

1 **Title:** A step-wise process of decision-making under uncertainty when implementing environmental
2 policy

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20
21 **Abstract**

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

34
35 **Keywords:** management; ecosystem approach; measures; indicators; socio-economics; governance

36
37 **Introduction**

38 There has been a proliferation of environmental management policies in Europe and worldwide, many
39 of which specify an ecosystem approach (Hassan et al., 2005). Environmental managers are obliged
40 to consider the impact of a management action – an action primarily designed to improve ecosystem
41 health - on existing social and economic systems (Samways et al., 2010). Moving from the aspirational

42 objectives of an environmental policy to the implementation of management actions to effect
43 ecosystem change requires decisions to be made with input (often independently) from environmental,
44 social, economy and governance stakeholders, who make considerable effort to provide best available
45 evidence. However, there is often uncertainty surrounding the evidence (Regan et al., 2005), and with
46 greater uncertainty, there is an increase in the number of possible outcomes (Tversky and Kahneman,
47 1992) making decision-making more difficult, especially when time is limited (Haynes, 2009). In
48 support, several frameworks have been developed for formal decision-making (see Regan et al., 2005
49 and references therein), but there remains little appreciation of how uncertainty can affect decision-
50 making or how to deal with it.

51

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

60

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

73

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

80

81 **The Marine Strategy Framework Directive: A Brief History**

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

92

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

109

110 **1. Identifying threats and risks to ecosystems, target setting and appropriate indicators**

111 The likelihood of an environmental objective being met will be dependent on the ability of management
112 action(s) to mitigate the impacts of human activities, where these are primary drivers of ecosystem
113 state (Halpern et al., 2008). However, not all drivers of ecosystem state change are manageable
114 (Figure 2), but are having marked effects on ecosystems (Firth and Hawkins, 2011; Harley et al.,
115 2006). A key step toward achieving ecosystem objectives must therefore be differentiation and
116 quantification of the contribution of manageable and non-manageable drivers to ecosystem state,
117 however uncertainty in the contribution of individual driver(s) to effect ecosystem state change can
118 limit our ability to identify what should be managed, and what the impact of management might be.

119

120 *1.1 Unmanageable environmental drivers of change*

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

133

134 The direction of effect that predicted changes in environmental conditions and/or human activities are
135 likely to have on indicators must be determined so that management actions can be assessed in light
136 of these changes; there is for instance a burgeoning literature on maladaptation to climate change (e.g.
137 Firth et al., 2013). In cases where species or habitats demonstrate conflicting responses to
138 environmental change (i.e. beneficial vs. detrimental effects on the indicator value, Rosset and Oertli,
139 2011), it is difficult to aggregate the response of state within a single generic evaluation, e.g. the
140 creation a single food web metric from multiple single stock datasets. The fact that environmental
141 drivers and human activities may interact resulting in the exaggeration or masking of effects of one or
142 both factors over temporal and/or spatial scales (Firth and Hawkins, 2011; Knights et al., 2012) further
143 complicates the assessment. Ideally, projections of the effect of climate change on an ecological
144 component should specify the full trajectory of the change for the ecological component in question
145 (Rosset and Oertli, 2011), the magnitude of effect (how far, how fast), and how the response (e.g.
146 mortality rate) varies among indicators. Such an analysis may then preclude, or make ineffectual, the
147 use of a particular (suite of) management action(s) as climate effects under a 'do nothing' scenario
148 might provide the benefits that the action itself was intended to stimulate.

149

150 *1.2 Manageable drivers of change: Linking human activities to ecosystem state*

151 In addition to non-manageable environmental drivers, human activities continue to impact our oceans
152 through direct and indirect means and affect large geographic areas (Halpern et al., 2008).

153 Understanding the impact those activities have on marine ecosystems is needed so that trade-offs can
154 be made between the continued exploitation of natural resources versus the protection of ecosystems
155 and provision of goods and services (MEA, 2005). Linkage frameworks, such as Driving force-
156 Pressure-State-Impact-Response (DPSIR), are commonly used to describe the link between human
157 activities and impact (e.g. Halpern et al., 2008; Knights et al., 2013; Oberle and Schaal, 2011).

