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Reconciling diverse viewpoints within systematic conservation planning

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Abstract:	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. 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. 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. 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.

Abstract

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.

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.

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.

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.

INTRODUCTION

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

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

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

The important step of identifying and balancing stakeholder viewpoints is typically undertaken at the beginning of the planning process in an attempt to agree weightings or goals for different priorities. One way of addressing these issues in a structured way is through using a planning framework such as systematic conservation planning (SCP) (Pressey and Bottrill, 2009; Watson *et al.*, 2011), which utilises network-scale and spatially explicit methods to inform important conservation planning decisions (Watson *et al.*, 2011). SCP provides a way to incorporate these techniques in a robust and auditable process, incorporating the principle of complementarity to design an optimal network for a given set of planning objectives (Margules and Pressey, 2000; Wilson, Cabeza and Klein, 2009). As part of a full SCP implementation there is opportunity to build consensus between differing stakeholders' viewpoints within the initial planning stages, but this is carried out primarily when conservation goals are being set rather than the prioritisation stage. However it is difficult to understand, at this early stage, whether different values translate into similar or distinct spatial priorities, potentially generating considerable debate about issues that have no practical impact.

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

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

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

METHODS

Feature layers

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

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

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

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

Viewpoint prioritisation

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

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

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

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

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

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

Viewpoint integration

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

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

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

Eq. 1

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

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

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

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

Eq. 2

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

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

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

RESULTS

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

We integrated the four viewpoints into single conservation strategies using two approaches (Figure 3). The inclusive approach selected landscapes in Scotland, upland Wales, north England, and southeast England. The pluralist approach contained similar priority areas, but higher priority landscapes were more concentrated in SE England. The pluralist approach had lower coverage of carbon and exclusion of agricultural value, but recreational value coverage was much higher than the inclusive approach (feature coverage, Figure 3; efficiency, Supplementary Figure 7). Species distributions received the best coverage through the ‘traditional’ viewpoint. The inclusive and pluralist integration approaches had higher species representation than the other viewpoints, although coverage was lower than the ‘traditional’ viewpoint (mean species distribution proportion coverage at 17% coverage threshold: TRAD 40.3%, NEW 17.9%, ECON 8.6%, SOC 25.9%, Inclusive 27.2%, Pluralist 28.8%; other thresholds: Fig. 2-3). The existing protected area network (Supplementary Figure 1) matched 'new' and 'market ecocentrism' viewpoint prioritisation ranks more closely than 'traditional' (TRAD $\rho = 0.379$, NEW $\rho = 0.423$, ECON $\rho = 0.413$, SOC $\rho = 0.068$), which was contrary to our expectation given that the rationale and goals for identifying potential SSSIs and NNRs closely align with a traditional approach.

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

DISCUSSION

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

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

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

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

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

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

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

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

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

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

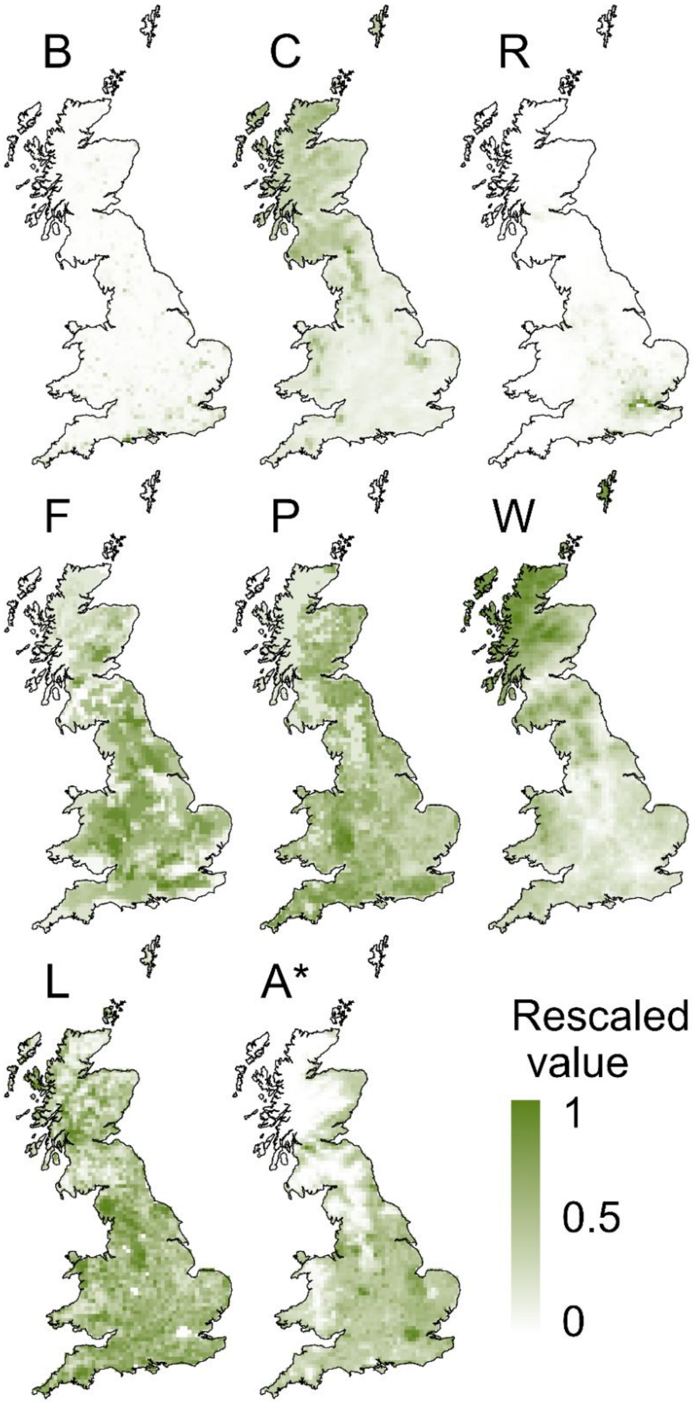

Figure 1 Rescaled ecosystem service, biodiversity and socio-environmental value feature layers included within the analysis including; mean priority species distribution proportion coverage (B), carbon storage (C), recreation (R), flood regulation (F), pollination (P), wilderness (W), landscape aesthetic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

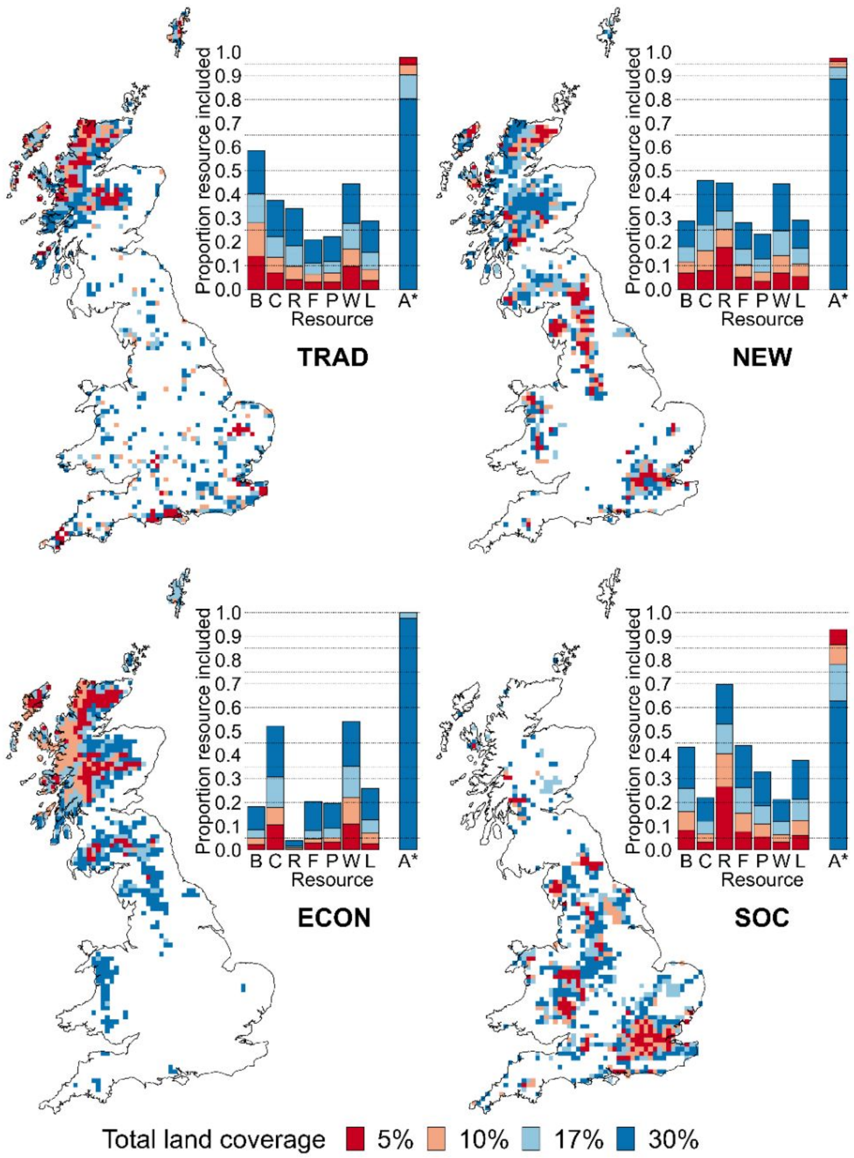

Figure 2 Feature coverage using spatial prioritisation for each of the four viewpoints; TRAD – 'traditional', NEW – 'new' conservation, ECON – 'international market ecocentrism', SOC – 'local social instrumentalism'. Maps indicate the priority areas for different coverage thresholds, and corresponding 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.

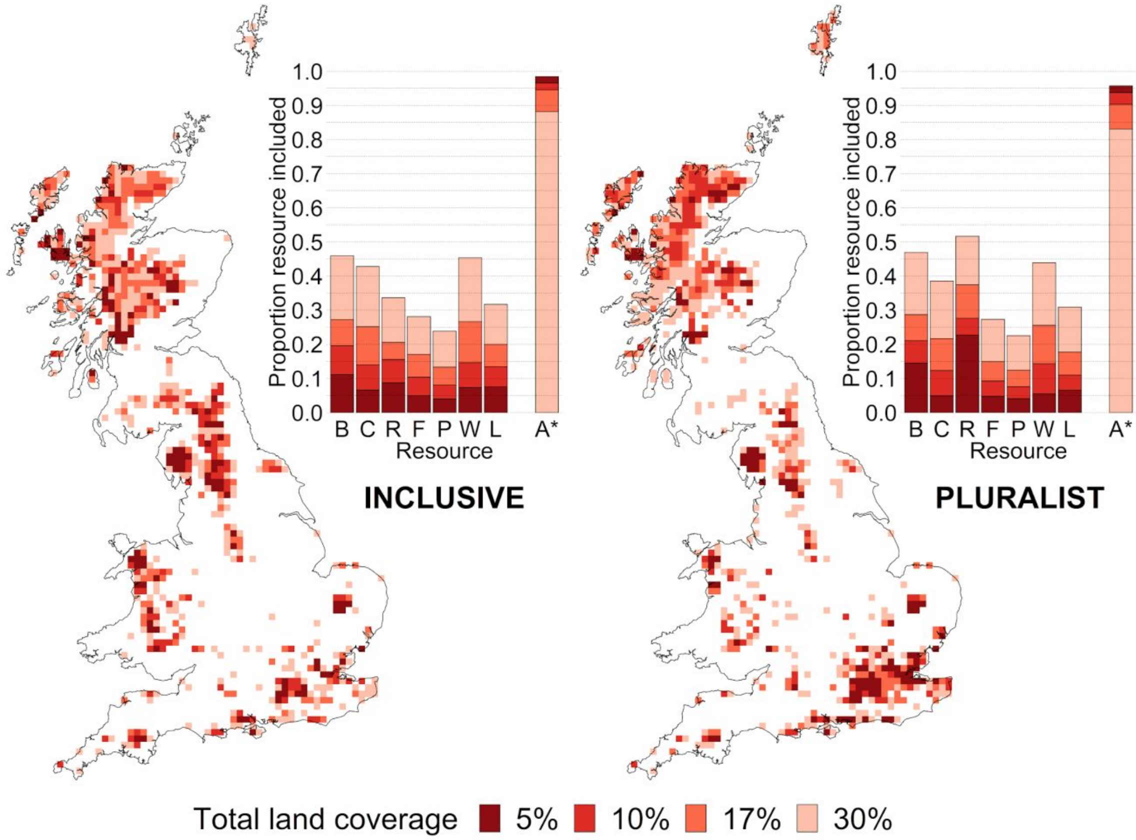

Figure 3 Spatially aggregating the four conservation viewpoint priorities (TRAD – 'traditional', NEW – 'new' conservation, ECON – 'international market ecocentrism', SOC – 'local social instrumentalism') using inclusive (vote counting) and pluralist (accounting for distinctiveness) integration methods. 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.

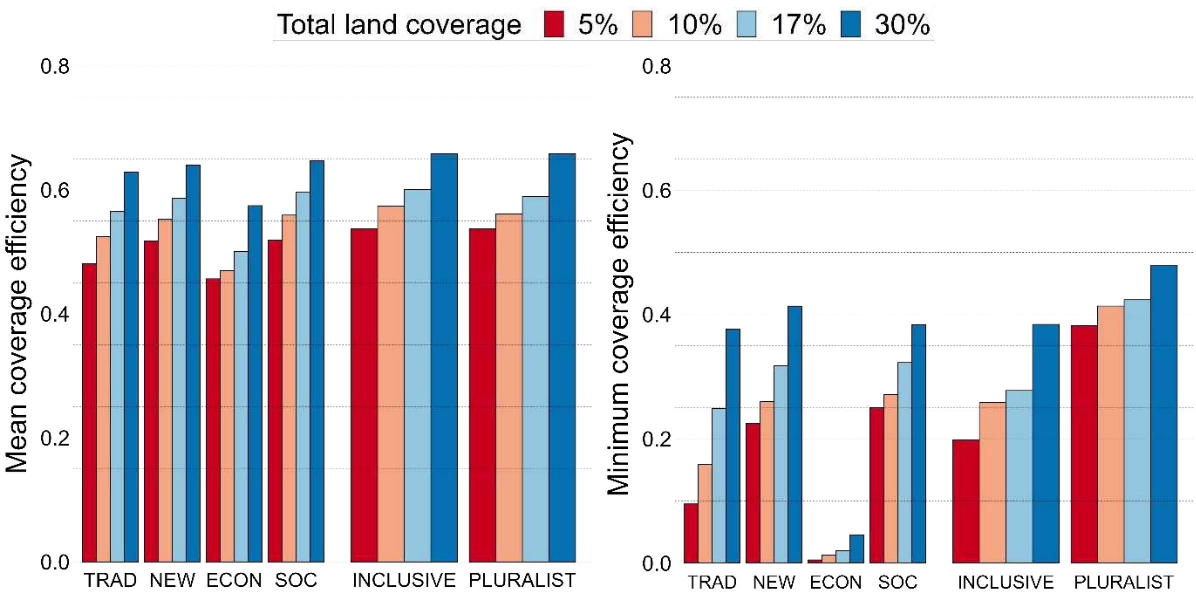

Figure 4 Mean and minimum feature coverage efficiency of viewpoint prioritisation performance (TRAD – 'traditional', NEW – 'new' conservation, ECON – 'international market ecocentrism', SOC – 'local social instrumentalism') and integration approach (inclusive and pluralist conservation). Efficiency is 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 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.

