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Introducing the index-based ecological condition assessment framework (IBECA)

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A B S T R A C T

Sustainable nature management and ecosystem conservation depends critically on scientifically sound and stakeholder-relevant analytical frameworks for monitoring and assessing ecological condition. Several general frameworks are currently being developed internationally, including the Essential Biodiversity Variables (EBV), and the UN's SEEA EEA Ecosystem Condition Typology (ECT). However, there has so far been few attempts to develop empirical implementations of these general frameworks, or to assess their applicability for environmental decision-making at national or regional scales. In this paper, we aim to fill this implementation gap by demonstrating a practical application of an empirically-based ecological condition assessment framework, the Index-Based Ecological Condition Assessment (IBECA). IBECA defines seven major classes of indicators of ecological condition, representing distinct ecosystem characteristics, and empirically synthesizes indicators for each of these characteristics from various monitoring data. We exemplify and explore the utility and robustness of IBECA using a case study from forest and alpine ecosystems in central Norway, and we investigate how IBECA aligns with the two international frameworks EBV and ECT. In particular, we analyze how the different approaches to categorize indicators into classes affect the assessment of ecological condition, both conceptually and using the case study indicators. We used eleven indicators for each of the two ecosystems and assessed the ecological condition according to IBECA for i) each individual indicator, ii) the seven ecosystem characteristics (indicator classes), and iii) a synthetic ecological condition value for the whole ecosystem. IBECA challenges key concepts of the international frameworks and illustrates practical challenges for national or regional level implementation. We identify three main strengths with the IBECA approach: i) it provides a transparent and management-relevant quantitative approach allowing assessment of spatio-temporal variation in ecological condition across indicators, characteristics and ecosystems, ii) the high degree of flexibility and transparency facilitates updating the ecological condition assessments, also back in time, as improved data and knowledge of indicators emerge, and iii) the quantitative and flexible procedure makes it a cost-effective approach suitable for fast management implementations. More generally, we stress the need for carefully choosing appropriate classification and aggregation approaches in ecological condition assessments, and for transparent and data-driven analytical approaches that can be adjusted as knowledge improves.

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1. Introduction

Alarming reports on biodiversity loss with dramatic consequences for ecosystems and society are feeding into the scientific literature. Substantial and rapid losses of taxonomic diversity, genetic diversity and abundance of organism groups are driven by human impacts such as land use change, exploitation, pollution and human-induced climate change (Newbold et al., 2015; Allentoft and O'Brien, 2010; Sánchez-Bayo and Wyckhuys, 2019; IPBES, 2019). Despite local, regional, and global conservation, restoration, and management initiatives, these biodiversity losses continue (Mace et al., 2018). It is widely acknowledged that we have failed to reach the 2020 Aichi targets (CBD, 2011; Tittensor et al., 2014) and the process of defining new, post-2020 biodiversity targets has been initiated (CBD, 2020). It is becoming increasingly clear that without relevant, sensitive and precise empirical indicators to assess progress towards these targets, we will likely continue to fail in our response to critical ecological changes (Pe'er et al., 2014; Tittensor et al., 2014). There is therefore an urgent need for scientifically sound and stakeholder-relevant monitoring and assessment frameworks to inform sustainable biodiversity and ecosystem management, as well as conservation (Reed, 2008).

Biodiversity policy and action plans for ecological sustainability worldwide aim to fulfill two critical goals: (i) maintain the diversity of natural habitats and species populations within their natural ranges; and (ii) maintain ecosystem structure and functioning (incl. productivity) (e.g. Millennium Ecosystem Assessment, 2005, Nature Diversity Act (NDA), 2009, TEEB, 2010). In order to reach these goals, managing for ecosystem resilience is critical (Diaz et al., 2006, Biggs et al., 2012). In practice, the concept of ecosystem resilience (Standish et al., 2014) is still difficult to apply and use across systems (Folke et al., 2010), as it requires the assessment of the overall ecological condition ('health' or 'state', cf. the Drivers, Pressures, States, Impacts, Response (DPSIR) framework (Smeets and Weterings, 1999)), including indicators of structure and function in addition to species diversity. Hence, if we could effectively and empirically assess state and trends in ecological condition, we could also better guide and adapt policy and management and thereby better link conservation goals and efforts (cf. Pollock et al., 2017).

The Essential Biodiversity Variables (EBV) is one major approach to assess progress towards the Aichi targets (Scholes et al., 2008, Pereira et al., 2013). The EBV concept proposes an indicator set with 21 indicators divided into six classes (genetic composition, species populations, species traits, community composition, ecosystem structure, ecosystem function; see Geijzendorffer et al., 2016). The Ecosystem Condition Typology (ECT) under development by the United Nations statistical group as part of the SEEA-EEA framework for ecosystem accounting (UN, 2012, Hein et al., 2020) aggregates indicators into seven broad classes (structural, compositional, function, physical and chemical state, and landscape patterns within and across ecosystems) (Maes et al., 2019; Czúcz et al., 2019). ECT is a flexible approach that can be adapted to data availability and the purpose of the ecological condition assessment. Indicator aggregation can also be adapted to better and more precisely inform policy and management at different resolutions.

Hence, the EBV concept provides a framework for developing indicator and monitoring systems, whereas the ECT provides a conceptual framework for using and combining indicators to assess ecological condition. However, the EBV does not propose approaches for aggregating indicators (e.g. to assess the overall ecological condition) and the ECT has merely started testing out its practical implementation using empirical datasets (https://seea.un.org/). In this paper, we present a newly developed index-based ecological condition assessment (IBECA) that combines a theoretically-derived general framework for defining indicators of good ecological state with a quantitative approach to derive and aggregate relevant empirical data from monitoring (Nybø and Evju, 2017). To demonstrate the utility of the approach, we test it empirically in a case study from central Norway (Nybø et al., 2019). Based on this exercise, we first exemplify and discuss how ecological condition assessments can be cost-effectively conducted using available monitoring data to develop empirical indicators for important aspects of the overall ecological condition. In contrast to previous studies (e.g. Vihervaara et al., 2017, Turak et al., 2017), our study fills an important implementation gap in that we link monitoring data, indicator development and ecological condition assessment (cf. Hein et al., 2020). Second, we discuss potential opportunities for improved management relevance arising from the parallel development of indicator networks at several spatial and conceptual scales. Third, we discuss how IBECA is linked to the EBV and ECT frameworks and evaluate its potential as an operationalization of these frameworks to better inform ecosystem management. Finally, we discuss the way forward, including how our analysis can benefit the operationalization of other frameworks.

2. Materials and methods

2.1. IBECA: an index-based ecological condition assessment framework

IBECA is an empirically-based ecological condition assessment framework. The framework is based around seven overarching ecosystem characteristics that together define and characterize ecosystem structure and functioning; primary production, biomass composition across trophic levels, functional groups within trophic levels, functionally important species and (biophysical) structures, biodiversity, landscape patterns and abiotic factors (Fig. 1, Table A1). For each of these seven characteristics, a number of underlying empirical indicators can be selected to reflect and represent the condition with respect to that characteristic (Nybø et al., 2019, Table A2). A quantitative estimate of ecosystem condition, for each of these characteristics or as an overall value, emerges from quantitative aggregation across indicators. To allow this aggregation and comparison across indicators and scales, all indicator values are scaled against a reference value and a limit for good ecological condition. The reference condition concept and limits for good ecological condition are detailed in 2.1.1, and the scaling procedure is explained in 2.1.2, followed by a description of indicator calculation and aggregation in 2.1.3.

2.1.1. Reference condition and limits for good ecological condition

All indicators are evaluated against a reference condition, defined as 'intact ecosystems' (sensu Nybø et al., 2019, see also Karr, 1981, Stoddard et al., 2006, EC, 2019). Nybø et al. (2019) define 'intact ecosystems' with respect to recent natural or semi-natural biodiversity and ecosystem functioning. Historic extinctions are not considered, species introduced before 1800 CE are regarded as native, climatic conditions

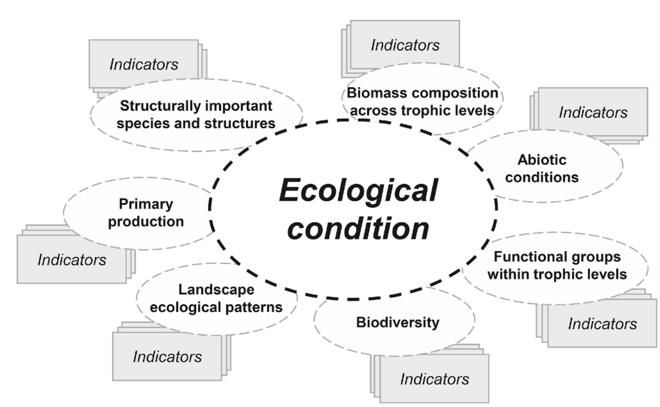

Fig. 1. The Index-Based Ecological Condition Assessment framework (IBECA) can be used to quantify and synthesize ecological condition at three levels: overall ecological condition within an ecosystem (central ellipse), seven ecosystem characteristics (outer ellipses) and individual indicators (stacks of rectangles).

follow the normal period (1961–1990), and modern intensive or large-scale human pressures are absent. Note that human impacts can be present according to this definition of intact ecosystems, but the impact type and intensity should be at a magnitude comparable to that of natural pressures or other organisms. In these systems, human management regimes are present as one among many environmental drivers and processes within an otherwise naturally functioning ecosystem, and are often seen as a historically integral part of the system. To increase policy and management relevance as well as ease of use, a limit for good ecological condition is set for each indicator (cf. Scheffer and Carpenter, 2003, Andersen et al., 2009). On the benign side of that limit "*the ecosystem's structure, function and productivity do not significantly deviate from the reference condition, defined as an intact ecosystem*" (Nybø and Evju, 2017), whereas on the adverse side they do. The normative values for the reference condition and the limit for good ecological condition can be quantitatively determined for each indicator and ecosystem based on empirical data, models, theoretical expectations, and/or expert judgement. For a comprehensive conceptualization of these approaches, see Jakobsson et al. (2020).