158 Linkage frameworks are reliant on accurate descriptions of linkages, and can be informed by
159 qualitative, quantitative or expert judgment assessments or a combination of these. However, an
160 inherent limitation of these frameworks is that they are constructed *a posteriori* ('after the fact') such

161 that the effect of gaps in our knowledge are not explicitly considered (*sensu* 'natural uncertainty' after
162 Walker et al., 2003). While there has been considerable work undertaken to further clarify these links
163 (e.g. Knights et al., 2013 in conjunction with Koss et al., 2011 have produced perhaps the most
164 comprehensive framework to date), if links are missing or those present are described in insufficient
165 detail, the contribution of a human activity may be inappropriately estimated or valued. In such cases,
166 a prospective management action may be insufficiently severe to achieve the management objective,
167 or worse still, the threat from that activity is missed entirely (Khalilian et al., 2010).

168

169 *1.3 Determining if a Management Action is needed: Identifying threats to ecosystems*

170 Under the MSFD, MSs are legally obligated to implement management action(s) where risk to a high-
171 level objective (in this case a GES descriptor, but equally could be a specific ecosystem component) is
172 identified. Risk is defined as the likelihood and the consequences of an event (Hope, 2006). Potential
173 sources of risk can be identified using, initially, a combination of tools such as linkage frameworks
174 (Knights et al., 2013) and pressure assessments (Robinson et al., In prep) to describe threat, which
175 can then be translated into risk (e.g. Samhoury and Levin, 2012; Smith et al., 2007)(Figure 1). The
176 identification of risk sources is a first step in managing the impact of human activities; the premise
177 being that a reduction in risk by management should result in an improvement in ecosystem state,
178 noting that this assumes the underlying assessment has encapsulated all possible threats and these
179 can be addressed (see 1.1 and 1.2 above). However, the need to make trade-offs between
180 environmental, economic and societal objectives makes it unlikely that management will attempt or
181 succeed in eliminating all risk sources. Instead, any reduction in risk is more likely to be targeted
182 toward risk sources that lead to consequences that are most acceptable to stakeholders.

183

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

201 that a suite of management actions (i.e. a strategy) will be required to control the threats of multiple
202 industries and activities (Knights et al., 2013) to improve ecosystem state. Recent efforts have focused
203 on predicting the performance of management strategies in mitigating such combined effects (e.g.
204 Goodsir et al., In prep; Stelzenmüller et al., 2010).

205

206 1.4. Target setting for ecosystem state indicators

207 Uncertainty in the performance of an action (i.e. the resulting ecosystem state post-management)
208 presents challenges to decision-makers in setting environmental targets. In the case of the MSFD,
209 each MS must set targets for specific and measurable indicators of each descriptor by 2020 (COM,
210 2010). Uncertainty in the state of an indicator following implementation of an action can lead to
211 uncertainty of the environmental, social and economic costs and benefits (Figure 1). This could affect
212 the level of support for an action (Bradshaw and Borchers, 2000). Targets must therefore be realistic
213 and achievable (Carwardine et al., 2009). Ideally, long-term data sets and historical data (e.g. Hawkins
214 et al., 2013) describing the trajectory of an indicator should be used, but such data do not guarantee
215 management success. There is often uncertainty of the future (forecasted) state, perhaps due to the
216 spatial and temporal variability in the state of a biological indicator i.e. the indicator displays “natural
217 variation”. Historically, “natural variation” has enabled resource managers to establish broad
218 management goals (i.e. not targeted toward a specific threat) to protect wildlife and other natural
219 resources (Landres et al., 1999), but the shift toward an ecosystem approach to management
220 emphasises that trade-offs need to be made between different choices and stakeholders’ priorities
221 (Röckman et al., Submitted), and thus necessitates management action(s) to be targeted toward
222 specific threats.

