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## Reconciling diverse viewpoints within systematic conservation planning

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Keywords:	conservation viewpoints, inclusiveness, pluralism, biodiversity, ecosystem services, consensus planning, systematic conservation planning, spatial prioritisation
Abstract:	<p>1. Conservation encompasses numerous alternative viewpoints on what to value (features such as biodiversity, ecosystem services or socio-economic benefits) and how to convert these values into conservation policies that deliver for nature and people. Reconciling these differing values and viewpoints in policy development and implementation is a perennial challenge.</p> <p>2. Balancing differing stakeholder viewpoints within a single conservation plan risks some viewpoints overshadowing others. This can occur as some dominant viewpoints may lead to more marginal views being suppressed, and also through social biases during the planning process.</p> <p>3. Here we develop four separate 'caricature' conservation viewpoints, and spatially quantify each of them in order to test different approaches to equitable reconciliation. Each viewpoint prioritises different locations, dependent on the extent to which they deliver a variety of different biodiversity, well-being and economic goals.</p> <p>4. We then show how these different viewpoints can be reconciled using numeric methods. We find that a pluralist approach, which accounts for the spatial similarities and differences between viewpoints, is able to deliver equitably for multiple conservation features. This pluralist approach provides a coherent spatial conservation strategy with the capacity to satisfy advocates of quite divergent approaches to conservation.</p>

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## 1 **Abstract**

- 2 1. Conservation encompasses numerous alternative viewpoints on what to value (features such  
3 as biodiversity, ecosystem services or socio-economic benefits) and how to convert these  
4 values into conservation policies that deliver for nature and people. Reconciling these  
5 differing values and viewpoints in policy development and implementation is a perennial  
6 challenge.
- 7 2. Balancing differing stakeholder viewpoints within a single conservation plan risks some  
8 viewpoints overshadowing others. This can occur as some dominant viewpoints may lead to  
9 more marginal views being suppressed, and also through social biases during the planning  
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- 11 3. Here we develop four separate 'caricature' conservation viewpoints, and spatially quantify  
12 each of them in order to test different approaches to equitable reconciliation. Each viewpoint  
13 prioritises different locations, dependent on the extent to which they deliver a variety of  
14 different biodiversity, well-being and economic goals.
- 15 4. We then show how these different viewpoints can be reconciled using numeric methods. We  
16 find that a pluralist approach, which accounts for the spatial similarities and differences  
17 between viewpoints, is able to deliver equitably for multiple conservation features. This  
18 pluralist approach provides a coherent spatial conservation strategy with the capacity to  
19 satisfy advocates of quite divergent approaches to conservation.

20

## 1 INTRODUCTION

2 Each individual worldview (here 'viewpoint') in biodiversity conservation  
3 encompasses its own set of specific values and conservation priorities (Sandbrook *et al.*,  
4 2019; Bhola *et al.*, 2021; Anderson *et al.*, 2022). As in other applied disciplines, different  
5 viewpoints can often seem contradictory and even irreconcilable in planning decisions  
6 (Matulis and Moyer, 2017), and hence there are trade-offs when seeking to reconcile  
7 different values during policy implementation (McShane *et al.*, 2011). This diversity of  
8 opinions must be addressed in planning so as to avoid socio-environmental conflicts,  
9 perceived injustices and hence ineffective policy interventions (Barton *et al.*, 2022). How to  
10 combine different values through an equitable process that ensures all viewpoints are fairly  
11 represented, but that nonetheless garners widespread support, is a pressing question for  
12 conservation and more widely in society (Termansen *et al.*, 2022).

13 Two different types of approach are commonly used to unify opposing viewpoints  
14 within conservation. Inclusive approaches seek to accommodate all viewpoints by building  
15 consensus and finding compromise between people holding different views, thus creating a  
16 single voice for conservation that carries greater weight (Tallis and Lubchenco, 2014).  
17 However, by not explicitly acknowledging differences, inclusivity could stifle debate and  
18 reinforce current dominant viewpoints if 'consensus' represents the viewpoint that is most  
19 frequently articulated and voted for (Matulis and Moyer, 2017). Pluralist approaches, by  
20 contrast, assert that we need to find better ways to accept and engage with diverse  
21 viewpoints on biodiversity thereby giving voice to marginalised values and increasing equity  
22 (Pascual *et al.*, 2021). The risk is that a pluralist approach results in a divided, and thereby  
23 potentially unconvincing, voice for conservation that makes systematic and coordinated  
24 action impossible (Matulis and Moyer, 2017).

25 Viewpoints within the conservation community are often considered in terms of  
26 'traditional' or 'new' conservation (Matulis and Moyer, 2017). 'Traditional' conservation  
27 follows an ecocentric viewpoint, conserving species diversity and natural habitats for their  
28 intrinsic value (Soulé, 1985; Taylor *et al.*, 2020). It is often regarded as the antithesis of 'new'  
29 conservation, which follows a more anthropocentric viewpoint motivated by achieving  
30 conservation action through attaining economic and social benefit (Marvier, 2014). However,  
31 this is a simplification of the diverse range of views on approaches to conservation. The  
32 Future of Conservation survey (<http://futureconservation.org>) sought to establish a  
33 framework to further categorise different viewpoints within conservation  
34 (<https://www.futureconservation.org/about-the-debate>) but in reality the views of  
35 conservation researchers and practitioners are spread over a continuum between and  
36 beyond these viewpoint groupings, with no clear 'camps' (Sandbrook *et al.*, 2019), making it  
37 difficult to evaluate potential approaches against each other (Hunter Jr, Redford and  
38 Lindenmayer, 2014). It is not this work's aim to revisit debate about the relative merit of any  
39 conservation viewpoint. Rather, it accepts that there exists a breadth of viewpoints that need  
40 to be reconciled during conservation policy development, whilst recognising that  
41 conservation is likely to be more successful if focused on common ground within the  
42 conservation community (Hunter Jr, Redford and Lindenmayer, 2014).

43 The important step of identifying and balancing stakeholder viewpoints is typically  
44 undertaken at the beginning of the planning process in an attempt to agree weightings or  
45 goals for different priorities. One way of addressing these issues in a structured way is  
46 through using a planning framework such as systematic conservation planning (SCP) (Pressey  
47 and Bottrill, 2009; Watson *et al.*, 2011), which utilises network-scale and spatially explicit  
48 methods to inform important conservation planning decisions (Watson *et al.*, 2011). SCP  
49 provides a way to incorporate these techniques in a robust and auditable process,  
50 incorporating the principle of complementarity to design an optimal network for a given set

51 of planning objectives (Margules and Pressey, 2000; Wilson, Cabeza and Klein, 2009). As part  
52 of a full SCP implementation there is opportunity to build consensus between differing  
53 stakeholders' viewpoints within the initial planning stages, but this is carried out primarily  
54 when conservation goals are being set rather than the prioritisation stage. However it is  
55 difficult to understand, at this early stage, whether different values translate into similar or  
56 distinct spatial priorities, potentially generating considerable debate about issues that have  
57 no practical impact.

58 An additional difficulty in building combined conservation plans is that social biases  
59 need to be accounted for in order to equitably reconcile viewpoints. Decision support tools  
60 can be used to facilitate decision-making between stakeholders, preferably using structured  
61 methods such as multi-criteria decision analysis (MCDA), which assesses performance of  
62 alternative solutions across criteria, and explores trade-offs (Davies, Bryce and Redpath,  
63 2013; Esmail and Geneletti, 2018); or the Delphi technique, which iteratively and  
64 anonymously surveys a panel of experts or stakeholders (Mukherjee *et al.*, 2015, 2018).  
65 Although these tools provide powerful methods to inform decision-making through  
66 reconciling different viewpoints, there are many social biases such as group-think and  
67 dominance effects that cannot be overcome completely (Mukherjee *et al.*, 2018), and hence  
68 consensus on conservation action attained may risk not providing an equitably integrated  
69 solution. Integrating differing viewpoints on how to carry out area-based conservation is vital  
70 in implementing a coherent and representative conservation framework (Bhola *et al.*, 2021),  
71 and it is important to minimise these social biases when building consensus conservation  
72 plans.

73 Spatial prioritisation methods provide a tool to evaluate potential spatial synergies  
74 and trade-offs between different conservation goals and is often an important stage of SCP.  
75 Although typically focused on improving representativeness of species distributions within  
76 area-based conservation measures, spatial prioritisation can also be used to investigate the

77 effect of including different socio-economic and ecosystem service (ES) information on  
78 spatial priorities to inform conservation policy (Naidoo *et al.*, 2008). In this way, trade-offs  
79 between protecting biodiversity and different societal or policy objectives can be assessed;  
80 such as carbon storage (Thomas *et al.*, 2013; Soto-Navarro *et al.*, 2020), or other ecosystem  
81 services and land use simultaneously (Anderson *et al.*, 2009; Moilanen *et al.*, 2011; Fastré *et*  
82 *al.*, 2020). Hence, although not previously used for this purpose, spatial prioritisation  
83 provides a potentially powerful quantitative tool to support spatially integrating different  
84 viewpoints in conservation. Here we implement spatial prioritisation combined with numeric  
85 aggregation methods, which avoids potential social biases, in order to fairly test different  
86 approaches to viewpoint integration that combine caricature viewpoints into single  
87 solutions.

88         Using the biological, environmental, social, and economic landscape conditions of  
89 Great Britain, we implement four caricature conservation viewpoint prioritisations  
90 ('traditional', 'new', 'local social instrumentalism', and 'international market ecocentrism';  
91 Table 1) at a national scale to illustrate the diverse range of perspectives within the  
92 conservation community. We assign weights to species distributions and other resources  
93 corresponding with the values of each viewpoint, and then carry out spatial prioritisation at  
94 a 10x10 km ('landscape') resolution for Britain. We expect prioritisations for each viewpoint  
95 to perform well at covering resource types (conservation feature layers) that are highly  
96 valued within that viewpoint, but they may overlook other features. For example, 'non-  
97 traditional' methods may perform relatively poorly in representing species distributions.  
98 Finally, we develop both 'inclusive' and 'pluralist' approaches to evaluate the extent to which  
99 it is possible to reconcile and integrate the four viewpoints into a collaborative and coherent  
100 conservation plan.

101



102 **METHODS**

103

104 **Feature layers**

105           We searched for and collated social, economic and ecological spatial data that: (i)  
106 was publicly available for the entirety of Great Britain (GB), (ii) had a resolution of 10x10km  
107 scale or finer, and (iii) could be used to create informative ecosystem service (ES) or socio-  
108 environmental value layers. After the data search, a total of seven non-biological layers were  
109 found to be suitable and are detailed below. We defined the study area as GB, excluding  
110 islands smaller than 20km<sup>2</sup>.

111           Five ES layers were adapted from published, publicly available resources: (i) carbon  
112 storage (Bradley *et al.*, 2005; Henrys, Keith and Wood, 2016), (ii) agricultural/land value  
113 (urban areas were assigned the highest 'agricultural value', indicating locations unsuitable  
114 for terrestrial conservation) (Natural Resources Wales, 2019; The James Hutton Institute,  
115 2019; Natural England, 2020), (iii) recreational services (Schägner *et al.*, 2016), (iv) flood  
116 regulation (Stürck, Poortinga and Verburg, 2014), and (v) pollination services (Schulp,  
117 Lautenbach and Verburg, 2014).

118           In addition, two socio-environmental value layers were included: (vi) wilderness  
119 (Kuiters *et al.*, 2013) and (vii) landscape aesthetic value (Van Zanten *et al.*, 2016). Full details  
120 of calculation, and data sources, of ES and socio-environmental value layers are provided in  
121 Supplementary Methods. All feature layers were rescaled to allow for direct comparison, and  
122 aggregated to 10x10 km (henceforth 'landscape') resolution for the analysis (Figure 1). Only  
123 landscapes with >50% land cover were considered

124           To incorporate biodiversity value, we included the interpolated distributions of 445  
125 priority species with distribution data available listed under Section 41 (Natural Environment  
126 and Rural Communities Act, 2006). Although the species that constitute 'priorities' may differ

127 between viewpoints, here we use the same species to allow for direct comparisons of  
128 different viewpoint prioritisation performance. Distribution data were provided by Butterfly  
129 Conservation (BC), Biological Records Centre (BRC) and breeding bird distributions by British  
130 Trust for Ornithology (BTO). Data were in the form of annual records between 2000 and 2014,  
131 except for two taxa where atlas data were only available for specific time periods (birds  
132 [2007-11] (Gillings *et al.*, 2019), and vascular plants [2010-2017]). We used the raw  
133 distribution records for 156 species that were very localised ( $\leq 10$  presence records) and for  
134 a further 77 species which could not be modelled (most of which were also very rare, and for  
135 which models did not converge). For the remaining 212 species with over 10 presence  
136 records, we interpolated their range using Integrated Nested Laplace Approximations (INLA)  
137 in the *inlabru* package (Bachl *et al.*, 2019). We used a joint model predicting distribution while  
138 accounting for recording effort (see Supplementary Methods for full details).

139

#### 140 **Viewpoint prioritisation**

141

142 Four conservation viewpoint caricatures were created that included ‘traditional’  
143 conservation (TRAD), ‘new’ conservation (NEW), ‘international market ecocentrism’ (ECON),  
144 and ‘local social instrumentalism’ (SOC) (see Table 1 for definitions). Viewpoints were  
145 constructed by varying the weightings of different feature layers (such as biodiversity, carbon  
146 or landscape aesthetics), indicating their relative importance, as this allows quantification of  
147 trade-offs when trying to reconcile viewpoints (Table 2). Weightings were not based upon  
148 wider consultation, and it must be emphasised that they are not designed to be accurate  
149 representations of the viewpoints of any group of conservationists. Instead, they capture an  
150 illustrative range of perspectives from across the conservation community. For real world  
151 implementation, viewpoint weightings could be developed with stakeholders and the public  
152 either through questionnaires and interviews to identify different individual viewpoints, or

153 through workshops or forums to collect the viewpoint of a particular stakeholder group or  
154 organisation.

155 In order to identify the highest priority areas for each viewpoint, we carried out a  
156 spatial prioritisation using the software Zonation (Moilanen, 2007) which produces a  
157 complementarity-based ranking of conservation priority over the study area. As it is  
158 important in joint species and ES prioritisations to ensure localised species are not  
159 overlooked (Thomas *et al.*, 2013), we used 'core area zonation' (landscape value based upon  
160 the single highest value feature) to ensure complementarity was incorporated. Although we  
161 present 'core area zonation' prioritisations in the main text, we also tested viewpoint  
162 prioritisations and integration approaches using the alternative 'additive benefit function'  
163 prioritisation algorithm (landscape value summed across all weighted features) within  
164 Zonation. The results from these analyses were qualitatively similar and we do not consider  
165 them further in the main text, although they are reported and discussed within the  
166 Supplementary Materials.

167 We incorporated ES, biodiversity and socio-environmental values into the viewpoint  
168 prioritisations through weightings commensurate with each viewpoint (Table 2). Weights for  
169 feature layers were generally positive, representing a desirable resource to include, with the  
170 exception of agricultural value (negative weights), which represented an alternative land use  
171 to conservation. Species distributions were collectively considered a single biodiversity  
172 feature layer for weightings, so that each species received a weighting corresponding to  
173  $(\text{biodiversity weighting})/(\text{number of species})$ , but were included as separate feature layers  
174 within the prioritisation.

175 We considered each prioritisation individually and tested feature coverage for the  
176 top 5%, 10%, 17% [corresponding to the Aichi 2020 target (CBD, 2010)] and 30% priority areas  
177 [corresponding to the first draft of the post-2020 global biodiversity framework (CBD, 2021)].

178 Coverage of biodiversity was calculated as the mean species distribution proportion  
179 coverage. The distribution of each ES feature is likely to have a large effect on prioritisation  
180 ranks for each viewpoint, as the more concentrated a feature is, the larger its effect on the  
181 prioritisation. Here we rescaled each feature but did not normalise the distribution, doing so  
182 would ensure each feature had an equal effect on prioritisations, but may mean return on  
183 coverage would be artificially inflated.

184 We also investigated the similarity of existing protected area (PA) coverage in Britain  
185 to the different viewpoint priority rankings of each landscape. We expected the existing PA  
186 network to match the 'traditional' viewpoint prioritisation most closely since the designation  
187 rationale for protected areas is typically to prioritise species and ecosystems  
188 representatively. We considered all Sites of Special Scientific Interest (SSSI) and National  
189 Nature Reserves (NNR) (<https://naturalengland-defra.opendata.arcgis.com>) as 'protected  
190 areas' (Supplementary Figure 1).

191

### 192 **Viewpoint integration**

193 Given that decision support tools risk being influenced by social biases of  
194 participants, we developed two novel numerical aggregation approaches to reconcile the  
195 individual viewpoints into single spatial conservation plans; one representing an inclusive  
196 approach and one a pluralist approach (Table 1). It is important to note that we tested these  
197 integration approaches using simulated caricature viewpoints, whereas real-world  
198 applications are likely to present additional complexities (see Discussion).

199 The inclusive approach produced an aggregate priority map by taking the individual  
200 viewpoint prioritisations (Figure 2), and summing the landscape priority ranks of each  
201 viewpoint (Eq. 1). This represents an integrated conservation solution generated through a  
202 vote counting method with equal weight given to each viewpoint.

203

204 
$$I_j = \sum_v r_{vj}$$

205 Eq. 1

206 where  $I_j$  is the inclusive value  $I$  for landscape  $j$ ,  $r_{vj}$  is the priority rank for viewpoint  $v$  and  
207 landscape  $j$ .

208

209 However, as there are correlations between viewpoints in their weighting of  
210 individual feature layers, inclusive vote counting methods may result in combined priority  
211 areas that are simply shared by more similar viewpoints, and therefore under-represent the  
212 level of importance of other features valued by more distinctive viewpoints. Hence, we also  
213 implemented a more equitable pluralist approach to integration accounting for correlation  
214 between feature layer choice (Table 2), weighting by the distinctiveness of each viewpoint,  
215 to ensure more marginal viewpoints were not overlooked.

216 For the pluralist approach we initially undertook a principle component analysis  
217 (PCA) to partition the variance from viewpoint weightings of feature layers (Table 2) into  
218 principal components (PC) (Supplementary Table 1). These PCs are linear combinations  
219 (eigenvectors) of the viewpoints, which were then used to weight the viewpoint landscape  
220 priority ranks (Eq. 2) to calculate the pluralist landscape value. The first PC is fitted in the  
221 direction that accounts for the maximum variance of the viewpoint weightings and further  
222 PCs, orthogonal to the previous PCs, maximise the remaining variance. Thus PCs are the  
223 combinations of viewpoints that explain the variance in weightings in the most efficient way.  
224 For each landscape, we multiplied the four viewpoint prioritisation rankings by the associated  
225 viewpoint eigenvectors of the first PC and took the sum (dot product) and iteratively added  
226 viewpoint rank/PC dot product absolute values until the 'main' PC of each viewpoint was  
227 included (Eq. 2), to ensure the distinctiveness of each viewpoint was represented.

$$P_j = \sum_{c=1}^{n_c} |r_{j1 \times v} \cdot W_{c_{v \times 1}}|$$

229 Eq. 2

230 where  $P_j$  is the pluralist value  $P$  for landscape  $j$ ,  $r_j$  is a  $1 \times v$  matrix of viewpoint priority ranks  
231 for landscape  $j$ , and  $W_c$  is the corresponding  $v \times 1$  eigenvector matrix from principal  
232 component  $c$  of a viewpoint feature layer weightings PCA (Supplementary Table 1).  $n_c$  is the  
233 smallest number of principal components where the highest PC loading for each viewpoint  
234 can be included (i.e. for all viewpoints we found the PC with the highest loading for that  
235 viewpoint, and included all PCs up to and including that viewpoint).

236 We evaluated the *efficiency* with which feature layers were included (or for  
237 agricultural value, excluded) within each individual prioritisation (Fig. 2) and aggregation  
238 approach (Fig. 3). Efficiency was calculated as the proportion of each feature included (or for  
239 agricultural value, excluded) by prioritisations at each coverage threshold, compared to the  
240 maximum amount possible. The mean efficiency across feature layers was used as a measure  
241 of the overall success of a given approach. For each approach, we also highlighted the feature  
242 layer that had the lowest coverage, to represent a measure of how equally feature layers  
243 were included (i.e., how ‘disappointed’ would someone be for whom this was their top  
244 priority?).

245 In addition to the inclusive and pluralist approaches listed in the main text, we also  
246 tested two other approaches to integrating viewpoints. Both performed less well under ‘core  
247 area zonation’ for most thresholds, and so are not discussed further here. See Supplementary  
248 Methods, Figures and Discussion for further details on these other approaches (and ‘additive  
249 benefit function’ prioritisations).

250

251 **RESULTS**

252

253           The viewpoint prioritisations selected different landscape priorities based upon their  
254 valued features (Figure 2). The ‘traditional’ conservation viewpoint priorities had the highest  
255 average proportion coverage of species distributions, primarily concentrated in northwest  
256 Scotland and scattered landscapes in the south of England. Conversely ‘local social  
257 instrumentalism’ spatial priorities were focused in landscapes in England close to large  
258 conurbations, especially London, maximising recreational value but resulting in lower  
259 exclusion of agricultural land; as well as landscapes in north England, which delivered  
260 landscape aesthetic value and flood protection services. ‘International market ecocentrism’  
261 priorities almost exclusively occurred in Scotland, and upland areas in Wales and northern  
262 England, driven by positive selection for carbon storage and avoiding the opportunity costs  
263 of more southerly productive farmland. The ‘new’ conservation spatial prioritisation selected  
264 landscapes appearing in both the ‘international market ecocentrism’ and ‘local social  
265 instrumentalism’ viewpoints, due the more balanced weightings across feature layers. These  
266 landscapes were primarily located in Scotland, upland areas in north England, and southeast  
267 England close to London.

268           We integrated the four viewpoints into single conservation strategies using two  
269 approaches (Figure 3). The inclusive approach selected landscapes in Scotland, upland Wales,  
270 north England, and southeast England. The pluralist approach contained similar priority  
271 areas, but higher priority landscapes were more concentrated in SE England. The pluralist  
272 approach had lower coverage of carbon and exclusion of agricultural value, but recreational  
273 value coverage was much higher than the inclusive approach (feature coverage, Figure 3;  
274 efficiency, Supplementary Figure 7). Species distributions received the best coverage through  
275 the ‘traditional’ viewpoint. The inclusive and pluralist integration approaches had higher  
276 species representation than the other viewpoints, although coverage was lower than the  
277 ‘traditional’ viewpoint (mean species distribution proportion coverage at 17% coverage

278 threshold: TRAD 40.3%, NEW 17.9%, ECON 8.6%, SOC 25.9%, Inclusive 27.2%, Pluralist 28.8%;  
279 other thresholds: Fig. 2-3). The existing protected area network (Supplementary Figure 1)  
280 matched 'new' and 'market ecocentrism' viewpoint prioritisation ranks more closely than  
281 'traditional' (TRAD  $\rho = 0.379$ , NEW  $\rho = 0.423$ , ECON  $\rho = 0.413$ , SOC  $\rho = 0.068$ ), which was  
282 contrary to our expectation given that the rationale and goals for identifying potential SSSIs  
283 and NNRs closely align with a traditional approach.

284 We found that both integration approaches had similar mean feature coverage  
285 efficiency (17% coverage threshold: inclusive 60.0%, pluralist 59.0%; other thresholds: Figure  
286 4) indicating similar overall optimality. However, the pluralist approach had higher minimum  
287 coverage efficiency for all thresholds, meaning features were included more equally (17%  
288 coverage threshold: inclusive 27.6%, pluralist 42.3%; other thresholds: Figure 4).

289

## 290 **DISCUSSION**

291

292 Different conservation viewpoints have passionate proponents and opponents  
293 (Kareiva and Marvier, 2012; Noss *et al.*, 2013; Soulé, 2013; Doak *et al.*, 2014; Sandbrook *et*  
294 *al.*, 2019) with differences that are difficult to balance within single coherent conservation  
295 plans. As expected we found that each of the four caricature viewpoints spatially prioritised  
296 different landscapes, depending on the values held, and resulted in different levels of feature  
297 coverage. We then aggregated viewpoint priorities by implementing inclusive and pluralist  
298 integration approaches, and we found that it is feasible to reconcile different viewpoint  
299 spatial priorities in a transparent manner. Although inclusiveness is typically associated with  
300 consensus building, our results demonstrate that applying pluralism approaches to  
301 systematic conservation planning methods can produce conservation plans that are just as  
302 coherent.



303 By accounting for conceptual similarities between viewpoints within a pluralist  
304 approach, similar viewpoints (which generate correlated spatial priorities) were prevented  
305 from 'crowding out' more marginal viewpoints. By preventing dominance by certain  
306 perspectives, the pluralist approach incorporated the most important locations for each  
307 viewpoint into the highest spatial prioritisation rankings (Figs. 1-3). The main difference  
308 between approaches was that the pluralist approach efficiently incorporated higher  
309 recreational value, concentrated around large conurbations, with minimal loss of other  
310 features. Although the pluralist approach performed less well for some features than the  
311 inclusive approach, overall it included features more equally while maintaining similar mean  
312 coverage efficiency. This shows, at least spatially, that a coherent conservation plan can be  
313 created while also representing potentially marginalised viewpoints.

314 Our approach is generalisable beyond conservation planning to many situations  
315 where perspectives need spatial reconciliation. Here we used a small number of caricature  
316 viewpoint weightings based on the conservation community to assess different integration  
317 methods, but applying these methods in a real-world setting would inevitably present  
318 additional complexities in reaching a compromise solution (Sandbrook *et al.*, 2019; Barton *et*  
319 *al.*, 2022). Depending upon the overall planning objectives, in real world situations, these  
320 would likely include involvement of a large number of participants. Many stakeholder  
321 viewpoints could be included through questionnaires and discussion fora, and involve the  
322 wider public, sampled through workshops and surveys (Rust *et al.*, 2021), or choice  
323 experiments (Badura *et al.*, 2020), but taking care to minimise social biases if determining  
324 viewpoints through group activities such as discussions and workshops.

325 Reconciling a large number of viewpoints is likely to require additional feature layers  
326 that may have to be co-developed with stakeholders, and some stakeholders may have  
327 unique interests in particular features. There are many possible conservation valuation  
328 methods and processes that can be used to produce feature layers, including different ES and

329 biodiversity modelling techniques, participatory mapping, and benefit transfer amongst  
330 others (Termansen *et al.*, 2022). However some priorities, and even entire viewpoints, may  
331 be difficult or impossible to quantify (Wyborn and Evans, 2021), and these non-quantifiable  
332 cultural and contextual values will still need to be incorporated within the process (Chaplin-  
333 Kramer *et al.*, 2021; Fleischman *et al.*, 2022; Strassburg *et al.*, 2022). Therefore, it may be  
334 difficult to entirely reconcile complex values of a large number of stakeholders using solely  
335 quantitative methods, but this could be at least partially redressed through stakeholder  
336 engagement by situating the approach within a systematic planning framework (SCP).

