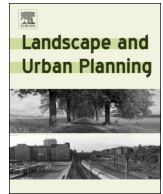




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Research Paper

Natural capital and the poor in England: Towards an environmental justice analysis of ecosystem services in a high income country

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ABSTRACT

Poorer communities tend to be located within lower quality natural environments, experiencing greater environmental burdens and fewer environmental amenities. To date, analysis of environmental inequalities has focussed on pollution, with less attention given to natural environment benefits that support human wellbeing. Here, the ecosystem service concept which identifies these benefits, and the natural capital (NC) which provides them, is applied within environmental inequality assessment. For England, 325 local authority districts were classified based on 14 indicators of NC, and the level of deprivation of districts within each class compared. Districts with extensive woodland or agriculture are the least deprived. The most deprived districts tend to be urban areas with lower extent and quality of NC, coastal districts, and rural uplands with extensive coverage of various higher quality NC. These findings demonstrate that the distribution of NC varies by social deprivation, with implications for social inequities and sustainable management of NC. However, whilst higher deprivation is often associated with a lower extent and quality of NC, this pattern is not consistent for all NC types or places. Given the lack of a consistent pattern of inequality nationally, this implies that equitable management of ecosystems should be driven at a local level. To achieve this, the relationship between environmental benefits and deprivation should be assessed at this level and analysis should move beyond NC to address the ecosystem services that flow from it.

1. Introduction

1.1. Natural capital, ecosystem services, and social justice

Natural capital (NC) as “stock of natural resources or environmental assets” (De Groot, Van der Perk, Chiesura, & van Vliet, 2003, p.188) and the ecosystem services (ES) it provides, is critical to people’s health and well-being (Fig. 1). Whilst dependence on the natural environment is widely acknowledged, universal access to high quality environments which support the health and wellbeing of everyone is lacking.

An extensive environmental justice (EJ) literature reveals that environmental quality is socially distributed, with low environmental quality and high environmental hazard typically found in minority and economically disadvantaged communities. Such patterns were first revealed in the USA (e.g. Freeman, 1972), and subsequently evidenced for many other countries (Walker, 2012).

Interest in EJ has traditionally focussed on environmental ‘bads’, but a broader conception of EJ has subsequently emerged which also

considers environmental ‘goods’ (Wolch, Byrne, & Newell, 2014). Analysis of the social distribution of such environmental benefits has been undertaken in many countries, but remains more limited than that of environmental burdens, and includes access to urban parks (Xiao, Wang, Li, & Tang, 2017), urban tree cover (Conway & Bourne, 2013), urban greenspace (Pham, Apparicio, Seguin, Landry, & Gagnon, 2012), bluespace (Raymond, Gottwald, Kuoppa, & Kytta, 2016), woodland (Morris et al., 2011), biodiversity (Davis et al., 2012) and tranquil places (Mitchell & Norman, 2012).

Conceiving of the environment as a source of benefit aligns with the concept of NC as a source of goods and services supporting health and wellbeing (see Missemmer, 2018 for a review), and suggests there is a clear case for addressing NC/ES and EJ within a common framework. Indeed, joint consideration of ES and EJ encompasses the ecosystem approach advocated by the Convention on Biological Diversity (CBD, 2004). Discussion of the social dimension within the ES discourse has focussed on its importance for poorer subsistence-based global communities (e.g. Sikor, 2013). However, the importance of fairness in all

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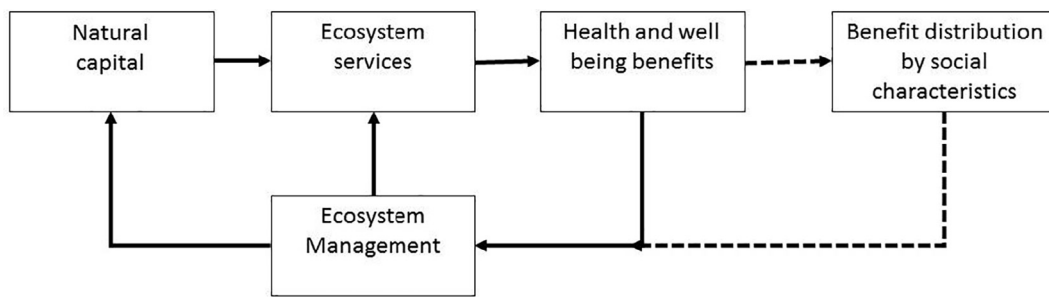


Fig. 1. Framework showing how the social distributions of benefits integrates within established linkages between natural capital, ecosystem services and humans as beneficiaries and actors.

