Proposal of indicator tools for GES descriptors on biodiversity food-webs and bottom integrity

Dissemination level

(Restricted to other programme participants, including the Commission Services)

LEAD CONTRACTOR

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Contents

1. Summary

This document reports the progress made by DEVOTES Task 3.3.3 (Develop tools/methodologies for setting reference and target values for biodiversity and food-web GES indicators where possible). A manuscript based on this report will be submitted to the journal Ecological Indicators under the title "Choosing indicators and their target ranges to assess sustainable use of marine ecosystems".

Table 1. Information on Milestone 13 of DEVOTES project.

Milestone number	Milestone name	Work package(s) involved	Expected date[1]	Means of verification[2]
13	Proposal of indicator tools for GES descriptors on biodiversity, food-webs and bottom integrity	3	39	Report

[1] Measured in months from the project start date (month 1: November 2012).

[2] To show how you will confirm that the milestone has been attained. Refer to indicators if appropriate. For example: a laboratory prototype completed and running flawlessly; software released and validated by a user group; field survey complete and data quality validated.

2. Choosing indicators and their target ranges to assess sustainable use of marine ecosystems

2.1. Introduction

2.1.1. From qualitative to quantitative criteria for indicator selection

Ecological indicators are increasingly being used in rule-based management schemes where indicator values outside their respective target ranges trigger management action. The specification of desirable properties of ecological indicators for this purpose has often been addressed in the literature (Elliott, 2011; Queirós et al., 2014; Rice and Rochet, 2005). An example relevant for assessment and management of marine ecosystems is the set of criteria proposed by ICES (2001), which was later developed into the Rice and Rochet (2005) criteria. These relate to concreteness, theoretical basis, public awareness, cost, measurability, representation through historic data, sensitivity, responsiveness, and specificity of indicators. A list by Elliott (2011) contains 18 criteria, and goes beyond the Rice and Rochet (2005) set, for example by requiring that indicators (and monitoring parameters) should be anticipatory, broadly applicable and integrative over space and time, interpretable, have low redundancy, be non-destructive, time-bounded and timely. A detailed review and analysis of indicator selection criteria that have been proposed was recently reported in DEVOTES Deliverable 3.2 (Queirós et al., 2014).

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

If, on the other hand, a quantitative, generic, and broadly accepted framework was available for choosing indicators and setting targets was available, which would allow quantitative comparisons among indicators across the wide variety of components and features of marine ecological communities, finding appropriate suites of indicators might become a well-specified quantitative problem that can be addressed using established scientific methods. Selection of sets of targets and indicators would become a research and development task to deliver a product according to specifications, rather than a social process of finding a common position in an uncertain space. Such a generic, quantitative framework on which society could agree, or which could at least provide a broadly agreed reference point for political deliberations, does currently not exist.

Environmental policy documents tend to specify their overall high-level objectives in a qualitative language. A way forward to arrive at an agreed quantitative specification of these high-level objectives is to first interpret this qualitative language quantitatively, and then to test the political acceptance of such an interpretation. This is the purpose of this Milestone: a tentative proposal for a quantitative interpretation of the concept of sustainable use in the context of indicator selection and target specification.

The DEVOTES project focuses on the European Marine Strategy Framework Directive (MSFD; EC, 2008). The principles being invoked for setting targets are not consistent within the community implementing the MSFD, with Cochrane et al. (2010) and ICES (2014a), for example, indicating that the target should be to have the ecosystem at or near unperturbed state, while Rogers et al. (2010) and ICES (2014b) refer to abundances that can recover from perturbation or have been observed to be historically stable, as well as Piet at al. (2010) interpreting the “safe biological limits” of fish stocks as fishing within MSY.

The MSFD requires from members states of the European Union (EU) to determine, in a collaborative manner, specific environmental targets and corresponding supporting quantitative indicators that together are representative of good environmental status (GES) in the marine waters of the EU. The definition of GES (cited below), in turn, refers to the concept of sustainable use, without specifically defining it. In previous work, DEVOTES has already proposed a pragmatic, operational definition of GES, including a concept of sustainability (Borja et al., 2013), cited in Section 2.5.1 below.

Here another, more systematic and quantitative interpretation of sustainability is proposed. It is deliberately constructed building on just a few generic principles, and therefore necessarily somewhat abstract and rigid, and should not be misunderstood as a direct prescription of policy. More plausibly, it will serve as a scientifically anchored orientation point in the process of political decision making.

2.1.2. The concept of sustainable use

The MSFD defines GES as follows:

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

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

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

Important is that this definition recognizes that the needs of future generations might be different from those of the present generation. By referring to "potential for uses and activities by [...] future generations", the MSFD follows this tradition. This concept of sustainability therefore differs from the purely economic concept, where it means just the possibility in principle to continue current patterns of use over indefinite times. Consistent with established conventions of environmental economics (Figge, 2005), we will call the former the strong concept and the latter the weak concept of sustainable use. The distinction between the two concepts is briefly summarized in Table 2.

