding the evidence (Regan et al., 2005), and with greater uncertainty, there is an increase in the number of possible outcomes (Tversky and Kahneman, 1992) making decision-making more difficult, especially when time is limited (Haynes, 2009). In support, several frameworks have been developed for formal decision-making (see Regan et al., 2005 and references therein), but there remains little appreciation of how uncertainty can affect decision-making or how to deal with it.

The scale of the challenge facing ecosystem approach policies is reflected by the limited examples of implementation (FAO, 2005) and an even fewer number of success stories (Tallis et al., 2010). Nevertheless, the belief in the underlying concepts and potential benefits of the ecosystem approach is such that despite this, stakeholders have not been dissuaded from attempting to develop novel concepts and frameworks to support the ecosystem approach objective (although this process has primarily been driven by the scientific community). To date, efforts have been numerous and varied, ranging from complex (e.g. ecological networks, Oberle and Schaal, 2011) to more simplified approaches (e.g. cluster analysis, Knights et al., 2013).

Assessment frameworks often lead to the identification of several possible management actions to reduce the risk of environmental degradation from human activities (Knights et al., 2013; Piet et al., in preparation). Possible actions are then assessed *a priori* to determine which action (or combination of actions) is most appropriate for the given objective and should be taken forward. The most appropriate action(s) is not necessarily the best for the environment, society or the economy. Rather, appropriateness is a trade-off between the environment, societal and economic factors (Samways et al., 2010) as determined by the costs and benefits associated with a given action. Appropriateness can be assessed using a variety of tools (e.g. Hussain et al., 2010), but often and despite best intentions, any uncertainty that surrounds the evidence underpinning the management action can moderate the evidence-based decision (e.g. Nickerson and Zenger, 2002) such that there is a potentially inferior outcome for that action, and in the long-term, could affect the level of support for future action(s) (Bradshaw and Borchers, 2000).

In this paper, we discuss some sources of uncertainty and their potential effect on decision-making that is undertaken prior to or during the implementation of environmental policies that require an ecosystem approach. We use the Marine Strategy Framework Directive (Directive 2008/56/EC, MSFD herein) as a case study example to give recent context and to illustrate how uncertainty could affect the choice of the management action(s) that will be implemented, although the arguments themselves are generic and can be applied to other policies.

2. The Marine Strategy Framework Directive: a brief history

The MSFD established a framework obliging European Union Member States (MSs) to take the necessary measures to achieve or maintain Good Environmental Status (GES) in the marine environment by 2020. MSs have to develop and implement strategies that: (a) protect and preserve the marine environment, and (b) prevent and reduce inputs in the marine environment. The MSFD introduced 11 qualitative descriptors of the marine environment and outlined an objective for each (COM, 2010). Each objective delivers either maintenance or an improvement in the state of an ecological component (also referred to as characteristics), and a sustainable level of pressure exerted on the ecosystem by human activities that is compatible with GES. Ecological components include features such as biodiversity, fish and shellfish, or seafloor integrity, whereas pressures include underwater noise, marine litter and chemical contamination (see Annex I of the MSFD).

The MSFD sets out a roadmap for MSs (Articles 9 and 10), whereby they have to: (1) undertake an initial assessment of a set of ecological components of their water body, (2) identify the human activities that are exerting pressures which impact those components, (3) establish a comprehensive set of environmental targets and indicators to act as a guide for progress towards GES of regional seas, which when devised, (4) should take into account existing legislation (national, community or international), and (5) be mutually compatible with the targets of other MSs in their region. This roadmap can be visualised in a step-wise manner (Fig. 1), and here, we consider the challenges faced at each step and identify ways in which those challenges could be addressed. First, we discuss the factors affecting the potential for a management action to achieve its objectives assuming it is implemented and appropriately supported. This includes the role of 'non-manageable' environmental change such as climate change and the evaluation of anthropogenic 'manageable' change. We then discuss how human barriers to the implementation of management actions including the cost and benefit of a particular (suite of) measures, the availability of infrastructure, political will or policy inaction, and the interaction required between stakeholders during implementation. Potential limitations are identified and areas where further effort may be required to support ecosystem-based management objectives highlighted.

