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Setting reference levels and limits for good ecological condition in terrestrial ecosystems – Insights from a case study based on the IBECA approach

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ABSTRACT

Effective evidence-based nature conservation and habitat management relies on developing and refining our methodological toolbox for detecting critical ecological changes at an early stage. This requires not only optimizing the use and integration of evidence from available data, but also optimizing methods for dealing with imperfect knowledge and data deficiencies. For policy and management relevance, ecological data are often synthesized into indicators, which are assessed against reference levels and limit values. Here we explore challenges and opportunities in defining ecological condition in relation to a reference condition reflecting *intact ecosystems,* as well as setting limit values for *good ecological condition*, linked to critical ecological thresholds in dose–response relationships between pressures and condition variables. These two concepts have been widely studied and implemented in aquatic sciences, but rarely in terrestrial systems. In this paper, we address practical considerations, theoretical challenges and possible solutions using different approaches to determine reference and limit values for good ecological condition in terrestrial ecosystems, based on empirical experiences from a case study in central Norway. We present five approaches for setting indicator reference values for *intact ecosystems*: absolute biophysical boundaries, reference areas, reference communities, ecosystem dynamics based models, and habitat availability based models. We further present four approaches for identifying indicator limit values for *good ecological condition*: empirically estimated values, statistical distributions, assumed linear relationships, and expert judgement-based limits. This exercise highlights the versatile and robust nature of ecological condition assessments based on reference and limit values for different management purposes, for situations where knowledge of the underlying relationships is lacking, and for situations limited by data availability.

1. Introduction

Ecological assessment tools and approaches that allow early detection of critical changes in biodiversity and ecosystems are key to effective, evidence-based nature management and policy at local, national and global scales (Tittensor et al. 2014). A key function of such tools is to translate and synthesize raw monitoring data (i.e. quantitative metrics of biodiversity and ecosystem functioning over time) into indicators, and to assess the condition and trends in biodiversity and functioning of the monitored ecosystems by comparing these indicators against pre-defined reference values representing a desired state such as *intact ecosystems* or limits for acceptable deviations from this state (e.g. Scholes & Biggs 2005, Loh et al. 2005). How to set these reference levels and/or reference limits is a topic of much debate.

One common approach is to define the reference level as the indicator value in a baseline year (e.g. EEA 2012, EBCC 2019). While this approach is conceptually straightforward, pragmatic, and allows for synthesized trend analyses, it is often less suitable for a comprehensive ecological condition assessment due to lack of relevant historical data (cf. Collins et al. 2020). More specifically, using baseline years is less

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appropriate for monitoring progress towards management goals that are set in absolute terms (e.g. evaluating restoration success) and for comparisons across sites or geographical areas as conditions, and hence baseline values, will generally differ for any given baseline year (cf. Soga & Gaston 2018). For the purpose of general applications and/or comparative purposes, a quantification of a universally defined desirable ecosystem *reference condition* is therefore preferable (Scholes & Biggs 2005, Nielsen et al. 2007).

Such universally-defined reference values may be developed to describe the system in *good ecological condition* as well as by associated limit values – or tipping points – beyond which the system is no longer considered to be in an acceptable condition (e.g. Becker & Hoffmann 2019). This conceptual framework originates from approaches that seek to identify critical ecological thresholds in relation to dose–response relationships between environmental pressures and indicators of ecological condition (Andersen et al. 2008). The reference condition concept is well-developed within freshwater science, where it is also implemented into policy (Stoddard et al. 2006), such as for example in the EU Water Framework Directive (WFD) (EC 2019).