337         SCP involves stakeholders through an interactive process and this is important when  
338 implementing both numeric and non-numeric approaches to reconciling viewpoints.  
339 Additional transparency in the process is achieved by developing the separate viewpoints  
340 into independent spatial priorities, enabling advocates of any particular viewpoint to  
341 compare the spatial consequences of their viewpoint with others. As the reconciliation  
342 process is numerical, each actor can compare alternative possible integrated solutions with  
343 that of their own initial viewpoint, leading to a more informed final decision. Decision-  
344 support tools could assist the balancing of different viewpoints by, for example, iteratively  
345 presenting integrated prioritisations to participants within the Delphi technique (Mukherjee  
346 *et al.*, 2015), or exploring trade-offs as part of a broader multi-criteria decision analysis  
347 (MCDA) approach (Esmail and Geneletti, 2018). This contrasts with situations where debates  
348 to reconcile viewpoints take place prior to the analysis, where consensus is generated by  
349 discussion rather than formal analytical reconciliation.

350         Importantly, this approach provides a decision-support tool and **not** an ultimate  
351 decision. Although we demonstrate how different viewpoints can be reconciled, ultimately  
352 the equity of any conservation plan depends upon the representation of stakeholders  
353 included within the planning process. Local stakeholder engagement, for example through  
354 local partnerships and public participation, can be especially overlooked within conservation

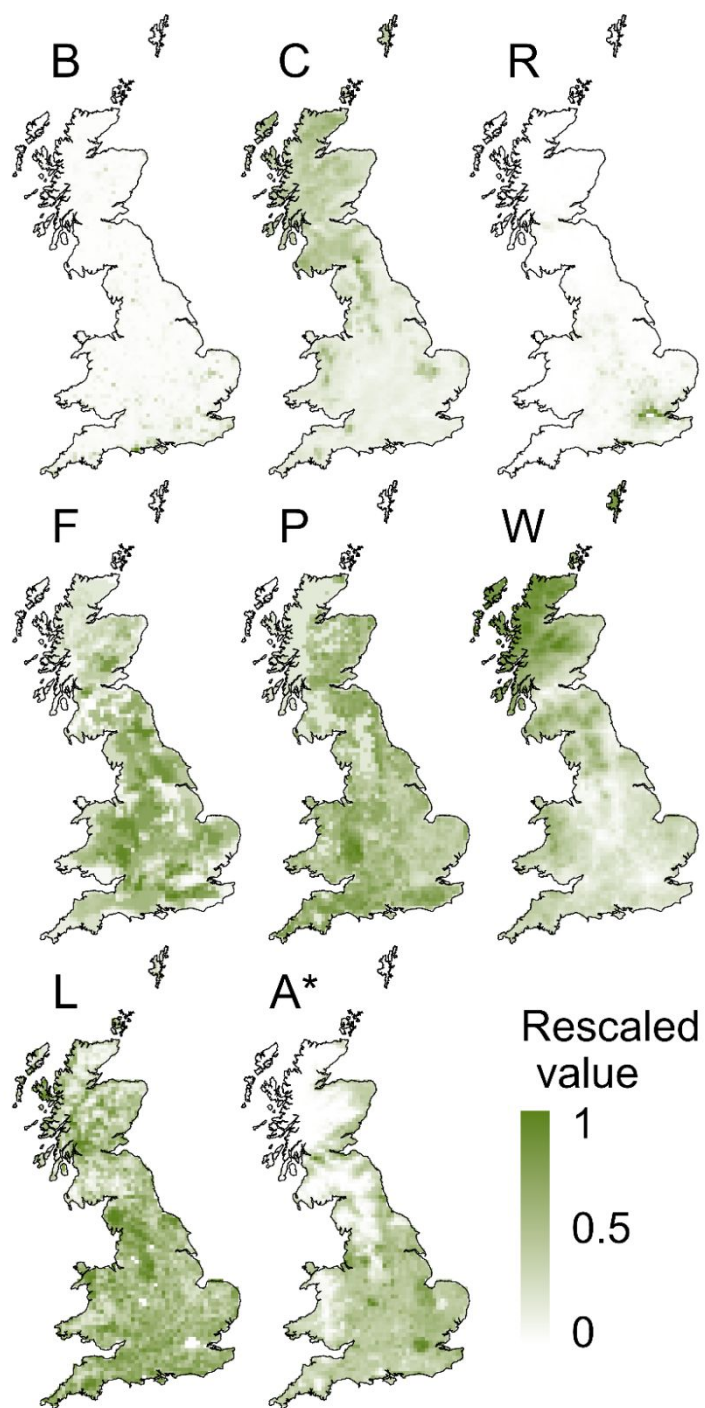
355 planning, but is also important to ensure a protected area network that delivers for all  
356 (Blicharska *et al.*, 2016; Sterling *et al.*, 2017; Barton *et al.*, 2022).

357         The British protected area network was designed primarily to protect species and  
358 habitats in a representative way, and we expected more spatial overlap with a prioritisation  
359 based on the values of conserving species diversity. The suite of notified sites might be  
360 inefficient from the 'traditional' viewpoint for several reasons. Although the rationale behind  
361 the initial network designation and subsequent expansion may have been 'traditional', it may  
362 be that sites were not identified optimally through the notification process in terms of  
363 species representation. Additionally, notification will have depended not only on the quality  
364 of feature, but also on other conservation planning considerations such as land ownership  
365 and local socio-economic context. For example large SSSIs are primarily in upland areas,  
366 because less intensive land management occurred here in the past, allowing more semi-  
367 natural habitat to persist; in contrast to the lowlands where much smaller fragments of  
368 habitat remained to be protected (Bailey *et al.*, 2022).

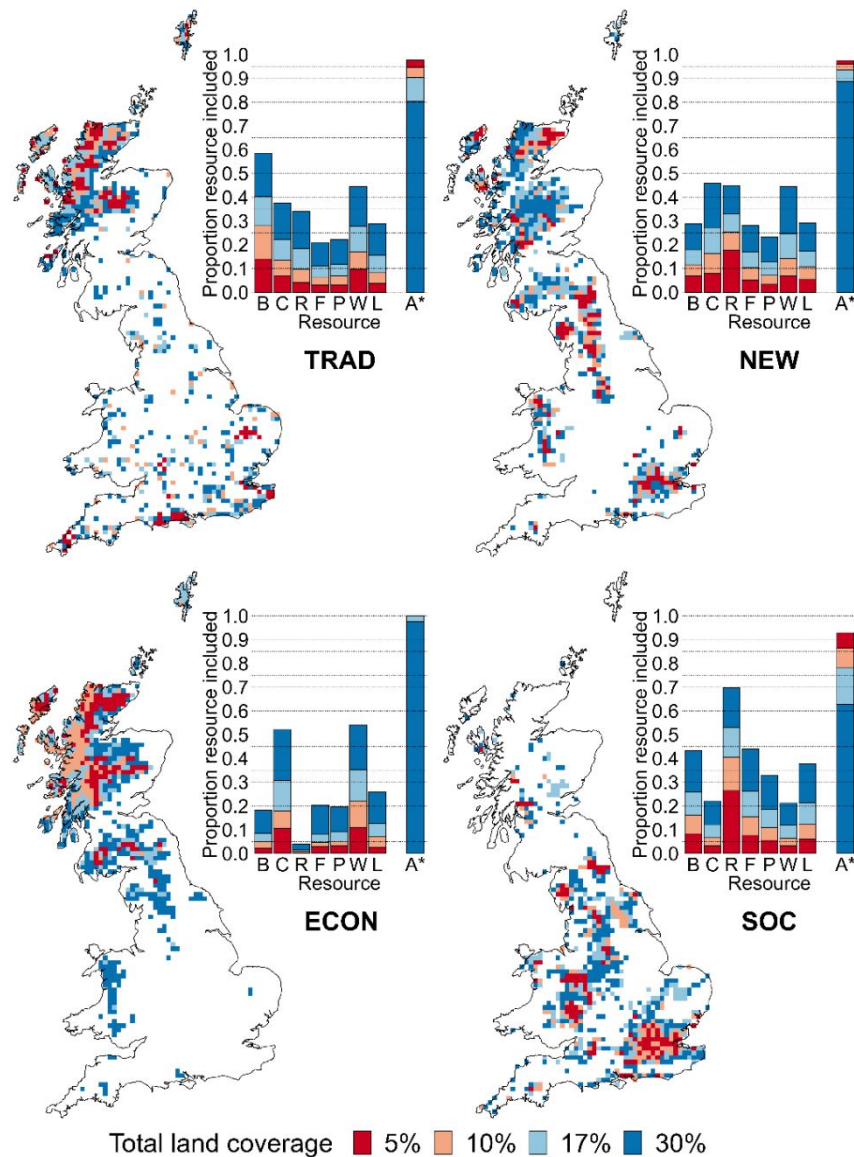
369         This work focuses on reconciling existing conservation feature values, not the  
370 establishment opportunities for potential future gains in feature values. Using carbon storage  
371 as an example, given its importance as a likely future driver for land use and management  
372 policy (Committee on Climate Change, 2020), these two distinct approaches are important:  
373 protecting and restoring existing high carbon habitats, particularly peatlands; and increasing  
374 carbon sequestration through the creation of new habitats, particularly woodlands (Gregg *et*  
375 *al.*, 2021). Our approach has only taken account of the first of these, but a conservation  
376 strategy using a combination of protecting existing high-value landscapes, and implementing  
377 habitat enhancement or creation in others is needed for both biodiversity and other  
378 ecosystem commitments (Soto-Navarro *et al.*, 2020).

379           Within each landscape, different types of action will be required depending on what  
380 is important and the local land use context, considering that the distributions of ecosystem  
381 carbon, biodiversity value and other ecosystem services may be positively correlated in some  
382 landscapes but negatively so in others (Anderson *et al.*, 2009). For example, if a low intensity  
383 agricultural landscape is prioritised for carbon storage, flood prevention or biodiversity, then  
384 enhanced protection and additional habitat management to further deliver on these  
385 ecosystem features may be implemented. However, strictly protected areas for biodiversity  
386 are unlikely to be the method to best incorporate all features, especially those valued by  
387 critical social science. Thus, other non-statutory area-based conservation measures may be  
388 needed to deliver for aspects such as human well-being. Similarly, other national schemes,  
389 such as tree planting, can also have hugely varying outcomes depending upon the spatial  
390 distribution of implementation (UNEP-WCMC and LWEC, 2014), and these could also be  
391 considered within a pluralist framework.

392           As well as balancing differing viewpoints on existing resource protection, it is  
393 important to consider future expected changes in landscape feature values due to climate  
394 change within any implemented conservation plan (Bateman *et al.*, 2013). In addition to  
395 conservation feature values changing over time; conservation perspectives, the needs of  
396 society, and value systems themselves change and develop (Mace, 2014; Anderson *et al.*,  
397 2022), and so joint conservation plans have to be re-evaluated periodically. However, whilst  
398 the weight attached to different conservation objectives will inevitably change, including a  
399 broad range of benefits in conservation planning will remain important. In developing the  
400 post-2020 global biodiversity framework (CBD, 2021), it is vital to acknowledge and carefully  
401 consider how to equitably integrate different viewpoints on how to implement area-based  
402 conservation.

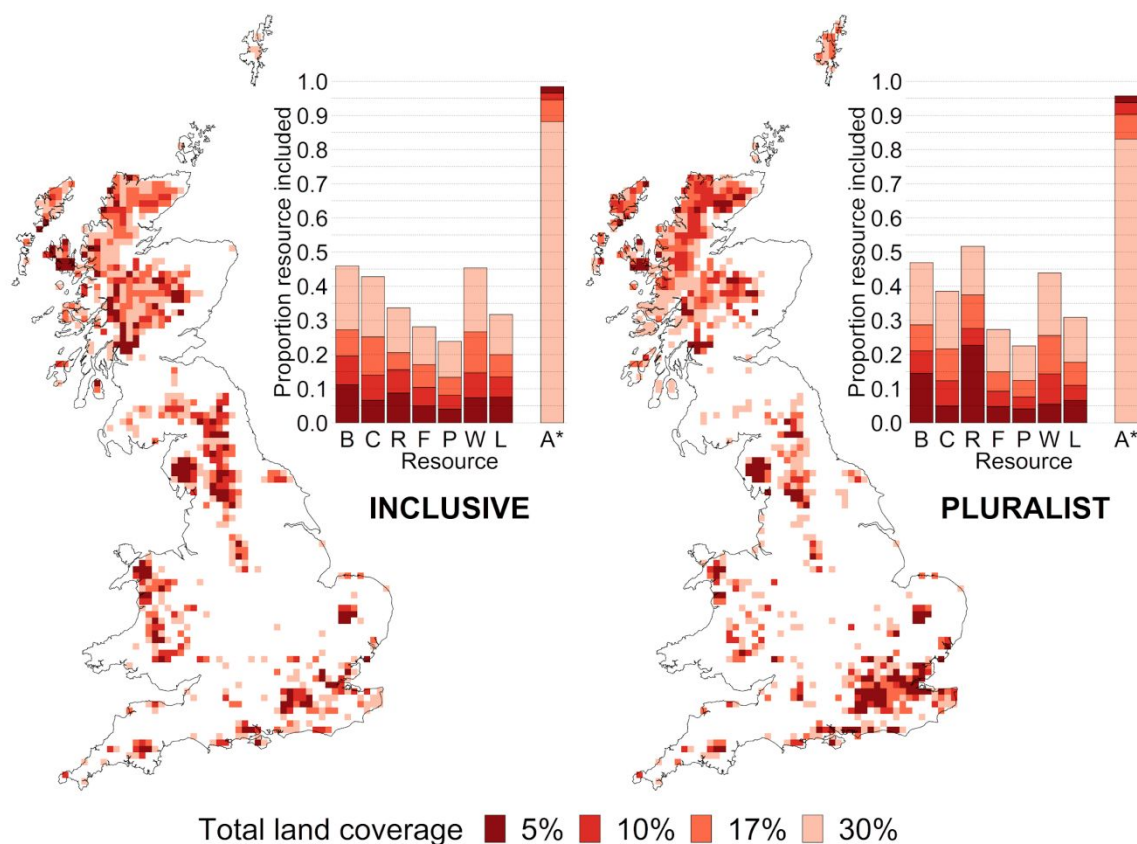


**Figure 1** Rescaled ecosystem service, biodiversity and socio-environmental value feature layers included within the analysis including; mean priority species distribution proportion coverage (B), carbon storage (C), recreation (R), flood regulation (F), pollination (P), wilderness (W), landscape aesthetic value (L) and agricultural/land value (A\*).

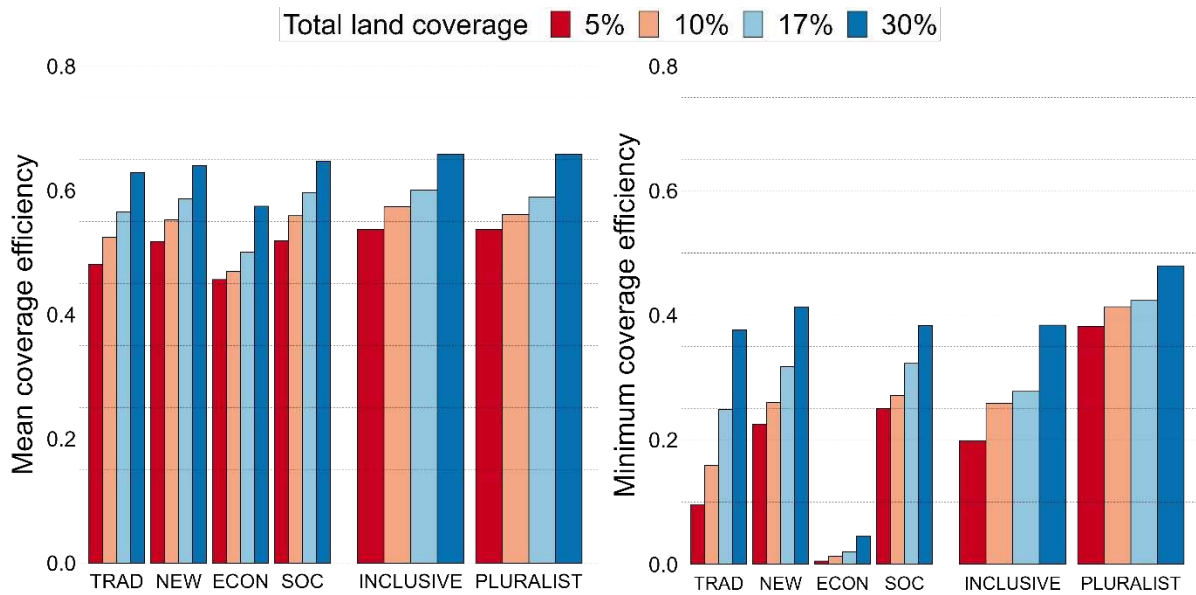


**Figure 2** Feature coverage using spatial prioritisation for each of the four viewpoints; TRAD – ‘traditional’, NEW – ‘new’ conservation, ECON – ‘international market ecocentrism’, SOC – ‘local social instrumentalism’. Maps indicate the priority areas for different coverage thresholds, and corresponding bar plots present proportion of feature included for each, including mean priority species distribution proportion coverage (B), carbon storage (C), recreation (R), flood regulation (F), pollination (P), wilderness (W), landscape aesthetic value (L) and agricultural/land value (A\*). A\*

indicates the only negative weighting where proportion excluded, not included, is shown and so higher land coverage results in lower proportion excluded.



**Figure 3** Spatially aggregating the four conservation viewpoint priorities (TRAD – ‘traditional’, NEW – ‘new’ conservation, ECON – ‘international market ecocentrism’, SOC – ‘local social instrumentalism’) using inclusive (vote counting) and pluralist (accounting for distinctiveness) integration methods. Maps indicate the priority areas for different coverage thresholds, and corresponding bar plots present proportion of feature included for each, including mean priority species distribution proportion coverage (B), carbon storage (C), recreation (R), flood regulation (F), pollination (P), wilderness (W), landscape aesthetic value (L) and agricultural/land value (A\*). A\* indicates the only negative weighting where proportion excluded, not included, is shown and so higher land coverage results in lower proportion excluded.



**Figure 4** Mean and minimum feature coverage efficiency of viewpoint prioritisation performance (TRAD – ‘traditional’, NEW – ‘new’ conservation, ECON – ‘international market ecocentrism’, SOC – ‘local social instrumentalism’) and integration approach (inclusive and pluralist conservation). Efficiency is calculated as the proportion of a feature covered by a prioritisation for each land coverage threshold (5%, 10%, 17%, and 30%), compared to the maximum possible if only that feature was prioritised. Features included are mean priority species distribution proportion coverage (B), carbon storage (C), recreation (R), flood regulation (F), pollination (P), wilderness (W), landscape aesthetic value (L) and agricultural/land value (A\*). The left-hand panel shows the mean performance across all resource types (conservation features), whereas the right-hand panel shows the efficiency of the feature that is least well covered by a particular approach. Inclusive and pluralist approaches perform similarly, but pluralism has a higher minimum feature coverage threshold.



**Table 1** Definitions of caricature viewpoints and viewpoint integration approaches.

<b>Conservation viewpoint</b>	A personal perspective that determines how nature is valued, and how to best conserve it. This analysis uses four arbitrary caricature conservation viewpoints to analyse approaches to viewpoint integration.
'Traditional' (TRAD)	<p>Ecocentric viewpoint, aiming to conserve species diversity and natural habitats for their intrinsic value and for their ability to regulate ecosystem services. Intrinsic value is ascribed to biotic diversity and ecological complexity, with a preference for 'natural' systems. Adapted from Soulé (1985).</p> <p>Weightings: species distributions and wilderness.</p>
'New' (NEW)	<p>Anthropocentric viewpoint, motivated by achieving conservation action through attaining economic and social benefit. Seeks to conserve biodiversity in human-modified as well as 'natural' landscapes, whilst also maximising human well-being and economic objectives. Adapted from Marvier (2014).</p> <p>Weightings: widest scope of the four viewpoints, including species and all economic and social value data, apart from wilderness.</p>
International market ecocentrism (ECON)	<p>Utilises capitalist economic arguments to deliver ecocentric conservation, but ignores human well-being and local benefits. Aims to protect intrinsic ecological value over a large area, typically 30-50% of land. This is achieved by employing a free market approach to resource extraction on the remaining land, with the view that this would maximise profit to resource consumption efficiency, and hence protect the 'spared' land. Adapted from Wilson (2016).</p> <p>Weightings: agricultural value (avoid) and related pollination service flow, as well as carbon storage and species distributions.</p>
Local social instrumentalism (SOC)	<p>Favours prioritising conservation benefitting human well-being at the local scale, but opposed to intrinsic value of nature arguments, economic objectives, and links with capitalism and corporations. Adapted from 'social instrumentalism' in Matulis and Moyer (2017).</p> <p>Weightings: ecosystem services that benefit the local population, i.e. flood prevention and recreation, as well as landscapes that are important to people, and a lower weighting for species distributions.</p>
<b>Viewpoint integration approach</b>	Numeric aggregation methods to spatially reconcile differences between individual viewpoints into a single, coherent conservation plan.

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Inclusive	Seek to embrace and bring together all perspectives, by building consensus and reducing disputes between people holding different views, and creating a single voice for conservation that is more unified, and hence carries more weight (Tallis & Lubchenco 2014). Here we implement this using an additive vote counting formula.
Pluralist	Accept and engage with diverse perspectives on biodiversity conservation, and give voice to marginalised values and views (Pascual et al. 2021). This is implemented by accounting for similarity between viewpoints and upweighting more distinct viewpoints.

**Table 2** Weightings for feature layers included within each of the four conservation viewpoints.

<i>Feature</i>	<b>Traditional conservation (TRAD)</b>	<b>‘New’ conservation (NEW)</b>	<b>International market ecocentrism (ECON)</b>	<b>Local social instrumentalism (SOC)</b>
Biodiversity (B)	1	1	1	0.5
Carbon (C)	0	1	1	0
Number of visits to recreation space (R)	0	1	0	1
Flood regulation (F)	0	1	0	1
Pollinator services (P)	0	0.5	0.5	0
Wilderness (W)	0.25	0	0	0
Landscape aesthetic value (L)	0	1	0	1
Agricultural land classification (A*)	0	-0.5	-1	0

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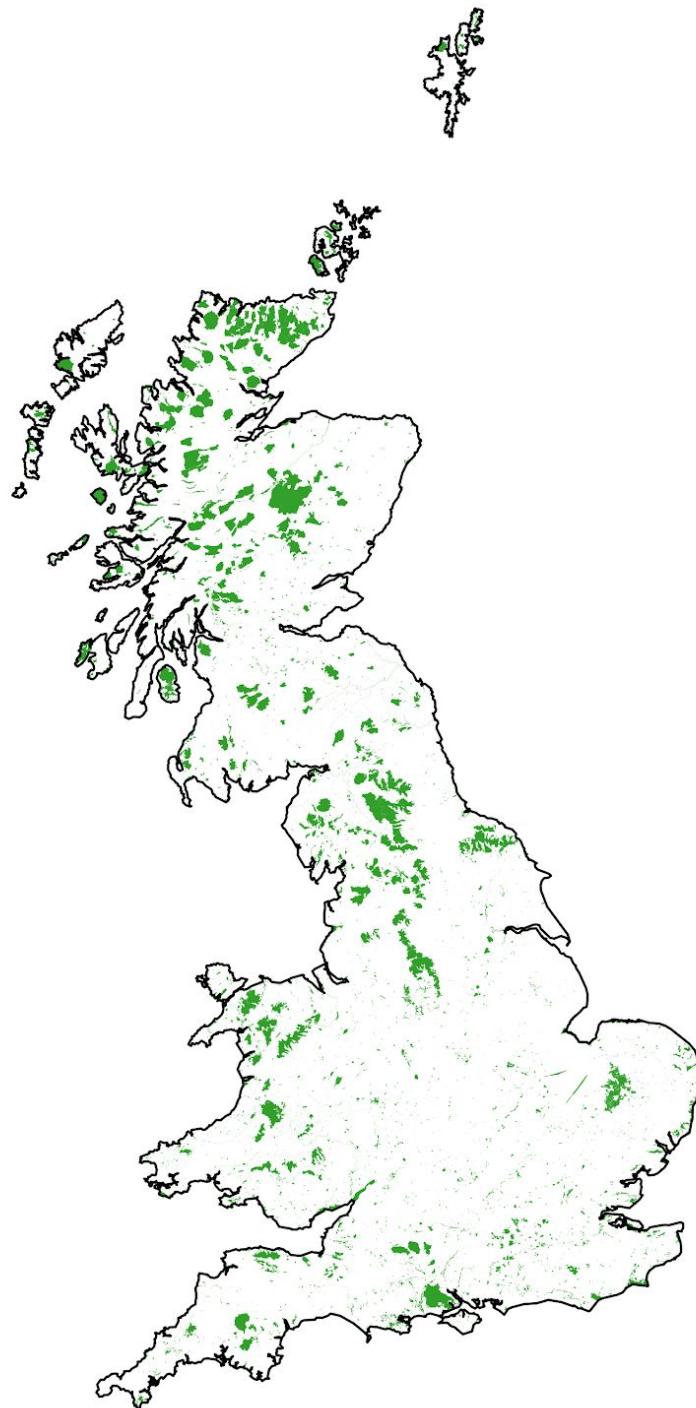
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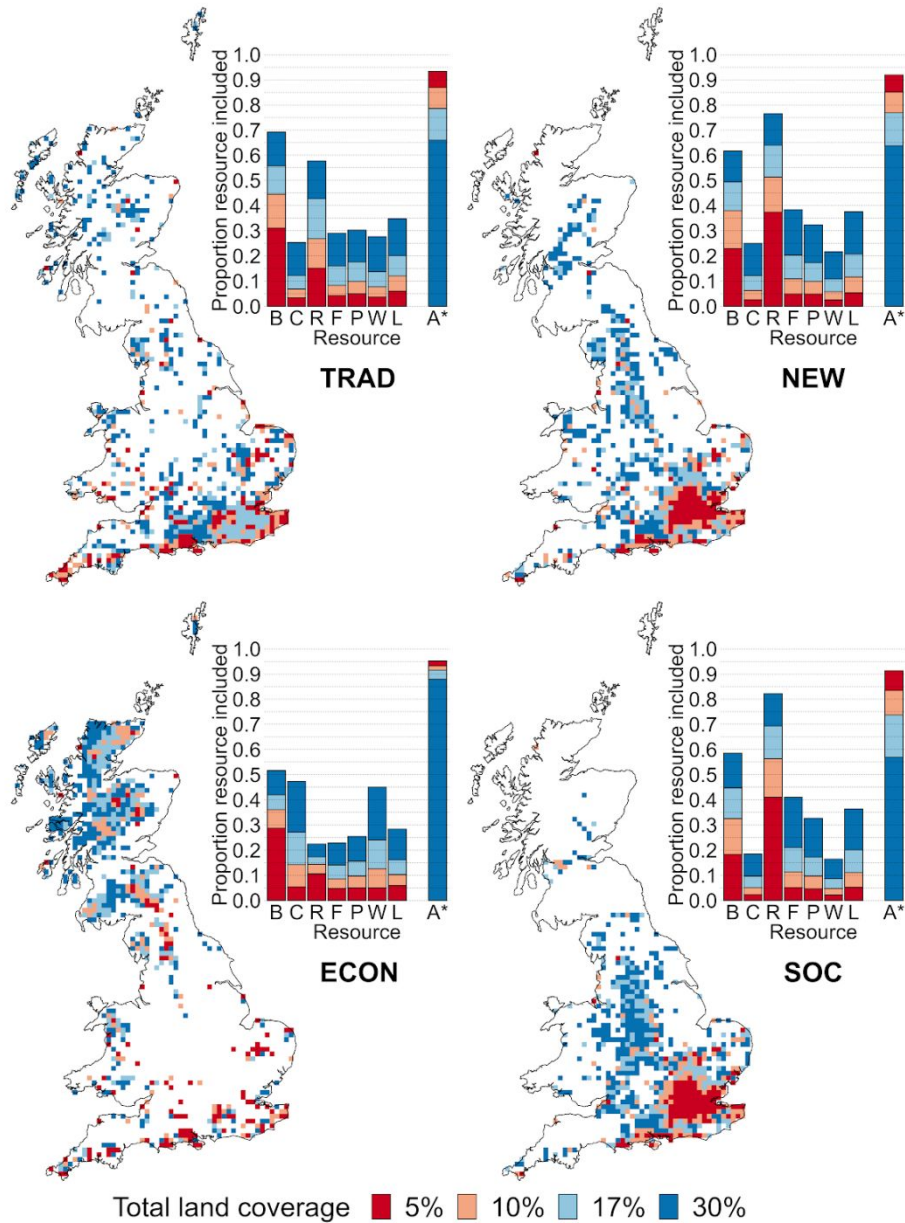
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## SUPPLEMENTARY MATERIALS