3. Identifying threats and risks to ecosystems, target setting and appropriate indicators

The likelihood of an environmental objective being met will be dependent on the ability of management action(s) to mitigate the impacts of human activities, where these are primary drivers of ecosystem state (Halpern et al., 2008). However, not all drivers of ecosystem state change are manageable (Fig. 2), but are having marked effects on ecosystems (Firth and Hawkins, 2011; Harley et al., 2006). A key step towards achieving ecosystem objectives must therefore be differentiation and quantification of the contribution of manageable and non-manageable drivers to ecosystem state, however uncertainty in the contribution of individual driver(s) to effect ecosystem state change can limit our ability to identify what should be managed, and what the impact of management might be.

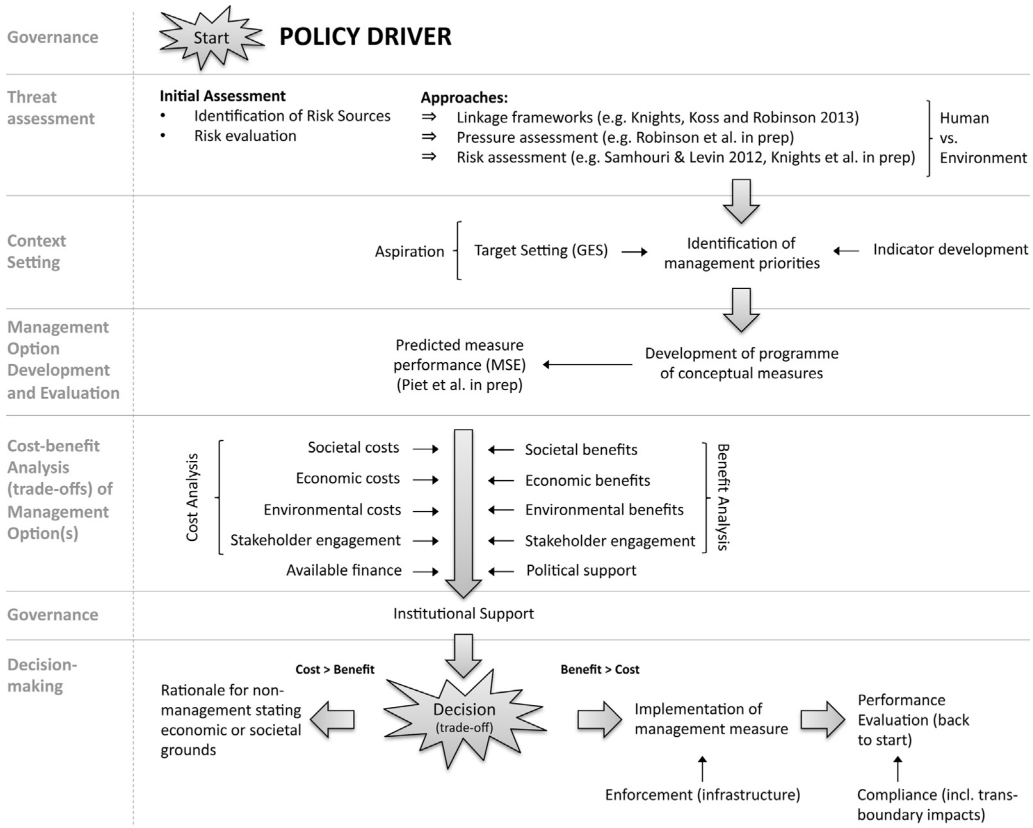

Fig. 1 – The steps undertaken by a decision-maker to determine whether a management measure should be adopted under an environmental policy. NB Some steps may have already been undertaken by managers (e.g. to fulfil the requirements of another policy driver) and therefore, decision-making may not need to 'start' where shown here. *Abbreviations*: GES, good environmental status; MSE, management strategy evaluation.

