



Historical influences on the current provision of multiple ecosystem services



Martin Dallimer^{a,*}, Zoe G. Davies^b, Daniel F. Diaz-Porras^d, Katherine N. Irvine^{e,f}, Lorraine Maltby^c, Philip H. Warren^c, Paul R. Armsworth^g, Kevin J. Gaston^h

^aSustainability Research Institute, School of Earth and Environment, University of Leeds, Leeds LS9 2JT, UK

^bDurrell Institute of Conservation and Ecology (DICE), School of Anthropology and Conservation, University of Kent, Canterbury CT2 7NR, UK

^cDepartment of Animal and Plant Sciences, University of Sheffield, S10 2TN Sheffield, UK

^dEscuela de Ciencias, Universidad Autónoma 'Benito Juárez' de Oaxaca, Oaxaca, Mexico

^eSocial, Economic and Geographical Sciences Research Group, James Hutton Institute, Craigiebuckler, Aberdeen AB15 8QH, UK

^fInstitute of Energy and Sustainable Development, De Montfort University, Leicester, UK

^gEcology and Evolutionary Biology, University of Tennessee, Knoxville, TN 37996, USA

^hEnvironment and Sustainability Institute, University of Exeter, Cornwall TR10 9FE, UK

ARTICLE INFO

Article history:

Received 20 April 2014

Received in revised form 31 August 2014

Accepted 15 January 2015

Available online 18 March 2015

Keywords:

Above-ground carbon
Cultural heritage
Historical ecology
Land-use change
Species richness
Urban greenspace

ABSTRACT

Ecosystem service provision varies temporally in response to natural and human-induced factors, yet research in this field is dominated by analyses that ignore the time-lags and feedbacks that occur within socio-ecological systems. The implications of this have been unstudied, but are central to understanding how service delivery will alter due to future land-use/cover change. Urban areas are expanding faster than any other land-use, making cities ideal study systems for examining such legacy effects. We assess the extent to which present-day provision of a suite of eight ecosystem services, quantified using field-gathered data, is explained by current and historical (stretching back 150 years) landcover. Five services (above-ground carbon density, recreational use, bird species richness, bird density, and a metric of recreation experience quality (*continuity with the past*)) were more strongly determined by past landcover. Time-lags ranged from 20 (bird species richness and density) to over 100 years (above-ground carbon density). Historical landcover, therefore, can have a strong influence on current service provision. By ignoring such time-lags, we risk drawing incorrect conclusions regarding how the distribution and quality of some ecosystem services may alter in response to land-use/cover change. Although such a finding adds to the complexity of predicting future scenarios, ecologists may find that they can link the biodiversity conservation agenda to the preservation of cultural heritage, and that certain courses of action provide win-win outcomes across multiple environmental and cultural goods.

© 2015 The Authors. Published by Elsevier Ltd. This is an open access article under the CC BY license (<http://creativecommons.org/licenses/by/4.0/>).

1. Introduction

Land-use change has led to substantive alterations in the amount and quality of services that ecosystems can provide (Dearing et al., 2012; Tianhong et al., 2010; Zhao et al., 2006). While mapping ecosystem services is a necessary first step in developing strategies for their maintenance (Chan et al., 2006; Davies et al., 2011; Naidoo et al., 2008; Raudsepp-Hearne et al., 2010a), it presents a static picture of current/contemporary distributions (e.g., Lautenbach

et al., 2011; Jiang et al., 2013). To understand how best to manage ecosystem service provision in response to land-use/cover (LU/LC) change, an expanding body of work has developed model-based scenarios to predict likely future consequences (Kareiva et al., 2011; Nelson et al., 2009; UKNEA, 2011). These studies often highlight a decline in some services as human development (e.g. urbanisation, intensification of agriculture) proceeds. Counter-intuitively, even though ecosystem services are increasingly degraded (MEA, 2005), human well-being continues to improve globally. One possible explanation may be that time-lags exist between the effects of human-driven land transformations and present-day provision of ecosystem services (Raudsepp-Hearne et al., 2010b).

Indeed, time-lags and feedbacks are a common and widespread response to perturbations in many biological systems (Foster et al., 2003; Nicholson et al., 2009). Historical land-use change has been shown to influence ecosystem function in a broad range of studies,

* Corresponding author. Tel.: +44 113 343 2846.

E-mail addresses: m.dallimer@leeds.ac.uk (M. Dallimer), z.g.davies@kent.ac.uk (Z.G. Davies), dfdporras@gmail.com (D.F. Diaz-Porras), katherine.irvine@hutton.ac.uk (K.N. Irvine), l.maltby@sheffield.ac.uk (L. Maltby), p.warren@sheffield.ac.uk (P.H. Warren), p.armsworth@utk.edu (P.R. Armsworth), k.j.gaston@exeter.ac.uk (K.J. Gaston).

with legacy effects that can last tens, hundreds or even thousands of years (Foster et al., 2003; Szabo and Hedl, 2011). For example, time-lags in extinction processes following habitat loss have been documented across several taxa (Kuussaari et al., 2009; Tilman et al., 1994). Given that many long-lived plants, or those with certain life-history traits (e.g. clonal propagation, extensive seed banks) are able to persist for long periods after conditions become unfavourable (Eriksson, 1996), services underpinned by vegetation are perhaps the most likely to be subject to a legacy of past land-use. For instance, plant species diversity in grassland is often heavily influenced by historical management (Gustavsson et al., 2007; Pärtel et al., 1999) and, similarly, harvesting and wildfires in forest habitats can limit annual carbon stored over 60 years later (Gough et al., 2007). Vegetation carbon storage is primarily determined by tree size (e.g., Davies et al., 2011) and thus has a strong link to past land-use and management.

