

Quantitative criteria for choosing targets and indicators for sustainable use of ecosystems

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Abstract

Wide-ranging, indicator-based assessments of large, complex ecosystems are playing an increasing role in guiding environmental policy and management. An example is the EU’s Marine Strategy Framework Directive, which requires Member States to take measures to reach “good environmental status” (GES) in European marine waters. However, formulation of indicator targets consistent with the Directive’s high-level policy goal of sustainable use has proven challenging. We develop a specific, quantitative interpretation of the concepts of GES and sustainable use in terms of indicators and associated targets, by sharply distinguishing between current uses to satisfy current societal needs and preferences, and unknown future uses. We argue that consistent targets to safeguard future uses derive from a requirement that any environmental state indicator should recover within a defined time (e.g. 30 years) to its pressure-free range of variation when all pressures are hypothetically removed. Within these constraints, specific targets for current uses should be set. Routes to implementation of this proposal for indicators of fish-community size structure, population size of selected species, eutrophication, impacts of non-indigenous species, and genetic diversity are discussed. Important policy implications are that (a) indicator target ranges, which may be wider than natural ranges, systematically and rationally derive from our proposal; (b) because relevant state indicators tend to respond slowly, corresponding pressures should also be monitored and assessed; (c) support of current uses and safeguarding of future uses are distinct management goals, they require different types of targets, decision processes, and management philosophies.

Keywords: Good Environmental Status, Marine Strategy Framework Directive, sustainable use, assessment, ecological indicators

60 1 Introduction

61 *1.1 From qualitative to quantitative criteria for indicator selection*

62 Ecological indicators are increasingly being used in rule-based management schemes where indicator
63 values outside their respective target ranges trigger management action. The question which properties
64 ecological indicators should have for this purpose has often been addressed in the literature (Elliott,
65 2011; Queirós et al., 2016; Rice and Rochet, 2005). An example relevant for assessment and
66 management of marine ecosystems is the set of criteria proposed by ICES (2001), which forms the basis
67 of the Rice and Rochet (2005) criteria. These relate to concreteness, theoretical basis, public awareness,
68 cost, measurability, representation through historic data, sensitivity, responsiveness, and specificity of
69 indicators. A list by Elliott (2011) containing 18 criteria goes beyond the Rice and Rochet (2005) list, in
70 requiring that indicators (and monitoring parameters) should be anticipatory, broadly applicable and
71 integrative over space and time, interpretable, have low redundancy, be non-destructive, time-bounded
72 and timely. For a detailed review and analysis of indicator selection criteria, see Queirós et al. (2016).

73 However, practically all published specifications of desiderata for ecological indicators and their
74 management targets remain at a qualitative level, despite containing some quantitative components
75 (e.g. reasonable cost in comparison with expected benefits). This has the advantage of flexibility to
76 accommodate variation in preferences and priorities of different stakeholder groups—after all, policies
77 manage human activities rather than the marine environment (Elliott, 2013). However, experts can vary
78 widely in their findings when evaluating indicators according to the same criteria (Rice and Rochet,
79 2005), which questions the idea that such criteria provide an objective basis for indicator selection.
80 Another disadvantage is that the scientific problem of developing indicators and monitoring programs
81 and the scientific and societal challenge of finding appropriate target ranges for these indicators remain
82 vaguely specified. This may lead to inconsistencies in specified target ranges, inefficient use of limited
83 monitoring capacity, and uncertainty about the most appropriate use of research capacity for refining
84 indicators and targets or filling potential gaps in indicator suites (Borja et al., 2012).

85 Ideally, a quantitative, generic, and broadly accepted framework was available for choosing indicators
86 and setting targets, so making this a research and development task to deliver a product according to
87 specifications, rather than a social process of finding common positions in an uncertain space. Such a
88 quantitative framework does currently not exist. Environmental policy documents tend to specify their
89 overall high-level objectives in a qualitative language. The purpose of this contribution is to propose, as
90 a way forward, a quantitative interpretation of this qualitative language, which can then be tested for
91 political acceptance. Being deliberately constructed building on just a few generic principles, our
92 proposal is necessarily somewhat abstract and rigid, and so should not be misunderstood as a direct

93 prescription of policy. More plausibly, it will serve as a scientifically anchored orientation point for
94 political decision making.

95 As a specific policy document which is currently widely discussed in Europe, we chose to focus here on
96 the Marine Strategy Framework Directive (MSFD; EC, 2008) of the European Union (EU). The principles
97 being invoked for setting targets are not consistent within the community implementing the MSFD. For
98 Cochrane et al. (2010), for example, the target is an ecosystem nearly unperturbed by humans, ICES
99 (2014a) primarily require that ecosystem functions are not degraded, Rogers et al. (2010) and ICES
100 (2014b) refer to abundances that can recover from perturbation or have been observed to be
101 historically stable, and Piet et al. (2010) interpret the “safe biological limits” of fish stocks as those
102 producing maximum sustainable yield. We shall here concentrate on policy needs under the MSFD.
103 However, the framework we proposed might be generally useful for linking assessments of aquatic or
104 terrestrial ecosystems to high-level policy goals.

105 ***1.2 The concept of sustainable use***

106 The MSFD requires from EU member states to determine, in a collaborative manner, specific
107 environmental targets and corresponding quantitative indicators that together represent “good
108 environmental status” (GES). It defines GES as:

109 the environmental status of marine waters where these provide ecologically diverse and
110 dynamic oceans and seas which are clean, healthy and productive within their intrinsic
111 conditions, and the use of the marine environment is at a level that is sustainable, thus
112 safeguarding the potential for uses and activities by current and future generations [...].

113 The last passage is a variation of the definition of sustainable development from the Brundtland Report
114 (World Commission on Environment and Development, 1987):

115 Sustainable development is development that meets the needs of the present without
116 compromising the ability of future generations to meet their own needs.

117 Important is that this definition recognizes that needs of future generations might be different from
118 current needs. By referring to “the potential for uses and activities by [...] future generations”, the MSFD
119 follows this tradition. Uncertainty about future uses, and so values, of resources naturally leads to
120 strong notions of sustainability¹ that aim at independent maintenance or enhancement of various forms
121 of natural and non-natural capital (Figge, 2005). Contrastingly, weak sustainability permits substitution
122 of natural with manufactured capital, implicitly assuming good knowledge of their respective future
123 values (Figge, 2005). Correspondingly, we say here “*strongly sustainable*” for use of the environment

¹ Others motivate strong sustainability by non-substitutability of critical natural capital, incomprehension of natural systems, irreversibility of losses, and ethically (Dietz and Neumayer, 2007) .