3.1. *Unmanageable environmental drivers of change*

Environmental factors can play an important role in determining the spatial and temporal distribution and abundance of ecological characteristics (Harley et al., 2006). Global climate change, in particular, is having profound implications on marine ecosystems such as shifts in the distribution and abundance of marine species (Parmesan and Yohe, 2003). Temperature is cited as a primary driver of these shifts, and the relationship between temperature and individual performance of species is often well described (Perry et al., 2005) and in several instances, species or habitats are in decline as a result of rising temperatures (Thuiller et al., 2008; Visser and Both, 2005). By contrast, other species are expected to benefit from climate change. For example, greater recruitment success of juvenile fish of some species may result in larger population sizes (e.g. Aprahamian et al., 2010). Where species benefit, this could directly lead to increases in stock size, and indirectly to greater sustainable levels of exploitation and increased seafood provision. In such cases, the objectives of an environmental policy may well be met without the need for management intervention.

The direction of effect that predicted changes in environmental conditions and/or human activities are likely to have on indicators must be determined so that management

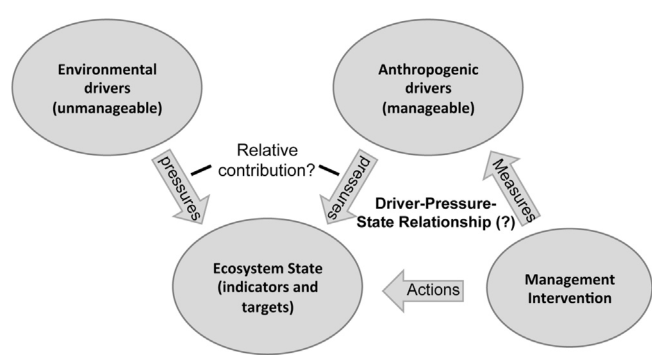

Fig. 2 – Drivers of ecosystem state and the potential for management intervention. Delineating the impact of environmental and anthropogenic drivers on ecosystem state is challenging. Linkage associations between anthropogenic drivers and ecosystem components are well described (e.g. Knights et al., 2013) and points where management interventions can be introduced are shown. However, the relative contribution and the driver-pressure-state relationship are often unknown limiting our ability to predict changes in ecosystem state following management intervention(s).

actions can be assessed in light of these changes; there is for instance a burgeoning literature on maladaptation to climate change (e.g. Firth et al., 2013). In cases where species or habitats demonstrate conflicting responses to environmental change (i.e. beneficial versus detrimental effects on the indicator value, Rosset and Oertli, 2011), it is difficult to aggregate the response of state within a single generic evaluation, e.g. the creation a single food web metric from multiple single stock datasets. The fact that environmental drivers and human activities may interact resulting in the exaggeration or masking of effects of one or both factors over temporal and/or spatial scales (Firth and Hawkins, 2011; Knights et al., 2012) further complicates the assessment. Ideally, projections of the effect of climate change on an ecological component should specify the full trajectory of the change for the ecological component in question (Rosset and Oertli, 2011), the magnitude of effect (how far, how fast), and how the response (e.g. mortality rate) varies among indicators. Such an analysis may then preclude, or make ineffectual, the use of a particular (suite of) management action(s) as climate effects under a 'do nothing' scenario might provide the benefits that the action itself was intended to stimulate.

