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Scaling up from protected areas in England: the value of establishing large conservation areas

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Abstract

Protected areas (PAs) are vital for conserving biodiversity, but many PA networks consist of fragmented habitat patches that poorly represent species and ecosystems. One possible solution is to create conservation landscapes that surround and link these PAs. This often involves working with a range of landowners and agencies to develop large-scale conservation initiatives (LSCIs). These initiatives are being championed by both government and civil society, but we lack data on whether such landscape-level approaches overcome the limitations of more traditional PA networks. Here we expand on a previous gap analysis of England to explore to what extent LSCIs improve the representation of different ecoregions, land-cover types and elevation zones compared to the current PA system. Our results show the traditional PA system covers 6.37% of England, an addition of only 0.07% since 2001, and that it is an ecologically unrepresentative network that mostly protects agriculturally unproductive land. Including LSCIs in the analysis increases the land for conservation more than tenfold and reduces these representation biases. However, only 24% of land within LSCIs is currently under conservation management, mostly funded through agri-environment schemes, and limited monitoring data mean that their contribution to conservation objectives is unclear. There is also a considerable spatial overlap between LSCIs, which are managed by different organisations with different conservation objectives. Our analysis is the first to show how Other Effective Area-Based Conservation Measures (OECMs) can increase the representativeness of conservation area networks, and highlights opportunities for increased collaboration between conservation organisations and engagement with landowners.

1. Introduction

Terrestrial biodiversity is under unprecedented pressure, despite intensifying conservation efforts. Protected areas (PAs) have long been used to mitigate these threats by separating biodiversity and incompatible land uses, and now cover 14.6% of the global terrestrial realm (Watson et al. 2014). Moreover, PA networks are continuing to expand, as most national governments have committed to increase the proportion of their land surface under conservation to 17% by 2020 (CBD, 2011). However, even with this new commitment, conservation success is far from guaranteed (Venter et al., 2014). This is because PA networks have often developed in an *ad hoc* manner and have three features that limit their effectiveness. First, many PAs are small and isolated, and so cannot maintain broad-scale ecological processes or sustain viable populations of wide-ranging species (Armsworth et al., 2011). Second, PAs are often placed in remote areas with little economic potential (Joppa and Pfaff, 2009), leaving many ecosystems and species poorly represented (e.g. Iojă et al., 2010; Jackson and Gaston, 2008). Third, PAs fix conservation efforts in space based on conditions at a certain time, while ecosystems and their threats are dynamic (e.g. Araújo et al., 2011).

These problems are evident in England, where much biodiversity is restricted to small, privately owned fragments of semi-natural habitats. Most of these habitats have been shaped over thousands of years by anthropogenic use and management, but have suffered significant fragmentation and degradation in the last century (Lawton et al., 2010). The English PA network is based on a restrictive zoning approach (Lawton et al., 2010), which uses planning legislation to identify National Natural Reserves (NNRs) and Sites of Special Scientific Interest (SSSIs) and then limit damaging development within them. Historically, this network has comprised of mostly small (< 1km^2) and isolated PAs (the median size of SSSIs and NNRs are 0.2 km^2 and 1.1 km^2 respectively), typically confined to uplands and ecoregions with low agricultural potential (Oldfield et al., 2004). To overcome these limitations, the United Kingdom (UK) has adopted a complementary approach based on agri-environment schemes and other incentive-based payment schemes. These pay landowners for income foregone and to cover the costs of management actions designed to improve landscape quality for conservation or other objectives, thereby providing an important source of funding for conservation inside and outside PAs. In England, the European Union's Common Agricultural Policy has funded agri-environment schemes since 1987 (Bright et al., 2015). Until recently, these schemes included Higher-Level Stewardship (HLS), which supported intensive habitat maintenance and restoration within target areas in production landscapes (Natural England, 2012), and English Woodland Grants that funded projects to restore and manage woodlands (Raum and Potter, 2015). Both of these were replaced in 2016 by the new Countryside Stewardship scheme (Natural England, 2015) and the UK's departure from the European Union could bring further changes.

Past research has shown that the English PA network is relatively effective at representing species and plays a major role in supporting species in response to climate change (Gaston et al., 2006; Gillingham et al., 2015; Jackson et al., 2009). However, 56% of species in the UK have declined since 1970 (Hayhow et al., 2016), underlining the limitations of the PA network and agri-environment schemes. Recognising this problem, the UK government commissioned work on how to improve nature conservation and ecosystem service provision (Lawton et al., 2010; NEA, 2011). These recommended a more proactive approach to improving England's ecological networks, based on landscape-scale habitat restoration

(Defra 2011) with five key steps identified to help achieve this objective: (i) improve habitat quality; (ii) increase the size of habitat patches; (iii) enhance connectivity; (iv) create new sites, and; (v) improve the wider environment (Lawton et al., 2010).

These government reviews provided renewed impetus to a trend that had been developing across the UK conservation sector. In particular, several conservation non-governmental organisations (NGOs) recognised the need for new large conservation areas, which should extend beyond the boundaries of existing PAs to encompass whole landscapes. These NGOs have established their own schemes to develop large conservation areas, such as the Royal Society for the Protection of Birds' "Futurescapes" (RSPB 2001) and the Wildlife Trusts' "Living Landscapes" (Wildlife Trusts 2007). There is also an increasing appetite for greater collaboration among and between conservation NGOs and local and national governmental agencies to support existing and new initiatives (Macgregor et al., 2012).