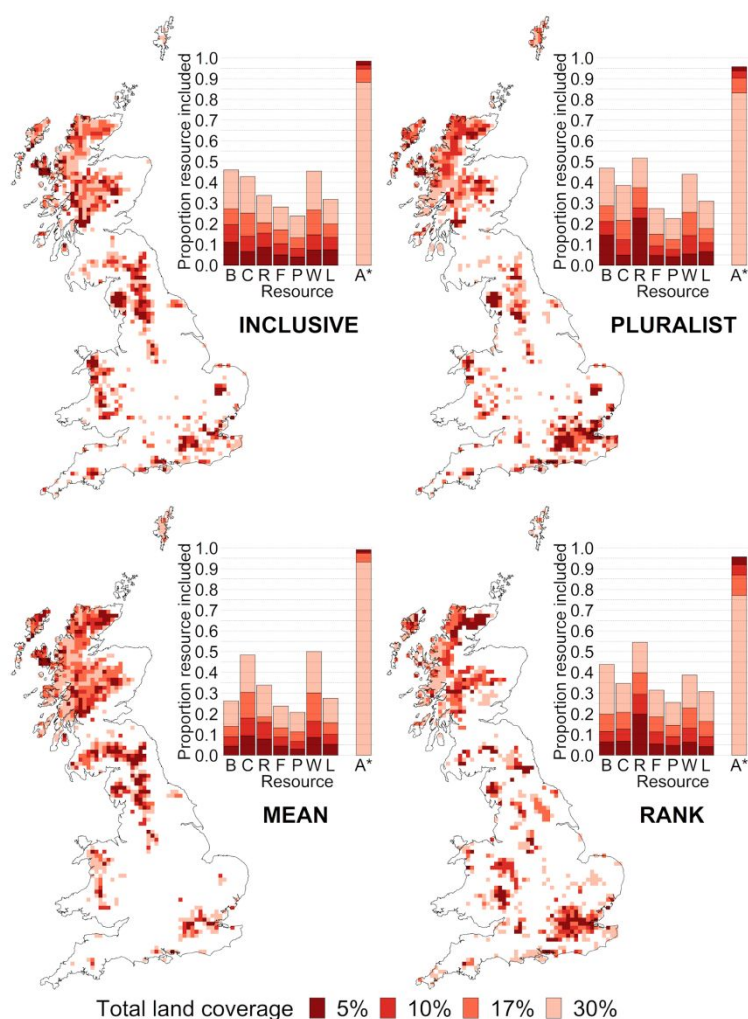
**Figure S1** Protected areas included within the analysis: Sites of Special Scientific Interest (SSSI) and National Nature Reserves (NNR) designated at the time of the study.



**Figure S2** Feature coverage using spatial prioritisation for each of the four viewpoints using the *additive benefit function* prioritisation method; TRAD – ‘traditional’, NEW – ‘new conservation’, ECON – ‘international market ecocentrism’, SOC – ‘local social instrumentalism’. Maps indicate the priority areas for different coverage thresholds, and corresponding bar plots present proportion of feature included for each, including mean priority species distribution proportion coverage (B), carbon storage (C), recreation (R), flood regulation (F), pollination (P), wilderness (W), landscape aesthetic value (L)

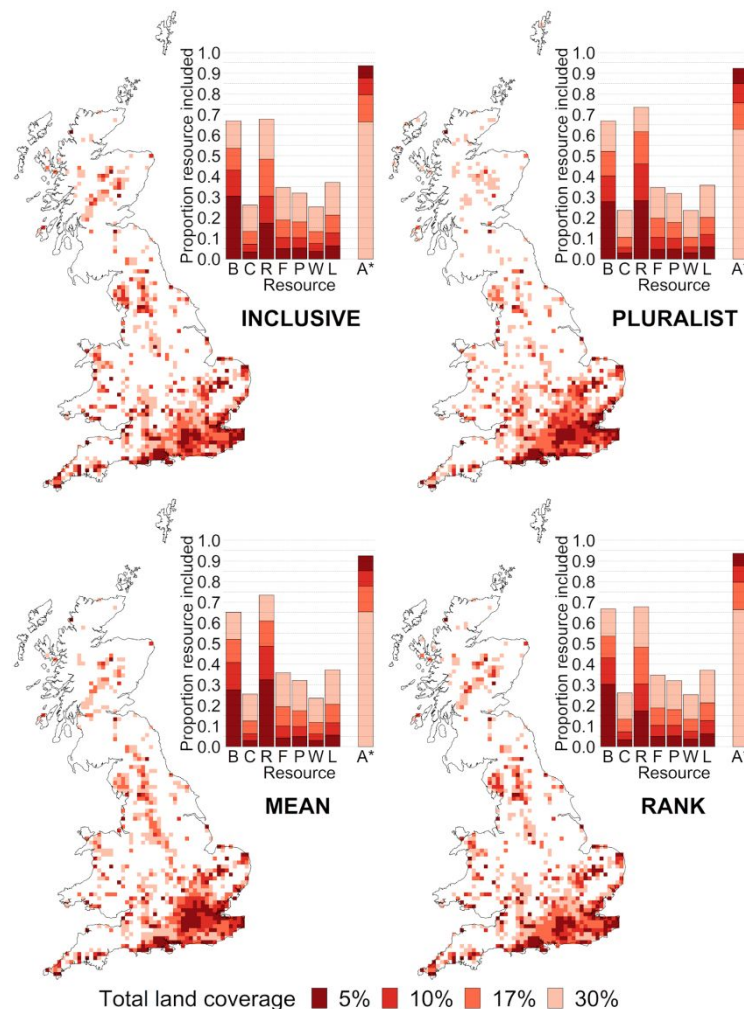


and agricultural/land value (A\*). A\* indicates the only negative weighting where proportion excluded, not included, is shown and so higher land coverage results in lower proportion excluded.



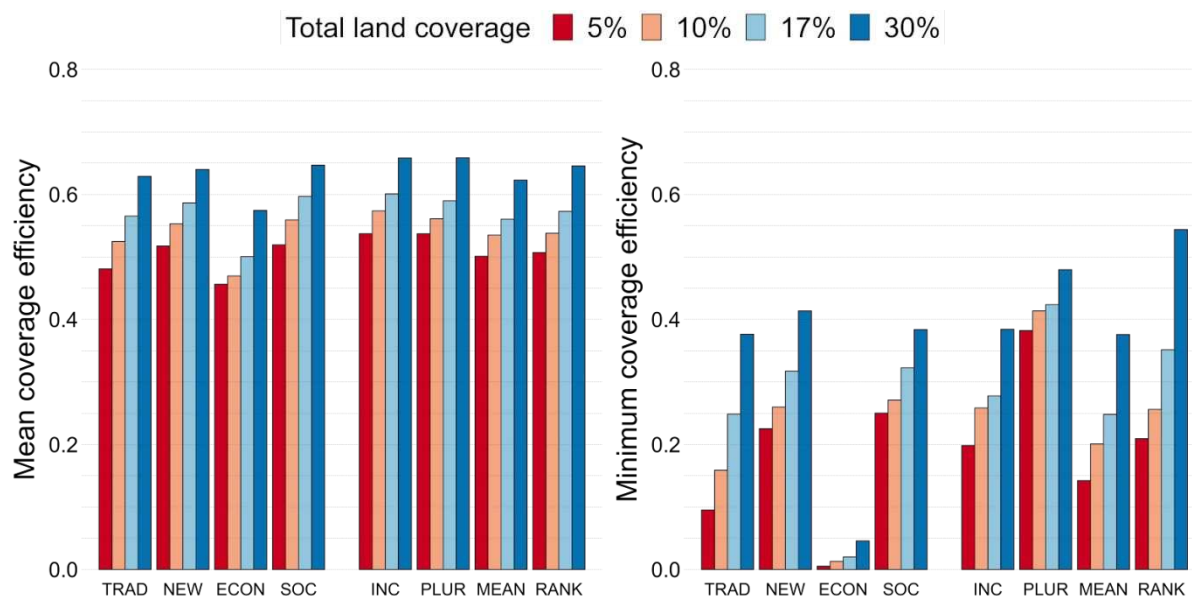
**Figure S3** Spatially aggregating the four conservation viewpoint priorities (TRAD – ‘traditional’, NEW – ‘new conservation’, ECON – ‘international market ecocentrism’, SOC – ‘local social instrumentalism’) using the *core area zonation* prioritisation method. We used inclusive (vote counting) and pluralist (accounting for distinctiveness) methods; as well as two additional integration approaches MEAN (averaging feature weightings before prioritisation), and RANK (undertaking additional prioritisation of viewpoint landscape ranks). Maps indicate the priority areas for different coverage thresholds, and corresponding bar plots present proportion of feature included for each, including mean priority species distribution proportion coverage (B), carbon storage (C), recreation

(R), flood regulation (F), pollination (P), wilderness (W), landscape aesthetic value (L) and agricultural/land value (A\*). A\* indicates the only negative weighting where proportion excluded, not included, is shown and so higher land coverage results in lower proportion excluded.

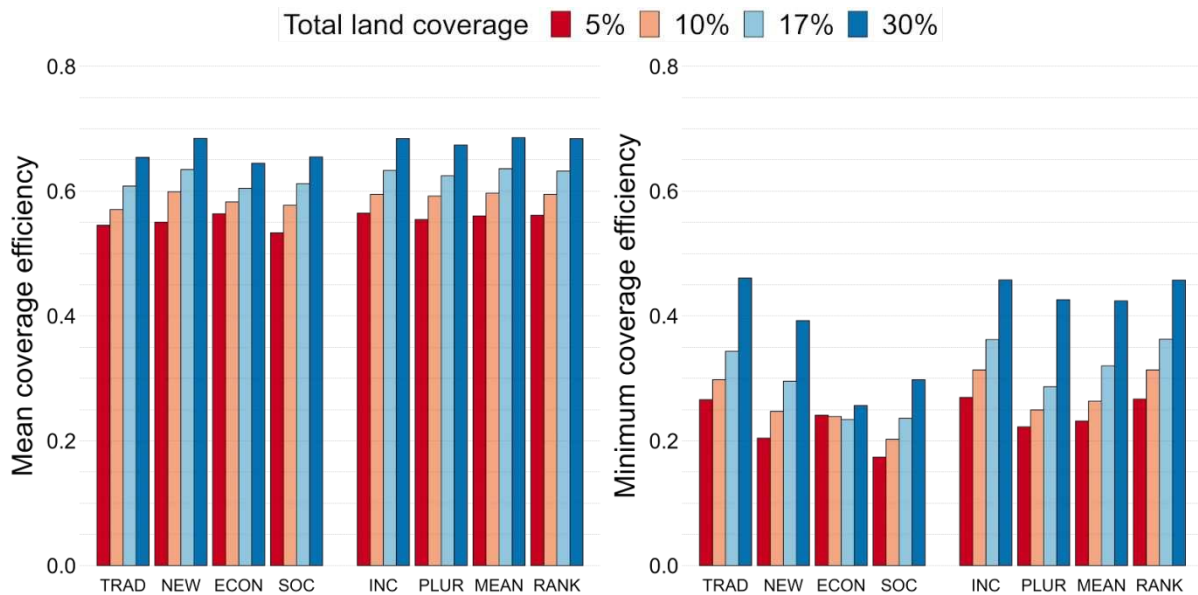


**Figure S4** Spatially aggregating the four conservation viewpoint priorities (TRAD – ‘traditional’, NEW – ‘new conservation’, ECON – ‘international market ecocentrism’, SOC – ‘local social instrumentalism’) using the *additive benefit function* prioritisation method. We used inclusive (vote counting) and pluralist (accounting for distinctiveness) methods; as well as two additional integration approaches MEAN (averaging feature weightings before prioritisation), and RANK (undertaking additional prioritisation of viewpoint landscape ranks). Maps indicate the priority areas for different coverage thresholds, and corresponding bar plots present proportion of feature included for each,

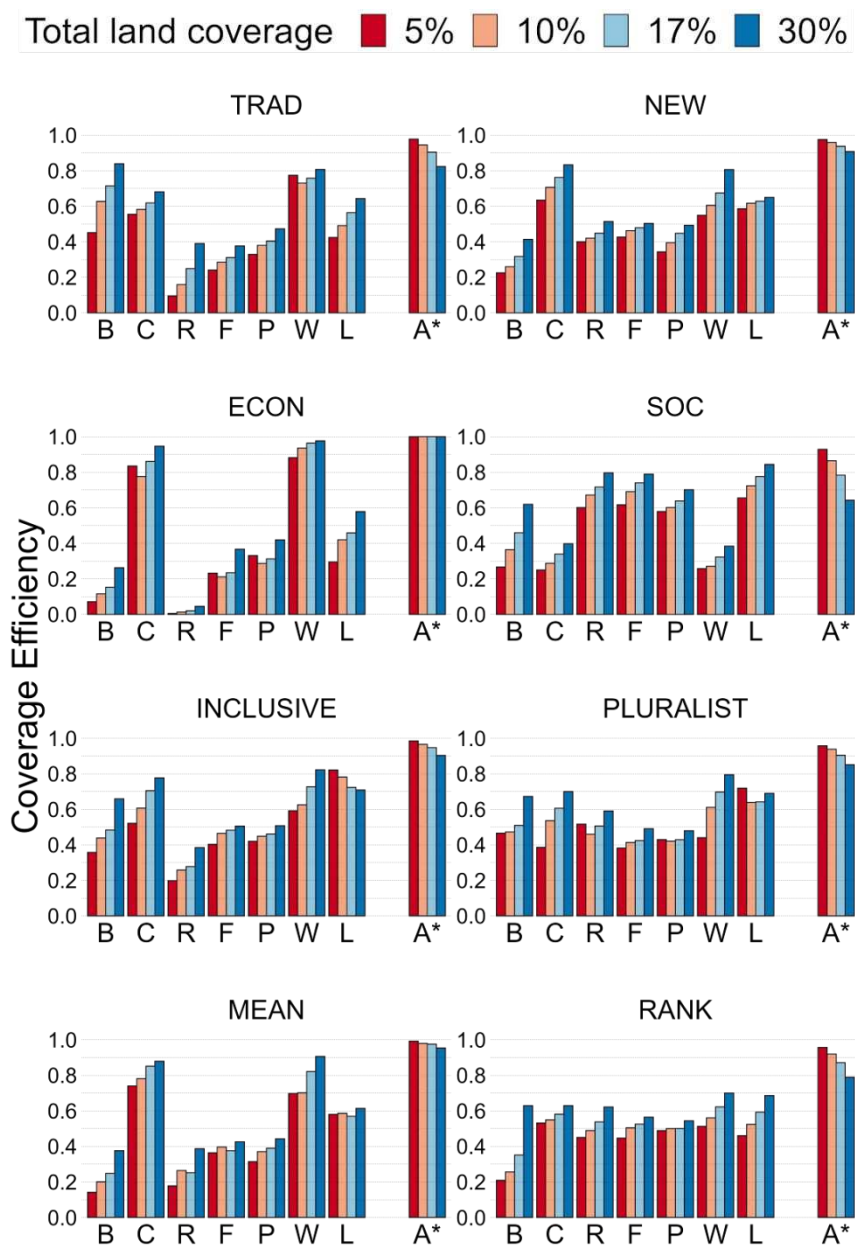
including mean priority species distribution proportion coverage (B), carbon storage (C), recreation (R), flood regulation (F), pollination (P), wilderness (W), landscape aesthetic value (L) and agricultural/land value (A\*). A\* indicates the only negative weighting where proportion excluded, not included, is shown and so higher land coverage results in lower proportion excluded.



**Figure S5** Mean and minimum feature coverage efficiency of core area zonation prioritisation performance (TRAD – ‘traditional’, NEW – ‘new conservation’, ECON – ‘international market ecocentrism’, SOC – ‘local social instrumentalism’) and integration approach (Inclusive and Pluralist conservation, as well as two additional integration approaches MEAN and RANK). Efficiency is calculated as the proportion of features covered by a prioritisation for each land coverage threshold (5%, 10%, 17%, and 30%), compared to the maximum possible if only that feature was prioritised. Features included are mean priority species distribution proportion coverage (B), carbon storage (C), recreation (R), flood regulation (F), pollination (P), wilderness (W), landscape aesthetic value (L) and agricultural/land value (A\*). The left-hand panel shows the mean performance across all resource types (conservation features), whereas the right-hand panel shows the coverage of the feature that is least well covered by a particular approach.

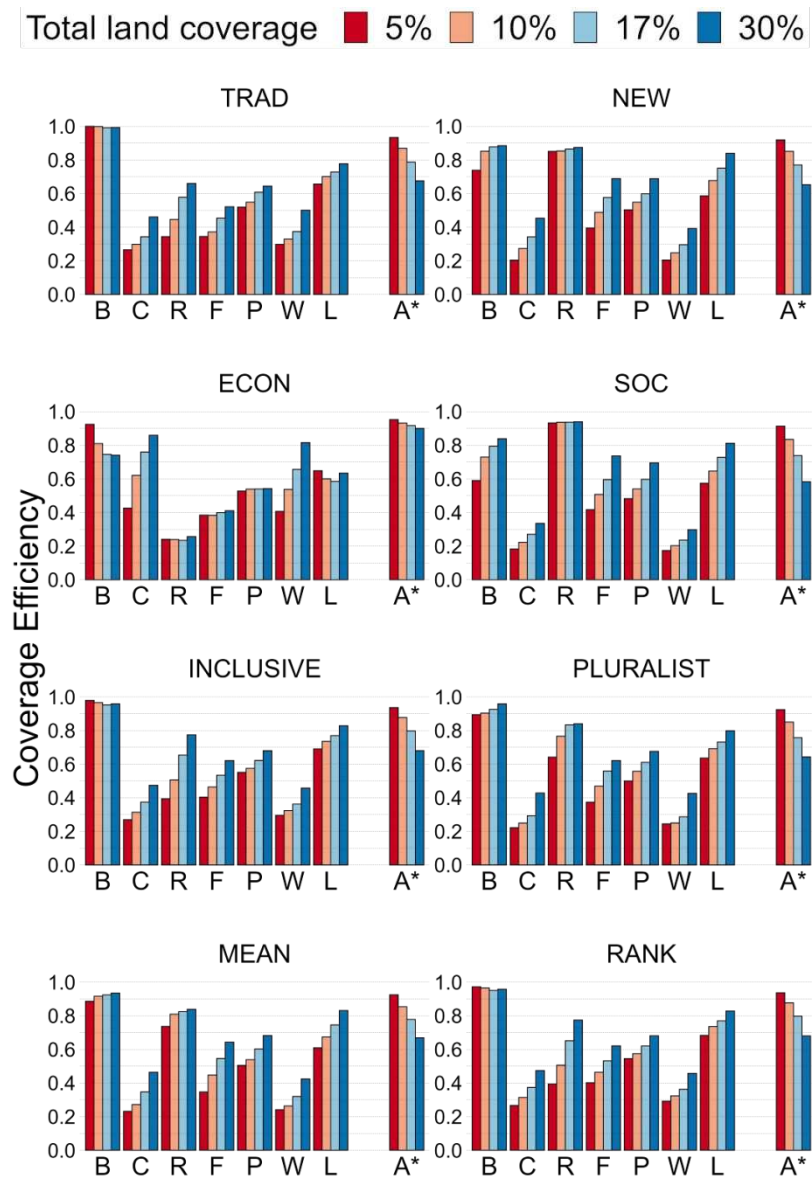


**Figure S6** Mean and minimum feature coverage efficiency of *additive benefit function* prioritisation performance (TRAD – ‘traditional’, NEW – ‘new conservation’, ECON – ‘international market ecocentrism’, SOC – ‘local social instrumentalism’) and integration approach (Inclusive, and Pluralist conservation as well as two additional integration approaches MEAN and RANK). Efficiency is calculated as the proportion of features covered by a prioritisation for each land coverage threshold (5%, 10%, 17%, and 30%), compared to the maximum possible if only that feature was prioritised. Features included are mean priority species distribution proportion coverage (B), carbon storage (C), recreation (R), flood regulation (F), pollination (P), wilderness (W), landscape aesthetic value (L) and agricultural/land value (A\*). The left-hand panel shows the mean performance across all resource types (conservation features), whereas the right-hand panel shows the coverage of the feature that is *least well covered* by a particular approach.



**Figure S7** Efficiency of *core area zonation* prioritisation performance (TRAD – ‘traditional’, NEW – ‘new conservation’, ECON – ‘international market ecocentrism’, SOC – ‘local social instrumentalism’) and integration approach (inclusive, and pluralist conservation as well as two additional integration approaches MEAN and RANK). Efficiency was calculated as the proportion of each feature covered compared to the maximum possible for each land coverage threshold (5%, 10%, 17%, and 30%). Features included are mean priority species distribution proportion coverage (B), carbon storage (C),

recreation (R), flood regulation (F), pollination (P), wilderness (W), landscape aesthetic value (L) and agricultural/land value (A\*).



**Figure S8** Efficiency of *additive benefit function* prioritisation performance (TRAD – ‘traditional’, NEW – ‘new conservation’, ECON – ‘international market ecocentrism’, SOC – ‘local social instrumentalism’) and integration approach (Inclusive, and Pluralist conservation as well as two additional integration approaches MEAN and RANK). Efficiency was calculated as the proportion of each feature covered compared to the maximum possible for each land coverage threshold (5%, 10%, 17%, and 30%). Features included are mean priority species distribution proportion coverage (B),

carbon storage (C), recreation (R), flood regulation (F), pollination (P), wilderness (W), landscape aesthetic value (L) and agricultural/land value (A\*).

**Supplementary Table 1** Output from the PCA analysis used to create the pluralist approach rankings. We partitioned variance from viewpoint weightings of feature layers, creating principal components (PC; columns). Cumulative proportion of variance explained by PCs included in brackets. We multiplied viewpoint prioritisation landscape rankings by corresponding PC eigenvectors, and took the absolute value of the sum (dot product). PCs were added iteratively until maximum viewpoint eigenvalue across PCs (bold) was included (PC3). The first PC is associated with the NEW and ECON viewpoints, the second PC is strongly associated with the SOC viewpoint, and the third PC is strongly associated with the TRAD viewpoint.

	<b>PC1</b>	<b>PC2 (0.911)</b>	<b>PC3 (0.999)</b>	<b>PC4 (1.000)</b>
	<b>(0.601)</b>			
<b>TRAD</b>	-0.168	0.234	<b>-0.927</b>	0.240
<b>NEW</b>	<b>-0.658</b>	-0.325	0.205	0.647
<b>ECON</b>	<b>-0.693</b>	0.515	0.129	-0.487
<b>SOC</b>	-0.241	<b>-0.758</b>	-0.286	-0.535

## Supplementary methods

### Feature layers

The complete list of seven ecosystem service and socio-environmental value layers were collated as follows:

Five ES layers were included; carbon storage (existing), agricultural value, recreational services, flood regulation, and pollination services. Carbon storage value was calculated as the sum of interpolated below-ground carbon from the CEH Soil Carbon Map to a depth of 100 cm (Bradley *et al.*, 2005), and estimated above-ground carbon using the 2007 Land Cover Map (Henrys, Keith and Wood, 2016). Agricultural value was assigned based upon agricultural land classifications for England (Natural Resources Wales, 2019; The James Hutton Institute, 2019; Natural England, 2020). Classifications were standardised between countries into an interoperable code, and the mean landscape value was then rescaled and subtracted from 1 to calculate the final agricultural value used for the spatial prioritisations [see Cunningham *et al.* (2021) for details]. Urban areas were then given the highest value, indicating unsuitable land use for terrestrial conservation. Recreation value was estimated from the predicted annual visits/ha for a potential new National Park, see Schägner *et al.* (2016).