124 that does not constrain usage choices and capabilities of future generations, and “*weakly sustainable*”
 125 for *use* that simply can be continued indefinitely in its current form (conceivable are even weaker
 126 notions). The distinction between the two concepts is briefly summarized in Table 1.

| Table 1 Comparison of concepts of weakly and strongly sustainable use | | |
|--|---|---|
| | Weakly sustainable use | Strongly sustainable use |
| Types of relevant services | Societal choice | <i>A priori</i> unknown |
| Value of services used | Mostly known | Unknown or uncertain |
| Value to be preserved | Anthropogenic capital plus natural capital | Natural capital |
| Nature of typical target | The <i>point</i> corresponding to optimal long-term use | The <i>range</i> allowing timely recovery |
| Management philosophy | Optimal control (as in control theory) | Limitation of pressures |

127 The best-known example of usage of “sustainable” in our weak sense in the marine ecology context is
 128 “maximum sustainable yield” (MSY). Management for MSY alone does not necessarily imply
 129 sustainability by the stronger definition, because changes to the wider ecosystem resulting from
 130 exploitation may be irreversible. The MSFD refers to weakly sustainable use, for example through the
 131 adjective “productive” in the GES definition above and in a clarifying Commission Decision (EC, 2010),
 132 which explicitly specifies exploitation at MSY as a target.

133 Our considerations here concentrate on strongly sustainable use, thus marking the limits within which
 134 weakly sustainable use options can be explored. From above considerations it follows that constraints
 135 imposed by strong sustainability will generally be weaker than those following from specific weakly
 136 sustainable use objectives; a potential source of confusion to keep in mind.

137 The operationalization of the strong concept of sustainable use in the context of marine management
 138 has been subject of extensive discussion in the work of the International Council for the Exploration of
 139 the Seas (ICES, 2005, 2010, 2013). ICES argued that, since the needs and preferences of future
 140 generations are unknown to us, sustainable use means not to perturb the ecosystem to such a degree
 141 that recovery from these perturbations is impossible or unacceptably slow (see also FAO, 2009). In other
 142 words, under sustainable use the system must remain capable of recovering to an unperturbed state
 143 over an acceptable time span.

144 When making this idea operational, two points need bearing in mind. Firstly, since the management
 145 objective is sustainable use in the present rather than in the past, the unperturbed state is not
 146 necessarily a historic or pre-historic state, but the state that would be reached in the long term if all
 147 anthropogenic pressures were removed. Secondly, the unperturbed state itself is not fixed but
 148 undergoes natural fluctuations.

149 Developing a quantitative interpretation of sustainable use, ICES (2010) proposed to focus indicator
 150 selection on ecosystem components that (1) are under pressure and (2) for which recovery from
 151 pressures is slow or impossible. Indicators are then chosen to quantify the state of these components or

152 features, called “vulnerable components” below, and the pressures on them. This method, however,
153 leaves open the problem of deriving target values for these indicators.

154 Here the approach of ICES is therefore reversed. A rule is proposed for setting target ranges for arbitrary
155 quantitative indicators of ecosystem state such that, for ecosystem components that are not vulnerable
156 in the sense above, the targets will “automatically” be met under almost all circumstances, while
157 indicators relating to vulnerable components are easily driven out of their target ranges under
158 inappropriate management, which is then interpreted as unsustainable use. That is, the rule for setting
159 target ranges implicitly selects indicators critical for monitoring sustainable use, and these implicitly
160 identify vulnerable ecosystem components, so focusing assessments and management on protecting the
161 latter.

162 The selected state indicators are complemented with a set of corresponding pressure indicators, and
163 potentially with additional indicators quantifying state along causal chains linking anthropogenic
164 pressures to vulnerable ecosystem components.

165 **2 The proposal**

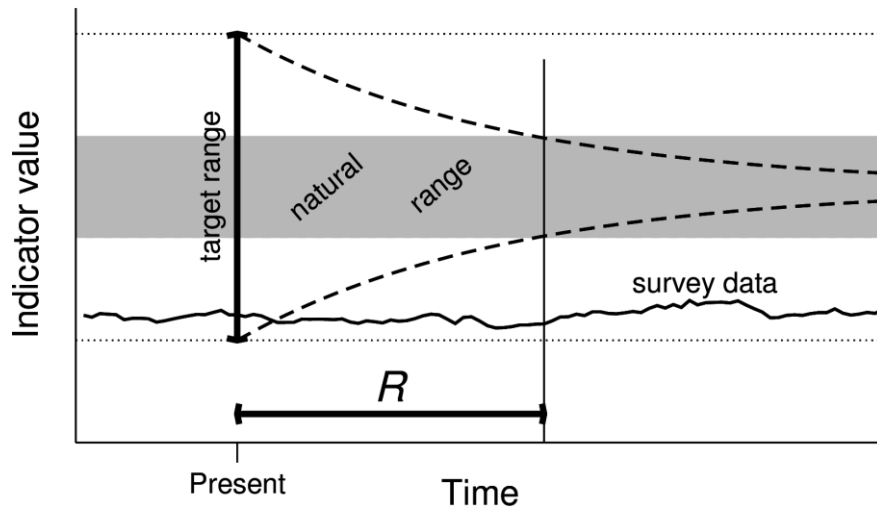
166 ***2.1 Choosing target ranges for state indicators***

167 The rule for choosing indicator target ranges proposed here contains a single free parameter, the
168 longest acceptable mean recovery time R (precisely: the largest acceptable expectation value of time to
169 recovery). The value of R is a matter of societal choice. It could be related, e.g., to the duration of policy
170 cycles or the human life cycle. According to a definition by the FAO (2009), for example, ‘significant
171 adverse impacts’ on ecosystems will typically have recovery times exceeding 5-20 years. Consistent use
172 of the same value of R when setting target ranges for different indicators improves consistency among
173 management goals. Society might require comparisons of the implications of different choices of R in
174 order to make an informed decision on its numerical value. We propose that, to remain consistent with
175 intergenerational freedom of choice, R should not exceed the approximate human generation time of
176 30 years, and assume $R \approx 30$ years in examples we discuss.