It was in this context that a recent project explored large-scale conservation initiatives (LSCIs) in England, Scotland and Wales, where LSCIs were defined as any area larger than an arbitrary threshold of 10 km^2 that is actively managed for biodiversity conservation goals (Eigenbrod et al., 2017). This research looked at the different categories and locations of LSCIs, the factors involved in their planning and management, and their environmental benefits (Adams et al., 2016; Eigenbrod et al., 2017; Macgregor et al., 2012). This analysis identified over 800 LSCIs in England, Scotland and Wales, which were subsequently categorised based on land tenure and management strategy (Macgregor et al., 2012). This large number of LSCIs highlights the growing interest in the approach in the UK. However, despite their number and appeal, there is little evidence on whether these new initiatives have resulted in a more representative PA network. The aim of this paper is thus to explore the extent to which LSCIs and agri-environment schemes have complemented the current network of PAs to reduce spatial biases.

The best way to explore this question is to undertake a gap analysis, a spatially resolved quantitative approach for measuring how well PA networks represent biodiversity and protect different biogeographic zones, land-cover types and species (e.g. Jenkins et al., 2015; Scott et al., 1993). Here we conduct the first ever gap analysis of the relative contribution of PA, LSCIs and agri-environment schemes, focusing on these different conservation area types in England. We begin by measuring how England's PA network has changed since a 2001 gap analysis in terms of extent and protecting different ecoregions and elevation zones (Oldfield et al. 2004). We then assess the contribution of two other major categories of conservation management initiatives: large-scale conservation initiatives (LSCIs), using the recently created LSCI database (Eigenbrod et al., 2017), and; incentive payment areas (IPAs) based on agri-environment and woodland improvement schemes. This involves measuring the overlap in the PA, LSCI and IPA networks, and the extent to which land under these management types cover the different ecoregions, land-cover types and elevation zones. In doing so, we test the hypothesis that Other Effective Area-Based Conservation Measures (OECMs), as highlighted in the Convention for Biological Diversity's Aichi target 11 (CBD 2011), reduce some of the limitations of the original PA network by better representing England's ecoregions and land with higher socio-economic value.

2. Methods

2.1. Types of conservation areas

We distinguished four categories of conservation areas in our analysis:

1. Protected areas (PAs). We focused on National Nature Reserves (NNRs) and Sites of Special Scientific Interest (SSSIs), the core statutory designations for biodiversity protection in England. We did not include European and internationally designated PAs in this analysis, because they are already included as NNRs or SSSIs, and we excluded National Parks and Areas of Outstanding National Beauty because non-PA land within such areas is normally not managed with conservation as a primary objective (Oldfield et al., 2004).
2. Type 1 Large Scale Conservation Initiatives (LSCIs). These consist of large, privately-owned land parcels that are managed by one or a few organisations or individuals, typically for long periods of time. Examples include the Great Fen Project, Wild Ennerdale and Wicken Fen Vision (Table S1). Type 1 LSCIs are currently managed primarily for conservation.
3. Incentive Payment Areas (IPAs). These are agricultural land parcels receiving HLS or woodland grant scheme payments (Natural England, 2012; Raum and Potter, 2015) under renewable ten year contracts. We excluded land under Entry-Level Stewardship schemes, as they cover only a small proportion of any land holding and support broader environmental improvement actions rather than conservation management (Davey et al., 2010).
4. Type 2 Large Scale Conservation Initiatives (LSCIs) represent large areas that are typically proposed to be managed for biodiversity conservation. They consist of many land parcels managed by different organisations or individuals, but guided through a single conservation initiative overseen by an organisation or partnership. Examples include the

UK Government's Nature Improvement Areas, the RSPB's "Futurescapes" and most of the Wildlife Trusts' "Living Landscapes" (Table S1). The majority of Type 2 LSCIs include PAs and farmland and thus have multiple management objectives. The conservation objectives are often achieved through shorter-term projects that encourage people to improve the conservation, ecosystem service and/or social capital value associated with their land. Project lengths are variable, often built from sequences of funding rounds, and benefits frequently only last as long as the funding.

These four conservation area categories are known to overlap, so we ranked them according to their conservation objectives, letting us report the amount of land belonging to the management category that gave the highest weight to conservation (Table 1). PAs were assigned the highest management category, followed by Type 1 LSCIs, Incentive Payment Areas and, finally, Type 2 LSCIs. This hierarchy was used because: PAs are managed for conservation; Type 1 LSCIs have similar goals to PAs, differing only in not having statutory obligations to manage the whole site for conservation; IPAs are likely to have more biodiversity benefits on land managed specifically for conservation, and; Type 2 LSCIs include land that is not currently managed for biodiversity, and the areas that are managed for conservation fall within existing PAs or IPAs.

2.2. Data collection and preparation

We used data held by Natural England on NNRs, SSSIs, Type 1 LSCIs and Type 2 LSCIs in 2013, as well as IPAs as of December 2013. Information was extracted from the existing database (Eigenbrod et al., 2017), for the 341 LSCIs that are found in England, have defined boundaries and meet the Type 1 or Type 2 criteria. We then used the Land Cover Map 2007 to exclude urban areas from each LSCI. The IPA boundaries were from maps of land holdings with HLS and woodland grant scheme agreements. We only considered those stewardship options which contribute to conservation. Where farm agreements contained at least one whole-farm option, we considered the entire farm as an IPA. If this was not the case, we used the HLS data to map the IPA land parcels (see Text S1 for further details). We clipped all of these datasets with the England political boundary to exclude any estuarine or marine areas (following Oldfield et al., 2004).

To determine the characteristics of the different conservation management categories, we used datasets describing elevation, slope, distance to infrastructure, ecoregion type, agricultural land quality and land-cover class. All of these data types were used in previous gap analyses to measure the representativeness of PA networks and the extent to which PAs are found in remote areas on land with low agricultural potential (e.g. Oldfield et al., 2004; Pressey and Tully, 1994). We did not use the available species distribution data because much of it has a spatial resolution of 10 km x 10 km, which is a great deal coarser than the majority of the PAs and agri-environment scheme land parcels, making it impossible to measure levels of species representation with precision.

