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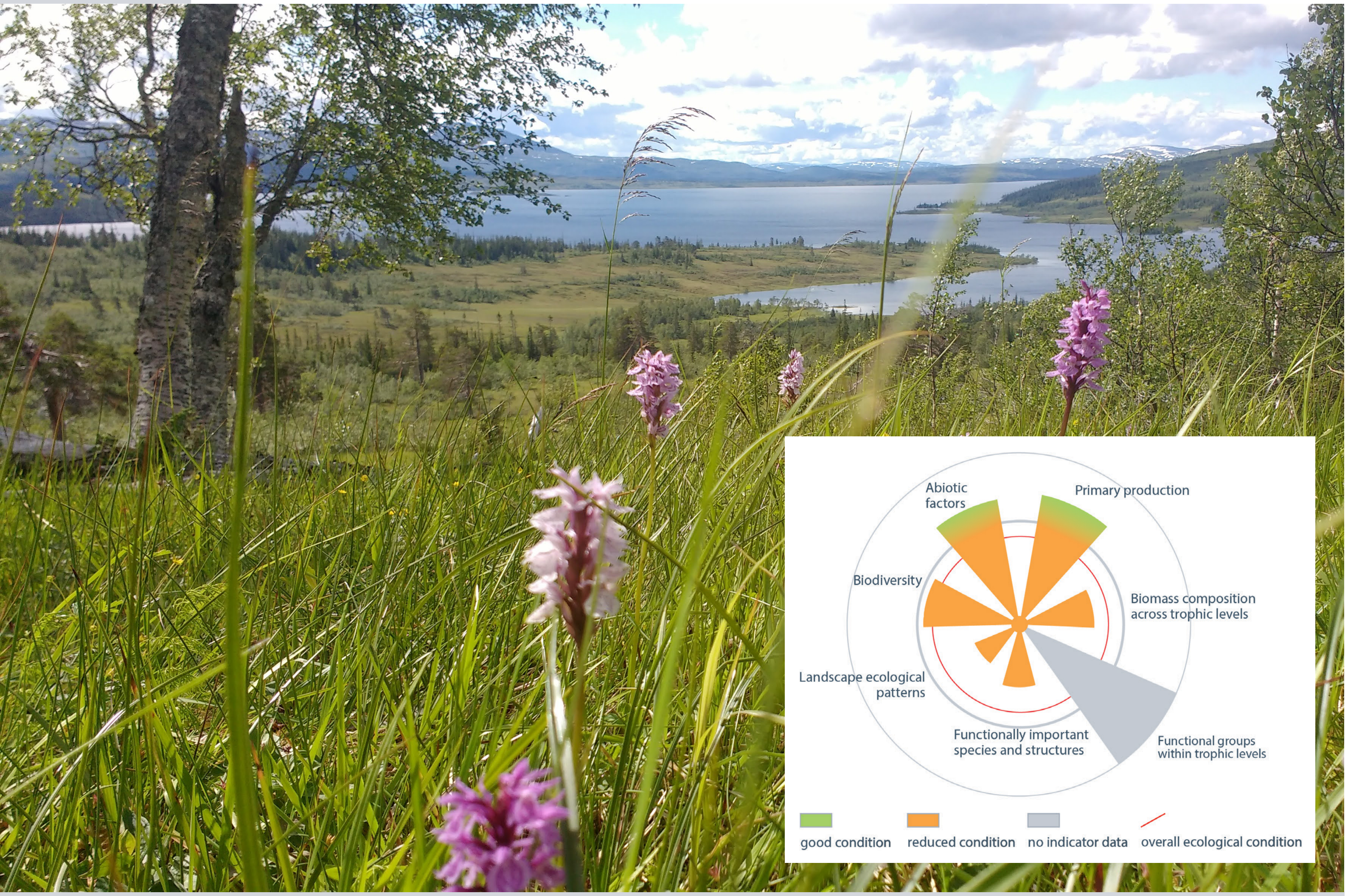
NINA Report

The Index-Based Ecological Condition Assessment (IBECA)

Technical protocol, version 1.0

Joachim P. Töpper* and Simon Jakobsson*

*Töpper & Jakobsson contributed equally to this work



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The Index-Based Ecological Condition Assessment (IBECA)

Technical protocol, version 1.0

Norwegian Institute for Nature Research

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Abstract

Töpper J. & Jakobsson S. 2021. The Index-Based Ecological Condition Assessment (IBECA) - Technical protocol, version 1.0. NINA Report 1967. Norwegian Institute for Nature Research.

Planet Earth experiences substantial and rapid losses of ecological values, primarily driven by human impacts such as habitat loss due to human land use, exploitation, pollution, introduction of alien species and anthropogenic climate change. Although science and politics have recognised these imminent threats to the world's ecosystems, and despite many initiatives for conservation, restoration, and sustainable management, the ecological degradation continues. One of the central challenges towards a sustainable future for our ecosystems is the knowledge basis, where we still fall short of effective and applicable frameworks for evaluating the ecological condition of ecosystems.

The Norwegian 'system for assessment of ecological condition' for terrestrial and marine ecosystems aims to fill such a function, by providing a sound and measurable representation of ecosystem condition. This should function as an ecological foundation for authorities when deciding management goals, and assist in designing measures to reach these goals. One of the methods developed within this system is the Index-Based Ecological Condition Assessment (IBECA). A key aspect of IBECA is the quantitative aggregation of scaled indicators into ecological condition indices. The scaling procedure relies primarily on the concept of a reference condition, representing intact nature, and limit values for what is regarded good ecological condition.

This technical report presents the methods for the IBECA framework. We address (i) how reference conditions for indicators in IBECA are defined, including reference values, limit values for good ecological condition, and minimum/maximum values of indicators, (ii) the scaling procedures that make the indicators comparable, (iii) the aggregation procedures for calculating estimates for ecological condition for the ecosystem as a whole as well as for single ecosystem characteristics and ecosystem pressures, (iv) how uncertainty is treated in IBECA, and finally (v) the requirements for indicators and data used in IBECA.

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Sammendrag

Töpper J. & Jakobsson S. 2021. The Index-Based Ecological Condition Assessment (IBECA) - Teknisk protokoll, versjon 1.0. NINA Rapport 1967. Norsk institutt for naturforskning.