Conservation viewpoint	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)	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). Weightings: species distributions and wilderness.
'New' (NEW)	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). Weightings: widest scope of the four viewpoints, including species and all economic and social value data, apart from wilderness.
International market ecocentrism (ECON)	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). Weightings: agricultural value (avoid) and related pollination service flow, as well as carbon storage and species distributions.
Local social instrumentalism (SOC)	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). 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.
Viewpoint integration approach	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.

Feature	**Traditional conservation (TRAD)**	**'New' conservation (NEW)**	**International market ecocentrism (ECON)**	**Local social instrumentalism (SOC)**
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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SUPPLEMENTARY MATERIALS

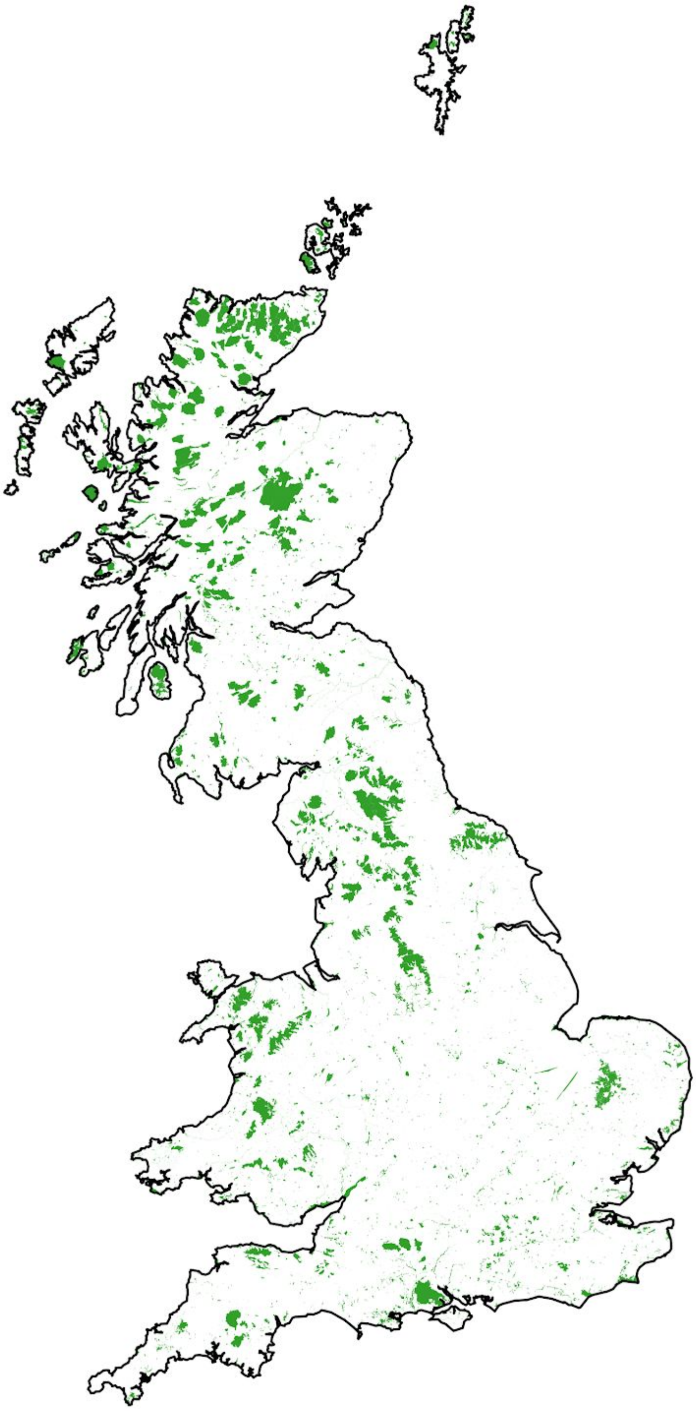

Figure S1 Protected areas included within the analysis: Sites of Special Scientific Interest (SSSI) and National Nature Reserves (NNR) designated at the time of the study.

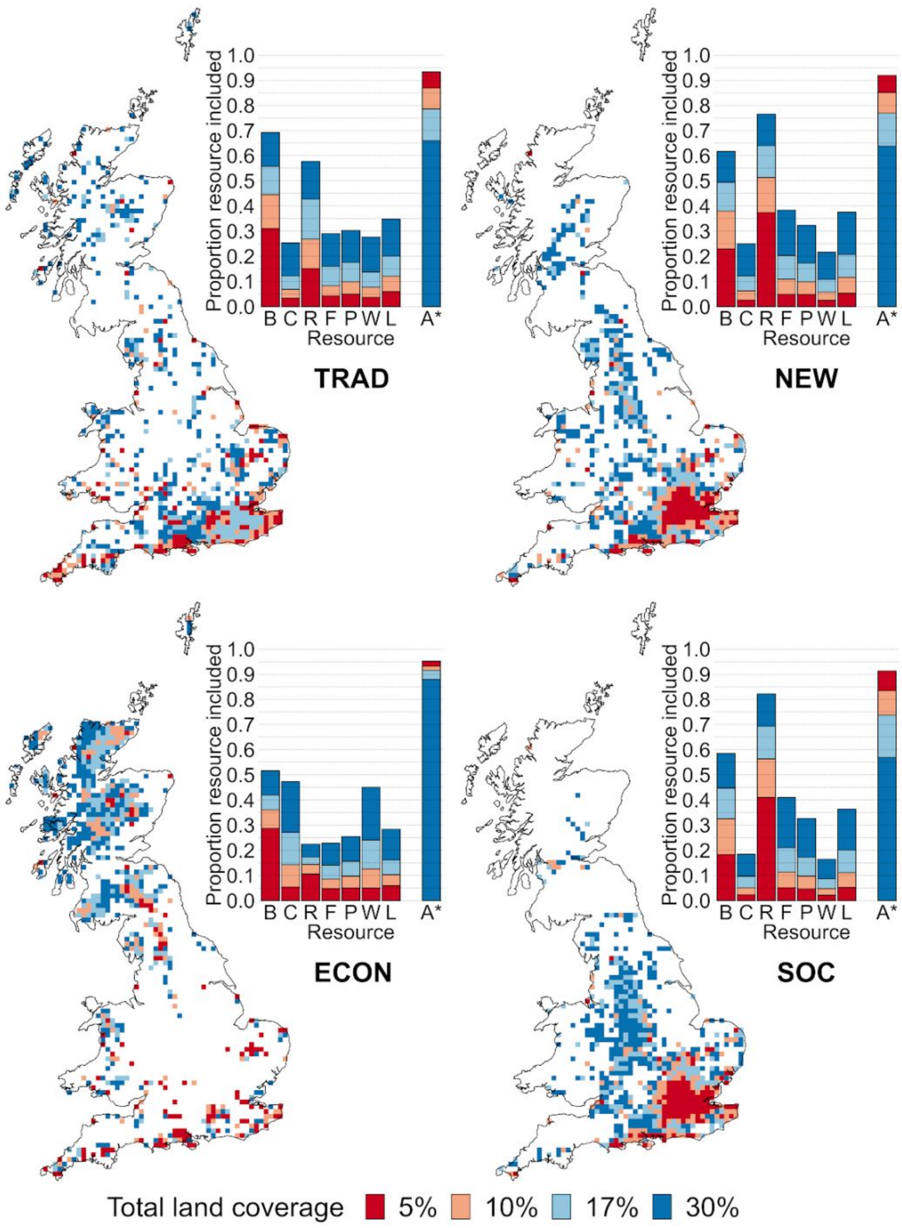

Figure S2 Feature coverage using spatial prioritisation for each of the four viewpoints using the *additive benefit function* prioritisation method; TRAD – 'traditional', NEW – 'new conservation', ECON – 'international market ecocentrism', SOC – 'local social instrumentalism'. Maps indicate the priority 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.

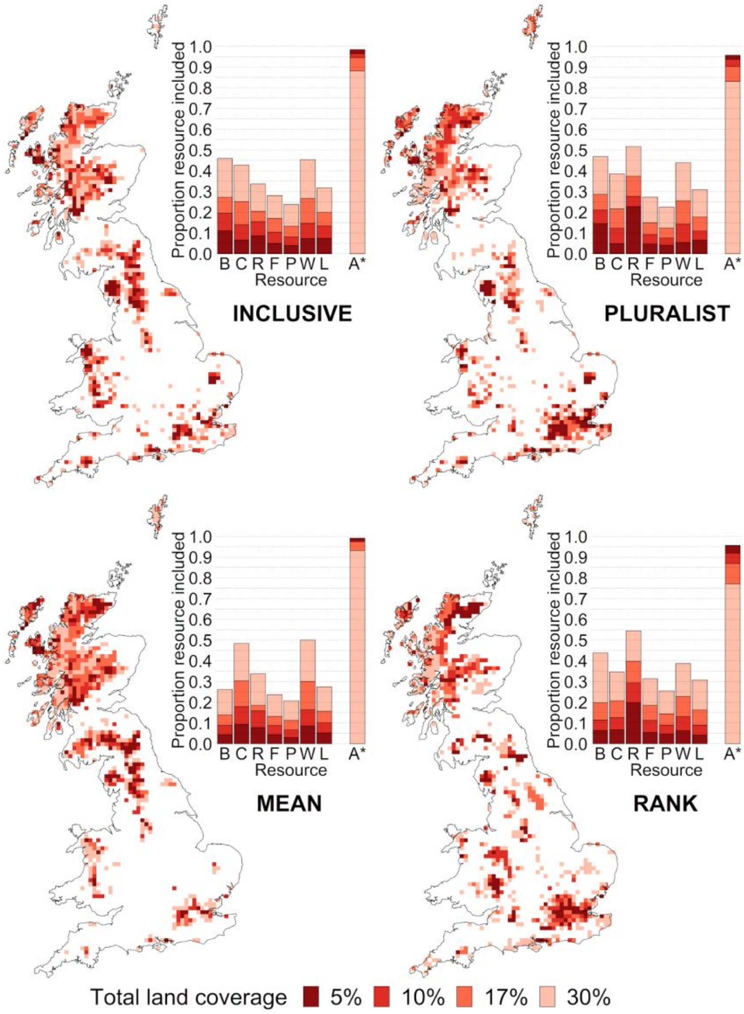

Figure S3 Spatially aggregating the four conservation viewpoint priorities (TRAD – 'traditional', NEW – 'new conservation', ECON – 'international market ecocentrism', SOC – 'local social instrumentalism') using the *core area zonation* prioritisation method. We used inclusive (vote counting) and pluralist (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.

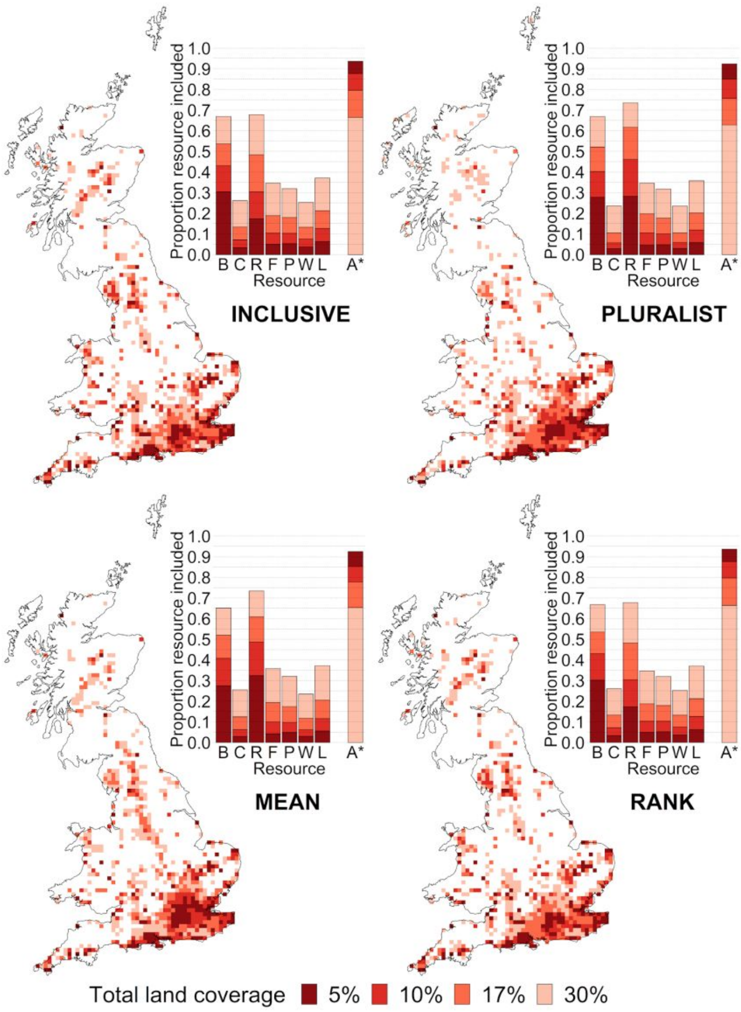

Figure S4 Spatially aggregating the four conservation viewpoint priorities (TRAD – 'traditional', NEW – 'new conservation', ECON – 'international market ecocentrism', SOC – 'local social instrumentalism') using the *additive benefit function* prioritisation method. We used inclusive (vote counting) and 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.

