Development of dynamic biodiversity indicators of success for Sites of Special Scientific Interest (SSSI)

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Foreword

This think piece was commissioned in order to support the Natural England's SSSI Future Reforms project. The requirement was to further explore the development of 'whole biodiversity assemblage monitoring' and what this could entail, in the form of a mini think-piece. More thinking is required and advice on next steps, investigating the practicalities and challenges associated with developing and implementing 'whole biodiversity assemblage monitoring' as a method used to assess SSSI / protected area condition.

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Executive summary

Biodiversity indicators are vital when assessing progress of conservation actions against policy goals and objectives. They form a key part of international frameworks, including the Convention on Biological Diversity (CBD), and future national frameworks such as the 25 Year Environment Plan. However, with species communities changing rapidly in response to Anthropocene drivers, it must be ensured that indicators represent both the gains and losses experienced by species communities. Indicators and the policies to which they are linked must also be firmly rooted in an international context. Based on these principles we propose expansions and adjustments to the current suite of indicators so that they enable acceptance and facilitation of ‘desirable’ future changes, as well as maintaining species and ecosystems that are globally rare and endangered.

1. Metrics that are widely used by ecologists (for example, species richness or evenness) are not often used as biodiversity indicators in a policy setting but have the advantage of being neither intrinsically good nor bad.

2. Species attributes have potential to be developed into additional indicators that are likely to be associated with different functional processes, by combining trait data with information on abundance and distribution trends.

3. We recommend that value-based indicators be aligned with international biodiversity conservation goals (internationally threatened and near-threatened species, and species for which Britain supports an important part of the distribution and/or population) so that conservation efforts to resist losses and facilitate gains are focussed on the highest priority species.

4. Factoring in the future changes via a combination of modelling and structured conservation assessment is recommended to highlight the potential to facilitate future gains and to ensure that potential local losses are not resisted if they are inevitable.

5. Ecological indicators, global value indicators and climatic potential can be applied nationally, for networks of protected sites and to individual sites. Indicators can be used as measures of change through time, and they can also be used to provide measures of value at a given time, which may be particularly useful when surveys to evaluate the condition of SSSIs or other protected areas are intermittent.

We recommend the inclusion of a range of public as well as professional perspectives so that indicators, and site management plans that stem from them, are co-produced in an inclusive manner. We suggest that a trial process could be created initially for the development of broad-scale indicators, of site and conservation network value, and of site management plans before new schemes are rolled out more widely.

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Introduction

Biodiversity indicators provide understanding of the status of biodiversity, and they are also used to develop conservation priorities and to evaluate the outcomes of conservation interventions. In the UK, biodiversity indicators have typically been applied to sets of priority species and to the composition or condition of ecosystems. Priority species are typically those with small geographic ranges in the UK, or with declining UK populations (Burns and others, 2021; Barwell and others, 2020; BRIG, 2007). Priority ecosystems or 'habitats' are commonly those which are localised and that have declined in the UK, or ecosystem types that are otherwise regarded as degraded. This approach was entirely rational under a presumption that species distributions and abundances and the characteristics of habitats were relatively stable. Protecting rare and declining species and representative habitats would be expected to stabilise changes to biodiversity, and hence indicators based on these attributes would represent a sensible measure of conservation success (relative to the expected performance of the indicators without such interventions - although this direct comparison is rarely made).

Biodiversity and conservation indicators are not easy to develop or interpret, however, especially in the context of ongoing environmental and ecological change. Some indices are self-fulfilling descriptions of the declines of particular species and habitats, rather than an overall assessment of the state of nature. Others risk 'confusing' departures from the historic state of biodiversity (replacement of some species with others) with net decline. For example, the C4a. Status of UK priority species – Relative abundance (Defra, 2020) indicator specifically collates information on priority species (for which measured decline is one qualifying criterion), and hence this metric is likely to decline even if the declining abundances of these species were replaced by increased abundances of other species. An example of the second type might be if the composition and relative abundances of species (such as National Vegetation Classification categories) is used as one metric to assess the condition of SSSIs or other protected areas, measuring the current composition against that which was described (or desired) when the protected area was first gazetted. The presence or arrival of new species may be deemed detrimental[1].

Changes to the distributions and abundances of species are widely observed in response to human-mediated and other environmental drivers. These responses are generally 'adaptive', in the sense that they increase the match between suitable environmental conditions and a species' distribution. Likewise, changes to biological communities at a given location normally increase the relative abundances and presence of species

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[1] The presence of "*species indicative of unfavourable site conditions and non-native species*". Jefferson, R.G., Smith, S.L.N. & MacKintosh, E.J. 2019. Guidelines for the Selection of Biological SSSIs. Part 2: Detailed Guidelines for Habitats and Species Groups. Chapter 3 Lowland Grasslands. Peterborough: JNCC..

adapted to the new conditions. These changes represent adjustments to the new conditions and consist of gains (local increases of some species, and range extensions and immigration into communities) as well as losses (declines and local extirpations). Hence, it is important that biodiversity indicators take account of gains in new features, as well as losses. Such gains are often ignored by the aforementioned indicators, unless they are already on a prescribed list of desired species, such as reintroductions due to conservation actions.

