A think piece on the effectiveness of protected areas in England

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Mosedale JR, Maclean IMD, Gardner AS, Gaston KJ, Hopkins JJ

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Natural England Project manager

Kimberly Owen

Contractor

University of Exeter

Author

Mosedale JR, Maclean IMD, Gardner AS, Gaston KJ, Hopkins JJ

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Executive summary

The design and establishment of protected areas has been a cornerstone of nature conservation policy. Protected areas have expanded both globally and within England over recent decades. However, biodiversity continues to decline across multiple taxa and the UK is expected not to meet the Aichi Biodiversity targets. England's wildlife is now among the most depleted in Europe. Furthermore, there is growing evidence that many species are already predestined for extinction due to levels of habitat fragmentation already reached and growing evidence that pervasive landscape-wide pressures, such as agricultural fertilisers and pesticides, are important drivers of biodiversity loss. In this think piece we ask if we can re-envisage and implement a network of protected areas able to reverse this decline.

Classic conservation planning theory was encapsulated by the 2010 '*Making Space for Nature*' report summary as '*bigger, better, more and joined'*. Implicit to these principles are important trade-offs that have not always been acknowledged. Advocating a particular principle, such as joining-up or improving the connectivity of protected areas, comes at the cost of a reduced emphasis on size and quality. There is a growing body of evidence to suggest that increasing the size and quality of protected areas will reduce the risk of extinction, improve dispersal and facilitate the adaptation of species to future climate, more effectively than improvements to structural connectivity through wildlife corridors.

The application of these design principles to England must be informed by the characteristics of the wider landscape in order to deliver an effective protected area network. The high degree of habitat fragmentation and the pervasiveness of intensely managed agricultural land throughout the English landscape presents particular challenges. A well-designed network of protected areas can slow biodiversity loss by protecting high concentrations of bio and geodiversity from localised impacts and facilitate dispersal and (re)colonization of habitats in the face of climate change and other forms of environmental change. Reversing the decline in biodiversity, however, will require a landscape wide approach that reflects the geography of land use, topographical and hydrological features that together define the risk landscape.

A strategy of delivering a more functional network of protected sites in England would be to improve the quality of National Parks and increase the size of existing SSSIs. Existing National Parks and Areas of Outstanding Natural Beauty are of sufficient size and compactness to be capable of maintaining large populations and diversity of species while managing landscape-wide pressures, but only if the enhancement of biodiversity becomes their primary objective. Currently, the do not differ from the wider landscape in their ability protect biodiversity. Existing SSSI sites have a median size of only 16Ha and therefore require not only enlargement but also mechanisms to mitigate drivers of biodiversity decline from outside their boundaries. Put simply, there is a need to make '*SSSIs bigger and National Parks better'*.

To deliver such a vision requires looking beyond approaches grounded in conservation biology to embrace a broader understanding of environmental management that views people as an essential element of nature conservation, both as participants in its protection and recipients of its benefits. There is also need to ensure continued public support for conservation goals is reflected in the delivery of effective policy. We detail how these wider societal views might be more objectively incorporated into the process of conservation planning. We also highlight the complexity of coupled socio-ecological systems and the need identify those parts of the combined system that are most critical to successful outcomes. We propose the concepts *of agents* and *agency* as means to do. A focus on ‘biological agents’ that have a disproportionate influence on ecosystem function, can help inform conservation practice. Identification of a ‘critical path’ of actions required by key biotic, abiotic and social agents provides a way of linking legislation, regulation and goals, to deliver an effective network of protected areas.

Contents

Introduction

The design and establishment of protected areas (or ‘reserves’) has been a cornerstone of nature conservation policy (Margules & Pressey 2000). By separating areas of high biodiversity from the processes that threaten their existence, a network of protected areas helps sustain viable populations of species and preserve habitats and geodiversity. Over recent decades there has been a significant expansion in the number of protected areas established globally, with the aim of helping to preserve and enhance biodiversity in line with Aichi Biodiversity Targets, which call for the conservation of at least 17% of terrestrial and inland water areas, especially areas of particular importance for biodiversity and ecosystem services (CBD 2010). In England, the total extent of land and sea protected through national and international designation increased from 1.2 million to 3.4 million hectares between 1999 and 2020. Most of this increase is, however, attributable to the designation of inshore and offshore marine sites (Defra 2020a). Nevertheless, 26.4% of the land area is currently designated in one form or another, albeit that the primary purpose of these designations is not always biodiversity protection.

Despite the expansion of protected areas, global biodiversity has continued to decline and the rate of extinctions and habitat loss has accelerated (WWF 2020, CBD 2020). The causes of this decline include habitat loss, climate change, pollution (particularly from synthetic pesticides and fertilisers), invasive species and pathogens, the effects of which are often synergistic and can occur far from their source or origin (Millennium Ecosystem Assessment 2005, Urban 2015). There is growing evidence that biodiversity decline is occurring across all taxa, with recent studies highlighting widespread loss in insect biodiversity, driven by multiple causes, across different regions and ecosystems (Sanchez-Bayo & Wyckhuys 2019, Goulson 2019, Hallmann et al 2017), including pristine rainforest environments (Lister & Garcia 2018). Protected areas have not proved immune to this loss of biodiversity, with a decline in the quality of habitat and the biodiversity of sites widely documented (Chape et al 2005, Laurence et al 2012). From monitoring across 63 protected areas of varying size and habitats across Germany, for example, Hallmann et al (2017) reported a 75% decline of flying insect biomass over 27 years. The magnitude and scope of biodiversity decline revealed in this study suggests that not only is the long-term persistence of many species endangered, but so too is the functioning of ecosystems and the services they provide (IPBES 2019). The continued decline in biodiversity and the role attributed to multiple, ‘permissive’ drivers of loss, raise questions about the adequacy of protected areas (Salah et al. 2000).

Given the importance placed on protected areas, determining their effectiveness in representing and maintaining biodiversity is of central importance in spatial conservation planning. Thus, spatial conservation planning principles and tools have been much used to apply ecological theory, such as island biogeography (MacArthur & Wilson 1967) and metapopulation dynamics (Levins 1969, Hanski & Gilpin 1991, Hanski 1999), to inform the location and design of protected areas around the world. Nevertheless, there are particular reasons for concern about the effectiveness of protected areas in England. Notwithstanding an established network of designated sites that far exceeds global protected area targets, long-established legal protections and widespread monitoring of biodiversity, there is substantial evidence that suggests systemic failure. Across the United Kingdom (UK), the abundance of over 40% of monitored species has declined over the past five decades, with trends in biodiversity similar in England to those for the UK as a whole (Hayhow et al 2019). As a result also of longer-term changes, England’s wildlife is now among the most depleted in Europe (Newbold et al. 2015). While large in overall extent, England’s protected areas compare unfavourably, in terms of quality and individual size, with those of many nations of similar population density and economic output (Gaston et al 2006). Remaining semi-natural habitat is highly fragmented. Given ongoing declines in England’s biodiversity, despite high protected area coverage, a key question that follows is whether these declines are due to a failure to apply conservation planning principles, a failure of their application, a more fundamental failure of policy priorities or awareness, or whether simply what has been proposed is not enough?