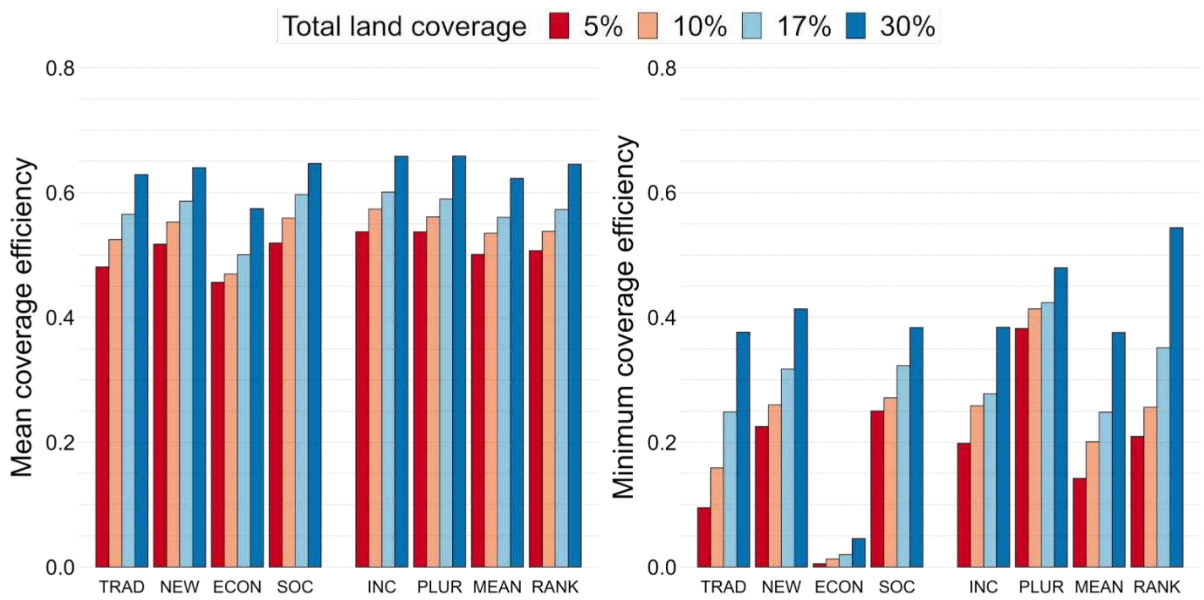

Figure S5 Mean and minimum feature coverage efficiency of core area zonation prioritisation performance (TRAD – 'traditional', NEW – 'new conservation', ECON – 'international market ecocentrism', SOC – 'local social instrumentalism') and integration approach (Inclusive and Pluralist conservation, as 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.

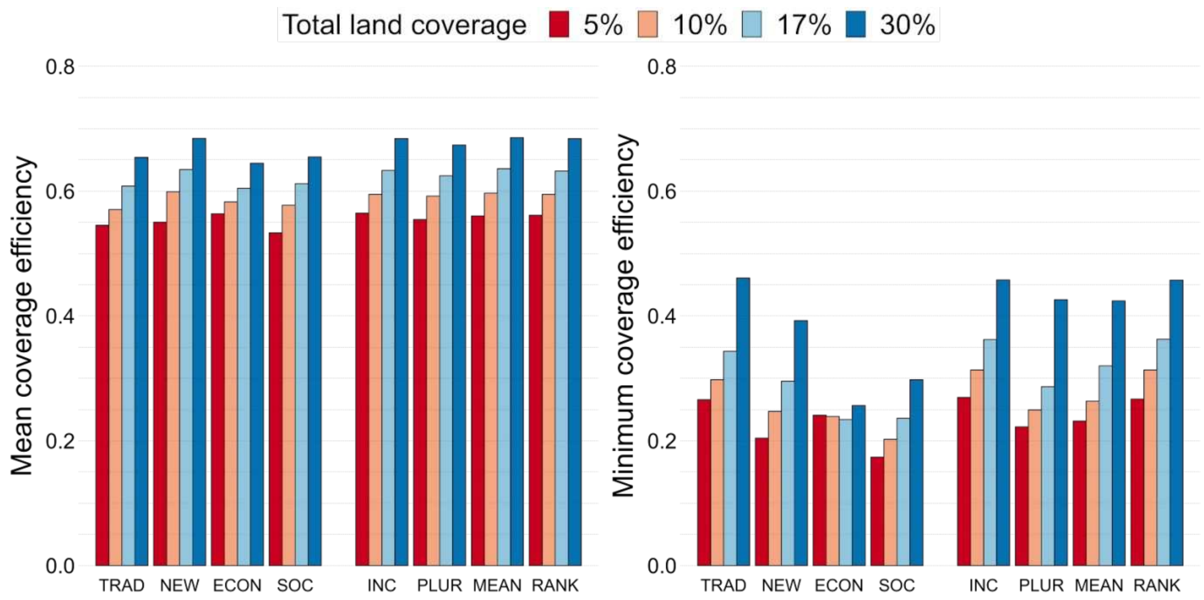

Figure S6 Mean and minimum feature coverage efficiency of *additive benefit function* prioritisation performance (TRAD – 'traditional', NEW – 'new conservation', ECON – 'international market ecocentrism', SOC – 'local social instrumentalism') and integration approach (Inclusive, and Pluralist conservation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

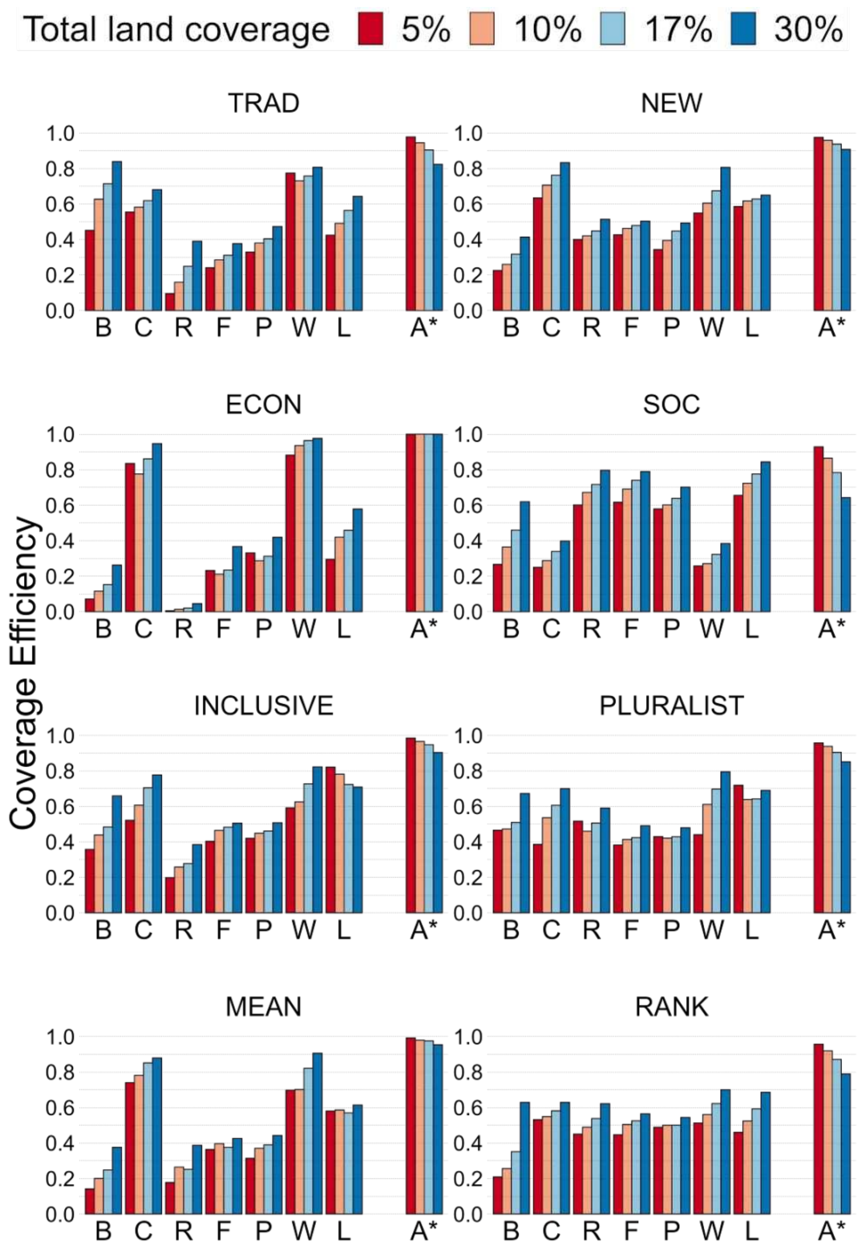

Figure S7 Efficiency of *core area zonation* prioritisation performance (TRAD – 'traditional', NEW – 'new conservation', ECON – 'international market ecocentrism', SOC – 'local social instrumentalism') and integration approach (inclusive, and pluralist conservation as well as two additional integration 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*).

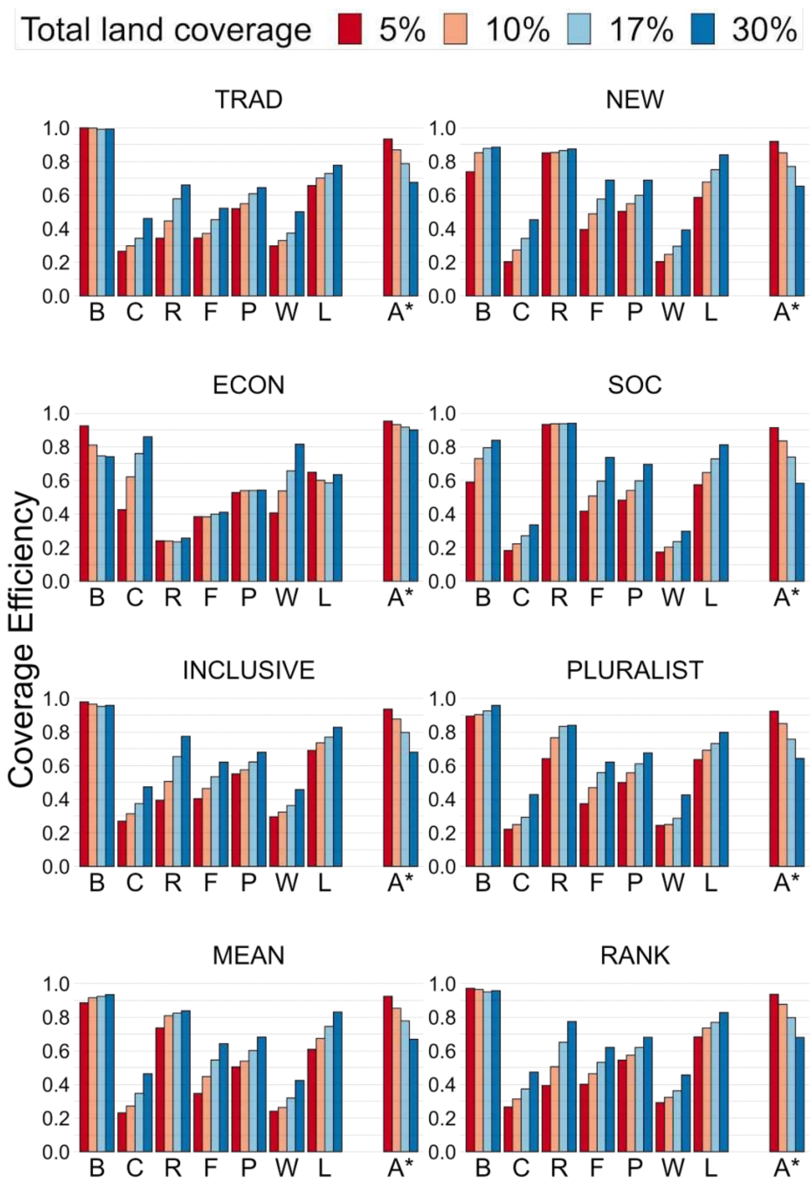

Figure S8 Efficiency of *additive benefit function* prioritisation performance (TRAD – 'traditional', NEW – 'new conservation', ECON – 'international market ecocentrism', SOC – 'local social instrumentalism') and integration approach (Inclusive, and Pluralist conservation as well as two additional integration 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 mulitplied 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.

	PC1 (0.601)	**PC2 (0.911)**	**PC3 (0.999)**	**PC4 (1.000)**
TRAD	-0.168	0.234	**-0.927**	0.240
NEW	**-0.658**	-0.325	0.205	0.647
ECON	**-0.693**	0.515	0.129	-0.487
SOC	-0.241	**-0.758**	-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 functio*n (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.

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INTRODUCTION

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

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

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

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

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

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

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

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

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

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

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

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

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

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METHODS

Feature layers

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

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

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

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

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

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

Viewpoint prioritisation

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

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

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

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

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

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

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

Viewpoint integration

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

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

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

Eq. 1

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

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

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

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

$$P_j = \cancel{\sum_{c=1}^{n_c} \left| r_{j_{1\times v}} \cdot W_{c_{v\times 1}} \right|} \sum_{c=1}^{n_c} \left| r_{j_{1\times v}} \cdot W_{c_{v\times 1}} \right|$$

Eq. 2

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

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

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

RESULTS

The viewpoint prioritisations selected different landscape priorities based upon their valued features (Figure 2). The 'traditional' conservation viewpoint priorities had the highest average proportion coverage of species distributions, primarily concentrated in ~~NW~~northwest Scotland and scattered landscapes in the south of England. Conversely 'local

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social instrumentalism' spatial priorities were focused in landscapes in England close to large conurbations, especially London, maximising recreational value but resulting in lower exclusion of agricultural land; as well as landscapes in ~~N~~north England, which delivered landscape aesthetic value and flood protection services. 'International market ecocentrism' priorities almost exclusively occurred in Scotland, and upland areas in Wales and northern England, driven by positive selection for carbon storage and avoiding the opportunity costs of more southerly productive farmland. The 'new' conservation spatial prioritisation selected landscapes appearing in both the 'international market ecocentrism' and 'local social instrumentalism' viewpoints, due the more balanced weightings across feature layers. These landscapes were primarily located in Scotland, upland areas in ~~N~~north England, and ~~SE~~southeast England close to London.