Judgements of decline or degradation are often made with respect to some measured or surmised historical (or otherwise preferred) state of species or ecosystems. For species, this may be the measured abundance of selected species at the time that monitoring started, such as the first years of the Common Bird Census (subsequently BBS), Rothamsted Insect Survey (for moths) and Butterfly Conservation transects. Comparison to a single time point can be problematic, especially for species that show fluctuating dynamics. The desirability of comparing to a certain baseline is also dependent on when that baseline was set relative to particular environmental drivers of change, and where. For example, ecosystems and taxa heavily affected by a particular form of pollution, which is then controlled by national or international legislation, would show very different trends depending on both where and when the baseline was chosen relative to the legislative changes (consider indicators for lichens and bryophytes or freshwater species (Hayhow and others, 2019; Outhwaite and others, 2020)). Historic baselines are also susceptible to data quality as many schemes have expanded greatly over the years, meaning that initial estimates may no longer be representative. The expansion of biological recording over time and increasing taxonomic coverage (Pocock and others, 2015) alongside advances in analysis methods (Isaac and others, 2014) has provided an increase in the representativeness of metrics but this also generates challenges when comparing recording schemes with different baselines and methodologies across taxa. Rapid changes across the whole biological recording sphere (Isaac and Pocock, 2015) mean that comparison to a particular baseline should always be treated cautiously. Use of such a baseline can also create other undesirable properties such as increasing width of confidence intervals over time reducing precision (Gregory and others, 2019). Hence there are challenges associated with baselines even before one considers the reality of increasing rates of change in the face of pressures such as climate change.

A clear conceptual issue is that any departure from a previous state is liable to be interpreted as a loss or deterioration. For example, if the total number of species and abundances summed across all species in a given location remains constant, indicators based on those species present at the start of a time series will invariably decline and indicators based on species that arrive will increase. Despite no net change, both of these outcomes are commonly treated as negative as they represent departures from the historical state, whether applied nationally or to individual sites (unless there is a specific reason why a new arrival is considered ‘desirable’). This can be illustrated by SSSI designations (Bainbridge and others, 2013), which may list particular NVC categories and the occurrence and abundances of notable species. NVC categories specify approximate relative abundances of particular species, and hence a change in composition might be interpreted as the site being in ‘poor condition’ even if there are just as many species present as previously - and they are regarded as equally important nationally (some notable species decline but others colonise). Colonising species are not included within those specified baseline conditions, and hence there is no compensation for losses, and a potential bias towards considering that the state of SSSIs has deteriorated during a period of biotic change. The new arrivals may even be treated as directly harmful because they represent a departure from the baseline. Such constraints could potentially impede conservation efforts. Faced with high levels of species turnover, resistance of change is no longer feasible in many cases and may have high opportunity costs. Conservation and management strategies must look at which changes to accept and those that can be facilitated (Thomas and others, 2021). Indicators are a key component of this process. However, they need re-alignment to ensure they are accurate portrayals of the state of biodiversity during a period of inevitable species turnover and abundance changes, they are not predisposed to suggest resistance as the most favourable conservation action and they embrace a global perspective so that conservation actions are targeted where they are most needed.

In this report we primarily consider species-based indicator metrics, starting with those that are regularly used by ecologists. We then outline how human preferences for particular species (for example, conservation priorities) and ecosystem services (for example, functional consequences) can be combined with species trends to establish a flexible set of indicators. In the concluding sections, we then briefly propose ‘next steps’ towards moving from these principles to designing specific metrics[2].

We propose the following extensions and additions to the existing indicator suite to increase their applicability in a time of rapid change:

a) The use of new indicators based on ecological metrics and adjustments to existing indicators so that they convey the complexity of change rather than referencing existing human values (values are only incorporated during subsequent interpretation).

b) Incorporating species attributes to better understand the causes and consequences of change.

c) Alignment with a global outlook.

d) Consider likely future changes to allow for facilitation of those changes that might be considered desirable.

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[2] Remote-sensed metrics of ecosystem extents and productivity are also valuable, but beyond the scope of this report.

Ecological indicators

The core metrics that ecologists use to measure biodiversity are surprisingly under-represented among indicators that are used in a policy context. In many respects, however, they are preferable to many existing indicators because they are simply measures of biological diversity. None of these metrics is intrinsically good or bad. Value judgements (if any) are only applied when they are interpreted, and hence underlying human ‘desires’ are only applied at a later stage, in the translation of information into subsequent proposed conservation interventions. In this section, we provide a non-exhaustive list of useful metrics, and provide thumbnail explanations of how they might be interpreted in a conservation context.

a) Species richness. This is a count of the number of species present in a given location, which might be a survey trap or observation transect, a site (for example, a SSSI), a grid square of a specified dimension, or any other defined region. There may be taxonomic inclusion criteria (linked to the availability of data), but it is a species count (controlling if required for recording effort). There are no value-based inclusion criteria for species (such as nativeness, current priority etc.). It is an unbiased metric. Species richness at recorded locations has increased in some locations in recent decades and declined in others (Dornelas and others, 2014). In the context of conservation, high or increasing species richness are usually regarded as ‘positive’, except where low diversity ecosystems provide other values (such as carbon sequestration, particular species, culturally-appreciated landscapes). In principle, other components of the Convention on Biological Diversity’s definition of ‘biodiversity' could also be measured, but this is less common: gene richness (but data rarely exist) and ecosystem or ‘habitat’ richness (the mixture of ecosystem types in a given area). All subsequent metrics also have genetic and ecosystem-level equivalents but, for brevity, we restrict the discussion to species - see metrics g to j).