In contrast, tools and approaches based on such predefined reference values and limits for good ecological condition are rare in terrestrial biodiversity and ecosystem monitoring (but see e.g. Scholes & Biggs 2005). This may reflect concerns amongst terrestrial ecologists about over-simplification of nature and, more pragmatically, a lack of data and scientific knowledge about dose–response relationships frustrating attempts to set evidence-based thresholds (cf. Lindenmayer & Luck 2005, Johnson 2013). While these concerns are valid and important, notably for avoiding misguided management, the inherent ability of reference-based approaches to assess and compare progress towards predefined goals is attractive due to high policy and management relevance (Rakocinski et al. 1997). Further, when such approaches are based on well-documented empirical driver-response relationships, as for the critical loads concept for nitrogen deposition (Bobbink & Hettelingh 2011), they provide rigor and reliability.

In this paper, we therefore present and discuss approaches for developing reference values and limit values for good ecological condition in terrestrial ecosystems (Box 1). The approaches are empirically exemplified by the newly-developed Index-Based Ecological Condition Assessment tool (IBECA) based on data from a case study of forest and alpine ecosystems in central Norway (Jakobsson et al., in prep; Nybø et al. 2019). We first introduce the rationale and the conceptual framework behind IBECA, and then discuss practical considerations, theoretical challenges, and possible solutions for setting values for the reference condition and limits for good ecological condition.

2. Case study framework

Recently we developed a new indicator-based ecological condition assessment approach (IBECA), illustrated through an empirical case study from forest and alpine ecosystems in central Norway (Jakobsson et al., in prep; Nybø et al. 2019). Throughout this paper we refer to the IBECA definition of ecological condition: "*the state and trends of structures and functions (incl. productivity) in an ecosystem*". IBECA defines and characterizes ecological condition with respect to seven ecosystem characteristics that encompass key aspects of the biodiversity, structure, and functioning of the ecosystem: primary production, biomass composition across trophic levels, functional groups within trophic levels, functionally important species and (biophysical) structures, biodiversity, landscape patterns and abiotic factors. Each of the seven characteristics are empirically assessed through indicators. In the case study, we used data on eighteen indicators, ranging from sample-based biodiversity-data and population or biomass estimates of important plants and animals, via landscape indicators to biophysical indicators (Appendix A).

The first challenge in developing IBECA was to operationalize the reference conditions and limits for good ecological condition (Box 1) for the indicators. IBECA defines the reference condition as *intact ecosystems* (sensu Nybø et al. 2019, see also Stoddard et al. 2006, EC 2019) characterized by recent historical biodiversity, climatic conditions (1961–1990 the normal period), and where modern intensive or large-scale human pressures are absent (Box 1). Note that this means that IBECA also covers semi-natural ecosystems where traditional extensive management regimes exist within an otherwise naturally functioning ecosystem, and is seen as an integral part of the system (see Jakobsson et al., in prep for details). Harmonizing with the EU WFD, the IBECA defines a state of good ecological condition as '*a condition that does not significantly deviate from the references condition*' (Box 1).

Box 1. Key concepts. Definitions correspond to the definitions used in the presented case study (Nybø et al. 2019).

1. *Ecosystem condition*: an overall estimate of the state and trends of structure and function (incl. productivity) of an ecosystem
2. *Reference condition* (case study): Intact ecosystems, understood as nature not significantly affected by human-driven pressures in the industrial era, with given climate normal (1961-1990) conditions and recent historical biodiversity (e.g. past extinctions are not considered, and species introduced before 1800 AD are regarded as native).
3. *Reference level*: the indicator value representing the reference condition.
4. *Good ecological condition*: A condition where the ecosystem's structure, function and productivity do not deviate significantly from the reference condition.
5. *Limit for good ecological condition*: the indicator value representing a significant deviation from the reference level.
6. *Target level*: management goal set by authorities.

Individual indicators can be (i) positive, i.e. a value decreasing from the reference condition value translates as reduced ecological condition, (ii) negative, i.e. a value increasing from the reference value translates as reduced ecological condition, or (iii) two-sided (e.g. unimodal), i.e. where the reference condition is characterized by an intermediate value and both an increase and decrease from this value represents reduced ecological condition.