Likewise, legacies can also be expected in a social context, which could influence the provision of cultural ecosystem services, such as the number of recreational visitors to a particular location, or the values that people associate with a certain site. For example, in the built environment, features can act as “icons” (Hull et al., 1994). Such icons can convey a connection with the past, self-identity and a sense of community for local residents. Indeed, a central aim of built cultural heritage preservation is to enhance the continuity of the historical environment. This, in turn, helps to connect people with both place and culture, thereby contributing to how desirable a place is to live and/or to visit (e.g., Ashworth, 2008).

There is therefore a need to quantify the extent to which historical land-use determines the distribution of present-day ecosystem service provision. An analysis of this type is particularly pertinent within human-dominated regions, such as urban areas, where shifts in landcover are dynamic, changing rapidly in response to policy (Dallimer et al., 2011; McDonald et al., 2010). Urbanisation is a major driver of land-use change globally (Seto et al., 2012), and will continue to be given that the proportion of the world’s population that lives in cities is predicted to rise to 70% over the next 40 years (United Nations, 2013). Furthermore, towns and cities are set to expand disproportionately, as increases in the area of urbanised land generally outpace population growth (Liu et al., 2003; ONS, 2012b).

Urban development has profound impacts on ecosystem service provision (Güneralp et al., 2013; Seto et al., 2012; Tianhong et al., 2010; Zhao et al., 2006), not least because the costs and benefits of green infrastructure (the network of greenspaces, water and other vegetated features in towns and cities) are rarely considered in expanding cities. This is despite the plethora of studies which have demonstrated the importance of urban green infrastructure in supporting the delivery of multiple services, such as temperature mitigation (Myint et al., 2013; Park et al., 2012; Susca et al., 2011), pollution reduction (Manes et al., 2012; Pugh et al., 2012), biological carbon storage (Davies et al., 2011), promoting human health and well-being (Mitchell and Popham, 2007; Ward-Thompson et al., 2012; Dallimer et al., 2012a; Irvine et al., 2013; Keniger et al., 2013), facilitating good social relations (Kuo and Sullivan, 2001; Sullivan et al., 2004), and the provision of habitat for biodiversity (Davies et al., 2011; Goddard et al., 2010; Dallimer et al., 2012b). The social and cultural value of urban greenspaces is also important (Barau et al., 2013; Gomez-Baggethun and Barton, 2013; Tzoulas et al., 2007).

Here, for a suite of eight ecosystem services delivered by urban greenspaces (Table 1), we examine the influence of historical landcover on present-day service provision within the city of Sheffield, UK. Using the UK National Ecosystem Assessment classification (UKNEA, 2011), we do this for one regulatory service (above-ground carbon storage), multiple dimensions of two cultural services (number of recreational users and the quality of recreational experience in terms of the self-reported psycho-

logical well-being of visitors to urban greenspaces) and three measures of wild species diversity (species richness and density of two highly visible and charismatic taxonomic groups; plants and birds). The choice of services was influenced by the desirability of having a spatially and temporally synchronous primary dataset likely to span a broad range of potential historical relationships. We were thus constrained to a combination of measures that was compatible with the resources available for data collection. However, if anything, the eight measures are biased towards those with a cultural dimension, which are often thought of as more difficult to quantify (UKNEA, 2011).

2. Materials and Methods

2.1. Study System

Sheffield (53°22'N, 1°20'W) is a typical large city in England (Jenks and Jones, 2010) and has a human population of 552,700 (ONS, 2012a). It lies at the confluence of five rivers, the Loxley, Rivelin, Porter, Sheaf and Don. A sixth, the Blackburn, enters the city on its eastern fringes where it joins the Don (Fig. 1). The rivers have a long history of human exploitation and their physical properties have been critical in determining the development of Sheffield (Crossley and Cass, 1989). Riparian zones therefore make an ideal system to investigate land-use legacies on ecosystem service provision and form the focus of this study.

Industrial output and the human population of the city peaked in the 1950s, and both contracted rapidly through the latter half of the 20th century, resulting in large areas of vacant former industrial land by the mid-1980s (Hey, 2005), much of which has subsequently been redeveloped. Pollution and environmental degradation followed the rapid urbanisation and, despite the early recognition of the importance of greenspaces associated with rivers (Abercrombie, 1924), the Don remained highly polluted until the 1980s (Firth, 1997). Much of this particular river is still dominated by large-scale industrial and commercial use. Despite this history of human exploitation, long-established public parks and networks of footpaths are located along the Porter, Rivelin and Sheaf that pass through residential areas of south and west Sheffield. More recent redevelopment initiatives have incorporated new public greenspaces and access routes along the city’s waterways. In parallel, there has been a renewed focus on the appreciation of the historical importance of the city’s rivers (e.g., Griffiths, 1999; Kendall, 2005). Given that riparian areas are distributed throughout the urban, suburban and more rural periphery of the city, they have the potential to deliver a range of ecosystem services to urban dwellers and we can expect that there would be an historical aspect to their provision.

2.2. Survey Design

To ensure that the sampling adequately covered the environmental variation across the riparian zones in the study area at the present time, we followed Gradsect survey design principles (Austin and Heyligers, 1989), by characterising Sheffield according to present-day landcover and river features (Dallimer et al., 2012b). This provided 81 survey points in the urban area and immediate rural surroundings. To further extend the variability covered, an additional 26 survey sites were placed along rivers at increasing distances from the urban centre, giving 107 locations in total (Fig. 1). Although we wished to generate data covering the complete suite of ecosystem services across all sites, sample sizes were constrained for a number of measures (Table 2). This was primarily due to logistical difficulties associated with resource-intensive data collection, or access restrictions being put in place while fieldwork was ongoing.

Table 1

Ecosystem services quantified across riparian greenspaces in the city of Sheffield, UK, and their hypothesised relationship with historical land-use, cover or management.