b) Evenness. Where the abundances (or cover, or biomass) of species are measured, evenness can be calculated, describing whether some species dominate communities or whether all species are approximately equally abundant (for example, Pielou’s Evenness Index). Evenness is typically most appropriate to calculate within taxonomic and trophic groups. *Relatively* high evenness is commonly regarded as ‘more biodiverse’, and hence favoured in conservation.

c) Beta diversity. Beta diversity is a measure of how different the species composition is from one location from another, measured either as species presences and absences (for example, the Jaccard Index) or abundances in different locations (for example, the Bray-Curtis Index). Changes in beta diversity reveal whether communities are becoming more or less homogeneous over time. High beta diversity is commonly regarded as valuable in conservation (for example, ‘habitat heterogeneity’ is commonly described in positive terms, while ‘homogenisation’ is described negatively) because the total number of species protected across a network of sites will be increased, increasing the regional pool of species. This is very important for the SSSI network due to its aims to be representative and contain characteristic examples of a range of species communities (Bainbridge and others, 2013).

d) Turnover. This is closely related to beta diversity, but measuring differences in the composition of communities over time rather than space (using the same metrics). Knowledge of how rapidly communities are changing in different locations and regions is a critical input to RAF assessments (Resist-Accept-Facilitate change; Thomas and others, 2021).

e) Gamma diversity. Gamma diversity is a count of the number of species in a region and is generated by a combination of species richness (number of species at each location) and beta diversity (compositional differences between locations), with all species counting equally. In principle, this would be one valuable metric when countries are reporting on CBD targets (is a country supporting populations of more or fewer species over time?). Newly-arriving species would be included in these metrics because metrics based on the number of species initially present (at the beginning of a time series) can only go down.

f) Species trends. Multi-species indicators, representing an overall average trend across a group of species, have been widely used by both government and non-governmental organisations (Fox and others, 2015; Hayhow and others, 2019; Defra, 2020; Burns and others, 2020). These indicators have the advantage of being able to provide a single metric of change and hence they can be used to evaluate progress against a specific target. However, simplicity comes at the cost of a large loss of detail (Rowland and others, 2021), and they suffer from the baseline issues already outlined. In addition, biodiversity indicators can be governed by extremes if many individual species trends are reduced to a single indicator (Leung and others, 2020), and it can be difficult to represent the uncertainty and variability associated with reductionist indicators (Rowland and others, 2021). Extremes are of interest to conservation and other managers, but such indices are not necessarily representative of the wider state of biodiversity. This problem could be exacerbated if species with extreme trends, very rare species and those with spatially-dynamic populations are more likely to be priorities for indicator inclusion. The inclusion of more species is likely to make the results increasingly representative, but there are circumstances where the inclusion of only certain species groups can be beneficial, as detailed in subsequent sections.

Single indicator values can be useful but require information presented alongside them, for example measures of the influence of each species on the trend. This could be achieved with methods based on individual exclusions and recalculations, or leverage calculations (Freeman and others, 2001; Gregory and others, 2019), making explicit the contribution of any outlying trends. The inclusion of the full frequency distribution of status changes (abundance and/or cover and/or distribution) of all species within a given indicator (including new arrivals) provides a balanced measure of changes to species (from the greatest losses to the largest increases). Methods to achieve better representation of this distribution are available (Dennis and others, 2019; Bowler and others, 2021) and can provide an improvement on the widely used categorical system. Greater consideration of the distribution of trends can also improve the representation of uncertainty and variability around indicator estimates with methods such as bootstrapping, jack-knifing and the use of quantiles relevant to asymmetrical distributions (Rowland and others, 2021).

Further work is needed to fully understand the implications of baseline use. The use of the first year may not be the best option, particularly in ‘noisy’ ecological systems, as can be seen when comparing UK moth and butterfly recording schemes, with the latter commencing in an extreme year (1976) for Lepidoptera abundances (Palmer and others, 2017). A number of options exist, including taking multiple start dates, an average of all years or incorporation of uncertainty into the baseline (Gregory and others, 2019; Freeman and others, 2021). Although all alternatives have limitations (Gregory and others, 2019), it is important that the implications of how baselines are chosen and used as are fully considered.

g) Phylogenetic diversity. Metrics a-f treat species as entities, but species only represent one step in a continuous variation in the hierarchy of biological life, ranging from differences among individuals and populations, through races, subspecies and species, to genera, families and higher taxa. Species metrics are practical, but versions of the above could also be estimated using measures of phylogenetic diversity (measures of average branch lengths among species found at a given location - full phylogenetic trees are increasingly available). These have potential to provide additional insight. For instance, the loss of the evolutionarily-distinct great auk (the only species in its genus) would represent a greater reduction in phylogenetic diversity than loss of the short-haired bumblebee (many other *Bombus* species survive). In principle, genetic diversity could be included directly but, if data are lacking, racial or subspecific variation can potentially be included heuristically.

h-i) ecosystem metrics. Given inevitable turnover of species over time, especially as the environment changes, and given that Britain lacks many fully endemic species, measures of ecosystems attributes directly can also have value. These are largely beyond the scope of the current report but can include the following: **h) Total multi-species abundance and cover; i) Primary productivity; j) Biomass**.

