Discussion and recommendations on the future of protected areas in England under climate change

June 2023

Natural England Commissioned Report NECR479

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Catalogue code: NECR479

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Keywords

Protected sites, biodiversity, conservation, protected sites, climate change, ecological network, SSSI

Citation

Gardner, A.S., Maclean, I.M.D.*, Hopkins, J.J. & Gaston, K.J., 2023. Discussion and recommendations on the future of protected areas in England under climate change NECR479. Natural England.

Foreword

Natural England's SSSI Future Reforms project commissioned several 'Think-Pieces' to inform discussion with stakeholders to develop a vision for what we want Sites of Special Scientific Interest (SSSI's) to deliver in future, and how we can best support the 25 Year Environment Plan to achieve 75% of protected sites in favourable condition by 2043, in the face of inevitable change to the natural world due to the Climate Crises. This report is one such think-piece providing a response to the question:

> We are interested in your thinking on how an 'Ecologically Connected' network (ECN) of protected sites / areas could work in England, based on the following draft vision:
>
> "Creating a large and 'Ecologically Connected' Network of Protected Sites / Areas as a key component of 30 x 30 and the Nature Recovery Network, that is actively monitored and adaptively managed to ensure its effectiveness at conserving bio-geodiversity in the face of dynamic change".

Natural England commission a range of reports from external contractors to provide evidence and advice to assist us in delivering our duties. The views in this report are those of the authors and do not necessarily represent those of Natural England.

Executive summary

Protected areas (PAs) have been established to protect species and ecosystems from threatening processes and will play an important role in nature conservation under climate change. However, there are concerns that climate change could alter the relationships between (dynamic) species range limits and (fixed) PA boundaries.

England's wildlife is already depleted and ensuring that PAs are effective now and in the future is a high priority. In this Think Piece we consider the current designation and monitoring framework for SSSIs. While recognising that SSSIs include both biological and geological features, the scope of this work is to focus primarily on biological features, and ask, 'how can we sustain biodiversity in PAs under climate change?'.

We challenge some of the existing paradigms regarding range shifts, demonstrating that range retractions are far less frequently documented than range expansions, and that most shifts are likely to be quite localised in response to local climate change. We propose more focus on maintaining the viability of regional species populations. Too much emphasis has been placed on enhancing connectivity, which comes at the expense of making sites bigger and better. A proportion of SSSIs are very small and connecting tiny fragments by thin wildlife corridors will not be the best strategy to protect their biodiversity.

To determine the best means of maintaining viability of regional species populations, we propose that England's PA network should be subjected to principles of systematic conservation planning. This type of assessment would provide a basis for identifying the protected sites needed to sustain biodiversity and their optimal configuration. To implement on-the-ground action, we discuss the merits of blending strategic and scientific advice with bottom-up participatory approaches that consider the needs of landowners and the conditions underpinning their willingness to engage in nature recovery.

We then explore how the monitoring of SSSIs could be improved to create a more functional network of protected sites that protects biodiversity under climate change. We propose that the monitoring of SSSIs would benefit from a shift in focus away from site-based condition. In so doing, however, we caution against an approach that would permit and accept climate-driven losses of species from sites and could lead to site de-designation. We argue that while it may be appealing to allow for more flexibility in feature designation to account for such turnover, there is the danger that a move in this direction would allow attribution of declines and extirpations to factors beyond a site manager's control, when this may not actually be the case. Instead, the focus of monitoring effort should be on the effectiveness of all sites in sustaining viable regional populations. Effective monitoring of SSSIs should thus be founded on two principles: (1) the extent to which they encompass an adequate (preferably large) sample of England's biodiversity, (2) on their capacity to sustain that sample into the future.

Finally, we consider how SSSIs might be managed to ensure the PA network is effective under climate change. We identify that almost all habitat management results in altered microclimatic conditions and may, therefore, be used as a tool to buffer species against macroclimatic warming. Management of England’s SSSI network for biodiversity has its challenges, however. A particular historic legacy of England’s high nature value habitats is their dependence on disturbance and therefore on sustained and expensive management. Over large areas, low-intensity (and lower cost) management may ensure that habitat of the right structure is present, even if not all of it is in optimal.

We conclude that the classic principles of conservation and PA design mostly hold true under climate change, but these approaches need to be implemented more effectively than at present. We provide practical discussions on how England’s PA network should be modified, monitored, and managed to achieve this and so create an ecologically connected PA network.

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Introduction

Climate change potentially poses significant threats to biodiversity (Pereira and others, 2010) because climate determines where species can live. Species that cannot tolerate changing conditions in their current range or successfully move out of areas that become unfavourable into suitable habitat elsewhere will face extinction. Recent changes in the abundance and distribution of species (Parmesan, 2006; Parmesan & Yohe, 2003; Wilson and others, 2007), as well as local extirpations of already declining species (Suggitt and others, 2018), have in part been attributed to climate warming.

Climate change is not only a concern, but also a challenge for nature conservation. A mainstay of such conservation has been the designation of protected areas (PAs), the strict definition of which is “a clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values” (Dudley, 2008). As a ‘clearly defined space’, PA boundaries are usually static, but species ranges are potentially geographically dynamic. Driven by climate change, species populations could decrease or disappear from some areas, increase in other areas, and/or expand into new regions (Johnston and others, 2013).

