Philosophy of Biodiversity

Chapter 16: Biodiversity indicators need to be fit for purpose

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Background

The International Union for Conservation of Nature (IUCN) estimates that roughly 22,000 species worldwide are threatened with extinction (IUCN, 2014). This number is predicted to increase due to the impacts of human activities (Sala et al., 2000), but also as more species are described and have their extinction risk assessed. As a consequence of the extinction of species, intergovernmental policies and agreements have been developed to conserve biological diversity, including the Convention on Biological Diversity (CBD), which has three main objectives: (1) conservation of biological diversity, (2) sustainable use of the components of biological diversity and (3) fair and equitable sharing of the benefits arising out of the utilization of genetic resources (Convention on Biological Diversity, 2003).

What is biodiversity? The term 'biodiversity' is a contraction of 'biological diversity' and is meant to capture the structural, functional or taxonomic heterogeneity of biology (Sarkar, 2008). The term was coined by Walter G. Rosen in 1986 (Sarkar, 2002) and its global legal framework defined by the CBD in 1992 (United Nations Environment Programme, 1992). The CBD define biodiversity as "the variability among living organisms from all sources including, *inter alia*, terrestrial, marine, and other aquatic ecosystems and the ecological complexes of which they are a part; this includes diversity within species, between species and of ecosystems" (Convention on Biological Diversity, 2003).

The term 'biodiversity' is inherently connected to the term 'conservation biology', which encompasses a social goal i.e. to conserve biodiversity. The concept of biodiversity thus has a normative component, along with scientific and descriptive components (Sarkar, 2008, Callicott et al., 1999, Norton, 1987). There are also normative decisions surrounding what components of biodiversity should be conserved (Sarkar, 2002). Justifications for conserving biodiversity range from biodiversity possessing intrinsic values (i.e. biodiversity has values 'in and for itself') to instrumental values (i.e. considered valuable by humans such as harvested forest products or species with medicinal values, Justus et al., 2009, Maguire and Justus, 2008, Norton, 1987, Minteer and Miller, 2011).

Biodiversity indicators are defined by the CBD to be "information tools that summarise data on complex and sometimes conflicting issues to indicate the overall status and trends of biodiversity" (Convention on Biological Diversity, 2003). Such indicators are employed to motivate changes in land use and management, and to assess the response of biodiversity to policy interventions and other actions (Butchart et al., 2010). Consequently, biodiversity indicators are a key component of efforts to mitigate biodiversity loss. Given the multiplicity of biodiversity, it can be measured in numerous ways (Sarkar and Margules, 2002, Caillon and Degeorges, 2007) and therefore no single indicator will likely describe biodiversity as a whole.

A range of criteria have been developed for appraising indicators (Heink and Kowarik, 2010b) including their feasibility, efficiency, responsiveness, timeliness, relevance and effectiveness (Gregory et al., 2005, Benedek, 2014, Lamb et al., 2009). One of the key criteria for indicators is that they are ***fit for purpose*** (Mace and Baillie, 2007, Vačkáři et al., 2012, Jones et al., 2011). Therefore, biodiversity indicators must measure the right things, as determined by the objective of the indicator or the purpose for which it will be applied (Mace and Baillie, 2007). Biodiversity indicators should also reflect the norms, values and goals of society and the ethical motivations for conservation (Minteer and Miller, 2011, Robertson and Hull, 2001). However, such normative components are inherently difficult to measure objectively (Heink and Kowarik, 2010a).

In this chapter we: (a) identify a core set of criteria for reviewing indicators, (b) critically review a suite of popular indicators according to these criteria, and (c) identify ways to improve the extent to which indicators are fit for purpose.