Service	Context	Historical influences	Hypothesised relationship with past land-use
Above-ground carbon storage	An important ecosystem service to which towns and cities can contribute. Although small compared with carbon emissions, the size of urban carbon reservoirs appears substantial (Davies et al., 2011).	Above-ground carbon storage is largely determined by the density, size and species of tree present (Davies et al., 2011). Tree cover in cities constantly changes due to natural processes (growth/regeneration) and human decisions to plant/remove individual trees (Nowak and Greenfield, 2012). Current vegetation structure and cover in urban areas is often better explained by past conditions (Hope et al., 2003; Luck et al., 2009; Pickett et al., 2008).	Strong
Wild species diversity	Biodiversity is considered central to supporting all ecosystem services (Balvanera et al., 2006), but is often not thought of as a service (Mace et al., 2012). We include the richness of two highly visible and culturally important groups (plants, birds) due to their associated use and non-use values for UK citizens (UKNEA, 2011). Large numbers of people actively participate in citizen science projects (e.g., over 600,000 people took part in the RSPB's annual "Garden Bird Watch" citizen science event; and/or are members of conservation/wildlife NGOs (e.g., around 3.7 million people are members of the National Trust).	Plant species richness is related to historical land-use, management and socio-economic characteristics across locations and habitat types (Gustavsson et al., 2007; Luck et al., 2009; Pärtel et al., 1999). Although the patterns are less clear than for plants, the persistence of bird species can be determined by past land-use changes and habitat fragmentation (Ford et al., 2009; Kamp et al., 2011).	Plant richness: Strong Bird richness and density: Weak
Recreation (number of users)	Government policies (EEA, 2009; ODPM, 2003) seek to encourage increased provision and usage of urban greenspaces.	The effort invested by human volunteers in protected areas in the region is positively related to how long the area has been managed for conservation (Armsworth et al., 2013). We therefore postulate that older greenspaces will receive more visitors, perhaps because they are better known and valued by the surrounding community. Cultural heritage preservation aims to connect people with places through the historical continuity of the built environment (e.g., Ashworth, 2008). Within Sheffield, a recent focus on encouraging people to visit historical locations along the city's rivers may result in the number of recreational users at a site being related to past land-use.	Weak
Recreation (quality of experience)	Greenspaces offer residents opportunities for improving their physical and mental health (Berman et al., 2008; Bowler et al., 2010). Self-reported psychological well-being can depend on the physical properties of the greenspaces (Dallimer et al., 2012a).	Attention restoration theory proposes that the natural world, including urban greenspaces, is cognitively restorative (Kaplan and Kaplan, 1989), something that is likely to be associated with the current features of a greenspace. The sense-of-place framework suggests that the relationship between people and greenspaces may be understood in terms of the site itself. We focus here on human emotional attachments with physical locations (Altman and Low, 1992) and on the sense of identity that may be developed by association with a particular location (Proshansky et al., 1983). We consider the relationship with place to include a cognitive, or conscious, dimension such that the meaning, thoughts, values and memories of a place held by an individual are linked to one's 'sense of place'. This relationship is likely to develop and strengthen through time.	Attention restoration theory well-being components: Weak Sense-of-place well-being components: Strong

2.3. Landcover Classification

We generated a timeline of landcover for the 107 sites using data dating back over 150 years. The decades chosen reflect the availability of historical topographical maps: the 2000s (the most recent map series), 1980s, 1960s, 1940s, 1890s and 1850s (Fig. 2). Landcover at each site was recorded in one of four categories: (a) developed (buildings, roads, other impervious surfaces); (b) urban greenspace (parks, gardens, playing fields, all green open space enclosed by urban development); (c) open land outside urban areas (agricultural land, moorland) and (d) woodland (all types of tree cover). More resolved landcover classification was not possible due to variability in the clarity of

the historical topographical maps. Equally, we restricted our dataset to variables that could be acquired from all time periods. As landscape-scale processes are likely to be important in determining biodiversity and ecosystem service provision in urban areas, the proportion of a circular buffer (100 m radius) around each site that was covered by urban land was also estimated.

2.4. Quantifying Present-day Ecosystem Service Provision

2.4.1. Above-ground Carbon Storage

Tree density across the study sites was highly variable (0.0005–0.0797 trees m⁻²), so we employed a variable radius