The best-known example of usage of "sustainable" in the weak sense in the context of marine ecology is the term maximum sustainable yield (MSY). Management for MSY alone does not necessarily imply sustainability by the stronger definition above, as it does not guarantee that the changes to the ecosystem resulting from exploitation are reversible. The MSFD does refer to weakly sustainable use as well, for example through the adjective "productive" in the definition above, or through the clarifying Commission Decision (EC, 2010), which explicitly specifies exploitation at MSY as a target.

Indeed, it is desirable that, within the boundaries imposed by the requirement of strong sustainability, the marine ecosystem provides a high level of services to society, which should be used sustainably in the weak sense. The particular nature of these services, however, depends on societal preferences. Some societies might have a stronger preference for recreational uses; others might value an ecosystem's ability to decompose pollutants higher. Management targets for weakly sustainable use and corresponding supporting indicators are therefore unlikely to be specifiable through a simple set of general criteria. Elliott (2011) provides an in-depth discussion of the complexities of management for weakly sustainably use. Our considerations here concentrate on strongly sustainable use, thus marking the limits within which weakly sustainable use options can be explored. From above considerations it follows that constraints imposed by strong sustainability will generally be weaker than those following from specific weak sustainability objectives; a potential source of confusion to keep in mind.

Table 2 Comparison of concepts of weakly and strongly sustainable use

	Weakly sustainable use	Strongly sustainable use
Objectives	Human wellbeing	Freedom of future choices
Types of relevant services	Societal choice	*A priori* unknown
Type of target	The *point* corresponding to optimal long-term use	The *range* allowing timely recovery
Value of services used	Mostly known	Unknown and uncertain
Value to be preserved	Anthropogenic capital and natural capital	Natural capital
Management philosophy	Optimal control (as in control theory)	Limitation of pressures

The operationalization of the strong concept of sustainable use in the context of marine management has been subject of extensive discussion in the work of the International Council for the Exploration of the Seas (ICES), in particular by its working group WGECO (ICES, 2005, 2010, 2013). The group argued that, since we do not know what the needs of future generations will be, sustainable use of ecosystem services should be usage that does not perturb the ecosystem to such a degree that recovery of the ecosystem from these perturbations is impossible or unacceptably slow (see also FAO, 2009). In other words, under sustainable use the system must remain capable of recovering to an unperturbed state over an acceptable time span.

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

2.1.3. Developing a quantitative interpretation of sustainable use

WGECO (ICES, 2010) proposed to focus indicator selection on ecosystem components that (1) are under pressure and (2) for which recovery from pressures is slow or impossible. Indicators are then chosen to quantify the state of these components or features (below "vulnerable components" for brevity) and the pressures on them. This method, however, leaves open the problem of deriving target values for these indicators.

To overcome this limitation, the approach of WGECO is here, in a certain sense, reversed. A rule is proposed for setting target ranges for arbitrary quantitative indicators of ecosystem state in such a way that, for ecosystem components that are not vulnerable in the sense above, the targets will "automatically" be met under almost all circumstances, while indicators relating to vulnerable components are easily driven out of their target ranges under inappropriate management. When indicators are driven out of their target ranges, this is interpreted as unsustainable use (or another kind of unsustainable interference with the ecosystem). If monitoring and assessment focus on the indicators likely to move or be outside these target ranges, this will therefore be a focus on the vulnerable components. That is, the rule for setting target ranges implicitly selects indicators critical for monitoring sustainable use, and these indicators implicitly specify vulnerable ecosystem components.

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

2.2. The proposal

2.2.1. Choosing target ranges for state indicators

The rule for choosing indicators proposed here contains a single free parameter, the longest socially acceptable mean recovery time R (precisely: the largest acceptable expectation value of time to recovery). The value of R is subject to societal choice. The magnitude of R could be related, e.g., to the duration of policy cycles or the human life cycle. According to a definition by the FAO (2009), for example, 'significant adverse impacts' on ecosystems will typically have recovery times exceeding 5-20 years. Consistent use of the same value of R when setting target ranges for different indicators improves consistency among management goals. Society might require comparisons of the implications of different choices of R in order to make an informed decision on its numerical value. To remain consistent with the idea of freedom of choice between generations, we propose the approximate human generation time of 30 years as a value that R should not exceed, and shall assume $\mathrm{R} \approx 30$ years in examples we discuss.

Now, let I stand for any univariate indicator of ecosystem state. The indicator is here understood as being defined directly in terms of ecosystem state variables, rather than by a protocol to measure it. Implications of measurement uncertainty are discussed below. Without anthropogenic pressures, the value of I would relax to and then naturally fluctuate around some typical value. The resulting distribution of values I can be called its natural distribution.