177 Now, let I stand for any univariate indicator of ecosystem state. The indicator is here understood as
178 being defined directly in terms of ecosystem state variables, rather than by a protocol to measure these.
179 Without anthropogenic pressures, the value of I would relax to and then naturally fluctuate around
180 some typical value. The resulting distribution of values I can be called its pressure-free, and, in this
181 sense, *natural distribution*.

182 One can define a natural range of variation $[I_{\text{low}}, I_{\text{high}}]$, for example by choosing I_{low} as the 2.5%
183 quantile of the natural distribution, and I_{high} as the 97.5% quantile. Under natural conditions, the
184 indicator is then in the natural range 95% of all times. Because direct observation data corresponding to

185 natural or pristine conditions does not necessarily exist, inferential methods to determine natural
 186 ranges will often be required. We now propose to choose the target range for any indicator as the range
 187 of values from where the mean time to reach the natural range when all pressures are, hypothetically,
 188 removed is not larger than the acceptable mean recovery time R . The idea is illustrated in FIGURE 1.
 189 Management under this rule implies that, after an average transition period R , future generations can
 190 use the corresponding ecosystem component in any form that would have been almost certainly
 191 possible under natural conditions, provided “almost certain” is interpreted as meaning 95% probability.



192
 193 **FIGURE 1** Illustration of proposed approach for choosing target ranges. The target range of an indicator is
 194 determined as the range of values from which it takes, on average, at most a time R to reach the natural range in a
 195 hypothetical situation without anthropogenic pressures. Dotted lines indicate the width of the target range,
 196 dashed lines hypothetical average relaxation trajectories, the grey area the natural range, and the ragged solid line
 197 a conceivable trajectory of the indicator for an ecosystem in strongly sustainable use. In practice, the target range
 198 may need to be narrowed to take measurement uncertainty and model uncertainty into account.

199 The indicator’s natural range depends on external factors, in the case of the MSFD described as “the
 200 associated physiographic, geographic, geological and climatic factors”. Complicating, Earth’s climate is
 201 on a trajectory of directed long-term change, and the natural range corresponding to current climatic
 202 conditions gradually changes. Target ranges should be chosen such that relaxation to the natural range
 203 within R on average is possible even though it changes over time.

204 **2.2 Choosing relevant state indicators**

205 By our proposal, all aspects of ecosystem state are potentially relevant. These including, e.g., the
 206 physical seascape, water temperature and flows, chemical water composition, the structuring elements
 207 of the ecosystem such as habitat-forming species, top predators, and key resource species, but also
 208 endangered species, groups or habitats, and high-level properties such as species richness, community
 209 biomass and production. It follows from our rule of choosing target ranges that among these the state
 210 indicators that are relevant in practice (below “relevant indicators”) are those which are outside their
 211 target ranges or likely to be pushed out of their target ranges by prevalent or foreseeable anthropogenic

212 pressures. Sets of candidate state indicators can initially be scanned for relevance by asking if their
213 recovery to the natural range can conceivably last longer than R .

214 ***2.3 Choosing relevant pressure indicators and their targets***

215 We propose to choose the combined target ranges of pressure indicators in such a way that, when
216 pressures are maintained indefinitely within target ranges, all ecosystem state indicators return to their
217 target ranges and then remain within these ranges during 95% of time.² To cope with empirical
218 uncertainty over pressure-state relationships, an adaptive management scheme where pressure target
219 ranges are iteratively revised based on observed changes in state will often be adequate. Analogously to
220 the state indicators, relevant pressure indicators are those which are outside or likely to be brought
221 outside their target ranges, and they can be found by a similar scanning procedure.

222 ***2.4 Causal relations and supporting indicators***

223 Some vulnerable ecosystem components are not or not only affected by direct anthropogenic pressures,
224 but also indirectly *via* causal chains through other ecosystem components (Borja et al., 2010b). A well-
225 known example are changes in populations at higher trophic levels caused, through bottom-up control,
226 by populations at lower trophic levels, in turn influenced, e.g. by fluvial nutrient input. If pressure-state
227 relationships along these causal chains are not well understood, monitoring of intermediate ecosystem
228 components, e.g. abundance of primary or secondary producers, can play an important role in
229 supporting decision making by managers. Existence of causal “webs” rather than linear chains heightens
230 this need (Borja et al., 2010b). Effective supporting indicators will have comparatively well-understood
231 causal links to both anthropogenic pressures and vulnerable ecosystem components, so maximising the
232 information on causal relations between pressures and states. Target ranges for such supporting state
233 indicators can be determined following the same logic as those for direct anthropogenic pressures.

234 ***2.5 Suites of indicators and correlations between indicators***

235 To adequately capture the status of complex marine ecosystems, large sets of indicators are often
236 proposed. The question then arises by which criterion potentially redundant indicators could be
237 identified and eliminated. Within the present framework, a natural answer arises as follows: consider a
238 situation where, under current and foreseeable pressures, some formula predicts the values of one
239 indicator I in a suite of state indicators from those of the other indicators up to a difference D . Then I

² A 5% probability of failing to meet the target for the state indicator must be admitted for consistency with the 5% probability that state indicators fall outside the natural range even in absence of pressures. To see this, consider state indicators with recovery rates much slower than $1/R$, for which the target range becomes essentially identical to the natural range (Appendix, FIGURE 2).

240 can be replaced by D in this suite without loss of information. When D is not a relevant indicator by our
241 proposal, D (and I) can be removed from the suite.

242 Situations can also arise where relevant state indicators are ecologically coupled so that the mean
243 relaxation time of one indicator depends on the values of other indicators, but the coupling is not strong
244 enough to justify disregarding any of them by the logic above. We suggest two ways of dealing with this
245 situation: (1) to set the target ranges of such indicators depending on the current values of other
246 indicators, or (2) to find target ranges for all indicators such that, as long as all are within target ranges,
247 each will relax to its natural range within R , no matter what the values of the others. Both options
248 reduce to our original proposal if indicators are uncoupled. Option 2 might lead to narrower target
249 ranges, but is more easily administered.

250 **2.6 Precautionary buffers**

251 A precautionary approach to management can be implemented following logic very similar to that
252 applied in traditional fisheries management (ICES, 1998): after determining the target range for an
253 indicator, it is narrowed down to take measurement errors in determining its value and model
254 uncertainties in the determination of the target range into account. Model uncertainties can affect
255 determination of both natural range and mean recovery time.