2.1.2. Indicator scaling

The raw indicator values are rescaled to allow quantitative comparison and combination of different indicators. IBECA indicator values are normalized by setting the reference values to 1, and the limit for good ecological condition to 0.6 (Fig. 2). The scaled values thus range from 0 to 1 (where 0 represents a real or conceptual boundary for a totally degraded ecosystem, and where values >1 are truncated to 1). Incorporating the limit for good ecological condition thus harmonizes IBECA with the EU Water Framework Directive (WFD) (EC, 2019). Note that the resulting framework is flexible, as it accommodates both linear and non-linear relationships between raw and scaled indicator values, and 'negative' indicators, i.e. when decreasing unscaled values translate into improved ecological condition. Indicators can be 'two-sided', meaning that indicator values both lower and higher than the reference value signify a decrease in ecological condition. In these cases, separate scaling procedures are applied on each side, and only the one with the largest deviation from the reference value is used when assessing the ecological condition (Fig. 2).

2.1.3. Indicator aggregation and uncertainty

Indicator values are aggregated at two levels, i) within each of the seven characteristics of ecosystem condition, and ii) all together for an overall ecosystem condition estimate. IBECA uses a flat aggregation approach (cf. *hierarchical aggregation*, described in 2.2), where the overall assessment is conducted separately from the allocation of indicators into characteristics (Fig. 3). For each aggregation, individual indicator values larger than the reference value (scaled value > 1) are truncated to value 1, and the mean aggregated condition value is extracted.

IBECA uses a bootstrapping approach to approximate uncertainty associated with the data for each indicator, allowing for inclusion of different data types (cf. case study dataset in Table A1). For quantitative datasets, each raw indicator dataset is bootstrapped (n = 10 000), resulting in indicator distributions yielding the median as the indicator estimate and 0.025 and 0.975 quantiles as the 95% confidence interval. If there is no original uncertainty associated with the data (i.e. not estimated from sample-based indicator data), IBECA implements either i) an elicitation process based on expert knowledge and published literature, or ii) a qualitative approximation of proportional (%) uncertainty (Figure A1). No uncertainty is allocated to the reference and limit for good ecological condition values as these are regarded as normative values based on best available knowledge (cf. Certain et al., 2011, Pedersen et al., 2016). To account for uncertainty in aggregated estimates for the seven characteristics and overall ecosystem condition, data values are re-sampled from each indicator distribution. This process is repeated 10 000 times, yielding a distribution of aggregated condition values with the median as the aggregated estimate and 0.025 and 0.975 quantiles as 95% confidence interval of this estimate. Individual indicator values and their associated uncertainty are, however, not truncated in visual presentations (e.g. Fig. 4). Spatial representation or data quality may require weighting of the indicator set of an ecosystem. While weighting was not necessary in our case study (2.2), IBECA can accommodate a weighted mean approach (cf. Certain et al., 2011).

We acknowledge the lack – in nature – of a clear distinction of what constitutes a *good* ecological condition for any given indicator. Within IBECA, the scaled value 0.6 represents the theoretical limit for good ecological condition, and when interpreting the results, we define a 'significant reduction' of ecological condition as values < 0.6 with a 95% CI not overlapping 0.6, and values where the 95% CI overlaps 0.6 as a 'marginal reduction' of ecological condition. Values > 0.6 with a 95% CI not overlapping 0.6, i.e. corresponding to values between the limit for good ecological condition and the reference condition, are referred to as 'good ecological condition' or 'no significant reduction'. These definitions relate to the scaled indicator values, and thus also apply to aggregated values for ecosystem characteristics and the overall assessment of ecosystem condition.

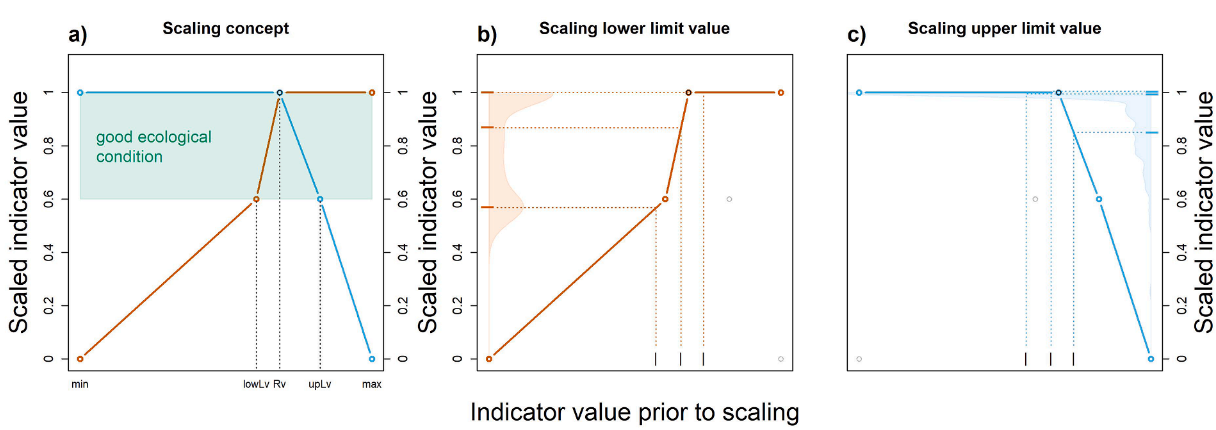

Fig. 2. Scaling of indicator values in relation to the reference value and the limit for good ecological condition. The scaling concept (a) builds on a reference value (Rv), where raw indicator values (x-axis) at Rv are scaled to 1. Furthermore, a lower (lowL) and/or upper (upL) limit (L) for good ecological condition is defined. Raw indicator values within the range of the Rv and L are then linearly scaled to values between 1 and 0.6, defining 'good ecological condition'. Values between min and lowL, or max and UpL, are scaled linearly between 0 and 0.6. Examples of scaling against a lowL (b) and upL (c) are given. For 'two-sided' indicators (i.e. with both a lower and upper limit for identifying a decrease in ecological condition), these separate scaling procedures are applied on each side, and only the one with the largest deviation from the reference value is used when assessing the ecological condition.

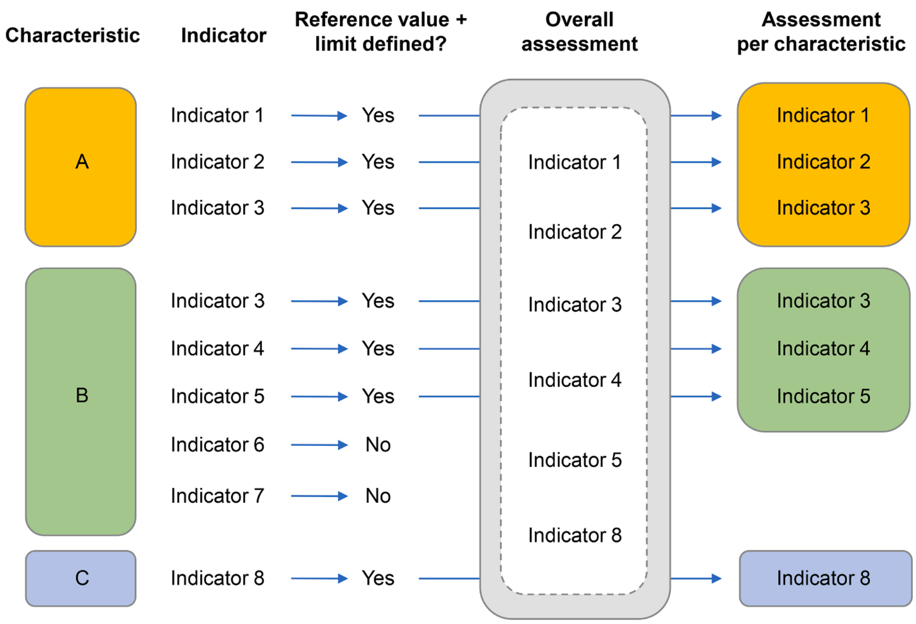

Fig. 3. Conceptual framework of the IBECA approach. Empirical indicators relate to ecosystem characteristics (potentially multiple characteristics per indicator). Development of reference values and limits for good ecological condition is required for the assessment of ecological condition. Rescaling allows for the combination of indicators into aggregated estimates of the seven characteristics and/or an overall assessment of ecological condition. Ecological condition can thus be estimated for i) each indicator, ii) each ecosystem characteristic, and iii) the overall ecosystem (using flat aggregation of indicator values).