To enable comparison and aggregation of results across individual indicators within the IBECA framework, the reference and limit values were used to rescale the raw indicator data into a 0 – 1 scale, where 1 represents the reference condition, 0.6 the limit for good ecological condition (this specific value was used to harmonize with the boundary between good and moderate condition in the EU WFD) and 0 represents a (theoretical or potentially realized) fully degraded condition (Fig. 1; cf. EC 2019). Here we focus on alternative approaches for quantifying and monitoring the state of ecosystems in or near the good ecological condition range (i.e. 0.6 – 1). We note that defining and characterizing the degraded state is a research topic in itself (cf. Ghazoul et al. 2015), and for the purpose of this paper we approach this quantitatively in relative simple terms: Operationally, for positive indicators IBECA considers the lowest possible value (or absence) of an indicator as representing the degraded condition. For negative indicators IBECA defines the highest possible value (realized or theoretical) as representing the degraded condition. For two-sided indicators IBECA defines the two realized or theoretical extreme values representing the degraded conditions.

In the following sections, we discuss opportunities, limitations, and consequences of different approaches and potential data sources for

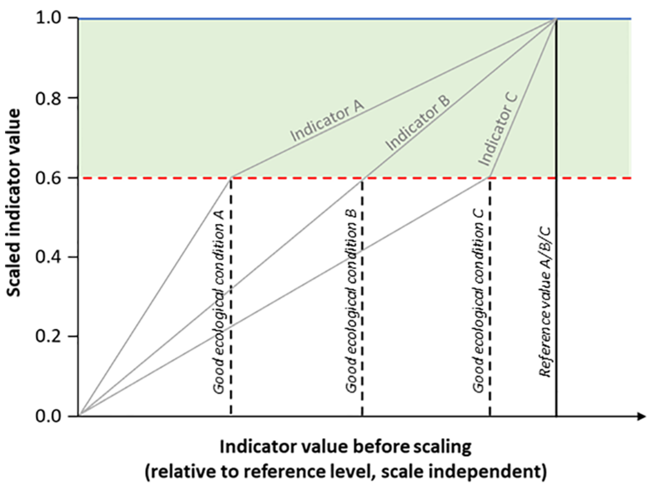

Fig. 1. The rescaling of IBECA indicator values uses three values: the reference level (solid vertical line), the limit for good ecological condition (dashed vertical lines: before rescaling, the three positive example indicators have different limits relative to the reference level) and the value representing a degraded condition. The scaling ensures that the rescaled values along the y-axis can be compared and combined across indicators. Light green = good ecological condition. The visualized scaling approach applies to positive indicators but can easily be applied for negative indicators by mirroring the x-axis, whereas two-sided indicators need a combination of these approaches (see 4.2). Modified from Nybø et al. (2019).

defining empirical reference and limit values for ecological condition indicators. For a complete overview of operationalized reference and limit values for indicators used in IBECA, see Appendix A. While this discussion is based on our experiences with developing the IBECA approach, we emphasize that all our approaches rely on sound empirical system knowledge and/or empirical data, with relevance more generally for other ecological condition frameworks. However, if the specific data needs for using the suggested approaches is not available, the associated method and indicator should not be used, neither alone nor within index-based frameworks such as IBECA.

3. Approaches for setting reference levels

Ideally, reference values should be based on empirical data from relevant reference systems or historical reference periods. In real life, such baseline data are scarce, incomplete or lacking, and our aim is to provide scientifically sound alternatives for tackling realistic situations in which complete empirical data from the reference condition is not available (cf. Section 1). In developing IBECA and testing it by means of the case study we explored five broad categories of approaches to estimate empirical reference levels for ecological condition indicators. The five approaches are summarized in Table 1 and discussed in detail below.