Planeten vår er utsatt for betydelige og raske tap av økologiske verdier, hovedsakelig drevet av menneskelige påvirkninger som tap av habitat på grunn av menneskelig arealbruk, beskatning, forurensning, spredning av fremmede arter og menneskeskapte klimaendringer. Selv om vitenskap og politikk har anerkjent disse pågående truslene mot verdens økosystemer, og til tross for mange initiativer for bevaring, restaurering og bærekraftig forvaltning, fortsetter tapet av økologiske verdier. En av de sentrale utfordringene mot en bærekraftig fremtid for våre økosystemer er kunnskapsgrunnlaget, der vi fortsatt mangler effektive og anvendbare rammer for å evaluere økologisk tilstand.

I Norge tar 'fagsystemet for vurdering av økologisk tilstand for terrestriske og marine økosystemer' sikte på å fylle en slik rolle ved å gi en faglig og målbar representasjon av økologisk tilstand i økosystemene. Dette skal danne et faglig grunnlag for myndighetene når det skal fastsettes forvaltningsmål og for å innrette den samlede virkemiddelbruken for å nå slike mål. En av metodene som er utviklet i dette systemet er 'Indeksmetoden' (Index-Based Ecological Condition Assessment, IBECA). Et sentralt aspekt ved IBECA er den kvantitative aggregeringen av skalerte indikatorer til økologiske tilstandsindekser. Skaleringsprosedyren relaterer i hovedsak til begrepet referansetilstand, som representerer intakt natur, og grenseverdier for det som anses som god økologisk tilstand.

Denne tekniske protokollen presenterer metodene for IBECA-rammeverket. Vi tar for oss (i) hvordan referansetilstanden for indikatorer i IBECA er definert, inkludert referanseverdier, grenseverdier for god økologisk tilstand og minimum/maksimumsverdier for indikatorer, (ii) skaleringsprosedyrene som gjør indikatorene sammenlignbare, (iii) aggregeringsprosedyrer for beregning av indekser for økologisk tilstand for økosystemet som helhet, samt for enkelte økosystemegenskaper og påvirkninger, (iv) hvordan usikkerhet behandles i IBECA, og til slutt (v) krav til indikatorer og data brukt i IBECA.

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Contents

Foreword

Since 2018, the Norwegian Environmental Agency has commissioned the development of methodology for the assessment of ecological condition based on the framework suggested by an expert panel appointed by the Norwegian government (Nybø & Evju 2017). In 2020, the Norwegian Institute for Nature Research (NINA) was charged with the national assessment of ecological condition for forest and mountain ecosystems, using the *Index-Based* approach for *Ecological Condition Assessment* (IBECA). In connection with this work, NINA was also commissioned with documenting the methodological protocol used in IBECA, which is presented in the report at hand. As for the specific indicators, development of new indicators and adjustments of existing indicators will happen in connection with actual evaluations of ecological condition of ecosystems as our data and knowledge base improves. Information about the specific indicators will thus be documented in connection to the ecological condition assessments themselves. The first comprehensive update of indicator protocols will be documented in the course of the work with the national assessment of ecological condition in forest and mountain ecosystems in spring and winter 2021, respectively. In the report at hand, we suggest the establishment of a digital solution for documenting and managing the developing methodology and protocols for assessments of ecological condition.

NINA-researchers Simon Jakobsson and Joachim Töpper have during the last three years been instrumental for the technical and analytical implementation of IBECA and collated the information necessary for performing an assessment of ecological condition using the IBECA framework in this report. Of course, there would be no method to write about without the efforts of the greater IBECA Working group during the recent years, and we thus thank Anders Lyngstad (NTNU University Museum), Hanne Sickel (NIBIO), Anne Sverdrup-Thygeson (NMBU), Vigdis Vandvik (University of Bergen), Liv Guri Velle (Møreforsking), and our NINA colleagues Tessa Bargmann, Marianne Evju, Erik Framstad, Markus Fjellstad Israelsen, Signe Nybø, Zander Venter, and Per Arild Aarrestad. The IBECA Working group comprises a wide range of ecological backgrounds and institutions, lending important scientific breadth to the framework. As a principal concept, the framework itself, as well as the applied methods for defining reference conditions, have been published as peer-reviewed journal articles in 2020 and 2021, and IBECA is thus starting to receive international attention.

NINA researcher Bård Pedersen, who has key expertise and experience from the Norwegian Nature Index, has performed the internal quality control for this report, ensuring methodological accuracy. His thorough and constructive review has contributed significantly to this report.

Finally, we thank the Norwegian Environmental Agency for excellent communication during the project. Else Løbersli and Eirin Bjørkvoll have been our contacts.

Bergen, mars 2021

Joachim Töpper & Simon Jakobsson

1 Introduction

During the last decades, planet Earth has transitioned into a world featuring substantial and rapid declines in biodiversity, on both the genetic, taxonomic and ecosystem level. These losses are all mainly driven by human impacts such as habitat loss due to human land use, exploitation, pollution, introduction of alien species and anthropogenic climate change (Sánchez-Bayo and Wyckhuys 2019, IPBES, 2018, Newbold et al. 2015, Allentoft and O'Brien 2010). Science and politics have recognised the imminent threats the biodiversity and climate crisis pose to human lives and livelihoods (Ruckelshaus et al 2020, IPBES 2018, Allen et al. 2014). Yet, despite various initiatives for conservation, restoration, and sustainable management – on both local, regional, and global scales – the biodiversity losses continue (Mace et al., 2018).

One of the central challenges towards sustainable management of our natural resources is the knowledge basis and its unsatisfactory availability to decision-makers in a relevant, aggregated and understandable form. While humankind has gathered impressive amounts of ecological understanding filling textbooks and scientific journals, we still fall short of effective and applicable frameworks for evaluating the ecological condition of ecosystems. 'Effective' as in providing a sound representation of ecosystems based on measurable and accessible metrics, and 'applicable' with respect to informing management and decision-makers on the actual causes of ecological degradation and effects of management actions. Internationally, there are two main concepts addressing this knowledge gap: (1) the Essential Biodiversity Variables concept (EBV) is one major approach to assess progress towards the Aichi targets (Pereira et al. 2013, Scholes et al. 2008) and provides a framework for developing indicator and monitoring systems. (2) the Ecosystem Condition Typology (ECT) within the SEEA-EEA framework for ecosystem accounting (Hein et al. 2020, Maes et al., 2019; Czúcz et al. 2019, UN 2012) 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 for aggregating indicators into ecological condition indices using empirical datasets (https://seea.un.org/).