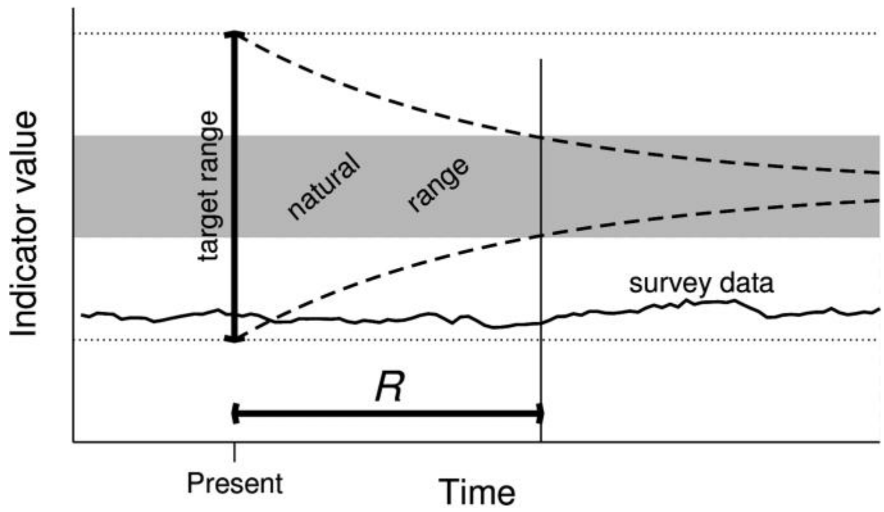

Figure 1 Illustration of proposed approach for choosing target ranges. The target range of an indicator is determined as the range of values from which it takes, on average, at most a time R to reach the natural range in a hypothetical situation without anthropogenic pressures. Dotted lines indicate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

One can define a natural range of variation $[\mathrm{I}_{\mathrm{low}}, \mathrm{I}_{\mathrm{high}}]$, for example by chosing $\mathrm{I}_{\mathrm{low}}$ as the 2.5% quantile of the natural distribution, and $\mathrm{I}_{\mathrm{high}}$ as the 97.5% quantile. Under natural conditions, the indicator would then be in the natural range 95% of all times. It is understood that direct observation data corresponding to natural or pristine conditions does not necessarily exist. We propose to choose the target range for any indicator as the range of values from where the mean time to reach the natural range when all pressures are, hypothetically, removed is not larger than the acceptable mean recovery time R. The idea is illustrated in **Figure 1**.

The indicator's natural distribution, and so the natural range, depend on external factors, in the case of the MSFD described as "the associated physiographic, geographic, geological and climatic factors". Complicating, Earth's climate is on a trajectory of directed long-term change, so that the natural range of variation corresponding to current climatic conditions gradually changes. Target ranges should be chosen such that relaxation to the natural range within R on average is possible even though the natural range changes over time.

In the future we might understand marine ecosystems well enough to gracefully control and change ecosystem states at will by applying appropriate pressures. For terrestrial and freshwater ecosystems, techniques for ecosystem restoration are already well developed (Mitsch, 2004). In marine ecosystems many efforts have been made in restoring different types of habitats (Hawkins et al., 1999; Zhang et al., 2010); however, although there has been some success for semi-enclose areas such as coastal bays or estuaries, other methods (Van Dover et al., 2014) will be needed for open coastal and marine habitats (Elliott et al., 2007; Hawkins et al., 1999). This might then justify choice of wider target ranges. The risk that society loses its ability to restore marine ecosystems should then be taken into account.

2.2.2. Choosing relevant state indicators

By the present proposal, all aspects of ecosystem state are potentially relevant. These including, e.g., the physical seascape, water temperature and flows, chemical water composition, the structuring elements of the ecosystem such as habitat-forming species, top predators, and key resource species, but also endangered species, groups or habitats and high-level properties such as species richness, community biomass and production and their distributions over body size or trophic levels. If follows from the rule above for choosing target ranges that among these the state indicators that are relevant in practice (below "relevant indicators") are those which are outside their target ranges or likely to be pushed out of their target ranges by prevalent or foreseeable anthropogenic pressures. Sets of candidate state indicators can initially be scanned for potentially relevant indicators by asking for each of the candidates if recovery to the natural range can conceivably last longer than R.

2.2.3. Choosing target ranges for pressure indicators

We propose to choose the combined target ranges of pressure indicators in such a way that, when pressures are maintained indefinitely within target ranges, all ecosystem state indicators return to their target ranges and then remain within these ranges in 95% of time[3] To cope with empirical uncertainty over pressure-state relationships, an adaptive management scheme where pressure target ranges are iteratively revised based on observed changes in state will often be adequate.

[3] 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 the absence of pressures. This follows from considering state indicators with recovery rates much slower than $1/R$, for which the target range becomes essentially identical to the natural range (Appendix, Observation 6).

2.2.4. Choosing relevant pressure indicators

Relevant pressure indicators are those which are outside or likely to be brought outside their target ranges. Sets of candidate pressure indicators can initially be scanned for potentially relevant indicators by asking how strong effects of the corresponding pressures on vulnerable aspects of ecosystem state could conceivably be. This scanning procedure complements that carried out in Deliverable 3.2 based on a set of qualitative indicator selection criteria.