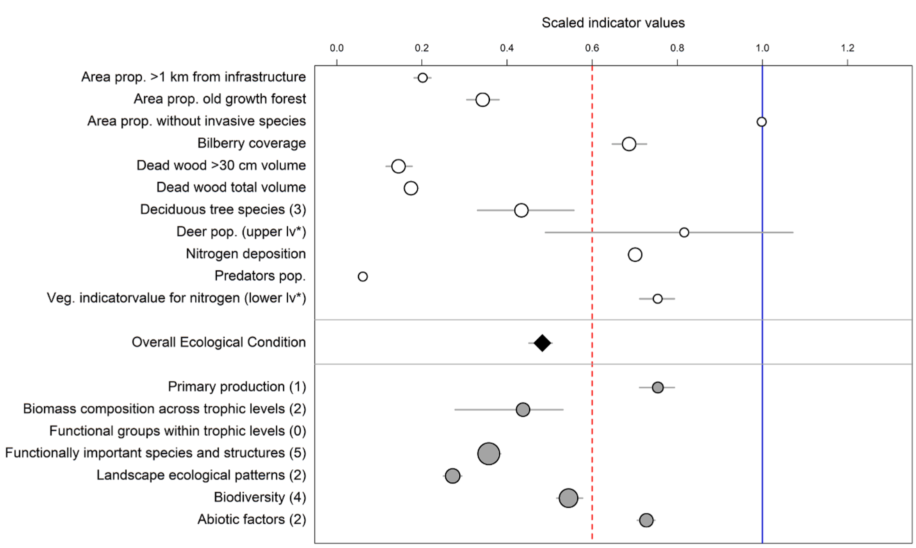

Fig. 4. Quantitative assessment of ecological condition of the forest ecosystem in Trøndelag county, central Norway, with scaled indicator values (circles/diamond), 95% confidence intervals (horizontal error lines), reference level (solid line at 1) and limit for good ecological condition (dashed line at 0.6). Values are given for individual indicators (white circles), overall ecological condition (black diamond) and the seven ecosystem characteristics (grey circles). Size of circles represent data quality (for individual indicators, see 2.2) and data quality combined with number of indicators (for each characteristic). Numbers in brackets after ecosystem characteristic names indicate number of indicators included. * two-sided indicators: lv = limit for good ecological condition, where 'upper' and 'lower' indicate that the indicator estimate either exceeds or undercuts the reference value (cf. Methods). 'Veg.' = Vegetation, 'prop' = proportion. Modified from Nybø et al. (2019).

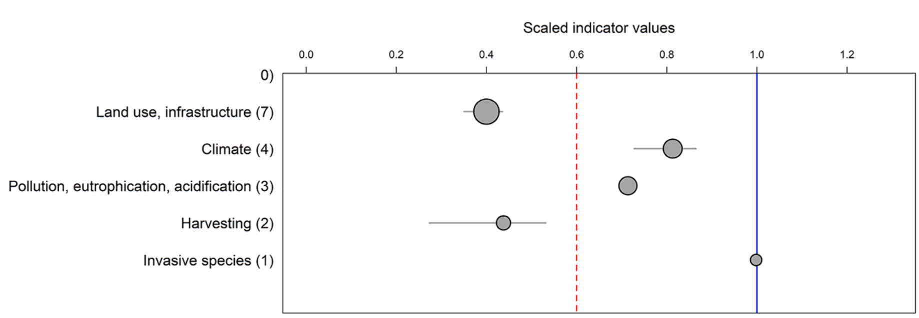

Fig. 5. Visualization of the most prominent pressures on the forest ecosystem condition. IBECA case study indicators were categorized into five pressure categories based on their expected vulnerability to pressures within the study area. Aggregation followed the same procedure as describe above for the IBECA ecosystem characteristics. For figure symbol explanations, see Fig. 4.

2.2. Analysis

First, we performed a case study in which we assessed the ecological condition of forest and alpine ecosystems in Trøndelag county, central Norway, using the IBECA approach (Nybø et al., 2019). This assessment was based on empirical data on eleven indicators for each ecosystem, compiled from the national forest inventory (Tomter et al., 2010), a regional vegetation monitoring project, species-specific monitoring programs, and mapped spatial data to calculate regional indicator values (Table A1). Hence, data were not collected at a common scale with a common methodology, but assembled from different sources as relevant indicators of different characteristics of ecological condition for each of the two ecosystems. Representativity varies, as some data are from established long-term monitoring programs (e.g. dead wood indicators; based on the national forest inventory) whereas other data are from less complete data sources (e.g. area proportion of invasive species; based on the first year of data from a new vegetation monitoring project). We visualize this variation by weighting indicator score circle sizes according to representativity (Figs. 4–7). For an overview of indicators used in the IBECA case study, including reference values and limits for good ecological condition, see Table A1. To supplement this analysis, we identified the pressures with most critical effects on each indicator (cf. Aslaksen et al., 2012), grouped into five pressure categories: land-use/infrastructure; climate; pollution/eutrophication/acidification; exploitation/harvesting; and invasive species. We used this evaluation to re-aggregate indicator values in relation to pressures, using the same methodological approach as for the ecosystem characteristics.

Further, we evaluated the IBECA indicator coverage in relation to EBV (indicators/classes) and ECT (classes), and compared our conceptual framework with ECT. For the indicator coverage evaluation, we re-classified the IBECA case study indicators into the EBV and ECT classes based on existing guidelines (https://geobon.org/ebvs/what-are-ebvs/; Czúcz et al., 2019; Maes et al., 2019). We used a mutually exclusive approach for the ECT classes, allocating each indicator to only one class (Czúcz et al., 2019), whereas each indicator could be allocated to several EBV classes (cf. Vihervaara et al., 2017). In addition to the existing indicators, the comparison included a set of candidate IBECA-indicators where reference values and limits for good ecological condition are currently lacking.

Lastly, we recalculated aggregated index values for the two ecosystems based on EBV and ECT classifications to quantitatively investigate how our indicator set behaved using these classification systems. In addition, we tested the hierarchical aggregation suggested for the ECT (Czúcz et al., 2019) for our original ecosystem characteristics classification, the EBV classification and the ECT classification. The hierarchical aggregation was done in a two-step process where indicator values were first aggregated into indicator classes and then aggregated into an overall ecological condition assessment, as opposed to the flat aggregation approach used for the overall assessment of ecological condition in IBECA.

We conducted all indicator calculations, aggregations and visualizations using R (R Core Team, 2018), including packages *gamlss.dist* (Stasinopoulos and Rigby, 2018), *dplyr* (Wickham et al., 2019) and *Hmisc* (Harrell, 2018).

3. Results

3.1. Results from the case study

Four of the eleven ecosystem indicators in the forest ecosystem indicated 'good' ecological condition, whereas six indicated 'significantly reduced' and one indicated 'marginally reduced' ecological condition (Fig. 4). The aggregated overall ecological condition was 'significantly reduced' from the reference condition, to a value of 0.48 (95% CI: 0.45–0.51). Five out of seven ecosystem characteristics indicated a 'significantly reduced' ecological condition, whereas no significant reduction was detected for *primary production* and *abiotic factors*. Land use/infrastructure and harvesting were the main pressures underlying these patterns in the ecological condition in the forest ecosystem (Fig. 5).

Five of the eleven indicators in the alpine ecosystem indicated 'good' ecological condition, three indicated 'marginally reduced' and three indicated 'significantly reduced' ecological condition. Among the 'marginally reduced' indicator values, the rodent populations had a low estimate but a large confidence interval. The aggregated overall ecological condition was 'marginally reduced' from the reference condition (0.61, 95% CI: 0.58–0.67). Within the alpine ecosystem, *primary production*, *landscape ecological patterns*, and *abiotic factors* were all in 'good' ecological condition, *biodiversity* and *functionally important species* were 'marginally reduced', whereas *biomass composition across trophic levels* were in 'significantly reduced' condition. We lacked data on indicators representing *functional groups within trophic levels* in both ecosystems (Fig. 6). Harvesting, and to some extent climate change, were the main underlying pressures on ecological condition in the alpine ecosystem. In contrast to the forest ecosystem, pressures from land use and infrastructure have not impacted the alpine indicators negatively (Fig. 7). For details on indicators and allocation of indicators across ecosystem characteristics, see Table A1.