In Norway, the development of the Norwegian 'system for assessment of ecological condition' for terrestrial and marine ecosystems (Nybø & Evju 2017) has led to the development of two methods to assess overall ecological condition. One of these methods is the Index-Based Ecological Condition Assessment (IBECA), with key emphasis on approaches for scaling and aggregating indicators into ecological condition indices (Jakobsson et al. 2021). This method was tested for the four major terrestrial ecosystems (forest, mountain, wetland, semi-natural) in Trøndelag county in 2019 (Nybø et al. 2019), and is currently being applied to assess the ecological condition of forest and alpine ecosystems nationally (e.g. Framstad et al. 2021).

1.1 A Norwegian System for assessment of ecological condition

In 2017, an expert committee appointed by the Norwegian Ministry for Climate and Environment launched a principal concept for the definition and evaluation of good ecological condition (Nybø & Evju 2017), complementary to other established systems like the Nature Index, the Water Framework Directive, the UN ecosystem accounts, and the Red Lists for species and habitat types (*cf.* Nybø et al. 2020). This system defines seven universal ecosystem characteristics covering the structure, productivity, and function of ecosystems (**Figure 1.1**). These seven ecosystem characteristics are too general to be measured directly and thus have to be addressed via various measurable indicators that allow the evaluation of the ecological condition of the single ecosystem characteristics and the ecosystem as a whole (Nybø et al. 2018). Good ecological condition is defined as a state in which *'the ecosystem's structure, function and productivity do not significantly deviate from the reference condition, defined as an intact ecosystem'*. This definition does allow human impact to a certain degree as long as structure and function still approximate the reference condition, which implies that either *'the ecosystem is robust enough so that human impacts do not change the ecological condition (robustness)'*, or that *'the ecosystems* *own internal processes easily can restore good ecological condition (resilience)'* (Nybø & Evju 2017). For the Norwegian system for assessment of ecological condition two operational methods for defining reference conditions and evaluating the ecological condition of an ecosystem against these have been developed: the 'Panel-based Assessment of Ecosystem Condition' framework (PAEC), which is described elsewhere (Jepsen et al. 2020), and the 'Index-Based Ecological Condition Assessment' framework (IBECA, Jakobsson et al. 2021). This report presents the methods for the IBECA framework.

In IBECA, ecological condition is primarily evaluated via aggregation of a set of scaled indicators (see Framstad et al. 2021 and Nybø et al. 2018 for examples), which are selected to cover the aspects of the ecosystem characteristics and potential ecological pressures in a representative way (see chapter 4 for details). As a consequence, the definition of a reference condition, as well as a quantification of key threshold values, for each indicator is central for the practical application of the framework. In the following chapters we will address (i) how reference conditions for indicators in IBECA are defined, including reference values, limit values for good ecological condition, and minimum/maximum values, (ii) the scaling procedures that make the indicators comparable, (iii) the aggregation procedures for calculating estimates for ecological condition for the single ecosystem characteristics and pressures as well as the ecosystem as a whole based on the single scaled indicators, (iv) how uncertainty is treated in IBECA, and (v) requirements for indicators and data used in IBECA.

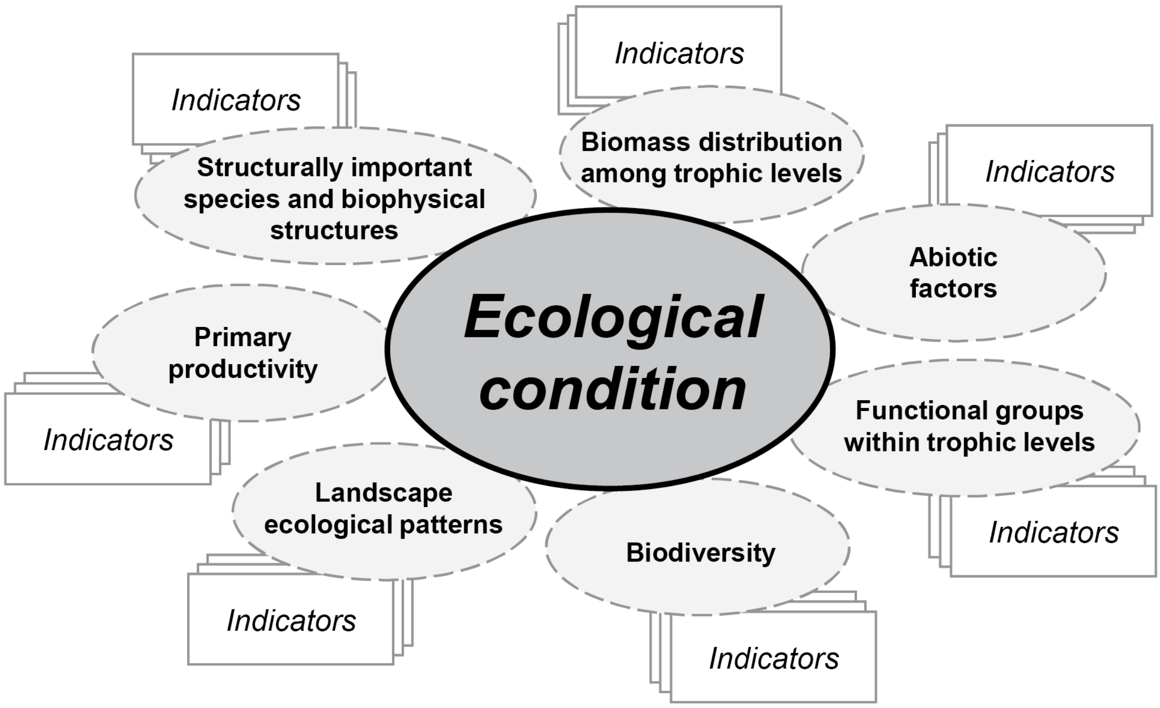

Figure 1.1. *A schematic representation of the Norwegian system for evaluation of ecological condition. The ecological condition of an ecosystem is evaluated through measurable indicators that themselves are related to seven defined ecosystem characteristics. Modified from Jakobsson et al. (2020).*

2 Reference conditions and quantification of threshold values

One possible way to approach an assessment of the ecological condition of an ecosystem implies the comparison of the ecosystem's *status quo* to a reference condition (Nielsen et al. 2007, Scholes & Biggs 2005). The reference condition can take various conceptual approaches (*cf*. McNellie et al. 2020), and within IBECA it relates to the concept of 'intact nature' (see Chapter 1). IBECA is a quantitative, indicator-based framework, and for every indicator it is therefore necessary to quantitatively define the functional relationship between the indicators' original measurements and the scale of the ecological condition index. In order to assess the indicators' deviation from what is defined as good ecological condition, this requires the quantification of threshold values, describing (i) the indicator value under a reference condition (reference value), (ii) the indicator value describing the border between what is defined as good and reduced ecological condition (limit value), as well as (iii) the minimum/maximum values an indicator can take under severely reduced ecological condition. This chapter provides a discussion of approaches for quantifying these indicator-specific threshold values. Complete lists containing concepts, data basis, as well as reference, limit, and minimum/maximum values for the specific indicators will be documented in the respective IBECA reports on specific ecosystems.