2.2.5. Causal relations and supporting indicators

Some vulnerable ecosystem components are not or not only affected by direct anthropogenic pressures, but also indirectly via causal chains through other ecosystem components (Borja et al., 2010b). A well-known example are changes in populations at higher trophic levels affected, through bottom-up control, by populations at lower trophic levels, which in turn can be influenced, e.g. by fluvial nutrient input. If pressure-state relationships along these causal chains are well understood, monitoring causally intermediate components might not be necessary. However, often pressure-state relationships along causal chains will be difficult to quantify, and existence of causal "webs" rather than linear chains complicates matters further (Borja et al., 2010b). Then monitoring of intermediate ecosystem components, e.g. abundance of primary or secondary producers, can play an important role in supporting decision making by managers. Effective supporting indicators will have comparatively well-understood causal links to both anthropogenic pressures and vulnerable ecosystem components, so maximising the information on causal relations between pressures and states. The target ranges for such supporting indicators can be determined following the same logic as those for direct anthropogenic pressure indicators.

2.2.6. Suites of indicators and correlations between indicators

Because of the complexity of typical marine ecosystems, large sets of indicators are often proposed to characterize their ecological status. This naturally raises the questions if all indicators in such suites are required and by which criterion potentially redundant indicators could be identified and eliminated. Within the framework of the present interpretation of sustainable use, such a criterion arises naturally from the interpretation itself.

Strong dependencies between two indicators A and B imply that indicator B can be predicted based on A up to a small residual D. Formally, this means that there is a function $\mathrm{f(A)}$ such that $\mathrm{B = f(A) + D}$, where D is small compared to typical variation in B. The indicator pair $(\mathrm{A, B})$ then provides the same information as the indicator pair $(\mathrm{A, D})$. The residual D, however, can have statistical and dynamic properties very different from those of A and B. In particular, it may be possible to choose $\mathrm{f(A)}$ such that D (i.e. $\mathrm{B - f(A)}$) on its own is not a relevant indicator by the present proposal, i.e., under prevailing or foreseeable conditions D is unlikely to attain values outside its target range. In such cases, the pair of indicators $(\mathrm{A, B})$, or, equivalently $(\mathrm{A, D})$, can be simplified to the single indicator A. This idea is easily generalised to entire indicator suites: an indicators within a suite can be eliminated if its value can be predicted based on other indicators in this suite up to a residual D such that D itself is not a relevant indicator by the present proposal.

Situations can also arise where two indicators A and B for the state of vulnerable ecosystem components are ecologically coupled, but the coupling is not strong enough to justify disregarding one in favour of the other. This complicates target setting, because the value of B can then affect the mean relaxation time of A to its natural range and vice versa. It might be possible to reduce the strength of coupling through a re-definition of the two indicators (a “change of variables”), but this would possibly lead to indicator definitions that are difficult to communicate. An alternative is to set target ranges of A and B jointly such that, for any value of B within its target range, the mean time for A to relax from its target range to the natural range is not larger than R when all pressures affecting A and B are hypothetically removed, and vice versa (exchanging the roles of A and B). This convention is easily extended to larger suites of indicators and, in the case of uncoupled indicators, reduces to our original proposal.

The coupling between two indicators A and B can be such that B affects the equilibrium value of A, but also such that B affects the relaxation time of A to its natural range, i.e. the resilience of the ecosystem component described by A. Both cases and combinations thereof can be treated by the convention for coupled indicators proposed above.

2.2.7. Precautionary buffers

A precautionary approach to management can be implemented following logic very similar to that applied in traditional fisheries management (ICES, 1998): after determining the target range for an indicator, this needs to be narrowed down to take measurement errors in determining the indicator value and model uncertainties in the determination of the target range into account. Model uncertainty contributes at two points in the process: during determination of the natural range of indicator values and when computing mean recovery times.

For example, when quantitative estimates of measurement and model uncertainty are available, the precautionary target range could be chosen so that (1) mean recovery time remains $\leq \mathrm{R}$ also when taking both kinds of uncertainty into account and (2) the correct indicator value will be within the correct target range in, say, at least 95% of cases. Depending on the circumstances, one or the other condition will be stronger[4]

For pressure indicators, uncertainty not only with regards to the target ranges of affected state indicators needs to be taken into account, but also additional uncertainty in determining the actual strengths of pressures and uncertainties in pressure-state relationships. The latter includes uncertainties resulting from natural fluctuations in state indicator values while pressures are applied.

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

2.2.8. Is our science ready?

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

The demands of the present proposal on the accuracy at which recovery times can be determined might be comparatively low. As shown in Appendix (Observation 6), rather coarse estimates of recovery time will often be sufficient, either because recovery is fast compared to R and so the target range too wide to be relevant, or because recovery is slow compared to R so that little variation beyond the natural range is tolerated. In the next chapter, we provide examples of how the proposed concept can be applied to determine target ranges for existing indicators of marine ecosystems, and also discuss some of its restrictions.