Furthermore, some PAs may lose habitat under climate change due to sea-level rise and coastal erosion (Geyer and others, 2011). Fujii & Raffealli (2008), for example, predicted that a sea-level rise of 0.3m would lead to a 6.7% loss of the intertidal area and a 6.9% loss of total macrobenthic biomass in the Humber Estuary, with consequences for species dependent on macrobenthos at higher trophic levels. Climate change could also reduce habitat quality and cause stress in otherwise unaffected species because of altered resource availability or species interactions. For example, warmer temperatures could facilitate the spread and colonisation of invasive species, thereby increasing pressures on native species (Gallardo and others, 2017).

Planning for climate change is explicit in the UK Government’s strategies for protecting biodiversity. For example, a key commitment of the 25-Year Environment Plan for England (Defra, 2018) is to “take all possible action to mitigate climate change, while adapting to reduce its impact” by “making sure that all policies, programmes and investment decisions take into account the possible extent of climate change this century”. However, there remains inevitable uncertainty as to exactly what the impacts would be on the SSSI portfolio in England and how, in turn, the designation and management of SSSIs should be adapted.

In this Think Piece, we consider what the likely climate change impacts will be on the UK’s species and whether the current designation and monitoring framework for SSSIs supports adaptation to climate change. In so doing, it must first be stated that SSSIs have been notified for both geological and biological features, and in some instances for both. However, in this report, our discussion is concerned primarily with biological features. We ask the question ‘how can biological features be maintained in an ecologically connected network of PAs under climate change?'. The means by which sites have been selected for their geological features (Ellis and others, 1996), and the appropriate means of monitoring and sustaining geological features under climate change are quite different from those associated with biological features.

First, to establish the merits of developing an 'ecologically connected' network (ECN) of protected sites, we provide a critical review of the main observed and anticipated impacts of climate change on biodiversity, namely shifts in habitat location, quantity, and quality. We scrutinise some of the existing paradigms regarding range shifts, particularly in relation to both range expansions and contractions. We then consider whether too much emphasis has been placed on enhancing connectivity, particularly at the expense of making sites bigger and better. We propose that, rather than considering the merits (and challenges) of developing an 'ecologically connected' network (ECN), the central question is really whether and how England's SSSI network could be modified to maintain viable populations of species under climate change (which in some may mean enhancing ecological connectivity). Second, we discuss whether England's SSSI network is indeed a coherent ecological network or whether it is primarily a portfolio of sites. A coherent PA network should adequately represent and sustain England's biota and ensure functional relationships between the different sites (Gaston and others, 2006). A PA portfolio, on the other hand, is simply a series or collection of PAs. To manage England's PA as a network would make national conservation more robust to climate change, and we propose monitoring solutions that could help to determine whether this is being achieved. Third, we examine how a coherent SSSI network might be managed. We critically evaluate whether a proposed move away from promoting the persistence / abundance of features that sites are designated for to embrace dynamism is the wisest course of action. We thus consider the scales at which turnover typically operates and the potential for dynamic approaches to weaken or strengthen conservation legislation. We then propose ways in which the PA network could be managed to ensure its effectiveness under climate change.

Range shifts in response to climate change

Shifts in species' spatial distributions (range shifts) are perceived as the most widely predicted and documented ecological responses to climate change (Morecroft & Speakman, 2015). It is commonly perceived that organisms face insurmountable range-shift barriers, leading to the conclusion that the only viable option to prevent extinctions is thus to create physical or functional means that allow them of their own accord to get to places where the climate is suitable (Loarie and others, 2009) or actively to translocate them there (Bellard and others, 2012; Dawson and others, 2011; Thomas and others, 2004). However, the expectation that places with suitable climate will lie outside the current range of many species is founded on the assumption that the patterns of and processes driving range expansions and contractions are similar. Comprehensive meta-analyses of recent climate-driven range shifts (e.g. Chen and others, 2011; Hickling and others, 2006) often fail to differentiate clearly between range expansions and contractions. For example, Chen and others (2011), examining 764 individual species responses to climatic changes, conclude that rapid range shifts are associated with high levels of climate warming. However, the only evidence specifically for range retraction was of localised retractions in four butterfly species (see also Franco and others, 2006). More recently, studies have sought to differentiate more clearly between range retractions and expansions and the evidence from these, for significant range retractions, is far weaker than is often thought. Lenoir and others (2020), for example, in examining more than 30,000 range shifts in response to climate change, found that range retractions among terrestrial organisms was very rarely more than a few 100 metres per year. What are the reasons for this discrepancy?

Assumptions of large latitudinal species' range shifts are generally founded on the assumption of strong and smooth geographic gradients in climate. These gradients would mean that directional temperature or precipitation changes make swathes of species' current ranges unsuitable and create swathes of suitable habitat elsewhere, further along the climatic gradient. While clear climatic gradients occur at coarse spatial resolution, at finer resolutions topography, soil conditions and vegetative shading exert great influence on local climatic conditions, resulting in considerable localised heterogeneity, particularly near the ground (Bramer and others, 2018). These fine-resolution variations can exceed the variability that occurs over continents in coarse-resolution climate, and greatly exceed the magnitude of climate change expected over the next 100 years (Maclean and others, 2017). For example, the difference in temperature on north and south facing slopes of a hill can be equivalent to that found over hundreds of kilometres in smoothed macroscopic models (Maclean and others, 2017). For the most part, organisms experience and respond to climate at resolutions where topography, soil, and shading determine conditions, and which are orders of magnitude finer than the scales at which shifts are measured or modelled (Potter, Arthur-Woods & Pincebourde, 2013). A related issue of scale is that patterns of extinction are usually localised and patchy and when driven by climatic changes, are most likely to occur first from low-quality habitat. Thus, rather than exhibiting clear-cut and systematic latitudinal range retractions, the more likely observed pattern is of increasingly sparse, and fragmented distributions across a significant portion of their geographic range (Wilson and others, 2004). By contrast, expanding species typically disperse from core populations and exhibit a clumped distribution (Hanski 1999; Wilson and others, 2004).