Though many SSSIs were designated prior to the widespread adoption of systematic conservation planning principles, more recent policy context and goals for biodiversity in England do, to an extent, draw upon these principles, particularly in terms of being shaped by the ‘Making Space for Nature’ report (Lawton et al 2010). This report proposed the need to improve the effectiveness of existing protected areas by means of a hierarchy of recommendations: (i) improving the quality of habitat, (ii) increasing the size and (iii) number of sites, and (iv) enhancing the connectivity among sites. Or, as the executive summary put it: ‘bigger, better, more and joined’. The recommendations were incorporated in biodiversity policy and most recently the 25-year Environment Plan (Defra 2018). Among the mechanisms for delivery of the plan is the development of ‘Nature Recovery Networks’ (NRNs) to deliver more habitat, in better condition, and in bigger patches that are more closely connected. The plan also reflects the work of the Natural Capital Committee, an independent advisory committee that ran from 2012 to 2020 focusing on the provision of ‘ecosystem services’. The additional services that Nature Recovery Networks are to deliver may include greater public enjoyment; pollination; carbon capture; water quality improvements and flood management. Similarly, the Environmental Land Management scheme (ELMS) is the intended mechanism by which farmers and other land managers may be paid for delivering public good including the restoration of habitats for endangered species and the recovery of soil fertility as part of a new, post-Brexit, system of support for agriculture. The current policy context and development of such mechanisms as NRNs and ELMs present a ‘once in a generation’ opportunity to change the way we do conservation in England. To realise the opportunity, we need to readdress and re-envisage the role of protected areas in enhancing biodiversity in England.

In this ‘think piece’ we return to the principles outlined by Lawton et al. (2010) and examine the implicit trade-offs between ‘bigger, better, more and joined’ when applied to England’s protected areas, their dynamic relationship to the wider landscape and in the context of climate change. We examine how the current state of biodiversity, distribution of remaining semi-natural habitat and characteristics of the wider landscape influence the application of the principles of conservation science and protected area design. The existing portfolio of protected areas is considered an essential element of the context that determines the realisation of design principles. In the second part of the ‘think piece’ we consider how to deliver a re-envisaged, and more fit-for-purpose, network of protected areas. We argue that there are too many barriers in place for the network to be expanded or improved without considering how to overcome these barriers, most of which stem from competing demands for land-use. We therefore examine the potential to integrate human activity into conservation planning approaches, recognising that there is a need to go beyond approaches that treat people as external to nature, and the need to adopt a wider perspective. An intuitive ability to identify these inter-relations through, for example, stakeholder consultation, has long been the mark of successful conservation. However, we argue that the approaches to enhancing protected areas would be improved through a more structured way of integrating information on the distribution of biodiversity and other natural assets with information on land tenure and management and the interactions and processes that shape land-use decisions. This will enable better understanding of the drivers that motivate engagement in biodiversity conservation and the underlying factors influencing perceptions, opinions and behaviours. In so doing there is a need to identify the key biological and social agents and mechanisms underlying biodiversity decline or best able to deliver nature recovery. By doing so, one is able to pinpoint the most significant ecological and social barriers to nature recovery and help organisations with the primary remit of protecting biodiversity better to focus policy and activities. Additionally, better recognition and understanding of the interactions between people and nature may be used positively to identify tipping points for nature recovery, namely those actions mostly likely to have a major positive effect.

1. Designing protected areas

Design Principles

The hierarchy of recommendations elaborated by Lawton et al. (2010) summarises many years of theory and practice about the design of protected areas and preservation of biodiversity: improving habitat quality, creating bigger and more sites, and enhancing connectivity among these sites. The geometric principles of design expressed in the recommendations reflect those summarised by Diamond (1975) that the extinction rate of species will be lower for single large, more compact protected areas than multiple smaller protected areas. In the case of multiple protected areas, extinction rates will be lower for connected or more aggregated sites than widely separated unconnected sites.

The benefit assigned to larger more compact protected areas reflects theories that larger habitats permit larger populations, which are less vulnerable to extinction from environmental or demographic stochasticity (Diamond 1975, Huxel and Hastins 1999, Franken & Hik 2004, Griffen & Drake 2008), and reduce the risks associated with lower genetic variability in small populations (Groeneveld 2005, Jarvinen 1982). More compact protected areas, as typically expressed by a low perimeter to area ratio or by measures of circularity (McGarigal 1995), reduce the exposure of species to edge effects (Ries and Sisk 2004) and the threats associated with the land-use of neighbouring areas. The more compact the protected area, the more it is likely to support larger populations and those species most vulnerable to external threats. Higher-quality, more suitable habitat also allows larger populations of a species to be sustained (Verboom et al 1991, Thomas et al 2001, Resetarits & Binckley 2013), but has the added benefit of enhancing metapopulation persistence across fragmented landscapes by enabling greater population growth and dispersal (Griffen & Drake 2008, Ye et al 2013, Hodgson et al 2009).

In general terms, the benefits of having more sites are intuitively obvious: the more land that is protected, the more biodiversity is contained within these protected areas. More specifically, a species will generally persist in a landscape if the metapopulation capacity of that landscape is greater than a threshold value determined by the properties of the species (Hanski & Ovaskainen 2000). While the size, quality and connectedness of habitats are key determinants of metapopulation capacity, in lieu of having large areas of intact habitat, having more sites protected will ensure that populations of species are more viable, especially if the sites are clustered together. Isolated habitat patches are more vulnerable to the extinction of local populations, and less readily recolonized from other habitat patches. Connectivity between protected areas is achieved by the establishment of habitat corridors linking protected sites or other methods of reducing the barriers to species movement and dispersal between sites. A landscape more permeable (Lees & Peres 2009) to species dispersal allows the easier colonisation of resource patches, greater gene flow and more viable metapopulations (MacArthur & Wilson 2001, Brown & Kodric-Brown 1977, Richards 2000, Eriksson et al 2014). Improving connectivity between protected areas has historically been viewed as essential for conservation in the face of long-term climate change. The projected shifts in species distributions, across altitude and latitude, as a response to a changing climate suggests that current biodiversity patterns, on which the designation of protected areas is based, may not reflect future patterns (Parmesan 2006; Chen et al. 2011; Hoffmann and Sgrò 2011; Pecl et al. 2017). Improving the connectivity between protected areas is therefore postulated as facilitating the expected ‘range shifts’ in species distributions and thereby reducing the risk of future extinctions (Heller and Zavaleta 2009; Thomas et al. 2012; Lawson et al. 2014).