2.3. Examples

2.3.1. The Large Fish Indicator

The Large Fish Indicator (LFI) is, for the North Sea, defined as the proportion by biomass of fish longer than 40cm in International Bottom Trawl Survey samples taken in Quarter 1 of each year (Greenstreet et

[4] The first condition is likely to be more stringent 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.

al., 2011). A target range LFI ≥ 0.3 has previously been set on the basis of pre-1980 data and the view that the early 1980s were "the last period when science experts considered fishing to be generally [weakly] sustainable in the North Sea." (Greenstreet et al 2011). Because recovery of fish community size structure is known to be slow (Fung et al., 2013; Rossberg, 2012; Shephard et al., 2013, 2012), it is desirable to identify a target range consistent with strong sustainability.

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

Besides being a state indicator for a vulnerable component of the marine ecosystem (fish community size structure), the LFI also signals pressures on marine biodiversity. Specifically, prolonged unselective fishing at a rate such that LFI remains near 0.25 leads to extirpation of nearly a third of the large fish species in simulations by Fung et al. (2013, Fig 6a). While these extirpations do no impede a nearly complete recovery of the LFI after removal of pressures within 35-40 years, recovery of local species richness through re-colonization processes might take longer.

2.3.2. Indicator species

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

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

2.3.3. Abundance of seals

Bounty hunting, encouraged in order to decrease the mortality of salmon and other valuable fish, caused the collapse of the Baltic grey seal (*Halichoerus grypus*) population from approximately 80,000-100,000 individuals in early 1900s to ca. 20,000 individuals in 1940s (Elmgren, 2001; Harding and Härkönen, 1999). Ceased hunting did not result in recovery of the population, however. Most probably due to environmental pollution harming reproduction, the population further decreased to approximately 2,000 in the late 1970s (Boedeker et al., 2002; Harding and Härkönen, 1999). As these pressures have been relieved or removed since early 1990s, the population has again increased to ca. 28,000 individuals today (Harding et al., 2007; Harding and Härkönen, 1999; Härkönen et al., 2013). The population growth rate has been >10% yearly between the early 1990s and mid 2000s, but decreased to about 6% in the 2010s, as the population abundance was higher than for decades (Härkönen et al., 2013).

Grey seal is listed in the EU Habitats Directive (EC, 1992) annex as a species of community interest, and the Baltic Marine Environment Protection Commission (HELCOM) monitors the seal population size and growth rate through a core indicator (Härkönen et al., 2013). A target for the population growth rate has been set to 10% yearly, but no target for the population size has been set. In addition to hunting and environmental pollutants, human-induced threats to seal populations include drowning in fishing gear, and decrease in food quality and spread of parasites due to changes in the food web, resulting in decreasing health of seal populations. These factors can potentially affect the growth rate of the population.

The 80-100 thousand individuals minimum population size evaluated by Harding and Härkönen (1999) can be used as an estimate of the natural population range of the Baltic Sea grey seal population. Assuming a constant 10% yearly population growth rate, a population size of 5050 individuals would be enough to enable rebuilding of the population to $\mathrm{N}_{\mathrm{low}}$=80,000 individuals in R=30 years, and $\mathrm{r} = 6\%$ yearly growth rate would require 15800 individuals or more. However, maintenance of such a high growth rate as the population is approaching its natural range is unrealistic. Assuming logistic growth with a carrying capacity of $\mathrm{K} = 100{,}000$ individuals, one obtains a lower limit of the strongly sustainable population target range of $N_{lim} = K[1 + e^{rR}(K/N_{low} - 1)]^{-1} = 40{,}000$ individuals. More detailed models might also take dependencies, e.g. on food availability, into account as explained in Section 2.2.6 above.

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

2.3.4. Secchi depth

Eutrophication is one of the major pressures in the Baltic Sea, where it affects several other ecosystem components, having deteriorating effects on the food web, sea-floor integrity, and biodiversity. Increase of phytoplankton biomass is primarily caused by increased nutrient concentrations in the water, caused by nutrient inputs from the drainage basin. Chlorophyll a and Secchi depth are often used as proxies for phytoplankton abundance. In the Baltic Sea, one of HELCOM ‘s key aims of the Baltic Sea Action Plan is a “Baltic Sea unaffected by eutrophication”, and two indicators related to this aim are water clarity (Secchi depth) and chlorophyll a concentration.

Secchi depth measurements from 1900-1920 in the northern Baltic Sea range between 5-15 m, with mean values around 9 m (Fleming-Lehtinen and Laamanen, 2012). This can be considered the natural range, as anthropogenic nutrient loading was low at that time. Secchi depth in these basins has since decreased, reaching 2-9 meters during the last decade (Fleming-Lehtinen and Laamanen, 2012). This change is concurrent with increases in nutrient loading and nutrient concentrations in the water. HELCOM targets for Secchi depth in the various basins of the Baltic Sea range between 5.5-8.5 meters depending on the sub-basin (Fleming-Lehtinen et al., 2014). These targets are set based on the principle of allowing 25% deviation from the undisturbed state.