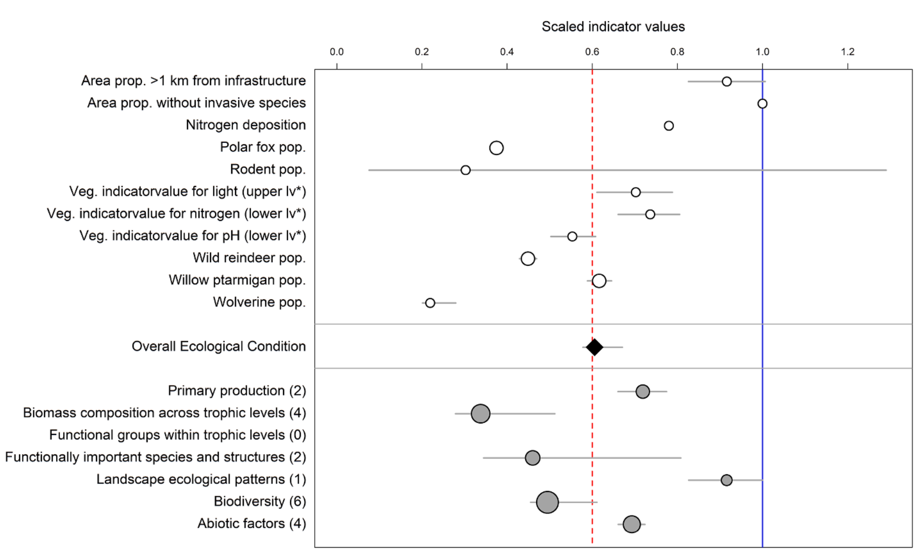

Fig. 6. Quantitative assessment of ecological condition of the alpine ecosystem in Trøndelag county, central Norway, with scaled indicator values (circles/diamond), 95% confidence intervals (horizontal error lines), reference level (solid line at 1) and limit for good ecological condition (dashed line at 0.6). Values are given for individual indicators (white circles), overall ecological condition (black diamond) and ecosystem characteristics (grey circles). Size of circles represent data quality (for individual indicators, see 2.2) and data quality combined with number of indicators (for each characteristic). Numbers in brackets after ecosystem characteristic names indicate number of indicators included. * two-sided indicators: lv = limit for good ecological condition, where 'upper' and 'lower' indicate that the indicator estimate either exceeds or undercuts the reference value (cf. Methods). 'Veg.' = Vegetation, 'prop' = proportion. Modified from Nybø et al. (2019).

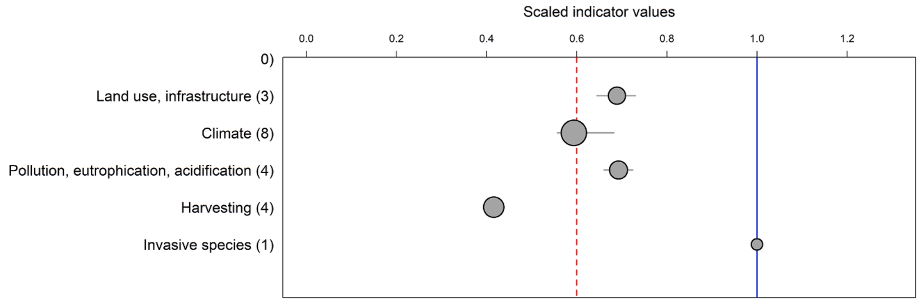

Fig. 7. Visualization of the most prominent pressures on the alpine ecosystem condition. IBECA case study indicators were categorized into five pressure categories based on their expected vulnerability to pressures within the study area. Aggregation followed the same procedure as describe above for the IBECA ecosystem characteristics. For figure symbol explanations, see Fig. 6.

Table 1
Conceptual links between IBECA characteristics and EBV and ECT classes.

IBECA characteristic	EBV classes	ECT classes
Primary production Abiotic factors	Ecosystem function	Functional state Physical state[1] Chemical state
Biodiversity	Species populations Community composition[1] Genetic composition[1]	Compositional state
Functionally important species and structures	Species populations Ecosystem structure	Compositional state Structural state
Functional groups within trophic levels[1]	Species populations Species traits[1]	Compositional state Structural state
Biomass composition across trophic levels	Species populations	Compositional state Structural state
Landscape ecological patterns	Ecosystem structure	Overall landscape characteristic[1,2] Ecosystem type specific landscape characteristic

[1]No indicators covering that class or characteristic in the case study dataset.
[2]Not relevant for the case study, as all assessments were conducted at the ecosystem type level.

3.2. Cross-comparison of IBECA and international frameworks

Conceptually, the IBECA characteristics *primary production* and *landscape ecological patterns* have their corresponding categories in both EBV and ECT, although they form sub-categories of ecosystem functions/processes and structure (Table 1). Similarly, the *biodiversity* characteristic corresponds to the ECT class *compositional state*, whereas it is matched by three EBV classes. The IBECA ecosystem characteristic *abiotic factors* has no corresponding EBV class but is similar to the ECT classes *physical state* and *chemical state*. In addition, the ECT class *overall landscape* has no counterpart in the current version of IBECA as all indicators are ecosystem specific. The remaining three IBECA characteristics (functional groups, biomass composition, and functionally important species and structures) correspond to two ECT classes (*compositional state* and *structural state*) and to the EBV class *species populations*. In addition, the EBV class *ecosystem structure* logically matches the IBECA characteristic *functionally important species and structures*, while the EBV class *species traits* is theoretically linked to the IBECA characteristic *functional groups within trophic levels* (Table 1).

Concerning specific indicators, the IBECA case study indicators covered the EBV classes *genetic composition*, *species traits* and *community composition* (Tables 2, 3). The indicators “area proportion > 1 km from infrastructure”, “dead wood” and “old growth forest” indicators in the forest ecosystem contributed to the class *ecosystem structure*. The EBV class *ecosystem function* was only represented by vegetation indicators (for light/nitrogen). It is worth noting that we find no appropriate class for our indicator on nitrogen deposition, although the critical loads estimates, on which the limits for good ecological condition are based, relate to nitrogen effects on vegetation (cf. Bobbink and Hettelingh, 2011, Austnes et al., 2018) (Tables 2 and 3).

Both the EBV class *species populations* and the ECT class *compositional state* corresponded well with the IBECA characteristic *biodiversity*, including indicators of mammal, bird and plant species (Tables 2, 3). The ECT classes *chemical state* as well as *ecosystem landscape* were represented by identical indicators such as IBECA characteristics *abiotic factors* (vegetation indicators for light and nitrogen, and nitrogen deposition) and *landscape ecological patterns* (area proportion > 1 km from infrastructure, and old growth forest). The IBECA case study used no indicators linked to the ECT class *physical state*. Furthermore, the ECT class *structural state* was only represented within the forest ecosystem, by the two dead wood indicators and area proportion of old growth forest.

3.3. Recalculations of ecological condition based on different classifications

Applying the ECT and EBV classifications both yield similar results to IBECA (Fig. 8). The IBECA *biodiversity* indicators are identical for the ECT *compositional state* and the EBV *species populations* classes. Furthermore, the ECT *landscape characteristic* class largely resembles the EBV *ecosystem structure*, both showing a ‘significantly reduced’ condition in the forest ecosystem driven by the area proportion >1 km from infrastructure and old growth forest indicators. The ECT allocates the forest dead wood indicators into a specific *structural state* class. This *structural state* class does not further disentangle what type of vegetation is considered, as opposed to the IBECA that includes a specific characteristic for *functionally important species and structures* for these types of key structures of an ecosystem. In addition, the ECT *chemical state* class reveals a similar estimate as the EBV *ecosystem function*, but with higher precision due to more indicators included.

IBECA showed a ‘significantly reduced’ overall ecological condition for the forest ecosystem, whereas the overall ecological condition for the alpine ecosystem was ‘marginally reduced’ (see 3.1). In those original calculations we used a flat aggregation approach. When instead adopting a hierarchical aggregation approach, the general interpretation of the overall ecological condition did not change noteworthy. Similarly, applying the hierarchical approach on the EBV and ECT classifications hardly changed the interpretation for the forest ecosystem (Fig. A2). In contrast, a 'good' condition value was found for the alpine ecosystem using the hierarchical approach, based on both the EBV and the ECT classifications. The recalculations of overall ecological condition were limited because of lack of indicators for three out of six EBV classes and three (forest ecosystem) to four (alpine ecosystem) out of seven ECT classes within the case study.

4. Discussion

4.1. The IBECA framework

In this study, we present a transparent and flexible index-based framework for assessing ecological condition and test it for forest and alpine ecosystems in central Norway. The analysis suggests that the forest ecosystem in the region has a 'significantly reduced' ecological condition, primarily driven by unfavorable landscape ecological patterns as a result of infrastructure development, loss of important structures due to forest management and alternative land use, and low populations of large predators because of high harvesting rates. The alpine ecosystem is in a marginally better condition, mainly explained by the lower impact of infrastructure expansion in the alpine areas of the region. We acknowledge that a more exhaustive coverage of indicators with respect to ecosystem characteristics would be desirable. Nevertheless, the framework and applied indicators communicate a picture of the ecological condition of these ecosystems within the study region that is consistent with the experts' qualitative assessments and the Norwegian Nature Index (cf. https://naturindeks.no/). We exemplify how the concept of ecosystem resilience can be used to assess multi-level condition estimates that are comparable across ecosystems, ecosystem characteristics and individual indicators in a transparent way (cf. Folke et al., 2010).

One of the key concepts – and thus also key challenges – within the IBECA framework is the use of a reference condition defined as intact ecosystems with negligible human impact (Karr, 1981, Stoddard et al., 2006). Many other assessments and monitoring projects make use of baseline years instead of the conceptual idea of a reference *condition* (e.g. EEA, 2012; EBCC, 2019; cf. discussion in Keith et al., 2019), where subsequent estimations of ecological condition are assessed relative to the condition in the baseline year. Such shifting – or different – baselines (cf. Soga and Gaston, 2018) are straight-forward to establish and use, but do not allow quantitative comparison of ecological condition assessments across geographical areas. We therefore argue that the use of baseline years are undesirable, in particular when historical data are limited (Collins et al., 2020). For example, areas that were in a poor ecological condition in the baseline year can show positive trends in subsequent years, despite being in a poorer 'absolute' condition than sites that were in a good condition in the baseline year and where no trend is apparent. The IBECA approach builds upon the previously developed Norwegian Nature Index (Certain et al., 2011; Pedersen et al., 2016) and, by using limits for good ecological condition, also harmonizes with well-developed concepts under-pinning the EU Water Framework Directive (EC, 2019). These two scaling parameters enable direct evaluations of single snapshot assessments of ecological condition, allowing the detection of critical ecological changes at an early stage and comparing of sites, systems and regions in absolute terms. For a thorough discussion on setting reference values and limits for good ecological condition, see Jakobsson et al. (2020).