2.1 Reference values

One of the probably most straightforward approaches for defining reference values for ecological indicators is to look at the indicator's values in a baseline year, where we know, or have reason to assume, that the ecological state of the respective ecosystem was good (e.g. EBCC 2019, EEA 2012). However, most indicators for comprehensive ecological condition assessments lack the necessary historical data for this approach (*cf*. Collins et al. 2020). In addition, using baseline years falls short in comparisons across sites or geographical areas as conditions, and hence baseline values, will generally differ for any given baseline year (*cf*. Soga & Gaston 2018), and may as well be less appropriate for evaluating progress towards absolute management goals (e.g. restoration success). In IBECA, we have explored several other principal approaches for defining reference values for ecological indicators: absolute biophysical boundaries, reference areas, reference communities, ecosystem dynamics based models, and habitat availability based models (**Table 2.1**, see Jakobsson et al. 2020 and references therein for details). The reference values of different indicators used in an ecological condition assessment may be quantitatively derived via different reference approaches. However, all indicators' reference values should at least theoretically relate to a common overarching reference condition. This would for instance not be the case when using different baseline years (e.g. 1900 vs. 1970), or when using reference areas that are not representative for the entire region in the assessment. For the practical implementation within IBECA, all these conceptual approaches relate to the concept of 'intact nature' for defining the reference condition (see above) and its associated reference values for indicators and thus build on similar principles as in the EU Water Framework Directive (Direktoratsgruppen vanndirektivet 2018, EC 2019). The reference value of an indicator can be (i) defined by e.g. absolute biophysical boundaries, (ii) derived statistically (e.g. statistical distributions for reference areas or communities), or (iii) calculated from models (e.g. based on empirically supported expert knowledge) (see **Table 2.1**). As stated above, the reference value is thought to represent the indicator value considered most optimal for a given ecosystem, but this may vary geographically, due to ecological differences caused by e.g. gradients in climate or edaphic factors, or between different ecosystem sub-types. Thus, an indicator may be represented by several reference values stratified by different realisations of the ecosystem.

Table 2.1. *The five approaches for defining reference conditions for ecosystems in good ecological condition in IBECA, including the logical concept, practical estimation of reference values, and examples of indicators. Modified from Jakobsson et al. (2020).*

Nr	Category	Conceptual idea	Estimation of reference values	Examples of indicators
1	Absolute biophysical boundaries	An *intact ecosystem* is represented by min. or max. of indicator values	Minimum or maximum value (e.g. 0/1 or 0/100 %)	Pollution levels, cover of vegetation types/species
2	Reference areas	Area(s) considered representing an *intact ecosystem*	Indicator data from reference areas (e.g. mean or maximum values)	Any
3	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	Species community data linked to quantitative data on species functional attributes
4	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
5	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

2.2 Limits for good ecological condition

Estimating a reference value for the intact ecosystem based on one of the above described conceptual approaches is often straight-forward (see **Table 2.1**). It is more difficult to define the limit value(s) for good ecological condition, representing the border beyond which indicator values deteriorating from the reference value indicate a reduced ecological condition of the ecosystem. For instance, the IBECA indicator on alien species is formulated as *% area without alien species*; here we can simply state the absence of alien species, i.e. 100% area without alien species, as the reference value, but where would we define the limit for good ecological condition? At 99% area without alien species? Or rather 95%, or down to 80%? In principle, the most optimal information to base informed limit values on would be a known dose-response relationship between an environmental pressure and a related indicator (Andersen et al. 2008). However, such dose-response relationships are often not well documented in the literature, and analytical, or more general approaches, to set indicator-specific limits for good ecological condition are needed. In IBECA, we have thus applied two additional approaches to define reference limits for good ecological condition: statistical distributions and expert judgement-based limits (**Table 2.2**, see Jakobsson et al. 2020 and references therein for details). Similar to reference values, also limit values may vary geographically or between different ecosystem sub-types. Thus, good ecological condition for an indicator may be represented by several limit values stratified by different realisations of the ecosystem.

Table 2.2. *The three approaches for estimating indicator threshold values for the limit for good ecological condition. The table includes the conceptual idea and the relevant scaling approach(es) in practice. For indicator-specific examples, see* ***Table S1.*** *Modified from Jakobsson et al. 2020.*

Nr	Category	Conceptual idea	Estimation of limit values
1	Empirically estimated values	Critical levels of the indicator can be directly linked to empirical data	Dose-response relationship
2	Statistical distributions	Distribution of indicator values within a reference data population used to estimate statistical deviance from the reference condition	95 % confidence interval
3	Expert judgement-based limits	a) Based on scientific expertise, the relationship between the reference condition and a degraded ecosystem is assumed to be linear	Expert-based
		b) Based on scientific expertise, the relationship between the reference condition and a degraded ecosystem is assumed to be non-linear	Expert-based

2.3 Minimum/maximum values of an indicator

It can be challenging to define the possible range of values suggesting reduced ecological condition for an indicator. However, identifying the complete range is key to accurately capture the degree of degradation suggested by the single indicators. In IBECA, this range is delimited by the limit value, indicating the border between good and reduced ecological condition, and the minimum/maximum values, indicating the lowest/highest measurable value for an indicator under severely reduced ecological condition. We apply three main approaches for quantification of the latter: (i) absence of the indicators measuring unit, (ii) lowest/highest empirically observed values, and (iii) lowest/highest principally possible values.

