Jiang, Mingkai; Bullock, James M.; Hooftman, Danny A.P.. 2013. **Mapping ecosystem service and biodiversity changes over 70 years in a rural English county [in special issue: The UK national ecosystem assessment]** *Journal of Applied Ecology*, 50 (4). 841-850. https://doi.org/10.1111/1365-2664.12093

Mapping ecosystem service and biodiversity changes over 70 years in a rural English county

Mingkai Jiang, James M. Bullock & Danny A. P. Hooftman[1]

NERC Centre for Ecology and Hydrology, Benson Lane, Wallingford, UK, OX10 8BB

[1]Corresponding author:

dann1@ceh.ac.uk

+44 (0)1491 692700

Running title: **Mapping ecosystem service changes**

Word count: 6,972

Summary: 348

Main text: 4,465

Acknowledgements: 38

References: 1,533

Tables: 319 (incl. legends)

Figure legends: 165

Number of tables: 2

Number of figures: 3

Number of references: 55

Summary (348 words)

1. Biodiversity and ecosystem services continue to be compromised by land-use change, which is often focussed on enhancing agricultural production. Assessment of losses would be aided by analyses of temporal changes in the extent and spatial pattern of services and biodiversity. To date, no studies have mapped long-term changes in ecosystem services using historical maps.
2. We mapped changes between the 1930s – before the considerable intensification of land-use in the UK starting in the 1940s – and 2000 in climate change amelioration services (carbon storage), provisioning services (agriculture and forestry) and plant species richness (biodiversity) for Dorset, a rural English county.
3. We combined land-use maps (1-ha resolution) with multiple proxies of service delivery for the 10 different Broad Habitats in the region. Biodiversity was mapped using plant survey data from the two time periods. We used bootstrapping to include uncertainty due to the different proxies and Gini-coefficients to quantify statistical changes in spatial pattern.
4. Overall, we found significant increases in agricultural provisioning and large losses in biodiversity over the period, which reflect widespread conversion and intensification of land-use. We found no change in Dorset's carbon store, because carbon lost through land-use intensification was balanced by increases in woodland over the 20th century.
5. The carbon storage and the delivery of provisioning services both became more unequally distributed, indicating a change from relatively homogeneous delivery of services to concentration into hotspots. The maps from the year 2000 showed spatial dissociation of hotspots for carbon, provisioning and biodiversity, which suggests that, compared to the 1930s, modern, intensive land use creates conflicts in delivery of multiple services and biodiversity.
6. *Synthesis and applications.* Detailed maps of historical changes in location-specific service delivery and biodiversity provide valuable information for land-use planning, highlight trade-offs and help to identify drivers. Furthermore, historical maps provide an important baseline to indicate the suitability and potential success of suggested actions, such as habitat

restoration, and their relevance to traditional land-use. Various frameworks could be informed by our approach, including the ecosystem service aims of the EU biodiversity strategy and the newly created UK Nature Improvement Areas.

Keywords:

Agricultural production; Biodiversity; Carbon sequestration; Ecosystem assessment; Historical maps; Landscape management; Land-use changes; Semi-natural habitats.

Introduction

As a consequence of worldwide land-use change, the capacity of ecosystems to provide the ecosystem services that are vital to human well-being have been undermined (MEA 2005; Tallis & Polasky 2009; UKNEA 2011). Many of these consequences have been the unintended result of management actions designed to maximise particular services, such as agricultural production (Rey-Benayas & Bullock 2012). Human societies have often overlooked the fact that ecosystems may support numerous services that are interrelated in complex and dynamic ways (Chan *et al.* 2006; Bennett, Peterson & Gordon 2009).

Policy responses to counter declines in the delivery of multiple ecosystem services – such as Defra's Ecosystems Approach in the UK (Defra 2010) and China's Grain for Green initiative (e.g. Feng *et al.* 2004) – require an understanding of the impacts of land use decisions on different services and biodiversity. At the landscape level, such an understanding necessitates the incorporation of credible estimates of ecosystem service changes into land-use maps to allow spatially-explicit planning for the delivery of bundles of ecosystem services (Chan *et al.* 2006; Nelson *et al.* 2009). Previous work has delivered methods for assessing and mapping the economic, social and ecological values of services (e.g. Kremen & Ostfeld 2005; Eigenbrod *et al.* 2011; Swetnam *et al.* 2011); identified spatial and temporal trade-offs and synergies (e.g. Anderson *et al.* 2009; Nelson *et al.* 2009; Raudsepp-Hearne, Peterson & Bennett 2010); and assessed the effects of land management decisions on the delivery of services and biodiversity (Egoh *et al.* 2008; Rey-Benayas *et al.* 2009; Birch *et al.* 2010; Newton *et al.* 2012a).

Both the Millennium Ecosystem Assessment (MEA 2005) and the United Kingdom National Ecosystem Assessment (UKNEA 2011) highlighted the importance of understanding trends in ecosystem services over time. However, to the best of our knowledge, no study has mapped long-term changes in the quantity and patterns of service delivery spanning a period of considerable agricultural intensification. Mapping ecosystem services at a reference point in the past will provide detailed understating of how service delivery has changed over time and indications as to where in a landscape actions might be targeted to enhance particular services (Dearing *et al.* 2011). Mapping projects have located geographic hotspots with high levels of one or more service and/or biodiversity (e.g. Egoh *et al.* 2008; Naidoo *et al.* 2008; Anderson *et al.* 2009; Bai *et al*. 2011). Such studies have generally suggested these hotspots should be targeted in plans to enhance biodiversity and/or ecosystem service delivery, and that the spatial coincidence of hotspots for different services and biodiversity may allow synergies in planning for multifunctional landscapes. Little is known, however, about the changes in the prevalence and pattern of these hotspots over time. Indeed, if hotspots develop as a result of anthropogenic landscape changes, such as fragmentation, they might rather be seen as a negative indicator for delivery of services and biodiversity conservation (e.g. Boakes *et al.* 2010). However, it may also be important to identify such hotspots to help prevent a further deterioration of services and biodiversity,

The UKNEA (2011) considered changes over the last 60 years, identifying the 1940s as a time in which the UK entered a phase of national reconstruction to enhance production, agricultural intensification, and to build homes and infrastructure, which resulted in massive land use changes. A snapshot of land use, services and biodiversity just before major changes occurred – as we consider here for the 1930s – provides an ideal reference for planning landscape management, and suggests to what extent and where ecosystem services might be restored. In this paper we use newly-digitized British land-use maps from the 1930s, which have allowed us to map, at a fine resolution, the extent, scale and spatial details of land-use patterns for Dorset, a typical rural English county. Comparison with the UK land cover map for 2000 showed huge losses and dramatic fragmentation in the area of semi-natural habitats as a consequence of agricultural intensification (Hooftman & Bullock 2012). Here, we use these maps combined with ecosystem service proxy data and plant surveys to map changes in ecosystem services and biodiversity between these two dates. We focus on: climate change amelioration services provided by carbon storage and net carbon change; provisioning services provided by agriculture and forestry; and plant species richness as a measure of biodiversity. By doing so we contrast changes in two services with those in biodiversity; following the argument that changes in biodiversity and in ecosystem services are not necessarily related (Bullock *et al.* 2011a).