The first step in the analysis was to produce six GIS layers derived from five spatial datasets, which were resampled to produce GIS layers with the same resolution of 80 m (matching the dataset with the coarsest resolution). Three of the layers described physical factors. We used the SRTM Digital Elevation Model (DEM) to produce the elevation zone layers (Table S2), where each elevation value was assigned to one of the following four classes: 0 to 200 m; 201 to 400 m; 401 to 600 m and > 600 m. We also used this DEM to produce the slope layers using the Slope function in ArcGIS (ESRI 2011; ArcGIS Desktop: Release 10. Redlands, CA). To produce the remoteness layer we used national data on public transport infrastructure (Table S2) and calculated distance from nearest transport node points (e.g., bus stops and train stations).

Another three layers described ecological and environmental factors. For ecoregions we used the National Character Areas (NCA) layer produced by Natural England (Table S2). The NCA layer subdivides England so each of the 159 NCAs (which we term "ecoregions" hereafter) represent a unique combination of landscape, biodiversity, geodiversity, cultural and economic activity. We also used the Provisional Agricultural Land Classification (Table S2) dataset, which divides England into five categories of agricultural land (with grade 1 representing the highest and grade 5 the lowest respectively) and two additional categories of land in non-agricultural use (i.e. non-arable and suburban). We used the Land Cover Map 2007 (Table S2), derived from satellite imagery, to produce the land-cover layer by reclassifying the original 23 land-cover types into seven: (i) coastal, salt and freshwater; (ii) mountains, heath and bog; (iii) woodland; (iv) semi-natural grassland; (v) arable; (vi) suburban, and; (vii) urban.

2.3. Data analysis

We calculated the percentage overlap between the different conservation area categories by converting the vector file for each into a raster format with an 80 m resolution, and using the Raster Calculator in ArcGIS to identify each combination of categories. Given the overlap between the conservation area categories, there were 15 combinations (e.g. PA + Type 1

LSCI), which were reclassified to the category that gave most weight to conservation based on the hierarchy described above and in Table 1.

We used ArcGIS to determine the characteristics of these different management categories based on the elevation, slope and remoteness layers. We did this by randomly selecting and extracting data from 1000 points of land belonging to each management category (i.e. PAs, Type 1 LSCIs, Type 2 LSCIs and IPAs) and land not within a conservation area. This helped ensure our sampling points were spatially independent and also avoided identifying statistically significant but negligible differences because of the large sample size. We then used non-parametric Kruskal–Wallis rank tests and *post-hoc* pairwise Wilcoxon rank tests with a Bonferroni correction to explore differences between the management categories, since homogeneity of variance and normality assumptions were not met. This random sampling with replacement of 1000 locations was repeated ten times for each environmental variable and the data were analysed using R.2.12.2 (R Development Core Team 2007). To provide an overview, we also reclassified the elevation, traffic node distance and slope layers into classes. We then calculated for each conservation management category the proportion of land that fell within each class, and compared this to the overall land that fell in each class of elevation, traffic node distance and slope across England (following Eigenbrod et al. 2009; see Table S3 for more details).

We conducted a gap analysis to assess the extent to which the different conservation management category networks represent surrogates associated with biogeographic differences in biodiversity. This involved calculating the percentage of each ecoregion, elevation zone, agricultural land quality class and land-cover class under each conservation management category, based on data extracted using the Tabulate Area function in ArcGIS.

Finally, we calculated the protection equality scores for the conservation area networks. This approach is based on the Gini coefficient (Barr et al., 2011), and describes how cumulatively adding land belonging to the different conservation management categories changes the extent to which every ecoregion is protected equally. We only used data on ecoregion coverage because protection equality scores are more robust when based on a large number of conservation features, and because the different ecoregions already represent the different elevation zones, land-cover classes and land quality classes (for further information on the calculation of protection equality scores see text S2).

3. Results

3.1. Temporal changes in PA coverage

The 4335 nationally designated terrestrial PAs (NNRs & SSSIs) cover 6.37% (8,322.4 km^2) of England's land surface (Figure 1), representing an increase of 83.6 km^2 (0.07%) since 2001 (Table 1). The increase had little impact on the median area of individual PAs, which at 0.17 km^2 is similar to that in 2001 (Oldfield et al., 2004). This is because 82% of the 4111 SSSIs and 46% of the 224 NNRs are smaller than 1 km^2. Many ecoregions are still poorly represented, with 78% of the 159 ecoregions having < 10% of their area protected by PAs (Figure 2a). Similarly, the percentage of PAs within the 0-200m elevation zone (Figure 3a), which represents 87% of England's terrestrial area, remains unchanged since the 2001 analysis at 3.5%, showing a consistent spatial bias in PAs towards upland areas.

3.2. Extent and overlap between the different conservation management categories

Land under LSCIs and IPAs is much larger than the land dedicated to formal PAs (Figure 1). Adding the large privately owned Type 1 LSCIs expands the net coverage of England by only 1%, because they cover < 1% of England's land surface and 37.9% of their area is already protected by PAs (Table 1). However, adding the IPAs nearly triples the land under conservation management from roughly 9,000 to 23,000 km^2, increasing coverage to 20.5%. Adding Type 2 LSCIs, which are managed by multiple different organisations or individuals, further increases this coverage to nearly 64% of England's terrestrial surface (Figure 1, Table 1), as 76% of the land in these Type 2 LSCIs is not part of a PA or an IPA and so it is only proposed to be managed for biodiversity conservation (Figure 1).

3.3. Characteristics of the different conservation management categories

Areas where conservation objectives were prioritised tended to be in upland areas, on land with lower agriculture quality and in more remote areas, e.g. coastal, wetland and montane areas (Figure 3). A greater proportion of PAs and Type 1 LSCIs contained woodland and semi-natural grasslands than was the case for Type 2 LSCIs. PAs and Type 1 LSCIs were on average higher, more remote, and steeper, while Type 2 LSCIs were lower, less remote and flatter (Figure 4; Table S3). These patterns were mirrored in the protection equality results. The PA network on its own had a protection equality score of 32%, because many ecoregions had negligible levels of protection, while a few upland and heathland ecoregions had PA coverage of > 40% (Figure 2). Including the Type 1 LSCIs made little difference to this result, increasing protection equality to 34%. However, adding land in IPAs increased protection equality to 62%, and also including land in Type 2 LSCIs increased it to 74% (Figure 2, Figure S1).