256 When quantitative estimates of measurement and model uncertainty are available, the precautionary
257 target range could be chosen so that (1) mean recovery time remains $\leq R$ also when taking both kinds
258 of uncertainty into account and (2) the correct indicator value will be within the correct target range in,
259 say, at least 95% of cases. Depending on the circumstances, one or the other condition will be stronger³.
260 Management aiming to respect target ranges of several indicators, while taking uncertainty in system
261 dynamics into account, could make use of the viability kernel method (Cury et al., 2005; Mullon et al.,
262 2004), which works independent of the criteria by which target ranges are defined.

263 For pressure indicators, not only uncertainty in target ranges of subsequently affected state indicators
264 needs to be considered, but also in the pressure-state relationships and the actual magnitude of
265 pressures.

266 It is an economic decision to balance the costs of monitoring and research to improve knowledge of
267 pressure-state relations with the opportunity costs of wider precautionary buffers when uncertainties
268 are high.

³ The first condition is likely to be stronger for state changes involving extinctions or ecosystem bi-stability, the second condition in situations where mean recovery time is a smooth function of the indicator value.

269 **2.7 *Is our science ready?***

270 The importance of recovery times for the management of marine resources has long been recognised in
271 the literature (Borja et al., 2010a; Duarte et al., 2013; Verdonschot et al., 2012). The quantitative
272 application of this concept for indicator selection proposed here is just the logical extension of this line
273 of thought, and can build on rich previous research determining recovery times and modelling recovery
274 processes.

275 The demands of our proposal on the accuracy at which recovery times can be determined might be
276 comparatively low. As shown in Appendix (FIGURE 2), rather coarse estimates will often be sufficient,
277 either because recovery is fast compared to R and so the target range too wide to be relevant, or
278 because recovery is so slow that little variation beyond the natural range is tolerated.

279 **3 Examples**

280 Next we apply our criteria to several types of candidate indicators to explore feasibility and likely
281 practical implications of our proposal. In each case we estimate the magnitude of relaxation times
282 and/or the approximate widths of target ranges, and, based on this, identify the candidates relevant for
283 strongly sustainable use. While the focus is on indicators likely to pass this test, not all candidates we
284 consider do. Overall, we find that sufficient ecological understanding is available to carry out the
285 proposal, and that computation of reliable target ranges would be possible with moderate extra effort.

286 **3.1 *The Large Fish Indicator***

287 The Large Fish Indicator (LFI) is defined as the proportion by biomass of fish caught in a given survey that
288 are longer than a defined length threshold. For the North Sea demersal fish community, sampled by the
289 International Bottom Trawl Survey in quarter 1, the agreed length threshold is 40 cm (Greenstreet et al.,
290 2011). A target range $LFI \geq 0.3$ has previously been set on the basis of pre-1980 data and the view that
291 the early 1980s were “the last period when science experts considered fishing to be generally [weakly]
292 sustainable in the North Sea” (Greenstreet et al., 2011). Because recovery of fish community size
293 structure has been shown to be slow (Fung et al., 2013; Rossberg, 2012; Shephard et al., 2013, 2012), it
294 is desirable to identify a target range consistent with strong sustainability.

295 The natural range of variability of the LFI is not known, but simulation studies (Fung et al., 2013; ICES,
296 2011) predict that indicator values of 0.5 or more could be reached if pressures were lower. Without
297 any fishing, simulations by Fung et al. (2013, Fig S5a) predict indicator values close to 0.8. Assuming a
298 coefficient of variation for LFI of 0.05 in its natural distribution, so that the 2.5% quantile corresponds to
299 about 90% of the mean undisturbed value, simulations by Fung et al. (2013, Fig 7) predict that recovery
300 from $LFI \approx 0.5$ would take around 30 years and recovery from $LFI \approx 0.25$ around 35-40 years. This
301 suggests that $LFI \geq 0.3$ is a reasonable target range if R is on the order of 30 years.

302 Besides being a state indicator for a vulnerable ecosystem component (fish community size structure),
303 the LFI also signals pressures on marine biodiversity. Specifically, prolonged unselective fishing at a rate
304 such that LFI remains near 0.25 leads to extirpation of nearly a third of all large fish species in
305 simulations (Fung et al., 2013, Fig 6a). These extirpations could represent declines of vulnerable
306 components of local biodiversity, even when they do not impede recovery of LFI itself.

307 **3.2 Indicator species**

308 **3.2.1 General considerations**

309 The use of population sizes (or the correlated spatial extent) of selected “indicator species” as indicators
310 for community or environmental status has drawn scepticism from both ecologists (Lindenmayer and
311 Likens, 2010) and jurists (Kelly and Caldwell, 2013). Our proposal supports this scepticism: population
312 sizes of species in communities tend to fluctuate, and exhibit little tendency, if at all, to revert to a
313 preferred value (Kalyuzhny et al., 2014; Korhonen et al., 2010). On longer time scales this leads to the
314 well-documented species turnover (Magnuson et al., 1994). The natural range of variation of species
315 population sizes thus extends from fairly large values (Rossberg, 2013, Sec. 14.6) down to effectively
316 zero. Corresponding indicators would not be relevant in the sense used here. This does, however, not
317 preclude the relevance of community-level indicators derived from population sizes or
318 presence/absence of member species (Faith and Pollock, 2014). In fact, alpha diversity is known to be
319 sensitive to pressures but in unperturbed communities remarkably stable through time (Vellend et al.,
320 2013), as theoretically expected from a control of alpha diversity through structural stability constraints
321 (Rossberg, 2013).

322 Population size or extent of an individual species can potentially be a relevant indicator when this
323 species is under a particular, manageable pressure, when the species is vulnerable to global or regional
324 extinction (from which recovery would be slow or impossible), or when the set of its actual or possible
325 competitors is so small that natural species turnover cannot unfold. For top predators, all three of these
326 criteria are likely to be satisfied, which justifies the use of species-level indicators in this case, as
327 illustrated by the next example.