We conducted an extensive literature search to compile a representative sample of the biodiversity indicators that are in current use. The literature search was performed using ISI Web of Knowledge and Google Scholar in March 2014. Search terms included 'biodiversity assessment', 'biodiversity indicator', 'biodiversity index' and 'biodiversity tool'. An effort was made to search for biodiversity indicators from all continents by combining the above search terms with country names. This exhaustive process revealed that there was a small set of distinct biodiversity indicators, which have been applied across numerous continents. The five indicators chosen for evaluation (Table 1) were selected because they: a) are focussed on biodiversity assessment; b) are measured at a national scale or above; c) are documented in the peer-reviewed literature; d) cover a wide range of biodiversity components among them (e.g. particular taxa through to whole ecosystems); and e) have distinct characteristics to the other indicators selected for review.

Table 1 The biodiversity indicators reviewed, with example applications and key references.

Indicator	Example applications	Key references
Red List Index (RLI)	• Status and trends of extinction risk of bird species	(Butchart et al., 2004, Butchart et al., 2007)
	• Status of grasslands in Japan	(Koyanagi and Furukawa, 2013)
Living Planet Index (LPI)	• Global vertebrate abundance	(Collen et al., 2009, Loh et al., 2005)
Nature Index (NI)	• Ecosystems of Norway	(Certain et al., 2011) (Skarpaas et al., 2012)
Natural Capital Index (NCI)	• Vegetation of Hungary	(Czúcz et al., 2012)
Wild Bird Index (WBI)	• Farmland birds in United Kingdom	(Gregory et al., 2004)
	• Birds of European countries	(Gregory and van Strien, 2010)

The biodiversity indicators were reviewed according to their: (1) objective and utility, (2) representativeness of biodiversity and (3) quality of information. Given the multiple conceptualisations, perceptions and possible measurements of biodiversity, appraising the level of 'fit' of indicators to their intended purpose requires appraisal from diverse viewpoints. We framed a specific perspective to reflect that commonly held by ecological scientists, who typically seek to accurately quantify changes in biodiversity. In contrast, we framed the general perspective to reflect that of policy makers and the general public who might be interested in only the general trajectory of biodiversity change.

Indicators have diverse objectives, but largely unknown utility

All of the indicators were established with normative objectives in mind: to communicate changes to policy-makers and the general public in order to raise awareness of biodiversity loss. The NCI is also noted to have the capacity to achieve a specific normative objective at a local scale of “improving the sense of place” (Czúcz et al., 2012). Typically, reference is made to international policy, such as the CBD, multi-country policies such as the European Union Common Agriculture Policy (Gregory et al., 2005) and national-level policies. To be policy relevant and meaningful, indicators ideally should be related to policy targets (Dennis et al., 2009). For example, the LPI, WBI and the RLI are connected to CBD Target 4[1], the RLI is connected to CBD Target 12[2] and a national application of the RLI to Australian birds demonstrated potential utility for reporting on CBD Target 13[3] (Szabo et al., 2012).

One of the key attributes of effective biodiversity indicators is that they are easily understood and amenable to clear presentation (Gregory et al., 2005, Normander et al., 2012). The WBI is an example of an indicator that has influenced national-level policy (Gregory et al., 2004, Aebischer et al., 2000, Chamberlain et al., 2000) and this has been attributed to the synthesis of scientific data into a simple, understandable and meaningful presentation (Gregory et al., 2004). Typically, the indicators are reported as a proportional change or on an ordinal scale (e.g. a dimensionless quantity between 0 and 100) to enhance communication. The 'apparency' of taxa used in an indicator (i.e. whether their decline is noticed by the public) and responsiveness to changes in management is also important for effective communication. The extent that indicators are understood can thus be enhanced through engaging the public in the collation of information underpinning an indicator. This so-called 'citizen science', where volunteers collect and/or process data as part of a scientific enquiry (Silvertown, 2009, Tulloch et al., 2013), is particularly helpful for addressing questions that have a large spatial or temporal scope, such as biodiversity loss (Bonney et al., 2009). Citizen scientists have contributed to bird species population monitoring across extensive regions in Europe allowing the development of indicators for birds, including the WBI (Gregory et al., 2005). The WBI is an example