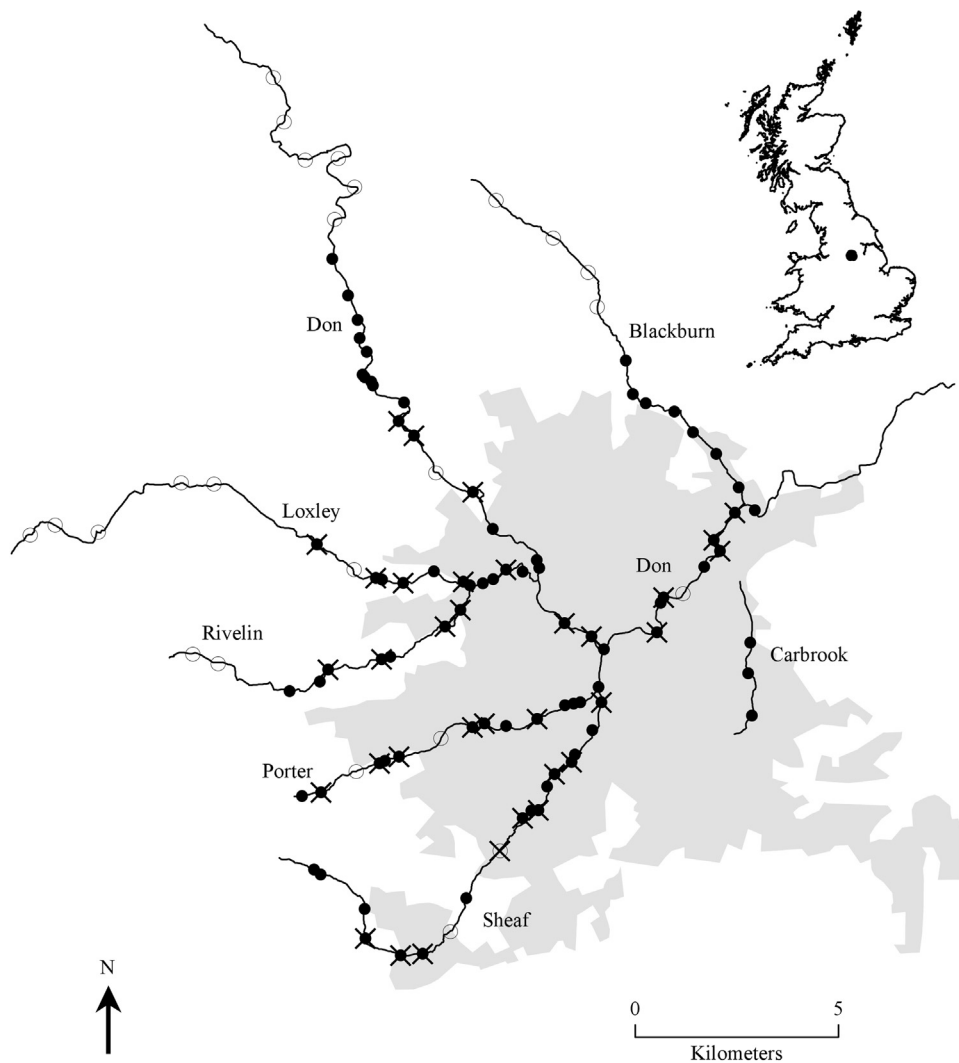


Fig. 1. The urban area of Sheffield (shaded) showing the surveyed riparian sites in the city (filled circles—recreational users, plants, birds and above-ground carbon density; open circles—plants, birds and above-ground carbon; crosses—quality of recreation experience). Rivers are indicated by solid lines. The inset shows location of the study area in Britain.

plot method (analogous to the distance sampling methods used for birds; see Section 2.4.2) with which to assess carbon storage. At each study point the distance to the five nearest trees was measured. These individuals were identified to species and their diameter at breast height and crown height recorded. Data were converted into above-ground dry-weight biomass for each tree using allometric equations obtained from the literature (Davies et al., 2011). A site-specific above-ground carbon density was calculated by dividing the total carbon stored in the five nearest trees by the circular area containing those trees (i.e., if the five trees lay within 10 m of the study point, then the total carbon stored was divided by the area of a 10 m radius circle to derive carbon density).

2.4.2. Wild Species Diversity

The density and richness of birds was surveyed across all 107 sites (Dallimer et al., 2012b; Rouquette et al., 2013). Following standard protocols, two visits were made in spring and early summer 2009 to coincide with the breeding season, with the second visit at least 6 weeks after the first. To ensure that the maximum number of species was encountered, visits began between one and three hours after sunrise (the time of highest bird activity) and were only carried out in suitable weather conditions (low wind, no rain or mist). A single observer (MD) recorded the identity of each bird that was seen or heard from the survey point over a 5 min period, excluding individuals that were flying over the site. A list of all species encountered during both visits was collated.

Table 2

Sample size (*N*), mean and standard errors (SE) and range for ecosystem services across the riparian greenspace study sites in Sheffield, UK.

Ecosystem service	Measure	<i>N</i>	Mean (\pm SE)	Range
Above-ground carbon storage	Carbon density (kg m^{-2})	102	7.66 \pm 1.01	0.01–56.50
Wild species diversity	Bird species richness	107	11.09 \pm 0.30	4–19
	Plant species richness	100	43.30 \pm 1.47	9–95
	Total bird density (birds ha^{-1})	107	21.99 \pm 0.93	5.82–74.37
Recreation	Number of users (index)	74	22.88 \pm 4.31	0–288
Quality of experience	Reflection	34	3.91 \pm 0.03	3.26–4.43
	Attachment	34	4.23 \pm 0.03	3.42–4.67
	Continuity with past	34	3.20 \pm 0.04	2.40–3.86