4. Discussion

Expanding conservation efforts beyond PAs is a step change in nature conservation policy for many countries (Boitani et al. 2007; Lawton et al. 2010; Reyers et al. 2012), but its importance is increasingly recognised. For example, the Convention on Biological Diversity's Aichi target 11 recognises that PAs are not the only approach for achieving goals for expanding land under conservation, and explicitly states the value of “other effective area-based conservation measures” (CBD, 2011). England is one of the pioneers, as shown by the development of hundreds of LSCIs, all of which aim to bring together different stakeholders and improve nature conservation through increased action and investment (Macgregor et al., 2012). One strength of this approach is that it is decentralised, allowing projects to match local conditions, but measuring the effectiveness of these LSCIs at a national level is important to inform general policies and strategies. This is why we used a gap analysis to explore the extent to which LSCIs help scale-up conservation efforts from PAs. We found LSCIs could substantially improve representation of less remote, flatter, lowland areas, with higher grades of agricultural suitability. However, the impact of LSCIs on conservation will depend on how they are planned and managed, which is an important caveat, because most of the land under Type 2 LSCIs is not currently managed for conservation. Our case study is the first to measure the relative contribution of LSCIs and land under agri-environment schemes to producing representative conservation area networks and provides a number of insights to inform policy and practice in human-dominated landscapes around the world.

4.1 Protected area coverage

A key step in improving the representativeness of any PA network is undertaking a gap analysis to identify species, habitats and ecoregions needing further protection. Such analyses should be undertaken periodically to evaluate progress (Margules and Pressey, 2000; Pressey et al., 2013). Our study adopts this approach by repeating a gap analysis for England undertaken over a decade ago (Oldfield et al., 2004). In England there are two main types of PA established for biodiversity conservation, namely NNRs and SSSIs. These covered 6.3% of England's land surface over a decade ago (Oldfield et al., 2004) and our results show how little this has changed, with only a marginal increase. The mean size of these PAs also remains small, although the maximum size has risen from 160 km^2 to 440 km^2, reflecting the success of several initiatives to join up existing areas.

Despite a decade of government and conservation NGO efforts, the PA network still poorly represents England's different ecoregions and elevation zones (Oldfield et al., 2004). For example, 78% of ecoregions have < 10% PA coverage with only 3.5% of English lowlands protected. These analyses also provide more detailed information on the spatial distribution of the current PA network, reinforcing that it is still biased towards remote, upland areas with lower agricultural potential. This helps explain why almost half of the PA network is composed of land-cover classes associated with relatively remote or inaccessible land, such as coastal, montane and wetland vegetation. It should be noted that many of these vegetation classes have conservation importance and the PAs also contain a high percentage of woodland and semi-natural grassland. This suggests that although the PA network is failing to represent different ecoregions adequately, it is protecting many important sites for biodiversity.

Such a bias in PA network coverage is common, as most national networks over-represent areas of low potential economic value (Joppa and Pfaff, 2009), but this tendency seems to be particularly strong in England. This is because the English PA system's protection equality score of 32% is lower than that of many other nations (Barr et al., 2011), although similar to some other countries in Western Europe, such as Italy (33%) and France (39%). However, comparison of equality scores requires caution, as they are based on the assumption that every conservation feature deserves equal protection and thus implicitly has equal conservation value. This is rarely the case but England, like most other countries, lacks nationally agreed targets on how much of each ecoregion should be protected. In the absence of such targets, the protection equality analysis provides a starting point to analyse the extent to which PA networks are representative.

4.2 The role of Large Conservation Areas

The LSCI approach is seen by many as one of the most effective ways of achieving the required change in conservation efforts, to meet both national and international obligations (CBD 2011; Macgregor et al., 2012). We investigated the current role of LSCIs by dividing them into two groups based on tenure and level of management for conservation objectives. Type 1 LSCIs are owned and managed primarily for conservation by one or a few landowners and are often based on several existing NNRs and SSSIs. There are relatively few of these LSCIs and nearly half of them have PA status, which explains why adding them to the gap analysis made little difference to the area dedicated for conservation or the spatial bias in the area conserved. This is probably because the mechanism for establishing such LSCIs is similar to the creation of large PAs, involving considerable land acquisition costs (Naidoo et al., 2006). Once established, management costs per unit area decline as PA size increases (Armsworth et al., 2011; Ausden and Hirons, 2002), suggesting Type 1 LSCIs have financial as well as ecological benefits when compared to a set of smaller PAs. However, creating such LSCIs requires the availability of large blocks of existing conservation land, or willingness on the part of adjacent landowners to sell or lease their land for conservation, which is unlikely on high-quality agricultural land (Adams et al., 2014; Knight et al., 2010).

In contrast, Type 2 LSCIs are much more widespread than PAs and Type 1 LSCIs and, partly because of this, do not show similar spatial biases. However, another reason for this lack of bias is that most Type 2 LSCIs are long-term initiatives for increasing land under conservation, and at present they are largely made up of land that is not managed for biodiversity. Our results show that only 24% of the land under Type 2 LSCIs is currently managed to achieve conservation objectives (i.e. PAs or IPAs). Caution is therefore needed when interpreting our results, as much of the higher quality agricultural land within Type 2 LSCIs is likely to have little current biodiversity value, nor much immediate prospect of being managed for conservation, given that individual landowners are not obliged to engage with or sustain any LSCI process. Moreover, even those who do manage their land for conservation might only do so on selected land parcels rather than across the entire holding. This means that at the moment a better measure of conservation land comes from IPA coverage, as these represent land parcels managed through specific suites of conservation mechanisms (Knight et al. 2010). Adding the IPAs to the gap analysis increases the land under conservation from 7.4% to 20.5%, when compared to a network of PAs and Type 1 LSCIs; substantially reducing spatial biases and improving protection equality.