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

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

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

DISCUSSION

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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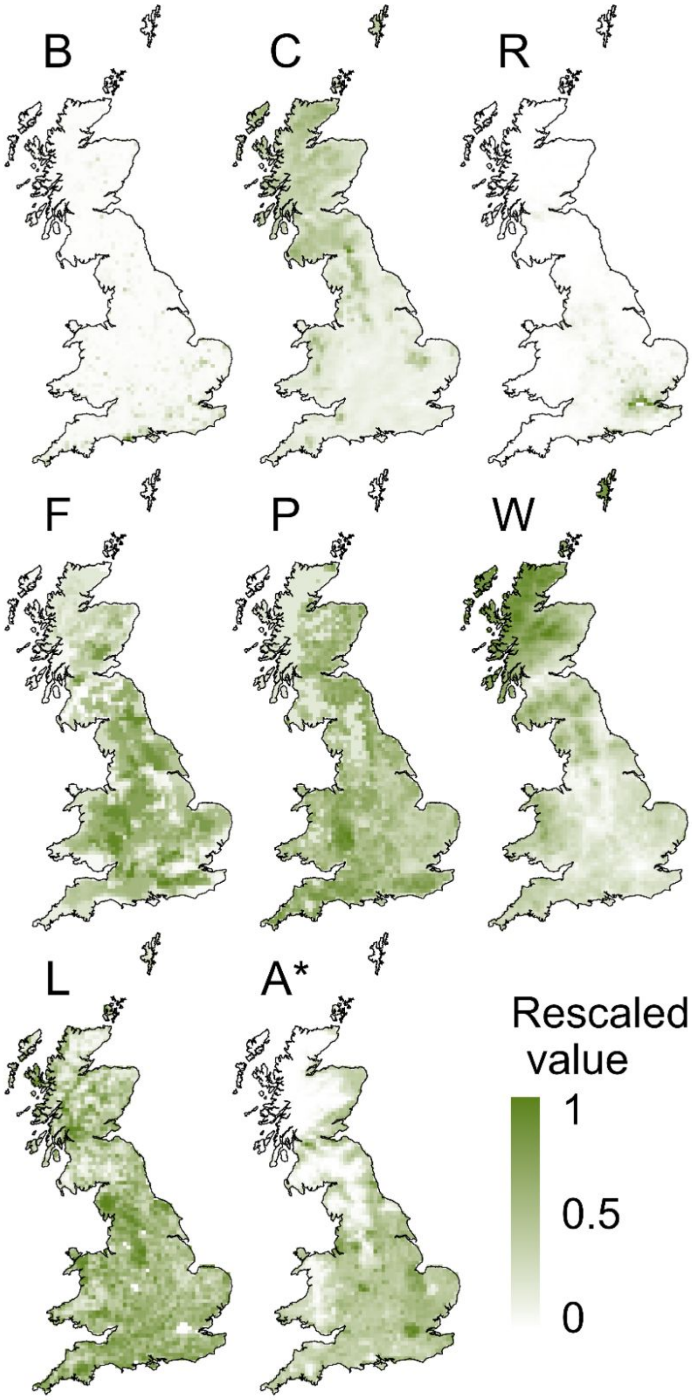

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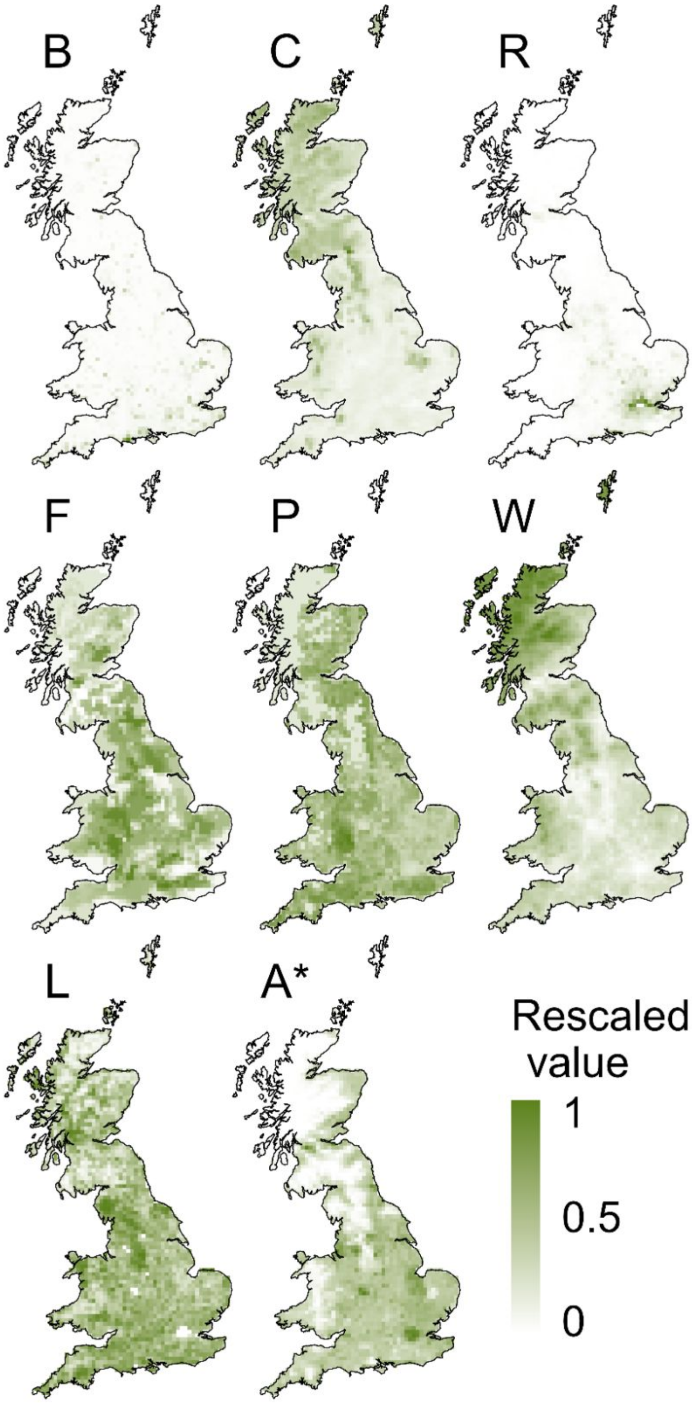

Figure 1 Rescaled ecosystem service, biodiversity and socio-environmental value feature layers included within the analysis including; mean priority species distribution proportion coverage (B), carbon storage (C), recreation (R), flood regulation (F), pollination (P), wilderness (W), landscape aesthetic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

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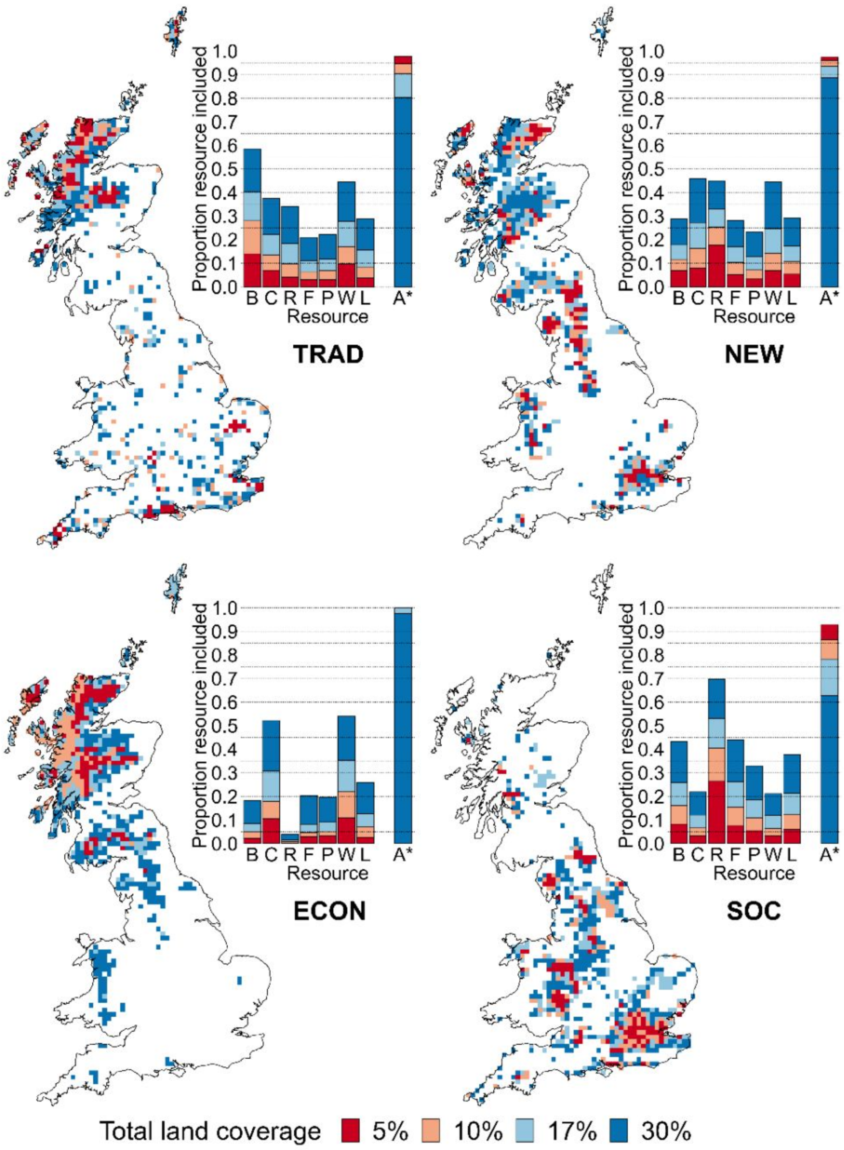

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Resource
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NEW
ECON
SOC
Total land coverage 5% 10% 17% 30%

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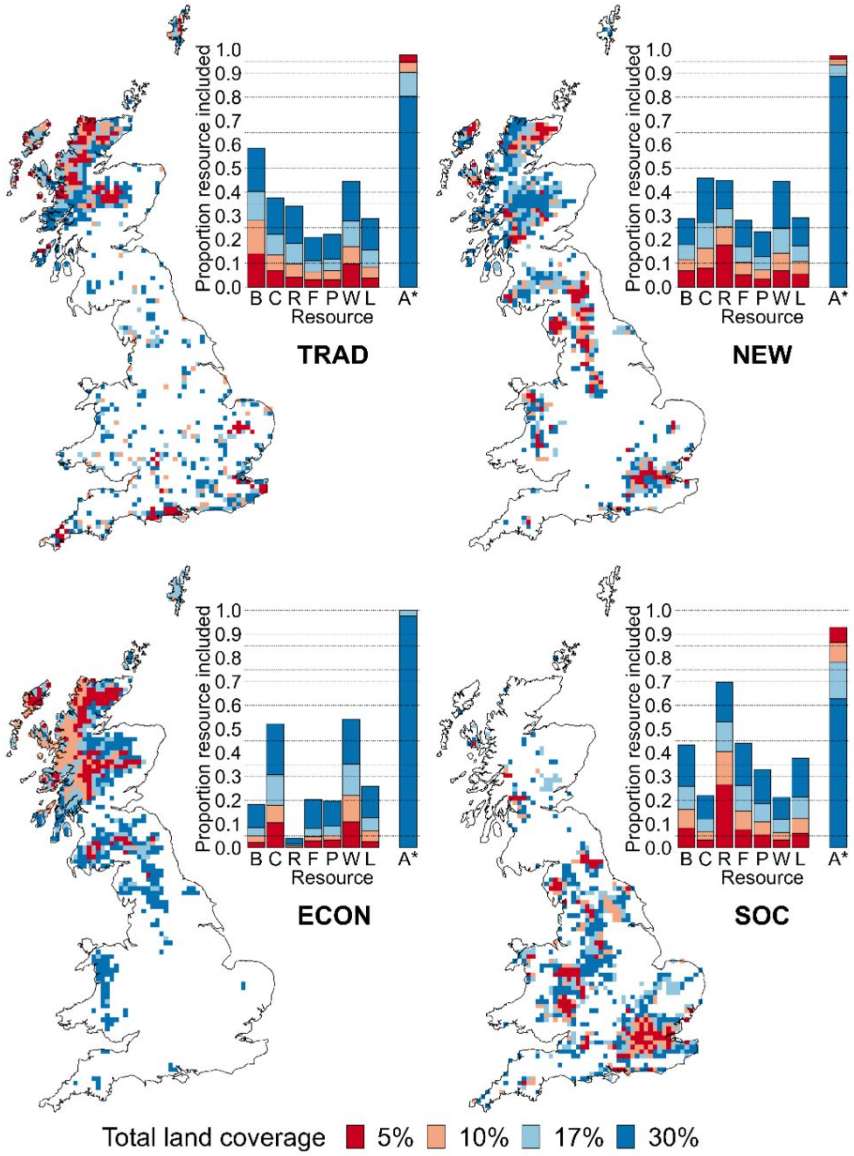

Figure 2 Feature coverage using spatial prioritisation for each of the four viewpoints; TRAD – 'traditional', NEW – 'new' conservation, ECON – 'international market ecocentrism', SOC – 'local social instrumentalism'. Maps indicate the priority areas for different coverage thresholds, and corresponding 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.