3.1. Absolute biophysical boundaries

For negative indicators, there are often absolute biophysical boundaries that allow us to define the reference level for the indicator value in a straight-forward way, i.e. the reference condition implies that the indicator is absent from the ecosystem. For example, IBECA includes an indicator for area proportion without alien species, for which the reference condition implies no alien species, hence the reference value is 100% of the area without alien species (Table 1). In our case study, this approach was used for three indicators in total. Robustness and transparency are major advantages in using biophysical or proportional boundary indicators, and the connection to ecological integrity is clear (cf. Stoddard et al. 2006). However, there are relatively few potential indicators for which this approach is appropriate.

Table 1
Five approaches to setting reference levels, as exemplified in IBECA and the regional case study in central Norway (Jakobsson et al., in prep; Nybø et al., 2019). The table lists generalized approaches (category), reference condition specification for that category, the approach used for defining reference levels, types of indicators for which the approach is applicable, and an indicator example from the case study.

Category	Reference condition specification	Estimation of reference levels	Type of indicator(s)	IBECA example
Absolute biophysical boundaries	*Intact ecosystem* represented by min. or max. of indicator values	Minimum or maximum value (e.g. 0/1 or 0/100%)	E.g. pollution levels, relative vegetation cover	Alien species coverage
Reference communities	Reference species communities used to reflect the functional signature of *intact ecosystems*	Mean (or median) of distribution of indicator values in a real or theoretical reference community	Any species community based indicator linked to quantitative data per species	Ellenberg-derived vegetation indicator for light
Reference areas	Area(s) representing an *intact ecosystem*	Indicator data from reference areas (e.g. mean or maximum values)	Any	Bilberry coverage
Data + ecosystem dynamics models	Indicator values from models where model predictor values represent an *intact ecosystem*	Reference level values estimated from data-driven expert knowledge on ecosystem dynamics	Species populations or ecosystem structures	Dead wood volume
Demography + habitat availability models	Indicator values from models where model predictor values represent an *intact ecosystem*	Reference level values estimated from data-driven expert knowledge on population dynamics and habitat availability.	Species populations	Wolverine population levels

3.2. Reference areas

If data from appropriate reference areas are available, these could be used to define reference values for basically any indicator. Selecting proper reference areas has been identified as a key step towards good ecological indicators (Soranno et al. 2011). The reference area approach is attractive in that it is data driven, and conceptually easy to relate to *intact ecosystems* as the meaning of a reference condition. However, with the current considerable human impact on natural ecosystems it could be questioned whether such areas exist, in many regions, and to what extent current potential reference sites represent *intact ecosystems* (Stoddard et al. 2006), and how this could be assessed and evaluated. In our case study, we found few indicators with data from areas which could be unequivocally classified as *intact*. In fact, this approach could only be applied to two of the indicators, among them percentage coverage of bilberry (*Vaccinium myrtillus*) based on reference areas within the National Forest Inventory (NFI) data (Tomter et al. 2010).

3.3. Reference communities

This approach is related to the previous category, but with a focus on the species composition (or other characteristics) of either a real or a theoretical reference community. A reference community approach could be based on statistical distributions of species data (cf. Stoddard et al. 2006), data from reference areas (as described above) or historical records. This approach is conceptually similar to the floristic quality index (Bourdaghs et al. 2006), clearly links to ecological integrity, and is in line with the optimal range approach to reference levels (Stoddard et al. 2006). Community composition can be treated as an indicator itself, but the reference community approach can also be used to estimate reference values for indicators based on species' environmental tolerances or functional species attributes (Lewis et al. 2014). We developed three IBECA indicators based on Ellenberg indicator values, where reference levels were estimated using this approach. Ellenberg values are classified ordinal values of species' position along environmental gradients in their realized ecological niche (Hill et al., 1999). We used representative species lists developed for Nature in Norway (Halvorsen et al. 2015) as a reference and calculated reference community weighted mean Ellenberg indicators for each nature type based on these lists. We resampled the representative species lists to generate reference community Ellenberg indicator distributions for each nature type, from which we used the median value to define the reference value (Appendix A, Töpper et al. 2018).

