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

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1 Abstract

2 Protected areas (PAs) are vital for conserving biodiversity, but many PA networks consist of
3 fragmented habitat patches that poorly represent species and ecosystems. One possible
4 solution is to create conservation landscapes that surround and link these PAs. This often
5 involves working with a range of landowners and agencies to develop large-scale
6 conservation initiatives (LSCIs). These initiatives are being championed by both government
7 and civil society, but we lack data on whether such landscape-level approaches overcome
8 the limitations of more traditional PA networks. Here we expand on a previous gap analysis
9 of England to explore to what extent LSCIs improve the representation of different
10 ecoregions, land-cover types and elevation zones compared to the current PA system. Our
11 results show the traditional PA system covers 6.37% of England, an addition of only 0.07%
12 since 2001, and that it is an ecologically unrepresentative network that mostly protects
13 agriculturally unproductive land. Including LSCIs in the analysis increases the land for
14 conservation more than tenfold and reduces these representation biases. However, only
15 24% of land within LSCIs is currently under conservation management, mostly funded
16 through agri-environment schemes, and limited monitoring data mean that their
17 contribution to conservation objectives is unclear. There is also a considerable spatial
18 overlap between LSCIs, which are managed by different organisations with different
19 conservation objectives. Our analysis is the first to show how Other Effective Area-Based
20 Conservation Measures (OECMs) can increase the representativeness of conservation area
21 networks, and highlights opportunities for increased collaboration between conservation
22 organisations and engagement with landowners.

23 1. Introduction

24 Terrestrial biodiversity is under unprecedented pressure, despite intensifying conservation
25 efforts. Protected areas (PAs) have long been used to mitigate these threats by separating
26 biodiversity and incompatible land uses, and now cover 14.6% of the global terrestrial realm
27 (Watson et al. 2014). Moreover, PA networks are continuing to expand, as most national
28 governments have committed to increase the proportion of their land surface under
29 conservation to 17% by 2020 (CBD, 2011). However, even with this new commitment,
30 conservation success is far from guaranteed (Venter et al., 2014). This is because PA
31 networks have often developed in an *ad hoc* manner and have three features that limit their
32 effectiveness. First, many PAs are small and isolated, and so cannot maintain broad-scale
33 ecological processes or sustain viable populations of wide-ranging species (Armsworth et al.,
34 2011). Second, PAs are often placed in remote areas with little economic potential (Joppa
35 and Pfaff, 2009), leaving many ecosystems and species poorly represented (e.g. Iojă et al.,
36 2010; Jackson and Gaston, 2008). Third, PAs fix conservation efforts in space based on
37 conditions at a certain time, while ecosystems and their threats are dynamic (e.g. Araújo et
38 al., 2011).

39

40 These problems are evident in England, where much biodiversity is restricted to small,
41 privately owned fragments of semi-natural habitats. Most of these habitats have been
42 shaped over thousands of years by anthropogenic use and management, but have suffered
43 significant fragmentation and degradation in the last century (Lawton et al., 2010). The
44 English PA network is based on a restrictive zoning approach (Lawton et al., 2010), which
45 uses planning legislation to identify National Natural Reserves (NNRs) and Sites of Special
46 Scientific Interest (SSSIs) and then limit damaging development within them. Historically,

47 this network has comprised of mostly small ($< 1\text{km}^2$) and isolated PAs (the median size of
48 SSSIs and NNRs are 0.2 km^2 and 1.1 km^2 respectively), typically confined to uplands and
49 ecoregions with low agricultural potential (Oldfield et al., 2004). To overcome these
50 limitations, the United Kingdom (UK) has adopted a complementary approach based on
51 agri-environment schemes and other incentive-based payment schemes. These pay
52 landowners for income foregone and to cover the costs of management actions designed to
53 improve landscape quality for conservation or other objectives, thereby providing an
54 important source of funding for conservation inside and outside PAs. In England, the
55 European Union's Common Agricultural Policy has funded agri-environment schemes since
56 1987 (Bright et al., 2015). Until recently, these schemes included Higher-Level Stewardship
57 (HLS), which supported intensive habitat maintenance and restoration within target areas in
58 production landscapes (Natural England, 2012), and English Woodland Grants that funded
59 projects to restore and manage woodlands (Raum and Potter, 2015). Both of these were
60 replaced in 2016 by the new Countryside Stewardship scheme (Natural England, 2015) and
61 the UK's departure from the European Union could bring further changes.