Overall, environmentalists are typically in favour of high and increasing values for all of these metrics (except turnover through time) with the caveat that some species are characteristic of species-poor and/or low productivity ecosystems. Low diversity ecosystems may provide other values (such as carbon sequestration, particular species, culturally-appreciated landscapes), so a ‘basket’ or suite of indicators is informative. Uneven communities dominated by one or a few species are sometimes regarded as aesthetically pleasing (for example, bluebell woods, stands of beech trees, heather moorland). Although conservationists are not typically in favour of high species turnover rates, in some senses it is a metric of the rate at which species and communities are adjusting to environmental change (a measure of ecological success/flexibility). The challenges associated with turnover are commonly cultural and administrative. Such changes may be interpreted negatively by conservationists and the wider public as community compositions move away from culturally-appreciated mixtures of species (including those described when SSSIs were gazetted and the identified special feature within them).

All of the above metrics also have genetic and ecosystem-level equivalents but, for brevity, we mainly restrict the discussion to species. None of these metrics has intrinsic value, and none of them is specifically biased towards providing increasing or decreasing trends. They are descriptors to inform subsequent decision making.

Species functions and traits

The majority of the metrics listed above do not require any information on the species monitored except their abundance or occurrence. When considering species richness, for example, all species are counted as equivalent. When considering communities and ecosystems, however, we know that different species have different roles and influences which depend on the attributes or traits of the species in question (Petchey & Gaston, 2006). Therefore, to understand how communities and ecosystems are changing we need to consider the roles and functions of the species that are increasing and declining (and lost and gained at a site level). Metrics based on functional traits can be more informative with respect to changes in ecosystem functions, compared to metrics such as species richness (Gagic and others, 2015). When it comes to indicators that reflect changes in functional diversity there are a number of possibilities.

It is possible to track the functional diversity of different groups of species performing different roles, for example pollination or pest control (Greenop and others, 2021). This can be done by calculating functional metrics for each species and, in its simplest form, could track community weighted mean (and variance) values of important traits. This approach is reliant on the availability of functional trait information, which may be lacking for many taxa. An alternative in this situation is to use richness, abundance or distribution metrics but only considering species which are known to perform a certain function. This approach has already been adopted with indicator D1c Status of Pollinating Insects (Defra, 2020). A functional group approach could be greatly enhanced by the production of look up tables detailing the functions of each species. This enables individual trends (past, recent, projected in future) to be converted into indices that reflect particular functions (for example, aquatic carnivores or terrestrial detritivores) and societal priorities (for example, known herbivores of crops versus natural enemies of crop pests). These can be applied to a region to assess progress in ‘desired’ directions, including focusing on species important for a particular function, iconic species, and so on. This look up approach is valuable because it facilitates retro-calculation of trends for whichever functions or traits are deemed to be of interest.

As well as being useful in providing indicators of functional change, species attributes can also be used to provide information on likely drivers of change. One of the main limitations of many indicators is that the cause of changes is difficult to understand. A possible method to fill this knowledge gap may be tracking species groups with known responses to certain drivers to indicate the magnitude of these drivers over time. The use of particular species or species groups as indicators of different pressures has long been established, particularly with regard to chemical pollution. Although measures such as total richness provide a good overall summary of trends, such response groups highlight patterns and causes of declines in particular sets of species. Methods to do this based on clustering and ordination techniques have already been used (Dennis and others, 2019; Bowler and others, 2021). Without clear indications of the drivers and their impact, the creation of policy is difficult.

The idea of response groups is already partially considered by existing biodiversity indicators with separate indicators for specialists and ‘wider countryside species’ as well as splits based on habitat associations (Defra, 2020), but note that some species that inhabit the wider countryside are ecological specialists (Platts and others, 2019) and hence measures of specialism should be based on numerical analysis. These could be taken further to focus on species associated with a particular priority or SSSI habitat types (i.e., the overall fates of sets of species that SSSIs originally aspired to protect across a network of sites, or in the UK, even if some of the original SSSI populations are lost). Further extensions could take the form of focusing on a wider range of species attributes, such as plant trends based on Ellenburg N index scores to provide information on the effects of nitrogen deposition (Hayhow and others, 2019). Note that these measures need to be represented neutrally. An increase in average Ellenburg N index scores or community thermal index, for example, likely represent adaptive community responses to nitrogen deposition and climate warming, respectively. They are indicators of the consequences of certain drivers, not interpretations of whether those changes are valued or unwelcome.