3.2. *Manageable drivers of change: linking human activities to ecosystem state*

In addition to non-manageable environmental drivers, human activities continue to impact our oceans through direct and indirect means and affect large geographic areas (Halpern et al., 2008). Understanding the impact those activities have on marine ecosystems is needed so that trade-offs can be made between the continued exploitation of natural resources versus the protection of ecosystems and provision of goods and services (MEA, 2005). Linkage frameworks, such as Driver-Pressure-State-Impact-Response (DPSIR), are commonly used to describe the link between human activities and impact (e.g. Halpern et al., 2008; Knights et al., 2013; Oberle and Schaal, 2011). Linkage frameworks are reliant on accurate descriptions of linkages, and can be informed by qualitative, quantitative or expert judgement assessments or a combination of these. However, an inherent limitation of these frameworks is that they are constructed *a posteriori* ('after the fact') such that the effect of gaps in our knowledge are not explicitly considered (*sensu* 'natural uncertainty' after Walker et al., 2003). While there has been considerable work undertaken to further clarify these links (e.g. Knights et al., 2013 in conjunction with Koss et al., 2011 have produced perhaps the most comprehensive framework to date), if links are missing or those present are described in insufficient detail, the contribution of a human activity may be inappropriately estimated or valued. In such cases, a prospective management action may be insufficiently severe to achieve the management objective, or worse still, the threat from that activity is missed entirely (Khalilian et al., 2010).

3.3. *Determining if a management action is needed: identifying threats to ecosystems*

Under the MSFD, MSs are legally obligated to implement management action(s) where risk to a high-level objective (in this case a GES descriptor, but equally could be a specific ecosystem component) is identified. Risk is defined as the likelihood and the consequences of an event (Hope, 2006). Potential sources of risk can be identified using, initially, a combination of tools such as linkage frameworks (Knights et al., 2013) and pressure assessments (Robinson et al., in preparation) to describe threat, which can then be translated into risk (e.g. Samhouri and Levin, 2012; Smith et al., 2007) (Fig. 1). The identification of risk sources is a first step in managing the impact of human activities; the premise being that a reduction in risk by management should result in an improvement in ecosystem state, noting that this assumes the underlying assessment has encapsulated all possible threats and these can be addressed (see Sections 3.1 and 3.2). However, the need to make trade-offs between environmental, economic and societal objectives makes it unlikely that management will attempt or succeed in eliminating all risk sources. Instead, any reduction in risk is more likely to be targeted towards risk sources that lead to consequences that are most acceptable to stakeholders.

Decision-makers prefer targeted questions (Wilson et al., 2007) such as, how much change is required to lower the risk significantly? The links between some management actions and the major drivers of change are sufficiently clear, such that realistic expectations of the performance of a management action can be made and do not require a quantified outcome; a qualified statement may suffice. For example, a reduction in the number and extent of activities that introduce underwater noise would lead to an immediate reduction in noise and would clearly satisfy an objective of noise reduction. However, in some cases risk cannot be easily translated into a description of ecosystem state. Rather, a quantified outcome is needed requiring an understanding of the pressure-state (cause–effect) relationship. Working examples of pressure-state relationships are rare in natural systems and are often undermined by the multiple interactions between different pressures and the ecosystem (e.g. Firbank et al., 2003; Knights et al., 2012), making it difficult to forecast the performance of management action(s) if those interactions are unknown or inappropriately described. For example, it is common that pressures are introduced by several industries and overlap in time or space (Stelzenmüller et al., 2010), such that efforts targeted towards the management of the detrimental effects of a single industry may be undermined by the unmanaged pressures of other unregulated industries (Smith et al., 2007). Therefore, it is unlikely that any one management action (or set of related actions) will control all drivers that influence ecosystem state. Rather, it is more likely that a suite of management actions (i.e. a strategy) will be required to control the threats of multiple industries and activities (Knights et al., 2013) to improve ecosystem state. Recent efforts have focused on predicting the performance of management strategies in mitigating such combined effects (e.g. Goodsir et al., in preparation; Stelzenmüller et al., 2010).