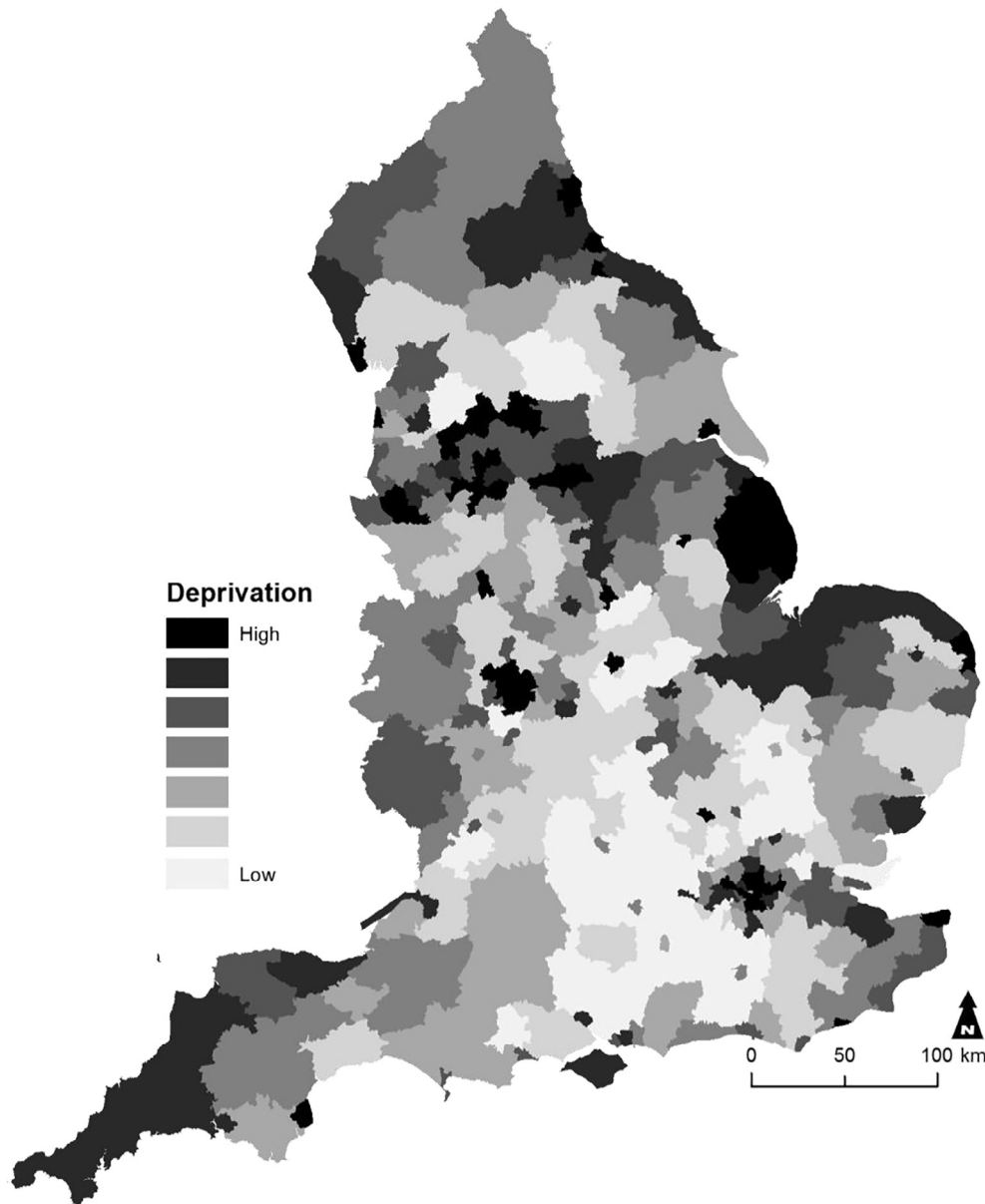


Fig. 2. Deprivation in English Local Authority Districts (IMD average ranks). ONS (2015).

valuations of ES was highlighted within Gretchen Daily’s seminal ‘Nature’s Services’ (Costanza & Folke, 1997) and more recently by Berbes-Blazquez, Oestreicher, Mertens, & Saint-Charles, 2014; Ernstson, 2013. Whilst recent empirical research incorporates a social dimension within ES analysis, the focus is on its relevance for production of and

demand for ES (e.g. Dittrich, Seppelt, Václavík, & Cord, 2017; Hamann, Biggs, & Reyers, 2016). Explicit consideration of social equity has been dominated by studies of participatory decision making (e.g. Wilson & Howarth, 2002), in equity appraisal of payment for ecosystem service schemes (McDermott, Mahanty, & Schreckenberg, 2013) and more

recently by qualitative research exploring structural and individual factors which mediate the distribution of ES (Horcea-Milcu, Leventon, Hanspach, & Fischer, 2016). There remains rather limited quantitative analysis of the socio-economic distribution of ES (Daw, Brown, Rosendo, & Pomeroy, 2011), although this is an emerging area of work.

Regulating service supply (as air pollutant removal, carbon sequestration and vegetative cooling) has been shown to increase with socio-economic status and fewer minority populations (Escobedo, Clerici, Staudhammer, & Corzo, 2015; Jenerette, Harlan, Stefanov, & Martin, 2011; Soto, Escobedo, Adams, & Blanco, 2016). However, the reliance of poorer communities on their natural surroundings is exemplified by greater ES use in low-income areas, as found by Hamann et al. (2016) in South Africa. Similarly for the Niger Delta, Nigeria, Adekola, Mitchell, and Grainger (2015) show how people directly depend on local ES and so suffer most from the environmental damage of oil production, yet benefit least from oil revenues. Such scale dependent distributions are also revealed by Gomes Lopes, dos Santos Bento, Correia Crisovao and Oliveira Baptista (2015)'s study in Portugal, where local people receive 45% of benefits derived from common land ecosystems, whilst 15% of benefits flow to global beneficiaries.

To date however, relatively few analyses have examined inequality in ES distribution (Lakerveld, Lele, Crane, Fortuin, & Springate-Baginski, 2015). The body of evidence remains too small and heterogeneous to draw general conclusions, but available studies do reveal asymmetry in distribution of ES benefits. They also point to the importance of socio-economic factors in ES provision (via management of NC), and raise questions about how to assess equity in the context of ES which are so scale dependent. Further work is necessary, addressing varied contexts (NC, ES, social factors, places and landscapes), and including high income countries where people are less directly dependent upon the supporting environment. It is reasonable to assume that NC (and so ES) is socially distributed in high income countries, but this remains to be tested.

1.2. Natural capital and environmental justice in England, UK

European EJ research began in Scotland, followed by England (Laurent, 2011), and focused on analyses of environmental hazard and social deprivation. Environmental inequalities have been found but vary by environmental measure (Wheeler, 2004). Equity analysis of environmental benefits has focussed on accessible greenspace, including parks and woodland (Barbosa et al., 2007; O'Brien & Morris, 2014), and outdoor recreational space (Natural England., 2015), concluding that ethnic minorities and people of lower socio-economic status visit greenspace less often due to it being of a lower quality or less accessible. Morse, Vogiatzakis, and Griffiths (2011) found lower quality countryside was associated with higher deprivation. Whilst evidence for an unequal social distribution of greenspace in England exists, it is not possible to draw conclusions on the distribution of the fuller range of NC. Prior inequality studies do not adopt a NC/ES framework and where relevant features are included, their social distribution is masked by the inclusion of other environmental metrics.

UK Government, as conveyed through the 25-year Environment Plan, aims to address environmental inequalities: "we want to ensure an equal distribution of environmental benefits, resources and opportunities" (HMG, 2018, p.16). It also seeks greater use of the NC approach in managing the environment, building on a wealth of existing work including the National Ecosystem Assessment (UKNEA, 2011), national monetary estimates of NC (ONS, 2015a) and reporting by the National Capital Committee. To date, work nationally has focussed on accounting for aggregate NC, based on the principle of 'environmental net gain' (HMG, 2018, Chap 1), and the distributional concerns are not yet fully embedded in the UK NC-ES discourse.

The UK can thus be characterised as a country with substantial interest in EJ, but with analyses that neglect NC/ES, and conversely substantial interest in NC and the benefits to people from ES, but with

little analysis of how those benefits are socially distributed. If England's NC is to be managed for all, it is important to develop an understanding of how planning and policy on environment and development might alter flows of benefits to people.

Distributive analysis of NC provides an essential first step in understanding how that NC may be equitably managed. Here we conduct such analysis nationally for England, aiming to provide a platform from which to build a more in depth understanding of the social distribution of ES derived from that NC, a more complex task due to spatial flows and scale dependencies.

Section 2 introduces the study area and describes the selection and mapping of indicators of NC and social deprivation. Section 3 discusses methods, detailing how NC is aggregated in space, then related to social deprivation. Section 4 presents results showing how NC varies by deprivation, and section 5 discusses implications for policy that seeks to manage NC in an efficient, socially just manner.

2. Study area and data

2.1. England's natural environment

Key features of England's landscape are summarised in Table 1. State of the environment reporting (ONS., 2015b; UKNEA, 2011) reveals a mixed picture with indicators variously revealing improving status (e.g. surface water quality, greenhouse gas emission), little change (wetland birds), or continued decline (farmland birds). A general trend for an increase in cultural and regulating ES and some decreases in provisioning services has been observed from 1993 to 2012 (Dick et al., 2016).