Conservation importance for biodiversity

In an age of rapid global change and species redistribution, it becomes increasingly important to take a global perspective on species conservation, given that national, provincial or other political ‘boundaries’ have little meaning to species. Although international conservation obligations and global or European extinction risk do feature in multiple criteria for species prioritisation, this is often alongside criteria for species in decline in the UK (Burns and others, 2021; Barwell and others, 2020; BRIG, 2007). When taken in combination with national red lists, high levels of focus can be dedicated to species that, although at risk in the UK, may be at low risk across the whole of their global range. It is important to distinguish between national and international declines - both are worthy of consideration. While some species may have considerable local or national significance, either ecologically (for example, ecosystem engineers) or culturally (for example, iconic species), conservation actions and expenditure that focus on ‘saving species’ would ideally take a global perspective. The basic issue is that existing metrics commonly conflate the national and international status of species. This is a problem because these two elements will often map differently onto resist, accept and facilitate conservation strategies. Therefore, we recommend a separation of national and international threat and status indicators - it is much easier to identify how the two, separately and in combination, map onto adaptive management strategies than if they are shoehorned into a single metric.

a) Global endangerment. We suggest analysing species considered globally at risk of extinction to provide trend metrics (locally or nationally) for these species. A full discussion will be required of the best approach to take (see Next Steps, below), so we here outline a few possible options for incorporating global threat into an indicator framework. The UK already reports on the status of species of European importance (Defra, 2020), which could be expanded to incorporate all species classed as at risk of extinction globally by IUCN (IUCN, 2021). It should be noted that *Britain* supports relatively few globally threatened and near-threatened species (many of the UK IUCN-listed species are found in UK Overseas Territories), based on IUCN red list designations, and hence *all* of these species are likely to be prioritised by British conservationists. Given the modest number of species involved, revealing the full distribution of species trends (with proportions increasing, stable, declining, as in some of the current indicators) will be insightful. How 'Britain is doing' at protecting this subset of species is currently obscured by the inclusion of nationally (but not internationally) threatened species within the same metrics. If an overall indicator is desired for reporting and policy perspectives, then individual species trends could be weighted by the IUCN designated level of threat (e.g., least concern = 0, near-threatened = 1, and so on).

Assessing the performance of SSSIs, National Parks, AONBs or other designations could use the same metrics, using spatially appropriate data for comparison with national trends. Where data are available, these can highlight the importance of individual sites as well as the network to these internationally threatened species. Conservation prioritisations (spatial planning and resource allocation) can be carried out to ensure that key locations for these species are prioritised and that the entire SSSI network provides significant contributions to international conservation. For internationally-threatened and near-threatened species, resist strategies may be required as a 'holding' measure while new facilitate approaches are developed.

The same approach can be carried out for species designated as nationally (but not internationally) rare and threatened. Establishing which species these are could, for example, be calculated by re-running species prioritisation algorithms without including elements of international endangerment. These can then be converted into trends and site prioritisation metrics, as for the internationally-threatened species.

A great advantage of separating these metrics is that it is possible then to identify which SSSIs (or other sites) are important for both internationally and nationally-threatened species, which are important for one or the other, and which are of relatively low importance to either group (where ecosystem service and wellbeing outcomes might be of greater importance); helping to establish conservation management approaches appropriate to each site. It also becomes possible to compare trends for these two groups of species among sites, enabling managers to establish whether conservation interventions for the two groups are synergistic or antagonistic.

b) Global range size. Global and national endangerment are important, but the UK may also support significant populations of additional species that are listed as Least Concern by IUCN and that are too widespread (or not declining enough) to register as nationally-threatened. National declines might still be regarded as important when a country contains a considerable proportion of the total global population or genetic variation of a particular species. Bluebells and gatekeeper butterflies would move up the conservation ranking by this approach, relative to early spider orchid or Lulworth skipper. The UK can be considered to have particular 'responsibility' for maintaining the non-threatened status of these species. A similar approach to 5a can be adopted to assess both the trends and site-specific importance of these species. These could be based on species weights in multi-species indicators, for example inverse weighting by global range size (a widely used numerical measure to quantify 'endemism'). This approach will identify sites, habitat types and regions that have distinctive biotas globally, and hence ensure that the representation of different biological communities within the SSSI network highlights those that are also required to safeguard international representation.

The purpose and criteria of any indicators produced must be clear. Many current indicators have multiple criteria for inclusion, increasing the difficulty of interpretation. These issues can however be solved by disaggregation into supplementary indicators. As already discussed, when considering the contribution of a nation to global conservation, two groups of interest are species that are globally at risk of extinction and species for which the nation represents a moderate to large proportion of their global range. These two criteria may be mixed to provide a list of priority species for global conservation. The two groups however provide different (if complementary) information - just because a species has a large proportion of their population within a single nation they are not necessarily at extinction risk (and hence one might in some instances prioritise wider-countryside regulations and incentives rather than site-specific conservation management). In general, our perspective is that conservation policy makers, planners and managers will be better placed to make informed decisions when information (and associated institutional guidelines) is deconstructed to ensure that local protection and management are appropriate.