223

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

240

241 **2. Society, Economics and Governance**

242 The number of human activities that can be/detrimental to an ecosystem are vast (Knights et al.,
243 2013), but the resources available to MS are finite such that only a proportion of these are likely to be
244 managed. Deciding which action(s) to implement necessitates a trade-off to be made between overall
245 ecosystem health and associated long-term economic benefits (measured in terms of enhanced
246 ecosystem service (ES) provisioning) on the one hand, and the costs of implementing a measure and
247 any detrimental impact of the measure on the other. Evaluating the contribution of each activity to
248 ecosystem state should initially help to prioritise (rank) the choice of management actions. If the
249 results are then juxtaposed with the economic and societal implications of those actions, then a
250 transparent and defensible decision-making process is achieved and support for management action
251 justified (Figure 1).

252

253 *2.1. Evaluating the Costs and Benefits of a Management Action*

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

264

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

272

273 Actions that support fish stock recovery can be used to illustrate the need to consider management
274 from an interdisciplinary perspective. The tangible provisioning ES of 'food' (MEA, 2005), and collapse
275 of the north Atlantic cod stocks can be used as an example. To maintain economic returns, fishermen
276 began targeting alternative species to cod (Gowdy et al., 2010). While no significant decline in net
277 financial returns from fishing effort were experienced (Hamilton, 2007), fewer fishermen were
278 supported by the industry. This altered human migration patterns, population distributions and
279 demographic structure, and undermined social cohesion (Hamilton and Haedrich, 1999). The intended
280 effect of management was displaced from its original purpose of protection of the ES 'food', to instead

281 having a disproportionate negative effect on cultural ESs associated with community cohesion. This
282 example highlights the links between ecological (fishing down the food chain), economic (capital costs
283 and foregone revenues) and social (community cohesion) components and indicates that treating any
284 one of these in isolation would lead to a false characterisation of risk in the integrated system.

285

286 *2.2. Management infrastructure: Implementation, Compliance and Enforcement*

287 Where management action(s) are deemed necessary, infrastructure is required to implement and
288 enforce the action, otherwise the regulatory objective is unlikely to be met (Heyes, 2000).

289 Infrastructure availability can vary markedly among national and international stakeholders depending
290 on factors such as the political will to implement the necessary controls (e.g. top-down control by
291 governing bodies), the availability of resources (both financial and human), and the prioritisation of an
292 action over other obligations.

293

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

305

306 *2.3. Institutional Support and Multi-national Collaboration*

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

318

319 The ambition level of environmental targets is increasingly linked with the economic and societal
320 implications of 'sustainability'. For example, the European Union's Horizon 2020 Strategy and Marine

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

333

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

344

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

355

356 **3. Conclusions**

357 We have highlighted several of the challenges to the success of an ecosystem approach-driven
358 environmental policy and have outlined a step-wise approach to aid decision-makers in making trade-
359 offs. There are a variety of tools available that aid decision-makers at each stage of the process,
360 whether supporting identifying threats to marine ecosystems from human activities (e.g. Knights et al.,

361 2013; Koss et al., 2011) or estimating the costs and benefits of management actions (e.g. Hussain et
362 al., 2010). The outcomes of such a step-wise approach can provide a transparent and defensible
363 evidence base for a specific decision, but the outcome of each step must be used to inform the next
364 step in the process, without which, a satisfactory trade-off between ecological, societal and economic
365 objectives is unlikely to be achieved and the over-arching objective of the environmental policy not met.

366

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

385

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

399

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

411

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417

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554 **FIGURE LEGENDS**

555

556 **Figure 1.** The steps undertaken by a decision-maker to determine whether a management measure
557 should be adopted under an environmental policy. NB Some steps may have already been undertaken
558 by managers (e.g. to fulfil the requirements of another policy driver) and therefore, decision-making
559 may not need to 'start' where shown here. Abbreviations: GES = Good Environmental Status; MSE =
560 Management Strategy Evaluation.

561

562

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

569

570

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

581