Trade-offs between design principles

Any application of the principles of the Lawton et al. (2010) review will imply trade-offs in terms of the emphasis placed on different aspects of individual principles, and in terms of the relative importance attributed to ‘bigger, better, more and joined’.

For example, two broad approaches have been suggested as means of enhancing quality: providing more optimal habitat or increasing heterogeneity. Studies have demonstrated the positive influence of creating more optimal habitat (Thomas et al. 2001; Griffen and Drake 2008; Ye et al. 2013; Ye et al. 2013), but have also shown that greater habitat heterogeneity buffers the effect of environmental fluctuation compared to homogenous habitats, encouraging population stability (Opdam and Wascher 2004; Donaldson et al 2017).

In terms of the trade-offs *between* the principles of the Lawton Review, it is simply the case that, given finite resources, advocating a particular principle may come at the cost of reduced emphasis on other principles. For example, by prioritising habitat quality one may not also be able to enhance the quantity and connectivity of sites.

Over recent decades, enhancing connectivity has emerged as the most commonly advocated approach for helping species adapt to climate change (Heller & Zavaletta 2009). On the basis of presumed benefits to metapopulation persistence and adaptive capacity to climate change, conservation measures that improve connectivity have been deemed beneficial irrespective of any measurable improvement to species resilience or persistence (for example Threadgill et al. 2020 on field verges). The emphasis in the Defra 25-year Environment Plan on creating a Nature Recovery Network risks perpetuating this emphasis of connectivity, at least in the minds of those involved in Local Nature Recovery Strategies. All too often the use of the word ‘network’, in the context of protected areas, evokes ideas of a series of habitat patches connected by wildlife corridors. Indeed, the purpose of Local Nature Recovery Strategies has sometimes been framed as to ‘restore and link up habitats’ (Cumbria County Council 2021).

However, such ideas run counter to the growing body of evidence that increasing the size and quality of protected areas will increase species dispersal, the persistence of metapopulations and adaptation to future climate, more effectively than improvements to structural connectivity through wildlife corridors or similar measures. Protected areas have not only continued to accommodate populations undergoing range retractions (Thomas & Gillingham 2015) but are also preferentially colonised by species, pointing to the importance of high quality habitat as a means of facilitating expansions (Thomas et al., 2012). Additionally, most assessments of species range shifts are made using coarse-resolution climate and biodiversity data. When finer-scale data are used, the patterns of extinction revealed are often quite localised and growing evidence suggests that species are able to survive in microrefugia (Suggitt et al. 2019). The assumption, therefore, that corridors of habitat are needed to facilitate significant poleward movements may be unwarranted. Instead, measures that ensure the survival of species within their existing geographic range may prove more effective. Larger sites also typically support larger populations, which in addition to being less extinction prone, have a greater capacity to (re)colonise surrounding habitat (Wilson et al. 2002, Lawson et al. 2012). As a result, larger populations augment the ability of species to shift their range in the face of climate change (Hodgson et al. 2011). In effect, therefore, local population dynamics can be as important as the distance between habitat patches in determining 'functional connectivity' of species (potential rates of immigration). The larger, more stable populations supported by habitat areas of greater size and quality, increase the rates of immigration to other sites (Hodgson et al. 2009).

England's wider landscape

More so than many other European countries (Büttner 2014), England is characterised by numerous small patches of semi-natural habitat located within a wider, primarily agricultural, landscape. The perceived need to enhance agricultural productivity and the associated commercial benefits is a major constraint on the land available for conservation options (Robinson & Sutherland 2002). Intensive agricultural land management renders the landscape less permeable to species dispersal, whereas less intensive, but more prevalent, agriculture can reduce land available solely for conservation and the establishment of protected areas.

What relatively large areas of semi-natural habitat remain are primarily located in upland areas of low agricultural productivity but nevertheless still subject to intense grazing pressures (or other management regimes such as for grouse shooting). The upland areas of England, in common with most of Europe, were once a predominantly forested landscape (Kirkby and Watkins 1998), but have long been largely denuded of trees. The historic loss of keystone species (Turvey 2009), particularly large predators, means that habitats are rarely self-sustaining ecosystems as grazing pressures, whether from domesticated herds or wild herbivore populations, are largely unregulated. As a result, maintaining the successional stage of habitats most suited to supporting high levels of biodiversity often requires significant management intervention.

The small size of many semi-natural habitat patches, especially in the lowlands, reflects both historic and more recent changes. Increases in agricultural productivity over recent decades resulted from an intensification in the management of the over 75% of the UK land area devoted to agriculture. Over 95% of flower meadows were lost between the 1930s and 1984 and about 90% of lowland ponds over a similar period, for example (Hayhow et al 2019). The pervasiveness of intensively managed agricultural land across the landscape, with for example few catchments not dominated by agricultural uses, also heightens the threats posed by diffuse pollution derived from intensive production even to semi-natural habitats relatively remote from neighbouring land use pressures. Other changes in land use have contributed to the erosion of sizeable areas of semi-natural habitat, including more widespread and intensively used transport networks and urban expansion, and the development of coastal defences and flood mitigation schemes, in part prompted by climate change, resulting in further fragmentation and loss of wetland habitat.

As a result, about 13% of assessed species in England are estimated to be threatened with extinction, and the majority have displayed strong declines in abundance, particularly over the short term. The declines in abundance have been particularly dramatic for priority species and habitat specialists and those associated with agricultural landscapes, with butterflies down by 68%, and farmland birds down by more than half on average since 1970 across the UK (Hayhow et al. 2019). The trends for biodiversity in England broadly mirror those for the UK as a whole (Hayhow et al. 2019). The steep declines in abundance but relatively low decline in species ranges suggests that species may be suffering from a 'hollowing out' across their range from marginal habitats as neither the quality nor the extent of remaining semi-natural habitat is sufficient to support viable populations of species.

A key consequence of the small size and fragmented nature of semi-natural habitat patches is the likelihood of a significant extinction debt - a future ecological cost of current habitat destruction (Tilman et al. 1994). Extinction debt occurs because of time delays between impacts on a species, such as destruction of habitat, and the species' ultimate disappearance. For long-term metapopulation persistence, a network of habitat fragments must be adequate in terms of the number, size, and spatial configuration of the fragments such that the rate of local extinction is exceeded by rates of colonisations (Hanski & Ovaskainen 2002). Following habitat loss, the species may survive for period of time in remaining patches, but the rate of localised extinction may exceed the rate of colonisations. This does not result in immediate extirpation of the metapopulation, but instead in a delayed response that may take many years or decades to occur (Kuussaari et al 2009). Evidence for the existence of an extinction debt in many taxa and for many regions is now well-documented. In England and Wales, for example, two-thirds of extant metapopulations of the Marsh Fritillary are predicted to go extinct without any further habitat loss (Bullman et al 2007). Thus, many of the species losses resulting from recent land-use changes may be yet to manifest. It is likely that the inadequacy of the present protected area system is yet to be fully revealed.