4.2. Framework comparison

IBECA assesses an ecosystem's structure and function (including productivity) with respect to seven ecosystem characteristics (Nybø and Evju, 2017), and we specifically collated our indicators to address these characteristics. Logically, reclassifying the indicator set according to other frameworks results in incomplete indicator coverage for several classes, but also highlights differences that are key to inform the indicator framework development for ecological condition assessments.

Comparing the IBECA indicator set and EBV, our study confirms the challenges in obtaining genetic and trait-based indicators for terrestrial ecosystems (Geijzendorffer et al., 2016; Vihervaara et al., 2017; Turak et al., 2017). Incorporating genetics (Schwartz et al., 2007) and traits (Kissling et al., 2018) in monitoring programs has been suggested, and in particular, the rise of open trait databases now offer promising opportunities (Gallagher et al., 2020). The EBV class *ecosystem structure* largely relies on indicators of the extent of various types of vegetation (Pereira et al., 2013), but we exclusively used structure indicators linked to what we regard being indicators of the condition *per se* (i.e. not the extent of general vegetation types). Although Vihervaara et al. (2017) included nutrient content indicators within the EBV class *ecosystem structure*, we argue that our vegetation indicators for light and nitrogen should instead be incorporated as proxies for *ecosystem function* (primary production). In general, our definition of abiotic factors does not suit the EBV classification system due to the biodiversity focus within the EBV, and hence our indicator of nitrogen deposition did not match any EBV class. Nitrogen deposition could indeed be questioned as a condition indicator (cf. Czúcz et al., 2019), but the good ecological condition limit for this indicator is based on experimental research on the sensitivity of vegetation to nitrogen fertilization (Bobbink and Hettelingh, 2011, Austnes et al., 2018), and hence our scaled indicator represents the condition of vegetation. In our case study, we also used a vegetation-derived nitrogen indicator (Ellenberg values; Töpper et al., 2018). Our results show that values of the two nitrogen indicators match well at the regional scale of our case study, despite completely different data collection methods, resolution and geographical coverage. However, because of the large spatial coverage of the nitrogen deposition indicator (Austnes et al., 2018), we encourage future use of it as an indirect indicator of ecological condition, at least as a supplement to indicators directly representing ecological condition. We achieved good indicator coverage for the EBV class *species populations* because of the close link to the many IBECA *biodiversity* indicators, but none of those indicators fit with the EBV *community composition* class. Future bird, insect and plant community indices will improve this composition criterion, which forms an important measure of biodiversity beyond single species populations (see 4.4). The last EBV class, *ecosystem function,* is difficult to conceptually separate from other classes based on current data (Czúcz et al., 2019), and more ecosystem functioning focused monitoring would be needed to make up a more complete indicator set for this class (cf. Mononen et al., 2016).

In contrast to IBECA, the ECT framework is based on mutually exclusive indicator classes, i.e. one indicator cannot be part of more than one class. Hence, the failure to capture indicators within the ECT class *functional state* was not surprising as it has already been highlighted that many potential EBV indicators fall into other ECT classes by definition (Czúcz et al., 2019). Concerning the *structural state* class, we acknowledge the lack of data based on remote sensing in the IBECA case study and believe that future development of the indicator set will help fill this gap (see 4.4). The remaining three classes of the ECT were well represented by the IBECA case study indicator set, which is not surprising with the high degree of characteristic/class overlap (i.e. we selected case study indicators to represent the IBECA characteristics, such that indicators for characteristics that resemble the ECT classes naturally cover also these classes). Currently, the ECT class *compositional state* represents the bulk part of the indicator set, among which many are directly or indirectly regulated by harvesting. We emphasize that using indicators sensitive to different pressures provides an opportunity to address the pressure-state relationship in the ecosystems, for example with sub-analyses on which pressures most strongly influence the ecosystem's

Table 2
Cross-comparison of the indicators used in the IBECA case study (including their allocation to ecosystem characteristics, and indicators that are suggested for the next update of the IBECA) and the EBV and ECT classifications, respectively, for the forest ecosystem. EBV classes sub-divided into the candidates for each class (GEO BON 2019), preceded by abbreviations of the classes: Genetic Composition (GC), Species Populations (SP), Species Traits (ST), Community Composition (CC), Ecosystem Structure (ES) and Ecosystem Function (EF). ECT classes are preceded by abbreviations of the three super-classes (Czúcz et al. 2019): Abiotic ecosystem characteristics (A), Biotic ecosystem characteristics (B) and Landscape and seascape characteristics (L). Shaded × = indicators used in the case study, non-shaded × = suggested future indicators, (x) = potential connection to indicator class.

IBECA

Indicator	Primary production	Biomass composition across trophic levels	Functional groups within trophic levels	Functionally important species and structures	Landscape ecological patterns	Biodiversity	Abiotic factors
Area prop. without invasive species						x	
Area prop. >1 km from infrastructure					x		
Deer species population levels		x					
Predators population levels		x				x	
Area prop. old growth forest				x	x		
Bilberry coverage				x		x	
Dead wood total volume				x			
Dead wood >30 cm volume				x			
Nitrogen deposition							x
Indicator value of vegetation for nitrogen	x						x
Deciduous tree species (3 species)				x		x	
(NDVI index)	x						
(Standing tree biomass)	x			x			
(Vegetation biomass)	x	x					
(Tree species/types composition)			x				
(Green infrastructure)					x		
(Bird index indicator)			x			x	
(Insect index indicator)			x			x	
(Plant index indicator)			x			x	

EBV

Indicator	GC. Co-ancestry	GC. Allelic diversity	GC. Population genetic differentiation	GC. Breed and variety diversity	SP. Species distribution	SP. Population abundance	SP.Population structure by age/size class	ST. Phenology	ST. Morphology (excl. biomass)	ST. Reproduction	ST. Movement	ST. Physiology	CC. Taxonomic diversity	CC. Species interactions	ES. Habitat structure/condition	ES. Ecosystem extent and fragmentation	ES. Ecosystem composition by functional type	EF. Net primary productivity	EF. Secondary productivity	EF. Nutrient retention	EF. Disturbance regime
Area prop. without invasive species					-	x															
Area prop. >1 km from infrastructure																x					
Deer species population levels						x															
Predators population levels						x															
Area prop. old growth forest																x					
Bilberry coverage						x															
Dead wood total volume															x		-				
Dead wood >30 cm volume															x		-				
Nitrogen deposition																				-	
Indicator value of vegetation for nitrogen															-			x		-	
Deciduous tree species (3 species)						x									x						
(NDVI index)																		x			
(Standing tree biomass)															(x)			x			
(Vegetation biomass)															(x)			x			
(Tree species/types composition)													x		x						
(Green infrastructure)																x					
(Bird index indicator)						x	(x)						x						(x)		
(Insect index indicator)						x	(x)						x						(x)		
(Plant index indicator)						x	(x)						x								

ECT

Indicator	A. Physical state	A. Chemical state	B. Compositional state characteristics	B. Structural state characteristics	B. Functional state characteristics	L. Landscape	L. Ecosystem type specific landscape
Area prop. without invasive species			x				
Area prop. >1 km from infrastructure							x
Deer species population levels			x				
Predators population levels			x				
Area prop. old growth forest							x
Bilberry coverage			x				
Dead wood total volume				x			
Dead wood >30 cm volume				x			
Nitrogen deposition		x					
Indicator value of vegetation for nitrogen		x					
Deciduous tree species (3 species)			x				
(NDVI index)				x			
(Standing tree biomass)				x			
(Vegetation biomass)				x			
(Tree species/types composition)			x				
(Green infrastructure)							x
(Bird index indicator)			x				
(Insect index indicator)			x				
(Plant index indicator)			x				

condition (Figs. 5 and 7).

The EBV classification scheme to a large extent corresponds to the IBECA approach, emphasizing main ecological processes. However, as the two schemes emphasize different aspects of ecological processes within ecosystems, applying the EBV classification to the IBECA case study data results in scarcely covered EBV classes. In particular, indicators of genetics and traits were lacking within the case study data, while some data ended up unused in the reclassification. The ECT classification is more indicator type-based compared to IBECA. In addition, allocating each indicator to a unique class adds a substantial risk of erroneously denying the potential relevance of single indicators for several aspects, functions and processes related to an ecosystem.