328 **3.2.2 Abundance of seals as an indicator**

329 Bounty hunting, encouraged in order to decrease the mortality of fish, caused the collapse of the Baltic
330 grey seal (*Halichoerus grypus*) population from approximately 80,000-100,000 individuals in the early
331 1900s to ca. 20,000 individuals in 1940s (Elmgren, 2001; Harding and Härkönen, 1999). Ceased hunting
332 did not result in recovery of the population, however. Most probably due to environmental pollution
333 harming reproduction, the population further decreased to approximately 2,000 in the late 1970s
334 (Boedeker et al., 2002; Harding and Härkönen, 1999). As these pressures have been relieved or removed

335 since the early 1990s, the population has increased to ca. 28,000 individuals today (Harding et al., 2007;
336 Harding and Härkönen, 1999; Härkönen et al., 2013). The population growth rate has been >10% yearly
337 between the early 1990s and mid-2000s, but slowed down to about 6% in the 2010s (Härkönen et al.,
338 2013).

339 The Baltic Marine Environment Protection Commission (HELCOM) monitors the seal population size and
340 growth rate through a core indicator (Härkönen et al., 2013). A target has been set for the population
341 growth rate to $\geq 10\%$ yearly, but none for population size. In addition to hunting and environmental
342 pollution, anthropogenic threats to seals include drowning in fishing gear, and decrease in food quality
343 and spread of parasites due to changes in the food web. A population size of 80,000-100,000 (Harding
344 and Härkönen, 1999) can be used as an estimate of the natural range for the Baltic Sea grey seal.
345 Assuming a constant 10% yearly population growth rate, a population size of 5,050 individuals would be
346 enough to rebuild the population to $N_{low}=80,000$ individuals in $R=30$ years, and $r = 6\%$ yearly growth
347 would require 15,800 individuals or more. Assuming, more realistically, logistic growth with a carrying
348 capacity of $K = 100,000$ individuals, one obtains a lower limit of the strongly sustainable population
349 target range of $N_{lim} = K[1 + e^{rR}(K/N_{low} - 1)]^{-1} = 40,000$ individuals. More detailed models might
350 take dependencies, e.g. on food availability, into account as explained in Section 2.5.

351 As the seal population has increased, predation on valuable fish and damages caused by seals to fishing
352 gear are increasingly seen as problems (Holma et al., 2014; Varjopuro, 2011). On the other hand, it has
353 been proposed that abundant seal populations could boost tourism in coastal communities. Finding a
354 balance between competing services and uses of the marine ecosystem has been recognized as a
355 challenge to be solved (e.g. the ECOSEAL project, <http://www.ecosealproject.eu/>). By our proposal, the
356 ultimately targeted size of the Baltic grey seal population should not lie below N_{lim} to be consistent with
357 strong sustainability.

358 **3.3 Secchi depth**

359 Eutrophication is one of the major pressures at sea, where it affects several other ecosystem
360 components: the food web, sea-floor integrity, and biodiversity (Cloern, 2001). Increases of
361 phytoplankton biomass are primarily caused by anthropogenic nutrient enrichment in the water. One of
362 the key aims of the Baltic Sea Action Plan is a “Baltic Sea unaffected by eutrophication”, and two
363 indicators related to this aim are water clarity (Secchi depth) and chlorophyll a concentration, which are
364 used as proxies for phytoplankton abundance.

365 Secchi depth measurements from 1900-1920 in the northern Baltic Sea range between 5-15 m, with
366 mean values around 9 m (Fleming-Lehtinen and Laamanen, 2012). This can be considered the natural
367 range, as anthropogenic nutrient loading was low at that time. Secchi depth in these basins has since
368 decreased, reaching 2-9 meters during the last decade (Fleming-Lehtinen and Laamanen, 2012). This

369 change is concurrent with increases in nutrient loading and nutrient concentrations in the water.
370 HELCOM targets for Secchi depth in the various basins of the Baltic Sea range between 5.5-8.5 meters
371 (Fleming-Lehtinen et al., 2014). These targets are set based on the principle of allowing 25% deviation
372 from the undisturbed state.

373 While anthropogenic nutrient enrichment is the major driver for nutrient concentrations in the water,
374 eutrophication abatement is complicated by internal loading, a process that recycles sedimented
375 nutrients back to the water column (Pitkänen et al., 2001). Internal loading forms a vicious cycle
376 (Vahtera et al., 2007), as it increases in non-oxygenated sediments, which again increase due to
377 increased sedimentation of phytoplankton biomass. Internal loading can delay the decline of nutrient in
378 the water after a reduction in anthropogenic input. A similar delay must be expected for Secchi depth.

379 Models suggest that response times of nutrient concentrations are of the order of 40 years (Ahlvik et al.,
380 2014; Kiirikki et al., 2006; Neumann and Schernewski, 2008). Linking these models to empirical models
381 for Secchi depth (Savchuk and Wulff, 2007), quantitative target ranges for Secchi depth consistent with
382 strong sustainability could be derived to inform the ongoing debate on target setting (Ahtiainen et al.,
383 2014).

384 ***3.4 Genetic diversity***

385 Operational indicators to quantify genetic diversity within populations have been defined since the
386 1990s (Chenuil, 2006; Petit et al., 1998). Loss of genetic diversity is of concern because of its detrimental
387 impacts on population resilience (Frankham, 2005; Frankham et al., 2002; Schwartz et al., 2007). Genetic
388 diversity will decline sharply during periods of small population size, and laboratory and field studies
389 have documented negative responses to various environmental and anthropogenic pressures (Ozerov et
390 al., 2013; Pini et al., 2011; Taris et al., 2006). Natural variability in populations and the environments can
391 be expected to determine natural variability in genetic diversity. Recovery dynamics of local genetic
392 diversity is understood to result from two processes, mutation and immigration, which exhibit
393 contrasting dynamics. The rate of accumulation of mutations is proportional to the product of the
394 mutation rate per locus per generation, the effective population size (Wright, 1938), and inverse
395 generation time (Baer et al., 2007; Kimura, 1984). For higher organisms, e.g. vertebrates, corresponding
396 time scales easily exceed 30 years. With regular immigration from neighboring or distant populations,
397 recovery can be much faster. Hence, genetic diversity can be a relevant indicator for small populations
398 of long-living species, in particular when these are relatively isolated or experience similar pressures
399 over broad spatial scales.