3.4. *Target setting for ecosystem state indicators*

Uncertainty in the performance of an action (i.e. the resulting ecosystem state post-management) presents challenges to decision-makers in setting environmental targets. In the case of the MSFD, each MS must set targets for specific and measurable indicators of each descriptor by 2020 (COM, 2010). Uncertainty in the state of an indicator following implementation of an action can lead to uncertainty of the environmental, social and economic costs and benefits (Fig. 1). This could affect the level of support for an action (Bradshaw and Borchers, 2000). Targets must therefore be realistic and achievable (Carwardine et al., 2009). Ideally, long-term data sets and historical data (e.g. Hawkins et al., 2013) describing the trajectory of an indicator should be used, but such data do not guarantee management success. There is often uncertainty of the future (forecasted) state, perhaps due to the spatial and temporal variability in the state of a biological indicator i.e. the indicator displays ''natural variation''. Historically, ''natural variation'' has enabled resource managers to establish broad management goals (i.e. not targeted towards a specific threat) to protect wildlife and other natural resources (Landres et al., 1999), but the shift towards an ecosystem approach to management emphasises that trade-offs need to be made between different choices and stakeholders' priorities (Röckman et al., submitted for publication), and thus necessitates management action(s) to be targeted towards specific threats.

Describing the natural variation in the state of an indicator plays an important role in the development of indicator targets and appropriate management action(s), but this variation may not have been considered in the development of the environmental policy objective(s) and its respective indicators. Uncertainty in indicator estimates can limit the ability to set achievable targets for an indicator or give an imprecise estimate of the indicator state. When an indicator is more variable, predicting its state in any given year is less certain and as the range of 'natural' values increases, our ability to detect change following implementation of a management action decreases (the 'effect size' *sensu* Underwood, 1997) (Fig. 3). More severe actions may be needed for more variable indicators, in order to move the state of that indicator outside the distribution of expected values such that improvement is 'seen', but these may be less socially or economically acceptable and may lead to higher enforcement costs and reduced compliance (see Section 2). Describing the natural variation in the state of an indicator prior to target setting should support the development of action(s) that will most likely move the indicator state beyond the expected range of values. If the variability is appropriately described, then the likelihood of the action appearing effective will increase and thus, minimise the risk that confidence will be lost in the action by relevant stakeholders (e.g. a 'miss' as defined by Rice, 2003).

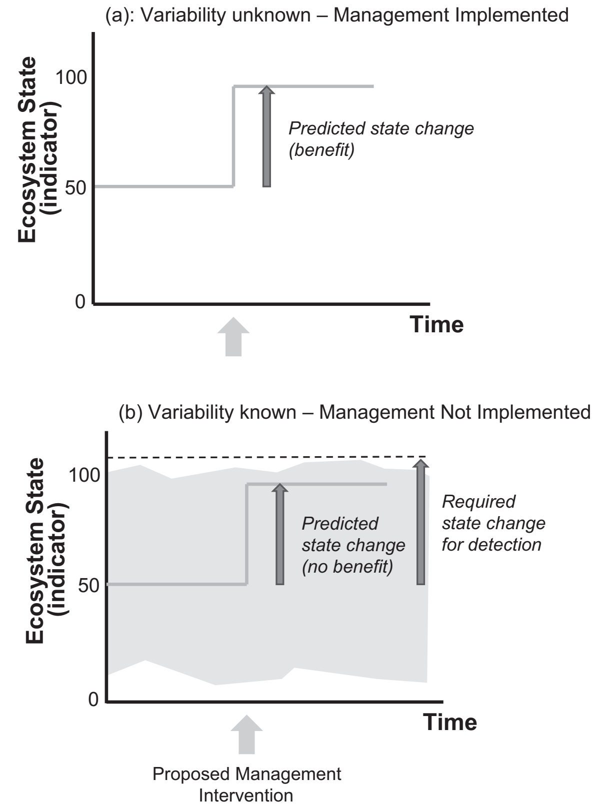

Fig. 3 – Schematic illustrating the effect of our ability to detect change in an indicator state and consequences for decision-making to implement or not implement a management intervention. (a) The variability of the indicator is unknown and management is implemented (light grey arrow) on the basis of the predicted increase in ecosystem state (+50 following the intervention), and (b) the variability is known (upper and lower confidence limits bounding shown by grey shading) but the proposed management intervention does not move the indicator outside of 'normal' expected values such that the management measure will appear ineffective and is therefore not implemented. A more severe management measure would be required to move the indicator outside normal values and thus indicate an improvement. A more severe measure may be less socially or economically acceptable. NB (b) assumes a working understanding of the driver-pressure-state relationship.