The value of protecting land for flood prevention depends on (a) supply: the degree to which upstream land reduces peak discharge volume (i.e. flooding risk); and (b) demand: the damage a flood could cause accounting for location within the catchment (i.e. aggregated damage within *and* downstream of each catchment). These factors interact such that if there is no valuable infrastructure downstream flood prevention action gains nothing, but equally if a location currently does little to reduce peak discharge then flood prevention value is again low. Hence, flood regulation value was estimated using a supply index (predicted total effect of upstream land on river discharge after precipitation events), and a catchment level demand index (downstream flood damage accounting for



upstream area); see Stürck et al. (2014) for details of supply and demand indexes used in this analysis. These indices do not provide an absolute measure of service flow; however, the relative distributions can be compared. Flood regulation flow was estimated by ranking the supply and demand indices separately, and then taking the minimum rank of the two. In this way, areas that had both relatively high supply and demand received higher value. Pollination service flow was similarly calculated with a supply index (estimated visitation probability by pollinators), and demand index (area of pollinator crops weighted by dependency level), see Schulp et al. (2014).

Additionally two socio-environmental value layers were added; wilderness and landscape aesthetic value. Wilderness was included from the 'wilderness register and indicator for Europe' map, created from a combination of naturalness, remoteness from settlements and access, and terrain ruggedness (Kuiters *et al.*, 2013). Landscape aesthetic value was quantified based on numbers of geolocated unique user uploads to three social media platforms, see Van Zanten et al. (2016). The mean landscape rank of the number of uploads to each platform was then taken as the 'landscape aesthetic value'.

### **Other viewpoint integration approaches**

In addition to the inclusive and pluralist approaches described within the main text, two additional multi-criteria decision analysis (MCDA) spatial approaches to integrating viewpoints together were tested. The first approach involved calculating the mean feature weightings between viewpoints (mean of the four weightings for each feature in Table 4.2) prior to any spatial prioritisation. These mean weightings were then used within a single spatial prioritisation using Zonation (MEAN), and hence this approach approximates deciding on conservation priorities prior to any spatial prioritisation. The other integration approach involved using the output landscape rankings from the four viewpoint prioritisations (TRAD, NEW, ECON, SOC) to seek an overall compromise (RANK). A further Zonation prioritisation was carried out on these ranks (each individual viewpoint was treated as an input feature layer). Neither of these two alternative methods outperformed the

inclusive and pluralist methods described in the main text in terms of mean or minimum feature coverage efficiency using CAZ (with the exception of higher RANK minimum efficiency at the highest [30%] area coverage threshold). MEAN consistently underperformed the other approaches using CAZ.

All four methods were tested using both the *core area zonation* (CAZ) and *additive benefit function* (ABF) prioritisation method. Both methods iteratively remove landscapes contributing the smallest value to the remaining landscapes. Through this removal, landscapes remaining within the solution longer complement other landscapes to a greater extent, in terms of contributing the most to underrepresented features. Using CAZ, landscape value is calculated as the *maximum* weighted proportion of any positive feature within the remaining landscapes (minus any negative alternative land use value within the landscape). Using ABF, this is *averaged across all positive features*, not just the maximum value. Inclusive and pluralist integration approaches using CAZ are presented in the main text, and all others are presented in Supplementary Figure 2 to Supplementary Figure 8. The following discussion considers similarities and differences between ABF and CAZ results.

## **Supplementary discussion**

### **Additive benefit prioritisation**

Since ABF averages across all features, it resulted in higher overall feature coverage but lower levels of complementarity between landscapes. Hence there was greater spatial similarity between the ABF viewpoint prioritisations than the CAZ prioritisations, with NEW and ECON prioritisations especially spatially correlated (Supplementary Figure 2). The greater convergence between viewpoints was due to ABF considering all landscape features, rather than the single highest weight\* (positive proportion) in CAZ. Due to these increased similarities, ABF viewpoint integration approaches were also more spatially similar compared to CAZ (Supplementary Figure 3 and Supplementary Figure 4), with a particular concentration within the south of England suggesting that this is an area with

potentially large gains in feature coverage, even if the most important landscapes for some features are not included.

Feature coverage was more consistent between the ABF integration approaches, and they provided a slightly higher mean feature coverage efficiency than CAZ (ABF 17% coverage efficiency range: 0.625-0.636; CAZ: 0.560-0.600; Supplementary Figure 5 to Supplementary Figure 8). For lower thresholds, minimum coverage efficiency was generally higher using ABF too (ABF 5% coverage efficiency range: 0.222-0.269; CAZ: 0.142-0.383). However, as the threshold rose CAZ minimum efficiency generally increased at a faster rate than ABF, and CAZ ultimately exceeded ABF for the pluralist and RANK approaches (ABF 30% coverage efficiency range: 0.424-0.458; CAZ: 0.376-0.545). This is illustrated by Supplementary Figure 5 and Supplementary Figure 6 (right hand panels), where ABF mainly outperforms CAZ at 5% area coverage (red columns) but not at 30% (dark blue columns), and some features may largely be 'missed' with the CAZ approach at 5% coverage if a single viewpoint is adopted. This reflects the fact that achieving multiple goals (satisfying multiple viewpoints and including many different features) is increasingly difficult at low coverage thresholds: CAZ priorities (aiming to include the very best examples of each feature included by a particular viewpoint) may be more difficult to reconcile than ABF (incorporating the places with the best mixture of features) when only a small percentage of the land is allocated to conservation. Nonetheless, the CAZ pluralist approach had relatively high minimum feature coverage efficiency for all area thresholds, ensuring that desired features (by any viewpoint) were not missed, even at low thresholds.

All ABF integration approaches resulted in high mean feature coverage efficiency and moderately high minimum efficiency. Hence, ABF could be considered a more inherently 'inclusive' prioritisation method in that the best combined-feature areas will be selected (most are well satisfied by any of the ABF integration approaches), but areas that are critically important for a single conservation feature may be disregarded (some individuals may be disappointed). Similarly CAZ could be considered a more 'pluralist' prioritisation method, in that the most important locations for each

feature and viewpoint are maintained, even if the solution is slightly less efficient overall. Both ABF and CAZ prioritisation methods could offer coherent conservation plans by integrating viewpoints, and the prioritisation method used should depend upon conservation objectives and spatial context. However, we focused on CAZ prioritisation in the main text, here, because CAZ combined with a pluralist approach generally resulted in the highest minimum coverage.

## 1 INTRODUCTION

2 In biodiversity conservation, individuals differ in the values they attribute to different  
3 conservation priorities (Sandbrook *et al.*, 2019; Bhola *et al.*, 2021). As in other applied  
4 disciplines, different perspectives often seem contradictory and even irreconcilable in  
5 planning decisions (Matulis and Moyer, 2017). Two different types of approach are  
6 commonly used to unify opposing viewpoints within conservation. Inclusive approaches seek  
7 to accommodate all perspectives, by building consensus and finding compromise between  
8 people holding different views, thus creating a single voice for conservation that carries more  
9 weight (Tallis and Lubchenco, 2014). In contrast, proponents of pluralist approaches contend  
10 that inclusive approaches reinforce current dominant perspectives and suppress marginal  
11 views (Matulis and Moyer, 2017) asserting that we need to find better ways to accept and  
12 engage with diverse perspectives on biodiversity and give voice to marginalised values  
13 (Pascual *et al.*, 2021). The risk is that a pluralist approach results in a divided, and thereby  
14 potentially unconvincing, voice for conservation. How to combine these different  
15 perspectives, in a way that ensures all viewpoints are represented but that nonetheless  
16 garners widespread support, is a pressing question for conservation and more widely in  
17 society (Pascual *et al.*, 2022).

18 Every individual viewpoint in conservation encompasses its own set of values and  
19 aims, and hence there are trade-offs when seeking to reconcile different viewpoints during  
20 policy implementation (McShane *et al.*, 2011). Each individual worldview (here 'viewpoint')  
21 in biodiversity conservation encompasses its own set of specific values and conservation  
22 priorities (Sandbrook *et al.*, 2019; Bhola *et al.*, 2021; Anderson *et al.*, 2022). As in other  
23 applied disciplines, different viewpoints can often seem contradictory and even  
24 irreconcilable in planning decisions (Matulis and Moyer, 2017), and hence there are trade-  
25 offs when seeking to reconcile different values during policy implementation (McShane *et*  
26 *al.*, 2011). This diversity of opinions must be addressed in planning so as to avoid socio-

27 environmental conflicts, perceived injustices and hence ineffective policy interventions  
28 (Barton *et al.*, 2022). How to combine different values through an equitable process that  
29 ensures all viewpoints are fairly represented, but that nonetheless garners widespread  
30 support, is a pressing question for conservation and more widely in society (Termansen *et*  
31 *al.*, 2022).

32 Two different types of approach are commonly used to unify opposing viewpoints  
33 within conservation. Inclusive approaches seek to accommodate all viewpoints by building  
34 consensus and finding compromise between people holding different views, thus creating a  
35 single voice for conservation that carries greater weight (Tallis and Lubchenco, 2014).  
36 However, by not explicitly acknowledging differences, inclusivity could stifle debate and  
37 reinforce current dominant viewpoints if ‘consensus’ represents the viewpoint that is most  
38 frequently articulated and voted for (Matulis and Moyer, 2017). Pluralist approaches, by  
39 contrast, assert that we need to find better ways to accept and engage with diverse  
40 viewpoints on biodiversity thereby giving voice to marginalised values and increasing equity  
41 (Pascual *et al.*, 2021). The risk is that a pluralist approach results in a divided, and thereby  
42 potentially unconvincing, voice for conservation that makes systematic and coordinated  
43 action impossible (Matulis and Moyer, 2017).

44 Viewpoints within the conservation community are often considered in terms of  
45 ‘traditional’ or ‘new’ conservation ~~(Matulis and Moyer, 2017).~~(Matulis and Moyer, 2017).  
46 ‘Traditional’ conservation follows an ecocentric viewpoint, conserving species diversity and  
47 natural habitats for their intrinsic value ~~(Soulé, 1985; Taylor *et al.*, 2020).~~(Soulé, 1985; Taylor  
48 *et al.*, 2020). It is often regarded as the antithesis of ‘new’ conservation, which follows a more  
49 anthropocentric viewpoint motivated by achieving conservation action through attaining  
50 economic and social benefit ~~(Marvier, 2014).~~(Marvier, 2014). However, this is a simplification  
51 of the diverse range of views on approaches to conservation. ~~Hence, the~~The Future of  
52 Conservation survey (~~<http://futureconservation.org>~~<http://futureconservation.org>) sought to

\_\_\_\_\_3  
\_\_\_\_\_3  
53 establish a framework to further categorise different viewpoints within conservation  
54 (<https://www.futureconservation.org/about-the-debate>). In  
55 <https://www.futureconservation.org/about-the-debate> but in reality, the views of  
56 conservation researchers and practitioners are spread over a continuum between and  
57 beyond these viewpoint groupings, with no clear 'camps' (~~Sandbrook et al.,~~  
58 ~~2019~~),([Sandbrook et al., 2019](#)), making it difficult to evaluate potential approaches against  
59 each other (~~Hunter Jr, Redford and Lindenmayer, 2014~~).([Hunter Jr, Redford and](#)  
60 [Lindenmayer, 2014](#)). It is not this work's aim to revisit debate about the relative merit of any  
61 conservation viewpoint. Rather, it accepts that there exists a breadth of  
62 ~~perspectives~~viewpoints that need to be reconciled during conservation policy development,  
63 whilst recognising that conservation is likely to be more successful if focused on common  
64 ground within the conservation community (~~Hunter Jr, Redford and Lindenmayer,~~  
65 ~~2014~~).([Hunter Jr, Redford and Lindenmayer, 2014](#)).

66 The important step of identifying and balancing stakeholder viewpoints is typically  
67 undertaken at the beginning of the planning process in an attempt to agree weightings or  
68 goals for different priorities. One way of addressing these issues in a structured way is  
69 through using a planning framework such as systematic conservation planning (SCP) (~~Pressey~~  
70 ~~and Bottrill, 2009; Watson et al., 2011~~),([Pressey and Bottrill, 2009; Watson et al., 2011](#)),  
71 which utilises network-scale and spatially explicit methods to inform important conservation  
72 planning decisions (~~Watson et al., 2011~~).([Watson et al., 2011](#)). SCP provides a way to  
73 incorporate these techniques in a robust and auditable process, incorporating the principle  
74 of complementarity to design an optimal network for a given ~~planning objective~~ (~~Margules~~  
75 ~~and Pressey, 2000; Wilson, Cabeza and Klein, 2009~~).set of planning objectives ([Margules and](#)  
76 [Pressey, 2000; Wilson, Cabeza and Klein, 2009](#)). As part of a full SCP implementation there is  
77 opportunity to build consensus between differing stakeholders' ~~perspectives~~viewpoints  
78 within the initial planning stages, but this is carried out primarily when conservation goals

79 are being set rather than the prioritisation stage. However, it is difficult to understand, at this  
80 early stage, whether different values translate into similar or distinct spatial priorities,  
81 potentially generating considerable debate about issues that have no practical impact.

82 An additional difficulty in building combined conservation plans is that social biases  
83 need to be accounted for in order to equitably reconcile viewpoints. ~~Decision support tools  
84 can be used to facilitate decision-making between stakeholders, preferably using structured  
85 methods such as multi-criteria decision analysis (MCDA), which assesses performance of  
86 alternative solutions across criteria, and explores trade-offs (Davies, Bryce and Redpath,  
87 2013; Esmail and Geneletti, 2018); or the Delphi technique, which iteratively and  
88 anonymously surveys a panel of experts or stakeholders (Mukherjee *et al.*, 2015, 2018).  
89 Although these tools provide powerful methods to inform decision-making through  
90 reconciling different perspectives, there are many social biases such as group-think and the  
91 dominance effect that cannot be overcome completely (Mukherjee *et al.*, 2018), and hence  
92 consensus on conservation action attained may risk not providing an equitably integrated  
93 solution. Integrating differing perspectives on how to carry out area-based conservation is  
94 vital in implementing a coherent and representative conservation framework (Bhola *et al.*,  
95 2021), and it is important to minimise these social biases when building consensus  
96 conservation plans.~~ Decision support tools can be used to facilitate decision-making between  
97 stakeholders, preferably using structured methods such as multi-criteria decision analysis  
98 (MCDA), which assesses performance of alternative solutions across criteria, and explores  
99 trade-offs (Davies, Bryce and Redpath, 2013; Esmail and Geneletti, 2018); or the Delphi  
100 technique, which iteratively and anonymously surveys a panel of experts or stakeholders  
101 (Mukherjee *et al.*, 2015, 2018). Although these tools provide powerful methods to inform  
102 decision-making through reconciling different viewpoints, there are many social biases such  
103 as group-think and dominance effects that cannot be overcome completely (Mukherjee *et*  
104 *al.*, 2018), and hence consensus on conservation action attained may risk not providing an



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105 equitably integrated solution. Integrating differing viewpoints on how to carry out area-  
106 based conservation is vital in implementing a coherent and representative conservation  
107 framework (Bhola *et al.*, 2021), and it is important to minimise these social biases when  
108 building consensus conservation plans.

109         Spatial prioritisation methods provide a tool to evaluate potential spatial synergies  
110 and trade-offs between different conservation goals and is often an important stage of SCP.  
111 Although typically focused on improving representativeness of species distributions within  
112 area-based conservation measures, spatial prioritisation can also be used to investigate the  
113 effect of including different socio-economic and ecosystem service (ES) information on  
114 spatial priorities to inform conservation policy (~~Naidoo *et al.*, 2008~~).(Naidoo *et al.*, 2008). In  
115 this way, trade-offs between protecting biodiversity and different societal or policy  
116 objectives can be assessed; such as carbon storage (~~Thomas *et al.*, 2013; Soto-Navarro *et al.*,~~  
117 ~~2020), or other ecosystem services and land use simultaneously (Anderson *et al.*, 2009;~~  
118 ~~Moilanen *et al.*, 2011; Fastré *et al.*, 2020).~~(Thomas *et al.*, 2013; Soto-Navarro *et al.*, 2020), or  
119 other ecosystem services and land use simultaneously (Anderson *et al.*, 2009; Moilanen *et*  
120 *al.*, 2011; Fastré *et al.*, 2020). Hence, although not previously used for this purpose, spatial  
121 prioritisation provides a potentially powerful quantitative tool to support spatially  
122 integrating different viewpoints in conservation. Here we implement spatial prioritisation  
123 combined with numeric aggregation methods, which avoids potential social biases, in order  
124 to fairly test different approaches to viewpoint integration that combine caricature  
125 viewpoints into single solutions.

126         Using the biological, environmental, social, and economic landscape conditions of  
127 Great Britain, we implement four caricature conservation viewpoint prioritisations  
128 ('traditional', 'new', 'local social instrumentalism', and 'international market ecocentrism';  
129 Table 1) at a national scale to illustrate the diverse range of perspectives within the  
130 conservation community. We assign weights to species distributions and other resources

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131 corresponding with the values of each viewpoint, and then carry out spatial prioritisation at  
132 a 10x10 km ('landscape') resolution for Britain. We expect prioritisations for each viewpoint  
133 to perform well at covering resource types (conservation ~~features~~feature layers) that are  
134 highly valued within that viewpoint, but they may overlook other features. For example,  
135 'non-traditional' methods may perform relatively poorly in representing species  
136 distributions. Finally, we develop both 'inclusive' and 'pluralist' approaches to evaluate the  
137 extent to which it is possible to reconcile and integrate the four viewpoints into a  
138 collaborative and coherent conservation plan.

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**140 METHODS**

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**142 Feature layers**

143 We searched for and collated social, economic and ecological spatial data that: (i)  
144 was publicly available for the entirety of Great Britain (GB), (ii) had a resolution of 10x10km  
145 scale or finer, and (iii) could be used to create informative ecosystem service (ES) or socio-  
146 environmental value layers. After the data search, a total of seven non-biological layers were  
147 found to be suitable and are detailed below. We defined the study area as GB, excluding  
148 islands smaller than 20km<sup>2</sup>.

149 ~~Five ES layers were adapted from published, publicly available resources: (i) carbon~~  
150 ~~storage (Bradley *et al.*, 2005; Henrys, Keith and Wood, 2016), (ii) agricultural/land value~~  
151 ~~(urban areas were assigned the highest 'agricultural value', indicating locations unsuitable~~  
152 ~~for terrestrial conservation) (Natural Resources Wales, 2019; The James Hutton Institute,~~  
153 ~~2019; Natural England, 2020), (iii) recreational services (Schägner *et al.*, 2016), (iv) flood~~  
154 ~~regulation (Stürck, Poortinga and Verburg, 2014), and (v) pollination services (Schulp,~~  
155 ~~Lautenbach and Verburg, 2014).~~

156 Five ES layers were adapted from published, publicly available resources: (i) carbon  
157 storage (Bradley *et al.*, 2005; Henrys, Keith and Wood, 2016), (ii) agricultural/land value  
158 (urban areas were assigned the highest 'agricultural value', indicating locations unsuitable  
159 for terrestrial conservation) (Natural Resources Wales, 2019; The James Hutton Institute,  
160 2019; Natural England, 2020), (iii) recreational services (Schägner *et al.*, 2016), (iv) flood  
161 regulation (Stürck, Poortinga and Verburg, 2014), and (v) pollination services (Schulp,  
162 Lautenbach and Verburg, 2014).

163 In addition, two socio-environmental value layers were included: (vi) wilderness  
164 ~~(Kuiters *et al.*, 2013)~~(Kuiters *et al.*, 2013) and (vii) landscape aesthetic value ~~(Van Zanten *et*~~

165 ~~et al., 2016).~~(Van Zanten et al., 2016). Full details of calculation, and data sources, of ES and  
166 socio-environmental value layers are provided in Supplementary Methods. All feature layers  
167 were rescaled to allow for direct comparison, and aggregated to 10x10 km (henceforth  
168 'landscape') resolution for the analysis (Figure 1). Only landscapes with majority>50% land  
169 cover were considered.

170 To incorporate biodiversity value, we included the interpolated distributions of 445  
171 priority species with distribution data available listed under Section 41 (Natural Environment  
172 and Rural Communities Act, 2006). Although ~~which~~the species that constitute 'priorities' may  
173 ~~change depending upon viewpoint~~differ between viewpoints, here we use the same species  
174 to allow for direct comparisons of different viewpoint prioritisation performance.  
175 Distribution data were provided by Butterfly Conservation (BC), Biological Records Centre  
176 (BRC) and breeding bird distributions by British Trust for Ornithology (BTO). Data were in the  
177 form of annual records between 2000 and 2014, except for two taxa where atlas data were  
178 only available for specific time periods (birds [2007-11] (~~Gillings et al., 2019~~), and ~~vascular~~  
179 ~~plants [2010-2017]~~)(Gillings et al., 2019), and vascular plants [2010-2017]). We used the raw  
180 distribution records for 156 species that were very localised ( $\leq 10$  presence records) and for  
181 a further 77 species which could not be modelled (most of which were also very rare, and for  
182 which models did not converge). For the remaining 212 species with over 10 presence  
183 records, we interpolated their range using Integrated Nested Laplace Approximations (INLA)  
184 in the *inlabru* package (~~Bachl et al., 2019~~)(Bachl et al., 2019). We used a joint model  
185 predicting distribution while accounting for recording effort (see Supplementary Methods  
186 for full details).

187

## 188 Viewpoint prioritisation

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190 Four conservation viewpoint caricatures were created that included 'traditional'  
191 conservation (TRAD), 'new' conservation (NEW), 'international market ecocentrism' (ECON),  
192 and 'local social instrumentalism' (SOC) (see Table 1 for definitions). Viewpoints were  
193 constructed by varying the weightings of different feature layers (such as biodiversity, carbon  
194 or landscape aesthetics), ~~representing~~indicating their relative importance, as this allows  
195 quantification of trade-offs when trying to reconcile viewpoints (Table 2). Weightings were  
196 not based upon wider consultation, and it must be emphasised that they are not designed to  
197 be accurate representations of the viewpoints of any group of conservationists. Instead, they  
198 capture an illustrative range of perspectives from across the conservation community.  
199 ~~Definitions of caricature viewpoints and integration approaches are provided in Table 1~~For  
200 real world implementation, viewpoint weightings could be developed with stakeholders and  
201 the public either through questionnaires and interviews to identify different individual  
202 viewpoints, or through workshops or forums to collect the viewpoint of a particular  
203 stakeholder group or organisation.

204 In order to identify the highest priority areas for each viewpoint, we carried out a  
205 spatial prioritisation using the software Zonation (~~Moilanen, 2007~~)(Moilanen, 2007) which  
206 produces a complementarity-based ranking of conservation priority over the study area. As  
207 it is important in joint species and ES prioritisations to ensure localised species are not  
208 overlooked (~~Thomas et al., 2013~~),(Thomas et al., 2013), we used 'core area zonation'  
209 (landscape value based upon the single highest value feature) to ensure complementarity  
210 was incorporated. Although we present 'core area zonation' prioritisations in the main text,  
211 we also tested viewpoint prioritisations and integration approaches using the alternative  
212 'additive benefit function' prioritisation algorithm (landscape value summed across all  
213 weighted features) within Zonation. The results from these analyses were qualitatively  
214 similar, and we do not consider them further in the main text, ~~reporting these analyses~~

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215 ~~in~~ although they are reported and discussed within the Supplementary ~~Methods, Figures and~~  
216 ~~Discussion~~ Materials.

217 We incorporated ES, biodiversity and socio-environmental values into the viewpoint  
218 prioritisations through weightings commensurate with each viewpoint (Table 2). Weights for  
219 feature layers were generally positive, representing a desirable resource to include, with the  
220 exception of agricultural value (negative weights), which represented an alternative land use  
221 to conservation. Species distributions were collectively considered a single biodiversity  
222 feature layer for weightings, so that each species received a weighting corresponding to  
223 (biodiversity weighting)/(number of species), but were included as separate feature layers  
224 within the prioritisation.

225 We considered each prioritisation individually and tested feature coverage for the  
226 top 5%, 10%, 17% [corresponding to the Aichi 2020 target ~~(CBD, 2010)~~ (CBD, 2010)] and 30%  
227 priority areas [corresponding to the first draft of the post-2020 global biodiversity  
228 framework ~~(CBD, 2021)~~ (CBD, 2021)]. Coverage of biodiversity was calculated as the mean  
229 species distribution proportion coverage. The distribution of each ES feature is likely to have  
230 a large effect on prioritisation ranks for each viewpoint, as the more concentrated a feature  
231 is, the larger its effect on the prioritisation. Here we rescaled each feature but did not  
232 normalise the distribution, doing so would ensure each feature had an equal effect on  
233 prioritisations, but may mean return on coverage would be artificially inflated.

234 We also investigated the similarity of ~~the~~ existing protected area ~~network~~ (PA)  
235 coverage in Britain to the different ~~viewpoints, expecting~~ viewpoint priority rankings of each  
236 landscape. We expected the existing PA network to match the ‘traditional’ viewpoint  
237 prioritisation most closely since the designation rationale for protected areas is typically to  
238 prioritise species and ecosystems representatively. We considered all Sites of Special

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239 Scientific Interest (SSSI) and National Nature Reserves (NNR) ([https://naturalengland-](https://naturalengland-defra.opendata.arcgis.com)  
240 [defra.opendata.arcgis.com](https://naturalengland-defra.opendata.arcgis.com)) as ‘protected areas’ (Supplementary Figure 1).

241

## 242 Viewpoint integration

243

244 Given that decision support tools risk being influenced by social biases of  
245 participants, we developed two novel numerical aggregation approaches to reconcile the  
246 individual viewpoints into single spatial conservation plans. ~~Firstly, an inclusive approach was~~  
247 ~~used. This; one representing an inclusive approach and one a pluralist approach (Table 1). It~~  
248 ~~is important to note that we tested these integration approaches using simulated caricature~~  
249 ~~viewpoints, whereas real-world applications are likely to present additional complexities (see~~  
250 ~~Discussion).~~

251 The inclusive approach produced an aggregate priority map by taking the individual  
252 viewpoint prioritisations (Figure 2), and summing the landscape priority ranks of each  
253 viewpoint (Eq. 1). This represents an integrated conservation solution generated through a  
254 vote counting method with equal weight given to each viewpoint.

255

$$256 \quad I_j = \sum_v r_{vj} \sum_v r_{vj}$$

257 Eq. 1

258 where  $I_j$  is the inclusive value  $I$  for landscape  $j$ ,  $r_{vj}$  is the priority rank for viewpoint  $v$  and  
259 landscape  $j$ .

260

261 However, as there are correlations between viewpoints in their weighting of  
262 individual feature layers, inclusive vote counting methods may result in combined priority  
263 areas that are simply shared by more similar viewpoints, and therefore under-represent the  
264 level of importance of other features valued by more distinctive viewpoints. Hence, we also

265 implemented a more equitable pluralist approach to integration accounting for correlation  
 266 between feature layer choice (Table 2), weighting by the distinctiveness of each viewpoint,  
 267 to ensure more marginal viewpoints were ~~more equitably represented~~ not overlooked.