62

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

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

74

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

84

85 It was in this context that a recent project explored large-scale conservation initiatives
86 (LSCIs) in England, Scotland and Wales, where LSCIs were defined as any area larger than an
87 arbitrary threshold of 10 km² that is actively managed for biodiversity conservation goals
88 (Eigenbrod et al., 2017). This research looked at the different categories and locations of
89 LSCIs, the factors involved in their planning and management, and their environmental
90 benefits (Adams et al., 2016; Eigenbrod et al., 2017; Macgregor et al., 2012). This analysis
91 identified over 800 LSCIs in England, Scotland and Wales, which were subsequently
92 categorised based on land tenure and management strategy (Macgregor et al., 2012). This
93 large number of LSCIs highlights the growing interest in the approach in the UK. However,
94 despite their number and appeal, there is little evidence on whether these new initiatives

95 have resulted in a more representative PA network. The aim of this paper is thus to explore
96 the extent to which LSCIs and agri-environment schemes have complemented the current
97 network of PAs to reduce spatial biases.

98

99 The best way to explore this question is to undertake a gap analysis, a spatially resolved
100 quantitative approach for measuring how well PA networks represent biodiversity and
101 protect different biogeographic zones, land-cover types and species (e.g. Jenkins et al.,
102 2015; Scott et al., 1993). Here we conduct the first ever gap analysis of the relative
103 contribution of PA, LSCIs and agri-environment schemes, focusing on these different
104 conservation area types in England. We begin by measuring how England's PA network has
105 changed since a 2001 gap analysis in terms of extent and protecting different ecoregions
106 and elevation zones (Oldfield et al. 2004). We then assess the contribution of two other
107 major categories of conservation management initiatives: large-scale conservation
108 initiatives (LSCIs), using the recently created LSCI database (Eigenbrod et al., 2017), and;
109 incentive payment areas (IPAs) based on agri-environment and woodland improvement
110 schemes. This involves measuring the overlap in the PA, LSCI and IPA networks, and the
111 extent to which land under these management types cover the different ecoregions, land-
112 cover types and elevation zones. In doing so, we test the hypothesis that Other Effective
113 Area-Based Conservation Measures (OECMs), as highlighted in the Convention for Biological
114 Diversity's Aichi target 11 (CBD 2011), reduce some of the limitations of the original PA
115 network by better representing England's ecoregions and land with higher socio-economic
116 value.

117

118 **2. Methods**

119 *2.1. Types of conservation areas*

120 We distinguished four categories of conservation areas in our analysis:

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

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

150

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

161

162 *2.2. Data collection and preparation*

163 We used data held by Natural England on NNRs, SSSIs, Type 1 LSCIs and Type 2 LSCIs in
164 2013, as well as IPAs as of December 2013. Information was extracted from the existing
165 database (Eigenbrod et al., 2017), for the 341 LSCIs that are found in England, have defined
166 boundaries and meet the Type 1 or Type 2 criteria. We then used the Land Cover Map 2007

167 to exclude urban areas from each LSCI. The IPA boundaries were from maps of land holdings
168 with HLS and woodland grant scheme agreements. We only considered those stewardship
169 options which contribute to conservation. Where farm agreements contained at least one
170 whole-farm option, we considered the entire farm as an IPA. If this was not the case, we
171 used the HLS data to map the IPA land parcels (see Text S1 for further details). We clipped
172 all of these datasets with the England political boundary to exclude any estuarine or marine
173 areas (following Oldfield et al., 2004).

174

175 To determine the characteristics of the different conservation management categories, we
176 used datasets describing elevation, slope, distance to infrastructure, ecoregion type,
177 agricultural land quality and land-cover class. All of these data types were used in previous
178 gap analyses to measure the representativeness of PA networks and the extent to which
179 PAs are found in remote areas on land with low agricultural potential (e.g. Oldfield et al.,
180 2004; Pressey and Tully, 1994). We did not use the available species distribution data
181 because much of it has a spatial resolution of 10 km x 10 km, which is a great deal coarser
182 than the majority of the PAs and agri-environment scheme land parcels, making it
183 impossible to measure levels of species representation with precision.