Taken together, these lines of evidence imply that while at the leading edge of species ranges there are indeed clear latitudinal range expansions, at the trailing edge, declines are more localised but may occur quite widely across a species' range, i.e., equatorward range margins may not always be the first place to exceed climatic limits. In any given region it may be difficult to predict with certainty whether a given species will go extinct. This fundamentally shifts how we should think about helping species cope with climate change, away from translocations and nation-wide ecologically connected networks, and more towards thinking about population viability at the regional scale, where some local physical connectivity may be important, but equally maintaining bigger and better sites may be more effective.

Moreover, for any given species we cannot rule out the possibility that in seemingly inhospitable parts of it range, it may persist, even under extreme climate change scenarios. Populations may persist for centuries and millennia. For example, spring gentian (*Gentiana verna*) is an Arctic alpine plant, with its only occurrence in Great Britain on the high Pennines, at Upper Teesdale. These populations look superficially vulnerable to climate change. At numerous PAs, post-glacial relict populations (e.g. Spring sandwort (*Minuartia verna*), butterwort (*Pinguicula vulgaris*), chives (*Allium schoenoprasum*), baltic stonewort (*Chara baltica*) can be found, including the Lizard in Cornwall, noted for its mild climate and Mediterranean component of its flora.

Maintaining viable populations of species under climate change

The 'Making Space for Nature' report on improving nature conservation and ecosystem service provision in England concluded that PAs should be made 'bigger, better and more joined' (Lawton and others, 2010). This hierarchy of recommendations reflects the classic principles of conservation planning (Diamond, 1975) and metapopulation theory - that extinction rates will be lower for single large PAs than many smaller PAs, and where there are multiple PAs, extinction rates will be lower if sites are connected rather than isolated. However, given limited resources for conservation, there will inevitably be trade-offs to consider and decisions to be made about whether to prioritise making protected sites bigger, better, or more joined (e.g. prioritising habitat quality may come at the cost of making sites bigger or more connected). In general, population viability in fragmented landscapes depends on all these factors. Bigger sites support larger populations for more species which will be less vulnerable to extinction and the risks associated with low genetic diversity in small populations (Diamond, 1975; Groeneveld, 2005). Higher quality habitat also supports larger populations (Verboom and others, 1991; Thomas and others, 2001). Connectivity of protected habitats may be important for long-term population persistence (Hanski & Ovaskainen, 2000) because populations in isolated habitat patches are more vulnerable to extinction, and less able to colonise or be recolonised from other habitat patches. However, the direct influence of connectivity in protecting species under climate change is rarely quantified and some authors (e.g. Hodgson and others, 2009) have argued that the benefits of improving PA quantity and quality can far exceed the effect of enhancing connections between protected sites. This is because the production of new individuals takes place within habitats, regardless of their location (Ovaskainen, 2002) so that, ultimately, the rate of dispersal is dependent on area and quality. Consequently, connected habitats can only partially compensate for deficiencies in habitat size and condition (Hodgson and others, 2009). If the site is too small or too poor quality to maintain viable species populations, there are likely to be too few individuals to successfully colonise a new site. Conversely, large sites support large source populations that are less vulnerable to stochastic extinction (Hanski, 1999) and are a more stable migrant source for future range expansion. Improving site quality also enhances species persistence by increasing population growth, resulting in larger propagule numbers, increased likelihood of colonisation, and higher population growth rates following colonisation and therefore enhances the capacity of biodiversity to cope with climate change (e.g., Macgregor & van Dijk., 2014). Improving connectivity to increase dispersal is only beneficial if populations at a new site can create a surplus of propagules. Otherwise, individuals could be enticed into suboptimal sites creating sink populations which are a drain on the regional population pool. Thus, connectivity does not have to be structural; local population dynamics, particularly the number and size of extant populations, can be as important as the distance between habitat patches in determining 'functional connectivity' of species (potential rates of immigration) (Hodgson and others, 2009). In support of this, Wilson, Davies & Thomas (2010) found silver-spotted skipper butterfly (*Hesperia comma)* range expansion rates in UK to be strongly related to habitat quantity and quality and Thomas and others (2012) observed that high quality sites are also the most likely to be colonised by expanding species.