2.2. Natural capital indicators

For the purposes of our study, the NC assets of interest are those that give rise to beneficial ES. Specifically, species, ecological communities, soils, freshwaters, land, natural processes and function (following Mace, Hails, Cryle, Harlow, & Clarke, 2015). We exclude oceans from the offset since these are beyond the extent of social data, although we acknowledge that they, alongside other external NC assets, provide goods and services to people within the study area.

NC indicators have largely been developed for monetary valuation (Costanza et al., 1997; ONS, 2015a) or for assessing its criticality (De Groot et al., 2003; Mace et al., 2015). To quantify how NC is distributed socially across England, we require spatially disaggregated, objective and relative measures of NC; a monetary value is not required. Existing indicators may not be spatially explicit (i.e. for monetary valuation) or may focus on the demand for goods and services provided by NC (i.e. for assessing NC criticality). The latter necessitates an understanding of the flows of services between assets and beneficiaries, which cannot be mapped for a full range of NC in England. Thus, we do not adopt a full set of indicators from existing studies but they inform our selection

Table 1
England's landscape.

Type	Extent ^a	Observations
Urban	9%	81.2% of the population live in urban areas
Agricultural	70%	Most is privately owned
Woodland	9%	One of the lowest afforestation rates in Europe
Wetland	4%	Almost half are protected
Upland	5%	
Rivers and streams	136,000 km	
Canals	2600 km	
Lakes and reservoirs	5700 (number)	In addition to an extensive coastline

Source: UKNEA (2011) and DCLG (2013). ^aPercentage of landcover (approximate).

Table 2
Natural capital indicators calculated for each LAD. ¹Extents correspond to UK Broad Habitats (Jackson, 2000). ²LCM2007 – Land Cover Map of Great Britain 2007 (www.ceh.ac.uk).

Quantity indicators	Natural capital	Natural capital indicator ¹	Relevant ecosystem service(s)	Data source(s)		
Quantity indicators	Land	Broadleaved and coniferous woodland (% coverage)	Fuel, fibre, climate regulation, hazard regulation, air, soil & water purification, noise regulation, aesthetic and education.	LCM2007 ²		
		Wetland and coastal land (% coverage)	Hazard regulation, water purification, climate regulation, recreation, aesthetic and education.	LCM2007		
		Low density built-up land (% coverage of suburban land cover class)	Recreation, noise regulation, hazard regulation.	LCM2007		
		High density built-up land (% coverage of urban land cover class)		LCM2007		
		Semi-natural grassland (% coverage)	Soil & water purification, recreation, aesthetic, hazard regulation.	LCM2007		
		Agricultural land (% coverage of enclosed farmland broad habitat)	Food, fuel, hazard regulation, aesthetic.	LCM2007		
		Mountain (% coverage)	Climate regulation, food, water purification, hazard regulation, aesthetic.	LCM2007		
		Freshwater (length of river or lake shoreline per km ²)	Freshwater, food, recreation, aesthetic.	OS OpenRivers, OS Meridian 2015 (www.ordnancesurvey.co.uk)		
		Quality indicators	Land	Land with protected status (% coverage)	Wild species diversity, pollination, education.	Natural England boundary datasets, downloaded 2015 – ‘Local Nature Reserves’, ‘National Nature Reserves’, ‘Ramsar Sites’, ‘Sites of Special Scientific Interest’, ‘Special Areas of Conservation’, ‘Special Protection Areas’ (www.gis.naturalengland.org.uk)
				Publicly accessible spaces (% coverage)	Recreation, education, cultural heritage.	‘RSPB UK reserves 2015’ (www.rspb.org.uk)
Agricultural land quality (% agricultural land modelled as good to excellent – grades 1 & 2)	Food			Natural England boundary datasets, downloaded 2015: ‘Country parks’, ‘CROW 4, 16, 15, access’, ‘Doorstep greens’, ‘Local Nature Reserves’, ‘National Nature Reserves’, ‘Millennium Greens’, ‘Registered parks and gardens’. (gis.naturalengland.org.uk)		
Density of carbon in topsoil (area weighted mean of carbon density tha^{-1})	Climate regulation, food.			Accessible woodland dataset (The Woodland Trust, BG v.11, 2014 enquiries @ woodlandtrust.org.uk) Agricultural land classification 2013 (gis.naturalengland.org.uk)		
Ecological communities and species	Land, soil	Ecological status (area weighted mean)	Wild species diversity, pollination.	Soil carbon natural capital map 2007 (catalogue.ceh.ac.uk)		
		Water quality (% length of waterbodies with good or high overall status)	Freshwater, recreation, food, aesthetic.	Ecological status grid 2014 – a modelled biodiversity index adjusted for recorder effort and environmental zone (catalogue.ceh.ac.uk) Water Framework Directive waterbodies’ status 2013 (www.geostore.com/environment-agency)		

(Table 2).

The first steps of the ‘Principals for Natural Capital Accounting’ (ONS., 2017) requires assessment of NC stock extent and condition. For creation of a risk register, Mace et al. (2015) add ‘spatial configuration’, whilst De Groot et al. (2003) consider the condition of the assets including naturalness, biodiversity, uniqueness, fragility, value for supporting life and renewability as ‘critical’ factors. On this basis, we sought indicators of the extent and quality (condition) of NC. We do not explicitly account for spatial configuration, nor natural processes or functions as these are specific to individual ES and the flows of goods and services. However, since we include the spatial extent of built-up areas to provide context, there is some indication of the number of potential beneficiaries of, or actors in the production of, local services.

We define distinct types of NC using the ‘UK broad habitats’ classification; an approach consistent with Mace et al. (2015) and UKNEA (2011). The extent of each broad habitat is measured from the Land Cover Map of Great Britain (LCM), except freshwater extents which are obtained from Ordnance Survey vector data.

Indicators regarding the quality of NC assets are less easily developed given the limited availability of spatially disaggregated datasets consistently measured across England. We sought quality indicators relevant for each broad habitat and for a range of ES comprising provisioning, cultural, regulating and supporting services. Some datasets explicitly measure the quality of specific types of NC (e.g. water quality), although several were omitted for reasons of parsimony. For example, multiple datasets indicate soil quality, however at the district level they correlate closely with soil carbon ($\rho_{\min} = 0.7$, $\rho_{\max} = 0.956$), which we use since it is also relevant for climate regulation. Some quality indicators are applicable across multiple types of NC and are relevant to conditions identified by De Groot et al. (2003). Ecological status indicates the level of biodiversity of each type of NC and protected areas, whilst denoting a management approach, by definition may imply greater naturalness, biodiversity, uniqueness and/or fragility. We include coverage of publically accessible land as this is an important condition for several cultural ES.

2.3. Deprivation data

We adopt the measure of social deprivation used by government in policy analysis and resource allocation, the 2015 Index of Multiple Deprivation (IMD). The IMD aggregates indicators across weighted domains of income, employment, education, skills and training, health and disability, crime, barriers to housing and services, and living environment. Fairburn, Maier, and Braubach (2016) give an account of its development and use in UK EJ studies.