4.3. Recalculations/communication

Recalculating the IBECA results based on alternative classification frameworks highlighted some key differences between these frameworks. For example, the landscape category used both for the IBECA characteristics and the ECT classes is absent from the EBV classification, blurring the difference between the two ecosystems concerning *landscape ecological patterns*. Nevertheless, given the dataset used, all the frameworks generate similar assessments of the condition for the following general ecosystem aspects: species indicators/biodiversity ('marginally reduced'), landscape/ecosystem structure ('no significant reduction' for the alpine ecosystem, 'significantly reduced' for the forest ecosystem), and abiotic conditions/functions ('no significant reduction'). The clearest difference that emerges from the reclassification exercise conducted in this study relates to the representation of structural attributes of an ecosystem. While EBV uses a general *ecosystem structure* class, ECT separates local and landscape scale structural attributes. Hence, adopting the ECT classification highlights that the low condition of the EBV *ecosystem structure* is not only a result of landscape scale infrastructure expansion, but also of the pressure from forest management practices (e.g. on the dead wood indicators). However, the IBECA framework increases the resolution concerning the structural components even more (besides local vs. landscape scale structural attributes) by splitting up the ecosystem characteristics in different types of species and structures, including allocating key intra-trophic and inter-trophic links into separate characteristics.

Concerning the overall ecosystem condition assessment, we noticed substantial effects of using hierarchical aggregation instead of flat aggregation of indicators for the alpine ecosystem because of unbalanced indicator classification. In particular, the *landscape characteristic* (ECT) and *ecosystem structure* (EBV) class scores were driven exclusively by the area proportion >1 km from infrastructure indicator. Being the only indicator in its class, it received one third of the weighting when calculating the overall ecosystem condition based on three classes. This effect of hierarchical aggregation highlights how sensitive ecological condition assessment is to indicator deficiencies within classes; under-represented indicator classes with few indicators will result in

Table 3
Cross-comparison of the indicators used in the IBECA case study (including their allocation to ecosystem characteristics, and indicators that are suggested for the next update of the IBECA) and the EBV and ECT classifications, respectively, for the alpine ecosystem. EBV classes sub-divided into the candidates for each class (GEO BON 2019), preceded by abbreviations of the classes: Genetic Composition (GC), Species Populations (SP), Species Traits (ST), Community Composition (CC), Ecosystem Structure (ES) and Ecosystem Function (EF). ECT classes are preceded by abbreviations of the three super-classes (Czúcz et al. 2019): Abiotic ecosystem characteristics (A), Biotic ecosystem characteristics (B) and Landscape and seascape characteristics (L). Shaded × = indicators used in the case study, non-shaded × = suggested future indicators, (x) = potential connection to indicator class.

	IBECA							EBV																					ECT						
Indicator	Primary production	Biomass composition across trophic levels	Functional groups within trophic levels	Functionally important species and structures	Landscape ecological patterns	Biodiversity	Abiotic factors	GC. Co-ancestry	GC. Allelic diversity	GC. Population genetic differentiation	GC. Breed and variety diversity	SP. Species distribution	SP. Population abundance	SP.Population structure by age/size class	ST. Phenology	ST. Morphology (excl. biomass)	ST. Reproduction	ST. Movement	ST. Physiology	CC. Taxonomic diversity	CC. Species interactions	ES. Habitat structure/condition	ES. Ecosystem extent and fragmentation	ES. Ecosystem composition by functional type	EF. Net primary productivity	EF. Secondary productivity	EF. Nutrient retention	EF. Disturbance regime	A. Physical state	A. Chemical state	B. Compositional state characteristics	B. Structural state characteristics	B. Functional state characteristics	L. Landscape	L. Ecosystem type specific landscape
Area prop. without invasive species						x						-	x																		x				
Area prop. >1 km from infrastructure					x																		x												x
Polar fox population levels		x				x							x								-										x				
Wolverine population levels		x				x							x																		x				
Willow ptarmigan population levels				x		x							x																		x				
Rodent population levels		x		x		x							x								-										x				
Wild reindeer population levels		x				x							x																		x				
Nitrogen deposition							x																				-			x					
Indicator value of vegetation for light	x						x															-			x					x					
Indicator value of vegetation for nitrogen	x						x															-			x		-			x					
Indicator value of vegetation for pH							x															-					-			x					
(NDVI index)	*x*																								x							x			
(Vegetation biomass)	*x*	*x*																				(x)			x							x			
(Wild reindeer habitat connectivity)					*x*																		x												x
(Bird index indicator)			*x*			*x*							x	(x)						x						(x)					x				
(Insect index indicator)			*x*			*x*							x	(x)						x						(x)					x				
(Plant index indicator)			*x*			*x*							x	(x)						x											x				

disproportional contribution of those few indicators to the overall ecological condition score. The opposite applies to the flat aggregation proposed within the IBECA; characteristics with many indicators will be over-represented in the overall condition assessment. Hence, we stress the importance of considering such challenges when defining classification typologies, where the final choice of approach is dependent on the purpose of the condition assessment. In our case with IBECA, the primary goal was to provide a transparent overview of the complexity behind the condition of ecosystems. Therefore, we provide results for three levels of ecological condition: i) individual indicator values (non-truncated), ii) index values for each ecosystem characteristic, and iii) an overall ecological condition assessment.

4.4. Future directions

Continued and improved monitoring of ecological indicators is essential to understand changes in ecosystem condition, and thus to address targeted and cost-effective measures to mitigate negative trends for biodiversity (Tittensor et al., 2014). Based on our results we emphasize key steps forward concerning regional and national level indicator data for assessing ecological condition. First, the majority of the indicators within the IBECA case study were species indicators based on measures of abundance of species and structures well represented by ongoing monitoring programs. For future ecological condition updates, a key step for better representation within and across trophic levels is to develop additional indicators based on species community data (Nybø et al., 2019). Second, better representation of ecosystem functions and processes as well as structural connectivity is needed. We believe the future development of the IBECA framework will improve the robustness of the condition assessments by incorporating additional relevant indicators representing these ecosystem characteristics. Here, an essential step forward is to capitalize on the opportunities that remote sensing technology offers (Vihervaara et al., 2015; 2017;; Sverdrup-Thygeson et al., 2016). Third, our use of vegetation-based indicators for light and nitrogen (see Töpper et al., 2018), exemplifies a novel, feasible and scientifically sound opportunity for improved ecological condition assessments, with three main advantages: i) they form a proper representation of condition *per se*, ii) they can easily be recorded simultaneously with other indicators, and iii) they are based on transparent and repeatable approaches for setting reference values and limits for good ecological condition (Jakobsson et al., 2020). We therefore stress the benefit of such types of vegetation indicators linked to species' response to the environment also in other ecological condition assessments.

Future development of indicators based on bird, insect and vegetation monitoring will include indicators linked to the EBV class *community composition* (potentially also *species traits* through indirect estimations, e.g. by using trait databases), whereas *genetic composition* most likely will remain unrepresented (Tables 2 and 3). These indicators will logically also increase the representativity of the ECT class *compositional state*. Species-based indicators like these were in the case study represented by mammal and bird species indicators in the alpine

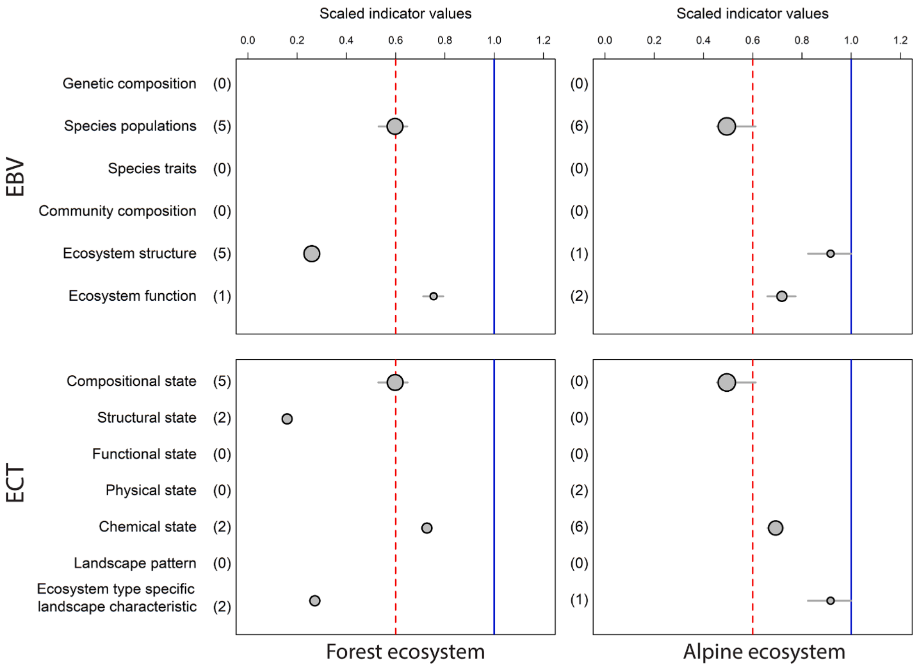

Fig. 8. The effect of different classification systems on aggregated estimates of indicator classes applying the EBV (upper) or the ECT (lower) classifications for the forest (left) and alpine ecosystems (right), using the IBECA case study indicators. Classification of indicators as in Tables 2 and 3, number of indicators in each class are given in brackets on the y-axis. Blue vertical line = scaled reference value (1), dashed red vertical line = scaled limit for good ecological condition (0.6). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

ecosystem, and mammal and plant species in the forest ecosystem, and mainly represent species directly or indirectly regulated by harvesting. The proposed future indicators of plant, insect and bird communities will represent other aspects of biodiversity, not directly affected by harvesting, and hence improve analyses on the effects of different pressures on ecological condition across indicators. Development of new indicators based on remote sensing will mainly contribute to the ECT classes *structural state* and *landscape characteristic* and the EBV classes *ecosystem function* and *ecosystem structure*, hence filling important gaps in the IBECA case study dataset in relation to ECT and EBV (Tables 2 and 3).