268 For the pluralist approach we initially undertook a principle component analysis  
 269 (PCA) to partition the variance from viewpoint weightings of feature layers, ~~creating a~~  
 270 ~~number of (Table 2) into~~ principal components (PC) ~~which (Supplementary Table 1). These~~  
 271 ~~PCs~~ are linear combinations (eigenvectors) of the viewpoints ~~(Supplementary Table 1),~~  
 272 ~~which were then used to weight the viewpoint landscape priority ranks (Eq. 2) to calculate~~  
 273 ~~the pluralist landscape value.~~ The first PC is fitted in the direction that accounts for the  
 274 maximum variance of the ~~data~~ viewpoint weightings and further PCs, orthogonal to the  
 275 previous PCs, maximise the remaining variance. Thus PCs are the combinations of viewpoints  
 276 that explain the variance in weightings in the most efficient way. For each ~~PC~~ landscape, we  
 277 ~~then~~ multiplied the four viewpoint prioritisation ~~landscape~~ rankings by the ~~corresponding~~  
 278 ~~PC~~ associated viewpoint eigenvectors of the first PC and took the sum (dot product). ~~We) and~~  
 279 iteratively added viewpoint rank/PC dot product absolute values until the 'main' PC of each  
 280 viewpoint was included (Eq. 2), to ensure the distinctiveness of each viewpoint was  
 281 represented.

$$282 \quad P_j = \sum_{e=1}^{n_e} |r_{j_1 \times v} \cdot W_{e_{v \times 1}}| + \sum_{c=1}^{n_c} |r_{j_1 \times v} \cdot W_{c_{v \times 1}}|$$

283 Eq. 2

284 where  $P_j$  is the pluralist value  $P$  for landscape  $j$ ,  $r_j$  is a  $1 \times v$  matrix of viewpoint priority ranks  
 285 for landscape  $j$ , and  $W_c$  is the corresponding  $v \times 1$  eigenvector matrix from principal  
 286 component  $c$  of a viewpoint feature layer weightings PCA (Supplementary Table 1).  $n_c$  is the  
 287 smallest number of principal components where the highest PC loading for each viewpoint  
 288 can be included (i.e. for all viewpoints we found the PC with the highest loading for that  
 289 viewpoint, and included all PCs up to and including that viewpoint).



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291 We evaluated ~~viewpoint and viewpoint integration approach performance as~~ the  
292 *efficiency* with which feature layers were included ~~into~~(or for agricultural value, excluded)  
293 within each individual prioritisation: (Fig. 2) and aggregation approach (Fig. 3). Efficiency was  
294 calculated as the proportion of each feature ~~covered~~included (or for agricultural value,  
295 excluded) by prioritisations at each coverage threshold, compared to the maximum amount  
296 ~~of feature coverage~~ possible. ~~Mean~~The mean efficiency ~~between~~across feature layers was  
297 used as a measure of the overall success of a given approach~~optimality, and minimum~~  
298 ~~efficiency was used as~~. For each approach, we also highlighted the feature layer that had the  
299 lowest coverage, to represent a measure of how ~~equitably features~~equally feature layers  
300 were included: (i.e., how ‘disappointed’ would someone be for whom this was their top  
301 priority?).

302 In addition to the inclusive and pluralist approaches listed in the main text, we also  
303 tested two other ~~integration~~ approaches to integrating viewpoints. Both performed less well  
304 under ‘core area zonation’ for most thresholds, and so are not discussed further here. See  
305 Supplementary Methods, Figures and Discussion for further details on these other  
306 approaches (and ‘additive benefit function’ prioritisations).

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## 308 RESULTS

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310 The viewpoint prioritisations selected different landscape priorities based upon their  
311 valued features (Figure 2). The ‘traditional’ conservation viewpoint priorities had the highest  
312 average proportion coverage of species distributions, primarily concentrated in  
313 NWnorthwest Scotland and scattered landscapes in the south of England. Conversely ‘local

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314 social instrumentalism' spatial priorities were focused in landscapes in England close to large  
315 conurbations, especially London, maximising recreational value but resulting in lower  
316 exclusion of agricultural land; as well as landscapes in [North](#) England, which delivered  
317 landscape aesthetic value and flood protection services. 'International market ecocentrism'  
318 priorities almost exclusively occurred in Scotland, and upland areas in Wales and northern  
319 England, driven by positive selection for carbon storage and avoiding the opportunity costs  
320 of more southerly productive farmland. The 'new' conservation spatial prioritisation selected  
321 landscapes appearing in both the 'international market ecocentrism' and 'local social  
322 instrumentalism' viewpoints, due the more balanced weightings across feature layers. These  
323 landscapes were primarily located in Scotland, upland areas in [North](#) England, and  
324 [SEsoutheast](#) England close to London.

325 We integrated the four viewpoints into single conservation strategies using two  
326 approaches (Figure 3). The inclusive approach selected landscapes in Scotland, upland Wales,  
327 [North](#) England, and [SEsoutheast](#) England. The pluralist approach contained similar priority  
328 areas, but higher priority landscapes were more concentrated in SE England. The pluralist  
329 approach had lower coverage of carbon and exclusion of agricultural value, but recreational  
330 value coverage was much higher than the inclusive approach (feature coverage, Figure 3;  
331 efficiency, Supplementary Figure 7). Species distributions received the best coverage through  
332 the 'traditional' viewpoint. The inclusive and pluralist integration approaches had higher  
333 species representation than the other viewpoints, although coverage was lower than the  
334 'traditional' viewpoint (mean species distribution proportion coverage at 17% coverage  
335 threshold: TRAD 40.3%, NEW 17.9%, ECON 8.6%, SOC 25.9%, Inclusive 27.2%, Pluralist  
336 28.8%); [other thresholds: Fig. 2-3](#)). The existing protected area network (Supplementary  
337 Figure 1) matched 'new' and 'market ecocentrism' [viewpoint prioritisation ranks](#)  
338 more closely than 'traditional' (TRAD  $\rho = 0.379$ , NEW  $\rho = 0.423$ , ECON  $\rho = 0.413$ , SOC  $\rho =$

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339 0.068), which was contrary to our expectation given that the rationale and goals for  
340 identifying potential SSSIs and NNRs closely align with a traditional approach.

341 We found that both integration approaches had similar mean feature coverage  
342 efficiency (17% coverage threshold: inclusive 60.0%, pluralist 59.0%; other thresholds: Figure  
343 4) indicating similar overall optimality. However, the pluralist approach had higher minimum  
344 coverage efficiency for all thresholds, meaning features were included more equitably  
345 (17% coverage threshold: inclusive 27.6%, pluralist 42.3%; other thresholds: Figure 4).

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## 347 DISCUSSION

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349 ~~Different conservation viewpoints have passionate proponents and opponents~~  
350 ~~(Kareiva and Marvier, 2012; Noss et al., 2013; Soulé, 2013; Doak et al., 2014) with seemingly~~  
351 ~~irreconcilable differences.~~ Different conservation viewpoints have passionate proponents  
352 and opponents (Kareiva and Marvier, 2012; Noss et al., 2013; Soulé, 2013; Doak et al., 2014;  
353 Sandbrook et al., 2019) with differences that are difficult to balance within single coherent  
354 conservation plans. As expected we found that each of the four caricature viewpoints  
355 spatially prioritised different landscapes, depending on the values held, and resulted in  
356 different levels of feature coverage. ~~However, we~~ We then aggregated viewpoint priorities  
357 ~~for the first time~~ by implementing inclusive and pluralist integration approaches, and we  
358 found that it is feasible to reconcile different viewpoint spatial priorities in a transparent  
359 manner. Although inclusiveness is typically associated with consensus building, our results  
360 demonstrate that ~~through~~ applying pluralism approaches to systematic conservation  
361 planning methods, ~~a can produce~~ conservation plan can be produced plans that ~~is~~ are just as  
362 coherent.

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363 By accounting for conceptual ~~similarity~~similarities between viewpoints within a  
364 pluralist approach, similar viewpoints (which generate correlated spatial priorities) were  
365 prevented from ‘crowding out’ more marginal viewpoints. ~~Here this included ‘local social~~  
366 ~~instrumentalism’ and, to a lesser extent, ‘traditional’ conservation.~~By preventing dominance  
367 by certain perspectives, the pluralist approach incorporated the most important locations for  
368 each viewpoint into the highest spatial prioritisation rankings (Figs. 1-3). The main difference  
369 between approaches was that the pluralist approach efficiently incorporated higher  
370 recreational value, concentrated around large conurbations, with minimal loss of other  
371 features. Although the pluralist approach performed less well for some features than the  
372 inclusive approach, overall it included features more equitablyequally while maintaining  
373 similar mean coverage efficiency. This shows, at least spatially, that a coherent conservation  
374 plan can be created while also representing potentially marginalised viewpoints.

375 ~~Importantly this approach provides a decision support tool and not an ultimate~~  
376 ~~decision. SCP involves stakeholders through an interactive process and this is just as~~  
377 ~~important when implementing a numeric approach to reconciling viewpoints. In fact,~~  
378 ~~additional transparency in the process is achieved by developing the separate viewpoints into~~  
379 ~~independent spatial priorities, enabling advocates of any particular viewpoint to compare~~  
380 ~~the spatial consequences of their viewpoint with others. The reconciliation process is then~~  
381 ~~numerical, and each actor can compare the combined solution with that of their own initial~~  
382 ~~viewpoint. The objective integrated solution maps developed here can be shown to~~  
383 ~~stakeholders to present one method of equitably including different viewpoints, leading to a~~  
384 ~~more informed final decision. This could potentially be achieved by incorporating the~~  
385 ~~integrated solution maps into decision support tools to assist balancing different~~  
386 ~~perspectives, for example either through iteratively presenting integrated prioritisations to~~  
387 ~~participants within the Delphi technique (Mukherjee *et al.*, 2015), or exploring trade-offs as~~  
388 ~~part of a broader MCDA approach (Esmail and Geneletti, 2018). This contrasts with situations~~

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389 ~~where debates to reconcile viewpoints take place prior to the analysis, where consensus is~~  
390 ~~generated by discussion rather than formal analytical reconciliation.~~

391 ~~Although we demonstrate how different viewpoints can be reconciled, ultimately~~  
392 ~~any conservation plan can only be as equitable as the range of stakeholders included within~~  
393 ~~the planning process. Our approach is generalisable beyond conservation planning to many~~  
394 ~~situations where perspectives need spatial reconciliation, and could incorporate a larger~~  
395 ~~number of viewpoints, for example through stakeholder questionnaires or discussion fora.~~  
396 ~~Here we use a small number of caricature viewpoint weightings based on the conservation~~  
397 ~~community, but a pluralist approach could also be used to incorporate many more~~  
398 ~~viewpoints from the wider public, sampled through workshops and surveys (Rust *et al.*,~~  
399 ~~2021), or choice experiments (Badura *et al.*, 2020). Local stakeholder engagement, for~~  
400 ~~example through local partnerships and public participation, can be especially overlooked~~  
401 ~~within conservation planning, but is also important to ensure a protected area network that~~  
402 ~~delivers for all (Blicharska *et al.*, 2016; Sterling *et al.*, 2017). Equally some priorities, and even~~  
403 ~~entire viewpoints, are difficult or impossible to quantify spatially (Wyborn and Evans, 2021),~~  
404 ~~and these non-quantifiable cultural and contextual values will still need to be incorporated~~  
405 ~~within the process, especially when translating global or national conservation plans into~~  
406 ~~local priority setting (Chaplin-Kramer *et al.*, 2021; Fleischman *et al.*, 2022; Strassburg *et al.*,~~  
407 ~~2022).~~

408 Our approach is generalisable beyond conservation planning to many situations  
409 where perspectives need spatial reconciliation. Here we used a small number of caricature  
410 viewpoint weightings based on the conservation community to assess different integration  
411 methods, but applying these methods in a real-world setting would inevitably present  
412 additional complexities in reaching a compromise solution (Sandbrook *et al.*, 2019; Barton *et*  
413 *al.*, 2022). Depending upon the overall planning objectives, in real world situations, these  
414 would likely include involvement of a large number of participants. Many stakeholder

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415 viewpoints could be included through questionnaires and discussion fora, and involve the  
416 wider public, sampled through workshops and surveys (Rust *et al.*, 2021), or choice  
417 experiments (Badura *et al.*, 2020), but taking care to minimise social biases if determining  
418 viewpoints through group activities such as discussions and workshops.

419 Reconciling a large number of viewpoints is likely to require additional feature layers  
420 that may have to be co-developed with stakeholders, and some stakeholders may have  
421 unique interests in particular features. There are many possible conservation valuation  
422 methods and processes that can be used to produce feature layers, including different ES and  
423 biodiversity modelling techniques, participatory mapping, and benefit transfer amongst  
424 others (Termansen *et al.*, 2022). However some priorities, and even entire viewpoints, may  
425 be difficult or impossible to quantify (Wyborn and Evans, 2021), and these non-quantifiable  
426 cultural and contextual values will still need to be incorporated within the process (Chaplin-  
427 Kramer *et al.*, 2021; Fleischman *et al.*, 2022; Strassburg *et al.*, 2022). Therefore, it may be  
428 difficult to entirely reconcile complex values of a large number of stakeholders using solely  
429 quantitative methods, but this could be at least partially redressed through stakeholder  
430 engagement by situating the approach within a systematic planning framework (SCP).

431 SCP involves stakeholders through an interactive process and this is important when  
432 implementing both numeric and non-numeric approaches to reconciling viewpoints.  
433 Additional transparency in the process is achieved by developing the separate viewpoints  
434 into independent spatial priorities, enabling advocates of any particular viewpoint to  
435 compare the spatial consequences of their viewpoint with others. As the reconciliation  
436 process is numerical, each actor can compare alternative possible integrated solutions with  
437 that of their own initial viewpoint, leading to a more informed final decision. Decision-  
438 support tools could assist the balancing of different viewpoints by, for example, iteratively  
439 presenting integrated prioritisations to participants within the Delphi technique (Mukherjee  
440 *et al.*, 2015), or exploring trade-offs as part of a broader multi-criteria decision analysis

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441 (MCDA) approach (Esmail and Geneletti, 2018). This contrasts with situations where debates  
442 to reconcile viewpoints take place prior to the analysis, where consensus is generated by  
443 discussion rather than formal analytical reconciliation.

444 Importantly, this approach provides a decision-support tool and **not** an ultimate  
445 decision. Although we demonstrate how different viewpoints can be reconciled, ultimately  
446 the equity of any conservation plan depends upon the representation of stakeholders  
447 included within the planning process. Local stakeholder engagement, for example through  
448 local partnerships and public participation, can be especially overlooked within conservation  
449 planning, but is also important to ensure a protected area network that delivers for all  
450 (Blicharska *et al.*, 2016; Sterling *et al.*, 2017; Barton *et al.*, 2022).

451 The British protected area network was designed primarily to protect species and  
452 habitats in a representative way, and we expected more spatial overlap with a prioritisation  
453 based on the values of conserving species diversity. The suite of notified sites might be  
454 inefficient from the ‘traditional’ viewpoint for several reasons. Although the rationale behind  
455 the initial network designation and subsequent expansion may have been ‘traditional’, it may  
456 be that sites were not identified optimally through the notification process in terms of  
457 species representation. Additionally, notification will have depended not only on the quality  
458 of feature, but also on other conservation planning considerations such as land ownership  
459 and local socio-economic context. For example large SSSIs are primarily in upland areas,  
460 because less intensive land management occurred here in the past, allowing more semi-  
461 natural habitat to persist; in contrast to the lowlands where much smaller fragments of  
462 habitat remained to be protected (Bailey *et al.*, 2022). (Bailey *et al.*, 2022).

463 This work focuses on reconciling existing conservation feature values, not the  
464 establishment opportunities for potential future gains in feature values. Using carbon storage  
465 as an example, given its importance as a likely future driver for land use and management

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20  
466 policy (~~Committee on Climate Change, 2020~~),(Committee on Climate Change, 2020), these  
467 two distinct approaches are important: protecting and restoring existing high carbon  
468 habitats, particularly peatlands; and increasing carbon sequestration through the creation of  
469 new habitats, particularly woodlands (~~Gregg et al., 2021~~),(Gregg et al., 2021). Our approach  
470 has only taken account of the first of these, but a conservation strategy using a combination  
471 of protecting existing high-value landscapes, and implementing habitat enhancement or  
472 creation in others is needed for both biodiversity and other ecosystem commitments (~~Soto-~~  
473 ~~Navarro et al., 2020~~),(Soto-Navarro et al., 2020).

474         Within each landscape, different types of action will be required depending on what  
475 is important and the local land use context, considering that the distributions of ecosystem  
476 carbon, biodiversity value and other ecosystem services may be positively correlated in some  
477 landscapes but negatively so in others (~~Anderson et al., 2009~~),(Anderson et al., 2009). For  
478 example, if a low intensity agricultural landscape is prioritised for carbon storage, flood  
479 prevention or biodiversity, then enhanced protection and additional habitat management to  
480 further deliver on these ecosystem features may be implemented. However, strictly  
481 protected areas for biodiversity are unlikely to be the method to best incorporate all  
482 features, especially those valued by critical social science. Thus, other non-statutory area-  
483 based conservation measures may be needed to deliver for aspects such as human well-  
484 being. Similarly, other national schemes, such as tree planting, can also have hugely varying  
485 outcomes depending upon the spatial distribution of implementation (~~UNEP-WCMC and~~  
486 ~~LWEC, 2014~~),(UNEP-WCMC and LWEC, 2014), and these could also be considered within a  
487 pluralist framework.

488         As well as balancing differing viewpoints on existing resource protection, it is  
489 important to consider future expected changes in landscape feature values due to climate  
490 change within any implemented conservation plan (~~Bateman et al., 2013~~). Similarly,  
491 ~~in~~(Bateman et al., 2013). In addition to conservation feature values changing over time;



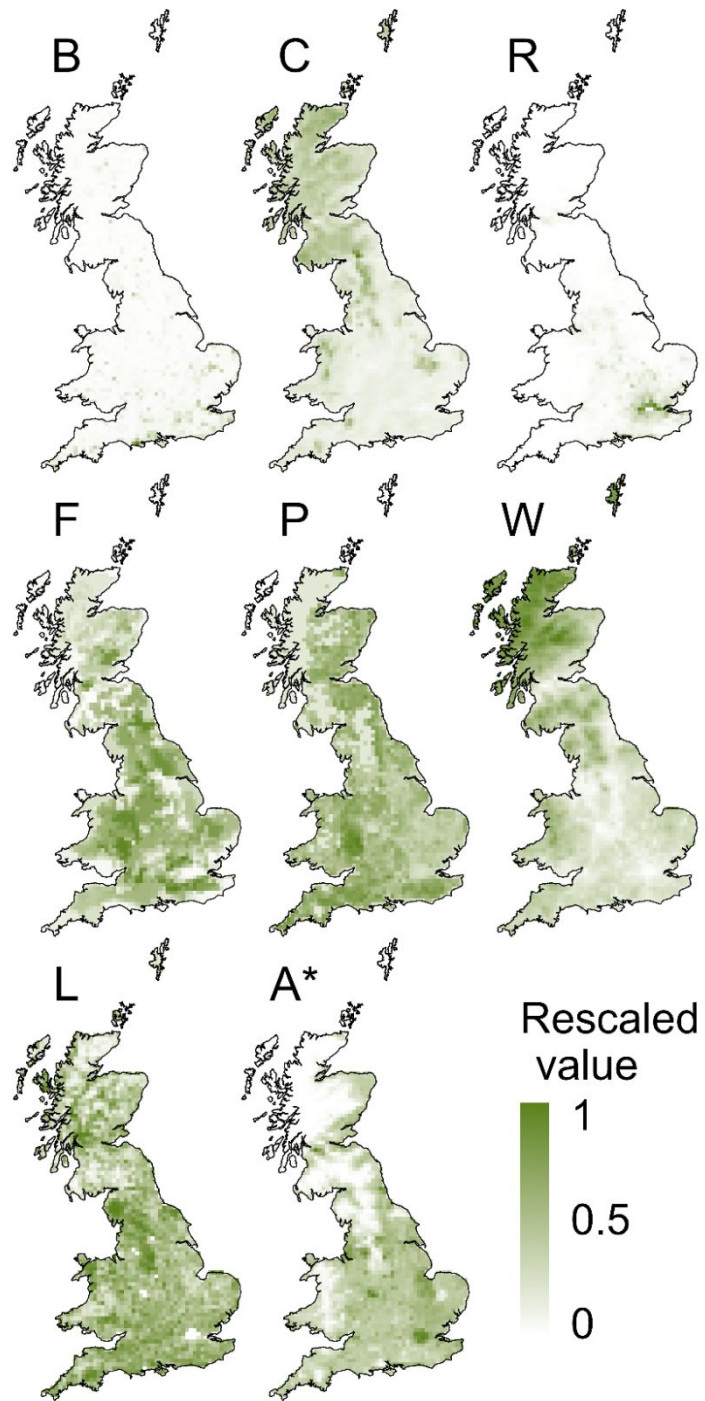
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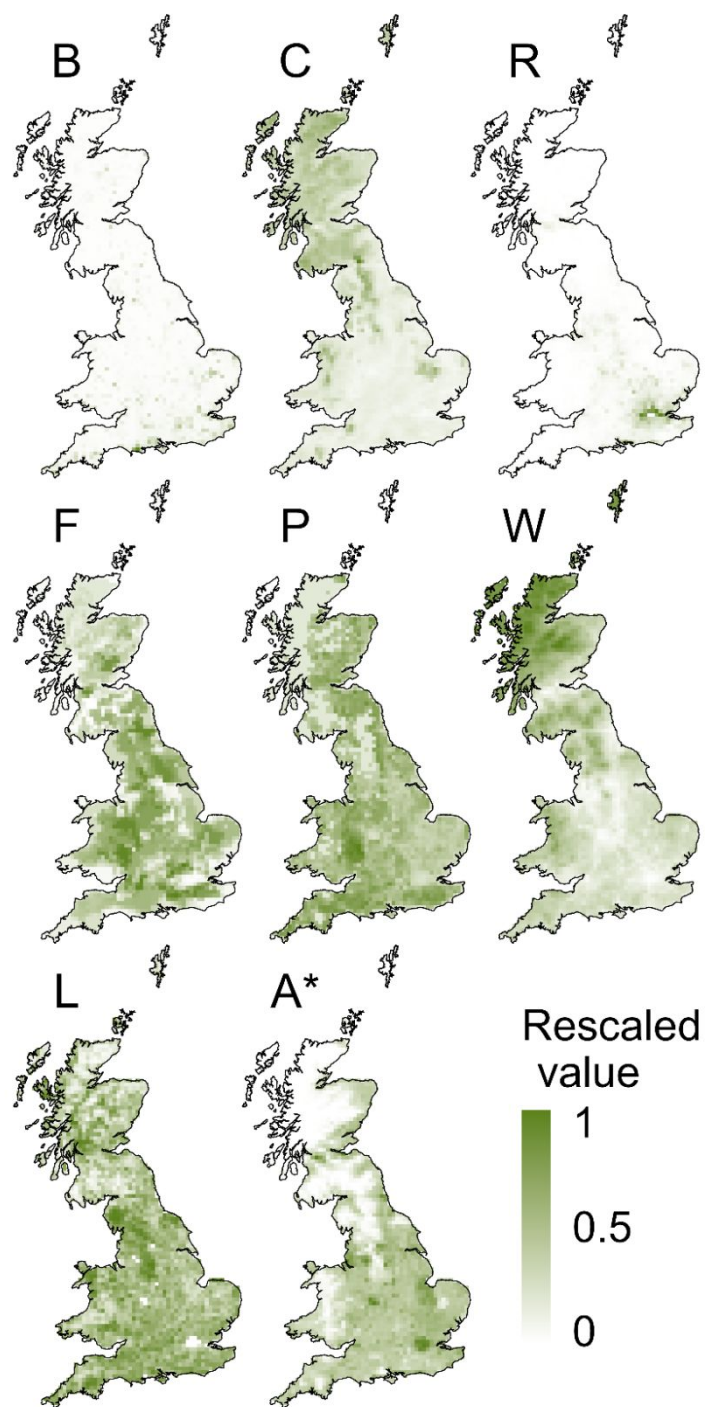
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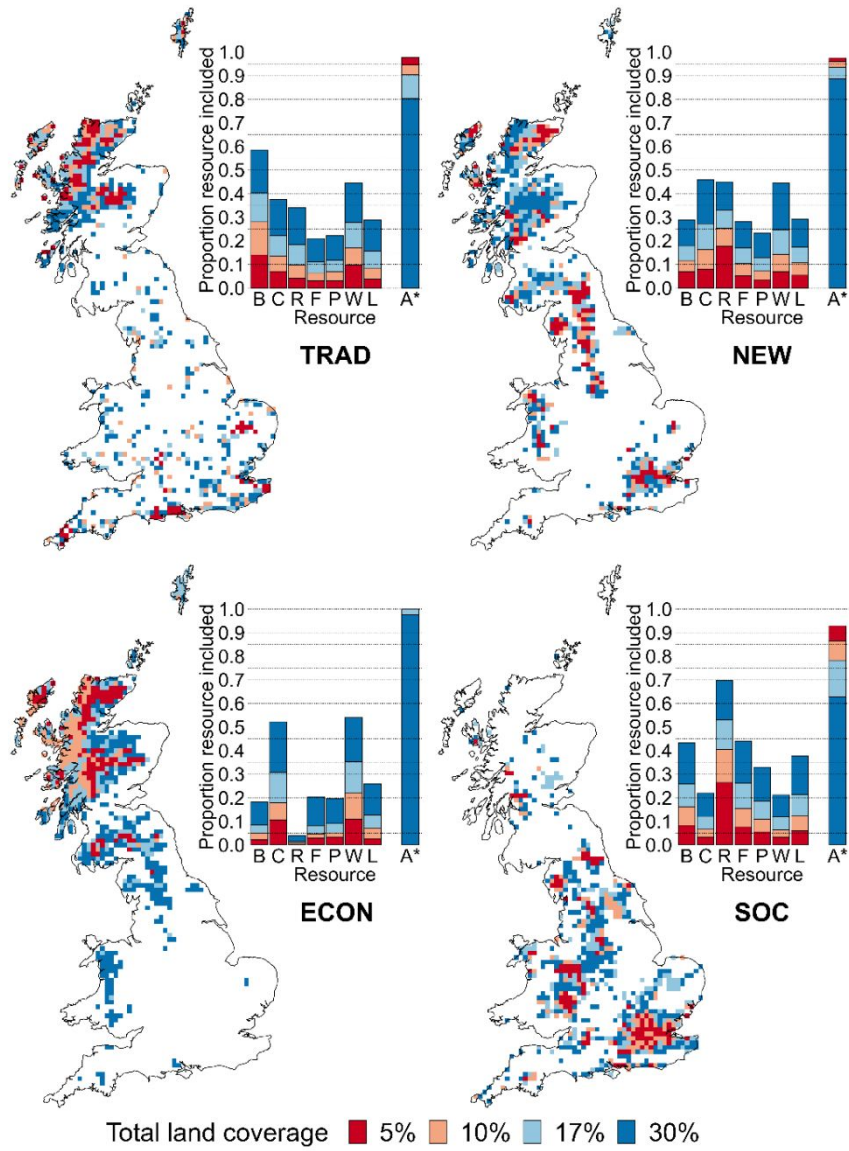
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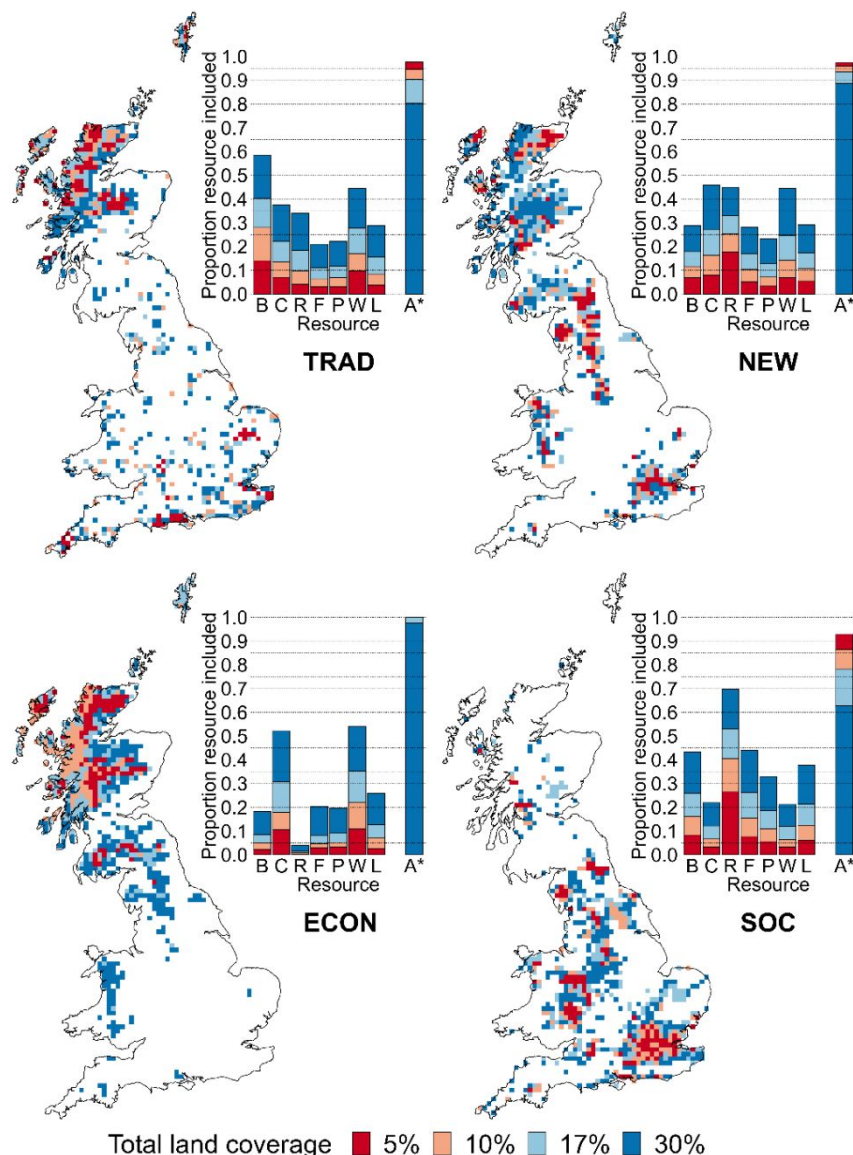
492 conservation perspectives, the needs of society, and value systems themselves change and  
493 develop (~~Mace, 2014~~), (~~Mace, 2014; Anderson et al., 2022~~), and so joint conservation plans  
494 have to be re-evaluated periodically. However, whilst the weight attached to different  
495 conservation objectives will inevitably change, including a broad range of benefits in  
496 conservation planning will remain important. ~~In developing the post-2020 global biodiversity~~  
497 ~~framework (CBD, 2021)~~, In developing the post-2020 global biodiversity framework (CBD,  
498 2021), it is vital to acknowledge and carefully consider how to equitably integrate different  
499 viewpoints on how to implement area-based conservation.