3.4. Ecosystem dynamics based models

With limited reference data (as described above) for indicators, alternative modelling approaches can be applied. This ecosystem dynamics based approach uses a combination of data for selected components of the ecosystem and modelling of ecosystem dynamics to allow calculation of reference values. Data from historical records and/or selected reference sites or empirical models can be used (Stoddard et al. 2006). This approach was used for five indicators in the case study, among which we find two indicators for the volume of dead wood in forests. These indicators were estimated from data on dead wood volume along forest productivity gradients in pristine forests (Siitonen 2001, Ranius et al. 2004), combined with models on age composition in pristine forests (Pennanen 2002). Assumptions based on these models were then applied on NFI productivity and age data (Tomter et al. 2010) in order to estimate expected dead wood amounts in pristine forests in Norway.

3.5. Habitat availability based models

Similar to the above described dynamics model, this approach relies on data to model reference levels, which can be seen as a combination of the *best professional judgement* and *extrapolation from empirical models* described by Stoddard et al. (2006). The approach is closely linked to species distribution modelling but implies making assumptions on potential population densities (under the reference condition) and depends on good data on habitat availability. Estimation of potential population densities should be data driven, and with adequate data and insight about populations, the uncertainty is on the scale of statistical precision. Habitat availability, on the other hand, can constitute a conceptual dilemma, as it can (i) be seen in the light of available habitat today, (ii) needs to be modelled back in time, or (iii) potential natural vegetation (Chiarucci et al. 2010) needs to be estimated. In the case study, we adopted the first of these approaches (but see assumptions in Box 1), and used habitat availability based models for five indicators. Among these is the wolverine population indicator, with model assumptions on habitat availability and potential population levels based on Lande et al. (2003).

4. Approaches for setting limits for good ecological condition

The dose–response relationship between an environmental pressure and the response in terms of change in ecological condition can take many different shapes (Andersen et al. 2008). In cases where dose–response relationships are established, the major challenge in setting indicator value limits for good ecological condition resides in agreeing how much the ecological conditions should be allowed to deviate from the reference before the condition is no longer *good*. For alien species, for example, it is relatively straight-forward to argue that the reference condition should be their absence, but should the limit for *good* ecological condition be set at 1, 5, 25% cover of alien species? This may vary between systems, but also between ecosystem characteristics or indicators within the same system. Challenges quickly exacerbate in real, often data-deficient systems: the dose–response relationships between pressures and indicators are often not well documented in the literature, and the data often do not allow analytical approaches to set limits for good ecological condition. We have identified four different conceptual approaches to set limits for good ecological condition (Table 2), and discuss these, giving examples from the IBECA project, in

Table 2
Examples of usage of the four approaches to setting limit values for good ecological condition in the IBECA case study on ecological condition in Trøndelag, Norway (Jakobsson et al., in prep; Nybø et al., 2019). The table lists the generalized approach (category), how it relates to the definition of good ecological condition, scaling possibilities, and an indicator example from the case study.

Category	Relation to limit for good ecological condition	Scaling	Example
Empirically estimated values	Critical levels of the indicator can be directly linked to empirical data	Flexible	Nitrogen deposition
Statistical distributions	Distribution of indicator values within a reference data population used to estimate statistical deviance from mean	Flexible, often two-sided	Ellenberg-derived vegetation indicator for light
Assumed linear relationships	Based on scientific expertise, the relationship between the reference condition and a degraded ecosystem is assumed to be linear	Linear	Dead wood volume
Expert judgement-based limits	Based on scientific expertise, the relationship between the reference condition and a degraded ecosystem is assumed to be non-linear	Non-linear	Alien species coverage

this section.

4.1. Empirically estimated values

When the required empirical evidence for determining the dose–response relationships for the important ecosystem indicators and characteristics exist, setting limits for good ecological condition are relatively straightforward. In our case study, the only indicator for which this approach was applied on was nitrogen deposition. Critical loads for nitrogen deposition (Bobbink & Hettelingh 2011) readily lend themselves to define limit values for good ecological condition, whereas this knowledge is lacking for most other terrestrial indicators.