The fragmented landscape of England and pervasiveness of intensively managed agricultural land also implies that remaining semi-natural habitat is heavily impacted by pressures from both neighbouring areas and the wider landscape. Furthermore, simply improving structural connectivity between habitat patches, without any increase in patch size, is insufficient to mitigate these pressures on biodiversity or enhance species persistence except in localized or specific cases. Much larger protected areas will allow adequate areas of highly biodiverse, successional habitats to be maintained, while reducing management interventions and creating more resilient ecosystems. However, the same competing land use demands that are major drivers of habitat fragmentation and biodiversity decline across England also present major obstacles to the realisation of designating large protected areas. One of the major reasons why improvements in connectivity have been so widely cited as conservation gains is that they are often more readily achievable than any significant gain in either habitat area or quality.

England's protected areas

Existing protected areas in England are typically divided into two different types that present contrasting characteristics when considered from a perspective of the principles of conservation planning.

The first set of protected areas include areas designated primarily on the basis of their biodiversity of geological features. These include nationally designated National Nature Reserves (NNR), Sites of Special Scientific Interest and Areas of Special Scientific Interest (SSSI) and internationally designated Special Protection Areas (SPA) and Special Areas of Conservation (SAC), under the EU's Birds and Habitats Directives, and sites designated under the Convention on Wetlands of International Importance (Ramsar). Although covering 6.5% of England they present a fragmented portfolio of sites, the majority of which are small size and vulnerable to pressures from neighbouring land uses and more remote areas that are sources or transits of permissive threats to biodiversity.

The median size of SSSI sites, at 16 Ha, is less than twenty percent of the size of the average farm holding in England of 86 Ha (Defra 2020b) and much smaller than the home ranges (territory) of many mammalian species and birds of prey. Rural foxes (*Vulpes vulpes*) have a mean territory size of 270 Ha (Reynolds & Tapper 1995), for example, and the home range size of breeding Barn Owls and Hen Harriers is 350 and 450-800 Ha respectively (Bunn et al. 2010; Arroyo et al. 2014). While some SSSIs lie adjacent to others to form contiguous blocks of protected habitat, many individual sites are not contiguous, instead comprising smaller patches of habitat fragmented by roads and other features. The median size of these fragments is even smaller: just 4.5 Ha (Figure 1). While it is true that mammals or birds of prey are rarely the features for which these sites have been notified, it serves to illustrate that remarkably few SSSIs are capable of supporting even single pairs of larger breeding species, let alone viable populations. Moreover, only half of the 4,132 SSSIs were most recently assessed as in favourable condition (Defra 2020a) – most likely due to a combination of limited resources for both management and enforcement of legislation, and their vulnerability to effects from the wider landscape.

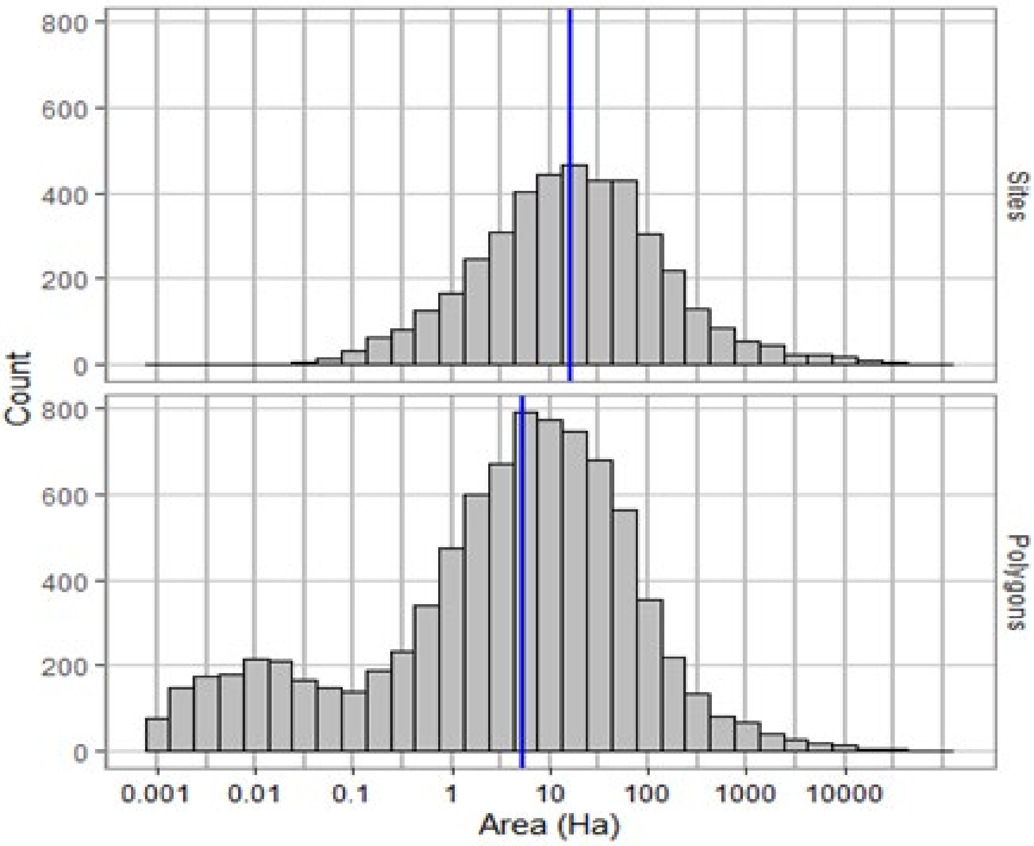

Figure 1: Size distribution of SSSI sites (top) and of contiguous SSSI ‘polygons’ (bottom), excluding areas <0.001Ha. Size is expressed on a log10 scale of area in hectares. Median size of sites and polygons shown in blue.

The second set of protected areas are landscape-scale designations. Areas of Outstanding Natural Beauty (AONB) and National Parks were established to conserve and enhance natural beauty, wildlife and cultural heritage and promote public enjoyment and understanding. Although there is a statutory requirement for governing bodies to have regard “to the purpose of conserving biodiversity” (Natural Environment and Rural Communities Act 2006 (c. 16 Part 3 section 40)) in exercising its functions, these designated areas are often as focussed on the protection of landscape character and heritage as on bio or geodiversity. Landscape scale designations cover much larger and more aggregated areas of land, covering approximately 24% of England. However, more than one-third of the land area of national parks comprises built up areas or farmland, and only one-third comprises habitat of principal importance for biodiversity (Table 1).