Our results also show that agri-environment payments are important for funding conservation within LSCIs, although there is limited information on the cost-effectiveness of these IPAs when compared to PAs (Batáry et al., 2015; Kleijn et al., 2006). Despite this knowledge gap, agriculture is likely to remain a key component of any type of LSCI in England and elsewhere in Europe, so short term incentives will remain vital for encouraging some landowners to manage their land for biodiversity. Thus, conservationists will need to focus efforts to ensure the most important areas are protected, and that connectivity is maintained and enhanced within these production landscapes. To achieve conservation objectives in the long-term, it is likely that other forms of funding will be needed and that conservation organisations will have to secure permanent conservation management on more land within LSCIs.

5. Conservation implications

The English government has set an ambitious goal to “halt overall biodiversity loss, support healthy well-functioning ecosystems and establish coherent ecological networks, with more and better places for nature, for the benefit of wildlife and people” (Defra, 2011). We found that Type 2 LSCIs, areas that are typically proposed to be managed for biodiversity conservation, cover extensive areas of England and so could play an important role in achieving this goal, complementing the current PAs and Type 1 LSCIs. Indeed, both NGOs and government agencies now see LSCIs as an essential part of conservation in England (Adams et al., 2016). However, the success of those initiatives in achieving these national goals depends heavily on the way they are funded, planned, managed and monitored (Macgregor et al., 2015). Finding solutions to these important issues is challenging, but could help inform every country seeking to implement LSCIs as a way of scaling-up their conservation efforts and achieving their international commitments (CBD 2011).

With regards to funding Lawton et al., (2010) argued that, in addition to their importance for biodiversity value, the value of ecosystem services provided by LSCIs outweigh the costs. However, like many other countries, England lacks mechanisms to transfer such funds, so the NGOs and government agencies that establish LSCIs receive little financial benefit for maintaining these ecosystems. Moreover, a recent study showed that restoration costs can exceed the market value of ecosystem services based on carbon storage, crops, livestock and timber (Newton et al., 2012), suggesting additional funding would be needed to establish LSCIs and restore functioning ecosystems within them. Our work highlights the potential contribution that agri-environment schemes could play in funding such efforts, although the effectiveness of current approaches is mixed and could be improved (Batáry et al., 2015; Kleijn et al., 2006; FERA, 2013). Funding for schemes in Type 2 LSCIs could boost landowner engagement and also be used to assist farmers with completing the paperwork associated with such funding schemes, which can be a significant barrier to participation (Christensen et al., 2011).

Planning and management of Type 2 LSCIs is similarly challenging, since they typically encompass a large number of individual land holdings and land owners, and our results show most of the land is not managed specifically for achieving conservation objectives. There is also a considerable temporal and spatial overlap between different LSCIs, with each overlapping project being overseen by different configurations of NGOs, government agencies and partnerships (Eigenbrod et al., 2017), but often with distinct conservation objectives. Thus, the conservation benefits of these schemes depend on integrating a multitude of stakeholder values and policies to prioritise and implement conservation action that complements the existing PA network (Adams et al., 2016). These complexities suggest a target-based spatial conservation prioritisation approach would be helpful, based on existing empirical data and expert knowledge, as such systems are designed to guide the prioritisation of conservation efforts, and to help understand and balance associated trade-offs (Carwardine et al., 2009; Metcalfe et al., 2015).

Such an analysis could usefully follow a two-tiered approach: a national-scale spatial conservation prioritisation to identify broad focal landscapes, followed by fine-scale analyses within each of these landscapes to identify when and how conservation action should be implemented. The second tier would involve local partnerships determining the best approach to take within these priority landscapes and the specific areas to focus on, based on local data and knowledge of opportunities and constraints (Smith et al., 2009). There are considerable benefits, in terms of building financial, human and intuitional capital, of adopting a systematic conservation planning approach at the landscape and LSCI level (Bottrill et al., 2012). This approach could be used to develop more detailed conservation goals, increase collaboration between individuals and organisations and so identify options for reducing overlap and costs. This would help to ensure that nationally important biodiversity was protected, but in a way that would maximise local buy-in and likelihood of implementation.

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Figures and Tables legend

Table 1:

Statistics describing the land under different conservation management categories found in England, presented in hierarchical order based on the weight given to conservation as a management objective (high to low). We present the total area, as well as the net cover for each category after accounting for overlaps with land in higher conservation categories. The percentage overlap is calculated as the net cover divided by the total area of each management category.

Figure 1:

Land area in England under the four conservation management categories, ordered by the weight given to conservation as a management objective, from highest (protected areas, PAs) to lowest (Type 2 Large-scale conservation Initiatives, LSCIs), and land not managed for conservation (unmanaged). Land belonging to these different categories often overlaps so the map shows the highest conservation management category for any land parcel.