There are further benefits of bigger and better PAs for species under climate change. Large sites can accommodate greater habitat heterogeneity, which is likely to lead to increased biodiversity because more species can find suitable habitat within the PA – the well-known species-area relationship (MacArthur & Wilson, 1967). Maintaining biodiversity is important as it increases ecosystem resilience against climate change. Species may perform different roles to support ecosystem processes (e.g. nutrient cycling and energy flow) and if one species is lost in a highly diverse system it is likely that another species can compensate so that the ecosystem service is not also lost (i.e., insurance hypothesis: Yachi & Loreau, 1999; Loreau and others, 2021). In this way, it is possible that a species of subsidiary importance may become critical to a process under climate change. Beech (*Fagus sylvatica*), for example, is currently very dominant in woodlands in Southern England but is sensitive to climate and may be killed by drought. Other presently minor trees may provide a replacement for beech in the future (Leuschner and others, 2020).

Larger sites may also have greater climate heterogeneity (Pyke, Andelman & Midgley, 2005; Pyke & Fischer, 2005) because of varied vegetation, hydrology, topographic and/or elevation zones. For example, summer thermal maxima may be 5 °C cooler under a woodland canopy than in an adjacent open habitat (Suggitt and others, 2011) and variations in topographic microclimate can provide cool north facing slopes near warm south facing slopes. Thus, fine-scale variation in microclimate conditions could allow species threatened by climate change to persist in localised 'microrefugia' (Suggitt and others, 2018) and buffer them against negative impacts of climate change (e.g., spring gentian (*Gentiana verna*) in Upper Teesdale and flora of the Lizard in Cornwall).

It is also worth noting that more species reach their northern range limit than those that reach their southern limit in England. The first group may be less affected by, or even benefit from warming temperatures providing their habitat requirements are met. Furthermore, much of Britain's flora and fauna, when considered in terms of its global range, tends to comprise species that are distributed over a relatively wide diversity of climatic conditions. Therefore, they may be able to adapt to climate change, especially if they are provided with a range of microclimates in their current range. For the Glanville Fritillary butterfly (*Melitaea cinxia)*, for example, the availability of suitable microclimates (as determined by the successional stage of vegetation) is almost twice as strong a predictor of butterfly abundance as is regional air temperature (Curtis & Isaac, 2015), probably because species can change habitat association in response to ambient temperatures (using cooler habitats more frequently when temperatures are warmer) (Suggitt and others, 2012).

High quality habitats can provide a similar buffer for species. For example, animals experiencing high temperatures, mostly have the capacity behaviourally to thermoregulate (Gates, 1980). Doing so reduces foraging time, but this could be compensated for if they had more food (Kearney, Shine & Porter, 2009). Plants can avoid high temperatures by opening their stomatal apertures, which increases cooling via evapotranspiration (Haworth, Elliot-Kingston & McElwain, 2011). They therefore use more water, but this would be compensated for if they had more water.

Making sites bigger and better could protect individuals and populations in SSSIs against other climate-driven stresses. These stresses might include habitat loss due to sea-level rise and coastal erosion or physical conversion of formerly inhabited areas due to erosion, landslides, and other extreme events. Species may also be threatened by efforts to mitigate or promote adaptation to climate change, such as changes in land and water management to increase carbon sequestration (Geyer and others, 2011).

Climate change is not the only threat to biodiversity and for species that have wide distributions and that are not at the edge of their range in England, other threats may be more important drivers of extinction than climate change (exceptions include, for example, mountain ringlet (*Erebia epiphron*)). Indeed, until recently, most conservationists agreed that the major causes of recent, current, and future species extinctions were habitat loss and fragmentation, the introduction of alien and invasive species, and over-exploitation (Pimm and others, 2001). The abundance of species in England has been declining since 1970 due to multiple pressures, including agriculture, pollution, urbanisation, and non-native species (Hayhow and others, 2016). British birds, for example, have shifted their distributions in multiple compass directions (Gillings, Balmer & Fuller, 2015), albeit most movements have been northwards, and this variability suggests multiple drivers of species movement including climate change. Climate change is likely to interact with and intensify the effects of other human-induced threats to biodiversity. For example, exposure to pest species makes affected species more vulnerable to drought-induced water stress (Breshears and others, 2005) and impedes the recovery of forests from extreme storm events (Pawson and others, 2013). Higher quality and larger sites can minimise threats posed by non-climate environmental drivers (Heller & Zavaleta, 2009; Bates and others, 2014), for the same reasons that they can help protect species under climate change.

Thus, while historically, improving connectivity to facilitate the dispersal of individuals moving in response to changing conditions between PAs has been viewed as the most important means of adapting for conservation to climate change (Heller & Zavaleta, 2009; Monzón, Moyer-Horner & Palamar, 2011), more recent evidence suggest that making sites bigger and better will be a far more effective strategy.

What does this mean for the current SSSI network?

Some SSSIs in England (e.g. the Broads, New Forest, STANTA in Norfolk, Salisbury Plain and the Dark Peak SSSIs) are relatively large in size. However, many SSSIs in England are small (<1km^2) and isolated and have suffered significant fragmentation and degradation in the last century (Lawton and others, 2010). The median size of SSSI sites currently is just at 16 Ha (Mosedale and others, 2021). Even within larger SSSIs, the size and quality of semi-natural habitat patches varies. Thus, the notion that SSSI performance would be significantly enhanced through connectivity (physical or otherwise) is likely to be false. The theories of island biogeography and metapopulation dynamics both suggest that when sites are small, a marginal increase in their size is an effective way of increasing species diversity and boosting population viability (by increasing the metapopulation capacity of the landscape) (Hanski & Ovaskainen., 2000). When sites are larger, but much of the habitat is in unfavourable condition, improving habitat quality is likely to be of primary importance, especially in cases where most of the habitat of interest is on a small proportion of sites. This is the case with Salisbury Plain and the New Forest for example. About 41% of all chalk grassland is on Salisbury Plain (Natural England, 2022) and the New Forest has about a quarter of all lowland heathland in England (Joint Nature Conservation Committee, 2022a, 2022b). Here improvements in habitat quality are of primary importance.