[1] By 2020, at the latest, governments, business and stakeholders at all levels have taken steps to achieve or have implemented plans for sustainable production and consumption and have kept the impacts of use of natural resources well within safe ecological limits.
[2] By 2020 the extinction of known threatened species has been prevented and their conservation status, particularly of those most in decline, has been improved and sustained.
[3] By 2020, the genetic diversity of cultivated plants and farmed and domesticated animals and of wild relatives, including other socio-economically as well as culturally valuable species, is maintained, and strategies have been developed and implemented for minimizing genetic erosion and safeguarding their genetic diversity.

of how the communication to the public and the media of the changes observed was enhanced through the process of engaging citizens in the collation of data.

To be easily communicated, indicators must simplify complex patterns, processes and phenomena. From an ecological perspective, the objective of the five biodiversity indicators reviewed can be classified in three ways: (a) to measure change over time only (RLI), (b) to measure the current 'state' relative to a baseline (NI and NCI) or (c) to measure change both over time and relative to a baseline (WBI and LPI). An important aspect of the utility of an indicator is whether this information can then be disaggregated (that is, separated into sub-categories or component data for further evaluation), particularly to allow the underlying drivers of change or impacts of policy and management to be evaluated.

All of the indicators reviewed have the stated capacity to be disaggregated. Data on habitat extent and quality is used to populate the NI and NCI and thus the results for these indicators are inherently connected to habitat conversion and degradation. The documentation for the NI suggests disaggregation be to the level of ecosystems rather than particular locations (Certain et al., 2011). The WBI can also be disaggregated to particular habitat types and has been used to evaluate agricultural policy in the United Kingdom (Gregory et al., 2004). Subsets of species for which extinction risk has changed most rapidly according to the RLI can be identified and hence important threatening processes can be inferred (Butchart et al., 2004, Szabo et al., 2012). The RLI has been used for assessing the effectiveness of CITES (the Convention on International Trade in Endangered Species of Wild Fauna and Flora, Bubb et al., 2009) and for differentiating the impacts of African protected areas policies (Nicholson et al., 2012). The LPI has been tested by assessing the impact of fisheries policies and was found to exhibit counter-intuitive behaviour due to over-representation of some taxonomic and functional groups in the indicator (Collen et al., 2013), and contrasting impacts of policies on different groups caused by trophic interactions (Nicholson et al., 2012). As only a handful of analyses such as these have been undertaken, it is possible that the stated capacity for disaggregation (and hence the utility of the indicators) might be different to the realised capacity.

A key criterion of an indicator is the capacity for users to assess the significance of changes relative to a baseline (OECD, 2003). All of the indicators reviewed, with the exception of the RLI, measure a change in state or trends relative to a baseline. For aggregated measures of biodiversity (i.e. the NI and NCI), there are challenges associated with choosing accurate baselines for all the components of biodiversity being evaluated.

The choice of baseline is important to facilitate comparisons and to obtain an accurate assessment of change (Bull et al., 2014). In terms of reference year, the length of the temporal sequence and when the observation commences relative to fluctuations in the data will be important. For example, for a given hypothetical indicator, if the reference year was prior to agricultural industrialization, then it may show a decline to the present time (Figure 1a), if it was at the height of human-induced modification, it may show no change to the present (Figure 1b) and if it was at a period of slowing modifications it may show a small increase to the present (Figure 1c). Thus, the choice of reference year ultimately determines whether changes are observed and therefore the usefulness of the indicator for informing or assessing policy. Across and within all indicators, there is however no common reference year as the choice of baseline is determined by data availability (e.g. 1970 for the LPI) or by experts (as is the case for the NCI). This limits the extent to which comparisons can be made among areas, with the documentation for the NCI explicitly expressing caution in relation to comparing regions with different baselines (Czúcz et al., 2012).