4.2. Statistical distributions

Statistical distributions of data can be used for setting limits for good ecological condition, in particular when using reference communities as a reference. In the case study, reference value distributions derived from representative species lists for different habitat types were generated for Ellenberg indicators. These reference distributions largely constitute two-sided distributions with tails on both sides of the Ellenberg scale, and thus provide both a lower and an upper limit for good ecological condition. For these indicators we used the median of the respective habitat type's distribution as the reference value (see above), and the lower and upper limits of the 95% confidence interval as the limit values for good ecological condition (Appendix A), as the probability of observing an indicator value outside this interval in a reference community is 0.05. An exception from two-sided distributions is Ellenberg salt tolerance: where the interval representing good ecological conditions includes zero.

4.3. Assumed linear relationships

In its simplest form, indicator values can be assumed to be linearly related to ecological condition (see Fig. 1). This was the case for most indicators in our case study, assuming that less than 60% of the reference value is critical for the indicator's condition. For some of these indicators this was a straightforward expert assumption (e.g. wolverine in alpine ecosystems), for others it was based on expert judgement supported by data reported in the literature (e.g. dead wood in forests). However, we acknowledge the need for indicator updates, including refining limit values, based on improved knowledge and data (Jakobsson et al., in prep, Nybø et al. 2019).

4.4. Expert judgement-based limits

Although linear approximations are commonly used in ecology, non-linear dose–response curves are often expected due to the complexity of ecosystems (Andersen et al. 2008). If data do not allow for quantitative dose–response estimations, but there is reason to expect a non-linear relationship between the indicator variable and ecological condition, expert judgement can be used to approximate this non-linear relationship. In our case, we used a simplified two-step linear regression approach to approximate non-linear relationships between indicator values and ecological condition (see Fig. 1), where the critical judgement was for setting the limit for good ecological condition. An example from the case study was the indicator area proportion without alien species, where the disproportionate negative effect of alien species on the ecosystem was accounted for when setting the limit value to 95%.

5. Conclusions

Robust assessments of ecological condition rely on integrating information from several indicators covering various structures and functions of the targeted ecosystem. Available data often vary in aspects like coverage, scale and data type. Hence, a toolbox of conceptualized approaches for setting reference and limit values for good ecological condition is necessary to facilitate the practical application and best possible use of data and integration of different data types in quantitative assessments of ecological condition. In this paper, we present five main approaches for setting reference values and four approaches for setting limit values for good ecological condition, and discuss our experiences concerning practical considerations, theoretical challenges and possible solutions. We hope that the approaches we present here will assist further development and practical implementation of quantitative, and testable, concepts for assessments of ecological condition.

CRediT authorship contribution statement

Simon Jakobsson: Conceptualization, Formal analysis, Investigation, Methodology, Project administration, Visualization, Writing - original draft, Writing - review & editing. **Joachim Paul Töpper:** Conceptualization, Methodology, Visualization, Writing - original draft, Writing - review & editing. **Marianne Evju:** Conceptualization, Data curation, Investigation, Methodology, Writing - review & editing. **Erik Framstad:** Conceptualization, Methodology, Writing - review & editing. **Anders Lyngstad:** Writing - review & editing. **Bård Pedersen:** Writing - review & editing. **Hanne Sickel:** Writing - review & editing. **Anne Sverdrup-Thygeson:** Methodology, Writing - review & editing. **Vigdis Vandvik:** Conceptualization, Investigation, Methodology, Writing - review & editing. **Liv Guri Velle:** Conceptualization, Investigation, Methodology, Writing - review & editing. **Per Arild Aarrestad:** Writing - review & editing. **Signe Nybø:** Conceptualization, Methodology, Project administration, Writing - review & editing.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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References

Andersen, T., Carstensen, J., Hernández-García, E., Duarte, C.M., 2008. Ecological thresholds and regime shifts: approaches to identification. Trends Ecol. Evol. 24, 49–57. https://doi.org/10.1016/j.tree.2008.07.014.