In the case of the *% area without alien species* indicator it is intuitive to define the minimum value as no area without alien species (0%). In this indicator, the maximum value of 100% area without alien species also represents the reference value; thus, it is a one-sided indicator with no possible values larger than the reference value. Indicators such as *Predator population level* are theoretically two-sided – ecologically speaking, there could principally be both too few or too many predators – but in practice only the minimum value (absence of predators) needs to be defined here due to the extremely low population levels among larger predator species in Norway. Similarly, only minimum values are defined for indicators like *Bilberry cover* or *Amount of dead wood* in forest ecosystems as any practically observable values larger than the reference value are not considered as ecological deterioration. In all the above types of indicators, the minimum value of the indicator equals zero (0), i.e. the absence of the species/group/structure.

Conversely, indicators based on the normalized difference vegetation index (NDVI) and Ellenberg values truly are two-sided indicators, and it is not possible to observe a value of zero (0). In the case of NDVI, the data for the indicator represent a complete remote-sensing sample of the entire area that is assessed, e.g. all forest in Norway (see Framstad et al 2021). In such a case, the minimum and maximum values for the indicator can be based on the minimum and maximum values observed in the data. In the case of Ellenberg values, it is of course impossible to obtain a complete sample of all plant communities in an ecosystem type, and we therefore revert to the minimum and maximum values possible on the scale for the respective Ellenberg indicator (in most cases 1 and 9, respectively, see Framstad et al 2021).

3 Indicator scaling and aggregation

As an index-based approach, IBECA relies on assembling indicators in order to assess the condition of an ecosystem, or an ecosystem characteristic. Different indicators may operate on very different scales and thus need to be standardised to a common scale from 0 to 1 prior to aggregation. Since IBECA is a quantitative framework, index uncertainty and indicator weighting also need thorough consideration when aggregating indicators. In this chapter, we outline the key steps to aggregated indices of ecological condition within IBECA.

3.1 Scaling principles

Scaling is based on the reference values, limit values for good ecological condition, and minimum/maximum value(s) discussed in Chapter 2. All indicators are standardised to a common scale of 0 – 1. The reference value of an ecosystem is represented by a scaled value of 1, the minimum/maximum value(s) are represented by a scaled value of 0, and the limit(s) for good ecological condition are represented by a scaled value of 0.6, harmonising with the classification of ecological condition within the EU Water Framework Directive (Direktoratsgruppen vanndirektivet 2018, EC 2019). Indicator values between the reference value and limit value are scaled according to a linear model between reference value and limit value and thus receive scaled values between 1 – 0.6. Indicator values between the limit value and minimum/maximum value are scaled according to a linear model between limit value and minimum/maximum value and thus receive scaled values between 0.6 – 0 (**Figure 3.1**).

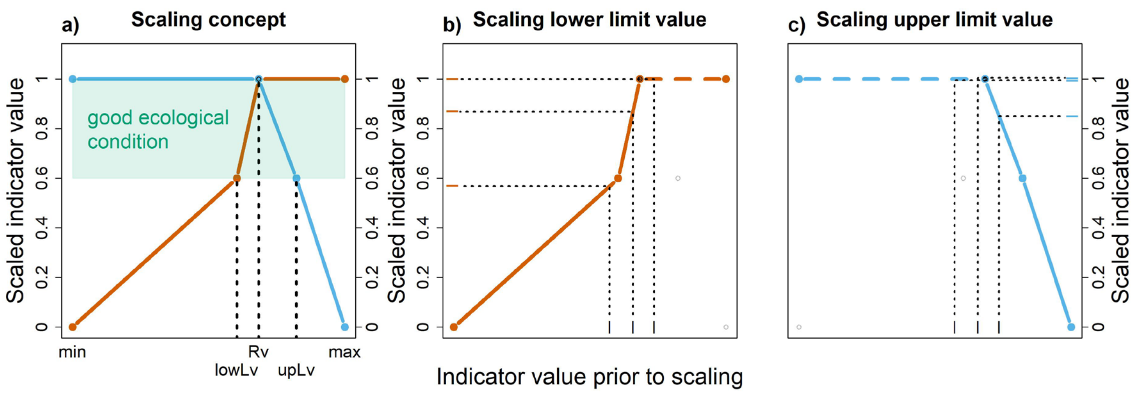

Figure 3.1. *Scaling of indicator values in relation to the reference value, the limit for good ecological condition, and the minimum/maximum values. The scaling concept (a) builds on a reference value (Rv), where raw indicator values (x-axis) at Rv are scaled to 1. Furthermore, a lower (lowL) and/or 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. Dashed coloured lines represent scaling truncated to 1 in one-sided indicators. For 'two-sided' indicators (i.e. with both a lower and upper limit), the indicator values lower or higher than the Rv are scaled against lowL and upL, respectively, and subsequently handled as two separate indicators. Modified from Jakobsson et al. (2021).*

For one-sided indicators, values larger than the reference value are scaled to 1 (i.e. truncated at 1), as values > 1 do not exist within the overall reference condition concept for the ecosystem. If one-sided indicators were scaled to values > 1 according to the linear model based on reference and limit value this may result in very high scaled values that would become highly influential in the subsequent aggregation procedure. Together, the piecewise linear scaling functions defined by the reference-, limit-, and minimum/maximum values thus represent an overall non-linear scaling function used to transform indicator observations from their original scale to the scale of ecological condition indices.

As mentioned above, indicators may be two-sided, i.e. indicators where values both larger and smaller than the reference value indicate an ecological deterioration from the reference condition. In IBECA, there are currently three such types of indicators used: *NDVI*, *Ellenberg indicators,* as well as *animal population levels*[1]. When developing IBECA, two different approaches to handle these indicators have been explored. Nybø et al. (2019) used a 'worst case scenario' approach. They split two-sided indicators into two separate side-indicators – one calculating deviations for all data from the reference condition and towards the lower limit value, and one calculating deviations for all data from the reference condition and towards the upper limit value. The upper indicator would thus detect ecological deterioration due to too high levels, and *vice versa* for the lower indicator. As all data for the respective indicator are used both towards the lower and the upper limit values, only the side with the largest deviation from the reference value was used in aggregated indices. One disadvantage with this approach is that it ignores the fact that different areas of a given ecosystem type may show positive or negative deviations from the reference condition. It is not meaningful for effective area management to only focus on the one side where the deviations are largest, as such informed general management measures likely would worsen the ecological condition in areas showing deviations on the other side. Therefore, we make use of both 'sides' of two-sided indicators in the current assessment of ecosystem condition of forests (Framstad et al. 2021)[1]. To do so without counting such an indicator twice, all indicator values are only scaled with respect to one limit value: indicator values below the reference condition are scaled towards the lower limit and minimum values, and indicator values larger than the reference condition are scaled towards the upper limit and maximum values (**Figure 3.1**). Indicator values that actually equal the reference value are utterly unlikely, but if they occur then they are scaled to 1 and count towards both sides. Hence, the lower indicator consists of observations up to and including the reference value, whereas the upper indicator consists of observations at and larger than the reference value.