The Office for National Statistics ranks all Lower Super Output Areas (defined to contain a population of c.1500) in the country by deprivation. IMD metrics are subsequently calculated for all 325 larger Local Authority Districts (LAD) in England. We use the LAD as our spatial unit as it is key for land use planning and largely coincides with areas defined for ecological management and economic growth. District deprivation ranks vary dependent on the measure used; only six districts feature in the 20 most deprived districts based on all three common deprivation measures. We therefore use these three measures, reflecting LAD deprivation rank (low ranks represent high deprivation), extent and severity (Table 3).

Table 3
District level representation of IMD data (Smith et al., 2015).

IMD measure	Description
Average rank	Population weighted mean of all LSOA ranks within the LAD. A whole area measure, but neglects within LAD variability.
Extent	% of LAD population in the 30% most deprived LSOAs nationally. Within the 30%, progressively more weight is given to the more deprived LSOAs.
Local Concentration	Average rank of the most deprived LSOAs within which 10% of the population of the district live. Measure focuses on most severe deprivation.

3. Methods

We next sought to explore the association of NC with deprivation at LAD level. Our analysis comprised calculation of selected NC indicators, their aggregation, and subsequent comparison to IMD metrics. Spatial analysis was performed using ESRI ArcGIS 10.3.1 and QGIS 2.14, with statistical analysis and aggregation of NC indicators executed using IBM SPSS 22 and R software.

3.1. Computation of natural capital indicators

Computation of NC indicators at the LAD level required several geospatial processing steps dependent on the resolution and format of the input data. To control for the variable size of LADs, indicators were normalised using percentages where possible, otherwise per unit area and area-weighted means were calculated. For quality indicators, calculations were required to handle different data types. In general, extent of publically accessible areas and land with protected status were given as a percentage of LAD area. These areas were defined by multiple independent datasets merged before the extent was computed. For quality of water and agricultural land, features classified as the highest and second highest quality were extracted and their extent relative to total classified waterbodies/agricultural land within each LAD computed as a percentage. Soil carbon and ecological status are provided as continuous data values and therefore area-weighted means were calculated.

3.2. Computation of aggregate natural capital

Aggregate NC is often conveyed by totalling individual components, however, we avoided this approach since this requires further value judgements (i.e. are different types of NC equally weighted?). Instead, we used clustering techniques to provide a comprehensive, quantitative and spatial summary of NC. Whilst potentially a less value driven aggregation approach, judgement cannot be wholly avoided. Clustering describes areas in terms of key characteristics as shown by the mean value of each indicator, allowing information about component characteristics to be simultaneously conveyed.

Clustering has previously been used to assess ES bundles (Hamann, Biggs, & Reyes, 2015; Turner et al., 2014). Hamann et al. (2016) further demonstrate how clusters can be used to compare ES to social data. Others have used clustering to explore who benefits from urban greenspace (Barbosa et al., 2007; Xiao et al., 2017).

Prior to applying a clustering algorithm, we calculated the Moran’s I global statistic of each indicator and assessed pairwise relationships using Spearman Rank correlation (Fig. 3). We also conducted a PCA analysis (see Supplementary material S1). We applied a two-step clustering approach (Fig. 3), similar to Green, Vickers, and Dorling (2014), which helps address some limitations of different clustering techniques. Criteria for optimal number of clusters includes relatively equal-sized and compact clusters, maximum separation between clusters and a stable solution (Green et al., 2014). Various cluster number optimisation metrics exist, of which 30 were accessed in the R NBCLust Package (Charrad, Ghazzali, Boiteau, & Niknafs, 2014). Beyond these metrics, the number of clusters selected should be based on results that are sensible, interpretable and resolved to an appropriate level of detail (Green et al., 2014).

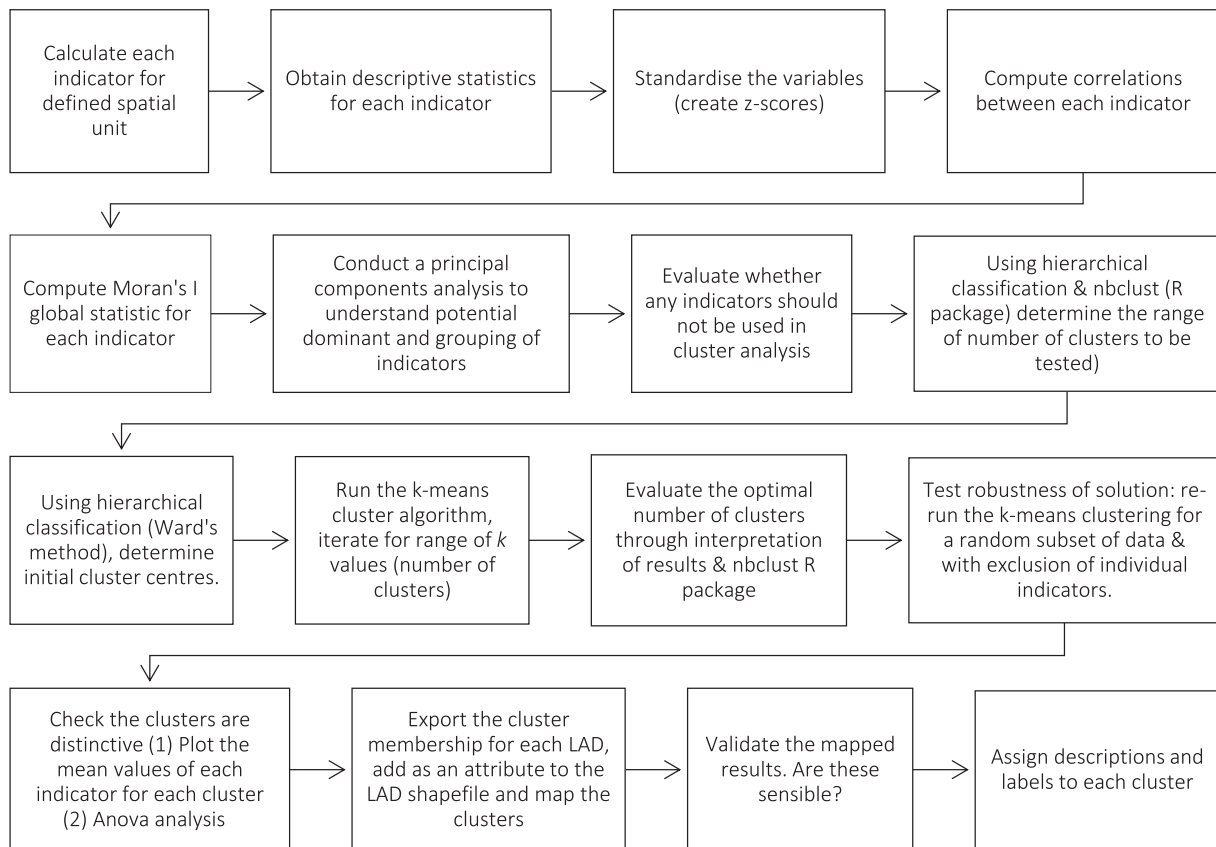


Fig. 3. Steps taken for analysis of natural capital classification of LADs using hierarchical and k-means clustering techniques.