We produce maps by bringing together multiple data sources for the present and past to provide estimates of ecosystem service delivery from different land-use classes in terms of UK Broad Habitat Types. We use an extended benefit transfer approach – explicitly incorporating variation in measures of services – to link habitat type (Jackson 2000) to ecosystem services. In line with general trends across the UK (UKNEA, 2011) and globally (Butchart *et al.* 2010; West *et al.* 2010), we expect this study to show that conversion of land for intensive agriculture along with advances in farming have increased agricultural outputs, but decreased biodiversity and carbon stocks in Dorset since the 1930s. However, the altered spatial patterns accompanying these changes are not understood, and we hypothesise that the fragmentation of habitats over the time period has led to increases in the prevalence of geographic hotspots for both services and biodiversity. In this way, we provide estimates of ecosystem service and biodiversity values across the landscape of Dorset, and highlight the importance of incorporating temporal changes in service delivery in land-use planning.

Materials and methods

Using an extended benefit transfer approach similar to, for example, the UKNEA (2011) and Newton *et al*. (2012a), we mapped: (i) the regulating ecosystem service of climate change amelioration in terms of carbon stock and net change, and (ii) the provisioning service of combined agricultural and timber production. We also used botanical surveys to map (iii) biodiversity in terms of plant species richness.

STUDY AREA AND LANDCOVER MAPS

We mapped these services and biodiversity between two periods (1930s and 2000) in the county of Dorset, southern England. Dorset (*ca* 2500 km^2) is a rural, maritime county which had approximately 400,000 residents in 2001 and roughly half this number in the 1930s. Dorset can be considered a typical rural landscape in north-western Europe, which has experienced some urbanisation, but in which most land-use change through the 20th century has been the modification of semi-natural habitats to highly intensive agriculture (Hooftman & Bullock 2012). An additional factor in our selection of Dorset is that detailed vegetation surveys were carried out here in the 1930s (see Keith *et al.* 2009; Newton *et al.* 2012b).

To map changes in combined agricultural and timber production and in carbon storage, we built upon detailed land cover maps (Hooftman & Bullock 2012). We used a map of land-use in the 1930s, prior to large-scale agricultural intensification (Dallimer *et al.* 2009) and the Centre for Ecology and Hydrology (CEH) Land Cover Map of 2000, which reflects current, highly intensive land-use. Hooftman & Bullock (2012) focussed on calculating changes in the area of semi-natural habitats between these periods. Results include a large increase of improved grassland and arable land at the expense of semi-natural grasslands and a 25% increase in woodland area. The mapping methods are summarised in Appendix S1 in the Supporting Information and the area of land-use for both periods can be found in Table 1. For our study, the original maps, which have a resolution of 25 x 25 m, were transformed into 1-ha grid cells using ARCGIS v9.3, based on the dominant land-use. The total number of grid cells is 250,146 in both maps.

AGRICULTURAL AND TIMBER PRODUCTION

We conducted an extensive search for proxies of annual agricultural and timber production in both periods; to obtain, as far as possible, estimates for all land-uses and crop types. We describe the procedure in Appendix S2; searches were performed in Google, the archives of Defra, Eurostat, FAOSTAT, the UK Forestry Commission as well as historical UK Government data archives. Where no data were found for the specific period, data for a nearby year were used (e.g. Agricultural Statistics 1929). Yield data were converted to economic values ("annual monetary production") using commodity prices for the year 2000. Hence changes in production values reflect land-use change and/or intensification but not changes in commodity prices. We will consider changes in relative values of commodities in the Discussion. The proxies *per* land-use type are given in the Tables S1, S2, S3 and S5.

The 2000 map uses specific data about agricultural land use, often including the exact crop planted; but the 1930s map does not, and there are only data on broad land-uses, such as arable. To address this, we bootstrapped among the values of the crops listed in the 1929 for Dorset (Agricultural Statistics 1929) – *i.e.,* barley, oats, wheat, field beans, potatoes and peas – with weighting according to the Dorset-wide cover of each crop (17%, 55%, 21%, 2%, 5% and 0.1% respectively; Tavener 1937). The bootstrapping was done over 50,000 runs and in each the value *per* land-use category was drawn randomly from the possible values. Furthermore, in both periods Dorset grasslands were used to support either livestock for meat (mostly beef, but also lamb and pork) or dairy cattle with the percentages of dairy to beef cows being 50% for each in 1929 (Tavener 1955) and 87% dairy to 13% beef in 2000 (Defra 2011). Per grid cell, we bootstrapped as above for grasslands among the production values for milk and meat production weighted by these ratios. For livestock meat production, different livestock generate different economic values. We addressed this by converting reported densities of each livestock type into Livestock Units (LSU; Table S4). The economic value of one LSU is the return obtained by selling the meat. For milk production from dairy cows, we multiplied the reported densities per hectare of cows with the milk production cow^{-1} year^{-1} (Tables S3 and S5). Wool production had very low value in both periods and was excluded for simplicity, i.e., wool production only accounted for 1% of the total agricultural economic output in 1925 over the UK as a whole (Cons. Archive 1940; Defra 2011).

CARBON STOCK AND NET CHANGE

A similar search was conducted for estimates of carbon stock (t ha^{-1}) for the different land-uses in Dorset, across four categories (adapted from Conte *et al.* 2011): above-ground biomass, below ground biomass, dead carbon (*i.e.* litter combined with other dead organic matter), and soil carbon. The sum of these categories estimates the carbon stock *per* 1-ha grid cell. Net carbon change (e.g. Ostle *et al.* 2009) was estimated as the difference *per* 1-ha grid cell in total carbon stock between both periods. The estimates are given in Tables S7, S8 and S9; the search procedure is described in Appendix S2. Although we used a wide variety of sources, we do not claim to include all published carbon stock figures. Our estimates represent different geographical ranges and most are not specific to the study area, although they encompass Dorset and are specific for the land-use type considered. Furthermore, we assumed the same carbon stock values *per* land-use for both 1930s and 2000. Therefore, we show differences driven by land-use change but not –unknown– temporal changes within land-uses.

BIODIVERSITY

Two independent vascular plant species distribution datasets were used to map biodiversity change between the two periods. For the 1930s we used the “Good data” (Good 1948; Keith *et al.* 2009; Newton *et al.* 2012b), which provides plant species data for approximately 7000 survey sites in Dorset in that period (Dorset Environmental Records Centre – DERC: http://www.derc.org.uk/projects/good.htm). This dataset covers approx. 7 % of the Dorset area and describes local presence/absence of all vascular plant species. For 2000, we used Bowen (2000), who recorded the presence of all plant species in 694 2 x 2 km cells covering the whole of Dorset in the period 1985–2000. These 2000 data were supplemented with a data-set of occurrences of 162 rare and declining species on a 1-ha scale, re-gridded to the 2 x 2 km cells (data courtesy of DERC).