Becker, H., Hoffmann, J. (Eds.). 2019. Sustainable development in Germany – Indicator Report 2018. Federal Statistical Office of Germany. Report, Order number: 0230002-18900-1, 155 pp.

Bobbink, B., Hettelingh, J.-P. (Eds.) 2011. Review and revision of empirical critical loads and dose-response relationships. Proceedings of an expert workshop, Noordwijkerhout, 23-25 June 2010. Coordination Centre for Effects, National Institute for Public Health and the Environment (RIVM).

Bourdaghs, M., Johnston, C.A., Regal, R.R., 2006. Properties and performance of the floristic quality index in great lakes coastal wetlands. Wetlands 26, 718–735. https://doi.org/10.1672/0277-5212(2006)26[718:PAPOTF]2.0.CO;2.

Chiarucci, A., Araújo, M.B., Decocq, G., Beierkuhnlein, C., Fernández-Palacios, J.M., 2010. The concept of potential natural vegetation: an epitaph? J. Veg. Sci. 21, 1172–1178. https://doi.org/10.1111/j.1654-1103.2010.01218.x.

Collins, A.C., Böhm, M., Collen, B., 2020. Choice of baseline affects historical population trends in hunted mammals of North America. Biol. Conserv. 242, 108421. https://doi.org/10.1016/j.biocon.2020.108421.

EBCC 2019. European Bird Census Council, http://www.ebcc.info. Accessed 20 Dec 2019.

EC 2019. European Commission. The EU Water Framework Directive – integrated river basin management for Europe. http://ec.europa.eu/environment/water/water-framework/index_en.html. Accessed 2 Dec 2019.

EEA 2012. European Environment Agency. Streamlining European biodiversity indicators 2020: Building a future on lessons learnt from the SEBI 2010 process. EEA Technical Report No 11/2012, 45 pp, Copenhagen.

Ghazoul, J., Burivalova, Z., Garcia-Ulloa, J., King, L.A., 2015. Conceptualizing forest degradation. Trends Ecol. Evol. 30, 622–632. https://doi.org/10.1016/j.tree.2015.08.001.

Halvorsen, R., Bryn, A., Erikstad, L., Lindgaard, A. 2015. Natur i Norge - NiN. Versjon 2.0.0. Artsdatabanken, Trondheim.

Hill, M.O., Mountford, J.O., Roy, D.B., Bunce, R.G.H. 1999. Ellenberg's indicator values for British plants. ECOFACT Volume 2 technical annex. Institute of Terrestrial Ecology, Huntingdon, 46 pp.

Jakobsson, S., Evju, M., Framstad, E., Imbert, A., Lyngstad, A., Sickel, H., Sverdrup-Thygeson, A., Töpper, J.P., Vandvik, V., Velle, L.G., Aarrestad, P.A., Nybø, S. In prep. An index-based assessment of ecological condition and its links to international frameworks.

Johnson, C.J., 2013. Identifying ecological thresholds for regulating human activity: Effective conservation or wishful thinking? Biol. Conserv. 168, 57–65. https://doi.org/10.1016/j.biocon.2013.09.012.

Lande, U.S., Linnell, J.D.C., Herfindal, I., Salvatori, V., Brøseth, H., Andersen, A., Odden, J., Andrén, H., et al., 2003. Utredninger i forbindelse med ny rovviltmelding. Potensielle leveområder for store rovdyr i Skandinavia: GIS-analyser på et regionalt nivå. NINA Fagrapport 64, 31 pp.

Lewis, R.J., Pakeman, R.J., Angus, S., Marrs, R.H., 2014. Using compositional and functional indicators for biodiversity conservation monitoring of semi-natural grasslands in Scotland. Biol. Conserv. 175, 82–93. https://doi.org/10.1016/j.biocon.2014.04.018.