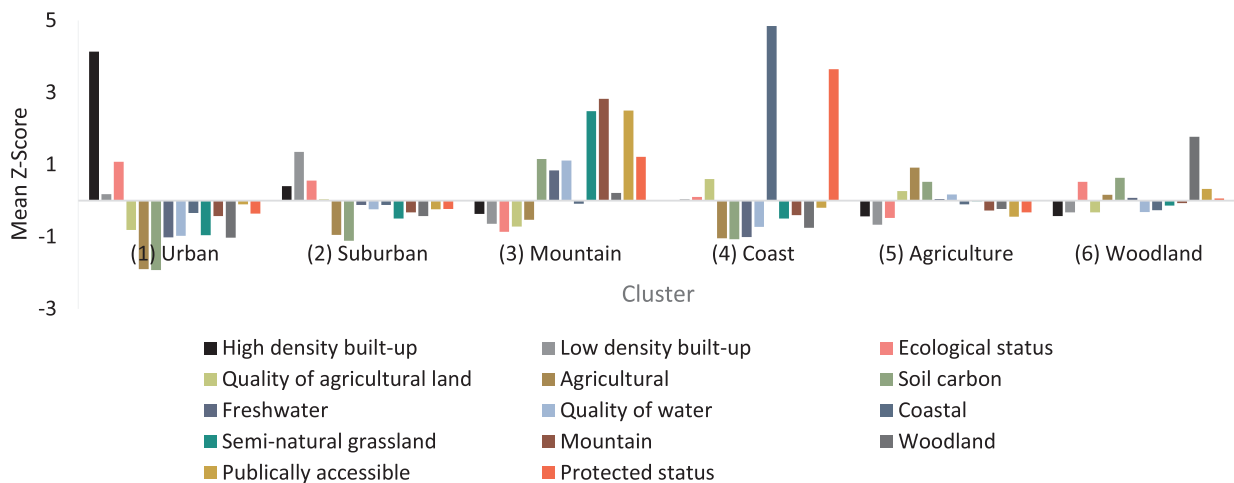


Fig. 4. Mean z-scores for each indicator by cluster. A z-score of zero represents the national average, whilst a score of +1 is one standard deviation above the average. Mean values of natural capital indicator z-scores for each of the six clusters describe the characteristics of each cluster. See Fig. 6 for cluster names.

We found no single optimal clustering of England’s NC. NBCLust metrics commonly indicate that three clusters are optimal, but this primarily identifies an urban/rural divide and our classification must reveal greater detail. A final classification of six clusters was selected; six clusters have a more evenly distributed cluster size, can be easily interpreted, and perform well in the Hubert’s Statistics and D Index plots (see [Supplementary content S2](#)). Critically, the six clusters are meaningful, retain sufficient distinction between clusters for descriptions to be assigned to each (Fig. 4), but are not overly complex. Results were replicated for subsets of data, indicating the classification is robust. Results of sensitivity testing using 5 and 7 clusters, and with removal of individual indicators are given in [Supplementary content \(S3 and S4\)](#).

3.3. Comparison of natural capital to deprivation

Variation of social deprivation by NC was explored by: using the mean values of each IMD measure for the districts assigned to each cluster; comparing which clusters contain the most and least deprived districts; and by visualisation of the distribution of IMD ranks in each cluster using boxplots. Kruskal-Wallis and 1-way Anova tests were applied to determine whether differences in deprivation between clusters are statistically significant.

Table 4
Spearman Rank correlation coefficients.

	Mountain	Coastal	Woodland	Ecological Status	Agricultural land	Quality agricultural land	Freshwater	Water quality	Publically accessible	Protected status	Semi-Natural Grassland	Soil carbon	Low-density built up	High-density built up
Mountain	1													
Coastal	-0.053	1												
Woodland	.305**	-0.095	1											
Ecological Status	-.223**	-0.025	-0.025	1										
Agricultural land	.141*	-.209**	-.022	.162**	1									
Quality agricultural land	-.152**	-.093	-.149**	.421**	.421**	1								
Freshwater	.439**	-.230**	.137*	-.089	0.083	-.110*	1							
Water quality	.461**	-.074	.123*	-.374**	0.015	0.078	.374**	1						
Publically accessible	.332**	0.004	.514**	0.042	-.330**	0.078	0.105	0.078	1					
Protected land	.165**	.532**	.148**	-0.048	-.271**	-0.032	-0.032	0.03	.496**	1				
Semi-Natural Grassland	0.669**	0.076	0.276**	-0.454**	0.349**	-0.129**	0.389**	0.451**	0.273**	0.129**	1			
Soil carbon	.531**	-0.023	.524**	-.445**	.642**	0.004	0.306**	.449**	.240**	0.098	0.707**	1		
Low-density built-up	-.305**	-.129*	-.243**	.555**	-.745**	-.172**	-.124*	-.417**	-0.018	-0.093	-0.536**	-.802**	1	
High-density built-up	-.333**	-0.036	-.411**	.469**	-.781**	-.189**	-.155**	-.425**	-0.046	-0.045	-0.536**	-.855**	.860**	1

** p < 0.05 *** p < 0.01.

4. Results: the social distribution of natural capital in England

4.1. Aggregate natural capital

4.1.1. Moran's I and pairwise correlations

All NC indicators exhibited significant spatial clustering (Appendix A), and each indicator exhibits significant moderate correlation ($\rho < -0.4$, $\rho > 0.4$) with at least one other indicator (wetland/coastal and agricultural land quality) and with as many as eight (soil carbon) (Table 4). Independence of variables is not a prerequisite for clustering but highly correlated variables may add 'weight' to a particular division. Whilst soil carbon is closely negatively correlated with built-up areas ($\rho = -0.802$ and $\rho = -0.855$ for low and high density respectively) we retain this indicator since it signifies quality and emphasises differences beyond the rural/urban divide. Low and high density built-up areas, ecological status and agricultural land quality are predominantly negatively correlated with other indicators, suggesting that districts dominated by these characteristics may form distinct clusters.

4.1.2. Clustering

For the six NC clusters distinguishing NC characteristics are indicated by high/low z-scores (Fig. 4). Spatial distribution of clusters is mapped in Fig. 5. Clusters 1 ('urban') and 2 ('suburban') are characterised by higher proportions of built-up land and lower NC. The 'urban' cluster with the greatest extent of high density built-up land overall exhibits the lowest extent and quality of NC, but has the highest ecological status. Few LADs are assigned to the 'urban' cluster and these are spatially concentrated in central London. The 'suburban' cluster has the highest extent of low density built-up land, the type and extents of most NC indicators are below average, except for ecological status. Clusters 3 ('mountain') and 4 ('coast') have the highest extent and quality of NC. The rural 'mountain' cluster exhibits the highest extents of mountain, freshwater, semi-natural grassland, land that is publically accessible and with protected status. It has the highest soil carbon and water quality but the lowest ecological status. Although only 28 districts are in this cluster, their rural character means they cover a large area, mostly in northern England. The 'coast' cluster has the highest area of coastal habitat, land with protected status and quality of agricultural land, but low extent of agricultural land. This is the smallest cluster and is scattered spatially in the northwest and southeast. Cluster 5 is predominantly rural and characterised by the highest extent of agriculture and above average quality of agricultural land, water and soil carbon. It is also the most commonly assigned cluster. Cluster 6, also rural, is characterised by the highest extent of woodland and an above average extent of freshwater, mountains, ecological status and soil carbon.