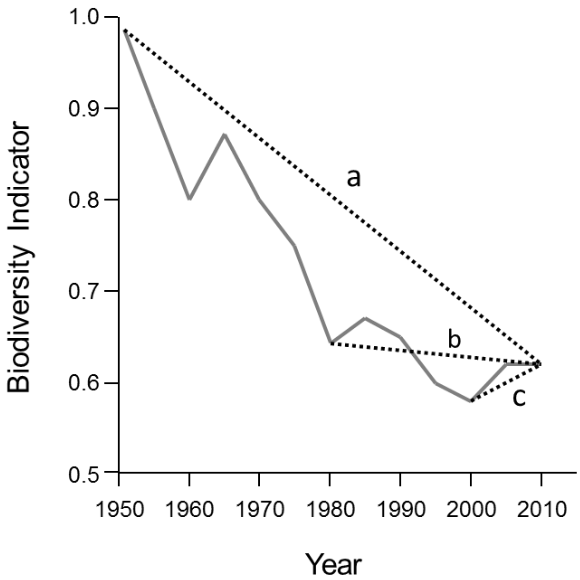

Figure 1. Impact of the reference year used to calculate a biodiversity index. Using an historical reference is likely to show a decline in biodiversity (a), whereas more recent years are likely to show smaller decreases (b), or increases (c) due to minor variations among years.

The choice of reference state for data on habitat quantity and quality is heavily reliant on expert assessment, and as such there is potential for ambiguity. For the NI and NCI, the reference state is assumed to be a ‘pristine’ or ‘intact’ state (e.g. for the NI it is a hypothetical undisturbed or sustainably-managed ecosystem). These notions are dynamic in relation to societal preferences and values and what is feasible to achieve (Hobbs and Norton, 1996). Important questions in relation to choice of reference state is “How does the ‘ideal’ state of an ecosystem differ among parts of society?” and “Is it feasible to return to the reference state?”

It is likely that there will be variation in the response of biodiversity to different drivers of change, with some species declining, some improving and some species displaying an intermediate response. Indicators should be sufficiently sensitive to detect these changes and thus explicitly account for and distinguish natural fluctuations that can arise due to seasonal variations or longer-term environmental phenomena (e.g. due to changes in the Southern Oscillation Index, Gregory et al., 2005). The WBI notes procedures for controlling natural fluctuations and this is achieved coarsely by smoothing the trend lines. However, mean trends (as delivered by the LPI, WBI and RLI) can hide substantial declines for some species if the majority of species are not declining. In the UK application of the WBI, a mean index was created by combining the species trends on a geometric scale, so that doubling of one species' index is balanced by halving of another (Gregory et al., 2004). As a result, the WBI will detect no change when half of the species increase, while the other half decrease, and upward trends could mask substantial declines in several species. In the case of the RLI, slow declines of common species are not well captured and recently-evaluated species may introduce bias (Szabo et al., 2012). While the impacts of using mean trends is not a stated limitation of the aggregated indicators (i.e. NI and NCI), it is possible that the impact is greatest for these types of indicators.

Only a limited subset of biodiversity is measured

Biodiversity is a complex and multidimensional concept that has to be estimated using surrogates, which are assumed to represent the distributions and trends in other components of biodiversity. The choice of biodiversity surrogate is driven by the availability of appropriate monitoring data. Consequently, there is a potential for bias, either explicitly or implicitly, towards commonly-surveyed taxa (e.g. birds, mammals and butterflies Boakes et al., 2010). Indeed, the biodiversity indicators reviewed that are focused on specific taxa are biased towards birds and mammals (RLI, WBI and LPI). There is also variation in the relative abundance of species that are included ranging from threatened species (for the RLI), common species (for the WBI) or a combination (for the LPI and NCI). For the WBI, common birds were employed as they were identified as being diverse, widespread and mobile, high in the food chain and responsive to environmental change (Gregory and van Strien, 2010). For the LPI, it is asserted that population trends for species may act as a proxy indicator for the state of the ecosystem that the species inhabits and by inference wider biodiversity (Loh et al., 2005). By assessing the ecosystem as a whole (in terms of habitat quantity and quality), the NCI and NI may be more representative of biodiversity than species-based indicators.