184

185 The first step in the analysis was to produce six GIS layers derived from five spatial datasets,
186 which were resampled to produce GIS layers with the same resolution of 80 m (matching
187 the dataset with the coarsest resolution). Three of the layers described physical factors. We
188 used the SRTM Digital Elevation Model (DEM) to produce the elevation zone layers (Table
189 S2), where each elevation value was assigned to one of the following four classes: 0 to 200
190 m; 201 to 400 m; 401 to 600 m and > 600 m. We also used this DEM to produce the slope

191 layers using the Slope function in ArcGIS (ESRI 2011; ArcGIS Desktop: Release 10. Redlands,
192 CA). To produce the remoteness layer we used national data on public transport
193 infrastructure (Table S2) and calculated distance from nearest transport node points (e.g.,
194 bus stops and train stations).

195

196 Another three layers described ecological and environmental factors. For ecoregions we
197 used the National Character Areas (NCA) layer produced by Natural England (Table S2). The
198 NCA layer subdivides England so each of the 159 NCAs (which we term “ecoregions”
199 hereafter) represent a unique combination of landscape, biodiversity, geodiversity, cultural
200 and economic activity. We also used the Provisional Agricultural Land Classification (Table
201 S2) dataset, which divides England into five categories of agricultural land (with grade 1
202 representing the highest and grade 5 the lowest respectively) and two additional categories
203 of land in non-agricultural use (i.e. non-arable and suburban). We used the Land Cover Map
204 2007 (Table S2), derived from satellite imagery, to produce the land-cover layer by
205 reclassifying the original 23 land-cover types into seven: (i) coastal, salt and freshwater; (ii)
206 mountains, heath and bog; (iii) woodland; (iv) semi-natural grassland; (v) arable; (vi)
207 suburban, and; (vii) urban.

208

209 *2.3. Data analysis*

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

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

216

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

233

234 We conducted a gap analysis to assess the extent to which the different conservation
235 management category networks represent surrogates associated with biogeographic
236 differences in biodiversity. This involved calculating the percentage of each ecoregion,

237 elevation zone, agricultural land quality class and land-cover class under each conservation
238 management category, based on data extracted using the Tabulate Area function in ArcGIS.

239

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

248

249 3. Results

250 *3.1. Temporal changes in PA coverage*

251 The 4335 nationally designated terrestrial PAs (NNRs & SSSIs) cover 6.37% (8,322.4 km²) of
252 England's land surface (Figure 1), representing an increase of 83.6 km² (0.07%) since 2001
253 (Table 1). The increase had little impact on the median area of individual PAs, which at 0.17
254 km² is similar to that in 2001 (Oldfield et al., 2004). This is because 82% of the 4111 SSSIs
255 and 46% of the 224 NNRs are smaller than 1 km². Many ecoregions are still poorly
256 represented, with 78% of the 159 ecoregions having < 10% of their area protected by PAs
257 (Figure 2a). Similarly, the percentage of PAs within the 0-200m elevation zone (Figure 3a),
258 which represents 87% of England's terrestrial area, remains unchanged since the 2001
259 analysis at 3.5%, showing a consistent spatial bias in PAs towards upland areas.

260

261 *3.2. Extent and overlap between the different conservation management categories*

262 Land under LSCIs and IPAs is much larger than the land dedicated to formal PAs (Figure 1).
263 Adding the large privately owned Type 1 LSCIs expands the net coverage of England by only
264 1%, because they cover < 1% of England's land surface and 37.9% of their area is already
265 protected by PAs (Table 1). However, adding the IPAs nearly triples the land under
266 conservation management from roughly 9,000 to 23,000 km², increasing coverage to 20.5%.
267 Adding Type 2 LSCIs, which are managed by multiple different organisations or individuals,
268 further increases this coverage to nearly 64% of England's terrestrial surface (Figure 1, Table
269 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
270 proposed to be managed for biodiversity conservation (Figure 1).