Metrics of global endangerment (potentially weighted by threat level) and global range size (weighted by proportion of range in Britain) can potentially be combined with the metrics in section 3 (ecological indicators) to provide a range of measures regarding the dynamics of species for which a given region has particularly significant populations. Increases by and colonisation (of Britain as a whole or of PAs within Britain) by species that are globally threatened or near-threatened, or that have localised global distributions, will tend to drive these metrics upwards, whereas colonisation by widespread generalists will have little influence. For policy purposes, decisions would be required on whether *additional* metrics are needed that consider the geographic origins of species. For example, water deer are listed as Vulnerable by IUCN, but were introduced to Britain from Eastern Asia and could be removed from some metrics. We see no particular reason to do so because nearly all invasive species are geographically widespread and not threatened, but that is one issue to consider in 'Next Steps' (section 8).

This international perspective highlights the importance of regularly updating UK priorities in the light of IUCN listings updates and engaging fully with the international community. This would include participation in the Bern Convention and working with global data providers and conservation bodies.

Assessing future risks and opportunities

It is important to consider the future trajectories of species under climatic and other drivers of change, given that sites and countries that are important for a species today may not be so in future, and vice-versa. This will inform decisions about when and how to intervene (resist, accept or facilitate change) in the context of observed and projected abundance, distribution and indicator changes, in such a way that positive change can be achieved. Nature 'recovery' needs to be a forwards-looking process rather than guided by attempts to replicate the past. It is important that any indicators or other metrics must remain useful under future changes and facilitate the forward-looking process.

The UK may lose some species which currently reach the southern extent of their global range in the UK but gains under a changing climate are also to be expected - dragonfly species, for example, are currently colonising Britain and Ireland faster than previously recorded (Taylor and others, 2021). Britain may also support a higher fraction of the world population for some species, such as the IUCN near-threatened Dartford warbler, which retreat from hotter and drier parts of southern and/or central Europe. Consideration of the future status of species can help distinguish between instances where conservation managers might decide to intervene, for example to retain and facilitate increased population sizes of populations of globally threatened and small range species (if feasible; if not it may be better to invest in *ex-situ* and *trans-situ* options) versus accept losses in the UK of species that are geographically widespread and secure across Eurasia. Equally, many species currently restricted to southern Britain might in future have the potential to become more widespread, suggesting that modest, short-term interventions may lead to longer-term success without the need for indefinite intervention. Understanding these likely future losses, gains and new opportunities are valuable conservation planning tools.

Risk and opportunity frameworks

We recommend adoption of a climate change (it can be extended to other drivers) risk and opportunity framework that has already been applied to over 3000 species within the UK (Pearce-Higgins and others, 2017) and is inspired by the IUCN red-listing approach that considers abundances and distributions (Thomas and others, 2011; Foden & Young 2016; Foden and others, 2019). Two key advantages of the approach are that it includes explicit consideration of benefits that arise from climate change as well as threats (thereby helping identify species for which opportunities exist to facilitate range expansions and population increases) and it combines recent observed data and modelled future (the balance between the two is considered, but the method can be applied even if historical trend data are lacking). It can use and build on knowledge of distribution and abundance changes in response to recent land use as well as climatic changes.

To date, such assessments of species in the UK have only been deployed for species historically present in the UK (Pearce-Higgins and others, 2017), but this should be expanded to introduced species (a few of which may have globally significant populations in the UK) and to the ‘pool’ of species that might potentially establish populations in Britain in the near to medium future. One would primarily consider European species, some of which may colonise unassisted, others of which may require assistance to cross the English Channel, and yet others that are globally endangered by climate change (e.g., in montane areas of southern or central Europe) but could potentially establish populations in Britain if assisted colonisation programmes were established. This is about identifying candidates for potential action at the geographic scales relevant to climate change. Conservation planning must be done at continental scales (for example, Barbet-Massin and others, 2012), especially during periods when distributions are dynamic. These can then be used to update priorities within each constituent region.

Future scenarios

Decisions on species prioritisation should include the future projected status of each species. Most such projections are based on modelling the distributions (and sometimes abundances) of individual species because species rather than biological ‘communities’ shift their geographic distributions in response to climate change. Multiple modelling frameworks are available, for example using different types of statistical (and similar) distribution or niche models, combined with different GCM climate models, and realised at different times in the future.

It is valuable if projections include information on land use (for example, affecting colonisation potential and likely eventual population size) as well as climate predictors, although land use change scenarios may be difficult to implement many decades into the future. Climate-only models for North American birds predicted extinctions better than colonisations (Illán and others, 2014), while models that incorporated both land use and climatic changes outperformed climate-only models (Betts and others, 2019). Platts and others (2019) demonstrated that up to about half of the between-species variation in the rates at which the range boundaries of southerly British insects were expanding northwards could be predicted using a combination of climate and land use predictors. Paradoxically, for species that are benefiting from climate change, climate may no longer be the most important consideration - the constraints on their expansions are commonly associated with the nature of the landscapes they are colonising.

It is valuable to test the predictive power of models on past data (how well did they predict past distribution changes, given the climate and land use of the last few decades) before adopting them for future scenarios. It is worth emphasising that projections are articulations of potential future conditions that can be used to inform near-future decision-making, rather than actual predictions. The outputs of these models should therefore be combined with expert knowledge to inform risk and opportunity frameworks.