The indicators reviewed mainly focus on terrestrial birds and mammals and therefore account for only a small proportion of total biodiversity (Figure 2). Ecologists have long debated the issue of whether single taxa should be used to make conservation decisions (Franklin, 1993). Different types of species have been theorised including 'ecological indicators' that respond to stressors, 'keystones' upon which a large part of a community's diversity depends, 'umbrellas' that if protected will conserve a suite of species, 'flagships' that are charismatic and can motivate political and societal change, and 'vulnerables' that are prone to extinction in human-dominated landscapes (Noss, 1990). However, there is limited evidence that a small subset of taxa can capture the breadth of biodiversity and may do no better than a random subset (Andelman and Fagan, 2000). For example, vertebrates have been found to be poor predictors of the distribution of plants and plants to be poor predictors of invertebrates (Oliver et al., 1998). Basing trend assessments on the abundance of common species may mask changes for vulnerable species. General rules for surrogacy are yet to be found (Lewandowski et al., 2010, Grantham et al., 2010, Margules and Sarkar, 2007) but existing evidence suggests that birds and mammals alone are unlikely to provide useful surrogates for all of biodiversity. Ecosystem indicators, which measure habitat extent and quality, may overcome some surrogacy issues since habitat area is a well-established predictor of biodiversity (MacArthur and Wilson, 1967). However, the relationship between biodiversity and habitat quality is still poorly defined and understood (Fischer and Lindenmayer, 2007).