271

272 *3.3. Characteristics of the different conservation management categories*

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

285

286 4. Discussion

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

306

307 4.1 Protected area coverage

308 A key step in improving the representativeness of any PA network is undertaking a gap
309 analysis to identify species, habitats and ecoregions needing further protection. Such
310 analyses should be undertaken periodically to evaluate progress (Margules and Pressey,
311 2000; Pressey et al., 2013). Our study adopts this approach by repeating a gap analysis for
312 England undertaken over a decade ago (Oldfield et al., 2004). In England there are two main
313 types of PA established for biodiversity conservation, namely NNRs and SSSIs. These covered
314 6.3% of England's land surface over a decade ago (Oldfield et al., 2004) and our results show
315 how little this has changed, with only a marginal increase. The mean size of these PAs also
316 remains small, although the maximum size has risen from 160 km² to 440 km², reflecting the
317 success of several initiatives to join up existing areas.

318

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

331

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

343

344 *4.2 The role of Large Conservation Areas*

345 The LSCI approach is seen by many as one of the most effective ways of achieving the
346 required change in conservation efforts, to meet both national and international obligations
347 (CBD 2011; Macgregor et al., 2012). We investigated the current role of LSCIs by dividing
348 them into two groups based on tenure and level of management for conservation
349 objectives. Type 1 LSCIs are owned and managed primarily for conservation by one or a few
350 landowners and are often based on several existing NNRs and SSSIs. There are relatively few
351 of these LSCIs and nearly half of them have PA status, which explains why adding them to
352 the gap analysis made little difference to the area dedicated for conservation or the spatial
353 bias in the area conserved. This is probably because the mechanism for establishing such
354 LSCIs is similar to the creation of large PAs, involving considerable land acquisition costs
355 (Naidoo et al., 2006). Once established, management costs per unit area decline as PA size

356 increases (Armsworth et al., 2011; Ausden and Hiron, 2002), suggesting Type 1 LSCIs have
357 financial as well as ecological benefits when compared to a set of smaller PAs. However,
358 creating such LSCIs requires the availability of large blocks of existing conservation land, or
359 willingness on the part of adjacent landowners to sell or lease their land for conservation,
360 which is unlikely on high-quality agricultural land (Adams et al., 2014; Knight et al., 2010).

361

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

378

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

390

391 5. Conservation implications

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

402 could help inform every country seeking to implement LSCIs as a way of scaling-up their
403 conservation efforts and achieving their international commitments (CBD 2011).

404

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

419

420 Planning and management of Type 2 LSCIs is similarly challenging, since they typically
421 encompass a large number of individual land holdings and land owners, and our results
422 show most of the land is not managed specifically for achieving conservation objectives.
423 There is also a considerable temporal and spatial overlap between different LSCIs, with each
424 overlapping project being overseen by different configurations of NGOs, government
425 agencies and partnerships (Eigenbrod et al., 2017), but often with distinct conservation

426 objectives. Thus, the conservation benefits of these schemes depend on integrating a
427 multitude of stakeholder values and policies to prioritise and implement conservation action
428 that complements the existing PA network (Adams et al., 2016). These complexities suggest
429 a target-based spatial conservation prioritisation approach would be helpful, based on
430 existing empirical data and expert knowledge, as such systems are designed to guide the
431 prioritisation of conservation efforts, and to help understand and balance associated trade-
432 offs (Carwardine et al., 2009; Metcalfe et al., 2015).

433

434 Such an analysis could usefully follow a two-tiered approach: a national-scale spatial
435 conservation prioritisation to identify broad focal landscapes, followed by fine-scale
436 analyses within each of these landscapes to identify when and how conservation action
437 should be implemented. The second tier would involve local partnerships determining the
438 best approach to take within these priority landscapes and the specific areas to focus on,
439 based on local data and knowledge of opportunities and constraints (Smith et al., 2009).

440 There are considerable benefits, in terms of building financial, human and intutional capital,
441 of adopting a systematic conservation planning approach at the landscape and LSCI level
442 (Bottrill et al., 2012). This approach could be used to develop more detailed conservation
443 goals, increase collaboration between individuals and organisations and so identify options
444 for reducing overlap and costs. This would help to ensure that nationally important
445 biodiversity was protected, but in a way that would maximise local buy-in and likelihood of
446 implementation.