Connectivity will typically enhance the viability of populations only when an adequate number of habitat patches with significant populations of dispersers exist in the landscape (Hanski, 1999). For sites that are already of adequate size and quality, connectivity may thus be an effective way to link local populations, but benefits diminish when connecting highly distant sites or those with few occupants, as is implicit in the equations governing metapopulation models. Climate change raises the general importance of site connectivity, but it is species that are undergoing range retractions rather than range expansions that should be of primary conservation concern. For these species, their range shifts are likely to be localised and/or could be accommodated by large sites. Although on small sites the future of declining species is far less secure, it would be difficult to predict the outcome at a given site as generalised narratives about patterns of change are unlikely to apply to every species and at every site. In general, maintaining an ecologically connected network of sites may not resolve the impacts of high degrees of fragmentation that result in very small PAs (Hannon & Schmiegelow, 2002).

Moreover, in some cases, connectivity should be avoided to protect against other threats to biodiversity, such as disease and spread of invasive species. For example, de facto we have adopted the island refugium strategy of New Zealand to conserve red squirrel (*Sciurus vulgaris*) in England and Wales on Anglesey, Isle of Wight and Brownsea. The survival of red squirrel in the Formby area is because the Lancashire Plain is so sparsely wooded, grey squirrels (*Sciurus carolinensis*) have not colonised. Site size and quality, as well as other (non-climate) threats to biodiversity need to be considered before connecting habitats, as poorly designed connectivity measures could be ineffective or have negative effects on species populations.

Thus, while it may seem pertinent ask if an 'ecologically connected' network of protected sites and protected areas could be created and managed to best protect biodiversity under climate change, a more appropriate framing would be to ask how such a network, whether connected or not, could maintain biodiversity.

While the qualitative message, that large, high-quality, and (sometimes) locally connected sites will help ensure the viability of biodiversity, is widely accepted, a quantitative multi-species approach to identify priority landscapes at the spatial scale of entire countries could be difficult. For most species in most landscapes, insufficient ecological data, population parameters, and/or habitat distribution information are available to allow calculation of the capacity of the landscape to support populations (Hanski & Ovaskainen, 2000; Moilanen and others, 2005). Fortuitously, however, multi-species landscape-scale conservation planning methods that target population persistence but have data requirements that do not preclude their use in the real world, are well-advanced. Perhaps the best-known and most-suited method is that embedded in the Zonation software (Di Minin and others, 2014). Zonation is a software tool for spatial conservation planning and prioritisation. It identifies areas or landscapes that are important for retaining habitat quality and connectivity simultaneously for many, potentially very many, biodiversity features. It can be used to quantify how best to expand existing PA networks, to prioritise sites including targeting of management or restoration to ensure maximum viability of populations, and to design optimally connected PA networks. Zonation produces balanced prioritisation scores and accounts for complementarity. It iteratively removes the least valuable cells from the landscape that minimise marginal loss of biodiversity. In other words, application of this software would provide a basis for identifying the protected sites and their configuration that would be needed to maintain population viability.

Perhaps surprisingly, given that there are few regions in the world where better data are available, England's PA network has not yet been subjected to principles of systematic conservation planning using tools such as Zonation. We advocate that such an approach is needed. To do so in the context of climate change would, however, require the prediction of species' distributions under climate change at scales relevant to site-based and landscape-scale management. While predictions of the future ranges of species have been made for numerous species across England (e.g. Pearce-Higgins and others, 2017), the resolution of such predictions have generally been ~100 km^2. However, given recent advances in our ability to model species distributions at higher resolution (Lembrechts, Nijs & Lenoir, 2019; Lembrechts & Lenoir, 2020), there is no reason why high-resolution predictions could not be made.

Of course, strategic priority maps of the type produced by systematic conservation planning tools, while adequate for targeting resources at regional scale, are a blunt instrument for guiding on-the-ground action. In practice, the feasibility of landscape-scale nature recovery is dictated by a complex suite of real-world opportunities and constraints that cannot easily be captured by conservation planning tools (Pressey & Bottrill, 2008). Understanding the conditions underpinning landowners' willingness to engage in nature recovery and co-designing strategies with them will be essential for successful implementation of recovery strategies (Pressey & Bottrill, 2008). However, if actions are guided solely by bottom-up participatory approaches, strategic oversight is lost, and one cannot adequately determine whether proposed actions are the most desirable or occurring in the right place. Moreover, even at a local scale it is difficult, without adequate scientific guidance, to determine whether a proposed activity will have the desired effect. The effects of spatial configuration and future climate change on biodiversity are not easy for land managers or advisors to grasp intuitively. Successful blending of strategic and scientific advice with bottom-up participatory approaches is therefore needed.

A portfolio of sites or a coherent network?