3.2 Scaling and spatial resolution

The spatial resolution and coverage of measurements/observations varies greatly among indicators. For instance, (i) *Ellenberg indicators* are based on ANO data which are collected in 1 m^2 plots (up to 18 of these plots make up a site of potentially varying ecosystem types, 1000 sites are distributed randomly on mainland Norway, see Tingstad et al. 2019), (ii) *Bilberry cover* are from the National Forest Inventory dataset at site scale (250 m^2), with more than 10.000 sites across the country (Viken 2018), while (iii) the data for the indicator *Deer population levels* are available at either the county (see www.naturindeks.no) or municipality level (see Speed et al. 2019). As a principal rule, in IBECA, scaling of indicator data is performed at the lowest possible spatial scale for each respective indicator, and thus prior to any spatial aggregation calculating mean indicator values for higher spatial scales. This rule is especially important when calculating spatially aggregated values for one-sided indicators that allow observations larger than the reference value, and for two-sided indicators. For instance, if scaling were performed at the regional level for the forest indicator *Bilberry cover* (the lowest spatial scale here is 'site'), the calculation of regional mean indicator values before scaling would allow local observations of higher bilberry cover than the indicators' reference value to compensate for local observations with less bilberry cover than the limit value. This would result in a regional estimate that is biased towards good ecological condition. Similarly, scaling an *Ellenberg* indicator at the site level (the lowest spatial

[1] In Framstad et al. (2021) the indicator *Deer population levels* could not be evaluated as a two-sided indicator due to lack of data from https://hjorteviltregisteret.no/. We had to revert to use data delivered to the Norwegian Nature Index 2020 and to the calculations therein (Jakobsson & Pedersen 2020).

scale here is ‘plot’) would allow plots with indicator values below the lower limit value and plots with indicator values above the upper limit value to compensate for each other. This would be highly problematic since one could potentially get a situation where an indicator makes a site (or region) appear as on average in good ecological condition while the sites’ plots (or the region’s sites) actually indicate positive and negative deviations from good ecological condition. Further, in some indicators the reference and limit values vary among regions (e.g. *Bilberry cover*), forest productivity class (e.g. *Amount of dead wood*), or basic ecosystem types (e.g. Ellenberg indicators). Scaling at the finest possible spatial resolution ensures that (i) the assessment covers local spatial variation in ecological condition, and (ii) all indicators are scaled according to the correct threshold values.

3.3 Indicator aggregation

In order to assess ecological condition for the ecosystem characteristics or the overall ecosystem in a chosen assessment area, the scaled indicator values need to be aggregated across indicators. In principle, an aggregated index represents the average across the relevant scaled indicators for the single ecosystem characteristic or the overall ecosystem, respectively. In IBECA, aggregated ecological condition indices are calculated as unweighted averages across scaled indicators, i.e. each indicator has the same weight in the calculation of the aggregated indices. However, two-sided indicators are in practice split into two indicators, a lower and an upper indicator (*cf*. Section 3.1). The two sides receive weight according to their respective proportions of unscaled observations summing to 1. Further, aggregated indices for the ecosystem characteristics and the overall ecosystem are calculated independently of each other (see Jakobsson et al. 2021), but can, in principle, also be calculated hierarchically, i.e. when indicators are aggregated into ecosystem characteristic indices and these are then aggregated into an overall ecosystem index (**Figure 3.2**).

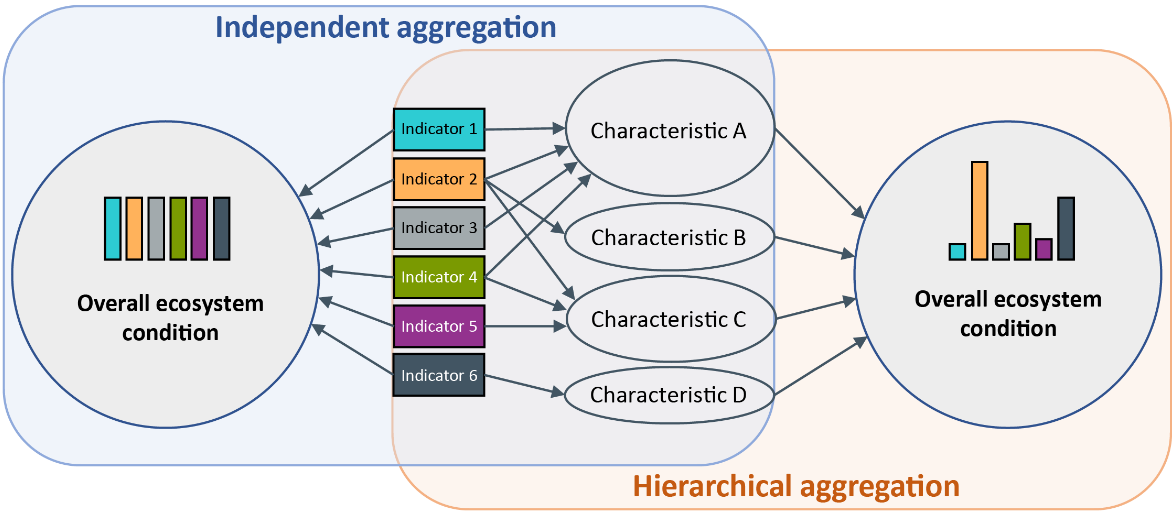

Figure 3.2. *Conceptual illustrations for independent (left) and hierarchical (right) aggregation of indicators for overall ecosystem condition. Empirical indicators relate to ecosystem characteristics, where one indicator may represent multiple characteristics. For graphical clarity, only four characteristics and six indicators are visualised here. Scaling of indicators (0 – 1; Chapter 3.1) allows the combination of indicators into aggregated estimates of the ecosystem 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 either aggregation of indicator values* ***independent of ecosystem characteristics*** *or* ***hierarchical aggregation via the ecosystem characteristics****. Bar plots in each circle represent the indirectly weighted contribution of each indicator in the overall ecosystem condition assessment for each approach.*

Indicators may be part of more than one ecosystem characteristic if their ecological role overlaps across characteristics. Thus, if the index for the overall ecosystem is calculated directly from the indicators, and not via the ecosystem characteristic indices, it is ensured that no indicator is used more than once for any aggregated index (**Figure 3.2**).