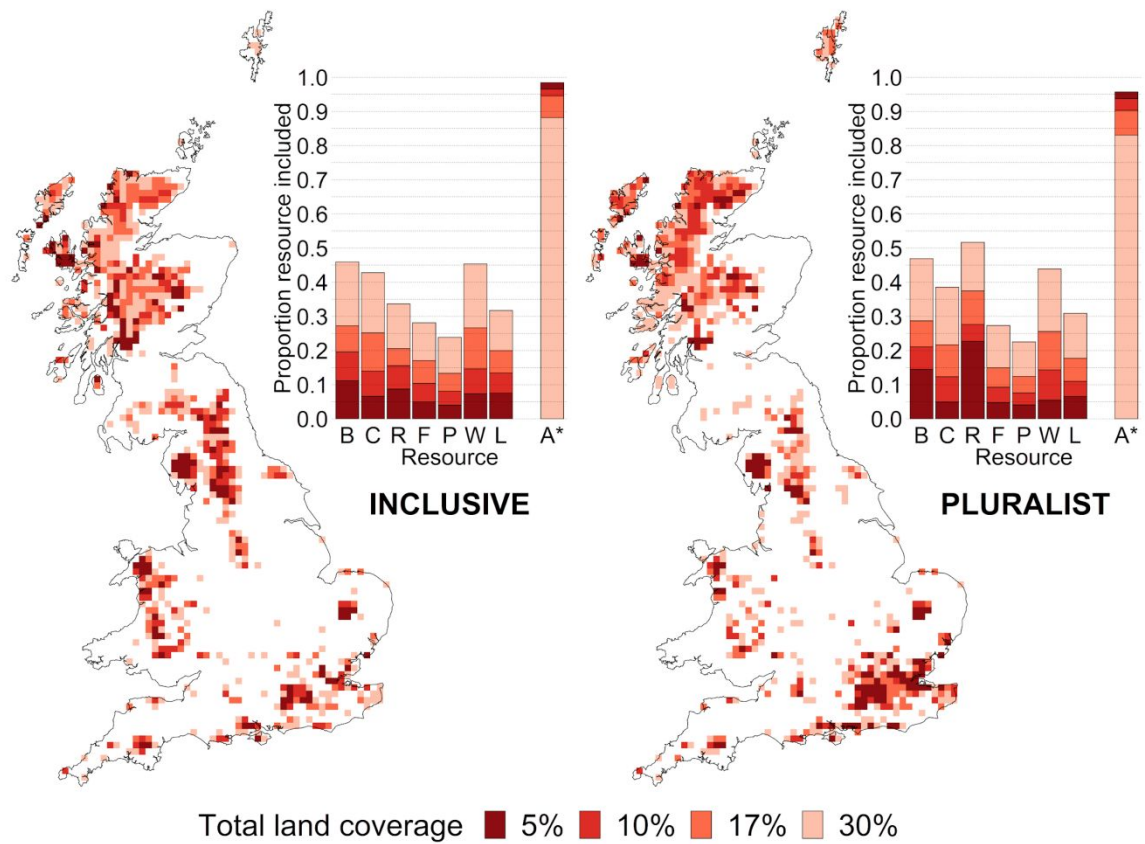
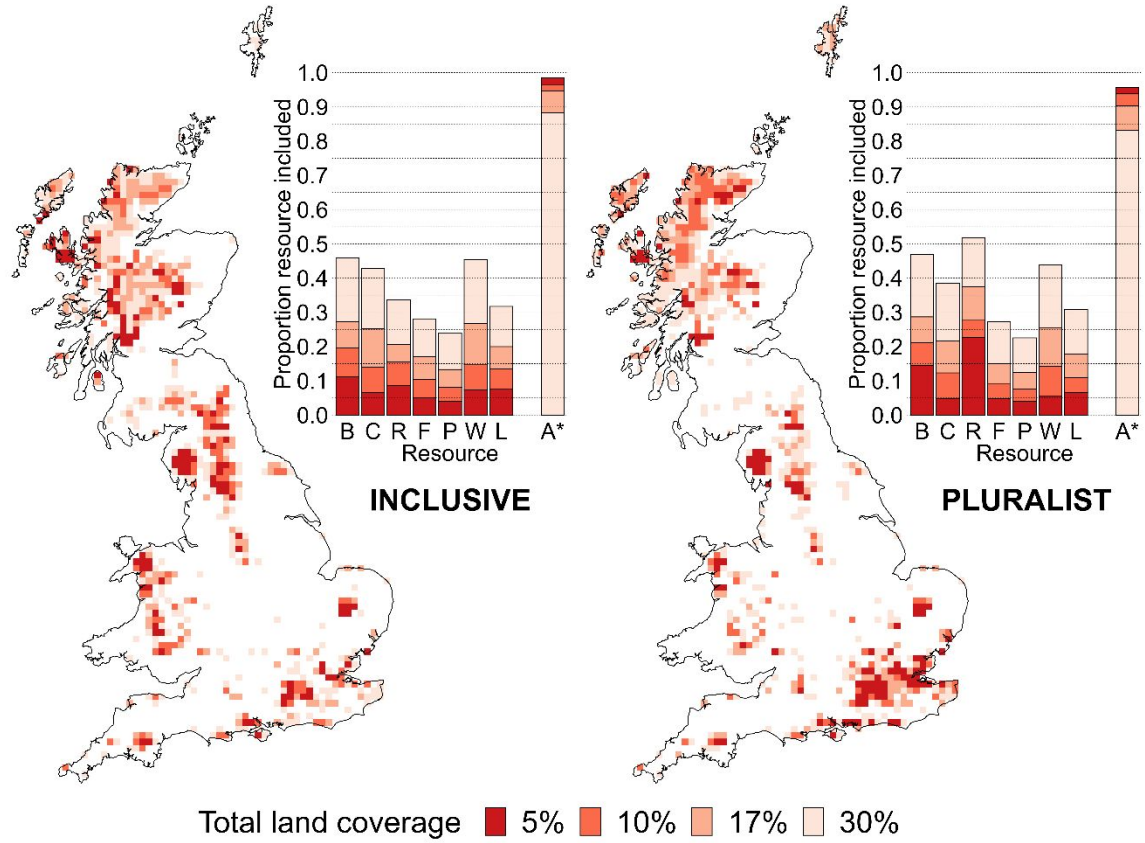
**Figure 1** Rescaled ecosystem service, biodiversity and socio-environmental value feature layers included within the analysis including; mean priority species distribution proportion coverage (B), carbon storage (C), recreation (R), flood regulation (F), pollination (P), wilderness (W), landscape aesthetic value (L) and agricultural/land value (A\*).



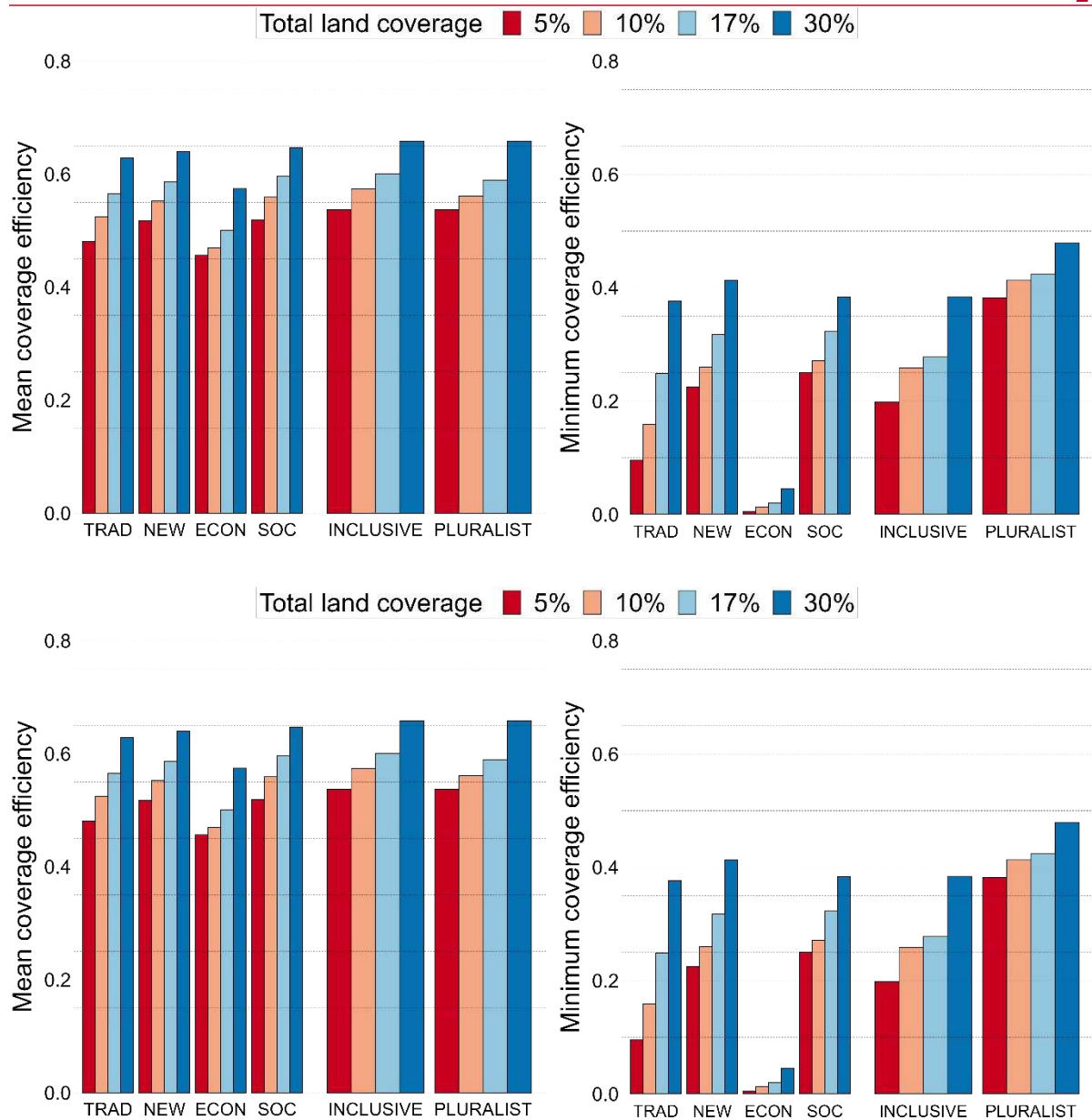


**Figure 2** Feature coverage using spatial prioritisation for each of the four viewpoints; TRAD – ‘traditional’, NEW – ‘new’ conservation, ECON – ‘international market ecocentrism’, SOC – ‘local social instrumentalism’. Maps indicate the priority areas for different coverage thresholds, and corresponding bar plots present proportion of feature included for each, including mean priority species distribution proportion coverage (B), carbon storage (C), recreation (R), flood regulation (F), pollination (P), wilderness (W), landscape aesthetic value (L) and agricultural/land value (A\*). A\* indicates the only negative weighting where proportion excluded, not included, is shown and so higher land coverage results in lower proportion excluded.





**Figure 3** Spatially aggregating the four conservation viewpoint priorities (TRAD – ‘traditional’, NEW – ‘new’ conservation, ECON – ‘international market ecocentrism’, SOC – ‘local social instrumentalism’) using inclusive (vote counting) and pluralist (accounting for distinctiveness) integration methods. Maps indicate the priority areas for different coverage thresholds, and corresponding bar plots present proportion of feature included for each, including mean priority species distribution proportion coverage (B), carbon storage (C), recreation (R), flood regulation (F), pollination (P), wilderness (W), landscape aesthetic value (L) and agricultural/land value (A\*). A\* indicates the only negative weighting where proportion excluded, not included, is shown and so higher land coverage results in lower proportion excluded.



**Figure 4** Mean and minimum feature coverage efficiency of viewpoint prioritisation performance (TRAD – ‘traditional’, NEW – ‘new’ conservation, ECON – ‘international market ecocentrism’, SOC – ‘local social instrumentalism’) and integration approach (inclusive and pluralist conservation). Efficiency is calculated as the proportion of a feature covered by a prioritisation for each land coverage threshold (5%, 10%, 17%, and 30%), compared to the maximum possible if only that feature was prioritised. Features included are mean priority species distribution proportion coverage (B), carbon storage (C), recreation (R), flood regulation (F), pollination (P), wilderness (W), landscape aesthetic value (L) and agricultural/land value (A\*). The left-hand panel shows the mean



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performance across all resource types (conservation features), whereas the right-hand panel shows the efficiency of the feature that is least well covered by a particular approach. Inclusive and pluralist approaches perform similarly, but pluralism has a higher minimum feature coverage threshold.

**Table 1** Definitions of caricature viewpoints and viewpoint integration approaches.

<b>Conservation viewpoint</b>	A personal perspective that determines how nature is valued, and how to best conserve it. This analysis uses four arbitrary caricature conservation viewpoints to analyse approaches to viewpoint integration.
'Traditional' (TRAD)	<p>Ecocentric viewpoint, aiming to conserve species diversity and natural habitats for their intrinsic value and for their ability to regulate ecosystem services. Intrinsic value is ascribed to biotic diversity and ecological complexity, with a preference for 'natural' systems. Adapted from Soulé (1985).</p> <p>Weightings: species distributions and wilderness.</p>
'New' (NEW)	<p>Anthropocentric viewpoint, motivated by achieving conservation action through attaining economic and social benefit. Seeks to conserve biodiversity in human-modified as well as 'natural' landscapes, whilst also maximising human well-being and economic objectives. Adapted from Marvier (2014).</p> <p>Weightings: widest scope of the four viewpoints, including species and all economic and social value data, apart from wilderness.</p>
International market ecocentrism (ECON)	<p>Utilises capitalist economic arguments to deliver ecocentric conservation, but ignores human well-being and local benefits. Aims to protect intrinsic ecological value over a large area, typically 30-50% of land. This is achieved by employing a free market approach to resource extraction on the remaining land, with the view that this would maximise profit to resource consumption efficiency, and hence protect the 'spared' land. Adapted from Wilson (2016).</p> <p>Weightings: agricultural value (avoid) and related pollination service flow, as well as carbon storage and species distributions.</p>
Local social instrumentalism (SOC)	<p>Favours prioritising conservation benefitting human well-being at the local scale, but opposed to intrinsic value of nature arguments, economic objectives, and links with capitalism and corporations. Adapted from 'social instrumentalism' in Matulis and Moyer (2017).</p> <p>Weightings: ecosystem services that benefit the local population, i.e. flood prevention and recreation, as well as landscapes that are important to people, and a lower weighting for species distributions.</p>
<b>Viewpoint integration approach</b>	Numeric aggregation methods to spatially reconcile differences between individual viewpoints into a single, coherent conservation plan.

Inclusive	Seek to embrace and bring together all perspectives, by building consensus and reducing disputes between people holding different views, and creating a single voice for conservation that is more unified, and hence carries more weight (Tallis & Lubchenco 2014). Here we implement this using an additive vote counting formula.
Pluralist	Accept and engage with diverse perspectives on biodiversity conservation, and give voice to marginalised values and views (Pascual et al. 2021). This is implemented by accounting for similarity between viewpoints and upweighting more distinct viewpoints.

**Table 2** Weightings for feature layers included within each of the four conservation viewpoints.

<i>Feature</i>	<b>Traditional conservation (TRAD)</b>	<b>'New' conservation (NEW)</b>	<b>International market ecocentrism (ECON)</b>	<b>Local social instrumentalism (SOC)</b>
Biodiversity (B)	1	1	1	0.5
Carbon (C)	0	1	1	0
Number of visits to recreation space (R)	0	1	0	1
Flood regulation (F)	0	1	0	1
Pollinator services (P)	0	0.5	0.5	0
Wilderness (W)	0.25	0	0	0
Landscape aesthetic value (L)	0	1	0	1
Agricultural land classification (A*)	0	-0.5	-1	0

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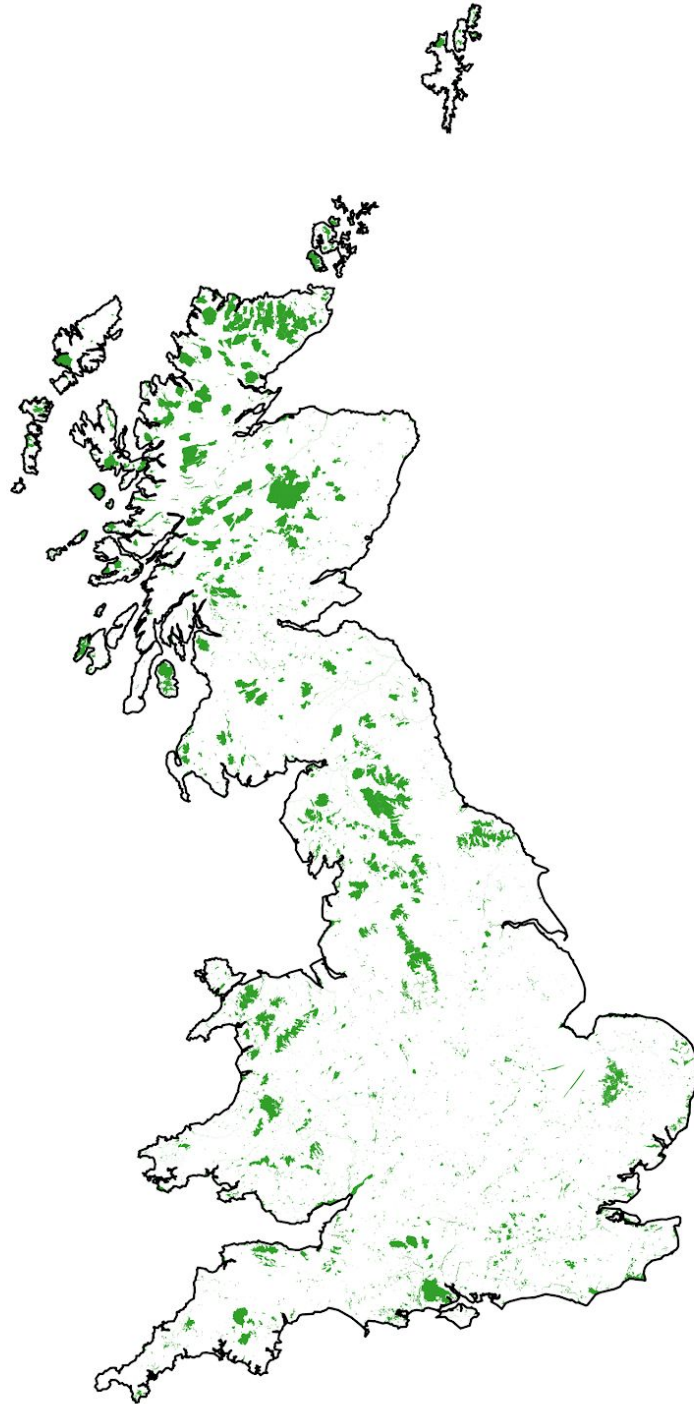
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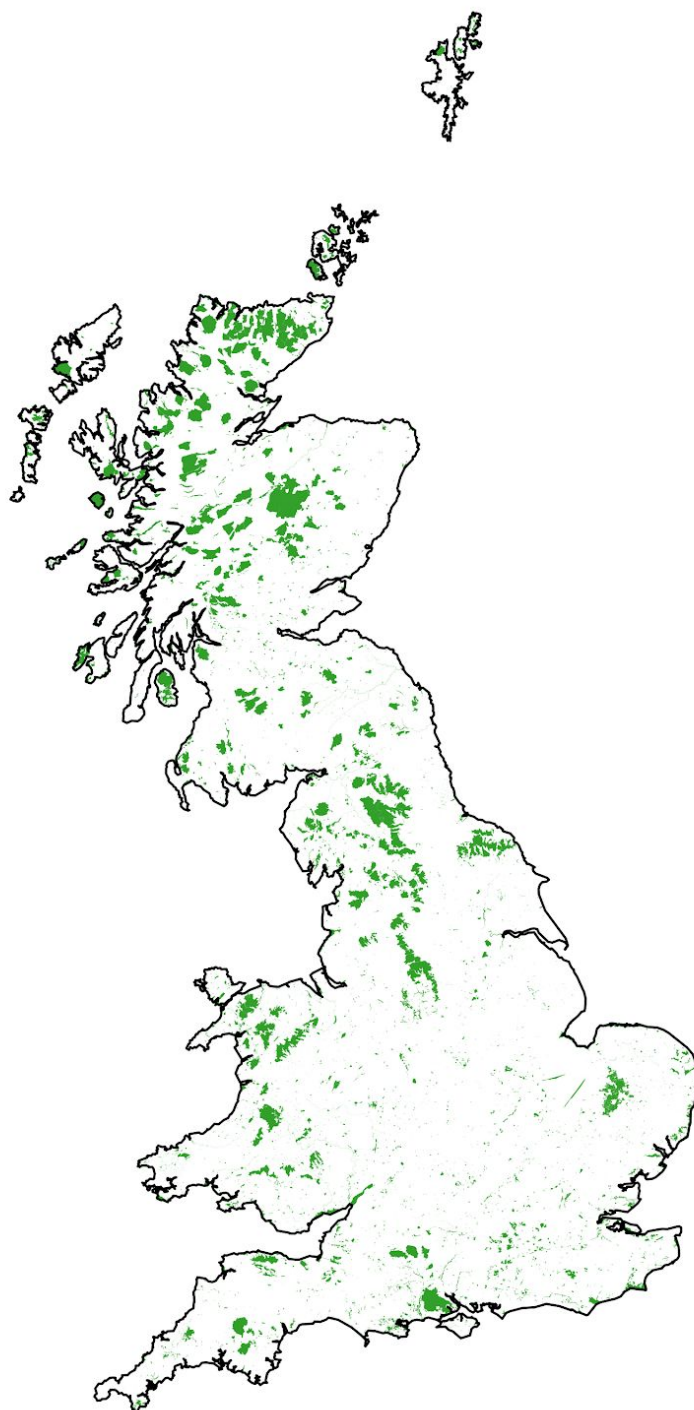
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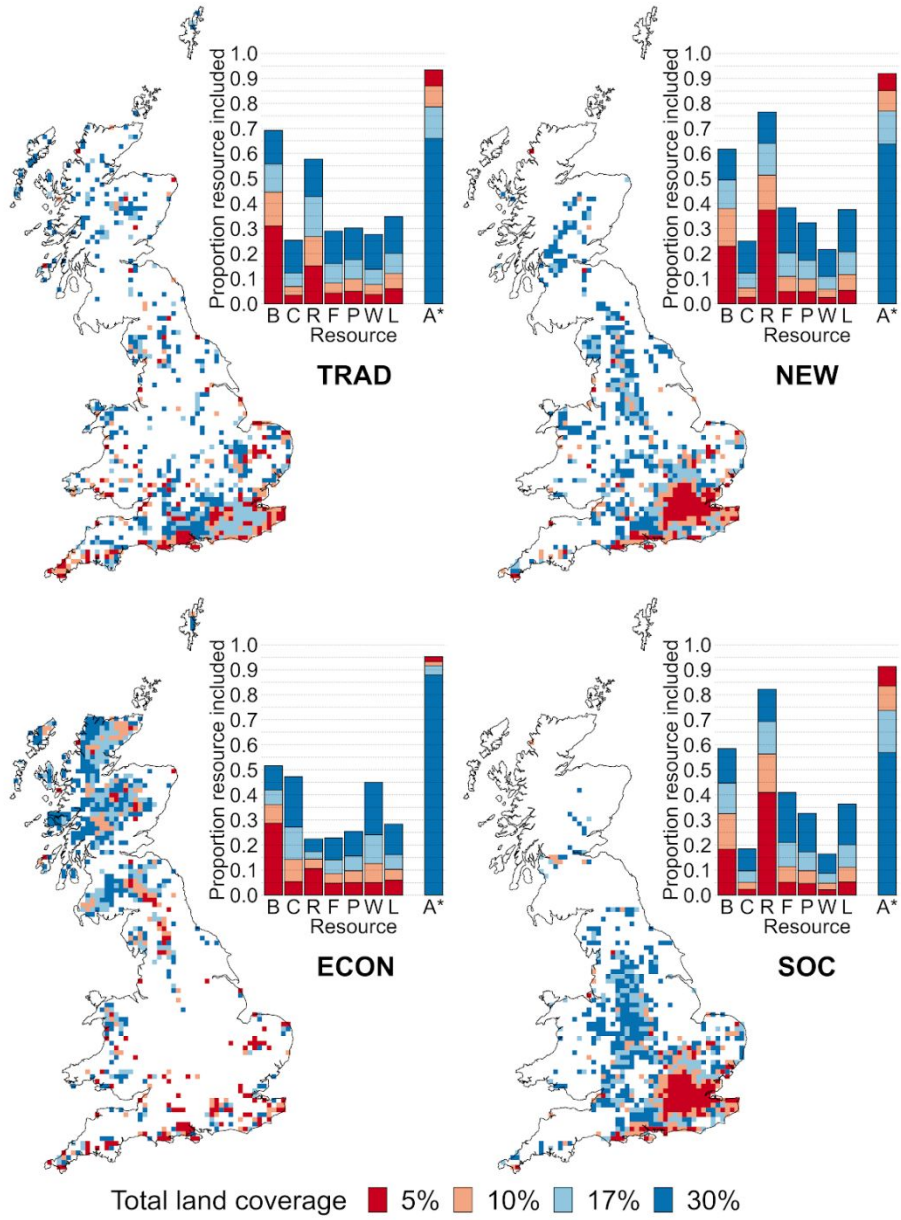
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**SUPPLEMENTARY MATERIALS**