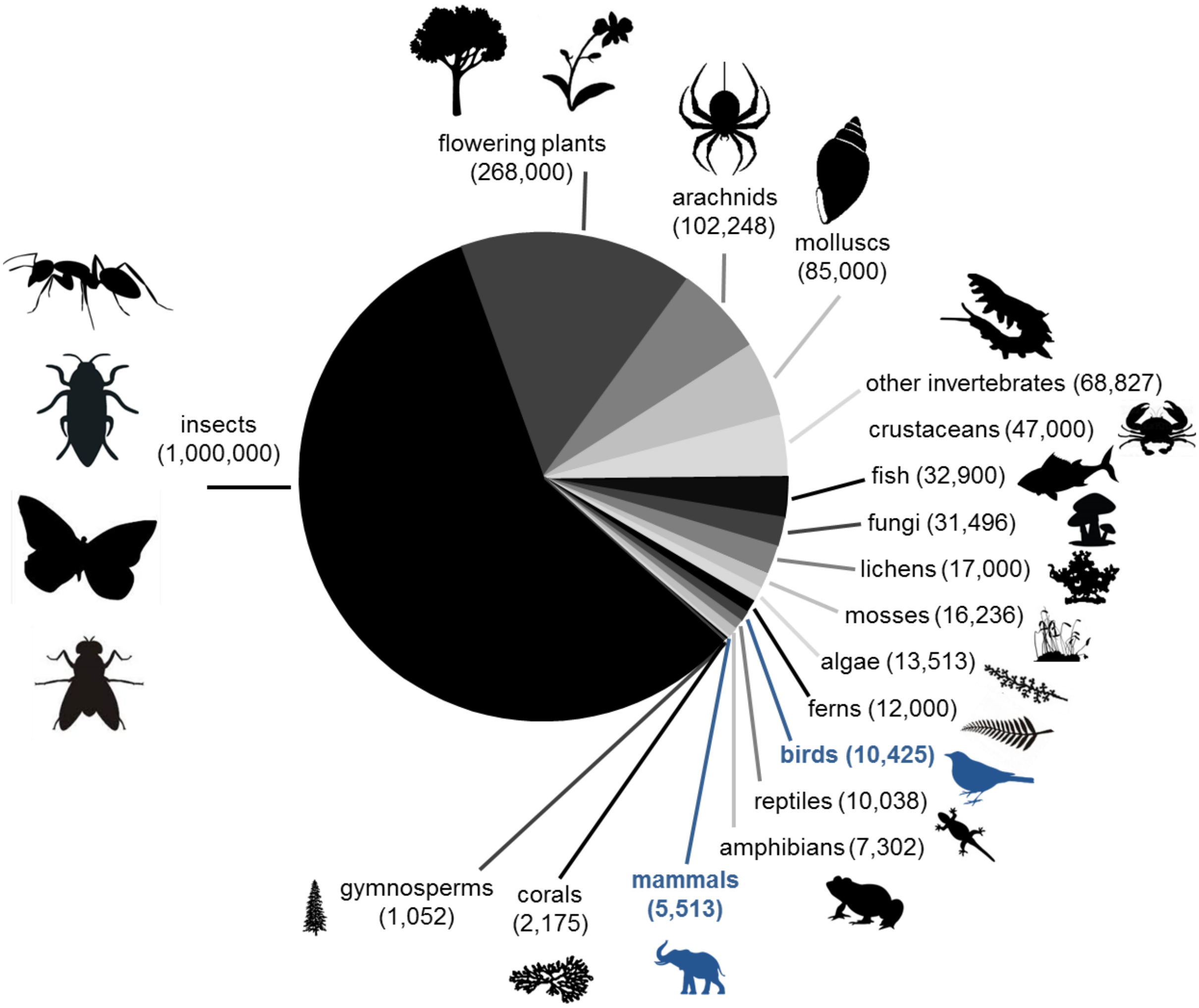

Figure 2. The number of described species for different taxa (number shown in brackets) (IUCN, 2014). The IUCN Red List, on which the RLI is derived, has assessed 76,000 and the goal is to extend this to 160,000 species by 2020.

There is little evidence that other cultural, ecological or evolutionary processes are embodied in the indicators, the exception being the NI, which when applied in Norway included ecosystem services (Certain et al., 2011). Abundant species are noted in the LPI to influence provisioning and regulating services provided by ecosystems (Collen et al., 2013), although this relationship was not quantified. For several of the indicators (LPI, NI and NCI), biodiverse ecosystems are assumed to be more resistant (e.g. to invasion by introduced species) and resilient (i.e. will recover after disturbance, (Certain et al., 2011, Collen et al., 2013, Czúcz et al., 2012). However, the role that biodiversity plays in delivering these values is poorly understood (Folke et al., 2004, Hooper et al., 2012) and it is argued by some that these values could be provided by simplified ecosystems (Kareiva et al., 2007, Worm et al., 2006, Isbell et al., 2011).

A broader ecological view would be to survey change in population abundance for a representative range of animals, plants and fungi from marine, freshwater and terrestrial environments. This is the approach taken for the Sampled Red List Index (Butchart et al., 2007). The sampled approach to the Red List Index (Butchart et al., 2007) was developed in order to determine the threat status and trends of lesser-known and less charismatic species groups in an attempt to provide a more broadly representative picture of biodiversity change. The index is based on a representative sample of 1,500 species selected for a number of taxonomic groups within vertebrates, invertebrates, plants and fungi. Furthermore, although indicators are driven by current data availability, they could be more targeted in the future as this is the rationale for the development of a consolidated set of variables referred to as the Essential Biodiversity Variables (EBVs, Pereira et al., 2013). The EBVs align with CBD targets and comprise measures of genetic composition, species composition and traits, community composition, and ecosystem structure and function. The indicators reviewed here are largely relevant to the candidate EBV classes of species population (RLI, LPI, WBI), and ecosystem structure and function (NCI, NI), with some relevance to genetic composition in the case of the RLI.