The need to evaluate the effectiveness of management strategies, such as of protected areas, means that the spatial patterns of biodiversity change are important. To be able to evaluate the effectiveness of the SSSI network, for example, we need to know where different changes are taking place. This will require both the use of existing empirical data and modelling. There is unlikely to be sufficient empirical data for particular taxa in many locations to be able examine ‘observed’ change at fine resolution. However, advances in remote sensing technology have enabled mapping of drivers of biodiversity change, often at high resolution. By combining high resolution remote sensing derived variables with what may be patchy empirical data there is an opportunity to predict changes in metrics such as species richness across large scales. This provides two important results. Firstly, it will help to build evidence on how drivers affect biodiversity and which are important. Secondly, biodiversity change can be mapped enabling understanding of where changes are happening and identifying areas likely in favourable and unfavourable condition for biodiversity as a whole, independent of the baseline state. Areas of particular concern for monitoring may be locations that represent the edge of many species ranges, important areas for connectivity or regions where human activity may be especially reliant on ecosystem services (for example, pollination or pest control).

Assessing projected effects of future distributions on indicators

In addition to informing risk and benefit assessment frameworks, the future projections can be used to simulate the future of biodiversity indicators. Characteristically, past distribution and abundance changes have been used to generate indicators. However, most of the above indicators could be constructed using forward-projections, including those for global endangerment and range sizes. By successively leaving species and groups of species out of the analysis, it is possible to identify those that are most likely to lead to either reductions or increases in a given indicator, and hence informing possible resist, accept or facilitate conservation strategies. Projections will also be needed to determine the importance of particular sites and the network as a whole in facilitating the required species movements. This requires the construction of null scenarios. For example, is any increase in the UK’s contribution to global conservation greater than would be expected based on species movements without conservation interventions.

National indicators and Protected Area assessments

Most if not all of the indicators mentioned above can be applied to any given area, be that a nation, administratively- or biologically-defined region, and to PA networks and even to individual sites, provided that adequate data are available. The management and policy requirements may however differ between scales, so complementary suites of indicators are likely to be required. For example, assessing the condition of an individual site is likely to require different considerations to evaluation of the whole network. The key will be to develop local (site, PA network, England) evaluations in an international context. Global conservation priorities and continental projections need to be able to filter down and inform the condition assessments undertaken by decision makers at multiple scales, including individual site managers.

To monitor and evaluate SSSIs, the condition(s) of the special feature(s) for which the site was designated is assessed (Williams, 2006). These special features may be an ecological community, a particular species or an ecosystem. By combining knowledge of a site's management team with information across scales, site assessment can be aligned with the wider framework. A clear possibility would be to define a feature or characteristic as special based on three main criteria - contribution to international biodiversity conservation, contributions to the protected area network as a whole, and future prospects. Based on this approach, an ongoing dynamic assessment of site (or network) features would be part of ongoing reviews, rather than referencing feature assessment to the date of site designation. Sites will lose certain features but gain others (Johnston and others, 2013), and so long as the network representation of priority (internationalised, dynamic) features is retained or increased, the performance of conservation actions might be regarded as acceptable.

Contributions to international biodiversity conservation can be considered as outlined in sections 5 and 6 (and many of the metrics in section 3). Lists of species that are globally endangered or for which the UK holds a substantial proportion of their global range are clear candidates for classification as special features to be monitored (the same approach can be applied to ecosystems, carbon or other ecosystem service features). The list would need to contain species for which the UK is of high future importance, and which can be expected to become more widespread in future (i.e., retaining source populations to fuel future increases). Where data are available, these considerations can be conducted at a more local scale; for example, which species will the protected areas in a selected region be important for in future? National and continental future predictions can be used to provide individual site managers and ecologists with information on how the site they manage will be important in future. Such projections can also be used to outline future visions based on potential scenarios for a given site, which can then be evaluated for its desirability from a local perspective as well as for its contribution to the network.

Periodic condition assessments

A challenge is that the condition of named features of protected areas may only be assessed periodically, whereas most indicators represent annual measures of change. Value (the status of features) and temporal trends in value are not necessarily very closely related. For instance, a species that only survives on protected areas, albeit at reduced numbers, may still be a positive feature despite a negative change since the date of site protection (it is in as ‘good’ a status as possible, given an even steeper national decline). **Therefore, there is utility in developing measures of ‘value’ (at a given time), ‘change in value’ (trend) and ‘relative performance’ (trend relative to unprotected locations)**, rather than merging them all into one overall metric of condition. Some monitoring sites that contribute to national indicators are within SSSI and other protected areas, and thus within-PA trends (indicators) can be calculated for comparison with outside-PA trends.

Contribution to the network

The second way in which the features of a site can be considered special is when network complementarity is considered. If a SSSI, for example, contains a feature not found across the vast majority of the network, then this feature/site is likely to make a contribution to metrics assessing the network effectiveness as a whole. The identification and prioritisation of such under-represented sites would be regarded as ‘favourable’ (valuable) at the network scale. As detailed in the ecological metrics section calculation of beta diversity is one method of identifying sites that are dissimilar to the rest of the network. An alternative option could be to calculate the metrics used to evaluate the network with and without each individual site to understand their contribution to the network as a whole.