The current assessment of ecological condition for Norwegian forest ecosystems applies both independent and hierarchical aggregation for estimation of the overall ecosystem index (Framstad et al. 2021). With hierarchical aggregation, it is important to be aware that indicators applied to more than one ecosystem characteristic may receive more weight in the overall assessment – their weight increases with the number of ecosystem characteristics they apply to. At the same time, the weight of an indicator is also moderated by the number of other indicators allocated to the same characteristics. Such weight differences may be representative of an indicator's generality or of its practical importance in the respective analysis, but they do not necessarily express higher or lower ecological importance. One way to handle undesired differences in indicator weight for the overall ecosystem assessment in hierarchical aggregation could be to adjust every indicator's weight going into the evaluation of ecosystem characteristics so that their summed weights are equal in the overall ecosystem evaluation. However, such an approach would distort the assessment of ecosystem characteristics as the different indicators would be weighted differently on completely arbitrary (for the ecosystem characteristic level) terms. Another approach could be to allow each indicator to represent only one ecosystem characteristic (*cf*. SEEA-EEA, Maes et al. 2019), but also here (i) the number of indicators per characteristic will affect the weight of an indicator in the overall assessment, and (ii) the simplified exclusive representation of indicators in single ecosystem characteristics reflects ecological complexities poorly. Also, data-related aspects varying among indicators, like spatial representation or data quality, may motivate a discussion on weighting of the indicator set of an ecosystem and/or the characteristics therein. Several alternatives of indicator-specific weighting have been discussed during the development of IBECA. However, relying on lessons learned from the SEEA-EEA framework (Czúcz et al. 2019, Maes et al. 2019), weighting was not conducted in the IBECA pilot (Nybø et al. 2019), and also the current national assessment of ecological condition of forest ecosystems primarily applies the simpler approach of calculating the indices for the ecosystem characteristics and the overall ecosystem independently (Framstad et al. 2021).

3.4 Aggregation for evaluating pressures

In IBECA, each single indicator is required to be responsive to ecosystem pressures that relate to one of the five main drivers of ecosystem change as described in the Millennium Ecosystem Assessment (MEA 2005): habitat loss due to land-use/infrastructure, climate change, pollution (incl. eutrophication and acidification), exploitation/harvesting, and invasive alien species. In order to estimate impacts of these pressures on ecological condition, IBECA combines aggregation of ecological condition indicators into pressure indices, using the indicators included in the assessment, with a trend analysis of available data on the pressures.

For the aggregation into pressure indices, indicators are grouped into the five main categories according to an evaluation of which are the most critical pressures on each indicator (may be more than one) (*cf.* Aslaksen et al. 2012). Here, it is important to note, that this grouping of indicators into pressure categories is based on expert considerations and does not represent empirically tested relationships. Then, using the same approach as for the ecological condition assessment of ecosystem characteristics (Chapter 3.3, see also 3.5 concerning uncertainty), indicator estimates are aggregated within each pressure category to calculate aggregated pressure-related indices. These aggregated estimates are used to evaluate the quantitative contribution of each pressure category to the deviation from the reference condition within the overall ecological condition assessment (Jakobsson et al. 2021).

To complement the index-based analysis, data on available pressures are compiled to allow either a quantitative or qualitative assessment of potential trends in these data. Ideally, data should cover all the above mentioned five main categories of ecosystem pressures, but data coverage and resolution may limit the potential use across spatio-temporal scales. Examples of data are time series of land use impact, climate variables, deposition of pollutants, hunting statistics and records of species invasions (see Framstad et al. 2021 for specific examples).

3.5 Accounting for uncertainty

In order to account for uncertainty arising from spatial variation in indicator observations, IBECA uses a bootstrap approach (*cf.* Davison & Hinkley 1997). For individual indicators, the scaled observations for a given region are resampled with replacement (N samples = number of observations), and then the mean value is calculated for every sample. Resampling is iterated 10 000 times to generate a distribution of mean values, where the median (0.5 quantile) of the distribution is used as the indicator estimate. The uncertainty linked to this estimate is presented as a 95% confidence interval (the 0.025 – 0.975 quantiles of the generated distribution), which means that we are 95% certain that this range of values contains the 'true' mean of the indicator. While this is straight forward for quantitative datasets, indicators may also be based on non-sample-based data (e.g. the IBECA indicator *Area proportion > 1 km from infrastructure*). If so, uncertainty should be estimated either by (i) elicitation from expert knowledge and published literature, or (ii) a qualitative approximation of proportional (%) uncertainty. Uncertainty is not allocated to the normative reference values and limit values for good ecological condition (*cf.* Pedersen et al. 2016, Certain et al. 2011).

In aggregated index estimates, IBECA accounts for uncertainty by a similar approach as for individual indicators. Values for each indicator distribution of mean values within a given set of indicators for aggregation (i.e. an ecosystem characteristic, pressure, or all indicators representing an ecosystem) are resampled with replacement: One value is drawn from each indicator distribution (of mean values), and the mean of these resampled values is calculated in line with the aggregation principles outlined in section 3.3. The resampling is repeated 10 000 times, with replacement between each resampling. This yields a distribution of resampled mean values representing the estimate of the aggregated index value (median; 0.5 quantile) with its associated uncertainty: 0.025 and 0.975 quantiles as the 95 % confidence interval (interpretation as described above, see **Figure 3.3**).

Estimation of uncertainties around indicator values as well as aggregated indices are used to evaluate deviations from the reference condition. Values < 0.6 with a 95 % confidence interval not overlapping 0.6 are referred to as 'significantly reduced' ecological condition, whereas values where the 95% confidence interval overlaps 0.6 are referred to as 'marginally reduced ecological condition'. Values > 0.6, with a 95 % confidence interval not overlapping 0.6, are interpreted as a 'negligible reduction' of the ecological condition.