Systematic removal of individual indicators to test sensitivity reveals overall a robust clustering (see S4), although some sensitivity to the exclusion of high and low density built up land is noted (19% and 15% LADs change cluster membership, respectively).

4.2. Deprivation and aggregate natural capital

Primarily, districts of lowest deprivation are located in southeast England, and the most deprived districts are in central London, Birmingham and the Northwest (Fig. 2). The spatial pattern of deprivation therefore in some cases shows some consistency with the spatial pattern of NC.

IMD values for districts in each cluster are shown in Fig. 6. Significant differences in median values for each IMD measure and the distribution of IMD values across clusters are observed. Differences are consistent under sensitivity testing.

'Urban' districts are on average the most deprived and rural 'agriculture' and 'woodland' districts the least deprived. 78% of the 10% most deprived districts are assigned to 'urban' and 'suburban' clusters

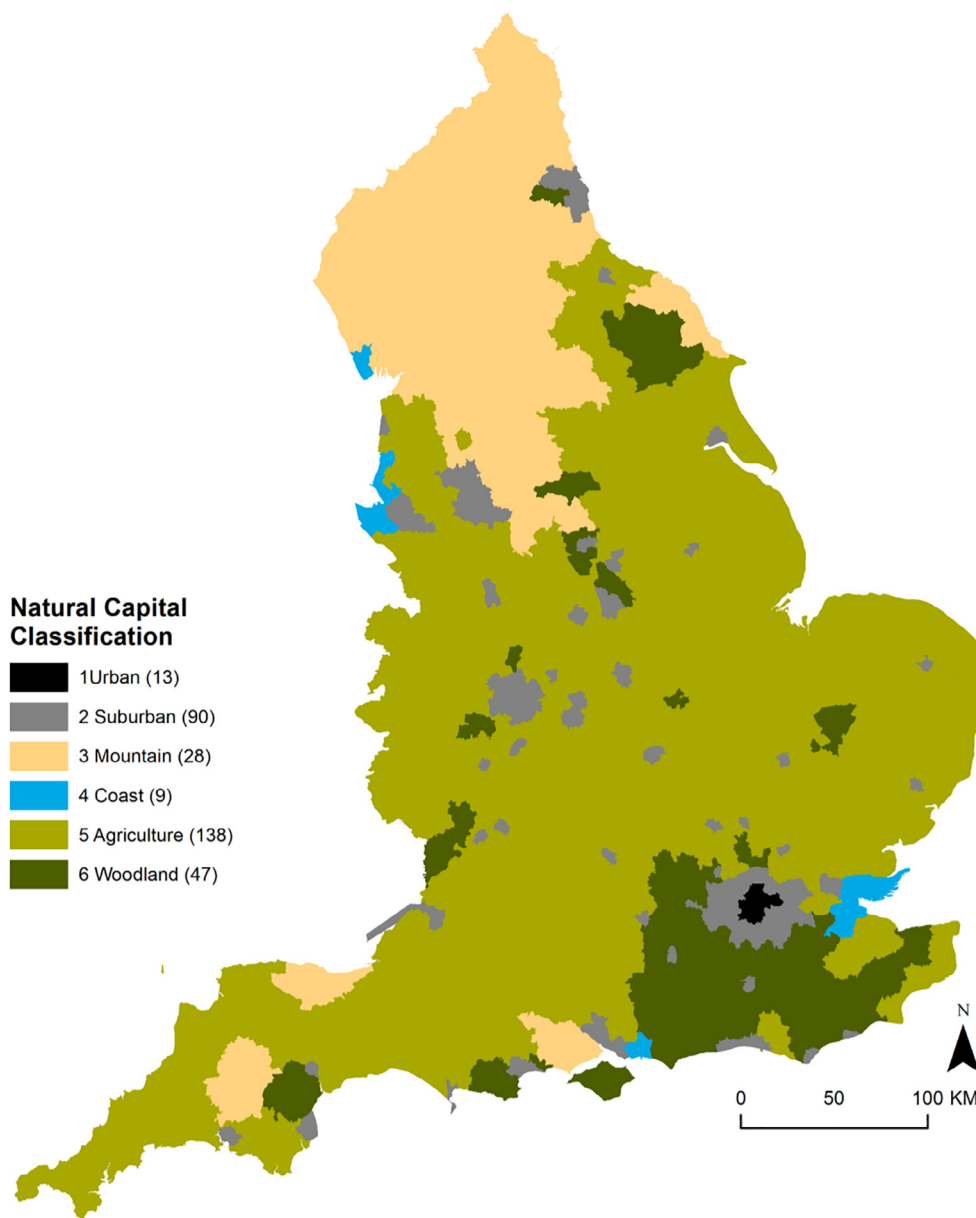


Fig. 5. Natural capital clusters in England (see text). Parenthesis denote number of LADs within each cluster.

compared to 88% of least deprived districts assigned to the ‘agriculture’ and ‘woodland’. None of the 30% least deprived districts are assigned to cluster 1 and only 5 of the 30% most deprived districts are assigned to the woodland cluster (Fig. 6).

The means of IMD average rank, extent and local concentration are consistently highest for the ‘agriculture’ and ‘woodland’ clusters. The ‘urban’ cluster has the lowest mean and median IMD average rank and extent, the mean and median of IMD local concentration is lowest for the ‘coast’ cluster. This suggests that the severity of deprivation for the most deprived districts is highest in coastal areas which are moderately built-up, although these are the smallest clusters with a much lower range of IMD values.

Of the more rural districts, the ‘mountain’ cluster has the highest deprivation, it also has higher severity of deprivation than the ‘urban’ and ‘suburban’ clusters. The least deprived areas have higher extents of woodland (cluster 6), ecological status (cluster 6) and agricultural land (cluster 5 & 6) and agricultural land quality (cluster 5) but overall, NC here is not as diverse and of high quality and extent as the ‘mountain’ cluster.

Overall, we find that the most deprived districts tend to have lower NC, but there are nuances and exceptions depending on the IMD measure used, with respect to some NC indicators. Notably, we find that some areas with very high extent, quality and diversity of NC have higher deprivation. The observed differences are largely noted with respect to measures of central tendency and there are large ranges in IMD ranks for most clusters. Thus the level of deprivation of districts within a particular cluster should not be assumed.