While the nutrient loading from the drainage basin is the major driver of the nutrient concentrations in the water, the eutrophication abatement issue is made more complicated by internal loading, a process that recycles sedimented nutrients back to the water column (Pitkänen et al., 2001). Internal loading forms a vicious cycle (Vahtera et al., 2007), as it increases in non-oxygenated sediments, which again increase due to increased sedimentation of phytoplankton biomass. Internal loading can delay the reduction of nutrient concentrations in the water after a reduction in nutrient loading from the drainage basin. A similar delay must be expected for the Secchi depth indicator.

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

2.3.5. Non indigenous species indicators

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

Our proposal can be adapted to the case of NIS if one assumes this boom and bust scenario to be the rule (Williamson, 1997), while disregarding those exceptional cases where the long-term impacts of a NIS remain high compared to those of natural turnover. One can then interpret the rate of NIS arrivals in an ecosystem as the pressure, and as the resulting change in state the aggregated disruptive impacts NIS cause before reaching their late adjustment phases. The impacts are then be considered strongly sustainable if the disruptions would, when NIS would cease to arrive, on average decline within time R to levels typical for natural turnover.

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

Recovery times in this interpretation depend on the pattern of boom and bust cycles, which may vary depending on intrinsic or extrinsic factors, e.g. specific biological traits of the species, changes within the invaded community, cumulative abiotic changes, interactions between the invasive species, and other variables that control the ecosystem (Strayer and Malcom, 2006). In general, recovery time is expected to be proportional to the generation time of the impacting NIS. For zebra mussels in Irish freshwater ecosystems, for example, Zaiko et al. (2014) document recovery times on the order of ten years since arrival and five years after maximum impact.

2.4. General implications

2.4.1. Target ranges vs natural ranges

Target ranges for indicators are frequently chosen as the indicator values thought to represent ecosystems unperturbed or only slightly perturbed by human interference, see, e.g., the European Water Framework Directive (EC, 2000). The present proposal supports this approach for ecosystem components with relaxation times that are much longer than R (see Appendix, Observation 6). For components that relax over time scales similar to R or faster, however, the present proposal lead to broader indicator target ranges; crucially, it provides a simple rationale for doing so. Ecosystem management to respect these target ranges, while taking uncertainty in system dynamics into account, could, for example, make use of the viability kernel method (Cury et al., 2005; Mullon et al., 2004), which works independent of the criteria by which target ranges are defined.

2.4.2. Importance of pressure indicators

A direct consequence of the preference for indicators with long recovery times in our interpretation of strongly sustainable use is that, in a sense specified below, the responsiveness of these indicators to management measures is slow, as well.

If an indicator has a long relaxation time, its current value can be interpreted as representing the cumulative effect of pressures over a time span comparable to this relaxation time (see Appendix, Observation 1). The indicator value can change rapidly only if, temporarily, pressures driving the indicator are strong enough to lead to a strong cumulative impact. If pressures become weaker or are entirely removed, the impact of the previous cumulative pressures initially remains and only slowly fades away as the indicator recovers. The analogy to “mining” has been invoked (Herrick et al., 2006). Because slow indicators will not recover immediately if pressures are removed, effectiveness of management must therefore, on shorter time scales, be assessed not only directly in terms of ecosystem state indicators, but also indirectly in terms of corresponding pressure indicators. Hence, pressure indicators have a particularly important role to play in the present interpretation of sustainable use.

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

2.4.3. Signal-to-noise ratio, monitoring intensity, and costs

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

Due to the inherently slow dynamics of relevant indicators and their high signal-to-noise ratio, monitoring intensity does not need to be high, unless there are concerns that persistent strong anthropogenic pressure could lead to rapid changes in indicator values. Relevant state indicators therefore tend to be among those that can be monitored at comparatively low cost.

2.4.4. Exceptionality of relevant indicators

Mathematical considerations suggest that, among potential state indicators, those with long relaxation times tend to be those with broad natural ranges of variability (Appendix, Observation 5). The reason is that slowly relaxing indicators do in effect integrate the impacts of natural fluctuations over long time and so tend to fluctuate widely. Co-occurrence of slow dynamics and small variability implies that corresponding ecosystem properties remain mostly unaffected by the inherent variability of other properties. Typically this will be the case only if indicator dynamics is governed by general ecological or physical principles (e.g. conservation laws) that inhibit strong fluctuations. Many quantifiable ecosystem properties do not fall into this category. State indicators of high relevance by the present proposal, i.e., indicators that relax slowly and have narrow ranges of variability, can be expected to be rather uncommon among conceivable state indicators at large.

For indicators with dynamics that are governed by general ecological or physical principles it is often possible to approximate these dynamics and their responses to pressures by simple management models. Such management models can be used to inform choices of pressure indicators and their target ranges, as well as management practices to ensure sustainable use. The indicators favoured by the present proposal can therefore be expected to be generally among those for which effective management schemes can be developed rather easily.