There is a tension between the quality and coverage of information

A desired quality of biodiversity indicators is that they are scientifically sound, being based on verifiable data that has been collected using standard methods, with known accuracy and precision (Dennis et al., 2009). Furthermore, accurate population trend analysis (as required by the RLI, WBI and LPI) relies on systematic monitoring programs with adequate repeat surveys (ideally annually) and extensive spatial coverage (Rhodes et al., 2006, Gregory et al., 2005, OECD, 2003). The availability of data on biodiversity that meet such criteria presents obstacles for achieving equitable coverage, particularly for data-poor areas.

Data poor areas typically coincide with the deforestation/degradation frontier and thus are where information is urgently needed to facilitate quick responses to mitigate the loss of biodiversity. Tropical regions are an epicentre of the global biodiversity crisis because they are the most specious and the most rapidly changing. In tropical regions monitoring data is typically sparse, along with general biological knowledge (Lawler et al., 2006), the availability of experts, and of how well land-use impacts are understood (Law and Wilson, In Press). Data coverage for the LPI, for example, is explicitly noted to be better in temperate than tropical regions (Collen et al., 2013). Future development of biodiversity indicators must prioritize the facilitation of timely and efficient assessment in biodiverse but information-poor regions.

The NI and NCI use data on habitat extent and quality, and these data types may be more readily obtained in data-poor areas through remote-sensing technology or land-use inventories (Kerr and Ostrovsky, 2003). However for the NCI, there is vagueness in relation to the metric that should be used to assess habitat quality (Czúcz et al., 2012) and even indirect proxies such as data on threatening processes (e.g. logging) can be included. Furthermore, relatively fine-resolution (e.g. < 1 km^2) ground measures or field validation of habitat quality data is required (Czúcz et al., 2012). The NI employs a range of information sources, which may be useful in data-poor areas, although care must be taken when combining data from different sources to ensure currencies are commensurable and to minimise duplication of information. Regardless of the type of input data, data that are measured in a consistent way (e.g. using comparable techniques at similar resolutions) are required to ensure valid comparisons among areas and between times. However, all of the indicators, with the exception of the WBI, are not underpinned by widely-accepted survey techniques.

While the rationale for the construction of the indicators and the process required to populate these with data is quite well documented, there is subjectivity in relation to the choice of weightings and how these are applied, particularly for the RLI, NCI and the LPI. For the LPI, all declines, regardless of how close they bring a species to extinction, are considered equal (Collen et al., 2013). For the NI, all major ecosystems are weighted equally while some component indicators are given more weight. By combining quantity and quality into one indicator, the NCI relies on a hypothetical equivalence in terms of ecological value between smaller intact patches of habitat and patches that are larger, yet degraded. This might be acceptable for species with generalist habitat requirements but not for species with specialist requirements such as those requiring intact habitat over expansive areas. All weightings, even equal weighting, represent a choice that requires clear justification. An important question is "who should choose weightings?" Should it be scientific experts, policy-makers or stakeholders that are impacted by biodiversity change?

All of the indicators are affected by error in the underlying data, sampling bias, sampling intensity and the timeframe assessed. The RLI, LPI and WBI allow for the estimation of confidence intervals on the rate of decline (and for the WBI trends lines are smoothed to control for sampling error). The NI allows confidence intervals to be generated for each ecosystem type based on three sources of data uncertainty: numerical uncertainty, data uncertainty and uncertainty because of lack of knowledge (Certain et al., 2011). In comparison, the NCI does not provide a measure of error although the results have been demonstrated to be sensitive to data and weightings employed for the quality measure (Czúcz et al., 2012). The establishment of systematic monitoring approaches would reduce uncertainty associated with reported values, although strict standards for data collection and analysis could further limit the spatial and temporal coverage of indicators.