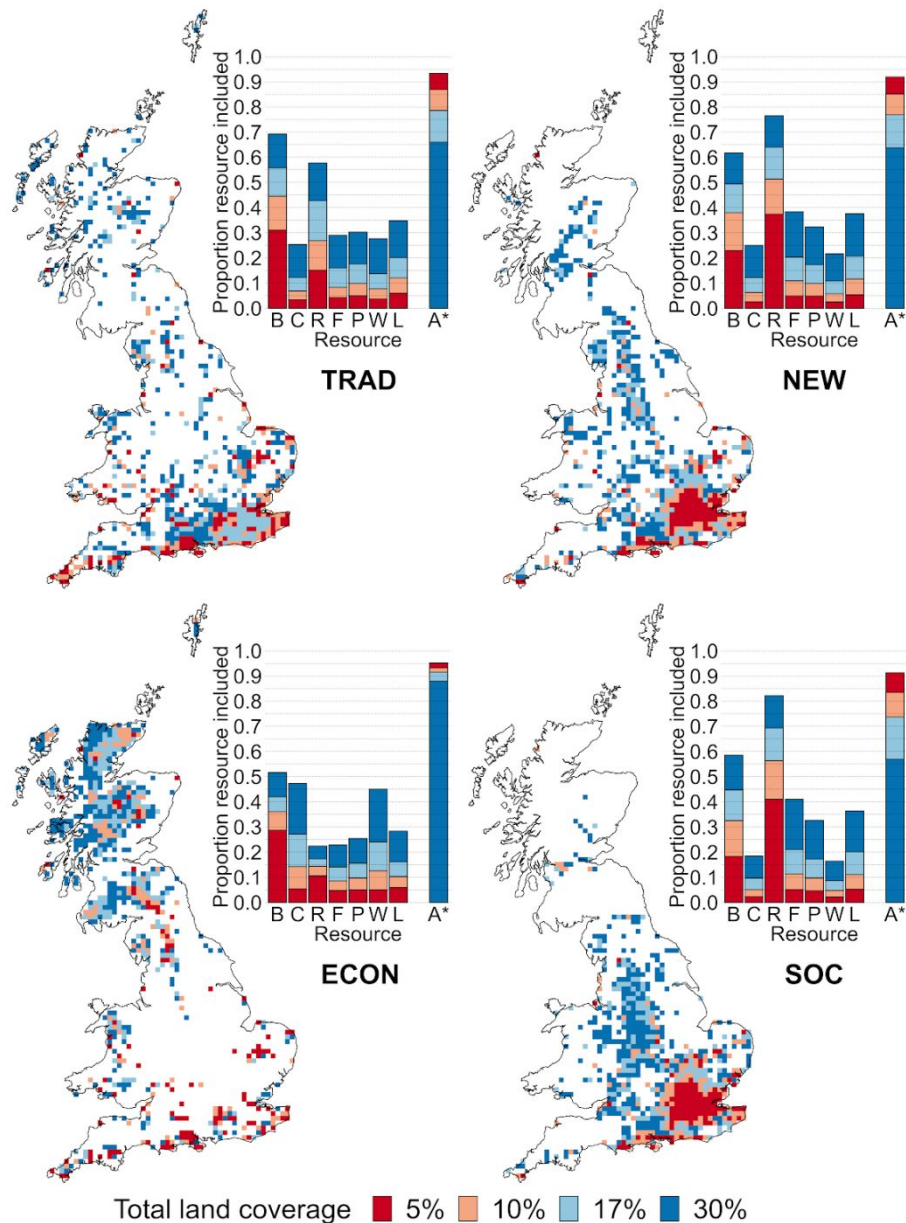




**Figure S1** Protected areas included within the analysis: Sites of Special Scientific Interest (SSSI) and National Nature Reserves (NNR) designated at the time of the study.

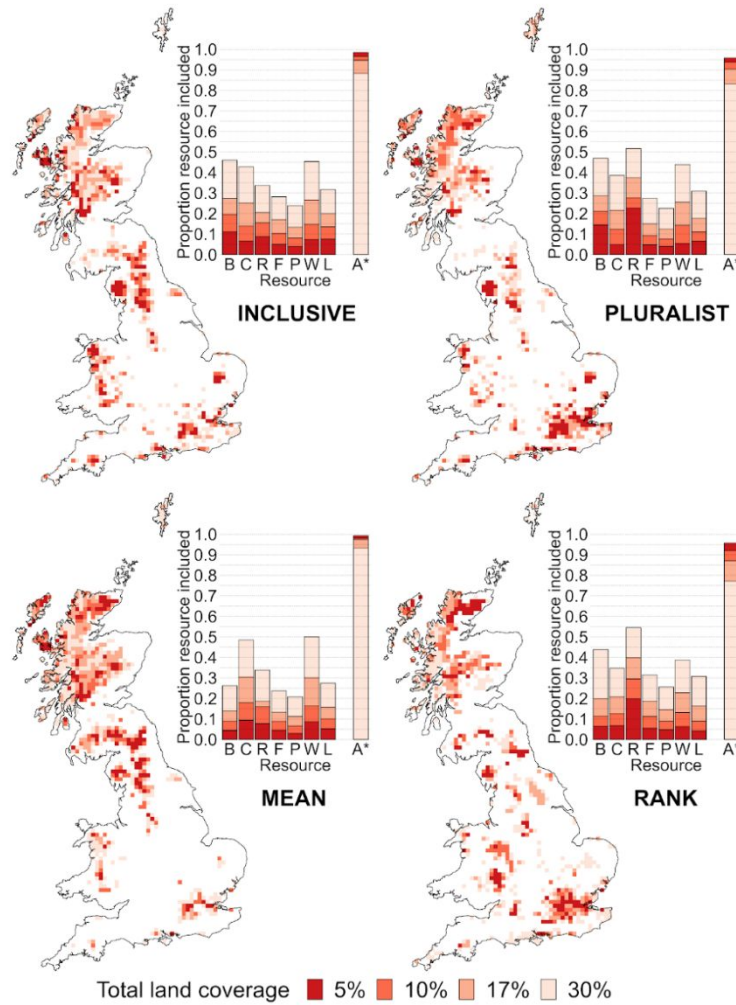




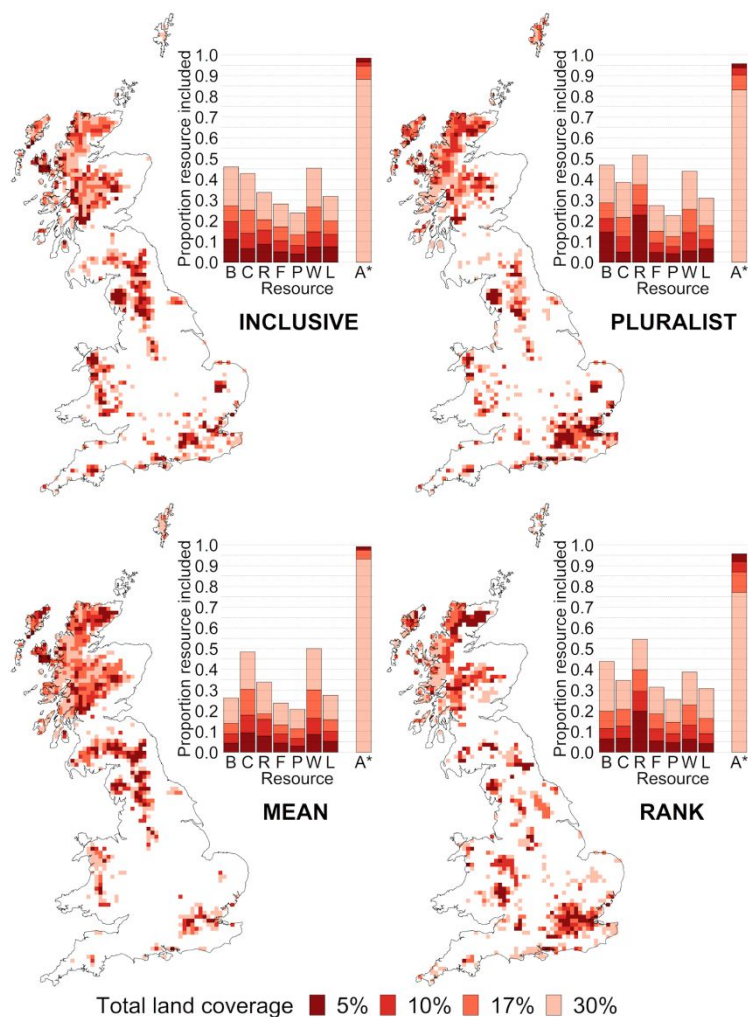


**Figure S2** Feature coverage using spatial prioritisation for each of the four viewpoints using the *additive benefit function* prioritisation method; TRAD – ‘traditional’, NEW – ‘new conservation’, ECON – ‘international market ecocentrism’, SOC – ‘local social instrumentalism’. Maps indicate the priority areas for different coverage thresholds, and corresponding bar plots present proportion of feature included for each, including mean priority species distribution proportion coverage (B), carbon storage (C), recreation (R), flood regulation (F), pollination (P), wilderness (W), landscape aesthetic value (L)

and agricultural/land value (A\*). A\* indicates the only negative weighting where proportion excluded, not included, is shown and so higher land coverage results in lower proportion excluded.

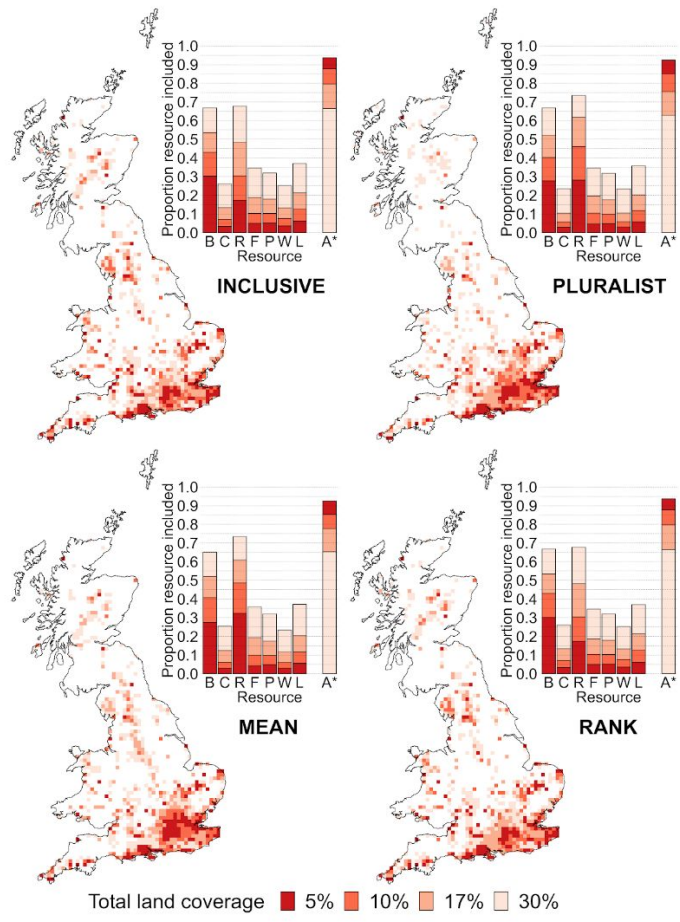


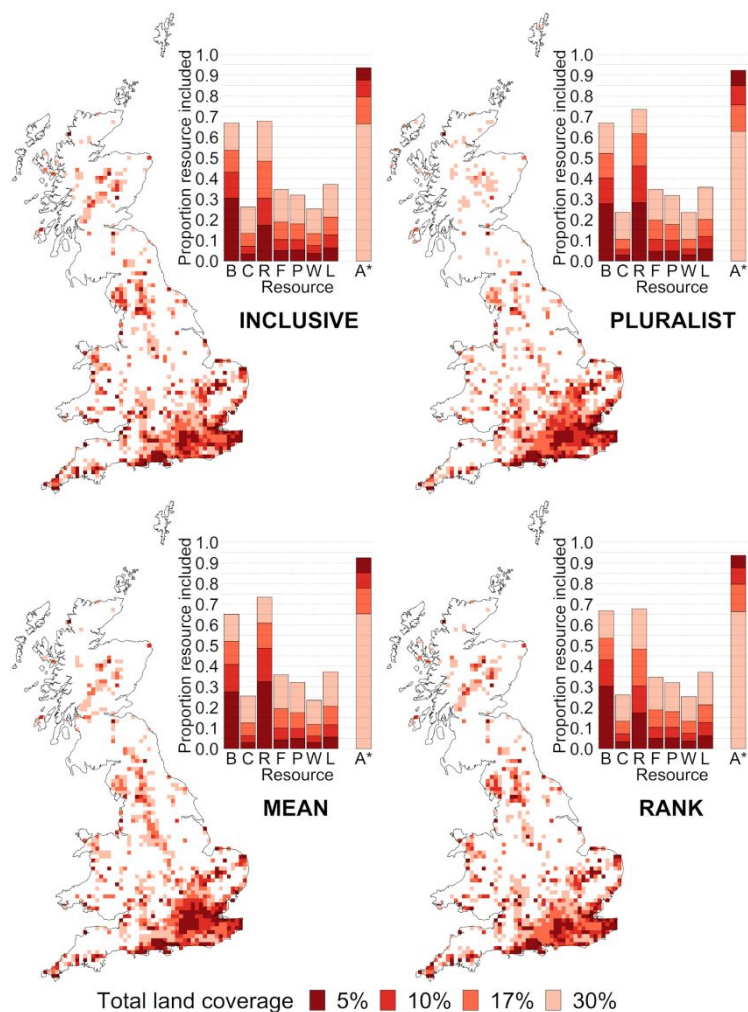




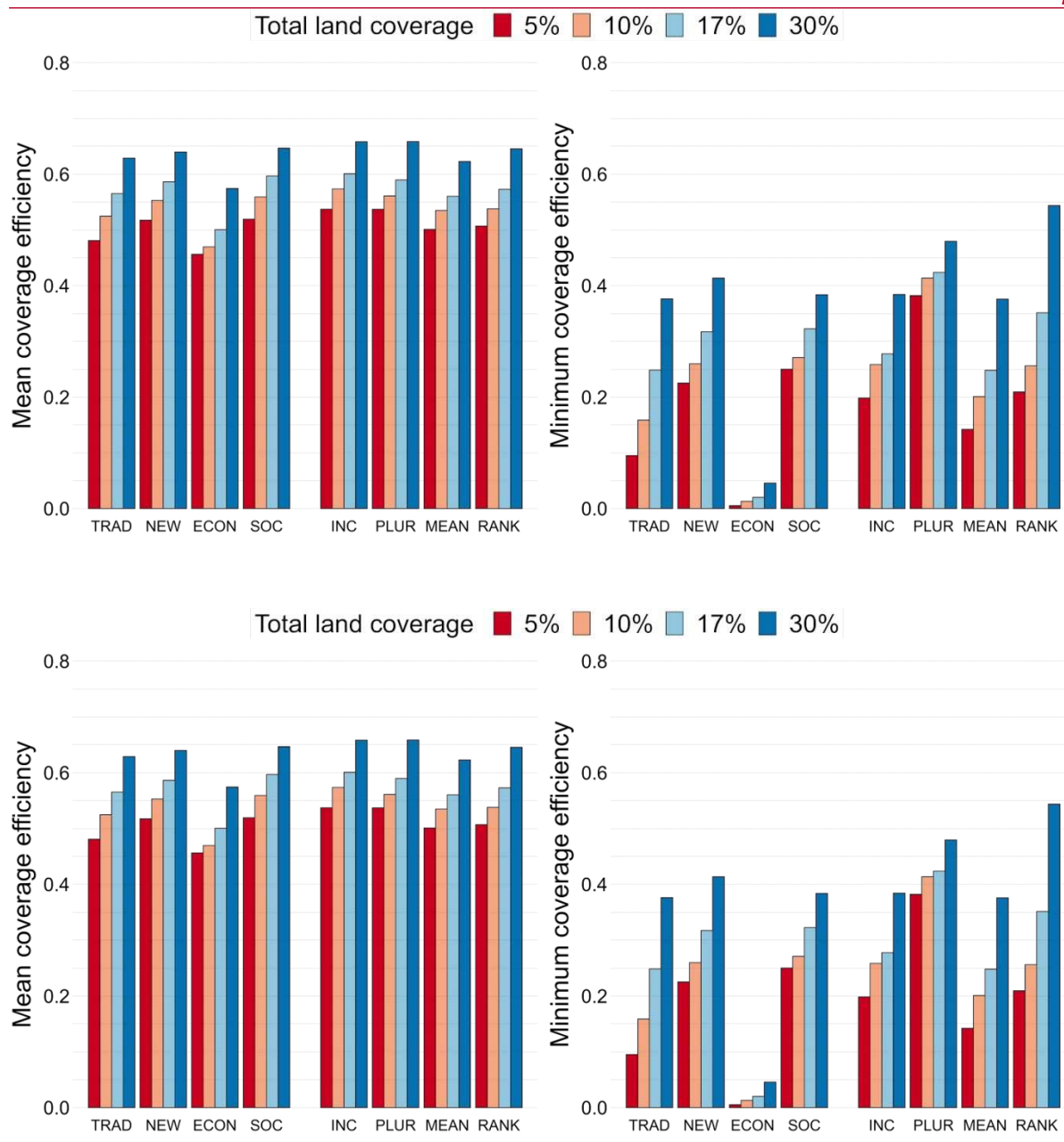
**Figure S3** Spatially aggregating the four conservation viewpoint priorities (TRAD – ‘traditional’, NEW – ‘new conservation’, ECON – ‘international market ecocentrism’, SOC – ‘local social instrumentalism’) using the *core area zonation* prioritisation method. We used inclusive (vote counting) and pluralist (accounting for distinctiveness) methods; as well as two additional integration approaches MEAN (averaging feature weightings before prioritisation), and RANK (undertaking additional prioritisation of viewpoint landscape ranks). Maps indicate the priority areas for different coverage thresholds, and corresponding bar plots present proportion of feature included for each, including mean priority species distribution proportion coverage (B), carbon storage (C), recreation (R), flood regulation (F), pollination (P), wilderness (W), landscape aesthetic value (L) and

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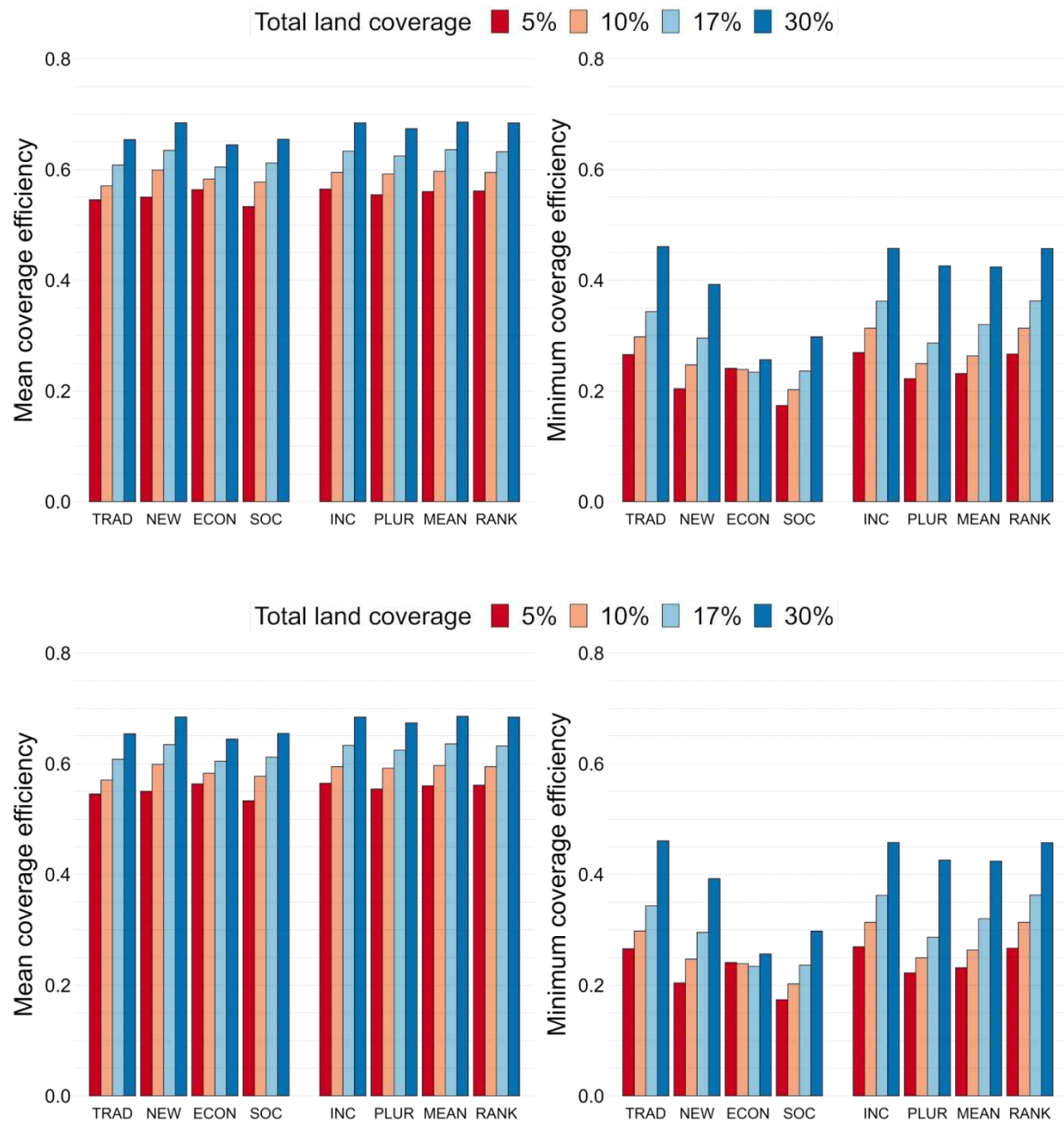


**Figure S4** Spatially aggregating the four conservation viewpoint priorities (TRAD – ‘traditional’, NEW – ‘new conservation’, ECON – ‘international market ecocentrism’, SOC – ‘local social instrumentalism’) using the *additive benefit function* prioritisation method. We used inclusive (vote counting) and pluralist (accounting for distinctiveness) methods; as well as two additional integration approaches MEAN (averaging feature weightings before prioritisation), and RANK (undertaking additional prioritisation of viewpoint landscape ranks). Maps indicate the priority areas for different coverage thresholds, and corresponding bar plots present proportion of feature included for each, including mean priority species distribution proportion coverage (B), carbon storage (C), recreation (R), flood regulation (F), pollination (P), wilderness (W), landscape aesthetic value (L) and agricultural/land value (A\*). A\* indicates the only negative weighting where proportion excluded, not included, is shown and so higher land coverage results in lower proportion excluded.