To allow comparison of the partial 1930s data with the complete spatial coverage for 2000, we used species–area curves to scale-up the 1930s data (see Harte *et al.* 2009). From the Bowen data-set, species counts at different resolutions were calculated (from 2 km square grid to 5 km, 10 km, 20 km, ¼ of Dorset and all Dorset) to fit a species–area relationship in 2000 following:

$$S = cA^z \qquad \text{eqn. 1}$$

with S: number of species; A the area in km^2; z the slope of the relationship in log-log space and c the scaling factor.

The species count and area of each site survey in the 1930s was combined with the *z* value calculated from the 2000 data to obtain a *c* value *per* survey. Using these values, the number of species in each site was extrapolated to its surrounding 2 x 2 km square. Good surveys were present in all but two (which were excluded from analysis) of the 2 x 2 km squares; in cases of multiple surveys in a square, the surveys were joined and the area summed. This method assumes a change over time in α-diversity but not β-diversity; we explore this assumption in Appendix S3.

STATISTICAL ANALYSES

For both production (agricultural and timber) and carbon storage, our search provided multiple proxy values per habitat type, which differ in geographical scale and location. To avoid making assumptions regarding the most relevant values (Eigenbrod *et al.* 2010), we used bootstrap assignments in mapping these services. We performed 50,000 runs and in each the value *per* land-use category was drawn randomly from the possible values. The values of all grid cells *per* run were summed to get the overall production and storage values. The derived value *per* grid cell was averaged among runs for mapping. To summarise over the whole area we calculated the average sum over all grid cells and the confidence intervals (95%, 99% and 99.9%). For diversity, confidence levels for both periods were provided by 100,000 random draws of the same number of data-points (694) with re-sampling.

We calculated inequalities in the distribution of values across grid cells for all three measures using the Gini-coefficient (*G*) following Gamboa *et al.* (2010),

$$G = \left(\frac{n}{n-1}\right) \times \left(\left(\frac{2}{\hat{\mu}n^2}\right) \times \left(\sum_{i=1}^{n} y_{(i)} \times \left(i - \frac{1}{2}\right) - 1\right)\right) \qquad \text{eqn. 2}$$

in which, n = number of data-points, y the original series sorted in increasing order (i); $\hat{\mu}$ is the estimated mean of y.

This coefficient reflects the shape of the histogram of all possible values and ranges from 0–1. A low Gini value indicates a skewed distribution, as would occur in a situation with many hotspots and areas of low service delivery or biodiversity. A high value indicates an equally heterogeneous distribution and fewer hotspots and/or low value areas. Note that this coefficient describes the distribution of the values, complementing interpretation of the mapped spatial patterns. Confidence intervals were calculated using the standard error and a z-distribution (eqn. 3). The standard error was calculated as

$$SE(G) = \sqrt{\left(\frac{1}{(n\hat{\mu})^2}\right)} \times \sum_{i=1}^{n}\left(Z_i - \hat{Z}\right) \qquad \text{eqn. 3}$$

with:

$$Z_i = -(G+1)y_{(i)} + 2 \times \left(\left(\frac{(2i-1)y_{(i)}}{2n}\right) - \left(n^{-1} \times \left(\sum_{j=1}^{i} y_j\right)\right)\right)$$

in which y_j is the cumulative value of the series in increased order and $\hat{Z}$ is the average value of Z_i.

We performed a validation test of this procedure, showing that patterns found are caused by (changes in) land use and are not statistical artefacts (Appendix S4). All statistical calculations were done in Matlab v.7.8.0.347; the code is available as Codes S1. The maps were created in ArcGIS v 9.3.

Results

AGRICULTURAL AND TIMBER PRODUCTION

The gross monetary production (in 2000 prices) of combined timber and agriculture in Dorset increased greatly, as expected, between 1930s and 2000 ($P < 0.001$; Table 1). Estimated annual agricultural production (including timber) was £219M for 2000 compared to £33M in the 1930s. Improved grassland contributed most to this increase (+ £141M; Table 1), together with a large increase in income from arable land (+ £65M). This reflects the higher income *per* ha caused by agricultural intensification with estimated annual production increasing from an average £141 ha^{-1} to £950 ha^{-1} (Table 1). Moreover the area of agricultural production was boosted by the large increases in area of arable land and improved grassland. The increase in woodland cover also caused a 30% greater monetary production from this land-use category compared to 1930s (Table 1).

The spatial detail of these results is mapped in Fig. 1. Regions with very low or no agricultural production, such as the heathlands in the south-east and the urban areas, are similar in both periods. However, the remaining area has seen a considerable rise in annual income *per* ha with clear hotspots in the north-western part of Dorset. This shift in spatial pattern is illustrated by a changed Gini coefficient, which indicated a much more unequal distribution of production in 2000 ($G = 0.46$; 5% CI ± 0.001; $P < 0.001$) than in the 1930s ($G = 0.85$; 5% CI ± 0.001).

Validation of these results in terms of net profit *per* farm in 2000 is presented in Appendix S5. These results show that an estimate – using the figures presented here – of yearly net profit for an average 53 ha farm of *ca*. £12,309 is likely an overestimate, but reasonably close to an independently-derived UK-wide farm estimate for 2000 (£8,700).

CARBON STOCK AND NET CHANGE

We found no significant difference in the total carbon stock of Dorset in the 1930s and 2000 (Table 2). We calculated stocks of 24.5 million tonnes in the 1930s and 22.6 million tonnes in 2000 (Table 2; $P > 0.1$). However, the distribution of carbon changed greatly between both periods (Figs. 2a & 2b). The total carbon stock in semi-natural habitats, especially in unimproved grasslands, was substantially reduced; reflecting the loss in area of these habitats. Much of these habitats were converted to land-uses containing lower carbon stocks*, i.e.* arable land and improved grassland (Hooftman & Bullock 2012). However, this reduction was balanced by an increase in woodland area, which has high carbon stocks (Table 1). Consequently, the carbon stock is now concentrated in hotspots of woodland fragments, in which 11% of the area contains approximately 50% of the Dorset carbon stock.

This shift into hotspots is demonstrated by a lower Gini coefficient in 2000 ($G = 0.80$; 5% CI ± 0.001) than in the 1930s (G =0.91; 5% CI ± 0.001). Further illustration is given by the map of the net. carbon change (Fig. 2c), which shows that the majority of land in Dorset lost carbon between the 1930s and 2000 while carbon was gained in woodland hotspots.

BIODIVERSITY

The distribution of species and the resulting mean α-diversity changed substantially, as expected ($P < 0.001$; Fig. 3). The average number of plant species *per* 2 x 2 km grid cell decreased from 393 (5% CI: 385–402) to 289 (5% CI: 284–294). This general decline is demonstrated by a slightly more equal distribution of diversity by 2000 (1930s: G = 0.825; 5% CI: 0.818 - 0.834 vs. 2000: G = 0.86 5% CI: 0.853–0.869; P <0.001). In both periods there were hotspots in a background of low diversity, but these hotspots had more species in the 1930s. However, the maps (Fig. 3) show only the south-east of Dorset maintained diversity, suggesting a single, major hotspot in 2000.