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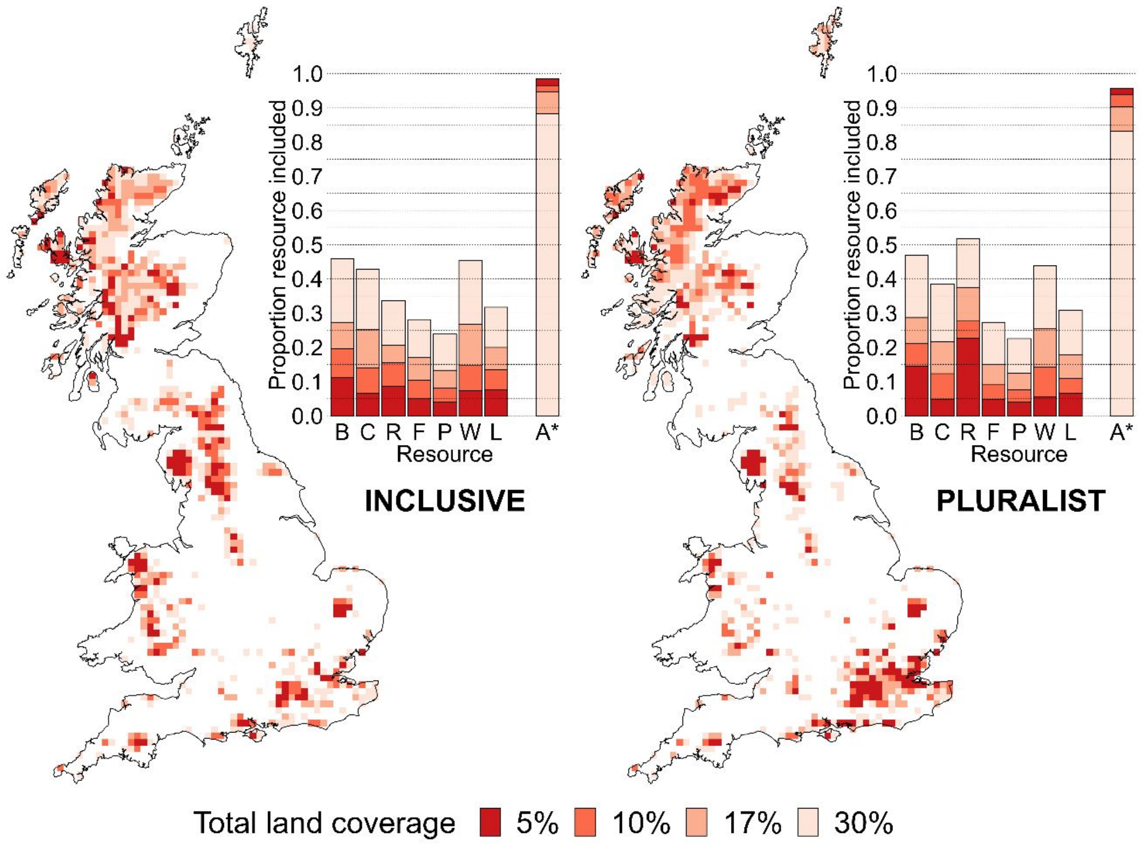
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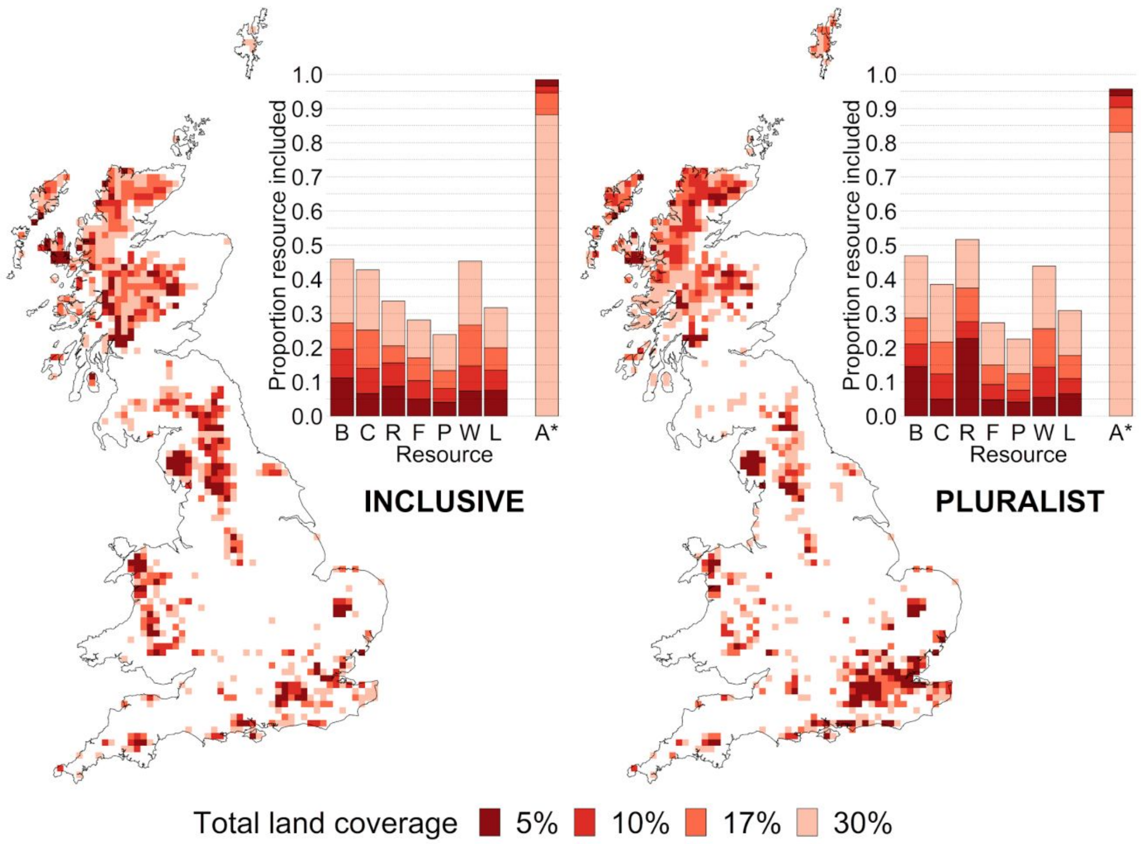
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Total land coverage 5% 10% 17% 30%

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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.

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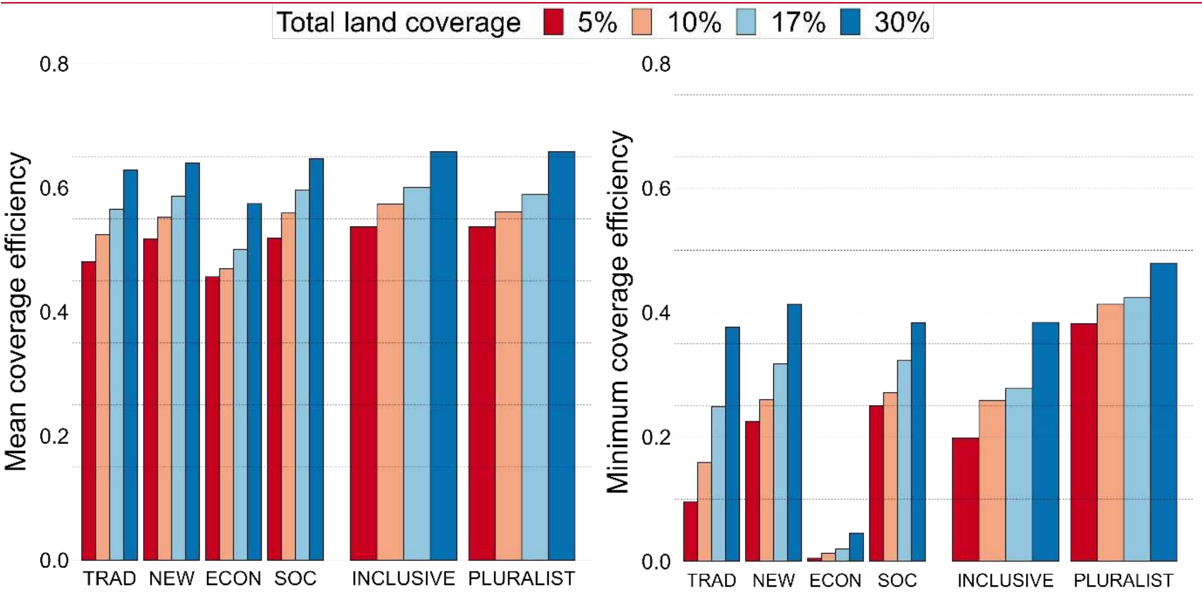

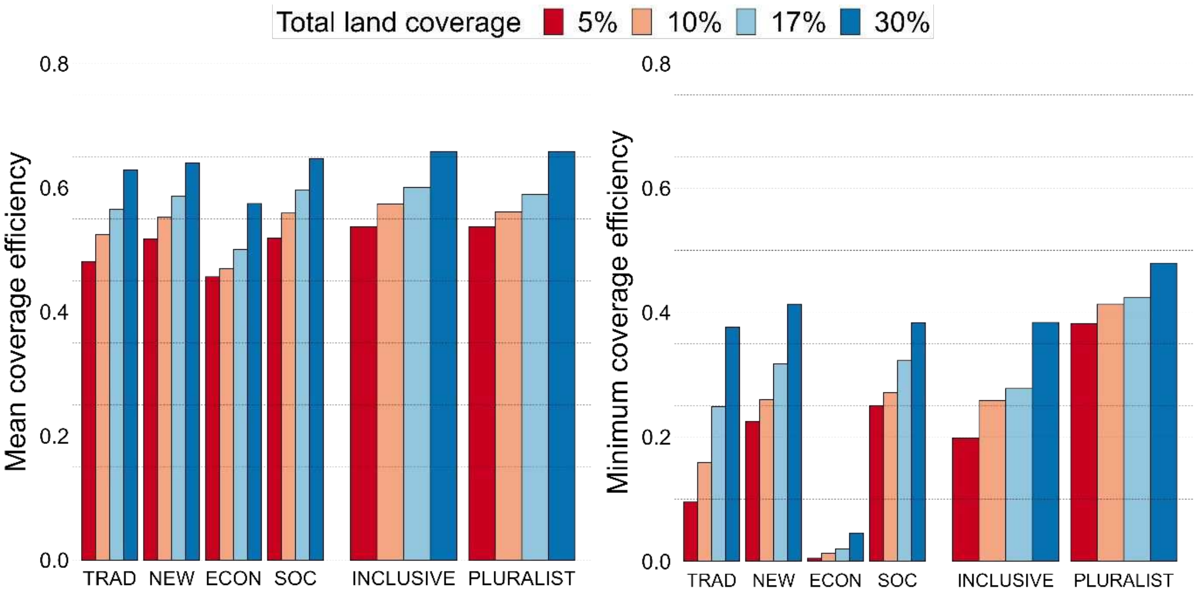

Figure 4 Mean and minimum feature coverage efficiency of viewpoint prioritisation performance (TRAD – 'traditional', NEW – 'new' conservation, ECON – 'international market ecocentrism', SOC – 'local social instrumentalism') and integration approach (inclusive and pluralist conservation). Efficiency is 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.

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Table 1 Definitions of caricature viewpoints and viewpoint integration approaches.

Conservation viewpoint	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)	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). Weightings: species distributions and wilderness.
'New' (NEW)	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). Weightings: widest scope of the four viewpoints, including species and all economic and social value data, apart from wilderness.
International market ecocentrism (ECON)	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). Weightings: agricultural value (avoid) and related pollination service flow, as well as carbon storage and species distributions.
Local social instrumentalism (SOC)	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). 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.
Viewpoint integration approach	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.

Feature	**Traditional conservation (TRAD)**	**'New' conservation (NEW)**	**International market ecocentrism (ECON)**	**Local social instrumentalism (SOC)**
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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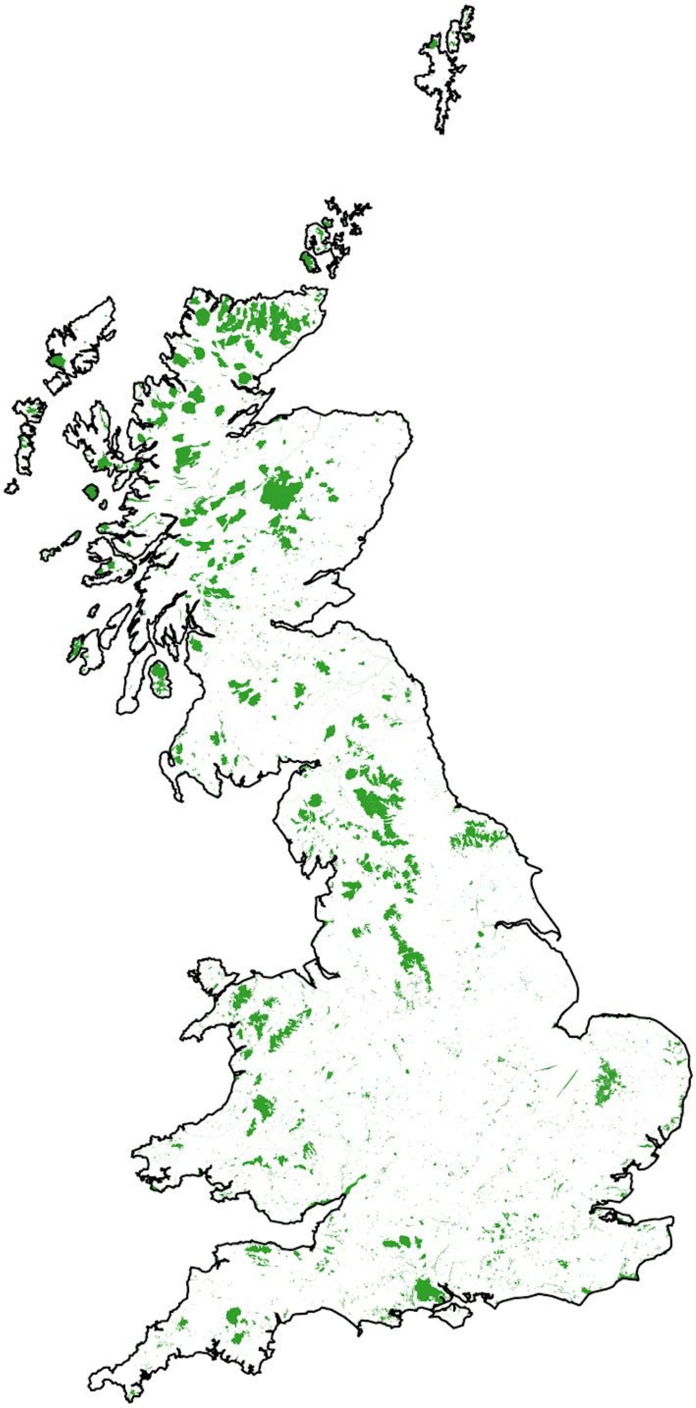
SUPPLEMENTARY MATERIALS