As is currently the case (Defra, 2020), condition assessments from individual sites can be collated and summarised to compare status over time, with the information also available for spatial comparisons. In addition to information on which sites in a protected area network are classed as in favourable condition, indicators can also be calculated across the network as a whole and where possible compared to the wider-countryside. A network approach when calculating ecological metrics such as species richness or beta-diversity has the advantage of increased data availability as well as embracing the dynamic nature of biodiversity change. It is inevitable that some species will decline at some sites and others will increase. By taking a network approach we can better understand if losses are offset by gains elsewhere or in other species. Indicators used for the protected area network as a whole can be compared with regional or national indicator trends, providing clarity over the contribution made by protected areas. Ensuring that the same principles operate across levels should ensure that outcomes considered positive at the site level contribute to positive outcomes at the network and national level (and ultimately the global level).

Network value is implied by the SAC and SPA designation thresholds that highlight proportions of the national or European population contained in a location (implicitly, it is linked to endemism metrics). However, inclusion of both European and UK status elements may make it hard to know what it is an indicator of. Clarifying and, where appropriate, separating the components of indicators and value-thresholds can avoid confusion and clarify targets of management.

Next steps and involvement of people

The next steps are to make progress towards updating indicators, evaluating implementation at site, network and national scales, and involving a wider-than-usual range of people in consultations and in the co-production of preliminary results. There is a trade-off when considering the number of indicators used (POST, 2021a). Too few can result in important information being missed and too many can lead to confusion. As discussed earlier, the ability to disaggregate is a possible solution. Although the suites of indicators vary between UK nations, there is a reasonable amount of consistency with many UK level indicators mirrored by those of constituent nations, and all indicator suites are designed to report against the same set of international agreements (POST, 2021b). This reaffirms the utility of the ecological indicators (Section 3) because they are independent of values (which may differ slightly among nations). The challenge is simultaneously to inform (via the elements included in measures of value and indicators) and empower. A development plan is needed.

It is beyond our scope to consider all possibilities here, but we can make a few points.

Indicators

Given the availability of species data that are already used in existing indicators, plus IUCN red listing and global distributional data (for example, from GBIF), it would be relatively straightforward and quick to calculate most of the indicators in sections 3 and 5, backdating them to the start of the relevant recording schemes. It would also be feasible to compare national trends for the subset of sites that are PAs, and to identify individual sites with positive or negative deviations from national trends (for those sites with individual monitoring). These and the historic indicators can then be discussed in professional, public and combined forums to identify preferences and suggest additional or different approaches. Due to the complexity of indicator creation and the data sets on which they are based, it is often difficult to rule out undesirable properties and non-intended counterproductive consequences; hence the desirability of trialling the process before rolling it out widely. The indicator suite must be regularly evaluated (using professional and public consultation) to ensure that it is still fit for purpose and care should be taken that the original goals and real world aims are not lost in pursuit of favourable indicator trends.

Site-specific development

Perhaps the simplest way to develop site-specific resist, accept, facilitate plans is again to try it out for a small number of exemplar sites (perhaps a couple of SSSIs with recently-updated plans and one or more nearby locations in AONBs/National Parks that are not SSSIs). This would enable discussion of how management plans based on historic indicators and current priority species/habitats might compare with those derived from a facilitate, accept, resist framework approach informed by updated ecological indicators and those based on global threat/range size and climate futures indicators. This would enable the development of a process, including the timing and nature of public involvement, and it would allow empirical assessment of whether the resultant management plans might be quite similar or different when informed by different input information and indicators, and adopting different processes.

People at the core

We have not advocated precise means of involving people, or who, in developing a set of indicators because the process will vary, depending on the specific goal. We note that, to date, public involvement tends to be via consultations more often than by co-production, and that participants in consultations are often biased towards landowners, conservation advocates and dedicated recreation groups (in some instances) more than towards more casual actual or potential users and beneficiaries of the environment. Informing and empowering people, which might be carried out nationally with respect to national indicator development and locally with respect to particular sites, may result in adjusted priorities, but this has many potential benefits. Such perspectives are likely to represent current needs and ideals as well as historic cultural visions and imaginings of the potential future state of nature.

Indicators of nature’s contributions to people

Given that conservation is typically justified by the dual criteria of intrinsic value and human wellbeing, there is a case for the development of ecosystem service (for example, ecosystem carbon storage) and wellbeing indicators (for example, recreation). None of the above indicators or value metrics are of this type, although their interpretation does represent a component of wellbeing for those who care about biodiversity. Again, the development of such measures would ideally be a co-production process. These indicators would allow the British government and devolved administrations to judge not only what we are doing to (and for) nature, but also what nature is doing for us.

Conclusions

Designing and implementing an indicator framework compatible with rapid dynamic change is a challenging and complex task that will require wide consultation and evaluation. In this piece we have detailed a range of ideas and concepts that we deem vital to this process.

We recommend that biodiversity indicators include ecological measures and that indicators and site measures of ‘value’ are realigned towards global scale threats. Indicators (and associated management) need to be robust to ongoing global change, and we suggest that a wider range of the public be invited to contribute to decision-making. These adjustments will have the double benefits of focusing attention on global rather than national or local concerns, hence increasing the contributions of British conservation on the global stage and enabling proactive forward-thinking management.

We recommend that a trial development of indicators and processes is conducted to ensure that robust procedures can be established.

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