Discussion

We studied changes in spatial patterns of two ecosystem services and biodiversity for Dorset in the 1930s and 2000 using extended benefit transfer and survey data. Biodiversity decreased over this period, while agricultural and timber production (provisioning) increased. Contrary to expectations, the estimated carbon stock did not decline despite large land-use change. It appears that carbon lost through conversion of semi-natural habitats to intensive agriculture was balanced by accumulation of carbon in the increased woodland area. The spatial distributions of these measures changed markedly: both carbon and provisioning became more unequally distributed among grid cells, indicating concentration of service delivery into hotspots, while biodiversity showed more even decreases.

The loss of biodiversity and increase in agricultural and timber production reflect the UK-wide trends reported in the UKNEA (2011) and global patterns (Ellis & Ramankutty 2008; Butchart *et al.* 2010). Indeed, the human requirement for increased provisioning is thought to be a major driver of biodiversity declines (Rey Benayas & Bullock 2012). By using 2000 commodity values for both periods we ensured that the observed spatial changes reflect land-use intensification but not changes in individual commodity prices. The prices of milk and beef increased 16-fold, of wheat and barley 7-fold and of potatoes 19-fold between 1929 and 2000 (Table S6). Since inflation over this period was 37-fold [safalra.com/other/historical-uk-inflation-price-conversion/], these demonstrate decreases in market values (see also Angus et al. 2009). Changes in the relative values of commodities may explain changes in agricultural practices, but are less relevant to our estimation of changes in the agricultural production service over the 70 year period.

The UKNEA (2011) makes no general statement about 20th century changes in the UK carbon stock , but the country-wide trends of carbon gain through increasing woodland area and loss through conversion of semi-natural habitats to intensive agriculture (Smith *et al.* 2011) are reflected in our Dorset analysis. The outcome that the carbon gains have balanced the losses in Dorset is a chance one as, clearly, land-use change was not done with carbon in mind.

Agricultural provisioning and carbon stocks became more concentrated into hotspots and high biodiversity was maintained only in the south-east. This supports our hypothesis that land-use change creates hotspots through fragmentation, highlighting a dynamic process which should be considered in land use planning. These results may suggest that the 'land sparing' approach to separating biodiversity conservation and agriculture (Phalan *et al.* 2011) could be extended such that different parts of the landscape are used for different ecosystem services. In Dorset, the maps for 2000 show separation of hotspots for agriculture (north of the county), carbon stocks (scattered woodland patches) and biodiversity (remaining semi-natural habitat). However, patterns of trade-offs between biodiversity and multiple ecosystem services are complex (e.g. Anderson *et al.* 2009; Nelson *et al.* 2009) and management for a particular service or biodiversity target may create further trade-offs with other services or aspects of biodiversity (Bullock *et al.* 2011a). We can see this occurring in Dorset: much heathland has been lost to encroaching woodland (Rose *et al.* 2000), and woodland contains most of the carbon stock. Therefore, on-going tree-felling to restore these biodiversity hotspots could have a clear impact on carbon storage. Restoring the 4000 ha heathland that converted to woodland over this period would reduce the Dorset carbon store by 5%. Conversely, tree planting may have negative effects on other services such as water supply and soil quality (Jackson *et al.* 2005). Furthermore, a land sparing approach is a compromise as production, carbon stocks and biodiversity were more spatially intermingled and evenly distributed in the 1930s, suggesting that a more historically-informed approach might seek to restore habitats which deliver multiple services and biodiversity. For example, species-rich grasslands may support moderate forage production, crop pollination and pest control, carbon sequestration and cultural services (Bullock *et al.* 2011b).

CAVEATS IN MAPPING CHANGES IN SERVICES AND BIODIVERSITY

A few studies have determined regional time trends in ecosystem services (e.g. Carreno *et al.* 2012; Dearing *et al.* 2012), including the UKNEA (2011), but none have created historical maps of service delivery. Making such maps using benefit transfer involves assumptions about relevant proxies (Eigenbrod *et al.* 2010). To mitigate any resulting biases we employed a Monte Carlo bootstrap procedure incorporating variation in proxies for carbon storage and agricultural and timber production. However, the outcomes remain dependent on the accuracy of the underlying data. We acknowledge that employing benefit transfer *per* land-use type may introduce noise because of variation in service values within land cover types (Eigenbrod *et al.* 2010).

A second caveat is that we analysed change using only two points in time. Simply, no similar maps are generally available for the 1950s or 1960s. However, Hooftman & Bullock (2012) showed that land-use change in Dorset showed a roughly linear trend over the last century. Nevertheless, patterns of production and the prices of agricultural commodities fluctuated over the study period (Edward-Jones *et al.* 2011), and our snapshots do not capture these temporal subtleties.

Lastly, we used current day estimates of carbon stock which may bias our estimates of change. We have no reason to believe that these values would have been different in the 1930s, but can speculate. As elsewhere, semi-natural habitats in Dorset are undergoing eutrophication (Keith *et al.* 2009; Newton *et al.* 2012b), which can increase carbon sequestration (de Vries *et al.* 2009). Conversely, arable carbon stocks may have declined since the 1930s due to factors such soil compaction and degradation, replacement of farmyard manure with inorganic fertilisers and reduced rotation with grass leys (Smith *et al.* 2011).

APPLYING HISTORICAL SERVICE CHANGES IN LAND-USE PLANNING

Our findings and approach can be applied to developing ecosystem service-based management and policy. For action 5 of the EU biodiversity strategy it is specified that maps should be valuable for prioritisation and problem identification, showing synergies and trade-off between services. Furthermore, maps can be used as visual communication tools to initiate discussions with stakeholders (Maes *et al.* 2012). While mapping is not a new approach we propose that it is imperative to add to this map-based information the changes that have occurred in location-specific delivery of services and their spatial patterns. For problem identification, local drivers and conflicts could thus be identified and tackled. Our maps can do this in Dorset, since land-use transitions have been identified (Hooftman & Bullock 2012).

In developing management plans, maps such as ours are valuable for framing optimisation strategies in land-use allocation and management based on synergistic and antagonistic effects among services, for example using GIS-based service modelling tools such as InVEST (Nelson *et al.* 2009; Goldstein *et al.* 2012). Such activities would be aided by understanding historical changes: the 1930s maps provide a baseline indicating the capacity of a local area for sustainable land-use change, while clarifying trade-offs such as potential production losses. Paleo-environmental methods may also provide information for such endeavours (Dearing *et al.* 2012). We envisage that management plans for the twelve newly-created UK Nature Improvement Areas (HM Government, 2011) would benefit from such information; indeed, our Dorset maps are being used by the Wild Purbeck NIA (Ian Rees pers. comm.; www.dorsetaonb.org.uk/our-work/wildpurbeck.html).

Considering the range of other services provided by Dorset's ecosystems – including tourism, clean water supply, flood mitigation and erosion control – a development of this study could involve mapping multiple services in the 1930s and 2000 together. This might be done using land-use based proxies for services such as recreational value (e.g. Newton *et al.* 2012a) or modelling using land-use, topography and other geographical variables for services such as flood mitigation (e.g. Eigenbrod *et al.* 2011).