Even where the proportion of arable and improved grassland within national parks appears relatively low, agricultural activities often predominate. For example, grazed acid grassland is excluded from estimates of agricultural land in Table 1 yet prevalent across certain National Parks such as the Yorkshire Dales, while approximately 90% of the Peak District is intensively grazed. In certain cases, the ability of these designated areas to reverse or limit biodiversity declines may be compromised by the very landscape they are seeking to preserve. The pastoral landscapes for preservation by the parks are often the result of relatively recent land management practices and of grazing regimes that are detrimental to any increase in biodiversity. Moreover, their ability to protect biodiversity from extinction is generally no better than the wider countryside, at least for wild bird species (see Box 1: extinctions of bird species in national parks, AONBs and the wider countryside).

Areas of Outstanding Natural Beauty are even more dominated by urban and agricultural land, comprising 67% of their total area (excluding Isles of Scilly) and feature less priority habitat (23% of total AONB area). Like National Parks they afford little additional protection to biodiversity, though for wild bird species, they do at least perform marginally better than the wider countryside (Box 1).

Table 1. Land area of England's 10 National Parks and the extent and percentage of urban and agricultural land and NERC41 priority habitat contained within each national park. Estimates of Urban and Agricultural land derived from CEH Landcover 2019 data while priority habitat estimated from Natural England data. Major conflicts in classification (for example large areas of the Broads national park are classified as coastal grazing pasture in Natural England data and improved grassland in CEH landcover) have been resolved by prioritising Natural England data.

National Parks	Area (km^2)	Urban and agricultural land	%	Priority habitat	%
South Downs	1,653	1,190	72	371	22
Exmoor	693	356	51	209	30
Yorkshire Dales	2,185	525	24	835	38
Peak District	1,438	491	34	545	38
Dartmoor	956	330	35	361	38
North York Moors	1,441	598	42	523	36
New Forest	567	191	34	304	54
Northumberland	1,051	171	16	302	29
Lake District	2,362	676	29	568	24
The Broads	302	102	33	170	56
All Parks	12,648	4,740	37	4,191	33

Box 1. Local extinctions of bird species in national parks, AONBs and the wider countryside.

Here, to quantify the effectiveness of National Parks and Areas of Outstanding National Beauty (AONBs), we sourced data on the distribution of breeding wild bird species in England from the two most recent BT0 Breeding Bird Atlases: 1988-91 (Gibbons *et al* 1994) and 2007-11 (Balmer et al 2013). In each 10 km x 10 km grid cell in which breeding was confirmed to have occurred in 1988-91, we calculated the proportion of the land area of the grid cell that was national park and AONB. For every species (*N* = 177), we then calculated the proportion of extinctions from each grid cell by the later period (1988-91) relative to the total number in which breeding occurred in the earlier period, to give a value for each species indicating the probability of extinction. We performed analyses separately for grid cells that were majority national park, majority AONB or majority neither AONB nor National Park and therefore effectively part of the wider countryside (Figure 2). Vertical blue lines in the figure below represent the median values across species.

(a) Outside National Park / AONB **(b) AONBs** **(c) National Parks**

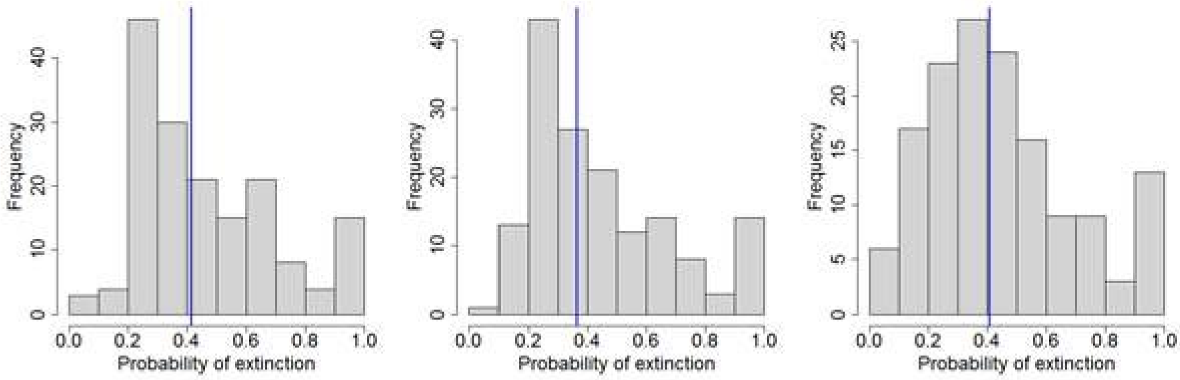

Figure 2: The calculated the proportion of extinctions outside National parks and ANOBS, ANOBs, and National parks.

It can be seen that extinctions from AONBs and areas outside National Parks and AONBs form a broadly similar pattern though the median within AONBs, at 37.3%, is marginally lower than for median for the wider countryside at 41.1%. In contrast, while there is evidence that a small number of species were more likely to experience very few extinctions from grid cells inside national parks, on average national parks perform less well than the wider countryside in preventing extinctions, with a median value of 40.1%. While some of the area classed as 'wider countryside' includes land designated as SSSIs National Nature Reserves, the same is true of land inside National Parks and AONBs, which on average have a higher proportion of their land cover protected by these designations. In summary AONBS afford only marginal additional protection to species than wider countryside and National Parks afford no more protection than the wider countryside.

An effective network of protected areas?

Historically, England's protected areas have been regarded largely as a portfolio of individual sites rather than a coherent network (Gaston et al 2006). Little consideration has been given to the broader aggregate properties of this portfolio, both in terms of its ability to sustain viable populations of species, and the extent to which it represents the full range and examples of a particular feature across a region or nationally. This is understandable, given that the goals of site designation have historically had little to do with the operation of the whole. Nevertheless, following the setting of Aichi Biodiversity Targets, which call for protected area to be ecologically representative and well-connected, the portfolio of sites has increasingly been referred to as a network (e.g. in Defra 2018, 2020a). Several studies have sought to determine the effectiveness of this network, particularly with regards to the extent of representation for selected components of England's portfolio (e.g. Oldfield et al 2004, Jackson et al 2004). Almost invariably, such studies conclude that while the portfolio of protected areas represents well the biodiversity features of concern, it could usefully (and sometimes markedly) be improved in this regard (Gaston et al 2006).

Allied to this concept, the monitoring of SSSIs is generally carried-out on a site-by-site basis without necessarily considering directly their context in the wider landscape, or the performance of the portfolio of sites as a whole. A common standard for monitoring and evaluation of SSSIs has been adopted, in which the site condition of SSSIs is assessed every six years. Condition is an indirect measure of site quality and of the management that is in place to maintain or enhance particular features (Joint Nature Conservation Committee 1998a, 2004; English Nature, 2003). The extent to which it indicates the long term viability of features within the site, or the ecological integrity of the sites is unclear (Gaston et al 2006). Given the small size of many SSSIs (Figure 1), there is every possibility that a SSSI in favourable condition is too small to sustain viable populations of its notified species in isolation, and their survival is thus contingent in part on the condition of the wider landscape, or indeed the effectives of protected area networks as a whole. The monitoring of SSSIs would therefore benefit from a shift in focus away from site-based condition, and more towards (i) measures more reflective of their capacity to sustain the features for which they are designated and (ii) the overall effectiveness of the portfolio of sites as a whole.