Figure 2:

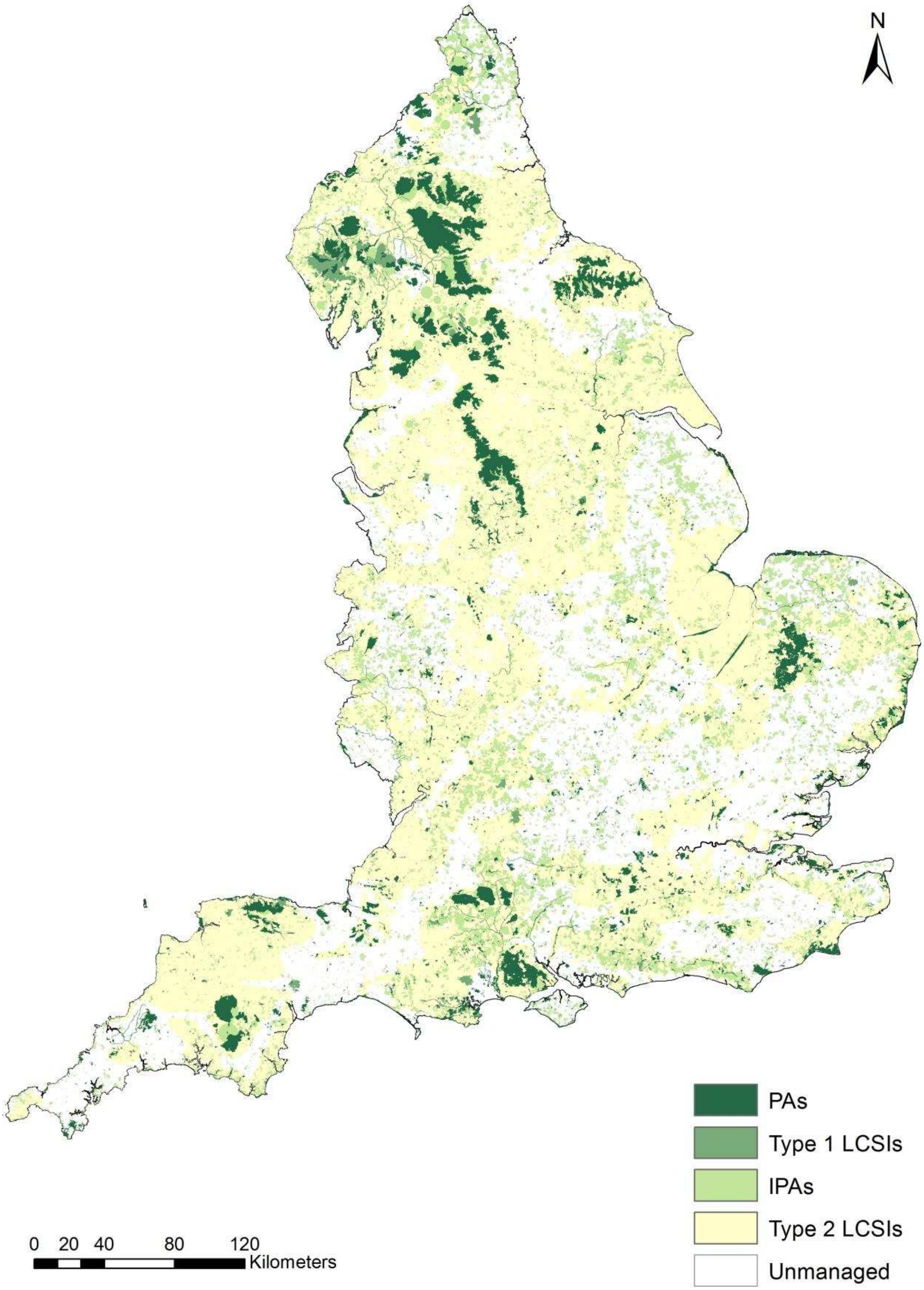
The cumulative percentage of protected area within each National Character Area (NCA) ecoregions for the four conservation management categories: a) protected areas (PAs); b) PAs and Type 1 Large-scale Conservation Initiatives (LSCIs); c) PAs, Type 1 LSCIs and Incentive Payment Areas (IPAs); d) PAs, Type 1 LSCIs, IPAs and Type 2 LSCIs.

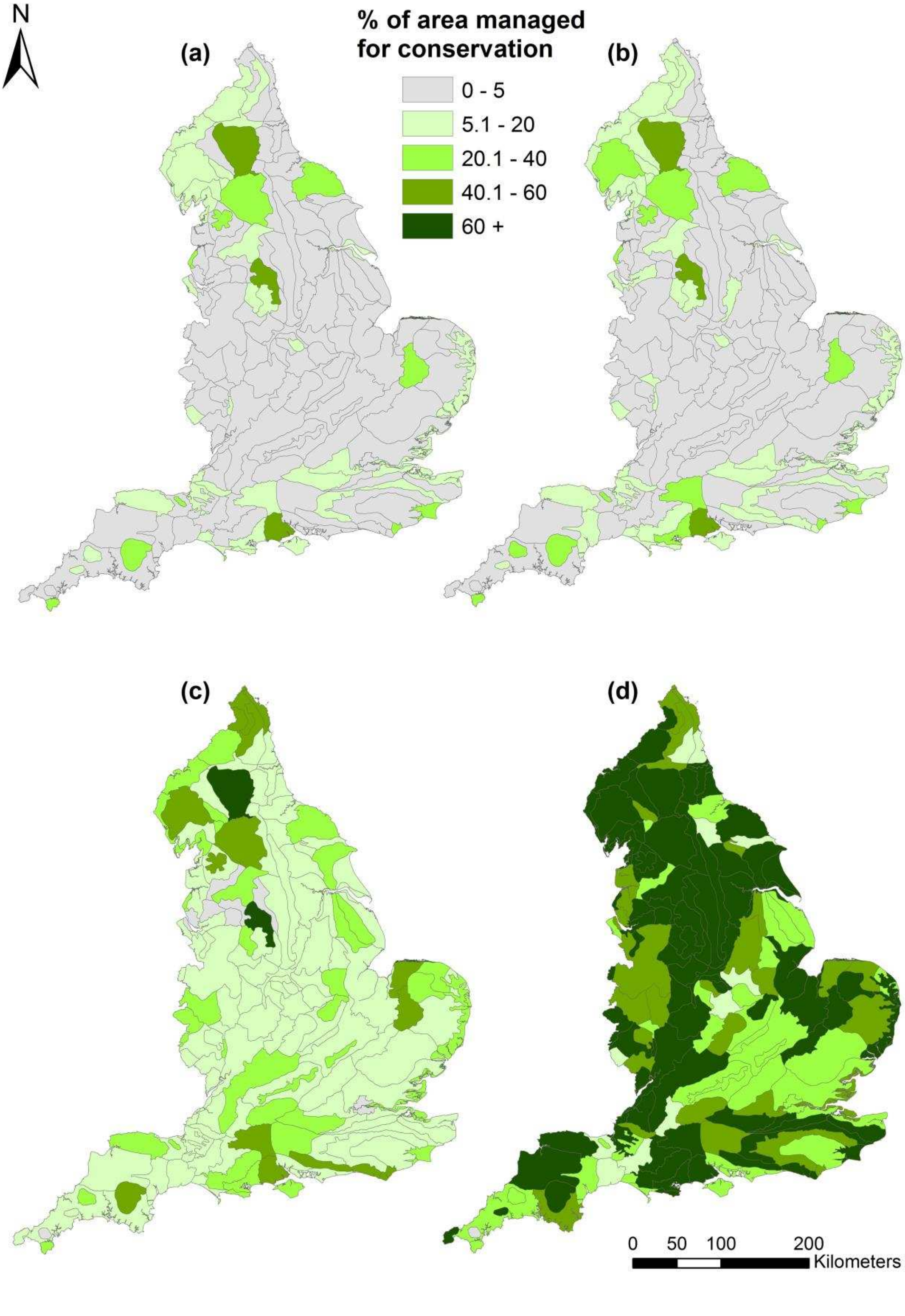
Figure 3:

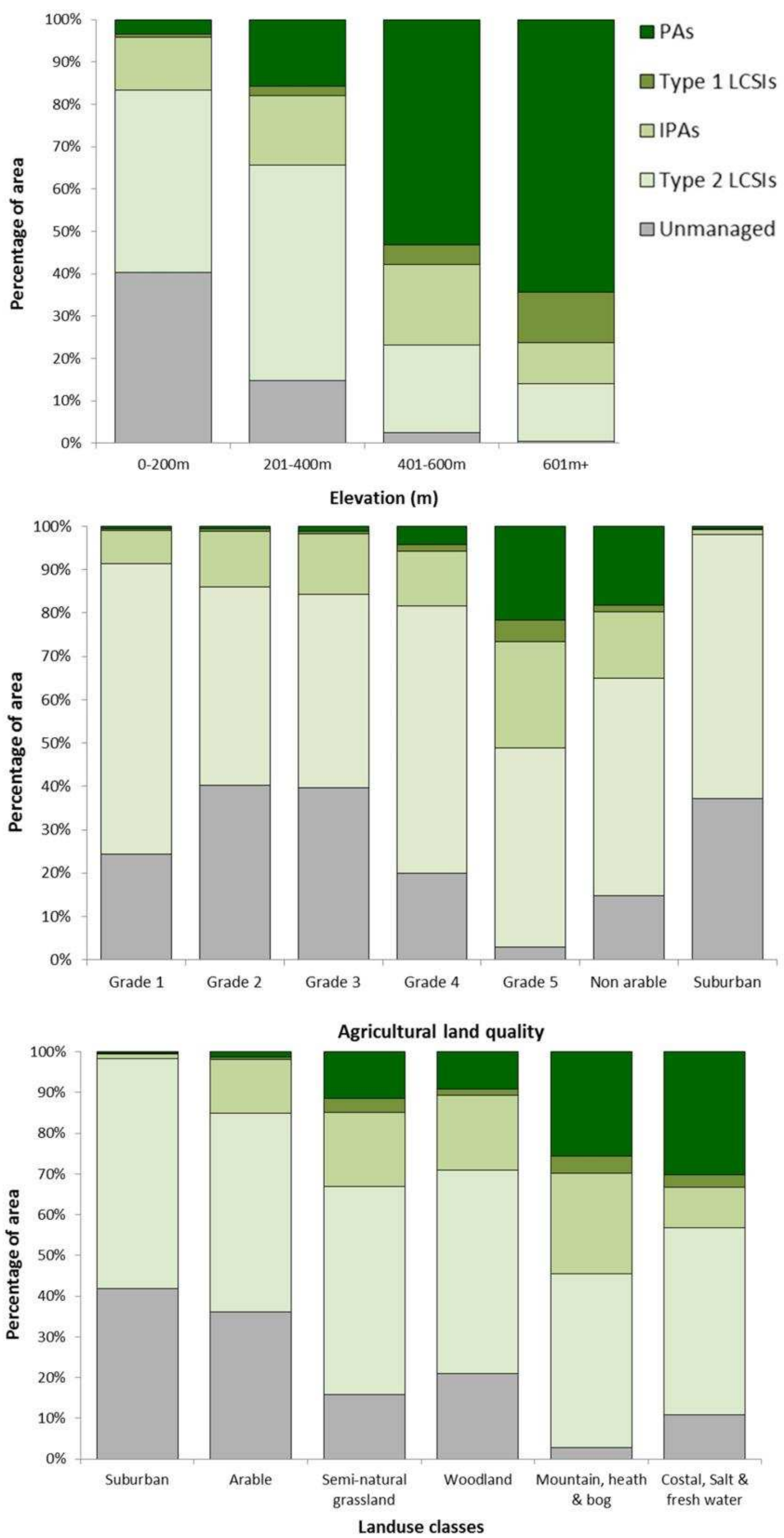
Details of the different conservation management categories by: (a) elevation classes, (b) agricultural land quality; and (c) landcover class. These categories are protected areas (PAs), Type 1 and Type 2 Large-scale Conservation Initiatives (LSCIs) and Incentive Payment Areas (IPAs).