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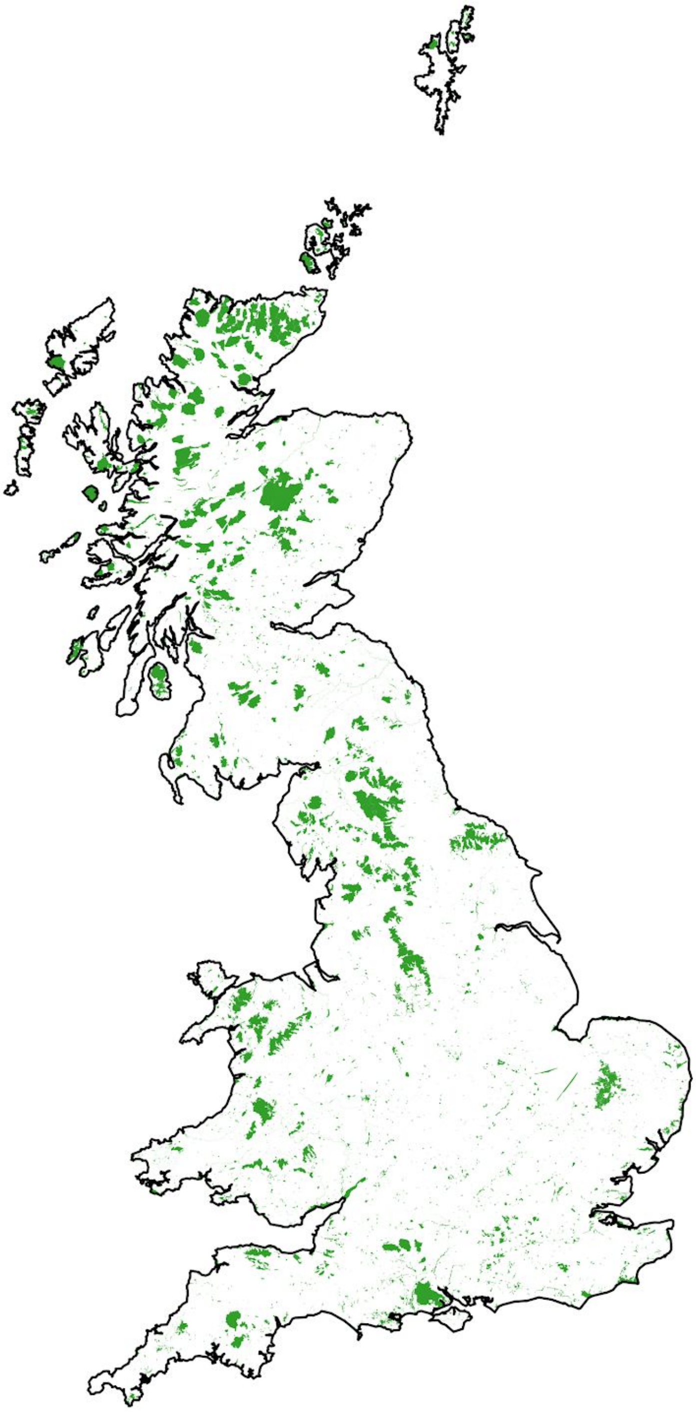

Figure S1 Protected areas included within the analysis: Sites of Special Scientific Interest (SSSI) and National Nature Reserves (NNR) designated at the time of the study.

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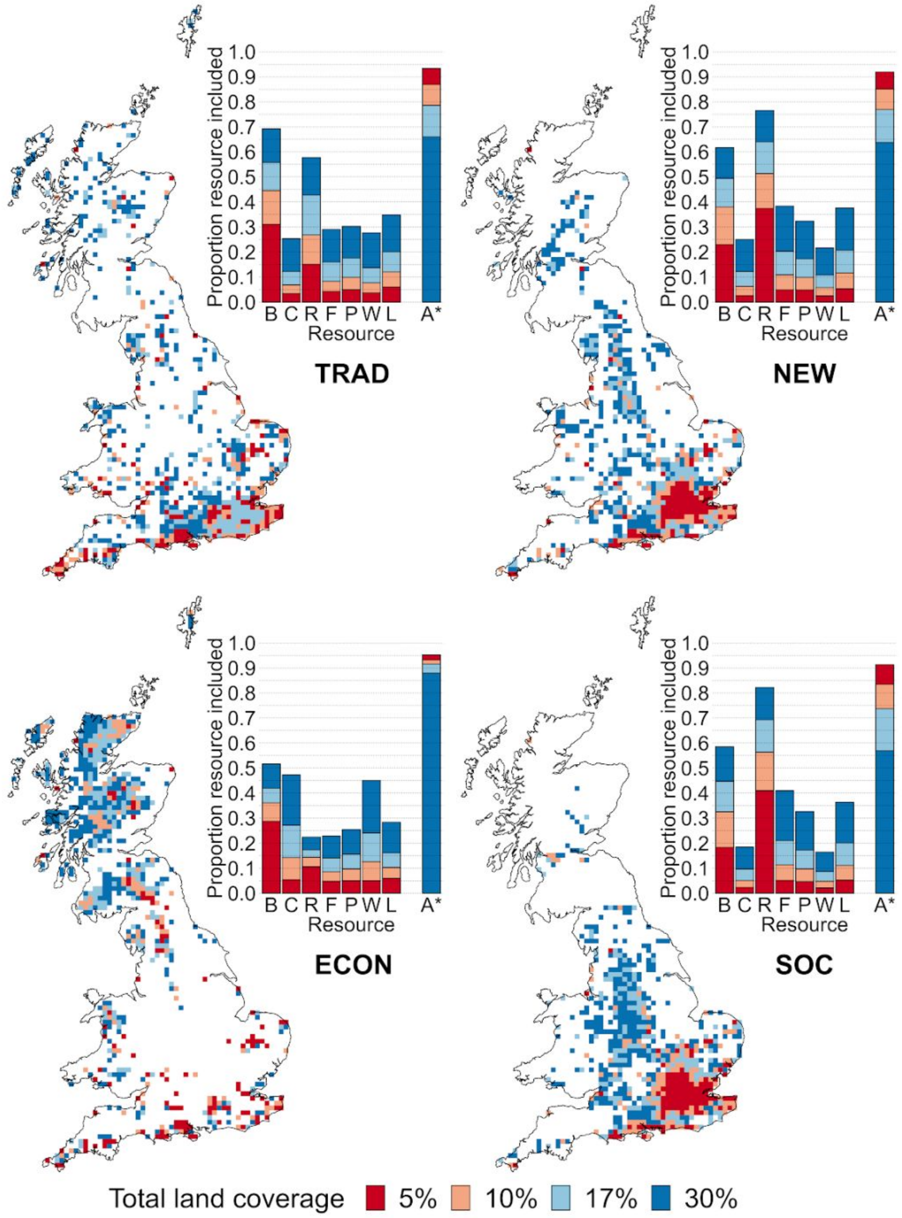
Proportion resource included
1.0
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0.0
B C R F P W L A*
Resource
TRAD
NEW
ECON
SOC
Total land coverage 5% 10% 17% 30%

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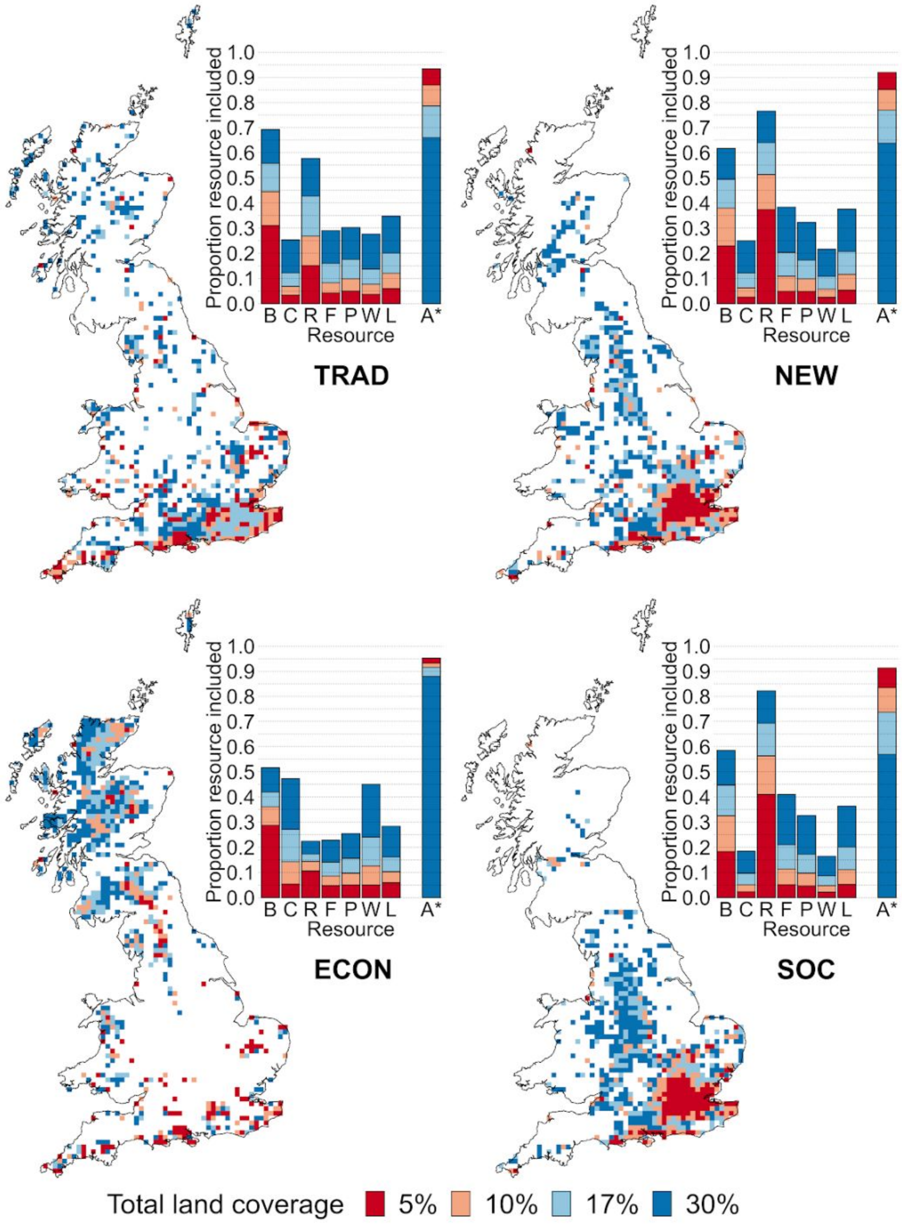

Figure S2 Feature coverage using spatial prioritisation for each of the four viewpoints using the *additive benefit function* prioritisation method; TRAD – 'traditional', NEW – 'new conservation', ECON – 'international market ecocentrism', SOC – 'local social instrumentalism'. Maps indicate the priority 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)

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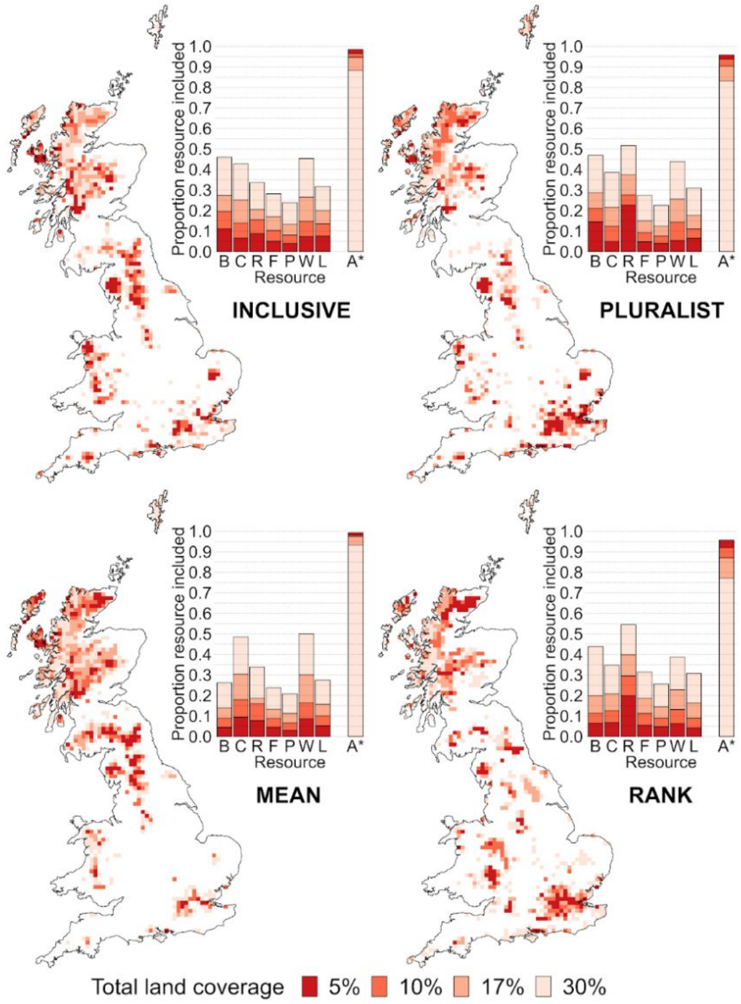
and agricultural/land value (A*). A* indicates the only negative weighting where proportion excluded, not included, is shown and so higher land coverage results in lower proportion excluded.