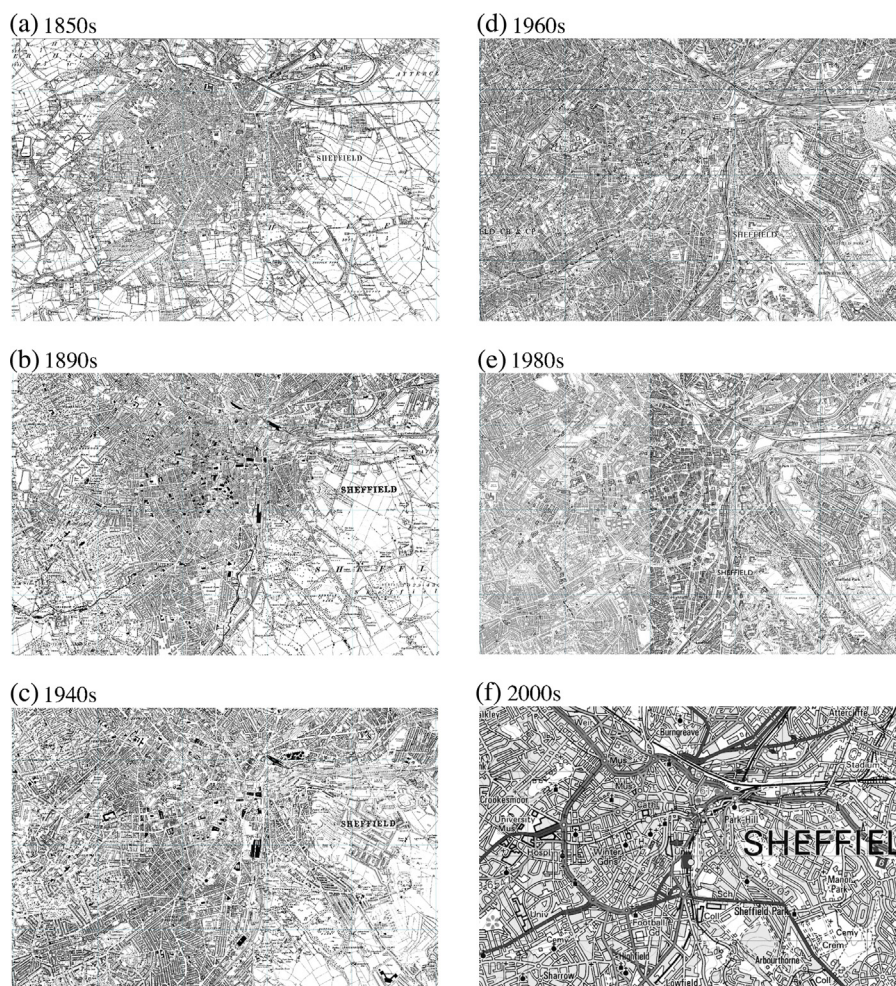


Fig. 2. Topographical maps of a part of the city of Sheffield, UK from: (a) the 1850s, (b) the 1890s, (c) the 1940s, (d) the 1960s, (e) the 1980s, and (f) the 2000s. All maps at a scale of 1:15,000. Maps (a) to (e) are from © Landmark Information Group Ltd and Crown Copyright 2014. Map (f) is © Crown Copyright/database 2014 Ordnance Survey/EDINA supplied service.

The radial distance from the observer to each bird was measured (using a laser rangefinder), which allowed distance sampling techniques to be employed (Thomas et al., 2010). Bird densities were calculated using Distance software (version 5.0, release 2), so that the variability in species' detectability can be incorporated into estimates of their density. Species-specific density functions were estimated for all species with 60 or more registrations. For less common species, a detection function was estimated using registrations for a group of similar species. We also controlled for the cue type used to locate the bird (i.e., whether the bird was only heard, or whether it was also seen). Subsequently, candidate models of the detection function were chosen and tested against the data. Model selection was based on minimum Akaike Information Criteria (AIC) and χ^2 goodness of fit tests. The detection function model was then applied to the number of encounters at each point location to give a species-specific estimate of the density. The density for all bird species recorded at a site was summed to give an estimate of the total bird density. For plant species richness, all forbs and woody plants were identified to species within a 40 m by 10 m area (long axis parallel to the river), centred on the bird survey location. Surveys were conducted at 100 sites (Dallimer et al., 2012b; Rouquette et al., 2013).

2.4.3. Number of Recreational Users

For 74 sites that had some recreational access (public parks or rights of way, informal paths), counts of all visitors within a 20 min period were made on three occasions during summer 2009. The

number of visitors in each period was summed to give an index of human use. Surveys were carried out in summer as this is the time of year when the greatest number of people is likely to be present. At the subset of 34 sites (see Section 2.4.4) where interviews were conducted, estimates of visitor numbers per hour were also recorded. The two metrics of human usage were highly correlated (Pearson's $r = 0.874$, $p < 0.001$), therefore in subsequent analyses we used the dataset with the higher sample size and broader spatial coverage (i.e., where counts took place over 20 min).

2.4.4. Quality of Recreation Experience

We used self-reported psychological well-being to measure the quality of the recreational experience. For a subset of 34 sites with good public access, we developed a questionnaire to derive estimates of individual visitor well-being (Dallimer et al., 2012a). The questionnaire was delivered face-to-face in situ to 1108 visitors. Seven closed-ended well-being statements measured reflection and contemplation, while a further 14 assessed emotional attachment and personal identity. All 21 statements used a 5-point Likert scale (1 = strongly disagree, 5 = strongly agree) in response to the stem question "Please indicate how much you agree with each statement about this stretch of river and the neighbouring banks". Factor analysis identified meaningful subscales of statements providing the following interpretable well-being axes: *reflection* (ability to think and gain perspective); *attachment* (degree of emotional ties with the stretch of river); and *continuity with past* (extent to which sense-of-identity is linked to the stretch of river through continuity

across time). Continuous measures were derived by calculating the participant's average rating of the set of statements forming each factor (for full details see Dallimer et al., 2012a).

2.4.5. Data Analysis

Taking a regression-based approach, and using the appropriate error structure (Poisson or quasiPoisson, to account for over-dispersion as necessary, for count data), we modelled present-day ecosystem service provision as a response to landcover in each of the six time periods (the 2000s, 1980s, 1960s, 1940s, 1890s, 1850s). Previous analyses revealed a lack of spatial autocorrelation in this system (Dallimer et al., 2012b), so it is not accounted for in our analyses. Explanatory variables included the landcover category at the survey site, the proportion of developed land in the 100 m buffer and their interaction. We included interactions to examine whether particular services had different forms of relationship with the proportion of developed land surrounding a survey point, dependent on the landcover at the site itself. For example, a site categorised as woodland may receive more recreational visitors as the proportion of the surrounding buffer covered by developed land increases, but the opposite may be true for a site that is categorised as developed. The smaller sample size for the recreation quality metrics precluded the inclusion of interaction terms.

As land-use at a site in one time period is likely to be related to the previous time period, data are not independent. We therefore refrained from including explanatory variables from different historical periods in the same regression model, instead electing to use AICc comparisons between full models for each time period to determine which historical landcover dataset offered the best explanation for present-day ecosystem service provision. We assumed that the model with the lowest AICc offered the best explanation for variation in present-day service provision. However, models that differ from this by $\Delta\text{AICc} < 2$ offer an equally plausible explanation for the data (Burnham and Anderson, 2002), with all models within this margin assumed to be equivalent. Thus, if the AICc from a landcover model from an historical time period fell within two AICc units of the 2000s model, we did not consider there to be evidence for historical landcover influencing current service levels. Similarly, where models from two time periods were equivalent according to AICc, we took the conservative approach of considering the most recent model as offering the best explanation for present-day service provision.