Acknowledgements

DERC provided the Good data. We thank Nick Isaac for advice on species–area curves. The comments of the reviewers and the associate editor greatly improved the manuscript. This project was supported by SCALES EU–FP7–226852.

References

Agricultural Statistics (1929) *General survey of agricultural production in England and Wales.* Conservative Party Archive at Bodleian Library, Oxford. Visited on 27/07/2011.

Anderson, B.J., Armsworth, P.R., Eigenbrod, F., Thomas, C.D., Gillings, S., Heinemeyer, A., Roy D.B. & Gaston, K.J. (2009) Spatial covariance between biodiversity and other ecosystem service priorities. *Journal of Applied Ecology*, **46**, 888-896.

Angus, A., Burgess, P.J., Morris, J. & Lindgard, J. (2009) Agriculture and land use: Demand for and supply of agricultural commodities characteristics of the farming and food industries, and implications for land use in the UK. *Land use Policy***, 26S**, S230-S242.

Bai Y., Zhuang C.W., Ouyang Z.Y., Zheng H. & Jiang B. (2011) Spatial characteristics between biodiversity and ecosystem services in a human-dominated watershed. *Ecological Complexity*, **8**, 177-183.

Birch, J.C., Newton, A.C., Aquino, C.A., Cantarello, E., Echeverría, C., Kitzberger, T., Schiappacasse, I. & Garavito, N.T. (2010) Cost-effectiveness of dryland forest restoration evaluated by spatial analysis of ecosystem services. *Proceedings of the National Academy of Sciences*, **107**, 21925-21930.

Bennett, E.M., Peterson, G.D. & Gordon, L.J. (2009) Understanding relationships among multiple ecosystem services. *Ecology Letters*, **12**, 1394-1404.

Boakes E.H., Mace G.M., McGowan P.J.K. & Fuller R.A. (2010) Extreme contagion in global habitat clearance. *Proceedings of the Royal Society B-Biological Sciences*, **277**, 1081-1085.

Bowen, H. (2000) *The Flora of Dorset*. Pisces Publications, Newbury, United Kingdom.

Bullock, J.M., Aronson, J., Newton, A.C., Pywell, R.F. & Rey-Benayas J.M. (2011a) Restoration of ecosystem services and biodiversity: conflicts and opportunities. *Trends in Ecology & Evolution*, **26**, 541-549.

Bullock, J.M., Jefferson, R.G., Blackstock, T.H., Pakeman, R.J., Emmett, B.A., Pywell, R.F., Grime, J.P. & Silvertown, J. (2011b) Semi-natural grasslands. *The UK National Ecosystem Assessment Technical Report*, pp. 161-196. UNEP-WCMC Cambridge.

Butchart, S.H.M., *et al.* (2010) Global biodiversity: indicators of recent declines. *Science*, **328**, 1164-1168.

Carreno, L., Frank, F.C. & Viglizzo, E.F. (2012) Tradeoffs between economic and ecosystem services in Argentina during 50 years of land-use change. *Agriculture Ecosystems & Environment*, **154**, 68-77.

Chan, K.M.A., Shaw, M.R., Cameron, D.R., Underwood, E.C. & Daily, G.C. (2006) Conservation planning for ecosystem services. *PLOS Biology*, **4**, 2138-2152.

Cons. Archive (1940) *Number of Cattle for milk production in England and Wales in 1929-30: Agricultural policy, 1930-40.* CRD 1/30/1-6. Conservative Party Archive, Bodleian Library, University of Oxford. Visited on 19/12/2012

Conte, M., Nelson, E., Carney, K., Fissore, C., Olwero, N., Plantinga, A.J., Stanley, B. & Ricketts, T. (2011) Terrestrial carbon sequestration and storage. *Natural capital: theory and practice of mapping ecosystem services* (eds P. Kareiva, H. Tallis, T.H. Ricketts, G.C. Daily & S. Polasky). pp. 111-128. Oxford University Press, Oxford.

de Vries, W., Solberg, S., Dobbertin, M., Sterba, H., Laubhann, D., van Oijen, M., Evans, C., Gundersen, P., Kros, J., Wamelink, G.W.W., Reinds, G.J. & Sutton, M.A. (2009) The impact of nitrogen deposition on carbon sequestration by European forests and heathlands. *Forest Ecology and Management*, **258**, 1814-1823.

Dallimer, M., Tinch, D., Acs, S., Hanley, N., Southall, H.R., Gaston, K. & Armsworth, P.R. (2009) 100 years of changes: examining agricultural trends, habitat change and stakeholder perceptions through the 20th century. *Journal of Applied Ecology*, **46**, 334-343.

Dearing, J.A., Yang, X., Dong, X., Zhang E., Chen X., Langdon, P.G. Zhang, K., Zhang W. & Dawson, T.P. (2012) Extending the timescale and range of ecosystem services through paleo-environmental analyses, exemplified in the lower Yangtze basin. *Proceedings of the National Academy of Sciences*, **109**, E1111-E1120.

Defra (2010) *Delivering a healthy natural environment*. Defra, London.

Defra (2011) *June survey of agriculture and horticulture*. http://archive.defra.gov.uk/evidence/statistics/foodfarm/landuselivestock/junesurvey/results.htm. Last accessed on: 25/01/2013.

Edwards-Jones, G., Cross, P., Foley, N., Harris, I., Kaiser, M., Le Vay, L., Rayment, M., Scowen, M. & Waller, P. (2011) Provisioning services. *The UK National Ecosystem Assessment Technical Report*, pp. 597-632. UNEP-WCMC Cambridge.

Egoh, B., Reyers, B., Rouget, M., Richardson, D.M., Le Maitre, D.C. & van Jaarsveld, A.S. (2008) Mapping ecosystem services for planning and management. *Agriculture, Ecosystems and Environment*, **127**, 135-140.

Eigenbrod, F., Armsworth, P.R., Anderson, B.J., Heinemeyer, A., Gillings, S., Roy, D.B., Thomas, C.D. & Gaston, K.J. (2010) The impact of proxy-based methods on mapping the distribution of ecosystem services. *Journal of Applied Ecology*, **47**, 377-385.

Eigenbrod, F., Bell, V.A., Davies, H.N., Heinemeyer, A., Armsworth, P.R. & Gaston, K.J. (2011) The impact of projected increases in urbanization on ecosystem services. *Proceedings of the Royal Society B-Biological Sciences*, **278**, 3201-3208.

Ellis, E.C. & Ramankutty, N. (2008) Putting people in the map: anthropogenic biomes of the world. *Frontiers in Ecology and the Environment*, **6**, 439-447.

Feng, Z., Yang, Y., Zhang, Y., Zhang, P. & Li, Y. (2004) Grain-for-green policy and its impacts on grain supply in West China. *Land Use Policy*, **22**, 301-312.

Gamboa, L.F., Garcia-Suaza, A. & Otero, J. (2010) Statistical inference for testing Gini coefficients: an application from Colombia. *Ensayos sobre Politica Economica*, **28**, 226-238.