Lindenmayer, D., Luck, G.W., 2005. Synthesis: Thresholds in conservation and management. Biol. Conserv. 124, 351–354. https://doi.org/10.1016/j.biocon.2005.01.041.

Loh, J., Green, R.E., Ricketts, T., Lamoreux, J., Jenkins, M., Kapos, V., Randers, J., 2005. The Living Planet Index: using species population time series to track trends in biodiversity. Phil. Trans. R. Soc. B 360, 289–295. https://doi.org/10.1098/rstb.2004.1584.

Nielsen, S.E., Bayne, E.M., Schieck, J., Herbers, J., Boutin, S., 2007. A new method to estimate species and biodiversity intactness using empirically derived reference condition. Biol. Conserv. 137, 403–414. https://doi.org/10.1016/j.biocon.2007.02.024.

Nybø, S., Framstad, E., Jakobsson, S., Evju, M., Lyngstad, A., Sickel, H., Sverdrup-Thygeson, A. et al. 2019. Test av fagsystemet for økologisk tilstand for terrestriske økosystemer i Trøndelag. NINA Rapport 1672.

Pennanen, J., 2002. Forest age distribution under mixed-severity fire regimes – a simulation-based analysis for middle boreal Fennoscandia. Silva Fennica 36, 213–231.

Rakocinski, C.F., Brown, S.S., Gaston, G.R., Heard, R.W., Walker, W.W., Summers, J.K., 1997. Macrobenthic responses to natural and contaminant-related gradients in northern Gulf of Mexico estuaries. Ecol. Appl. 7, 1278–1298. https://doi.org/10.1890/1051-0761(1997)007[1278:MRTNAC]2.0.CO;2.

Ranius, T., Jonsson, B.G., Kruys, N., 2004. Modelling dead wood in Fennoscandian old-growth forests dominated by Norway spruce. Can. J. For. Res. 34, 1025–1034. https://doi.org/10.1139/x03-271.

Scholes, R.J., Biggs, R., 2005. A biodiversity intactness index. Nature 434, 45–49. https://doi.org/10.1038/nature03289.

Siitonen, J., 2001. Forest management, coarse woody debris and saproxylic organisms: Fennoscandian boreal forests as an example. Ecol. Bull. 49, 11–41.

Soga, M., Gaston, K.J., 2018. Shifting baseline syndrome: causes, consequences, and implications. Front. Ecol. Environ. 16, 222–230. https://doi.org/10.1002/fee.1794.

Soranno, P.A., Wagner, T., Martin, S.L., McLean, C., Novitski, L.N., Provence, C.D., Rober, A.R., 2011. Quantifying regional reference conditions for freshwater ecosystem management: A comparison of approaches and future research needs. Lake Reservoir Manage. 27, 138–148. https://doi.org/10.1080/07438141.2011.573614.

Stoddard, J.L., Larsen, D.P., Hawkins, C.P., Johnson, R.K., Norris, R.H., 2006. Setting expectations for the ecological condition of streams: the concept of reference condition. Ecol. Appl. 16, 1267–1276. https://doi.org/10.1890/1051-0761(2006)016[1267:seftec]2.0.co;2.

Tittensor, D.P., Walpole, M., Hill, S.L.L., Boyce, D.G., Britten, G.L., Burgess, N.D., Butchart, S.H.M., et al., 2014. A mid-term analysis of progress toward international biodiversity targets. Science 346, 241–244. https://doi.org/10.1126/science.1257484.

Tomter, S.M., Hylen, G., Nilsen, J.E., 2010. Norway. In: Tomppo, E., Gschwanter, T., Lawrence, M., McRoberts, R. (Eds.), National Forest Inventories, Pathways for Common Reporting. Springer, pp. 411-424.

Töpper, J., Velle, L.G., Vandvik, V. 2018. Developing a method for assessment of ecological state based on indicator values after Ellenberg and Grime (revised edition). NINA Report 1529b, 25 pp.