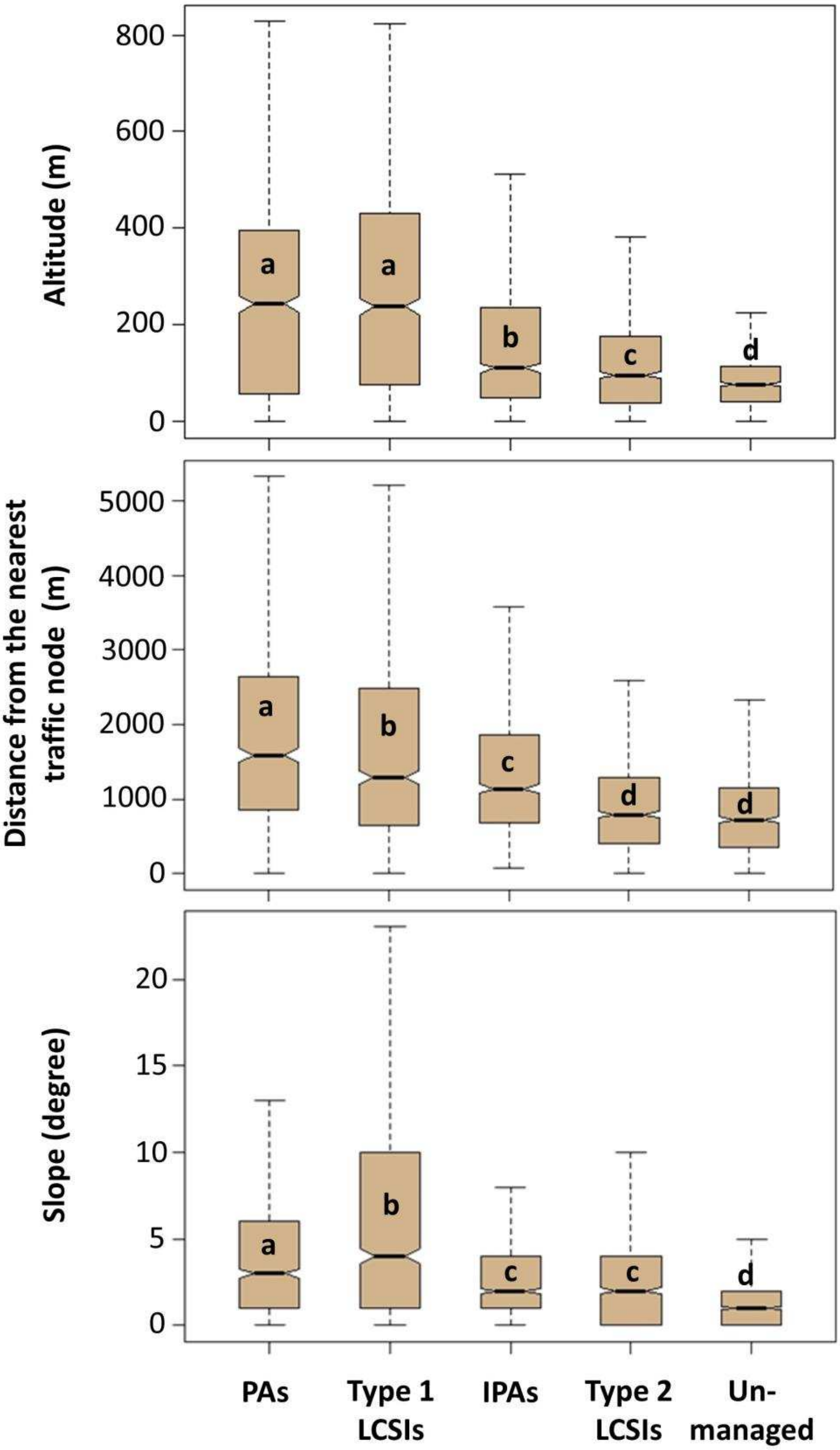
Figure 4:

Altitude, distance from traffic nodes and slope of the four conservation management categories (PAs, Type 1 and Type 2 LSCIs and IPAs) and unmanaged land. We used pairwise Wilcoxon tests to explore differences between all possible management category pairs and used the Bonferroni correction to account for multiple testing. Significant differences ($p<0.05$) between management categories are indicated by letters.


N
PAs
Type 1 LCSIs
IPAs
Type 2 LCSIs
Unmanaged
0 20 40 80 120
Kilometers


N
% of area managed
for conservation
0 - 5
5.1 - 20
20.1 - 40
40.1 - 60
60 +
(a)
(b)
(c)
(d)
0 50 100 200
Kilometers


PAs
Type 1 LCSIs
IPAs
Type 2 LCSIs
Unmanaged
Percentage of area
100%
90%
80%
70%
60%
50%
40%
30%
20%
10%
0%
0-200m
201-400m
401-600m
601m+
Elevation (m)
Grade 1
Grade 2
Grade 3
Grade 4
Grade 5
Non arable
Suburban
Agricultural land quality
Suburban
Arable
Semi-natural grassland
Woodland
Mountain, heath & bog
Costal, Salt & fresh water
Landuse classes



Altitude (m)
800
600
400
200
0
a
a
b
c
d
Distance from the nearest traffic node (m)
5000
4000
3000
2000
1000
0
a
b
c
d
d
Slope (degree)
20
15
10
5
0
a
b
c
c
d
PAs
Type 1 LCSIs
IPAs
Type 2 LCSIs
Un-managed

675

Table 1

Conservation management category	Median area (and range) in km^2	Total area (km^2)	Net additional cover (km^2)	Overlap with higher CMI categories	Cumulative % of England
Protected areas	0.17 (<0.01-440.9)	8957.5	8957.5	-	6.37
Type 1 Large Conservation Areas	18.91 (6.38-120.5)	2108.2	1276.0	39.5	7.36
Incentive Payment Areas	0.02 (<0.01-220.1)	22961.0	17085.6	25.6	20.45
Type 2 Large Conservation Areas	156.36 (9.25-5381.4)	112248.9	56429.2	49.7	63.71