Goldstein, J.H., Caldarone, G., Duarte, T.K., et al. (2012) Integrating ecosystem-service tradeoffs into land-use decisions. *PNAS*, **109**, 7565-7570.

Good, R. (1948) *A geographical handbook of the Dorset flora*. The Dorset Natural History and Archaeological Society, Dorchester.

Harte, J., Smith, A.B. & Storch, D. (2009) Biodiversity scales from plots to biomes with a universal species-area curve. *Ecology Letters*, **12**, 789-797.

HM Government. (2011) *The natural choice: securing the value of nature.* The Stationary Office Ltd. http://www.defra.gov.uk/environment/natural/whitepaper/. Last accessed on 25/01/2013.

Hooftman, D.A.P. & Bullock, J.M. (2012) Mapping to inform conservation: A case study of changes in semi-natural habitats and their connectivity over 70 years. *Biological Conservation*, **145**, 30-38.

Jackson, D.L. (2000) *Guidance on the interpretation of the Biodiversity Broad Habitat Classification (terrestrial and freshwater types): Definitions and the relationship with other classifications.* http://jncc.defra.gov.uk/page-2433. Last accessed on 02/09/2012.

Jackson, R.B., Jobbagy, E.G., Avissar, R., Roy, S.B., Barrett, D.J., Cook, C.W., Farley, K.A., le Maitre, D.C., McCarl, B.A. & Murray, B.C. (2005) Trading water for carbon with biological sequestration. *Science*, **310**, 1944-1947.

Keith, S.A., Newton, A.C., Morecroft, M.D., Bealey, C.E. & Bullock, J.M. (2009) Taxonomic homogenization of woodland plant communities over 70 years. *Proceedings of the Royal Society B-Biological Sciences*, **276**, 3539-3544.

Kremen, C. & Ostfeld, R.S. (2005) A call to ecologists: measuring, analyzing, and managing ecosystem services. *Frontiers in Ecology and the Environment*, **3**, 540-548.

Maes, J., *et al.* (2012) *Mapping and Assessment of Ecosystem and their services: an analytical framework for ecosystem assessment under action 5 of the EU biodiversity Strategy to 2020.* http://www.eccb2012.org. Last accessed on 25/01/2013.

Naidoo, R., Balmford, A., Costanza, R., Fisher, B., Green, R.E., Lehner, B., Malcolm, T.R. & Ricketts, T.H. (2008) Global mapping of ecosystem services and conservation priorities *PNAS*, **105**, 9495-9500

Nelson, E., Mendoza, G., Regetz, J. et al. (2009) Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. *Frontiers in Ecology and the Environment*, **7**, 4-11.

Newton, A.C., Hodder, K., Cantarello, E., Perrella, L., Birch, J.C., Robins, J., Douglas, S., Moody, C. & Cordingley, J. (2012a) Cost–benefit analysis of ecological networks assessed through spatial analysis of ecosystem services. *Journal of Applied Ecology*, **49**, 571-580.

Newton, A.C., Walls, R.M., Golicher, D., Keith, S.A., Diaz, A. & Bullock, J.M. (2012b) Structure, composition and dynamics of a calcareous grassland meta-community over a 70-year interval. *Journal of Ecology*, **100**, 196-209.

MEA (2005) *Millennium Ecosystem Assessment*. Washington DC, Island Press.

Ostle, N.J., Levy, P.E., Evans, C.D. & Smith, P. (2009) UK land use and soil carbon sequestration. *Land Use Policy,* **26**, S274-S283.

Phalan, B., Onial, M., Balmford, A. & Green, R.E. (2011) Reconciling food production and biodiversity conservation: land sharing and land sparing compared. *Science*, **333**, 1289-1291.

Raudsepp-Hearne, C., Peterson, G.D. & Bennett, E.M. (2010) Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. *Proceedings of the National Academy of Sciences of the United States of America*, **107**, 5242-5247.

Rey Benayas, J.M. & Bullock, J.M. (2012) Restoration of biodiversity and ecosystem services on agricultural land. *Ecosystems*, **6**, 883-899.

Rey Benayas, J.M., Newton, A.C., Diaz, A. & Bullock, J.M. (2009) Enhancement of biodiversity and ecosystem services by ecological restoration: a meta-analysis. *Science*, **325**, 1121-1124.

Rose, R.J., Webb, N.R., Clarke, R.T. & Traynor, C.H. (2000) Changes on the heathlands in Dorset, England, between 1987 and 1996. *Biological Conservation*, **93**, 117-125.

Smith, P., *et al.* (2011) Regulating services. *The UK National Ecosystem Assessment Technical Report*, pp. 535-596. UNEP-WCMC Cambridge.

Swetnam, R.D., Fisher, B., Mbilinyi, B.P., Munishi, P.K., Willcock, S., Ricketts, T., Mwakalila, S., Balmford, A., Burgess, N.D., Marshall, A.R. & Lewis, S.L (2011) Mapping socio-economic scenarios of land cover change: a GIS method to enable ecosystem service modelling. *Journal of Environmental Management*, **92**, 563-574.

Tallis, H. & Polasky, S. (2011) Assessing multiple ecosystem services and integrated tool for the real world. *Natural capital: theory and practice of mapping ecosystem services* (eds Kareiva, P., Tallis, H., Ricketts, T.H., Daily, G.C., Polasky, S.), p.p. 34-50, Oxford University Press, Oxford.

Tavener, L.E. (1937) Land classification in Dorset: a study of land classification with special reference to agricultural distributions in Dorset. *Transactions (Institute of British Geographers)*, **6**, 1-61.

Tavener, L.E. (1955) Dorset farming, 1900-1950. *Proceedings of the Dorset Natural History and Archaeological Society*, **75**, 91-114.

UKNEA (2011) *The UK National Ecosystem Assessment Technical Report*. UNEP-WCMC Cambridge.

West, P.C., Gibbs, H.K., Monfreda, C., Wagner, J., Barford, C.C., Carpenter, S.R. & Foley, J.A. (2010) Trading carbon for food: Global comparison of carbon stocks vs. crop yields on agricultural land. *Proceedings of the National Academy of Sciences of the United States of America*, **107**, 19645-19648.

Table 1. Monetary values (in British pounds) of annual agricultural and timber production for each relevant land-use type in Dorset for the 1930s and 2000, with totals and averages *per* hectare derived using a Monte Carlo algorithm. Production values are based on 2000 commodity prices

Land-use type	**1930s**			**2000**		
	Area (km^2)	Total (million £)	ha^{-1} (£)	Area (km^2)	Total (million £)	ha^{-1} (£)
Unimproved grassland	1541	23.26	151	138	2.91	211
Improved grassland	-[†]			809	140.6	1,737
Arable cropland	449	7.32	163	1026	72.1	703
Woodland (= timber & non-wood products)[‡]	205	2.34	114	268	3.05	114
Heathland[‡]	138	0.045	3	62	0.020	3
Total	2334[§]	32.93	141	2303[§]	218.7	950
95% Confidence interval		32.01 33.03			208.3 276.5	

† The area of improved grassland in the 1930s was none to negligible (Hooftman & Bullock 2012).