4. Society, economics and governance

The number of human activities that can be/are detrimental to an ecosystem are vast (Knights et al., 2013), but the resources available to MS are finite such that only a proportion of these are likely to be managed. Deciding which action(s) to implement necessitates a trade-off to be made between overall ecosystem health and associated long-term economic benefits (measured in terms of enhanced ecosystem service (ES) provisioning) on the one hand, and the costs of implementing a measure and any detrimental impact of the measure on the other. Evaluating the contribution of each activity to ecosystem state should initially help to prioritise (rank) the choice of management actions. If the results are then juxtaposed with the economic and societal implications of those actions, then a transparent and defendable decision-making process is achieved and support for management action justified (Fig. 1).

4.1. *Evaluating the costs and benefits of a management action*

Actions can be evaluated, firstly by determining (any) economic benefits gained based on a projection of how the supply of ESs might improve following management intervention (e.g. Fig. 3). A comparison of these benefits with the expected cost of implementation and compliance will also be necessary (Hussain et al., 2010). Where the cost of implementation exceeds the expected ES benefits and assuming decisions are informed by the economic appraisal, it is unlikely that the action will be considered viable in an economic sense, but may still have political and/or social support (Baral and Guha, 2004) (Fig. 1). Many ESs do not have a direct market value (these are referred to as a non-marketed ESs) or even a proxy, but changes in the provisioning of non-marketed ESs can affect human welfare and thus constitute an important element of economic decision-making with methods available to value some of these (Hussain et al., 2010).

The financial implications of introducing management interventions are wide-ranging and effects may be both positive and negative at different times of the implementation cycle. For example, an improvement in the condition of an ecosystem component to a sustainable level (e.g. GES), should in theory improve ecosystem resilience although evidence of such improvements following implementation remain inconclusive (Ives and Carpenter, 2007). Nevertheless, in the long-term and assuming that there is no erosion of the per unit benefit (e.g. the sale price of fish), such an increase could lead to greater economic productivity in the form of annual turnover.

Actions that support fish stock recovery can be used to illustrate the need to consider management from an interdisciplinary perspective. The tangible provisioning ES of 'food' (MEA, 2005), and collapse of the north Atlantic cod stocks can be used as an example. To maintain economic returns, fishermen began targeting alternative species to cod (Gowdy et al., 2010). While no significant decline in net financial returns from fishing effort were experienced (Hamilton, 2007), fewer fishermen were supported by the industry. This altered human migration patterns, population distributions and demographic structure, and undermined social cohesion (Hamilton and Haedrich, 1999). The intended effect of management was displaced from its original purpose of protection of the ES 'food', to instead having a disproportionate negative effect on cultural ESs associated with community cohesion. This example highlights the links between ecological (fishing down the food chain), economic (capital costs and foregone revenues) and social (community cohesion) components and indicates that treating any one of these in isolation would lead to a false characterisation of risk in the integrated system.

4.2. *Management infrastructure: implementation, compliance and enforcement*

Where management action(s) are deemed necessary, infrastructure is required to implement and enforce the action, otherwise the regulatory objective is unlikely to be met (Heyes, 2000). Infrastructure availability can vary markedly among national and international stakeholders depending on factors such as the political will to implement the necessary controls (e.g. top-down control by governing bodies), the availability of resources (both financial and human), and the prioritisation of an action over other obligations.