The design and management of individual protected sites and the realisation of a coherent network needs to be informed not solely by the distribution of biological or geodiversity, but also by the spatial distribution and flows of threats to biodiversity across the landscape. Up-stream catchments, for example, can threaten the quality and diversity of habitats of downstream protected areas through sediment deposition or pollution transfer (Cook et al 2018).

There are few instances where SSSI boundaries reflect catchment boundaries or other key aspects of the 'risk landscape' (Boon 1991) and limited means for modifying land use management outside of SSSI boundaries to mitigate impacts. The process of SSSI designation has usually focused on the spatial distribution of features of interest, rather than the processes that threaten or sustain these in the wider landscape. While the Impact

Risk Zones for Sites of Special Scientific Interest (Natural England 2020) provide a means to regulate planning consents in regions adjacent to a SSSI, there is limited legislation in place to prevent other activities from occurring. Regulations governing consents for land management, for example, focus solely on operations occurring within SSSI boundaries. For SSSIs to be effective, greater heed must be given to land use and management in regions surrounding SSSI.

In contrast, many Natural Parks comprise relatively coherent hydrological units, often being located in upland catchment areas. The geography of these national parks could facilitate the mitigation of many permissive threats to biodiversity and therefore offer the potential of delivering robust, self-regulating habitat, ecosystems and services. To do so, however, requires mechanisms by which the underlying drivers and sources of permissive risks can be effectively addressed. In turn this requires the preservation and enhancement of geo and biodiversity to be the overriding priority for these protected areas.

To summarise, the existing protected area portfolio in England can be characterised as a combination of (i) a number of small, high quality, generally homogeneous habitats within sites, that are unsustainable without active management and vulnerable to pressures from the wider landscape, and (ii) much larger areas of heterogenous, but lower quality habitats but of sufficient size and compactness that they could be resilient to external pressures and capable of maintaining large populations and diversity of species if only they were of sufficient quality. For the most part, these former should be regarded largely as a collection of individual sites rather than coherent and functioning network and is unlikely to sustain viable populations of many species. The latter, while being of sufficient size to sustain viability of species is rarely of sufficient quality. Taken together, and given the pervasiveness of many permissive threats to biodiversity, it is clear that the existing portfolio of protected areas will not reverse biodiversity declines unless markedly improved.

Despite their limitations, these contrasting elements of protected areas in England offer the potential to realise a more effective network of protected areas to reverse biodiversity decline. To do so requires a re-envisaging of both landscape-scale designations and SSSIs within the mechanisms and opportunities presented by recent policy development. The recommendations of the Landscape Review (DEFRA 2019) underline the need for reform of AONBs and national parks and the adoption of a renewed mission to ‘recover and enhance nature’. The scale of the existing national parks, and an overlapping network of SSSIs and other designated sites, combined with the proposed mechanisms for delivering on the 25-year environment plan (ELMS, NRNs) present an opportunity to realise policies that make England’s protected area network the cornerstone of effective conservation policy that reverses centuries of biodiversity decline. Given the growing body of evidence that increasing the size and quality of protected areas is of greater importance than connectivity in sustaining regional populations of species, the ‘re-tasking’ may be simply expressed as making ‘*SSSIs bigger and National Parks better’*.

2. Delivering bigger and better protected areas

We have emphasised above ways in which the spatial properties of designated sites and their associated habitats and species populations, may explain the unfavourable condition of protected areas. Within-site changes, imposed by external land management and other activities in the risk landscape, are also problematic, not least pollutants and climate change. Such changes would require broad public and political support.

A further cause of unfavourable condition is the failure to restore on site-ecological or geophysical conditions. For example failure to halt the spread of scrub in grasslands and on geological exposures, or to prevent drying out and peat erosion in blanket bogs. This may have two main causes: i.) a failure to identify the appropriate management prescriptions and ii.) a failure to implement them, or they are implemented inadequately. To enlarge SSSIs and improve the biodiversity of National Parks and AONBs would not only need the areas for land use change to be defined but also require new management prescriptions to be identified and the necessary planning and actions to implement them. It is ensuring the connectedness between human activity and ecosystem change or maintenance which is at the heart of successful conservation.

Application of a systems approach to the improvement of protected areas therefore requires that we see them as a linked ecological-social system, in which optimally a complex set of social activities, including policy development, planning and land and water management, facilitate necessary ecosystem functioning. Increasingly it has been recognised there are many human benefits from this social investment in protected areas and other parts of the environment, further emphasising the linkage between ecosystems and society in the form of outputs. We need therefore to see protected areas in a broader socially engaged context, reflecting these recent, but now well established shifts in understanding and policy, as reflected in Government's 25-year environment plan.

Nature and people in conservation biology

Implicit to the delivery of effective protected area reform are assumptions about the underlying relationships between people and nature (Mace 2014). Historically nature conservation has largely been premised on a notion of nature as isolated from people, either in terms of isolated, self-sustaining natural wilderness or reserves protected against the effects of human activity. There was often a tacit assumption within the conservation community that legal protection of areas will enhance the long-term persistence of habitats and species (Gaston et al 2006).

Over the past two decades, however, emphasis has been placed on the benefits of nature to people, by attributing explicit social and economic value to semi-natural habitat through the valuation of ecosystem services. Following the publication of the Millennium Ecosystem Assessment (2005) and UK National Ecosystem Assessment (2011), government policy in England has recognised the value of the many ecosystem services delivered by the "natural capital" within protected areas and other semi-natural habitats. The geological features within protected areas logically form part of this natural capital and also deliver ecosystem services, such as coastal protection and water purification (Gordon & Barron 2013).

Although it has its origin among natural scientists (Mooney & Ehrlich 1997), the concept of ecosystem services has been developed mainly in the field of environmental economics (e.g. Dasgupta 2021; UKMEA 2011, chapter 2). By better capturing the societal value of the services provided by protected areas and semi-natural habitats, these areas are integrated more readily into spatial planning – rather than remaining largely detached from such processes due to their legal status. A strength of this economics led approach is that it recognises that ecological and social systems are interdependent and has helped to reveal the fuller value of biodiversity and geodiversity to society. However, any such framing of the relationship between people and nature can at best be considered a convenient premise on which to elaborate theories, but insufficient to reveal the processes within the linked ecological and social systems which might be most important in enabling or inhibiting necessary conservation actions to be carried out.