5. Discussion

5.1. Evaluation of methodology

Benefits gained from NC are driven by scale dependent, direct and indirect physical, social and economic processes (Rova & Pranovi, 2017). We sought to include a full range of NC but acknowledge that at this scale, it is not practical to concurrently map the flows of benefits for all NC and associated ES. Indeed, existing national classifications of ES bundles (Dittrich et al., 2017; Turner et al., 2014) tend not to examine

flows to beneficiaries for most services. Whilst it is reasonable to speculate that the social distribution of NC reflects the social distribution of multiple ES (with direct and local flows) generated by that capital, this remains to be tested.

We implemented a clustering approach, however, this is subject to indicator selection and thus data availability, hence future work which includes other indicators of NC quality is desirable (De Groot, Alkemade, Braat, Hein, & Willemsen, 2010). Aggregation to administrative boundaries neglects ‘natural’ boundaries (Dittrich et al., 2017), but if greater attention is to be given to issues of inequality in local planning (Pinoncelly, 2016) some compromise is unavoidable. Nonetheless, exploring the relationship between NC and deprivation for other administrative or landscape scales may be revealing.

5.2. Implications for environmental equity

There is substantial environmental inequality in the UK but to date the evidence base neglects a full range of environmental benefits (see Section 1.1). We hypothesised that more deprived communities would occur in areas with less, and/or lower quality, NC, which our district level, national analysis supports for some forms of NC, but not all.

We find deprivation is lowest in districts characterised by high woodland cover with greater accessibility, whilst deprivation is highest in urban areas with less NC, but of high ecological status. In contrast, we also find high deprivation for coastal areas with large swathes of protected land and upland rural areas with multiple types of higher quality NC; this finding is consistent with literature highlighting issues of rural poverty (e.g. Shucksmith, 2012). We also find some rural areas with lower deprivation but also lower NC according to several indicators. For example, some urban districts have higher ecological status compared to agricultural dominated, rural districts, which also tend to be the least accessible. The higher ecological status in urban and suburban areas is likely due to the high potential biodiversity of urban gardens (Goddard, Dougill, & Benton, 2010). It is therefore possible that benefits provided by urban gardens in more deprived areas could exceed those provided by agricultural land in less deprived areas, dependent on how the land is managed (Power, 2010). Such opposing patterns demonstrate the importance of exploring environmental inequality for different contexts, including within and between urban and rural areas, and for multiple types of NC.

Some of the observed associations between deprivation and NC are spatially driven and correspond with known north-south economic inequalities (Whitehead, 2014). However, intertwined social, political, economic and environmental processes shape variation in deprivation and NC across the country. To disentangle such processes is beyond the scope of this paper. Rather, we have sought to draw attention to social patterns in NC distribution relevant to the social objectives of current NC strategies. Nonetheless, there also remains a need for greater evidence to develop current understanding of the mechanisms relating ES and poverty (Suich, Howe, & Mace, 2015).

5.3. Implications for planning and land management

The social inequality in NC distribution has implications for sustainable management of NC in terms of the production of and demand for ES (Bennett et al., 2015; Ernstson, 2013), and trade-offs between welfare and conservation objectives (Daw et al., 2011). Humans are both users and actors in the production and consumption of goods and services derived from NC (Rova & Pranovi, 2017). For example, Soto et al. (2016) found that higher education, income and age had a beneficial effect upon carbon stock of forests. Lin et al. (2017) found that connection to nature and vegetation cover in gardens is greater for households in less deprived communities and Morse et al. (2011) argue that deprivation impacts upon environmental degradation. Indeed, a national survey in England found that those in the lowest social grades were least likely to engage in pro-environmental behaviour (Natural

England, 2015). Conversely, Ernstson (2013) highlights risk to NC in wealthier areas from development pressure. Given these multiple feedbacks, social inequalities should be a critical consideration in NC management. A NC approach is being increasingly adopted for management of the natural environment, and the impacts of different social distributions of NC need to be encapsulated when NC values and thresholds are calculated.

Synergistic ecological, social and economic outcomes are desirable but not always achievable and whether social justice and environmental sustainability are compatible objectives has been a key debate in the EJ literature (Dobson, 2003). Similarly, not all ES can be maximised concurrently (Seppelt et al., 2012) and an increase in one service can reduce another which may impact poverty alleviation objectives (Daw et al., 2011). A NC/ES approach can help identify synergies and conflicts (De Groot et al., 2010) but this is better achieved with understanding the relationship of NC (and ES) to social characteristics of communities. For England, we identify deprived districts with both high and low NC, and therefore anticipate both synergies and conflicts in social and ecological outcomes exist. For example, the 25-year environment plan seeks to increase tree cover in urban areas; this may simultaneously increase local benefits available to deprived