2.4.5. The importance of weakly sustainable use targets

Coming back to the analogy between the precautionary approach to fisheries management on our proposal here, the inherent risks need to be highlighted as well. It was not long after ICES (1998) established their formulation of the precautionary approach that official ICES advice warned of its shortcomings (ICES, 2002), increasingly so since 2004 :

> *Risk aversion, based on the precautionary approach, defines the boundaries of management decisions for sustainable fisheries. Within these boundaries society may define objectives relating to benefits such as maximised long-term yield, economic benefits, or other ecosystem services. The achievement of such objectives may be evaluated against another set of reference points, <u>target reference points</u>, which may be measured in similar dimensions as limit reference points but which may also relate to money, food, employment, or other dimensions of societal objectives. […] setting targets for fisheries management involves socio-economic considerations. Therefore, ICES does not propose values for Target Reference Points […]. This means that […] exploitation of most stocks is likely to be sub-optimal, i.e. the long-term yield is lower than it could be.*
>
> *[…] Managers are invited to develop targets and associated management strategies.*
>
> *ICES (2004), original emphasis*

Indeed, managers tended to misinterpret the proposed precautionary limits as targets. It took another nine years until MSY as a use objective was incorporated into the EU's Common Fisheries Policy (EU, 2013).

This experience demonstrates the importance of defining specific weakly sustainable use targets consistent current societal needs and preferences, alongside the target ranges corresponding to strongly sustainable used that we have focused on here. The management objective of strong sustainability on its own is insufficient for achieving the societal benefits it is meant to enable.

2.5. Comparison with current approaches

2.5.1. Common criteria for indicator selection

Comparing the approach laid out here with commonly proposed qualitative criteria for choosing indicators (Queirós et al., 2014), one finds it to be generally either aligned with these criteria or to be unrelated to them. An example for good alignment is the criterion of cost-efficiency, which, as explained above, is expected to be naturally satisfied by many indicators for the state of vulnerable ecosystem components. Examples for criteria that appear unrelated to the current proposal are the concretes of and the easy interpretability of the metrics used (Elliott, 2011).

These and other unrelated criteria can be taken into account alongside the one proposed here when defining specific indicators.

The only criterion for indicator selection that is frequently mentioned in the literature but perhaps incompatible with the present proposal is the responsiveness of indicators to management measures. As explained above, state indicators with long relaxation time effectively integrate pressures over time, rather than being representative of current pressures. The immediate effectiveness of management must be monitored through pressure indicators and other supporting indicators.

2.5.2. The common understanding of GES

In previous work (Borja et al., 2013), the DEVOTES collaboration derived an operational definition of "good environmental status" (GES) in the context of the MSFD, based on a review of policy documents and relevant ecological and management considerations (see also Deliverable 6.2 on "Potential definition of good environmental status"). Their summary characterization of GES is a good description of the general current understanding:

> *[GES] is achieved when physico-chemical (including contaminants, litter and noise) and hydrographical conditions are maintained at a level where the structuring components of the ecosystem are present and functioning, enabling the system to be resistant (ability to withstand stress) and resilient (ability to recover after a stressor) to harmful effects of human pressures/activities/impacts, where they maintain and provide the ecosystem services that deliver societal benefits in a sustainable way (i.e. that pressures associated with uses*

cumulatively do not hinder the ecosystem components in order to retain their natural diversity, productivity and dynamic ecological processes, and where recovery is rapid and sustained if a use ceases).

The present proposal develops these thoughts further by separating the characterizations of weakly and strongly sustainable use. We avoided reference to the concept of ecosystem functioning, because "functioning" might be difficult to explicate without implied reference to the purpose of a function and so ultimately to particular ecosystem services we enjoy at present (ICES, 2012). Views of what a functioning ecosystem is might change with changing usage preferences, so it is difficult to base strong sustainability consideration on this concept.

Another distinction of our proposal from the current general understanding is the recognition that not all characteristics of ecosystems are naturally resilient (i.e. recover rapidly and predictably from pressures). Management should pay attention to potential mechanisms that deteriorate resilience further, but the primary focus should be on ecosystem components with naturally low resilience.

2.6. Policy implications

Above, we proposed a quantitative approach to selection of indicators and their target ranges for the purpose of assessing strongly sustainable ecosystem use. To close, we highlight three implications of this interpretation that are likely to stand out in future developments of MSFD and similar policy instruments.

Firstly, proposals for targets of MSFD indicators often still aim at restoring natural or near-natural ecosystem states. This is not always necessary when the policy goal is sustainable use. We here provided a rationale for the choice of alternative, broader target ranges.