447

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631 **Figures and Tables legend**

632

633 Table 1:

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

640

641 Figure 1:

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

647

648

649 Figure 2:

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

654

655 Figure 3:

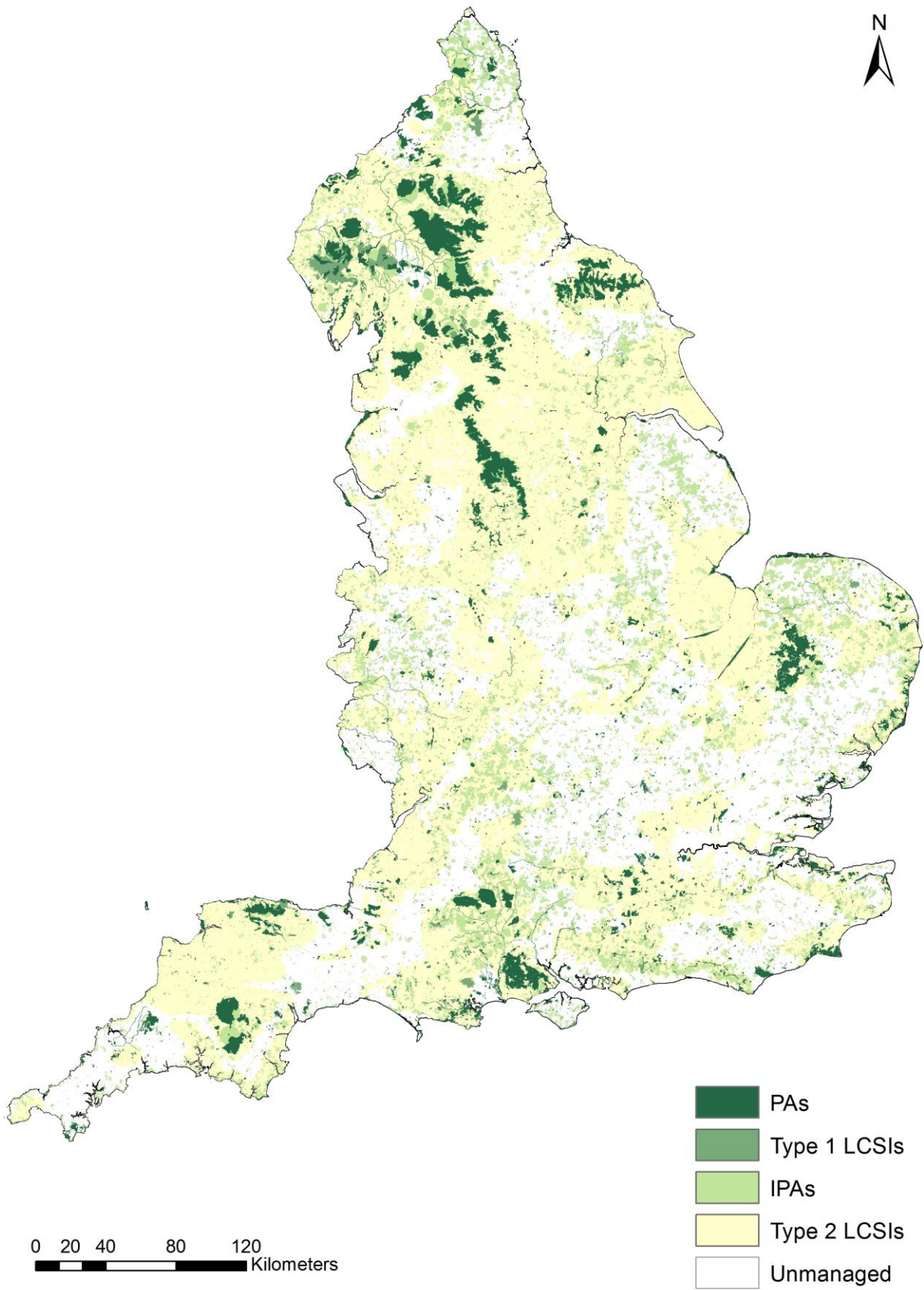
656 Details of the different conservation management categories by: (a) elevation classes, (b)
657 agricultural land quality; and (c) landcover class. These categories are protected areas (PAs),
658 Type 1 and Type 2 Large-scale Conservation Initiatives (LSCIs) and Incentive Payment Areas
659 (IPAs).