400 ***3.5 Non-indigenous species indicators***

401 Finally, we consider an important example for which application of the proposed framework is not
402 obvious: choices and target ranges for pressure and state indicators related to non-indigenous species
403 (NIS). Invasion of NIS is often irreversible and so direct recovery impossible (Thresher and Kuris, 2004).
404 Yet, compared with natural species turnover, the fact alone that NIS invade local communities and there
405 compete with native residents might, at regional level, not be an issue (loss of global biodiversity
406 through homogenization of communities notwithstanding). However, invasions by NIS differ from
407 species turnover by natural dispersion in being more likely to go through a phase of rapid population
408 expansion with strong impacts on the ecosystem. At the climax of this expansion phase the affected
409 ecosystem can be driven out of its natural range of variation, but these disruptions differ from case to
410 case. Fortunately, there is mounting evidence that the expansion phase is generally followed by an
411 adjustment phase at which the invader's population and its impact on the ecosystem decline to less
412 disruptive, in cases even beneficial, levels (Blackburn et al., 2011; Reise et al., 2006; Zaiko et al., 2014).

413 Our proposal can be adapted to the case of NIS if one assumes this boom and bust scenario to be the
414 rule (Williamson, 1997), while disregarding cases where the long-term impacts remain high compared to
415 those of natural turnover. One can then interpret the rate of NIS arrivals in an ecosystem as the
416 pressure, and the aggregated disruptive impacts NIS cause before reaching their late adjustment phases
417 as the resulting change in state. The impacts can be considered strongly sustainable if these disruptions
418 would, without new arrivals of NIS, on average decline within time R to levels typical for natural
419 turnover.

420 The quantification of the level of disruptions is complicated by the idiosyncrasy of NIS impacts. Among
421 frameworks suggested for quantifying bioinvasion impacts (Copp et al., 2009; Molnar et al., 2008;
422 Nentwig et al., 2010), the Biological Pollution Level (BPL) assessment method (Olenin et al., 2007) has
423 been recommended as a robust and standardized indicator in the context of the MSFD (Olenin et al.,
424 2010). It has been tested in assessments of the impacts of single and multiple NIS at various scales
425 (Olenina et al., 2010; Zaiko et al., 2011). However, no unambiguous target range has been proposed for
426 it, yet.

427 Recovery times in this interpretation depend on the pattern of boom and bust cycles, which may vary
428 depending on intrinsic or extrinsic factors (Strayer and Malcom, 2006). For zebra mussels in Irish
429 freshwater ecosystems, for example, Zaiko et al. (2014) document recovery times on the order of only
430 ten years since arrival and five years after maximum impact.

431 **4 General implications**

432 ***4.1 Target ranges can differ from natural ranges***

433 Target ranges for indicators are frequently chosen as the indicator values thought to represent
434 ecosystems unperturbed or only slightly perturbed by human interference, see, e.g., the European
435 Water Framework Directive (EC, 2000). The present proposal supports this approach for ecosystem
436 components with relaxation rates much slower than $1/R$ (see Appendix, FIGURE 2). For components
437 that relax faster, the proposal leads to broader indicator target ranges and, crucially, supports this by a
438 simple rationale.

439 ***4.2 Importance of pressure indicators***

440 If an indicator has a long relaxation time, its current value can be interpreted as representing the
441 cumulative effect of pressures over a corresponding time span (see Appendix, Observation 1). The
442 indicator value can change rapidly only when, temporarily, pressures far exceed the level corresponding
443 to strongly sustainable use. If pressures become weaker or are entirely removed, the impact of previous
444 cumulative pressures initially remains and only slowly fades away as the indicator recovers. The analogy
445 to “mining” has been invoked (Herrick et al., 2006). Effectiveness of management on shorter time scales
446 must therefore be assessed not only directly through state indicators, but also indirectly based on
447 corresponding pressure indicators. Hence, pressure indicators have a particularly important role to play
448 in the present interpretation of sustainable use.

449 A pattern one expects to see frequently in time series of state indicators for vulnerable ecosystem
450 components is a rapid decline in phases of unmanaged overuse, followed by slow recovery to a baseline
451 after management became effective (Duarte et al., 2013). Recovery at the same rate as collapse can not
452 generally be expected. Symmetric patterns of decline and recovery are more characteristic of rapidly
453 recovering indicators or natural fluctuations under managed sustainable use.

454 ***4.3 Signal-to-noise ratio, monitoring intensity and costs***

455 Relevant state indicators have narrow ranges of natural variability, and yet are responsive to lasting
456 pressures. In the language of engineering, their signal-to-noise ratio is high.

457 Due to the inherently slow dynamics of relevant indicators and their high signal-to-noise ratio,
458 monitoring intensity does not need to be high, unless there are concerns that present anthropogenic
459 pressures lead to rapid changes in indicator values. Relevant state indicators therefore tend have
460 comparatively low monitoring costs.

461 **4.4 Exceptionality of relevant indicators**

462 Mathematical considerations suggest that, potential state indicators with long relaxation times tend
463 have broad natural ranges of variability (Appendix, Observation 5), because they integrate the impacts
464 of natural fluctuations over long time. Co-occurrence of slow dynamics and small variability, as required
465 for indicator “relevance”, implies that underlying ecosystem properties remain mostly unaffected by the
466 inherent variability of other properties. Often this will be the case because indicator dynamics is
467 governed by general ecological or physical principles (e.g. conservation laws) that inhibit strong
468 fluctuations. Indicators of high relevance by the present proposal therefore can be expected to be
469 rather uncommon among conceivable state indicators at large.

470 When indicator dynamics are governed by general ecological or physical principles it is often possible to
471 approximate dynamics and responses to pressures by simple management models. These management
472 models can inform choices of pressure indicators and their target ranges, as well as management
473 practices to ensure sustainable use. We therefore expect that relevant indicators by the present
474 proposal are among those for which effective management schemes can rather easily be developed.

475 **4.5 The importance of specific use targets**

476 It is desirable that, within the boundaries of strong sustainability, the marine ecosystem provides high
477 levels of services to society. These should be used sustainably in the weak sense. The particular mix of
478 services, however, depends on societal preferences. Some societies might have strong preferences for
479 recreational uses; others might value decomposition of pollutants higher. Management targets for
480 weakly sustainable uses and corresponding indicators are therefore unlikely to derive from simple
481 general criteria. The problem is much more complex (Elliott, 2011), yet addressing it is paramount,
482 because the management objective of strong sustainability on its own is insufficient for achieving the
483 societal benefits it is meant to enable.