3. Results

Between the 1850s and the 2000s, the number of sites classified as urban/developed increased by an order of magnitude (Table 3). Urban greenspace sites rose fourfold and those classified as woodland doubled. These changes were matched by a concomitant decrease in the open land category. Similarly, the median proportion of urban development in the 100 m buffer surrounding each survey point rose from 0 to 50% (Table 3). Our study sites are representative of current riparian landcover and river features (see

Section 2.2). Nevertheless, until the 1960s, the pattern of historical landcover change we observed was broadly similar to that experienced by Sheffield as a whole (Table 3) (Díaz Porras, 2013). After this date, landcover trajectories diverged, with a higher proportion of the city as a whole classified as “developed” when compared to our study sites. Different landcovers exhibited varying degrees of stability through the time period of our study (Fig. 3). Developed land rarely changed categorisation, although in more recent time periods some developed sites were re-classified as urban greenspace, generally reflecting the presence of abandoned former industrial sites in the city. As would be expected, there was a notable movement of sites from the open land into developed and urban greenspace categories. Sites categorised as woodland tended to remain constant.

Ecosystem service provision varied across the study region (Table 2). Historical landcover offered a better explanation for the variation in five out of the eight ecosystem service measures than current landcover (Figs. 4 and 5). For example, R^2 for above-ground carbon density modelled against current landcover was 0.14 compared to 0.27 for data from the 1890s (Fig. 4a). AICc comparisons also indicated that landcover from the 1890s offered the best explanation for present-day variation in carbon density. Indeed, there was a strong signal from historical landcover as, with the exception of the 1850s, models from all historical periods offered a better explanation for current patterns of above-ground carbon storage than data from the 2000s. The strength and direction of the relationship between above-ground carbon density and the explanatory variables was similar across different time periods (again with the exception of the 1850s; Tables 4 and S1). Similarly, apart from the 1850s, above-ground carbon density was highest for landcovers classified as developed or woodland. For sites classified as urban greenspace in 1890, carbon density was negatively related to the proportion of urban land in the surrounding buffer (Table S1).

Across the three measures of wild species diversity (Figs. 4b–d), the influence of historical landcover was either absent or modest. The best explanation for current-day variation in bird density (Fig. 4b) and richness (Fig. 4c) was offered by historical landcover from the 1980s. However, the improvement in AICc (and increase in explanatory power) relative to 2000s was modest, and the strength and direction of the relationships with landcover variables was similar (Table S1). For bird density, landcover in all other time periods offered a substantially worse explanation of the data than either 2000s or 1980s data. Bird density was lower for sites classified as developed in the 1980s, but there was little difference in bird species richness between landcover categories (Table 4). Historical landcover was not related to present-day variation in plant richness any more strongly than 2000s landcover, with little difference in QAICc values across all time periods. Explanatory power peaked at 0.16 across all plant richness models (Table 4), with the parameter estimates for the landcover categories broadly similar across all time periods (Table S1).

There was substantial variation in the ability of current and historical landcover data to explain the number of recreational

Table 3
Survey sites ($N = 107$) classified according to landcover for each time period, and the median (interquartile range) percent coverage by urban development in a 100 m buffer surrounding each site. Across the entire currently urbanised area of the city 4.2% of the currently urbanised area of the city was classified as developed in 1850. This increased to 10.1, 14.9, 22.6, 31.6 and 41.5% for the 1890s, 1940s, 1960s, 1980s and 2000s, respectively (Díaz Porras, 2013).

Time period	Landcover				%Urban in 100 m buffer
	Developed	Urban greenspace	Woodland	Open land	
1850s	3 (2.8% of 107 sites)	5	15	84	0 (0–5)
1890s	14 (13.1%)	6	20	67	5 (0–20)
1940s	21 (19.6%)	13	19	54	10 (0–90)
1960s	23 (21.5%)	16	21	47	30 (5–95)
1980s	24 (22.4%)	21	26	36	40 (5–97.5)
2000s	34 (31.8%)	20	29	24	50 (5–100)