**Figure S5** Mean and minimum feature coverage efficiency of core area zonation prioritisation performance (TRAD – ‘traditional’, NEW – ‘new conservation’, ECON – ‘international market ecocentrism’, SOC – ‘local social instrumentalism’) and integration approach (Inclusive and Pluralist conservation, as well as two additional integration approaches MEAN and RANK). Efficiency is calculated as the proportion of features covered by a prioritisation for each land coverage threshold (5%, 10%, 17%, and 30%), compared to the maximum possible if only that feature was prioritised. Features included are mean priority species distribution proportion coverage (B), carbon storage (C),

recreation (R), flood regulation (F), pollination (P), wilderness (W), landscape aesthetic value (L) and agricultural/land value (A\*). The left-hand panel shows the mean performance across all resource types (conservation features), whereas the right-hand panel shows the coverage of the feature that is least well covered by a particular approach.



**Figure S6** Mean and minimum feature coverage efficiency of *additive benefit function* prioritisation performance (TRAD – ‘traditional’, NEW – ‘new conservation’, ECON – ‘international market ecocentrism’, SOC – ‘local social instrumentalism’) and integration approach (Inclusive, and

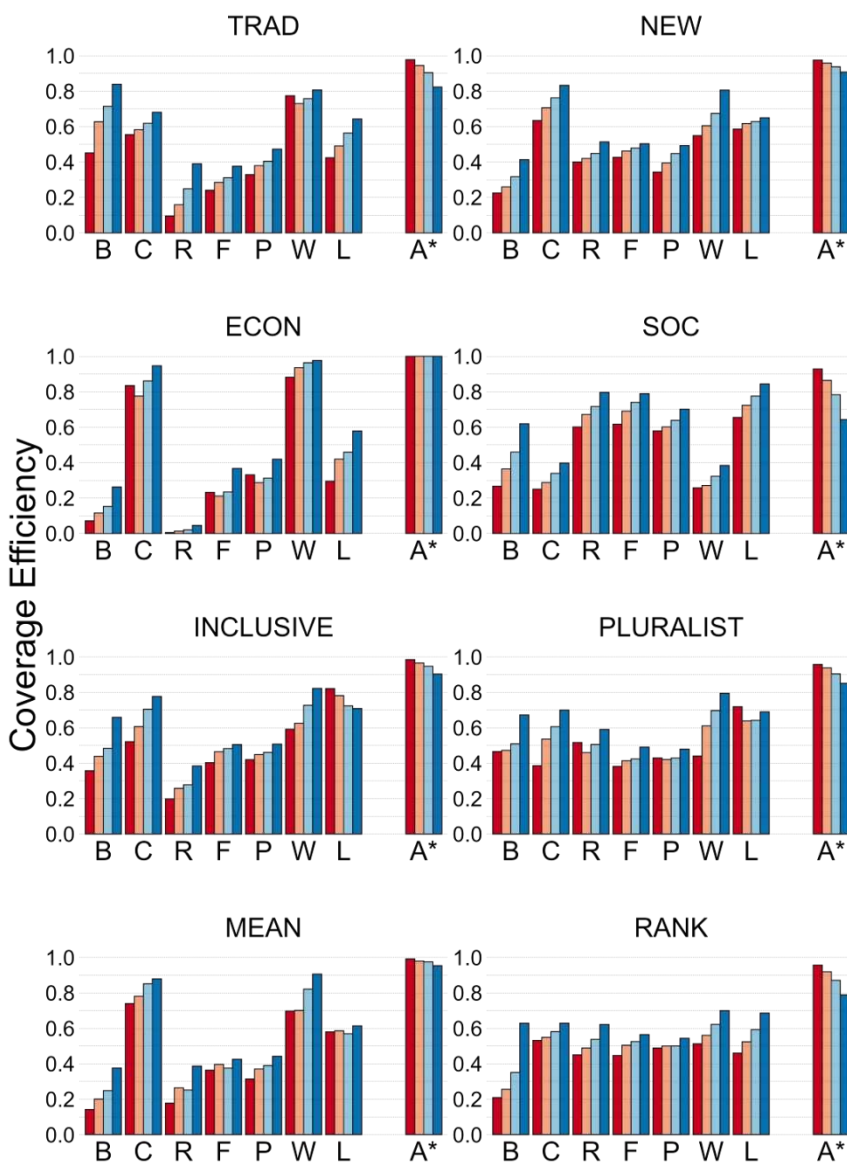
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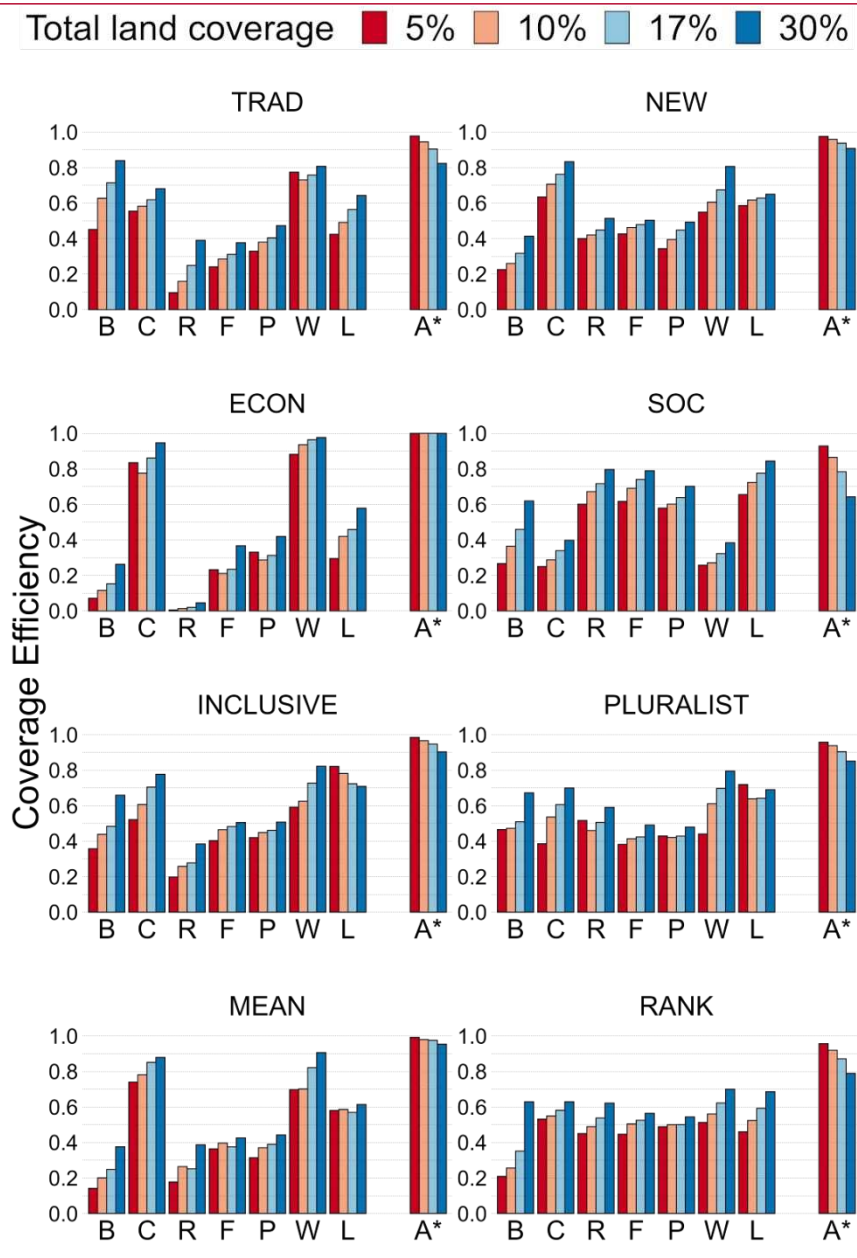
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Pluralist conservation as well as two additional integration approaches MEAN and RANK). Efficiency is calculated as the proportion of features covered by a prioritisation for each land coverage threshold (5%, 10%, 17%, and 30%), compared to the maximum possible if only that feature was prioritised. Features included are mean priority species distribution proportion coverage (B), carbon storage (C), recreation (R), flood regulation (F), pollination (P), wilderness (W), landscape aesthetic value (L) and agricultural/land value (A\*). The left-hand panel shows the mean performance across all resource types (conservation features), whereas the right-hand panel shows the coverage of the feature that is *least well covered* by a particular approach.

Total land coverage 5% 10% 17% 30%



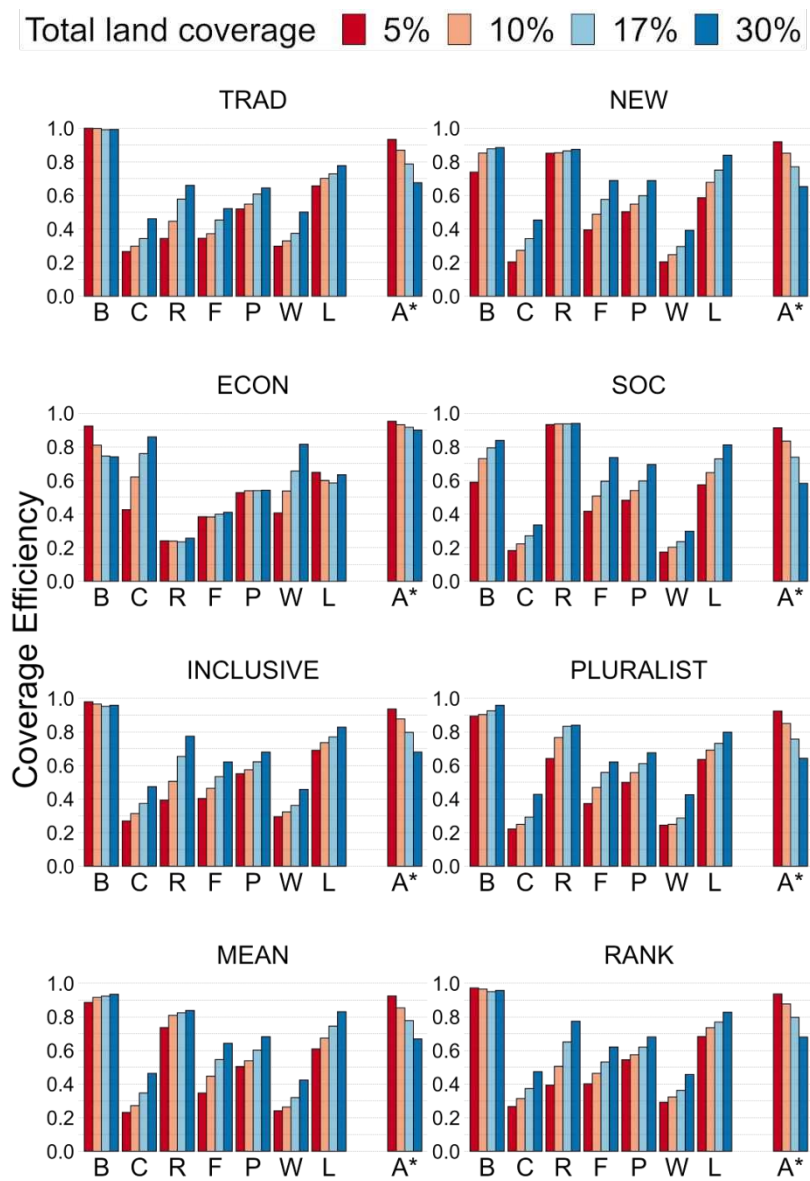


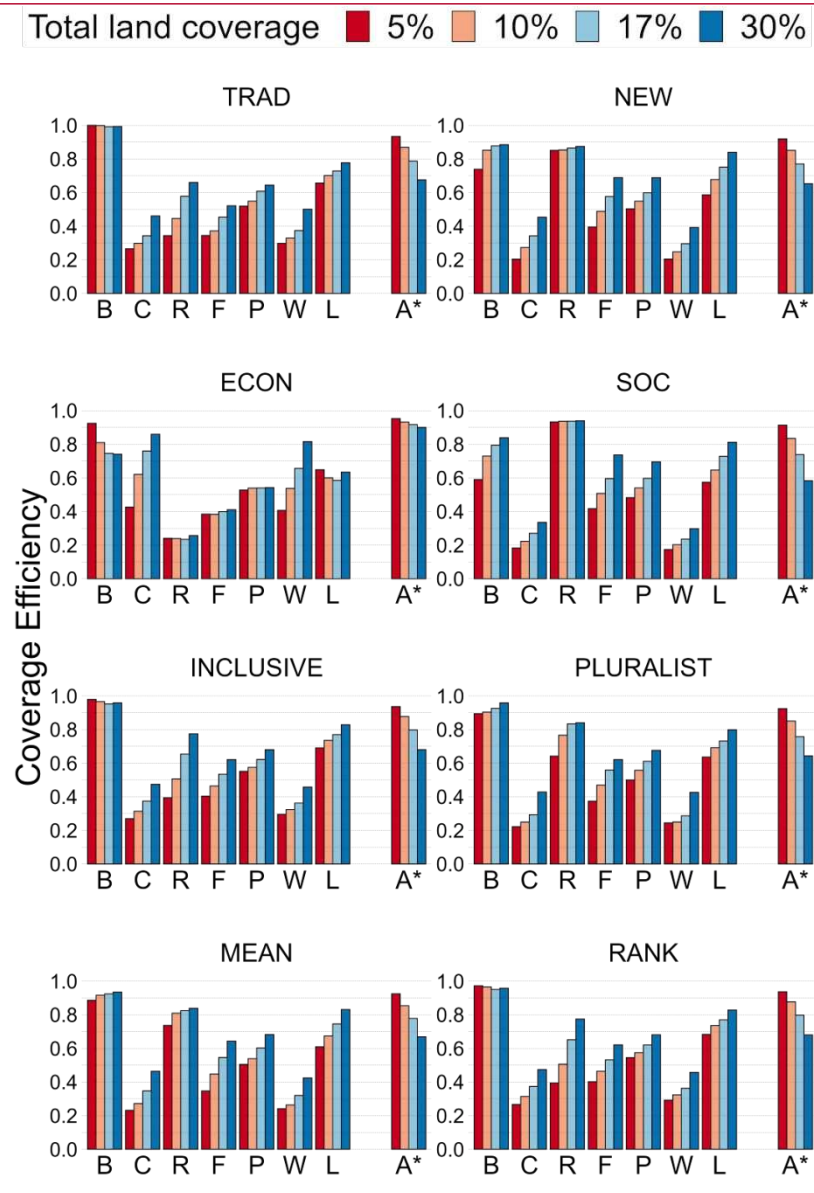


**Figure S7** Efficiency of *core area zonation* prioritisation performance (TRAD – ‘traditional’, NEW – ‘new conservation’, ECON – ‘international market ecocentrism’, SOC – ‘local social instrumentalism’) and integration approach (inclusive, and pluralist conservation as well as two additional integration approaches MEAN and RANK). Efficiency was calculated as the proportion of each feature covered compared to the maximum possible for each land coverage threshold (5%, 10%, 17%, and 30%). Features included are mean priority species distribution proportion coverage (B), carbon storage (C),



recreation (R), flood regulation (F), pollination (P), wilderness (W), landscape aesthetic value (L) and agricultural/land value (A\*).





**Figure S8** Efficiency of *additive benefit function* prioritisation performance (TRAD – ‘traditional’, NEW – ‘new conservation’, ECON – ‘international market ecocentrism’, SOC – ‘local social instrumentalism’) and integration approach (Inclusive, and Pluralist conservation as well as two additional integration approaches MEAN and RANK). Efficiency was calculated as the proportion of each feature covered compared to the maximum possible for each land coverage threshold (5%, 10%, 17%, and 30%). Features included are mean priority species distribution proportion coverage (B), carbon storage (C), recreation (R), flood regulation (F), pollination (P), wilderness (W), landscape aesthetic value (L) and agricultural/land value (A\*).

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**Supplementary Table 1** Output from the PCA analysis used to create the pluralist approach rankings. We partitioned variance from viewpoint weightings of feature layers, creating principal components (PC; columns). Cumulative proportion of variance explained by PCs included in brackets. ~~We used each PC to multiply~~We multiplied viewpoint prioritisation landscape rankings by corresponding PC eigenvectors, and took the absolute value of the sum (dot product). PCs were added iteratively until maximum viewpoint eigenvalue across PCs (**bold**) was included (PC3). The first PC is associated with the NEW and ECON viewpoints, the second PC is strongly associated with the SOC viewpoint, and the third PC is strongly associated with the TRAD viewpoint.

	<b>PC1 (0.601)</b>	<b>PC2 (0.911)</b>	<b>PC3 (0.999)</b>	<b>PC4 (1.000)</b>
<b>TRAD</b>	-0.168	0.234	<b>-0.927</b>	0.240
<b>NEW</b>	<b>-0.658</b>	-0.325	0.205	0.647
<b>ECON</b>	<b>-0.693</b>	0.515	0.129	-0.487
<b>SOC</b>	-0.241	<b>-0.758</b>	-0.286	-0.535

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## Supplementary methods

### Feature layers

The complete list of seven ecosystem service and socio-environmental value layers were collated as follows:

Five ES layers were included; carbon storage (existing), agricultural value, recreational services, flood regulation, and pollination services. Carbon storage value was calculated as the sum of interpolated below-ground carbon from the CEH Soil Carbon Map to a depth of 100 cm (Bradley *et al.*, 2005), and estimated above-ground carbon using the 2007 Land Cover Map (Henrys, Keith and Wood, 2016). Agricultural value was assigned based upon agricultural land classifications for England (Natural Resources Wales, 2019; The James Hutton Institute, 2019; Natural England, 2020). Classifications were standardised between countries into an interoperable code, and the mean landscape value was then rescaled and subtracted from 1 to calculate the final agricultural value used for the spatial prioritisations [see Cunningham *et al.* (2021) for details]. Urban areas were then given the highest value, indicating unsuitable land use for terrestrial conservation. Recreation value was estimated from the predicted annual visits/ha for a potential new National Park, see Schägner *et al.* (2016). (Bradley *et al.*, 2005), and estimated above-ground carbon using the 2007 Land Cover Map (Henrys, Keith and Wood, 2016). Agricultural value was assigned based upon agricultural land classifications for England (Natural Resources Wales, 2019; The James Hutton Institute, 2019; Natural England, 2020). Classifications were standardised between countries into an interoperable code, and the mean landscape value was then rescaled and subtracted from 1 to calculate the final agricultural value used for the spatial prioritisations [see Cunningham *et al.* (2021) for details]. Urban areas were then given the highest value, indicating unsuitable land use for terrestrial conservation. Recreation value was

estimated from the predicted annual visits/ha for a potential new National Park, see Schägner et al. (2016).

The value of protecting land for flood prevention depends on (a) supply: the degree to which upstream land reduces peak discharge volume (i.e. flooding risk); and (b) demand: the damage a flood could cause accounting for location within the catchment (i.e. aggregated damage within *and* downstream of each catchment). These factors interact such that if there is no valuable infrastructure downstream flood prevention action gains nothing, but equally if a location currently does little to reduce peak discharge then flood prevention value is again low. Hence, flood regulation value was estimated using a supply index (predicted total effect of upstream land on river discharge after precipitation events), and a catchment level demand index (downstream flood damage accounting for upstream area); see Stürck et al. ~~(2014)~~(2014) for details of supply and demand indexes used in this analysis. These indices do not provide an absolute measure of service flow; however, the relative distributions can be compared. Flood regulation flow was estimated by ranking the supply and demand indices separately, and then taking the minimum rank of the two. In this way, areas that had both relatively high supply and demand received higher value. Pollination service flow was similarly calculated with a supply index (estimated visitation probability by pollinators), and demand index (area of pollinator crops weighted by dependency level), see Schulp et al. ~~(2014)~~(2014).

Additionally two socio-environmental value layers were added; wilderness and landscape aesthetic value. Wilderness was included from the 'wilderness register and indicator for Europe' map, created from a combination of naturalness, remoteness from settlements and access, and terrain ruggedness ~~(Kuiters et al., 2013)~~(Kuiters et al., 2013). Landscape aesthetic value was quantified based on numbers of geolocated unique user uploads to three social media platforms, see Van Zanten et al. ~~(2016)~~(2016). The mean landscape rank of the number of uploads to each platform was then taken as the 'landscape aesthetic value'.

#### **Other viewpoint integration approaches**

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In addition to the inclusive and pluralist approaches described within the main text, two additional [multi-criteria decision analysis \(MCDA\)](#) spatial approaches to integrating viewpoints together were tested. The first approach involved calculating the mean feature weightings between viewpoints (mean of the four weightings for each feature in Table 4.2) prior to any spatial prioritisation. These mean weightings were then used within a single spatial prioritisation using Zonation (MEAN), and hence this approach approximates deciding on conservation priorities prior to any spatial prioritisation. The other integration approach involved using the output landscape rankings from the four viewpoint prioritisations (TRAD, NEW, ECON, SOC) to seek an overall compromise (RANK). A further Zonation prioritisation was carried out on these ranks (each individual viewpoint was treated as an input feature layer). Neither of these two alternative methods outperformed the inclusive and pluralist methods described in the main text in terms of mean or minimum feature coverage efficiency using CAZ (with the exception of higher RANK minimum efficiency at the highest [30%] area coverage threshold). MEAN consistently underperformed the other approaches using CAZ.

All four methods were tested using both the *core area zonation (CAZ)* and *additive benefit function (ABF)* prioritisation method. Both methods iteratively remove landscapes contributing the smallest value to the remaining landscapes. Through this removal, landscapes remaining within the solution longer complement other landscapes to a greater extent, in terms of contributing the most to underrepresented features. Using CAZ, landscape value is calculated as the *maximum* weighted proportion of any positive feature within the remaining landscapes (minus any negative alternative land use value within the landscape). Using ABF, this is *averaged across all positive features*, not just the maximum value. Inclusive and pluralist integration approaches using CAZ are presented in the main text, and all others are presented in Supplementary Figure 2 to Supplementary Figure 8. The following discussion considers similarities and differences between ABF and CAZ results.

## Supplementary discussion

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**Additive benefit prioritisation**

Since ABF averages across all features, it resulted in higher overall feature coverage but lower levels of complementarity between landscapes. Hence there was greater spatial similarity between the ABF viewpoint prioritisations than the CAZ prioritisations, with NEW and ECON prioritisations especially spatially correlated (Supplementary Figure 2). The greater convergence between viewpoints was due to ABF considering all landscape features, rather than the single highest weight\* (positive proportion) in CAZ. Due to these increased similarities, ABF viewpoint integration approaches were also more spatially similar compared to CAZ (Supplementary Figure 3 and Supplementary Figure 4), with a particular concentration within the south of England suggesting that this is an area with potentially large gains in feature coverage, even if the most important landscapes for some features are not included.

Feature coverage was more consistent between the ABF integration approaches, and they provided a slightly higher mean feature coverage efficiency than CAZ (ABF 17% coverage efficiency range: 0.625-0.636; CAZ: 0.560-0.600; Supplementary Figure 5 to Supplementary Figure 8). For lower thresholds, minimum coverage efficiency was generally higher using ABF too (ABF 5% coverage efficiency range: 0.222-0.269; CAZ: 0.142-0.383). However, as the threshold rose CAZ minimum efficiency generally increased at a faster rate than ABF, and CAZ ultimately exceeded ABF for the pluralist and RANK approaches (ABF 30% coverage efficiency range: 0.424-0.458; CAZ: 0.376-0.545). This is illustrated by Supplementary Figure 5 and Supplementary Figure 6 (right hand panels), where ABF mainly outperforms CAZ at 5% area coverage (red columns) but not at 30% (dark blue columns), and some features may largely be 'missed' with the CAZ approach at 5% coverage if a single viewpoint is adopted. This reflects the fact that achieving multiple goals (satisfying multiple viewpoints and including many different features) is increasingly difficult at low coverage thresholds: CAZ priorities (aiming to include the very best examples of each feature included by a particular viewpoint) may be more difficult to reconcile than ABF (incorporating the places with the best mixture of features) when

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only a small percentage of the land is allocated to conservation. Nonetheless, the CAZ pluralist approach had relatively high minimum feature coverage efficiency for all area thresholds, ensuring that desired features (by any viewpoint) were not missed, even at low thresholds.

All ABF integration approaches resulted in high mean feature coverage efficiency and moderately high minimum efficiency. Hence, ABF could be considered a more inherently ‘inclusive’ prioritisation method in that the best combined-feature areas will be selected (most are well satisfied by any of the ABF integration approaches), but areas that are critically important for a single conservation feature may be disregarded (some individuals may be disappointed). Similarly CAZ could be considered a more ‘pluralist’ prioritisation method, in that the most important locations for each feature and viewpoint are maintained, even if the solution is slightly less efficient overall. Both ABF and CAZ prioritisation methods could offer coherent conservation plans by integrating viewpoints, and the prioritisation method used should depend upon conservation objectives and spatial context. However, we focused on CAZ prioritisation in the main text, here, because CAZ combined with a pluralist approach generally resulted in the highest minimum coverage.



## How to satisfy as many people as possible when conservationists have different priorities

While those involved in biodiversity conservation tend to agree over their broad aims, there are often some important differences of opinion and perspectives on how to tackle these aims and the priorities that should be given to each. These opinions result in different values, such as whether an individual would prioritise the protection of rare species, maintaining as much carbon as possible in ecosystems, or providing recreational opportunities that will increase human wellbeing. All of these priorities have merit, but there are trade-offs between them. It is important, therefore, that different viewpoints are considered and balanced against each other within coherent plans that minimise any sense of unfairness and help to avoid future conflict. But how can they be reconciled?

We developed and tested different quantitative methods to balance opposing viewpoints on how to value nature. First we created four simulated “caricatures” of what different conservationists might favour (based on studies from the social science literature) and then used numerical analyses (of where species, carbon and recreational values exist, for example) to identify where would be prioritised for protection by people with these different sets of values within Britain. While there was some overlap, people with different viewpoints would often want to protect different parts of Britain. To reconcile these differences, we developed a new, numerical method which represents a “pluralist approach” to join these viewpoints into a single plan. This approach ensured that all of the caricatures still ‘got most of what they wanted’ and prevented the ‘least supported’ (or ‘most unusual’) set of priorities from being ignored, as might happen if priorities were simply weighted by the number of people which care about each. The analyses generated a coherent spatial conservation plan which appears to include different conservation values efficiently and satisfy all the different viewpoints quite well.

This new analytical method represents an important development in incorporating diverse viewpoints within conservation planning and provides a new tool to support decision making. Including this quantitative method within broader approaches to conservation decision-making (e.g. systematic planning frameworks) would facilitate the development of increasingly satisfactory compromise solutions through transparent engagement with stakeholders.

*Photo caption: Landscape features with differing values, depending upon your conservation viewpoint, along the River Swale in North Yorkshire, UK: biodiverse meadows, carbon sequestering woodlands, traditional pasture, and a water provisioning river. Photo by Charles Cunningham.*



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