660

661 Figure 4:

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

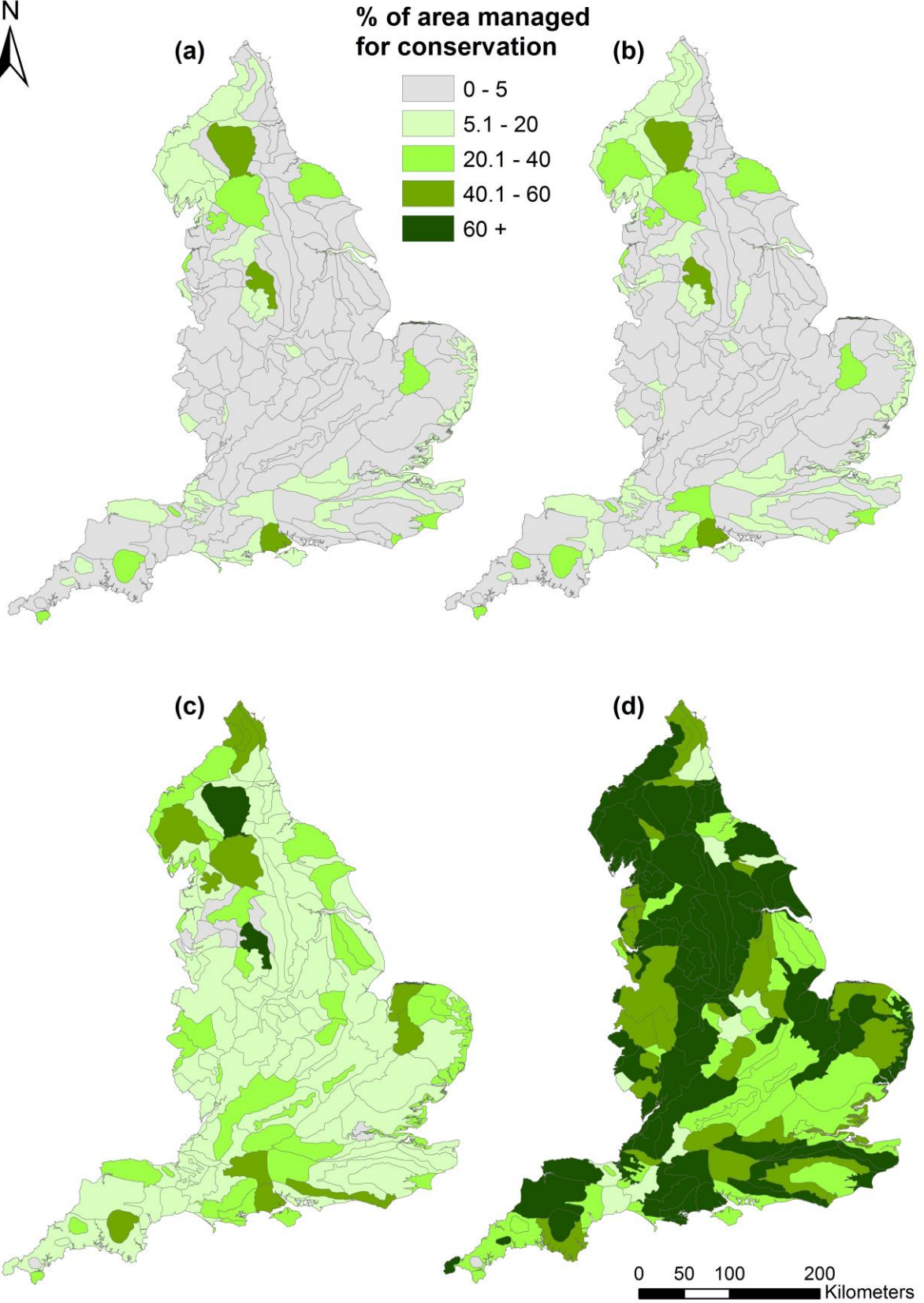
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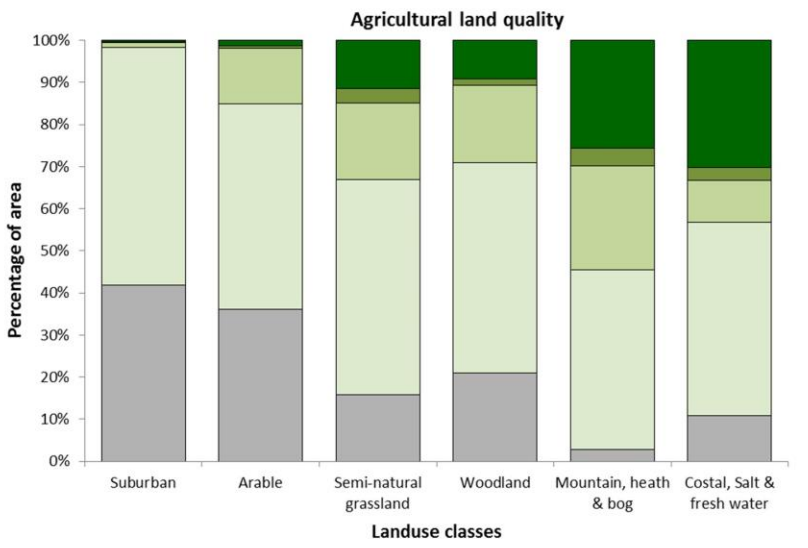