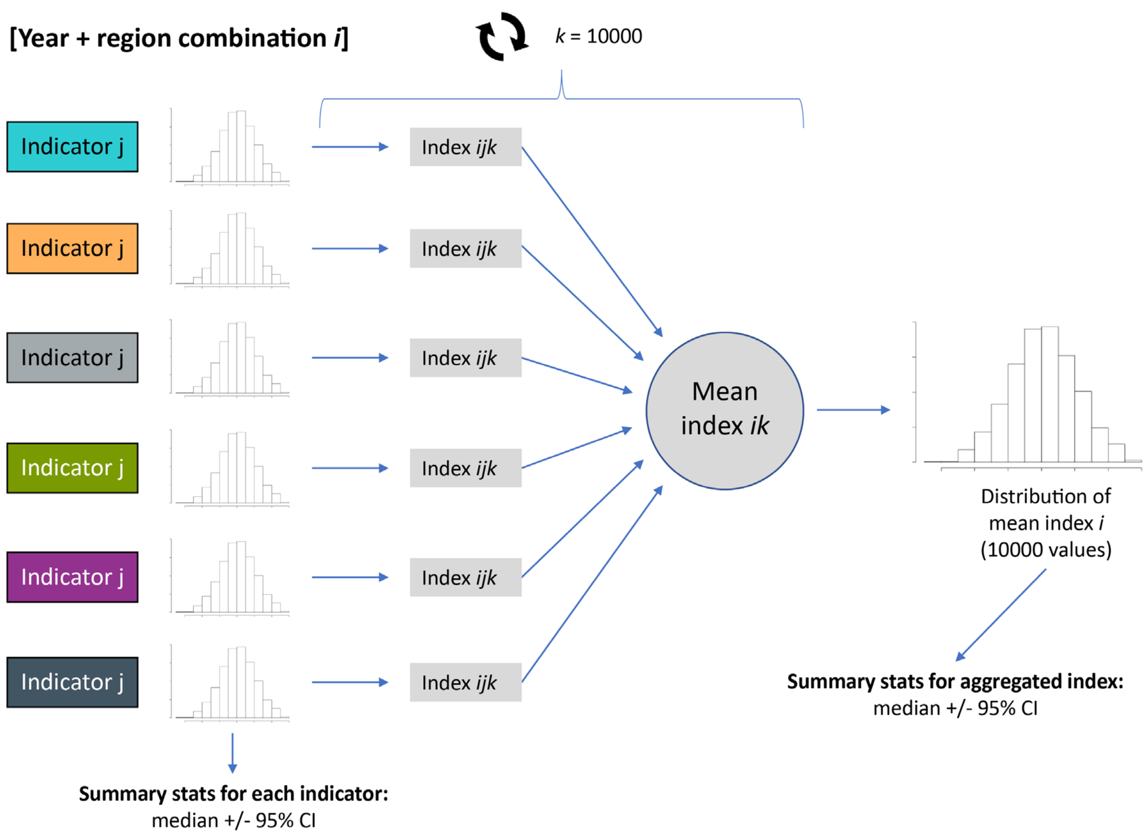

Figure 3.3. *Conceptual illustrations of the resampling strategy for aggregated indices of ecological condition within IBECA. For each indicator, a distribution of index estimates is calculated, which represents the uncertainty around each indicator estimate. For aggregated indices (either for a characteristic, pressure, or the overall ecosystem), index values are drawn from the distribution of mean values for each indicator included in the aggregated index to estimate an aggregated mean. This process is repeated 10 000 times, with replacement between each resampling, to generate a distribution of aggregated index mean values.*

4 Requirements for indicators and data

To facilitate a sound evaluation of ecological condition with IBECA, the indicator data feeding into the framework need to meet a number of requirements with respect to applicability, reliability, representativeness, temporal resolution, and accessibility (Nybø & Evju 2017). The bottom line for indicators in IBECA is that they must relate to one of the seven **ecosystem characteristics**, be **responsive to external pressures** on the ecosystem, and that it must be possible to **define a reference condition** for each indicator, which allows the **quantification of a reference value, limit value(s) and minimum/maximum value(s)**.

Indicator data need to be **spatially representative** for the evaluated area. Further, indicator data need to be representative for the respective ecosystem subject to assessment and reliable with respect to **data quality**. Indicator data should thus be retrieved from professionally performed data collections in monitoring or research projects. Ideally, indicators of ecological condition should be represented by **time series of data** to allow for trend analyses of ecological condition. That is rarely the case though, as historical data are lacking for many relevant indicators. However, the use of defined quantitative reference and threshold values within the IBECA framework allows for inclusion of indicators with low temporal resolution and coverage, making it flexible in relation to different monitoring efforts and data availability across ecosystems. Due to temporal mismatch across monitoring systems and other data sources, implementations of IBECA have so far worked with defined periods for condition assessments, without analysing trends for ecosystem characteristics or the ecosystem as a whole. Instead, qualitative discussions concerning data quality and coverage support the single period quantitative assessments in order to discuss potential changes over time (*cf.* Framstad et al. 2021). Data that do not fulfil the above described requirements for indicators within IBECA, but in principle can be indicative of ecological condition, can be used as **supplementary variables** adding qualitatively to the evaluation of ecological condition (see Framstad et al. 2021). Even though they cannot be scaled and aggregated like operationalized indicators, assessing such supplementary variables may support the index-based assessment, and lead to the development of additional indicators, especially concerning data with good spatial representativeness and temporal coverage but without defined reference- and limit values.

Good **data availability and transparency** facilitate a smooth and efficient assessment of ecological condition, and are instrumental for IBECA to reach its full potential. **IBECA is testable** and both indicators (i.e. their threshold values) and principles for scaling, aggregation and estimation of uncertainty **can be updated** as the empirical knowledge basis improves. Once updated, earlier assessments performed with IBECA can easily be re-run to provide **consistent and thus comparable assessments** in both time and space. Therefore, continuously developing FAIR data standards (*cf.* Wilkinson et al. 2016) for the data used in or relevant for IBECA are key to ensuring (i) optimal access to data on both existing and prospective ecological indicators, and (ii) better dataflow. Along the same lines, we also suggest the installation of an openly accessible digital solution for **documenting key information on indicators** for ecological condition: indicator concept, data basis, as well as reference, limit, and minimum/maximum values for each indicator in a given ecosystem. This information will continuously be subject to additions as new indicators are being developed, and revisions as additional data/insights on existing indicators will improve our knowledge basis.

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