MSs may not have the capacity to implement and enforce a management action when in fact it is required and there may be reluctance to invest in environmental policy and ecosystem management, especially when the perceived costs outweigh the benefits (Fig. 1). A failure to implement management action(s) could have major consequences beyond not meeting the high-level objectives of an environmental policy. Persistent and continued environmental degradation could lead to cascading detrimental effects to the economy and society such as industry closures, unemployment or loss of cultural services. In Europe, failure to implement an EU Directive, such as the MSFD, might result in significant financial penalties being imposed on a MS (Article 258 of the Treaty on the Functioning of the European Union (TFEU)), but there is no guarantee that the necessary infrastructure will be put in place to support the implementation and enforcement of a management action, such that the risk of continued ecosystem degradation will remain (Smith et al., 2007).

4.3. *Institutional support and multi-national collaboration*

At the time of writing, there is still uncertainty as to how GES descriptors of the MSFD should be interpreted, which may lead to difficulties in assessing the support for action(s) prior to their environmental, economic or societal evaluation (unclear governance, Fig. 1 start). A varied interpretation of descriptors will further complicate this. For example, while our knowledge of human-induced pressures is relatively advanced and there is broad agreement across Europe on appropriate discharge thresholds for nutrients or certain contaminants (OSPAR, 2009), no such understanding is available for descriptors of relatively new pressures such as noise and marine litter. Moreover due to the complexity of the ecosystem such thresholds are almost entirely lacking for just about any aspect of state (e.g. biodiversity, foodweb functioning) for which the environmental policies have stated high-level objectives. This uncertainty, coupled with difficulties in measuring political will prior to management actions being suggested, might undermine implementation of a specific action or actions.

The ambition level of environmental targets is increasingly linked with the economic and societal implications of 'sustainability'. For example, the European Union's Horizon 2020 Strategy and Marine and Maritime Agenda for growth and jobs (''the Limassol Declaration'') have explicitly moved the focus from environmental targets towards a more economic focus of employment and growth (Freire-Gibb et al., 2014). Predicting changes in the economic and societal value of a resource (e.g. Smith, 1993) following the implementation of management action(s) may act as an effective proxy for predicting political support for a measure (i.e. evaluating costs and benefits). However, this is reliant on the impact of the action on ESs being visible to decision makers (i.e. they are marketed), rather than invisible, in which case, the impact of an environmental management programme on the provision of ESs is ignored or is unknown to the decision maker. The level of confidence (uncertainty) in the state of the ecosystem following measure implementation could also act as a similar proxy. A worst-case example might be that when uncertainty is high, the likelihood of implementation is predicted to be low and vice versa (Bradshaw and Borchers, 2000) although more likely, the confidence with which targets are set will be lower and the resultant state will be unknown.

The ability of any individual MS to meet its environmental objectives may be affected by the level of collaboration between MSs. Setting GES or indicator targets at the national level poses a significant challenge to the MSFD and its success, especially where a resource is shared among two or more MSs or straddles international boundaries. If GES targets are less stringent in one MS's waters than in its neighbours, then the costs incurred by the more stringent MS (e.g. a pressure reduction such as a spatial restriction imposed on an industry) may be undermined by the continued exploitation of the resource by the other MS. The efforts of a MS may be further undermined if the industry that targets the shared resource is of particular significance to the economy or society of another MS or when shared resources are distributed unevenly between territorial waters, leading to a mismatch between the beneficiaries of the measure and those that incur the cost.

In several cases, regional bodies such as OSPAR, HELCOM and Regional Advisory Councils (RACs) have already coordinated regional efforts for monitoring and could play an important role in facilitating MS interactions including negotiations on targets and management measures and in providing a regional perspective of the issues. The MSFD, however, does not provide any specific legal framework nor specifies governing structures to ensure cooperation and coordination and calls for new modes of governance (e.g. Raakær et al., in preparation). In order to achieve effective regionalisation, coherent, repeatable and transparent approaches for assessing the level of pressure from (overlapping) human impacts and the risks to the ecosystem at a regional sea scale are required. Without this, national perspectives will be based on subjective opinion rather than through objective structured assessments.