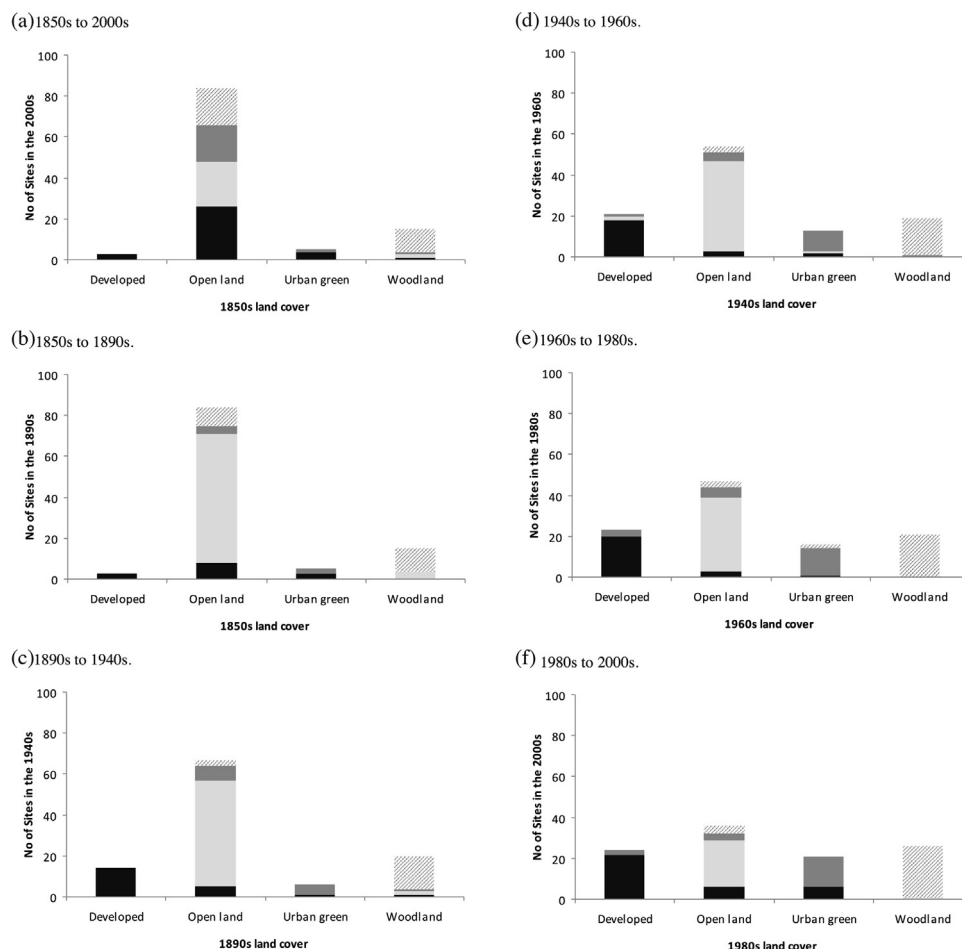


Fig. 3. The fate of sites classified as the four landcover categories between different time periods across 107 survey sites situated along the urban riparian corridors in Sheffield, UK, for between the: (a) 1850s and 2000s (the entire time period), (b) 1850s and 1890s, (c) 1890s and 1940s, (d) 1940s and 1960s, (e) 1960s and 1980s and (f) 1980s and 2000s. In each case, the figures show the number of sites falling into the four landcover categories in the earlier time period (listed on the x-axis) which are subsequently classified as developed (black), open land (light grey), urban greenspace (dark grey) and woodland (striped) in the later time period (listed on the y-axis).

users at our study sites (Fig. 5a). The landcover model with the lowest QAICc was from the 1940s, with pR^2 varying from 0.15 (1850s) and 0.17 (2000s), to 0.37 (1940s). With the exception of the 1850s data, models from all time periods offered a better explanation (lower QAICc and higher pR^2) of patterns of recreational use than the measures of landcover from the 2000s. Present-day usage was higher for sites that were classified as open, woodland or urban greenspace in the 1940s than those classified as developed. Usage increased with the proportion of land surrounding the site that was already urban in the 1940s (Table 4).

For one measure of the quality of the recreational experience (*continuity with past*), the best explanation for current-day variation was offered by landcover in the 1960s (Fig. 5), with the measure of well-being negatively related to the proportion of the surrounding 100 m classified as urban in that time period. The improvement in AICc relative to present-day landcover was substantial and was matched by an increase in R^2 . For the remaining two measures of well-being (*reflection* and *attachment*), the 2000s landcover models offered both the lowest AICc and highest R^2 (Fig. 5). For all well-being measures, the direction (positive/negative) of the relationship with landcover variables was similar across models (Table S1).

4. Discussion

The provision of ecosystem services varies temporally (Launtenbach et al., 2011; Jiang et al., 2013; Holland et al., 2011).

Nonetheless, the prevailing approach to their study has been to quantify provision based on analyses that generally ignore time-lags and feedbacks within and between social and ecological systems (Carpenter et al., 2009; Nicholson et al., 2009), using contemporary land-use proxies to map the spatial distribution of services (Seppelt et al., 2011). Yet we report relatively modest explanatory power for several relationships between current landcover and ecosystem service provision; something not uncommon in the literature (e.g., Eigenbrod et al., 2010). Our results demonstrate that, for some ecosystem services, past landcover is a better predictor of current provision than present landcover, and highlight the need to incorporate legacy effects into ecosystem service provision models.

For five out of the eight ecosystem services examined here, past landcover offered a better explanation for present-day variation in service provision when compared to current landcover (cf. Table 1). The strength and length of the time-lag varied according to the service; ranging from more than 100 years for above-ground carbon density, through 60 years for human visitor numbers, 40 years for *continuity with past* well-being, to 20 years for total bird density and richness. There was no evidence of a time-lag for the remaining measures of well-being (cf. Table 1). Any ecosystem is likely to be subject to a number of historical influences, which will be due to its inherent properties, as well as human-induced alterations (Holland et al., 2011). When examining multiple services, we should thus expect that each service will respond to landcover from different points in the past.