484 Returning to the analogy between the precautionary approach to fisheries management and our
485 proposal here, the historic lesson must be recalled that the boundaries of strong-sustainability target
486 ranges might effectively become management targets in the policy process, with detrimental effects for
487 ecosystem functioning and services. It was not long after ICES (1998) established their formulation of
488 the precautionary approach that official ICES advice warned of this issue (ICES, 2002), increasingly so
489 since 2004:

490 *Risk aversion, based on the precautionary approach, defines the boundaries of management*
491 *decisions for sustainable fisheries. Within these boundaries society may define objectives relating*
492 *to benefits such as maximised long-term yield, economic benefits, or other ecosystem services.*
493 *The achievement of such objectives may be evaluated against another set of reference points,*
494 *target reference points, which may be measured in similar dimensions as limit reference points*

495 *but which may also relate to money, food, employment, or other dimensions of societal*
496 *objectives. [...] setting targets for fisheries management involves socio-economic considerations.*
497 *Therefore, ICES does not propose values for Target Reference Points [...]. This means that [...]*
498 *exploitation of most stocks is likely to be sub-optimal, i.e. the long-term yield is lower than it*
499 *could be.*

500 *[...] Managers are invited to develop targets and associated management strategies.*

501 *ICES (2004), original emphasis*

502 Only recently MSY as a use objective was incorporated into the Common Fisheries Policy (EU, 2013).

503 **4.6 Alignment with prevailing approaches**

504 Comparison of the approach laid out here with commonly proposed qualitative criteria for choosing
505 indicators (Queirós et al., 2016) shows them to be either aligned with these criteria or to be unrelated to
506 them. An example for good alignment is the criterion of cost-efficiency, which, as explained above, is
507 expected to be naturally satisfied by many indicators for the state of vulnerable ecosystem components.
508 Examples for criteria that appear unrelated to the current proposal are the concreteness and the easy
509 interpretability of the metrics used (Elliott, 2011). The unrelated criteria can be taken into account
510 alongside those proposed here.

511 The only criterion for indicator selection that is frequently mentioned in the literature but perhaps
512 incompatible with the present proposal is the responsiveness of indicators to management measures.
513 We proposed to address this using pressure indicators and other supporting indicators.

514 Our proposal develops earlier suggestions for an operational definition of GES presented by Borja et al.
515 (2013) by separating the characterizations of weakly and strongly sustainable use. Another distinction of
516 our proposal from the current general understanding is the recognition that not all characteristics of
517 ecosystems are naturally resilient (i.e. recover rapidly and predictably from pressures). Management
518 should pay attention to potential further deterioration of resilience, but of primary concern should be
519 ecosystem components for which resilience is naturally low.

520 **5 Conclusions and policy implications**

521 We proposed a systematic, quantitative approach to select indicators and their target ranges for the
522 purpose of assessing strong sustainability of ecosystem use. The approach offers a rationale for
523 improving consistency among targets and focusing investments into indicator research and monitoring.
524 To close, we highlight three overarching implications of the proposal that are likely to stand out in
525 future developments of MSFD and similar policy instruments.

526 Firstly, proposals for targets of MSFD indicators often still aim at restoring natural or near-natural
527 ecosystem states. This is not always necessary when the policy goal is sustainable use. Here we provided
528 an argument for the choice of alternative, broader target ranges.

529 Secondly, relevant state indicators, by our proposal, will almost always be paired with corresponding
530 pressure indicators or sets of pressure indicators. Situations where either a state or a pressure indicator
531 are sufficient to characterise the status of an ecosystem component are those where the relevant
532 recovery times are comparatively small (Appendix, Observation 3), implying that these ecosystem
533 components are likely to be resilient to pressures and therefore not of primary conservation concern
534 (Appendix, Observation 4).

535 Thirdly, the setting of indicator target ranges for strongly sustainable use and of target ranges or values
536 corresponding to particular use objectives should be clearly distinguished in the policy process.
537 Authority for setting these types of targets might even be assigned to different bodies. An example
538 where such a separation is *de facto* in place is EU fisheries management. The Common Fisheries Policy
539 (EU, 2013) now regulates the setting of fishing quotas in accordance with MSY objectives, while
540 respecting environmental constraints are defined, among others, by the MSFD. The MSFD, in turn,
541 leaves room for pragmatic fisheries management. The two policy instruments are administered by
542 different departments of the European Commission.

543 **6 Appendix: mathematical analyses**

544 In this appendix a minimal mathematic model is introduced that describes relaxation of state indicators
545 to some natural range and responsiveness of state indicators to pressures and environmental
546 fluctuations. The model is then analyzed mathematically in order to develop an understanding of the
547 general relationships between state indicator dynamics, their responsiveness to pressures, and the
548 implications for indicator target ranges.

549 In the model, the indicator value changes because of (i) natural recovery to a value corresponding to an
550 undisturbed state, (ii) external pressures and (iii) uncontrolled natural fluctuations. Specifically, it
551 assumes a dependence of the value of an indicator $I(t)$ on time t to follow

$$\frac{dI(t)}{dt} = -\frac{[I(t) - I_0]}{T} - cP(t) + \text{noise}. \quad (1)$$

552
553 This model is a direct translation of our general understanding of indicator dynamics: The indicator
554 value changes (" $dI(t)/dt$ ") because of ("=") natural recovery (" $-[I(t) - I_0]/T$ ") to a value
555 corresponding to an undisturbed state (" I_0 "), because of external pressures (" $P(t)$ ") and because of
556 uncontrolled natural fluctuations ("noise"). It is legitimate to think of the three terms on the right hand

557 side to be mechanically independent contributions with magnitudes controlled by independent
 558 mechanisms, so that the values of the constants T and c and the strength of the noise are independent
 559 parameters. Equations of the type above are mathematically well studied. An excellent exposition of the
 560 relevant mathematics in easily accessible form can found in the book by Gardiner (1990).

561 The constant c denotes the sensitivity of the indicator to the pressure $P(t)$. The value of this constant
 562 can in principle be determined by monitoring the rate at which the indicator changes $(dI(t)/dt)$ when
 563 suddenly a large constant pressure $P(t) = P$ is applied. The value of c then follows as $c \approx -[dI(t)/dt]/$
 564 P . It can be positive or negative. For simplicity, c is here assumed positive, so that the indicator declines
 565 when a pressure is applied.

566 The parameter T denotes the relaxation time constant of the indicator. When noise is negligible, T is the
 567 time it takes the indicator $I(t)$ to reduce the distance to the equilibrium I_0 from its current value to 40%
 568 ($= \exp(-1)$) of this value in absence of pressures.