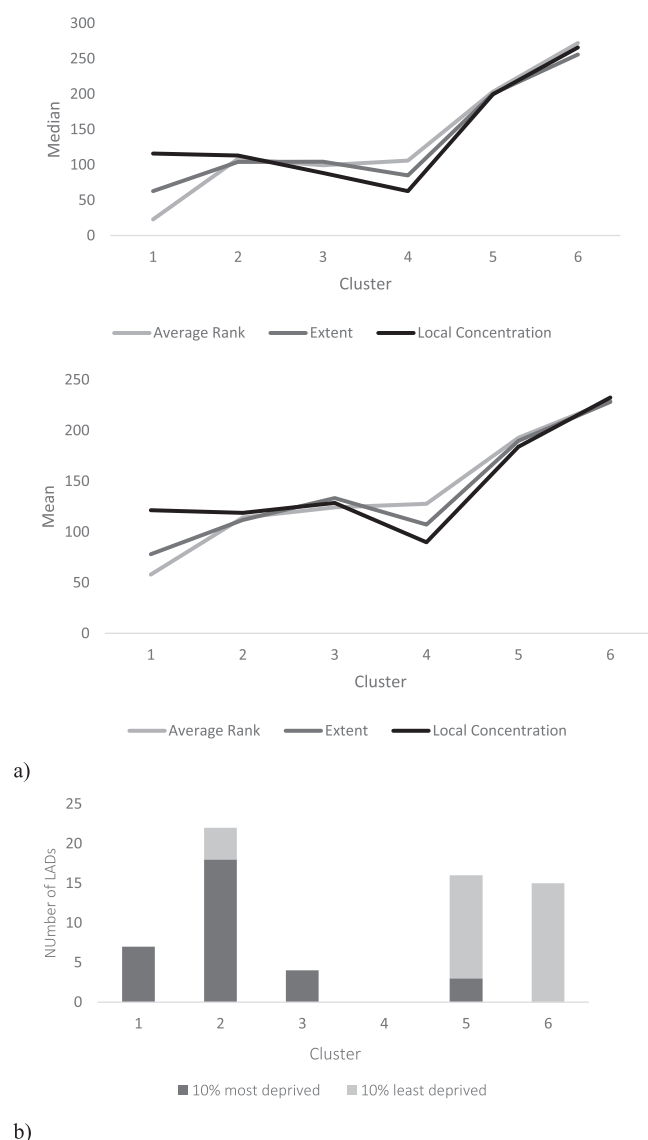
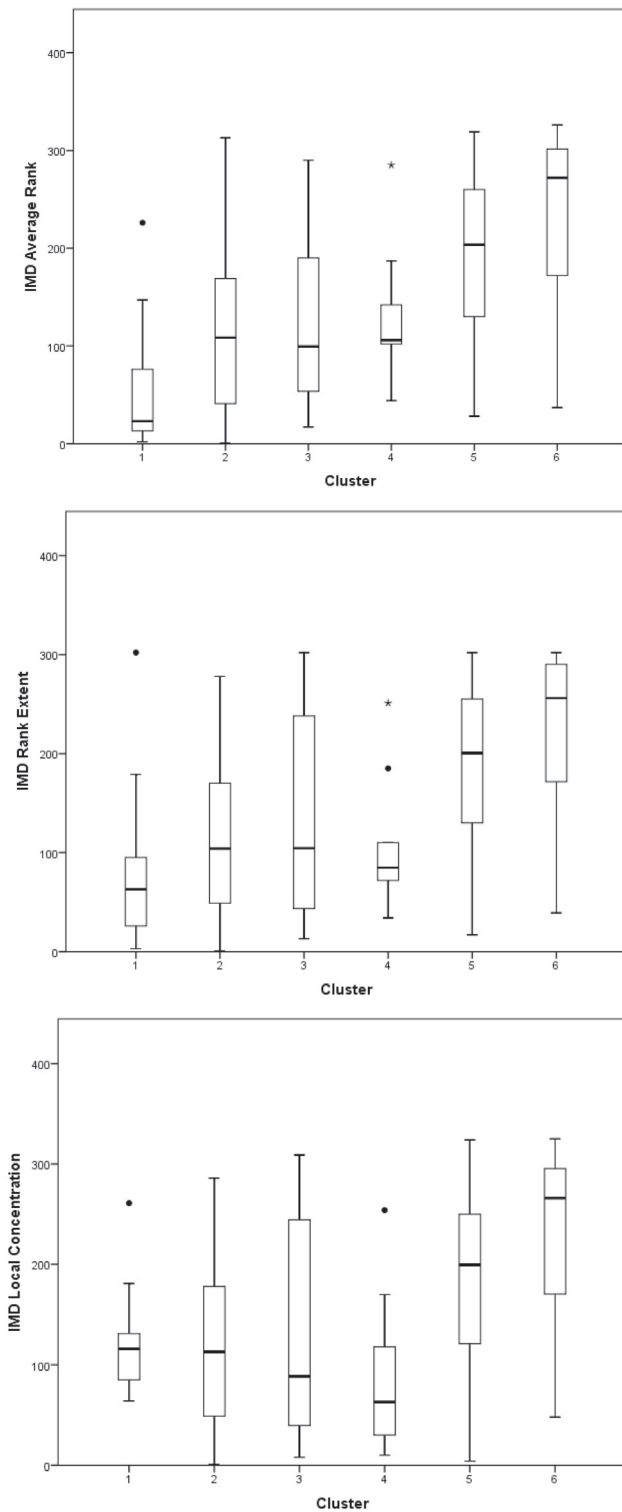


Fig. 6. Variation in IMD across clusters. a) Mean IMD for each cluster b) Number of the most and least deprived LADs assigned to each cluster c) Boxplots of the values of each IMD measure for the 10 clusters.



c)

Fig. 6. (continued)

populations, although it is important to avoid the attendant risk of benefits not reaching the target group due to environmental improvement induced gentrification (Wolch et al., 2014). Conversely, woodland management for carbon sequestration has been shown to be most effective in areas of lowest deprivation (Soto et al., 2016). If this holds

true for England, then woodland policy for climate change mitigation should target wealthier areas which would limit the flow of other more localised benefits to more deprived neighbourhoods. We also find many of the districts with the most accessible high quality green space are also the most deprived. Increasing publically accessible spaces where it is lowest (agricultural districts) may therefore not be relevant from a distributional justice perspective and may conflict with food provision objectives.

Whilst there are potential synergistic outcomes, a location specific approach which accounts for existing NC, its services and social conditions is required and the outcomes may not be equally weighted in all respects of social, ecological or economic gains. Thus we advocate further research in other regions and countries of the spatial patterns in social, economic conditions, NC and ES.

6. Conclusion

Environmental inequalities have been widely reported in many high income countries, including England, with respect to a broad range of environmental harms. However, knowledge of inequalities in the social distribution of the goods and services provided by NC is lacking. Studies of how ES are socially distributed have largely taken a livelihoods perspective for low income countries where there is often more direct dependence on NC, or upon a single type of NC such as urban green-space.

Our analysis develops preliminary insights into the social distribution of ES in a high-income, urbanised country through an environmental inequality analysis of NC in England. We find the most deprived communities experience very low extent and quality of NC ('urban poor'), but also the highest quality and extent of NC (the 'rural upland and coastal poor'). We also find that deprivation is lowest for highly wooded publically accessible areas, and districts which are largely agricultural but inaccessible.

Overall, considering a wider range of NC reveals some interesting patterns nationally, and whilst some inequality in the social distribution of environmental 'goods' is evident, this is not consistent for all place or all types of NC. Thus a 'one-size fits all' national policy to address inequalities in ecosystem goods and services is not appropriate. Rather national policy should encourage incorporation of equity concerns within planning and ecosystem management at sub-national scales, with the intention of identifying synergies between social, economic and ecological outcomes, and minimising conflicts between justice and environmental sustainability objectives.

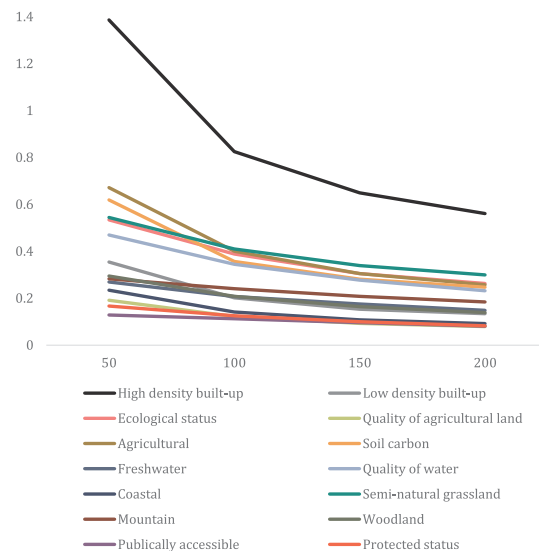
Since a complex set of social, economic and ecological interactions drive the production of ES and environmental inequalities, analysis of the social distribution of ES at multiple sub-national scales is needed. Methods for assessing the social distribution of multiple ES need developing, and then analysis can ascertain whether inequalities occur with respect to particular services and/or locations and the appropriate means and scale at which to address inequalities. Our subsequent work will begin to address some of these challenges through sub-national analysis of the social distribution of ES, which we hope will develop understanding required for decision making to promote sustainable and equitable implementation of the ecosystem approach.

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Appendix A

Moran's I global statistics

Moran's I global statistic calculated with 50, 100, 150 and 200km threshold ($p < 0.001$)

Appendix B. Supplementary data

Supplementary data associated with this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.landurbplan.2018.03.022>.

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