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

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

2.7. Appendix: mathematical analyses

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

In the model, the indicator value changes (i) because of natural recovery to a value corresponding to an undisturbed state, (ii) because of external pressures and (iii) because of uncontrolled natural fluctuations. Specifically, it is assumed the dependence of the value of an indicator I(t) on time t follows the model

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

This model is a direct translation of our general understanding of indicator dynamics: The indicator value changes (“dI(t)/dt”) because of (“=”) natural recovery (“-[I(t)-I_0]/T”) to a value corresponding to an undisturbed state (“I_0”), because of external pressures (“P(t)”) and because of uncontrolled natural fluctuations (“noise”). It is legitimate to think of the three terms on the right hand side to be in general mechanically independent contributions, so that their magnitudes are controlled by independent mechanisms, and so the values of the constants T and c and the strength of the noise. Equations of the type above are mathematically well studied. An excellent exposition of the relevant mathematics in easily accessible form can be found in the book by Gardiner (1990).

The constant c denotes the sensitivity of the indicator value to the pressure P(t). The value of this constant can in principle be determined by monitoring the rate at which the indicator changes (dI(t)/dt) when suddenly a large constant pressure P(t)=P is applied. The value of c can then be obtained as c≈-

[dI(t)/dt]/P. The value of c can be positive or negative. For simplicity, it is here assumed to be positive, so that the indicator value declines when a pressure is applied.

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

The solution of Equation (1) is

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

Observation 1: The deviation of I(t) can from I_0 is proportional to a weighted sum over previous pressures and previous noise, with weights decaying exponentially as exp[-(t-τ)/T], where τ denotes some point in time in the past (i.e. before t). The length of the time span over which this weight factor is of the order of magnitude of 1, before it decays to smaller values, is approximately T.

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

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

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

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

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

Most types of pressure are not expected to remain constant or approximately constant over the time R. With this in mind, we arrive at

Observation 4: Direct functional relations between pressures P(t) and state indicators I(t) hold only for state indicators with relaxation times T considerably shorter than R.

The "noise" term in Equation (1) describes effects that drive natural fluctuations in the indicator value.[5] In the presence of noise the indicator does not reach the equilibrium value I_{eq} given by Equation (3) when the pressure is constant or absent, but fluctuates around this value. The width of the range of fluctuation (which is, for the present model, independent of pressure P) increases not only with increasing strength of the natural fluctuations of the environment, but, complicating, also with increasing autocorrelation in these fluctuations, i.e. the slower these fluctuations, the stronger their impact on $I(t)$.[6] Yet, as a general rule, it follows, by Equation (2), from the additively of the effects on $I(t)$ of noise over the recent time interval of approximate duration T, and the randomness of the noise (by definition), that the mean squared deviation of $I(t)$ from I_{eq} resulting from noise increases as T. This supports the following

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

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

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

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

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

> **Observation 6**: As can be seen in Figure 2, the target range quickly becomes very wide when T is less than about half as large as R, and differs only little from the natural range for $T > 10R$. The actual value of T therefore typically matters only when it is within about 0.5R to 10R.

[5] The "noise" term is assumed to have a long-term mean value of zero. If not, this can be enforced by adjusting the value of I_0.

[6] This assumes autocorrelation time is not much larger than T.

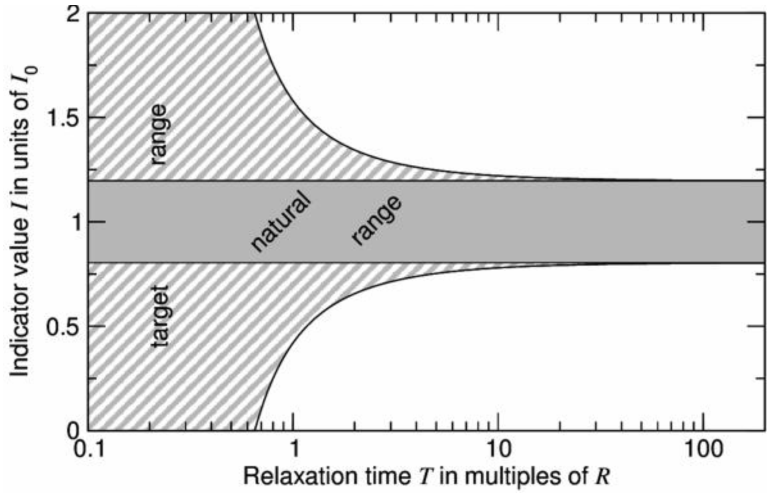

FIGURE 2 Dependence of target range for strongly sustainable use (hatched & gray area) on indicator relaxation time ***T*** for the linear model Equation (1). The natural range (gray area) is shown for comparison. Calculation assumes a coefficient of variation for the natural distribution of 0.1.

For relaxation times T much smaller than the maximal mean recovery time R, corresponding to $(I_1 - I_0)$ much larger than σ, noise can be disregarded and the pressure-free relaxation of $I(t)$ be approximated by a simple, exponential relaxation. For the case that $I(t) = I_1$ at $t = 0$, one gets $I(t) - I_0 = (I_1 - I_0)\exp(-t/T)$. If I_1 equals the lower bound of the target range, the condition $I(R) = I_{low}$ then leads to $I_1 = I_0 - 1.96\,\sigma \exp(R/T)$.

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