‡ Timber and non-forest product values *per* hectare used are identical for the 1930s and 2000

§ Area of productive land.

Table 2. Total carbon stock in tonnes for each relevant land-use type in Dorset for the 1930s and 2000, with totals and averages *per* hectare, derived using a Monte Carlo algorithm (Supporting Information). Net carbon change is calculated *per* grid cell as the difference between the two periods

Land-use type	**1930s**			**2000**		
	Area (km^2)	Total (million tonnes)	ha^{-1}	Area (km^2)	Total (million tonnes)	ha^{-1}
Unimproved grassland	1541	10.00	65	138	0.72	52
Improved grassland	-†			809	2.27	28
Arable cropland	449	2.27	62	1026	6.32	62
Woodland (timber & non-wood products)	205	8.19	400	268	10.78	403
Heathland	138	1.09	79	62	0.49	79
Other land-uses	168	0.66	39	198	0.60	30
Total	2501	24.48	98	2501	22.63	91
95% Confidence interval		17.10 37.10			14.73 32.31	
Net carbon change‡ (95% Confidence interval)					-1.98 (-14.62 7.92)	

† The area of improved grassland in the 1930s was none to negligible (Hooftman & Bullock 2012).

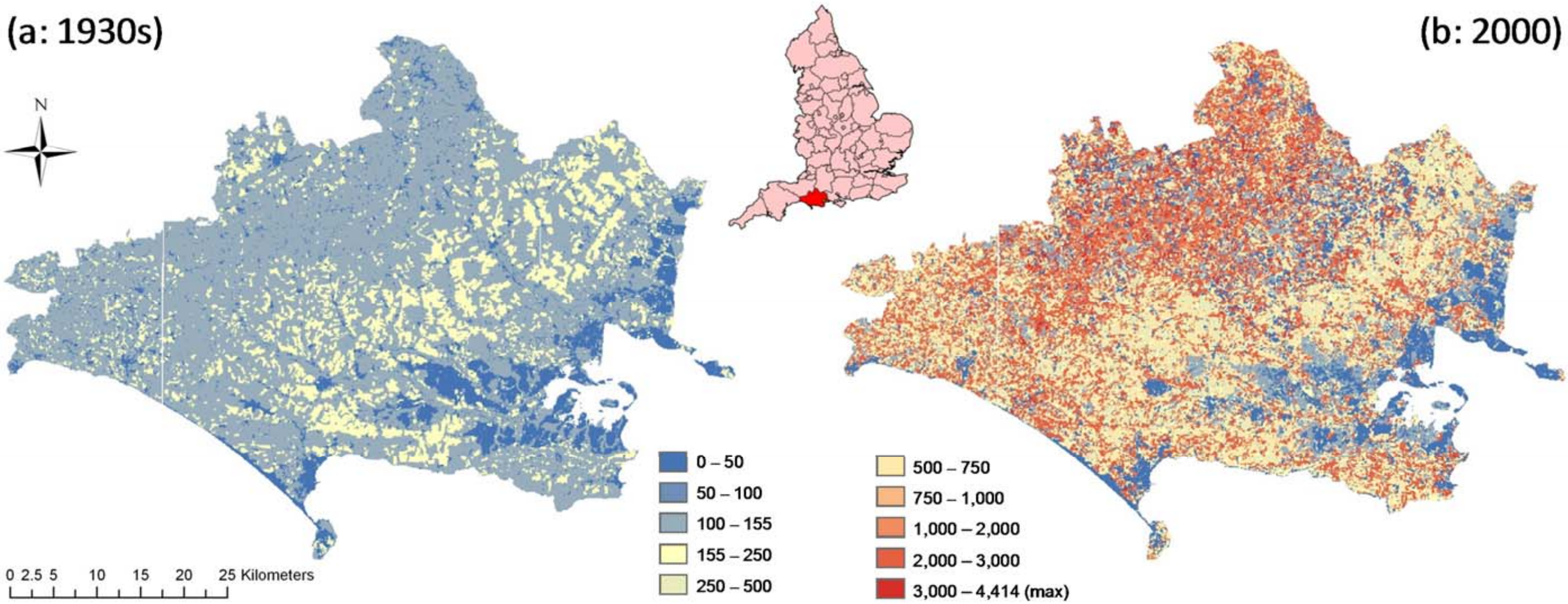
‡ Calculated *per* grid cell and summed over all cells.

Figures

Fig 1. Combined annual agricultural and timber production in 1930s (a) and 2000 (b) in British pounds *per* hectare. Grid size is 100 x 100 m (1 ha). Values are the mean of 50,000 Monte Carlo runs based on the different estimates *per* land-use type.

Fig 2. Carbon stock in 1930s (a) and 2000 (b) *per* hectare, being the sum of carbon in above-ground biomass, below-ground biomass, dead carbon (*i.e.,* litter and other dead organic matter), and soil carbon. (c) The net carbon change is the difference between the two periods *per* grid cell. Grid size is 100 x 100 m (1 ha). Values are the mean of 50,000 Monte Carlo runs based on the different estimates *per* land-use type.

Fig. 3 Species richness of vascular plant species in the (a) 1930s and (b) 2000 on a 2 x 2 km grid. 1930s data are extrapolated from the smaller vegetation surveys using species-area curves. 2000 data are collated records for 1985–2000 (Bowen 2000).



(a: 1930s)
(b: 2000)
N
0 2.5 5 10 15 20 25 Kilometers
0 – 50
50 – 100
100 – 155
155 – 250
250 – 500
500 – 750
750 – 1,000
1,000 – 2,000
2,000 – 3,000
3,000 – 4,414 (max)

Figure 1.