There is also growing evidence of how exposure to wildlife affects people's support for pro-biodiversity policies and management actions (Evans et al 2018; Gaston & Soga 2020). As such, increasing interactions with nature is a key goal for a wide range of proposed nature-based interventions (McCurdy et al, 2010; Shanahan et al., 2019). However, neither an approach that considers ‘nature despite people’, nor an approach that considers ‘nature for people’ is able to capture how interactions between people and nature can provoke positive changes (Gaston et al. 2018). Coupled human-nature systems exhibit emergent dynamics that frustrate approaches that focus on aspects of environmental management in isolation (Lawton et al. 2010). An intuitive ability to identify these inter-relations has long been the mark of successful conservation. However, approaches to enhancing protected areas would be improved through a more quantitative and structured way of integrating information on the distribution of biodiversity and other natural assets with information on land ownership and management and the social interactions and processes that shape land-use decisions. We therefore consider how a more integrated, people-in-nature approach to conservation planning would focus on critical ecological features, individuals and organisations with the greatest influence on the performance of designated sites.

Nature and people in protected area design

To date, broadly two approaches have been used in order to try and integrate humans into conservation planning approaches seeking to design protected areas, a top-down or bottom- up approach. On the hand one, where formal systematic conservation planning tools and algorithms are used, socio-economic opportunities and constraints are quantified and incorporated as spatial information during the planning process. The objectives of these exercises are usually to satisfy biodiversity targets with minimum cost (Ball *et al* 2009), to produce priority rankings of areas most important for sustaining biodiversity

(Moilanen *et al* 2009), or to balance alternative land uses that have both a positive or negative effect on biodiversity (Moilanen *et al* 2011). In this way, provided that costs and opportunities can be appropriately quantified and mapped, including for example the forgone opportunities in terms of agricultural yield, the ease with which a protected area could be established or expanded to encompass a given area of land can be estimated. The second approach is one in which stakeholder engagement is placed at the heart of decision-making processes and the practical constrains and opportunities are instead used to dictate what is done and where. Here it is generally assumed that successful implementation of conservation management is unlikely to be achieved by top-down edict. Instead, it is assumed to require widespread social support, particularly from those most directly affected by any changes that result, is necessary for successful conservation.

In reality, both approaches have their merits and drawbacks. On the hand, the push for adoption of systematic techniques has been driven largely by the recognition that the ways in which protected areas identified in the past were largely opportunistic. Protected areas were predominantly established on land where the opportunity costs of setting aside land for nature conservation were minimal (Balmford & Whitten 2003). Scientifically defensible techniques were thus seen as the panacea for this problem of poorly targeted nature-conservation efforts (Knight & Cowling 2007). Nevertheless, in the real world, the successful selection and implementation of protected areas is the product of a complex suite of factors that are not reliably predictable and cannot easily be quantified and mapped *a priori*. Pressures of economic forces, organizational and institutional capacity, the willingness of land-owners to engage in conservation and the extent to which this is influenced by decision-making processes and other factors, mean that the recommendations of systematic conservation assessments are often difficult to implement in practise (Knight & Cowling 2007).

How might one therefore draw upon the merits of each? An initiative that sets out to expand protected area networks purely from a biological perspective is likely to fail because many of those who are influential in delivering this may not share the priorities produced by technical conservation planning exercises. Similarly, attempts to expand protected areas based purely on the opportunities presented or the views of land-owners and managers will almost inevitably fail to enhance biodiversity in the most effective way.

We argue that there is a need to more effectively bridge the gap between information generated by the technical activities of conservation planning assessments and the activities of stakeholder collaboration, strategy development and practical implementation carried-out by most conservation practitioners. Bridging this gap will enable conservation planners to acknowledge the importance of the fusion of knowledge traditions—and not only apply the natural sciences to conservation planning problems, but also embrace the knowledge and techniques of social research that will be essential for gathering the information to support the identification and implementation of conservation opportunities (Balmford & Cowling 2006). It will also require mapping of conservation opportunities to assist in decision making about not only where conservation action is required, but also when and how to implement actions when opportunities appear.

The flexibility of systematic approaches to configure alternative networks of proposed protected areas present an opportunity for meaningful stakeholder involvement. It is possible to present possible priority rankings and then update these in real-time to reflect the views of stakeholders on opportunities and constraints. In doing so one is able to recast the obstacles to achieving protected area enhancement as an opportunity and to improve the assessment's acceptability. In this way the historical nemesis of opportunism may be embraced and turned into an advantage (Knight & Cowling 2007).

Agency in protected area sustainability

While traditional systematic conservation planning tools allow identification of where conservation action is required, the complexity of both ecosystem and social systems present significant barriers to determining when and how to implement actions. The challenge for policy makers and practitioners is thus to identify those parts of the combined system which are most critical to successful outcomes. In this way one is able to pinpoint the most significant barriers to nature recovery and help individuals and organisations with the remit of protecting biodiversity better to focus policy and activities. One is also better placed to identify tipping points for nature recovery, namely those actions mostly likely to have a major positive effect. Fundamentally this relies upon carrying out key management interventions to restore of maintain ecosystem function, but the indivisibly of ecosystem and social systems requires working with those societal actors most able to deliver or frustrate this outcome.

We propose that the concept *of agency* may have value as an analytical tool that bridges the knowledge traditions and breaks down the barriers between compartmentalised disciplinary analyses. The terms agent and agency have been widely used in the social sciences, with slight differences of meaning. Broadly the term *agent* has been used to signify individuals and *agency* the term used to signify their ability to make choices (economics) or decisions (sociology). At a deeper level we consider both usages of the term agency relate to an ability to change or maintain a system.

Biological and physical agents

Using this broader definition we can therefore see that biological and even chemical and physical entities within the combined ecological-social system also have agency. Deer, for example, have the ability to alter profoundly the state or function of an ecosystem by affecting the regeneration dynamics of vegetation (Côté et al 2004), as does nitrogen by altering the diversity of grassland (Stevens et al 2006). Most important, is that agency is not evenly distributed and by understanding its distribution in the combined ecological-social system, the critical agents and the actions needed to influence them are revealed.

A particularly clear ecological example of the uneven distribution of biological agency is shown by those species identified as *ecosystems engineer*, a diverse group of organisms that individually "create" the habitat in which they occur and have an influence disproportionate to their biomass or abundance. A topical example is the European beaver

(*Castor fiber*) which has recently been reintroduced at a number of locations in Britain. This is in large part due to the beneficial ecosystem level change its dam building has upon biodiversity and the associated flood regulation benefits (Brazier et al. 2021). Another example of mammalian agency at ecosystem level is through the stocking of many protected areas with cattle and sheep to maintain or restore open habitats. On extensive complex suites such as the New Forest, livestock behaviour, including differences between seasons, has an important influence upon habitat, and by implication species, distribution (Putman 1986). In other circumstances, the presence of predatory species can enhance the diversity and the sustainability of ecosystems by restricting grazing intensity through the control of herbivore populations. In North America, for example, at the continental scale, wolf (*Canis lupus*) predation limits moose (*Alces alces*) to low densities, resulting in low browsing levels (Messier 1994). Extirpation of wolves is thought to have resulted in unprecedentedly high browsing pressure on plants in areas of the continent where wolves have disappeared (Crête 1999), though in recent years the extent to which they do so both within and outside of North America is widely recognised as being context-dependent (Ausilio *et al* 2021). The concept of rewilding is partially based in the premise that 'rewilded' species have agency on habitats and ecosystems via trophic-cascade (Bakker & Svenning 2018)

This uneven distribution of agency is also found among plants and microorganisms. For example *Sphagnum* mosses have a high level of agency. Through their ability to modify the acidity of the soil surface and store water; they create the conditions for bog development and maintenance. On the coast, marram grass (*Ammophila arenaria*) has a similar ecological role in dune systems.