Having established that principles of systematic conservation planning have yet to be applied the SSSI network, it is worth asking the question “is England’s current suite of SSSIs a network or portfolio? A PA portfolio is simply a series or collection of PAs, whereas a coherent PA network ensures functional relationships between the different sites to support viable regional populations of species (Gaston and others, 2006). To be self-sustaining and contribute to the ecological coherence of the network, PAs must supply individuals to themselves, to the rest of the network and outside of the network and individuals arriving from other protected or non-protected sites must be retained (Ross, Nimmo-Smith, & Howell, 2017). The network properties of PAs become more critical as intervening areas become less hospitable to species due to climate change or other pressures (Gaston and others, 2008).

Though often referred to as a network, the historical process of site designation means that England’s SSSIs should more strictly be regarded largely as a portfolio of individual sites rather than a coherent network (Gaston and others, 2006). This is understandable, given that the original purpose of England’s SSSIs was to protect a “representative sample” of species, habitats, and geological features across the country. The SSSI portfolio was never intended to be a comprehensive or holistic nature conservation mechanism and consequently, many sites are too small and isolated to support viable populations.

Monitoring the SSSI network?

Reflecting this ‘portfolio’ status, SSSIs are currently monitored on a site-by-site basis, without considering the interactive role with the wider landscape. This monitoring focuses on site condition, an indirect measure of site quality and of the management that is in place to maintain or enhance its features (Joint Nature Conservation Committee 1998, 2004; English Nature, 2003). The extent to which this can indicate the long-term viability of features within the site, or the ecological integrity of the sites is unclear (Gaston and others, 2006). In many circumstances, however, the drivers of species decline on a SSSI may have little to do with site condition. It is likely, instead, that in many circumstances it is simply a by-product of the site being too small.

In consequence, and given also perceived dynamism in the ranges of species, it is argued commonly that more dynamic approaches to site designation are needed (e.g., Crick and others, 2020). However, a drawback of any designation or de-designation approach that permits and accepts species turnover as a response to climate, is that species will often be disappearing from a site for reasons associated with deterioration in site size and/or quality, rather than because of climate change, which in many situations is currently not the most severe threat. For example, farmland bird declines have been linked to increased agricultural intensification (Gregory, Noble & Custance, 2004) and the decline of the red squirrel (Sciurus vulgaris) has been attributed to the introduction of the North American grey squirrel (Sciurus carolinensis) and disease (Tompkins and others, 2002). In most instances it would be possible to limit the climate change effect either by making sites bigger and therefore less susceptible to turnover, or through in situ management (Greenwood and others, 2016) and mitigation of damage from surrounding land uses, that removes other sources of harm and/or improves habitat quality. Moreover, in the absence of robust means of assessing performance across a SSSI network and given also renewed emphasis on natural capital in the 25-Year Environment Plan (Defra, 2018), there is the temptation to monitor PAs for ecosystem function rather than for individual species. A key consideration here is that if ecosystem functions are sustained at relatively low species richness, then arguing for the conservation of ecosystem function, no matter how important in its own right, does not provide a compelling argument for the conservation of species. Though the relationships between function and diversity are complex, positive relationships between species richness and ecosystem function are widely documented (Cardinale and others, 2012; Chisholm and others, 2013). However, few empirical studies demonstrate improved function at high levels of species richness (Schwartz and others, 2000). Likewise, the vast majority of theoretical studies predict saturation of ecosystem function at a low proportion of local species richness (Schwartz and others, 2000, Loreau and others, 2001). Thus, while high diversity is crucial for maintaining the resilience of ecosystems (Jump & Peñuelas, 2005), an ecosystem function-led approach to managing SSSIs would allow for significant biodiversity loss and fail to measure the Government's contribution to the global objective of improving biodiversity and the intrinsic culture attached to biodiversity.

The conservation impact and effectiveness of the PA network would thus be better serviced through empirical observations of biodiversity. A growing body of evidence highlights the importance of biodiversity for ecosystem stability, with additional species that currently contribute little function providing an insurance against future species losses (Downing & Leibold, 2010; Jump & Peñuelas, 2005). Any approach based on ecosystem function would make it challenging to capture these insurance benefits, as they may not manifest until significant biodiversity losses have occurred. Additionally, quantifying the function of ecosystems is not a simple process. It relies on detailed understanding of the flows of energy, materials, and nutrients through an ecosystem, which in turn are influenced in complex ways by attributes of a landscape (Best and others, 2011). Almost inevitably, therefore, an ecosystem function-based approach to managing SSSIs would require monitoring that resorts to rather simplistic habitat-condition based proxy measures.

This would only serve to exacerbate the current limitations of monitoring using condition-based assessment.

So how should the protected area network be monitored to determine its effectiveness under climate change? Broadly speaking, the ecological effectiveness of PAs can be considered in terms of different biodiversity features they support and from an ecological perspective, it seems reasonable to regard the SSSIs in aggregate as having the potential to provide a mechanism for maintaining a sample of biodiversity (preferably large) into the future (Gaston and others, 2006). To fulfil their role of maintaining a sample of biodiversity into the future, PAs must achieve two things. (1) They must capture that biodiversity within their boundaries, and (2) they must buffer it from processes that threaten its persistence (Margules & Pressey, 2000). There are thus two broad groups of measures of the ecological effectiveness of PAs that require monitoring.