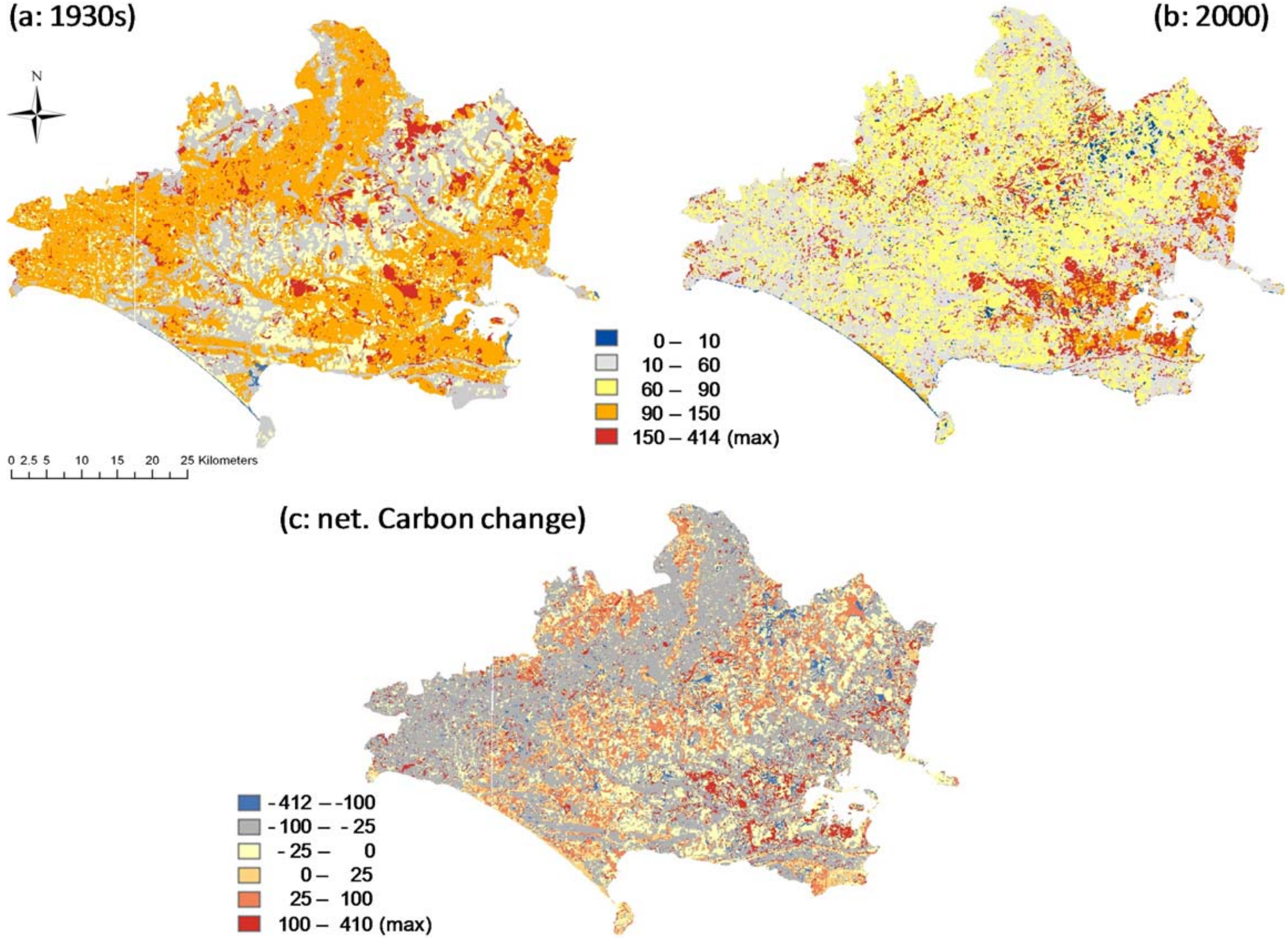

(a: 1930s)
(b: 2000)
N
0 2.5 5 10 15 20 25 Kilometers
0 – 10
10 – 60
60 – 90
90 – 150
150 – 414 (max)
(c: net. Carbon change)
-412 – -100
-100 – -25
-25 – 0
0 – 25
25 – 100
100 – 410 (max)

Figure 2.

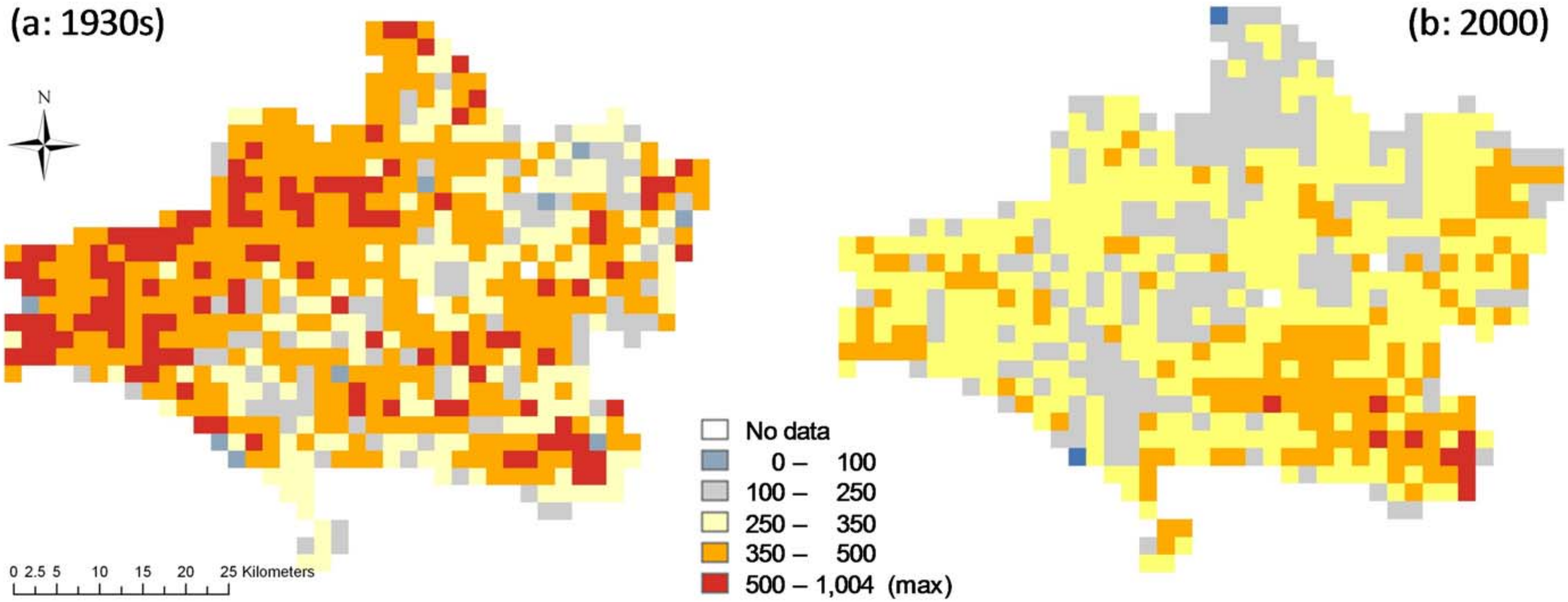

(a: 1930s)
(b: 2000)
N
0 2.5 5 10 15 20 25 Kilometers
No data
0 – 100
100 – 250
250 – 350
350 – 500
500 – 1,004 (max)

Figure 3.

Supporting Information

Additional supporting information may be found in the online version of this article.

Appendix S1. Description of mapping method

Appendix S2. Proxy search procedure

Appendix S3. Biodiversity distribution and species-area relationship evaluations

Appendix S4. Validation runs for bootstrap and Gini procedures

Appendix S5. Agricultural production validation

Table S1. Agricultural and timber production values (yield and price) for woodlands & heathlands.

Table S2. Prices and yields for the crops planted in Dorset.

Table S3. Annual agricultural production values on grasslands in the LCM2000 map.

Table S4. Conversion of livestock to Livestock Units.

Table S5. Differences in agricultural production on semi-natural grasslands in the 1930s map between livestock and dairy.

Table S6. Relative price changes of agricultural commodities.

Table S7. Above ground carbon stock values per land-use category used in this study.

Table S8. Soil carbon stock values per land-use category used in this study.

Table S9. Below ground and dead material carbon stock values per land-use category used in this study.

References used in the Supporting Information

Codes S1. Matlab codes used for the Monte Carlo (bootstrap) procedure and Gini calculations.