5. Conclusions

In this paper, we provide an example of an index-based ecological condition assessment tested out in practice based on available data. Related to, but not fully aligned with current international frameworks, it challenges key concepts of these frameworks for national level implementation. IBECA illustrates the need for carefully choosing appropriate classification and aggregation approaches, but also highlights the value of flexible approaches that facilitate adjustments and recalculations, as exemplified by our analytical exercise. We identify three main strengths of the index-based IBECA approach. First, a transparent outcome in terms of standardized quantitative estimates of ecological condition that can be compared across ecosystems and indicators, giving it high management relevance, e.g. for prioritization purposes. Second, new and improved monitoring data (of existing or new indicators), or updated knowledge of reference conditions and limits for good ecological condition, can easily be added to the assessment framework and can be used to update calculations back in time for trend analyses. Third, the overarching framework is flexible, and updating, for example, the aggregation approach will have negligible effects on other parts of the analytical process, making IBECA a cost-effective approach for ecological condition assessments.

CRediT authorship contribution statement

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

Allentoft, M.E., O'Brien, J., 2010. Global amphibian declines, loss of genetic diversity and fitness: a review. Diversity 2:47-71.

Andersen, T., Carstensen, J., Hernández-García, E., Duarte, C.M., 2009. Ecological thresholds and regime shifts: approaches to identification. Trends Ecol. Evol. 24 (1), 49–57. https://doi.org/10.1016/j.tree.2008.07.014.

Aslaksen, I., Framstad, E., Garnåsjordet, P.A., Nybø, S., Skarpaas, O., 2012. Knowledge gathering and communication on biodiversity: developing the Norwegian Nature Index. Norsk Geografisk Tidsskrift – Norwegian J. Geography 66 (5), 300–308. https://doi.org/10.1080/00291951.2012.744092.

Austnes, K., Lund, E., Sample, J.E., Aarrestad, P.A., Bakkestuen, V., Aas, W., 2018. Overskridelser av tålegrenser for forsuring og nitrogen for Norge. Oppdatering med perioden 2012–2016 (Exceedances of critical loads for acidification and nitrogen in Norway. Update including 2012–2016). NIVA Rapport 7239-2018, Miljødirektoratet M-966 | 2018.

Biggs, R., Schlüter, M., Biggs, D., Bohensky, E.L., BurnSilver, S., Cundill, G., Dakos, V., Daw, T.M., Evans, L.S., Kotschy, K., Leitch, A.M., Meek, C., Quinlan, A., Raudsepp-Hearne, C., Robards, M.D., Schoon, M.L., Schultz, L., West, P.C., 2012. Toward principles for enhancing the resilience of ecosystem services. Annu. Rev. Environ. Resour. 37 (1), 421–448. https://doi.org/10.1146/annurev-environ-051211-123836.

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 (RIUM), www.rivm.nl.cce.

CBD, 2011. Convention on Biological Diversity: Strategic Plan for Biodiversity 2011-2020, Including Aichi Biodiversity Targets. https://www.cbd.int/sp/.

CBD, 2020. Convention on Biological Diversity: Zero draft of the post-2020 global biodiversity framework. CBD/WG2020/2/3, 6 January 2020. Retrieved 27 Jan 2020.

Certain, G., Skarpaas, O., Bjerke, J-W., Framstad, E., Lindholm, M., Nilsen, J-E., Norderhaug, A., Oug, E., Pedersen, H-C., Schartau, A-K., van der Meeren, G.I., Aslaksen, I., Engen, S., Garnåsjordet, P-A., Kvaløy, P., Lillegård, M., Yoccoz, N.G., Nybø, S., 2011. The Nature Index: A General Framework for Synthesizing Knowledge on the State of Biodiversity. PloS ONE 6:e18930.

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.

Czúcz, B., Keith, H., Jackson, B., Maes, J., Driver, A., Nicholson, E., Bland, L., 2019. Discussion paper 2.3: Proposed typology of condition variables for ecosystem accounting and criteria for selection of condition variables. Paper submitted to the SEEA EEA Technical Committee as input to the revision of the technical recommendations in support of the System on Environmental-Economic Accounting. Version of 18 October 2019. 27 pp.

Diaz, S., Fargione, J., Chapin III, S., Tilman, D. 2006. Biodiversity loss threatens human well-being. PLoS Biology 4:e277.

EBCC. 2019. European Bird Census Council. http://www.ebcc.info. Retrieved 2 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 Retrieved 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.

Folke, C., Carpenter, S.R., Walker, B., Scheffer, M., Chapin, T., Rockström, J., 2010. Resilience thinking: integrating resilience, adaptability and transformability. Ecol. Soc. 15, 20.

Gallagher, R.V., Falster, D.S., Maitner, B.S., Salguero-Gómez, R., Vandvik, V., Pearse, W. D., et al., 2020. Open Science principles for accelerating trait-based science across the Tree of Life. Nat. Ecol. Evol. 4 (3), 294–303. https://doi.org/10.1038/s41559-020-1109-6.

Geijzendorffer, I.R., Regan, E.C., Pereira, H.M., Brotons, L., Brummitt, N., Gavish, Y., Haase, P., Martin, C.S., Mihoub, J.-B., Secades, C., Schmeller, D.S., Stoll, S., Wetzel, F.T., Walters, M., Cadotte, M., 2016. Bridging the gap between biodiversity data and policy reporting needs: an essential biodiversity variables perspective. J. Appl. Ecol. 53 (5), 1341–1350. https://doi.org/10.1111/1365-2664.12417.

GEO BON, 2019. Essential Biodiversity Variables. https://geobon.org/ebvs/what-are-ebvs/ Retrieved 2 Dec 2019.

Harrell F.E.Jr., 2018.; with contributions from Charles Dupont and many others. Hmisc: Harrell miscellaneous. R package version 4.1-1.

Hein, L., Bagstad, K.J., Obst, C., Edens, B., Schenau, S., Castillo, G., Soulard, F., Brown, C., Driver, A., Bordt, M., Steurer, A., Harris, R., Caparrós, A., 2020. Progress in natural capital accounting for ecosystems. Science 367 (6477), 514–515. https://doi.org/10.1126/science:aaz8901.

IPBES, 2019. Summary for Policymakers of the Global Assessment Report on Biodiversity and Ecosystem Services. Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES), Paris.

Jakobsson, S., Töpper, J.P., Evju, M., Framstad, E., Lyngstad, A., Pedersen, B., Sickel, H., Sverdrup-Thygeson, A., Vandvik, V., Velle, L.G., Aarrestad, P.A., Nybø, S., 2020. Setting reference levels and limits for good ecological condition in terrestrial ecosystems – experiences from practical implementation. Ecol. Ind. 116, 106492.

Karr, J.R., 1981. Assessment of biotic integrity using fish communities. Fisheries 6 (6), 21–27. https://doi.org/10.1577/1548-8446(1981)006<0021:AOBIUF>2.0.CO;2.

Keith, H., Maes, J., Czúcz, B., Jackson, B., Driver, A., Bland, L., Nicholson, E., 2019. Discussion paper 2.1: Purpose and role of ecosystem condition accounts. Paper submitted to the SEEA EEA Technical Committee as input to the revision of the technical recommendations in support of the System on Environmental-Economic Accounting. Version 5 September 2019. 50 pp.

Kissling, W.D., Walls, R., Bowser, A., Jones, M.O., Kattge, J., Agosti, D., Amengual, J., Basset, A., van Bodegom, P.M., Cornelissen, J.H.C., Denny, E.G., Deudero, S., Egloff, W., Elmendorf, S.C., Alonso García, E., Jones, K.D., Jones, O.R., Lavorel, S., Lear, D., Navarro, L.M., Pawar, S., Pirzl, R., Rüger, N., Sal, S., Salguero-Gómez, R., Schigel, D., Schulz, K.-S., Skidmore, A., Guralnick, R.P., 2018. Towards global data products of Essential Biodiversity Variables on species traits. Nat. Ecol. Evol. 2 (10), 1531–1540. https://doi.org/10.1038/s41559-018-0667-3.

Mace, G.M., Barrett, M., Burgess, N.D., Cornell, S.E., Freeman, R., Grooten, M., Purvis, A., 2018. Aiming higher to bend the curve of biodiversity. Nat. Sustainability 1, 448–451.

Maes, J., Driver, A., Czúcz, B., Keith, H., Jackson, B., Bland, L., Nicholson, E., Dasoo, M. 2019. Discussion paper 2.2: Review of ecosystem condition accounting case studies: Lessons learned and options for developing condition accounts. Paper submitted to the SEEA EEA Technical Committee as input to the revision of the technical recommendations in support of the System on Environmental-Economic Accounting. Final version. 25 pp.

Millennium Ecosystem Assessment. 2005. Ecosystems and human well-being: current state and trends: findings of the Condition and Trends Working Group. Edited by Rashid Hassan, Robert Scholes, Neville Ash.