First, measures that encapsulate the amount of biodiversity present within the network. This might encompass numbers of individuals or species, or the area of a particular habitat type, and how well this samples the full variety in the wider landscape. While a seemingly simple exercise, efforts to do so are hampered by the fact that, despite England having some of the best documented and monitored biota in the world, beyond particular designated features, it is not necessarily known what species occur on any given site. In general, however, the distribution of many taxa is known at a grid resolution of 10km x 10km (e.g., birds: Balmer and others, 2013; butterflies: Asher and others, 2001; flora: Preston, Pearman & Dines, 2002). While the resolution of such data is, in general, too coarse to establish with certainty whether a given species occurs within an SSSI, it is possible to acquire some insights into the likely distributions of species within the SSSI network with knowledge of the habitat requirements of species and the habitats present within a given SSSI. Such analyses would be further refined by predictive species distribution modelling. If carried out at sufficiently high resolution, one could relatively easily establish whether a given species is likely to occur within an SSSI. Regular updating of such models to account for changes in climate and the spatial configuration and quality of habitat would provide a means of monitoring such changes through time.

Second, reflecting the need to buffer biodiversity from processes that threaten its persistence, measures of condition or persistence are needed. Here one must assess the status of biodiversity features, using metrics such as population sizes and viability and species occurrences, or the condition of a habitat or vegetation type. The two groups of measures are, of course, related. Over time, inventory measures can give indirect indications of condition or persistence. Given the potential for climate-driven turnover in species, a key distinction from present assessments of site condition would be to consider the performance of the network as a whole or at least its performance over a particular region.

This is because it is quite likely that there are limits to the extent that increasing local population viability can accommodate all climate-induced changes, and even if individual PAs protect single species or communities, the network as a whole may fail to be effective at protecting biodiversity. To create a more functional network of protected sites and deliver on the commitment in the government's 25-Year Environment Plan to create a 'Nature Recovery Network' (Defra, 2018), the monitoring of SSSIs would thus benefit from a shift in focus away from site-based condition, and more towards the effectiveness of all sites in sustaining viable regional populations. This type of monitoring would consider not only where species are in the landscape, but also the spatial distribution and flows of threats to biodiversity across the landscape, i.e., are sites adequate in size and quality, as well as in the right places in the landscape to prevent extinctions irrespective of what happens in the wider countryside? It would serve the dual benefit of making assessment more robust to climate change, but also emphasise the extent to which PAs can and should act as the cornerstone for protecting biodiversity in England.

Assessing this in practise is more challenging. For some taxa, notably for birds, there is moderately good monitoring of populations across the SSSI suite (e.g. Conway and others, 2015; Maclean & Austin, 2006), but for the majority of taxa such data are lacking. Assessment of population viability across England's PAs and for multiple taxa would thus require a substantial increase in monitoring effort. Alternatively, modelling approaches of the type embedded in systematic conservation planning tools such as Zonation (Di Minin and others, 2014) would be required. A combination of both is likely to be the most effective solution. Field-based monitoring could be restricted to appropriate indicator species with model-based approaches covering a broader suite of species associated with particular habitat types. The attraction of using modelling approaches as a complement to conventional monitoring is that one is able simultaneously to identify how best to ensure population viability using classic metapopulation principles of reserve design (Hanski & Ovaskainen, 2000).

Managing the protected area network

A de facto consequence of habitat management of almost any form is that microclimatic conditions are altered (e.g., Suggitt and others, 2013; De Frenne and others, 2013) and may provide a means of buffering species against macroclimatic changes. The evidence that such an approach may be effective, while indirect in many cases, is growing. Davies and others (2006), for example, report how the silver spotted skipper butterfly (*Hesperia comma*) can adjust microhabitat usage in response to temperature and will choose warm host plants for oviposition at low ambient temperature and cooler host plants at high ambient temperatures. This would suggest that habitats can be manipulated through active management to buffer species against the adverse effects of climate change. Similarly, using paired measurements of temperatures under the canopy versus in the open at 98 sites across five continents, De Frenne and others (2019) show that forests function as a thermal insulator, cooling the understory when ambient temperatures are hot. In aquatic ecosystems the maintenance of riparian shade can reduce temperatures sufficiently to offset the effects of climate change (Greenwood and others, 2015). Broadmeadow and others (2011) demonstrated that even relatively low levels of shade (20–40%) can be effective in keeping summer temperatures below the incipient lethal limit for brown trout (*Salmo trutta*).

Similar options for manipulating the hydrology systems have been demonstrated. In grazing marshes in the east of England, for example, artificial shallow drains have been used to divert water to the middle of marshes. This process creates areas of flooding and damp habitat that can potentially provide a mosaic of nesting habitat and profitable feeding areas for breeding waders when conditions are dry (Eglington and others, 2010). Similarly, Caroll and others (2011) show that drain blocking in upland peatland can increase soil moisture and cranefly abundance. Craneflies constitute a key component of peatland biological communities; they are important herbivores and a major prey item for breeding birds.

Management of England's protected area network for biodiversity has its challenges, however. A particular historic legacy of England's high nature value habitats is their dependence on disturbance and therefore on sustained and active management (Hobbs & Huenneke, 1992; Vera, 2000). This can be expensive. The cost of meeting environmental land management priorities, for example, has been estimated at £2.2 billion and £2.3 billion per year (Rayment and others, 2019). It is worth, therefore, revisiting the assumptions that underpin the need for active management.