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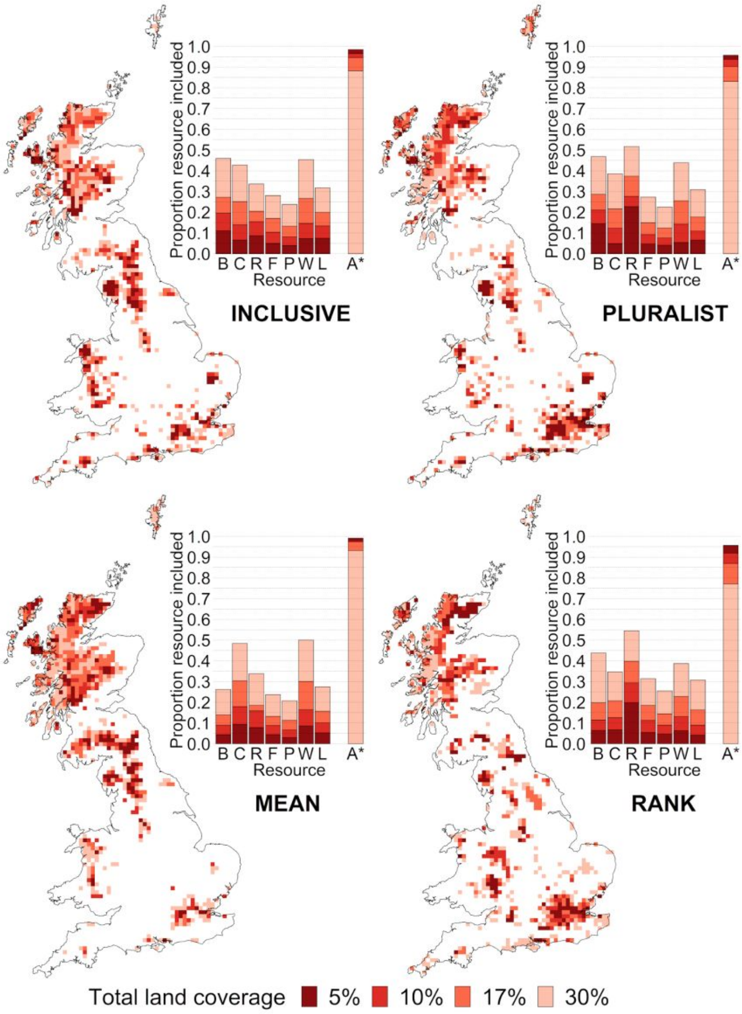

Figure S3 Spatially aggregating the four conservation viewpoint priorities (TRAD – 'traditional', NEW – 'new conservation', ECON – 'international market ecocentrism', SOC – 'local social instrumentalism') using the *core area zonation* prioritisation method. We used inclusive (vote counting) and pluralist (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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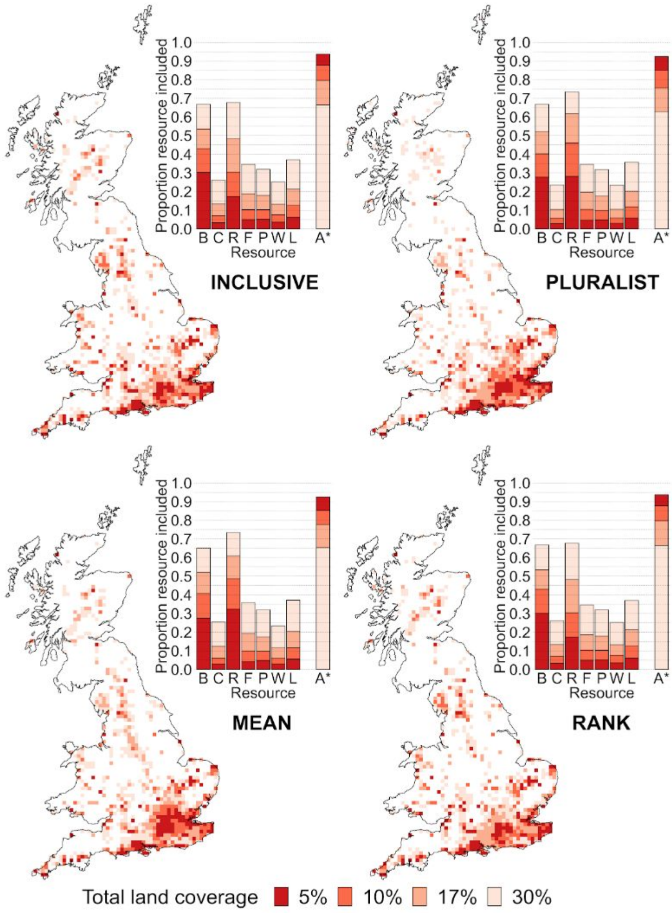
agricultural/land value (A*). A* indicates the only negative weighting where proportion excluded, not included, is shown and so higher land coverage results in lower proportion excluded.



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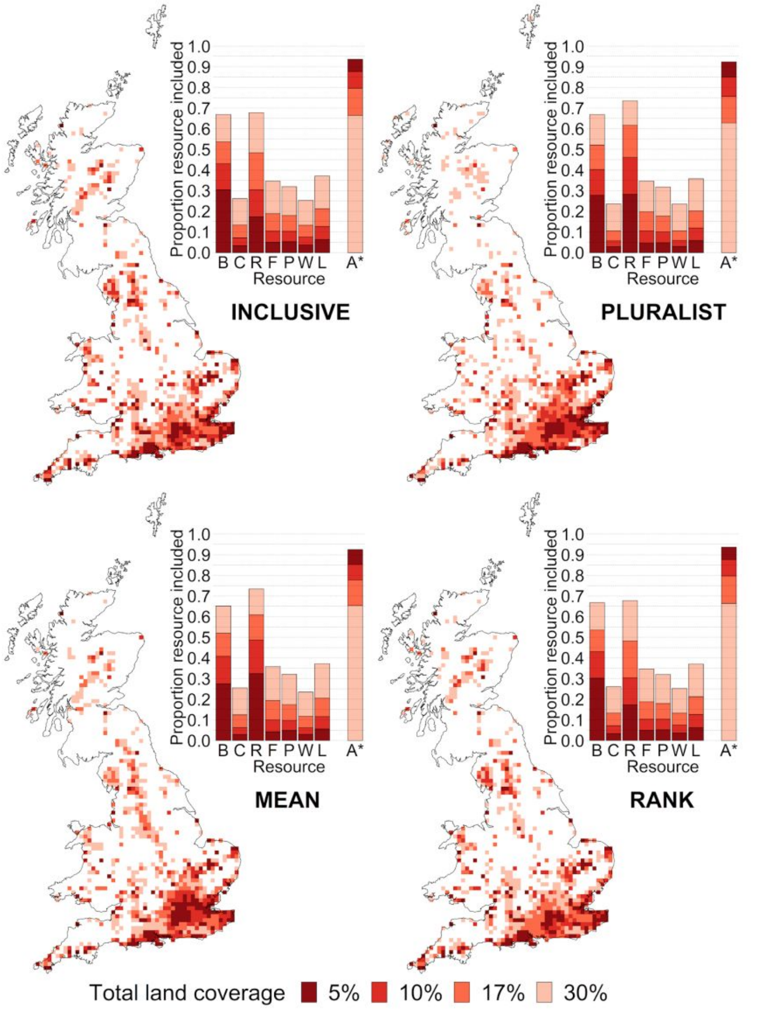

Figure S4 Spatially aggregating the four conservation viewpoint priorities (TRAD – 'traditional', NEW – 'new conservation', ECON – 'international market ecocentrism', SOC – 'local social instrumentalism') using the *additive benefit function* prioritisation method. We used inclusive (vote counting) and 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.

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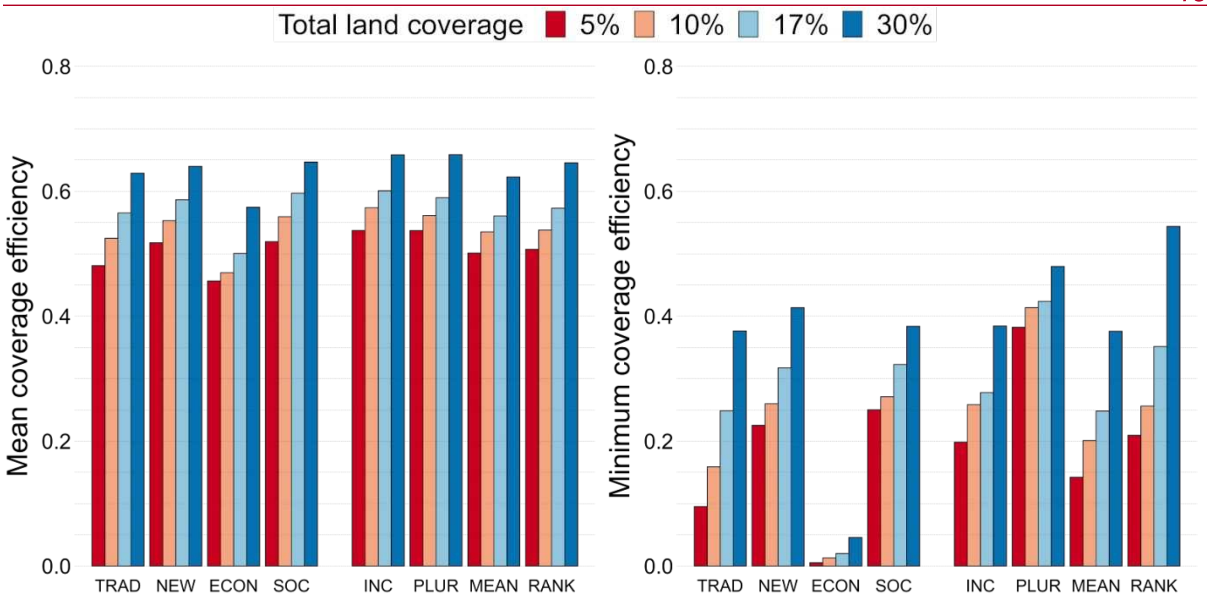

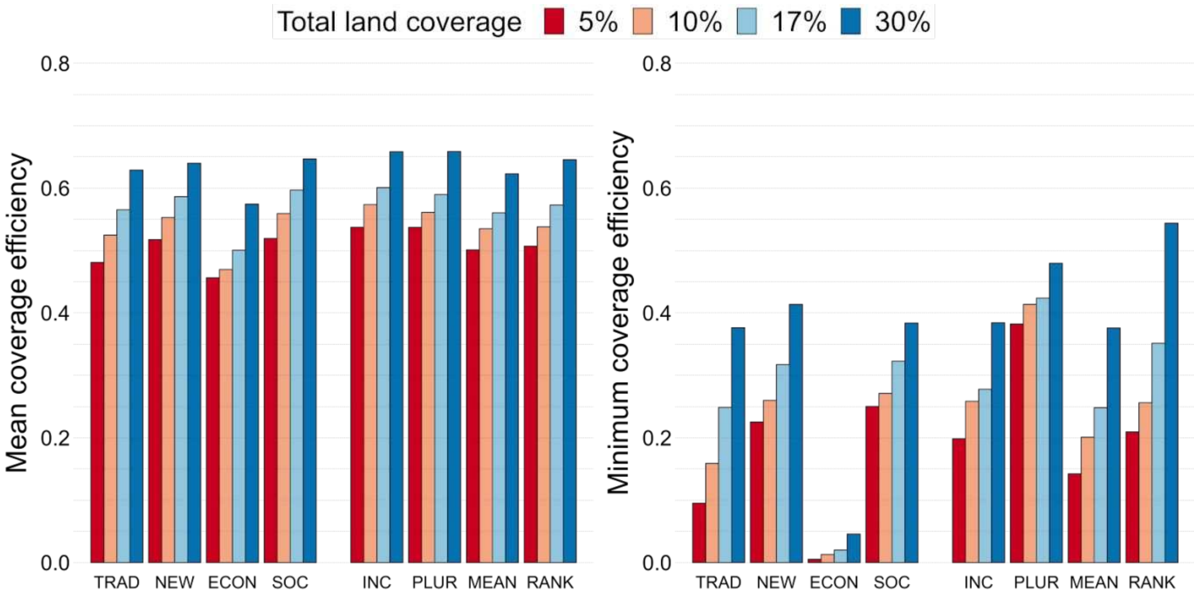

Figure S5 Mean and minimum feature coverage efficiency of core area zonation prioritisation performance (TRAD – 'traditional', NEW – 'new conservation', ECON – 'international market ecocentrism', SOC – 'local social instrumentalism') and integration approach (Inclusive and Pluralist conservation, as 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),

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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.

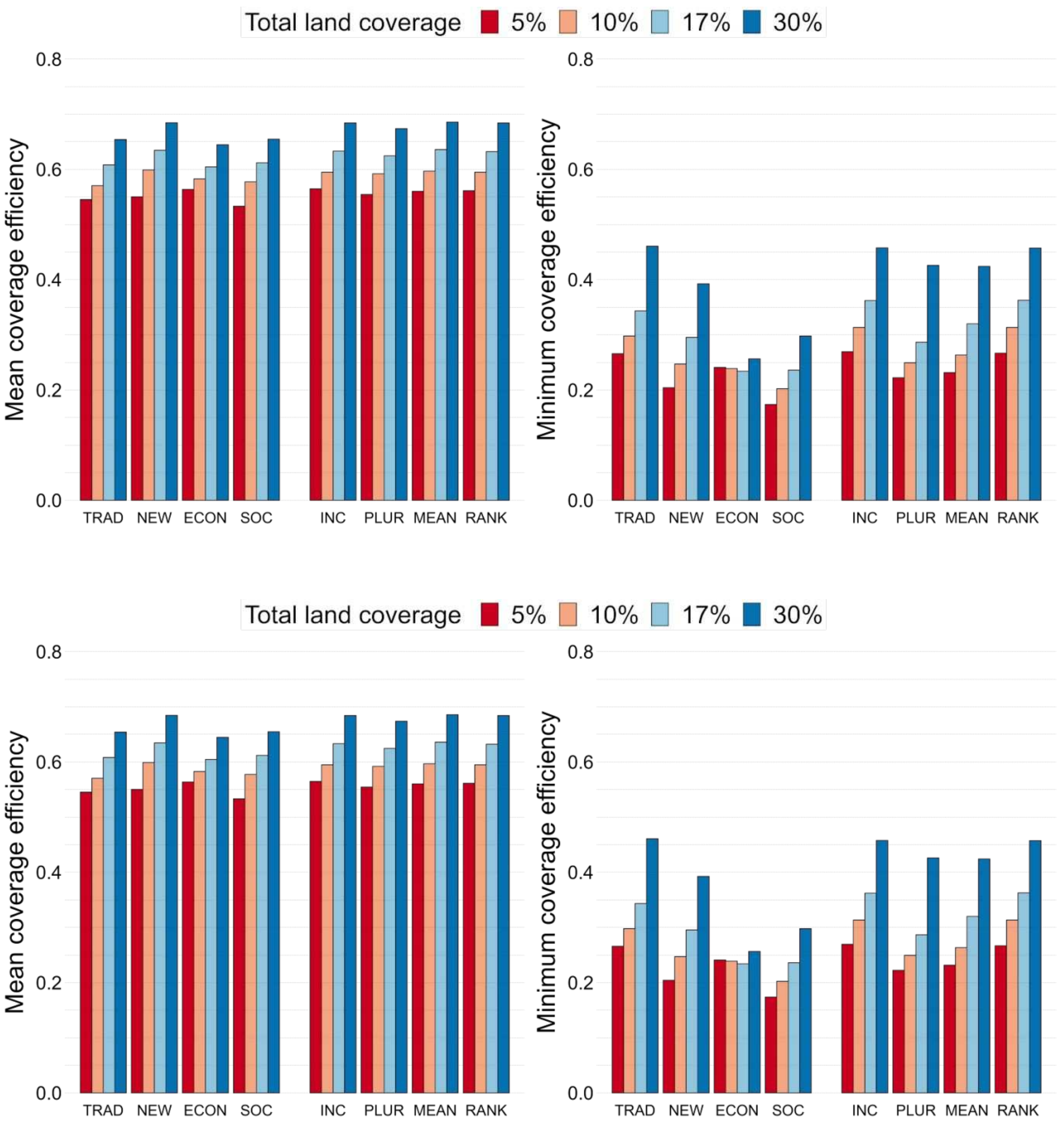

Figure S6 Mean and minimum feature coverage efficiency of *additive benefit function* prioritisation performance (TRAD – ‘traditional’, NEW – ‘new conservation’, ECON – ‘international market ecocentrism’, SOC – ‘local social instrumentalism’) and integration approach (Inclusive, and

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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.