We can extend the concept of agency to the abiotic components of the ecological-societal system. The most biodiverse habitats found in protected areas are mainly systems in which productivity is limited by nitrogen and/or phosphorous levels in the soil and water. Increasing the levels nitrogen and /or phosphorous has a disproportionately large positive impact upon productivity and negative impact on biodiversity of most ecosystems (Sterner and Elser 2002).

Social agents and agency

In society, agency can be assigned to both individuals and organisations. An individual farmer will have agency over the land they farm and areas downstream of this, or an organisation, such as Defra may have agency over the way in which land is managed through the administration of The Basic Payment Scheme to the agri-sector.

An important aspect of conservation discourse has been seeking influence upon an ill-defined group of social agents described as 'policy makers'. By implication this includes legislators, government ministers, civil servants and those who influence their opinions and choices. In principle, in a functioning democracy, the legislature has potential control over all aspects of environmental management, through its ability to pass legislation, enact policy and allocate resources. The way democratic government's agency is used reflects in part an environmental narrative which is perceived to have public support, but also a need to accommodate those in civic society with a more strongly vested interest in a policy outcome; due to its more direct influence upon their lives, most notably their income.

A consequence of a focus on policy-makers is that it restricts the view of agency in society and underestimates the importance of other actors in the chain of events needed to protect and manage biodiversity and geodiversity. In practice, and of necessity, other social agents often exercise considerable autonomy. These local actors have traditionally been far removed from technical conservation planning exercises, and include those who work on nature reserves, manage livestock and carry out other farm operations, or carry out groundworks in civil engineering projects. Many of these practices have developed by trial and error among practitioners and their environmental effectiveness is poorly evidenced (Rose et al 2019, Pullin et al 2004). An emphasis on top-down systematic techniques for identifying where protected areas are in need of expansion or enhancement is likely to feed a sense of disjunction and suspicion of the conservation discourse among those most directly involved in delivery. Where such top-down approaches have been applied in the past, the suspicion has been largely founded on the perception that there is a bottom-up one-way flow of information and a reverse top-down flow of 'advice' and legislation. Rather than feeling engaged by the discourse and helping to shape means of delivering conservation, many of those affected feel ignored or dictated to.

A second group of agents are those who produce and manage plans and projects, often involving allocation of resources. This may, for example, include at a local scale those who write nature reserve management plans or manage small and medium sized commercial enterprises, notably farm businesses. In these situations, practical action and planning is often vested in the same individuals. In larger organisations, such as government agencies, district councils and large businesses, this agency is often vested in distinct professional groups, such as planning officials and land agents, frequently involving specialist consultants. Where the private sector has an involvement with protected areas, this exercise of agency often involves property and other rights over natural assets. In the public sector such agency often involves regulatory authority.

Difference in the motivations, opportunities and resources of different agents along the critical path to ecosystem sustainability are a major obstacle to delivering better protected areas. The costs and benefits of enhancing biodiversity are not evenly distributed. In the case of SSSI designation or repurposing of AONBs and national parks, wider society may benefit from the ecosystem services provided by biodiversity, such as carbon capture, and a downstream community may benefit significantly from flood alleviation. However, costs may be incurred by those on whose land a SSSI is being proposed. Only by the identification of such connections can we address potential conflict issues and provide appropriate mechanisms for closing the loop between beneficiaries and those negatively affected to ensure management issues and decisions can be "shared as one" (Brazier et al 2021, Redpath et al., 2015).

Ultimately, nearly all policy, legislation and regulation, as well as the policies adopted by commercial and voluntary sector organisations, aims to influence ecological processes within specific protected areas and/or across the wider landscape. For this to be achieved it is necessary to identify a critical path, which sets out necessary actions, including those brought about by ecological agents. This should form a logical, often chronological, series linking legislation, regulation and goals to environmental outcomes. We suggest by understanding where critical social and ecological agents are located on the critical path, and how their agency might be engaged, the goals set by government and others might be better achieved.

Conclusions

Protected areas have been the centrepiece of nature conservation policy. However, despite a significant expansion in the number of protected areas, including in the UK, biodiversity has continued to decline. This points to a systemic failure in the way that biodiversity conservation has been carried-out, with both an ecological and social dimension. Nevertheless, the development of a 25-year environment plan and mechanisms for delivering this plan present a ‘once in a generation’ opportunity to change the way in which conservation is carried-out. To realise the opportunity, we need to readdress and take stock of the role of protected areas in enhancing biodiversity across the whole of the landscape. To do so requires re-envisaging both the design and delivery of protected areas.

With respect to improving the design and function of existing protected areas, this may be expressed simply as make ‘*SSSIs bigger and National Parks better’*. Because of their relatively small size, few SSSIs host self-sustaining populations of species. Moreover, only half are in favourable condition– most likely due to a combination of thin-spreading of limited resources and their vulnerability to effects from the wider landscape. For landscape-scale designations, including National Parks, there is a statutory requirement for governing bodies to have regard to the purpose of conserving biodiversity. In practise, however, their ability to protect biodiversity, is little different than that in the wider landscape.

The realisation of improved and expanded protected areas will not be achieved by top-down edict. Instead, it will require widespread social support, particularly from those most directly affected by any changes that result. There is thus a need to ensure that information generated by the technical activities of conservation planning assessments is coupled with the usual approaches of stakeholder collaboration and implementation carried-out by most conservation practitioners. This is crucial for achieving widespread acceptance and to ensure protected area enhancements is viewed as an opportunity as opposed to a cost. In so doing, however, there is a need to capture this information in a more structured and systematic way so that it can be fed into conservation planning process objectively.

While traditional conservation planning has entailed mapping of where conservation action is required, there is also a need to understand when and how to implement actions when opportunities arise. Doing so requires the identification of key agents and mechanisms that are either causes of biodiversity decline or effective agents of restoration. Identifying existing organisational agents capable of change can help better join up the policy, legal and economic responsibilities for reversing biodiversity decline, as well as better targeting resources, policy mechanisms and funding to organisations and individuals best able to deliver restoration and mitigation strategies. Overall, more holistic approaches to protected area management are needed, rooted in greater understanding of the inter-relations between people and biodiversity.

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