It is recognised widely that most of the original natural vegetation has long since been destroyed or substantially modified. Resultantly, modification of the original natural vegetation, especially in productive lowland regions, has filtered biodiversity through an 'ecological bottleneck', exemplified by the loss of Urwald fauna (Buckland & Dinnin, 1993; Fuller and others, 2017). The species assemblages associated with the resulting semi-natural habitats comprise a subset of the original biodiversity (Buckland & Dinnin, 1993; Fuller and others, 2017). As these habitats support rich (albeit filtered) biodiversity, including many species that are now scarce or threatened, it is generally believed that they should be managed as early to mid-successional habitats in ways that mimic the largely obsolete land-use practices that created and maintained them (Fuller and others, 2017). It is also thought that some of these historic management practices sustain species because they provide important natural ecological processes that have otherwise been lost since the arrival of humans, such as disturbances created by very large herbivores (Vera, 2000). It is this desire to recreate historic land management practises that often results in higher costs.

However, in adopting perceived 'traditional' management practices, modern conservation rarely achieves the range and complexity of conditions that were present in the past (Fuller and others, 2016). Rather than seeking to mimic the management itself, an alternative is to give greater emphasis to physical disturbance and variability in prescriptions both temporally and spatially and the conditions that these disturbances create for species through a variety of means. Inevitably, creating such variation within small sites will require intensive and therefore expensive management. Over large areas, however, low-intensity (and lower cost) management may ensure that habitat of the right structure is present, even if not all of it is in optimal. Likewise, if the collection of SSSIs is indeed management as a functioning ecological network, the overall requirement is that enough habitat of the right structure is present within the network, irrespective of the site in which it is located. It is also worth noting that the subset of the original biodiversity present in our current semi-natural habitats often comprise thermophilous species associated with early successional habitats (Buckland & Dinnin., 1993; Fuller and others, 2017). Under warming temperatures, the requirement to maintain these early successional habitats through active management may diminish.

However, just because it is possible to manage a PA in theory, does not mean that such management will occur in practise. Though approximately one third of all SSSIs are managed by non-governmental conservation organisations, with a remit to enhance biodiversity (Gaston and others, 2006), a significant proportion of SSSIs are on farmland. To a large extent, therefore, the potential for effective management is determined by whether and how farmers are prepared to carry-out management (Staddon and others, 2021). While incentives and legislation are important, farmers' own cultural dispositions (Burton, Kuczera & Schwarz, 2008) and their management knowledge and skills (Rois-Díaz and others, 2018) play a key role. For example, management decisions for nature recovery on farmland often depend on farmers' belief in their own capacity to achieve it successfully (Ambrose-Oji and others, 2018). It can also depend on how farmers self-identify and the extent to which environmental concerns make up their image of a 'good farmer' (Van Dijk and others, 2016). The norms that form part of a farmer's social environment are crucial in shaping this self-identity (Moseley and others, 2014). Given the complex and heterogeneous character of farmers' attitudes towards nature recovery, successful management will depend largely on alignment with farmers' dispositions. Management of SSSIs on farmland must therefore be attuned to the characteristics of farmers and consider 'structural' factors such as farm tenure. Ultimately, we need to know more about the willingness of and incentives required for different types of farmers to carry-out different forms of nature recovery and the reasons underpinning this.

Conclusions

For SSSIs to protect biodiversity in a changing climate will not require an overhaul of the classic principles of PA design (e.g. Diamond, 1975). These principles just need to be better implemented into the current system. There is much to be gained from considering how species may be able to cope with climate change in their existing ranges, as the problems that limit the effectiveness of PAs under current conditions also limit their ability to protect species under climate change. Sites should be made bigger and better so that species may be able to buffer themselves against climate change in their current ranges. On large sites, 'better' (involving the reduction of threats other than climate change) should be the priority. Concerns about ecological connectivity should not override making sites bigger and better. The primary objective should be to ensure that biodiversity persist in the face of climate change in the most effective way.

A shift towards performance metrics based on de-designation, would almost certainly weaken conservation legislation, and while appropriate in some circumstances, should always be applied with caution and with the consideration of the performance of the network as a whole in mind. Likewise, approaches based on ecosystem function would almost certainly weaken the protection of species and populations that are critical to ensuring the resilience of these systems to future environmental changes. General habitat monitoring will not pick up the critical fine variation in habitats which species do through their niche requirements. The effective monitoring of SSSIs should be founded on two principles: (1) the extent to which they encompass an adequate (preferably large) sample of England's biodiversity, (2) on their capacity to sustain that sample into the future.

Almost all habitat management of any form results in altered microclimatic conditions and may, therefore, be used as a tool to buffer species against macroclimatic warming. Management of England's SSSI network for biodiversity has its challenges, however. A particular historic legacy of England's high nature value habitats is their dependence on disturbance and therefore on sustained and expensive management. Over large areas, however, low-intensity (and lower cost) management may ensure that habitat of the right structure is present, even if not all of it is in optimal.

PAs have high conservation value and should continue to be the cornerstone of biodiversity protection and enhancement in England. Commitments in the Post-2020 Biodiversity Framework and the 25-Year Environment Plan provide an unprecedented opportunity to develop future-oriented and realistic long-term conservation goals to reverse species declines.

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