Mononen, L., Auvinen, A.-P., Ahokumpu, A.-L., Rönkä, M., Aarras, N., Tolvanen, H., Kamppinen, M., Viirret, E., Kumpula, T., Vihervaara, P., 2016. National ecosystem service indicators: measures of social–ecological sustainability. Ecol. Ind. 61, 27–37. https://doi.org/10.1016/j.ecolind.2015.03.041.

Nature Diversity Act (NDA). 2009. Retrieved from Ministry of Climate and Environment. Kingdom of Norway.

Newbold, T., Hudson, L.N., Hill, S.L.L., Contu, S., Lysenko, I., Senior, R.A., B€orger, L., Bennett, D.J., Choimes, A., Purvis, A., 2015. Global effects of land use on local terrestrial biodiversity. Nature 520, 45–50.

Nybø, S., Evju, M., (Eds.), 2017. Fagsystem for fastsetting av god økologisk tilstand. Forslag fra et ekspertråd (Norwegian System for Assessment of Ecosystem Condition. Suggestion from an expert council). Ekspertrådet for økologisk tilstand, 247 s.

Nybø, S., Framstad, E., Jakobsson, S., Evju, M., Lyngstad, A., Sickel, H., Sverdrup-Thygeson, A., Töpper, J., Vandvik, V., Velle, L.G., Aarrestad, P.A., 2019. Test av fagsystemet for økologisk tilstand for terrestriske økosystemer i Trøndelag (Test of the system for assessing ecological condition for terrestrial ecosystems in Trøndelag). NINA Rapport 1672. Norsk institutt for naturforskning.

Pe'er, G., Dicks, L.V., Visconti, P., Arlettaz, Báldi, A., Benton, T.G., Collins, S., Dieterich, M., Gregory, R.D., Hartig, F., Henle, K., Hobson, P.R., Kleijn, D., Neumann, R.K., Robijns, T., Schmidt, J., Shwartz, A., Sutherland, W.J., Turbé, A., Wulf, F., Scott, A.V., 2014. EU agricultural reform fails on biodiversity: Extra steps by Member States are needed to protect farmed and grassland ecosystems. Science 344, 1090–1092.

Pedersen, B., Nybø, S., Sæther, S.A., (Eds.), 2016. Nature Index for Norway 2015. Ecological framework, computational methods, database and information systems – NINA Report 1226. 84 pp.

Pereira, H.M., Ferrier, S., Walters, M., Geller, G.N., Jongman, R.H.G., Scholes, R.J., Bruford, M.W., Brummitt, N., Butchart, S.H.M., Cardoso, A.C., Coops, N.C., Dulloo, E., Faith, D.P., Freyhof, J., Gregory, R.D., Heip, C., Höft, R., Hurtt, G., Jetz, W., Karp, D.S., McGeoch, M.A., Obura, D., Onoda, Y., Pettorelli, N., Reyers, B., Sayre, R., Scharlemann, J.P.W., Stuart, S.N., Turak, E., Walpole, M., Wegmann, M. 2013. Essential Biodiversity Variables. Science 339:277-278.

Pollock, L.J., Thuiller, W., Jetz, W., 2017. Large conservation gains possible for global biodiversity facets. Nature 546, 141-157.

R Core Team (2018). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/.

Reed, M.S., 2008. Stakeholder participation for environmental management: a literature review. Biol. Conserv. 141 (10), 2417–2431. https://doi.org/10.1016/j.biocon.2008.07.014.

Sánchez-Bayo, F., Wyckhuys, K.A.G., 2019. Worldwide decline of the entomofauna: a review of its drivers. Biol. Conserv. 232, 8–27. https://doi.org/10.1016/j.biocon.2019.01.020.

Scheffer, M., Carpenter, S.R., 2003. Catastrophic regime shifts in ecosystems: linking theory to observation. Trends Ecol. Evol. 18 (12), 648–656. https://doi.org/10.1016/j.tree.2003.09.002.

Scholes, R.J., Mace, G.M., Turner, W., Geller, G.N., Jürgens, N., Larigauderie, A., Muchoney, D., Walther, B.A., Mooney, H.A., 2008. Toward a global biodiversity observing system. Science 321, 1044–1045.

Schwartz, M.K., Luikart, G., Waples, R., 2007. Genetic monitoring as a promising tool for conservation and management. Trends Ecol. Evol. 22 (1), 25–33. https://doi.org/10.1016/j.tree.2006.08.009.

Smeets, E., Weterings, R., 1999. Environmental Indicators: Typology and Overview. Technical Report No. 25. EEA, Copenhagen.

Soga, M., Gaston, K.J., 2018. Shifting baseline syndrome: causes, consequences, and implications. Front. Ecol. Environ. 16 (4), 222–230. https://doi.org/10.1002/fee.1794.

Standish, R.J., Hobbs, R.J., Mayfield, M.M., Bestelmeyer, B.T., Suding, K.N., Battaglia, L. L., Eviner, V., Hawkes, C.V., Temperton, V.M., Cramer, V.A., Harris, J.A., Funk, J.L., Thomas, P.A., 2014. Resilience in ecology: abstraction, distraction, or where the action is? Biol. Conserv. 177, 43–51. https://doi.org/10.1016/j.biocon.2014.06.008.

Stasinopoulos, M., Rigby, R., 2018. gamlss.dist: Distributions for generalized additive models for location scale and shape. R package version 5.1-1.

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 (4), 1267–1276. https://doi.org/10.1890/1051-0761(2006)016[1267:SEFTEC]2.0.CO;2.

Sverdrup-Thygeson, A., Ørka, H.O., Gobakken, T., Næsset, E., 2016. Can airborne laser scanning assist in mapping and monitoring natural forests? For. Ecol. Manage. 369, 116–125. https://doi.org/10.1016/j.foreco.2016.03.035.

TEEB, 2010. The Economics of Ecosystems and Biodiversity. Earthscan, London & Washington.

Tittensor, D.P., Walpole, M., Hill, S.L.L., Boyce, D.G., Britten, G.L., Burgess, N.D., Butchart, S.H.M., Leadley, P.W., Regan, E.C., Alkemade, R., Baumung, R., Bellard, C., Bouwman, L., Bowles-Newark, N.J., Chenery, A.M., Cheung, W.W.L., Christensen, V., Cooper, H.D., Crowther, A.R., Dixon, M.J.R., Galli, A., Gaveau, V., Gregory, R.D., Gutierrez, N.L., Hirsch, T.L., Höft, R., Januchowski-Hartley, S.R., Karmann, M., Krug, C.B., Leverington, F.J., Loh, J., Lojenga, R.K., Malsch, K., Marques, A., Morgan, D.H.W., Mumby, P.J., Newbold, T., Noonan-Mooney, K., Pagad, S.N., Parks, B.C., Pereira, H.M., Robertson, T., Rondinini, C., Santini, L., Scharlemann, J.P.W., Schindler, S., Sumaila, U.R., Teh, L.S.L., van Kolck, J., Visconti, P., Ye, Y., 2014. A mid-term analysis of progress toward international biodiversity targets. Science 346: 241-244.

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. https://doi.org/10.1007/978-90-481-3233-1.

Turak, E., Brazill-Boast, J., Cooney, T., Drielsma, M., Delacruz, J., Dunkerley, G., Fernandez, M., Ferrier, S., Gill, M., Jones, H., Koen, T., Leys, T., McGeoch, M., Mihoub, J.-P., Scanes, P., Schmeller, D., Williams, K., 2017. Using the essential biodiversity variables framework to measure biodiversity change at national scale. Biol. Conserv. 213, 264–271. https://doi.org/10.1016/j.biocon.2016.08.019.

Töpper, J., Velle, L.G., Vandvik, V., 2018. Utvikling av metodikk for økologisk tilstandsvurdering basert på indikatorverdier etter Ellenberg og Grime (revidert utgave) (Developing a method for assessment of ecological state based on indicator values after Ellenberg and Grime (revised edition)). NINA Rapport 1529b. Norsk institutt for naturforskning.

UN, 2012. United Nations, European Union, Food and Agriculture Organization of the United Nations, Organisation for Economic Co-operation and Development, World Bank (2012). System of Environmental-Economic Accounting 2012 — Experimental Ecosystem Accounting. United Nations, New York. https://seea.un.org/sites/seea.un.org/files/seea_eea_final_en_1.pdf.

Vihervaara, P., Mononen, L., Auvinen, A.-P., Virkkala, R., Lü, Y., Pippuri, I., Packalen, P., Valbuena, R., Valkama, J., 2015. How to integrate remotely sensed data and biodiversity for ecosystem assessments at landscape scale. Landscape Ecol. 30 (3), 501–516. https://doi.org/10.1007/s10980-014-0137-5.

Vihervaara, P., Auvinen, A.-P., Mononen, L., Törmä, M., Ahlroth, P., Anttila, S., Böttcher, K., Forsius, M., Heino, J., Heliölä, J., Koskelainen, M., Kuussaari, M., Meissner, K., Ojala, O., Tuominen, S., Viitasalo, M., Virkkala, R., 2017. How Essential Biodiversity Variables and remote sensing can help national biodiversity monitoring. Global Ecol. Conserv. 10, 43–59. https://doi.org/10.1016/j.gecco.2017.01.007.

Wickham, H., François, R., Henry, L., Müller, K., 2019. dplyr: A Grammar of Data Manipulation. R package version 0.8.3. https://CRAN.R-project.org/package=dplyr.