5. Conclusions

We have highlighted several of the challenges to the success of an ecosystem approach-driven environmental policy and have outlined a step-wise approach to aid decision-makers in making trade-offs. There are a variety of tools available that aid decision-makers at each stage of the process, whether supporting identifying threats to marine ecosystems from human activities (e.g. Knights et al., 2013; Koss et al., 2011) or estimating the costs and benefits of management actions (e.g. Hussain et al., 2010). The outcomes of such a step-wise approach can provide a transparent and defensible evidence base for a specific decision, but the outcome of each step must be used to inform the next step in the process, without which, a satisfactory trade-off between ecological, societal and economic objectives is unlikely to be achieved and the overarching objective of the environmental policy not met.

Successful implementation of an environmental policy is reliant upon the objectives of the policy being clearly defined with realistic and pragmatic targets. However, clarity in the objectives does not necessarily mean that all of the objectives can or will be met. This may be the case for several reasons. Most simply, environmental drivers may be the cause of state change and by definition, are unmanageable. Alternatively, the target state set by the policy may be too ambitious, in that they are ecologically unattainable, or the action(s) required to achieve that target may be too costly to be socially or economically acceptable. In the latter case, a trade-off could be made by lowering the target (and by definition requiring a less severe management action), but any cost reduction will be at the expense of ecosystem state and ES provision benefits. The choice of indicator and the variability of that indicator may also affect our ability to detect an improvement in state and our choice of management strategy. If an indicator is highly variable, then the cost of a management programme that achieves a discernable outcome may be great and outweigh any benefit, such that it is unacceptable to stakeholders. In such cases, no trade-off in benefits can be made as, if the measure were less severe, benefits would be undetectable (Underwood, 1997) and thus the measure would appear ineffective. Implementing a measure would therefore serve no purpose beyond managers appearing to be taking action in support of a particular goal; an approach that could back-fire in the longer term as no evidence of improvement may in time lead to an erosion of political, societal or economic support (Davies et al., 2010).

There is inherent uncertainty with each step of the decision-making process, some of which is known (i.e. known-unknowns), yet decision-makers must continue to make management decisions on the basis of this 'uncertain' evidence, whereby the costs and benefits are weighed up with a view to meet, or at least progress towards, the objectives of the environmental policy. Measurement of uncertainty and estimations of the cost and benefit of management action plays a valuable role in supporting decision-making (Walker et al., 2003), especially given the high financial and human resources cost of implementing an ecosystem approach-based environmental policy (Smith et al., 2007). The process we have outlined provides a structured framework for developing an evidence base for decision-making, which starts by making clear and explicit links between human activities and their impact on the environment based in the policy objective. This is a fundamental precursor to an evaluation of the environmental, societal and economic costs and benefits of management actions, which in turn, is followed by an assessment of institutional support (Raakær et al., in preparation). Only once all steps are complete can transparent and evidence-based decisions be made.

There are numerous pathways to an environmental policy objective, in terms of the type of management action implemented (Piet et al., in preparation), the severity of the actions, and the impact that the actions have on ESs (Hussain et al., 2010). Outlining the available options and an assessment of the costs and benefits of each from the outset will allow actions to be compared and contrasted in a transparent and defensible manner. Only once this is done can appropriate governance structures be formed to deal with uncertainty and to make the necessary trade-offs (Raakær et al., in preparation; Tattenhove et al., in preparation). Our framework uses an interdisciplinary approach to ecosystem-based management that draws on a wide range of expertise including ecology, social science, economics and governance that operate in collaboration, rather than mutually exclusively. Such an approach is required if the environmental, societal and economic objectives of ecosystem-approach environmental policies are to be realised.

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