569 The solution of Equation (1) is

$$I(t) = I_0 + \int_{-\infty}^t \exp[-(t - \tau)/T] [-cP(\tau) + \text{noise}(\tau)] d\tau. \quad (2)$$

570 **Observation 1** The deviation of $I(t)$ from I_0 is proportional to a weighted sum over previous pressures
 571 and previous noise, with weights decaying exponentially as $\exp[-(t - \tau)/T]$, where τ denotes
 572 points in time in the past (i.e. before t). This weight factor is of the order of magnitude of 1 over an
 573 approximate time span T , and then decays to smaller values.

574 When the “noise” is negligible and a constant pressure $P(t) = P$ is applied over a time that is long
 575 compared to T , the indicator will eventually relax to a constant value

$$I(t) = I_{\text{eq}} = I_0 - TcP(t). \quad (3)$$

576 When pressure changes though time but these changes are slow compared to T , this formula is still a
 577 good approximation.

578 **Observation 2** Equation (3) implies that, in general, large relaxation times T imply a high sensitivity of
 579 the equilibrium value I_{eq} to pressures.

580 **Observation 3** For pressures that change slowly compared to T , there is a direct functional relationship
 581 (here linear) between the pressure P and the state indicator $I(t)$.

582 Most kinds of pressures are not expected to remain constant or approximately constant over the time R .
 583 With this in mind, we arrive at

584 **Observation 4** Direct functional relations between pressure $P(t)$ and state indicators $I(t)$ hold only for
585 state indicators with relaxation times T considerably shorter than R .

586 The “noise” term in Equation (1) describes environmental effects that drive natural fluctuations in the
587 indicator value.⁴ In the presence of noise the indicator does not reach the equilibrium value I_{eq} given by
588 Equation (3) when the pressure is constant or absent, but fluctuates around this value. The width of the
589 range of fluctuation (which is, for the present model, independent of pressure P) increases not only
590 with increasing strength of the “noise”, but, complicating, also with increasing autocorrelation in these
591 fluctuations: the slower these fluctuations, the stronger their impact on $I(t)$.⁵ Yet, as a general rule, it
592 follows, by Equation (2), from the additivity of the effects of noise on $I(t)$ over a recent time interval of
593 approximate duration T , and the randomness of the noise (by definition), that the mean *squared*
594 deviation of $I(t)$ from I_{eq} resulting from noise increases as T . This supports the following

595 **Observation 5** All else equal, indicators with larger relaxation times T tend to have wider natural
596 ranges of variation.

597 For typical forms of the noise, the distribution of $I(t)$ in the absence of pressures follows a normal
598 distribution with mean I_0 . If σ is the standard deviation of this distribution, the natural range according
599 to the definition above is given by $I_{\text{low}} = I_0 - 1.96 \sigma$ and $I_{\text{high}} = I_0 + 1.96 \sigma$.

600 The problem of computing the mean time to recovery is mathematically a problem of computing the
601 mean first passage time of a univariate random process. In the special case that “noise” in the model
602 above is white noise, the mean first passage time for reaching I_{low} from a starting value $I_1 < I_{\text{low}}$ for
603 $P = 0$ is (Gardiner, 1990)

$$\sqrt{\frac{\pi}{2}} T \int_{(I_1 - I_0)/\sigma}^{-1.96} \exp\left(\frac{y^2}{2}\right) \left[1 - \operatorname{erf}\left(\frac{y}{\sqrt{2}}\right)\right] dy, \quad (4)$$

604
605 with $\operatorname{erf}(x)$ denoting the so-called error function. The lower bound of the indicator target range is the
606 value of I_1 for which the expression above equals R . Figure 2 illustrate the resulting dependence of I_1 on
607 T .

⁴ The “noise” term is assumed to have a long-term mean of zero. If not, this can be enforced by adjusting the value of I_0 .

⁵ This assumes autocorrelation time is not much larger than T .

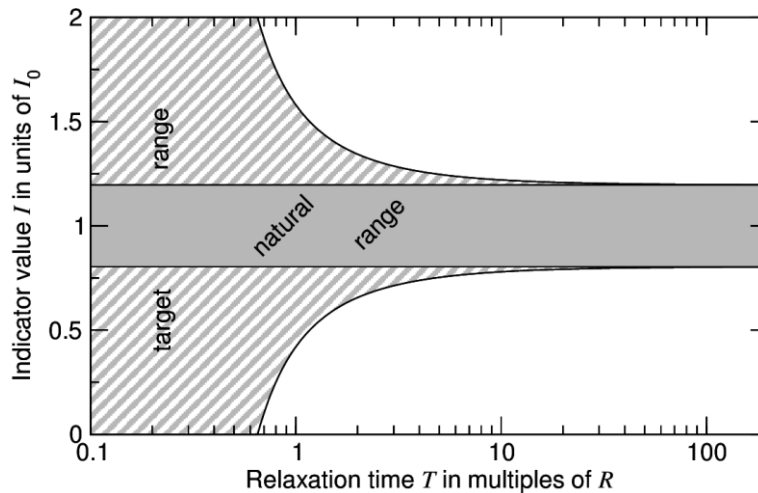


FIGURE 2 Dependence of target range for strongly sustainable use (hatched & grey area) on indicator relaxation time T for the linear model Equation (1). The natural range (grey area) is shown for comparison. Calculation assumes a coefficient of variation for the natural distribution of 0.1.

608 As can be seen in Figure 2, the target range quickly becomes very wide when T is less than about half as
 609 large as R , and differs only little from the natural range for $T > 10R$. The actual value of T therefore
 610 typically matters only when it is within about $0.5R$ to $10R$.

611 For relaxation times T much smaller than the maximal mean recovery time R , corresponding to $(I_1 - I_0)$
 612 much larger than σ , noise can be disregarded and the pressure-free relaxation of $I(t)$ approximated by
 613 a simple, exponential relaxation. For the case that $I(t) = I_1$ at $t = 0$, one gets $I(t) - I_0 = (I_1 -$
 614 $I_0) \exp(-t/T)$. Interpreting I_1 as the lower bound of the target range, the condition $I(R) = I_{low}$ then
 615 leads to $I_1 = I_0 - 1.96 \sigma \exp(R/T)$. Correspondingly, the condition for sustainable use becomes
 616 $I_0 - 1.96 \sigma \exp(R/T) < I(t) < I_0 + 1.96 \sigma \exp(R/T)$.

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