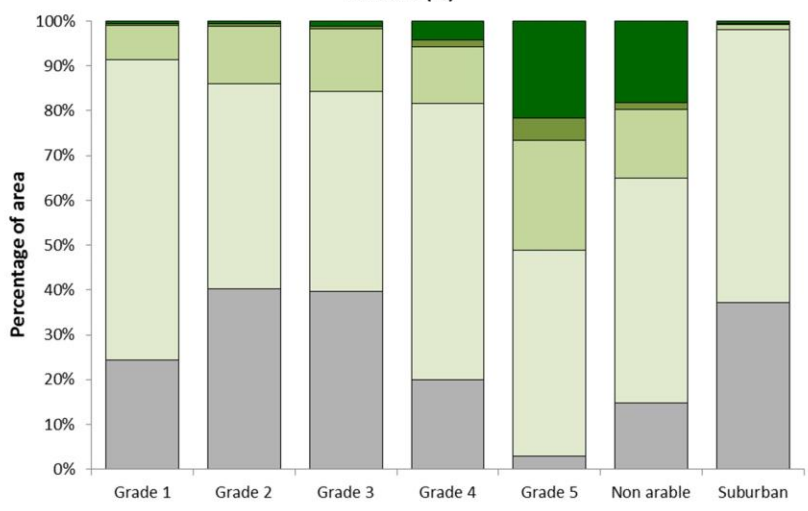
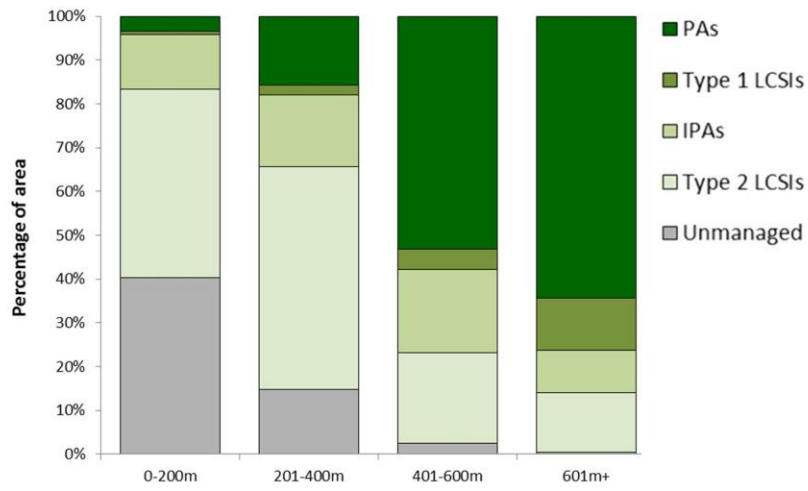
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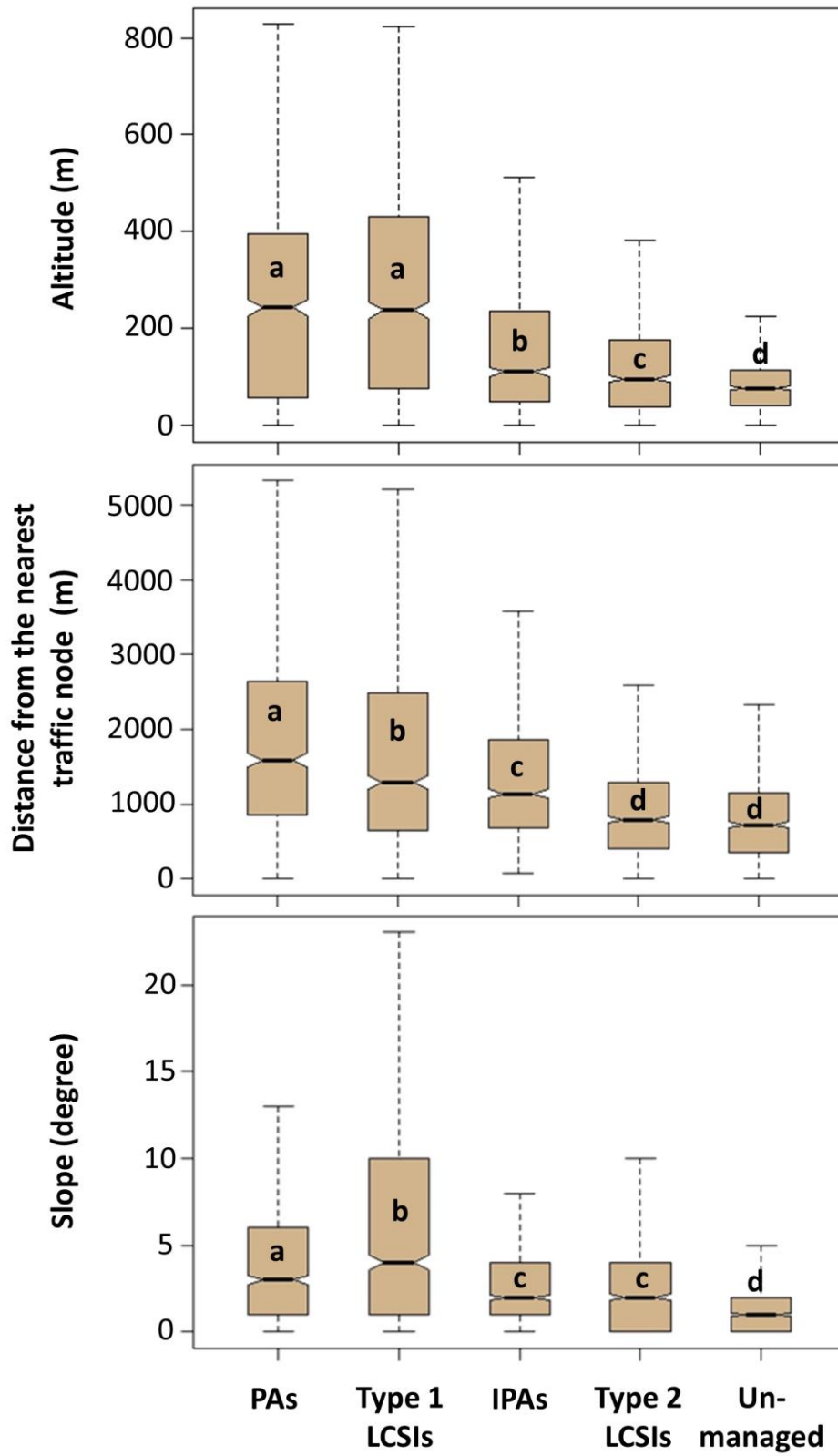


Table 1

Conservation management category	Median area (and range) in km²	Total area (km²)	Net additional cover (km²)	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