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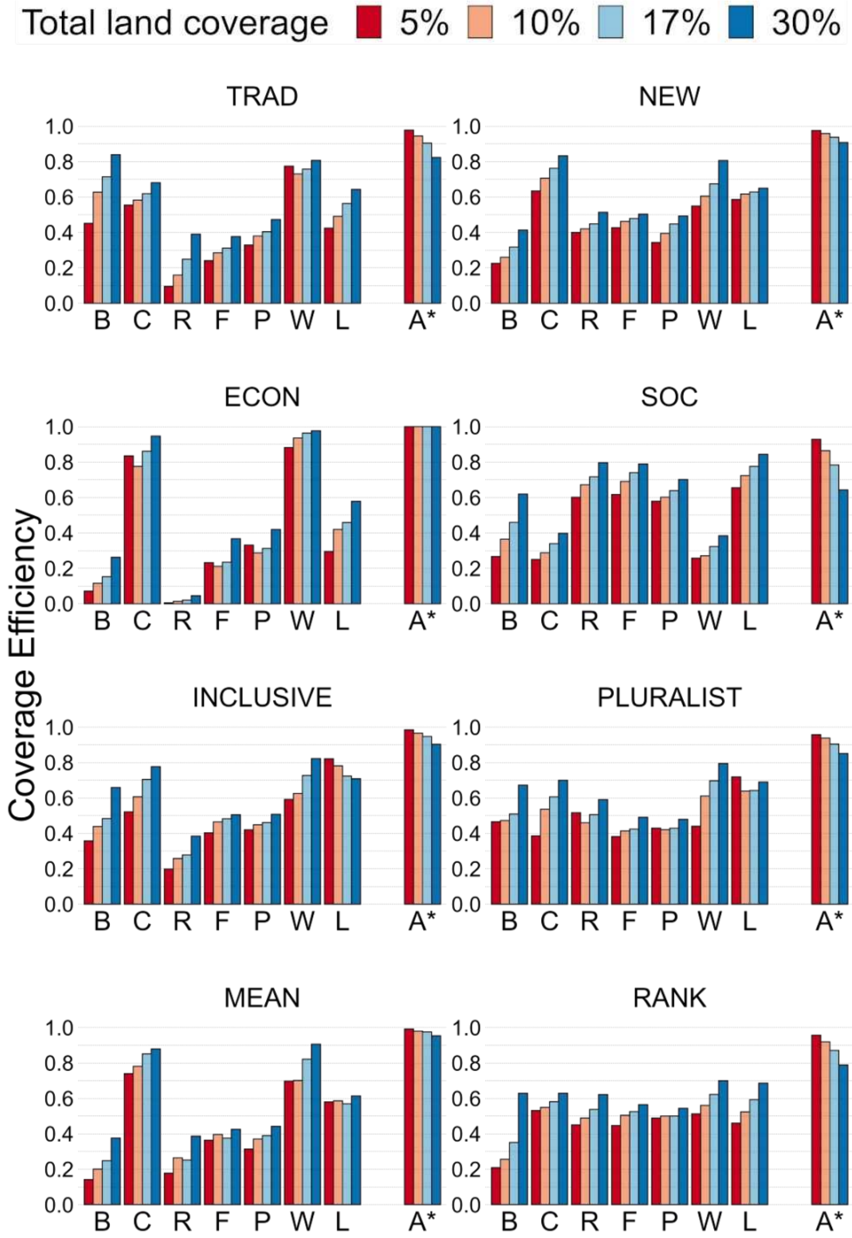
Total land coverage 5% 10% 17% 30%
TRAD
NEW
ECON
SOC
INCLUSIVE
PLURALIST
MEAN
RANK
Coverage Efficiency
1.0
0.8
0.6
0.4
0.2
0.0
B C R F P W L A*

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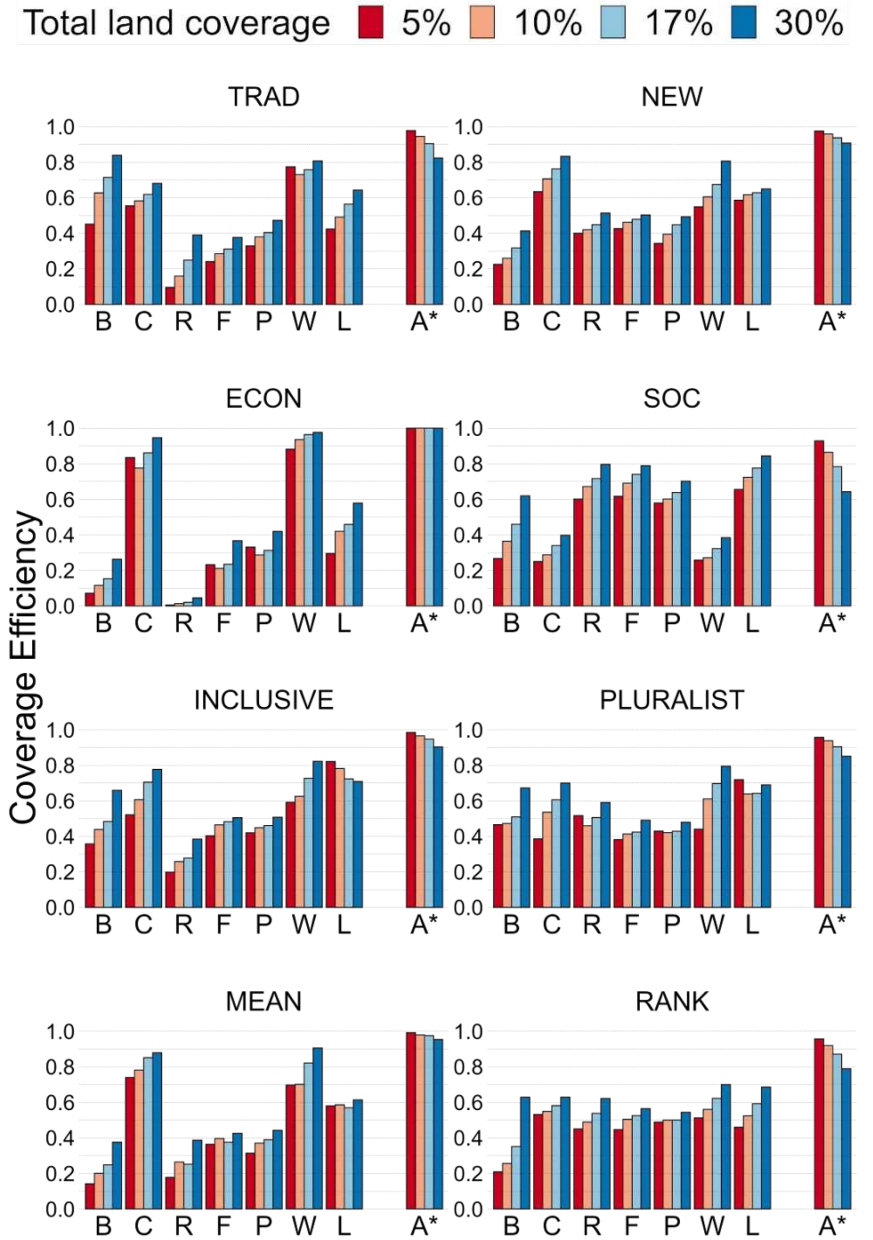

Figure S7 Efficiency of *core area zonation* prioritisation performance (TRAD – 'traditional', NEW – 'new conservation', ECON – 'international market ecocentrism', SOC – 'local social instrumentalism') and integration approach (inclusive, and pluralist conservation as well as two additional integration 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),

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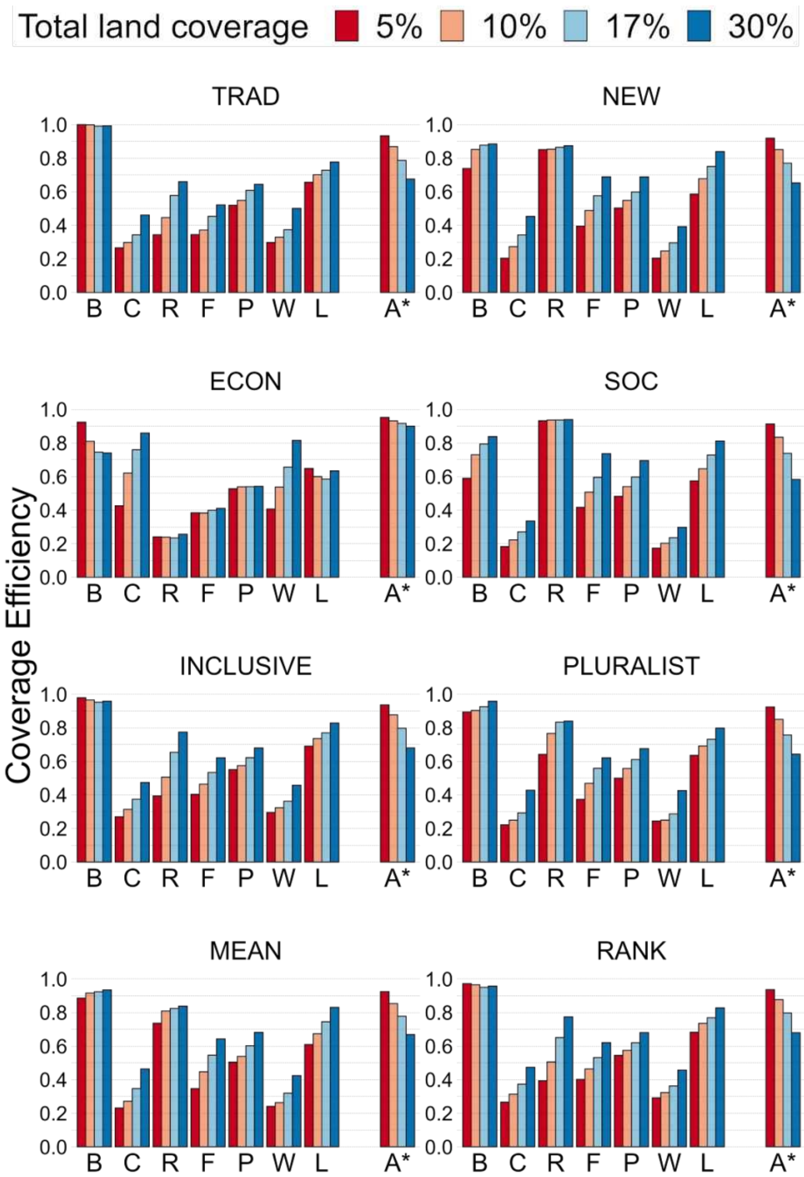
recreation (R), flood regulation (F), pollination (P), wilderness (W), landscape aesthetic value (L) and agricultural/land value (A*).



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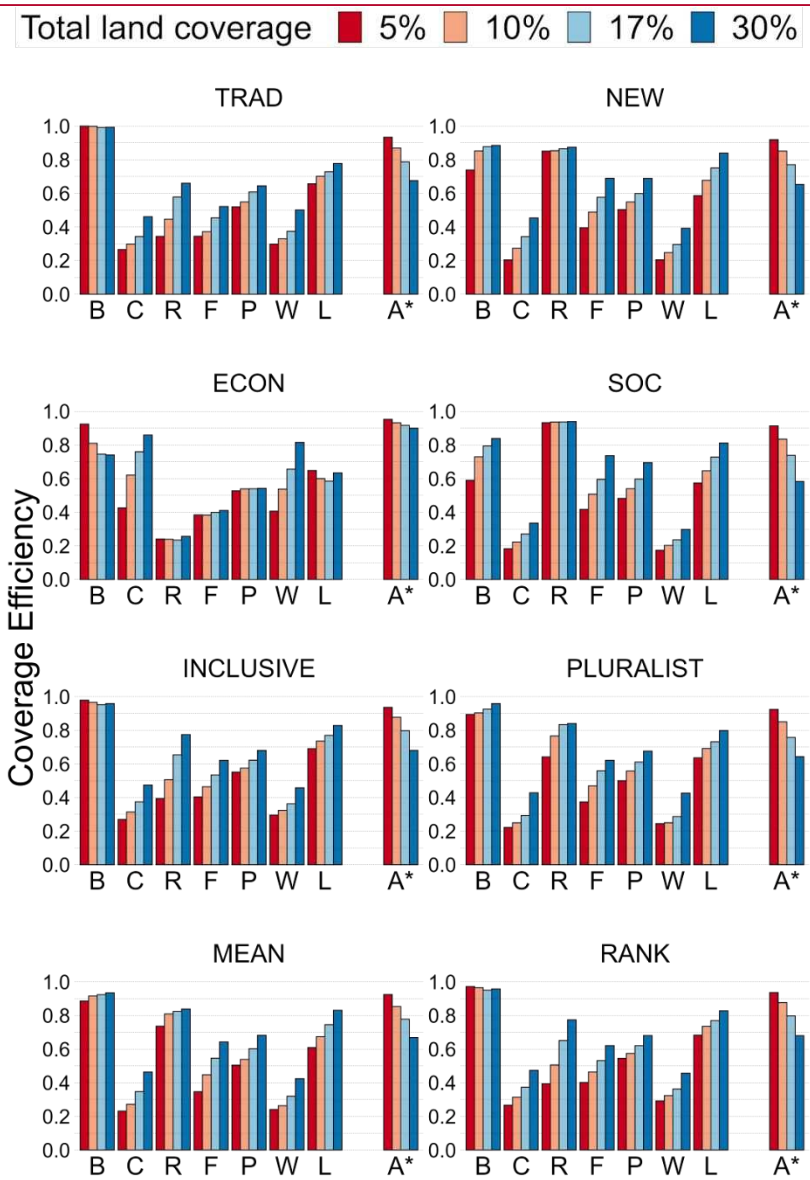

Figure S8 Efficiency of *additive benefit function* prioritisation performance (TRAD – 'traditional', NEW – 'new conservation', ECON – 'international market ecocentrism', SOC – 'local social instrumentalism') and integration approach (Inclusive, and Pluralist conservation as well as two additional integration 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 mulitplied 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.

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

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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 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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