Conclusions

All of the indicators reviewed here have been developed with a normative objective in mind (mostly to inform international biodiversity policy and associated targets or for communication). Ideally, biodiversity indicators should connect the field of ecological science with policy making or science communication. There are four main points of concern that ecologists and policy makers should keep in mind while assessing indicators. First, there is a tension between the needs of ecologists and those who form policy. Second, the indicator species that are often used are unlikely to be representative of biodiversity. Third, the spatial and temporal coverage of information is uneven. Finally, the indicators have been rarely used to evaluate the impacts of policy interventions.

Ecologists are generally concerned with the quantitative attributes of an indicator, such as whether an indicator includes accurate data with an assessment of error or whether the indicator is representative. In terms of its development, the LPI represents a special case. The LPI was developed initially as a communications tool for a World Wildlife Fund campaign. However, the increasing role of the LPI as a policy tool for monitoring progress toward the CBD and other biodiversity targets, initiated efforts to improve the robustness, sensitivity and representativeness of the indicator (Collen et al., 2009, Collen et al., 2013). While the accuracy of indicators is important to scientists, it is potentially less important for the normative purposes of biodiversity indicators. The WBI is the only indicator that uses survey data of a defined standard and all indicators report mean trends, but does this matter? If this question is resolved in the early stages of indicator development then the level of fit of the indicator will be enhanced. Ultimately, indicators are intended to influence political decisions about the allocation of resources to an issue that society deems important (Robertson and Hull, 2001). However, the utility of the indicators to identify the underlying drivers of change and the impacts of policy is yet to be extensively tested. Instead a large amount of research and development has been focussed on the accuracy and representativeness of indicators. Ultimately, to be fit for purpose, the design and testing of indicators should be user-driven (Figure 3), involving policy makers, scientific experts, environmental groups, citizens and other important stakeholders (Dennis et al., 2009).

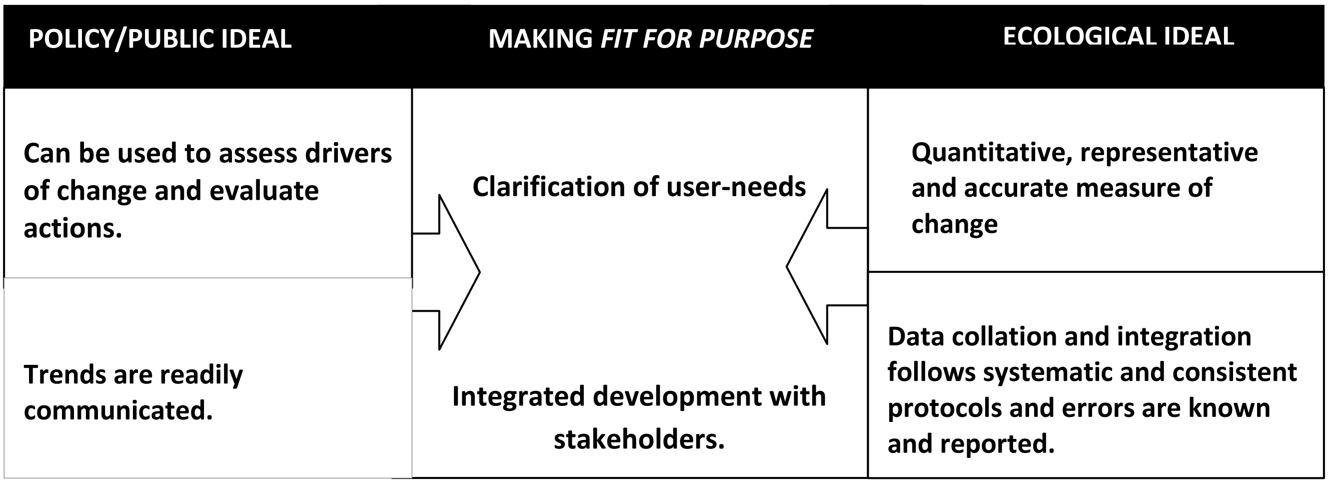

Figure 3. An integrated approach to improve the extent to which biodiversity indicators are fit for purpose.

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