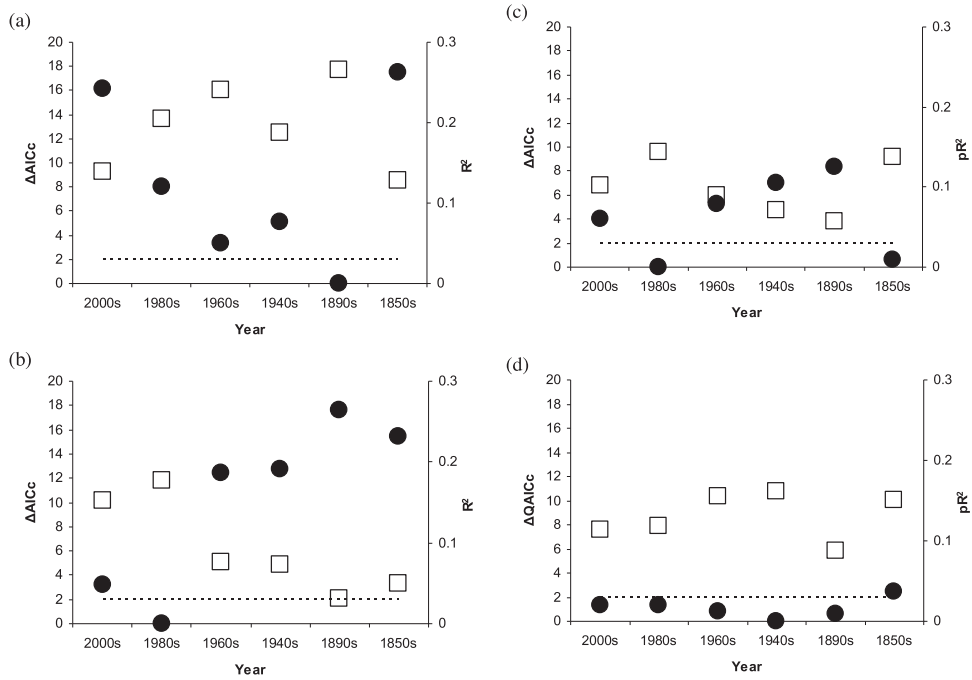


Fig. 4. $\Delta AICc$ (solid circles—lower $\Delta AICc$ represents more plausible models) and R^2 (open square—higher R^2 represents better explanatory power) for present-day ecosystem service measures across the riparian areas of Sheffield for: (a) carbon density; (b) bird density; (c) bird richness; and (d) plant richness. The horizontal dotted line indicates $\Delta AICc = 2$. Where multiple time periods offer plausible explanations for the data (difference in $AICc < 2$ between models), we took the conservative approach of considering the most recent model as offering the best explanation for present-day service provision (see Section 2.5; Table 4).

Within cities, many dimensions of vegetation are often better explained by past characteristics, with temporal lags often due to social and ecological changes happening at different rates (Luck et al., 2009; Troy et al., 2007). Although we find no conclusive

evidence that plant species richness is related to past landcover, two different biodiversity metrics, bird richness and density, were more strongly predicted by landcover from the 1980s than from the present-day, although the improvement in model fit was

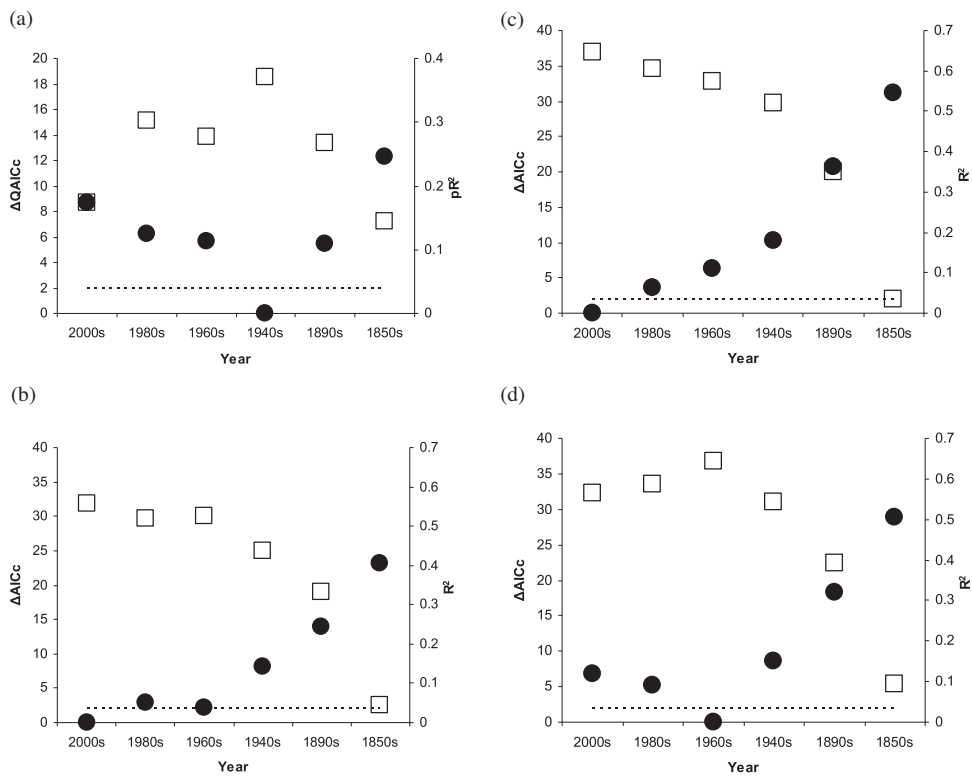


Fig. 5. $\Delta AICc$ (solid circles—lower $\Delta AICc$ represents more plausible models) and R^2 (open square—higher R^2 represents better explanatory power) for present-day ecosystem service measures across the riparian areas of Sheffield for: (a) number of recreational visitors; (b) reflection well-being; (c) attachment well-being; and (d) continuity with past well-being. The horizontal dotted line indicates $\Delta AICc = 2$. Where multiple time periods offered plausible explanations for the data (difference in $AICc < 2$ between models), we took the conservative approach of considering the most recent model as offering the best explanation for present-day service provision (see Section 2.5; Table 4).

Table 4
Year, explanatory power and parameter estimates (\pm standard error) for eight ecosystem services for the year of landcover data that offered the best explanation of present-day service provision. Where multiple time periods offered plausible explanations for the data (difference in AICc < 2 between models), we present parameter estimates for the most recent data. Interaction terms indicate b for %Urban cover for each site classification. For example, carbon density at a site classified as developed in the 1890s is predicted to be $2.659 - 0.033(\%Urban) \text{ kg m}^{-2}$, while for an urban greenspace site, carbon density = $0.843 + 0.015(\%Urban)$. Interaction terms were not calculated for the quality of experience measures. For full model outputs, including standard errors for parameter estimates, see Table S1.

	Carbon storage		Wild species diversity		Recreation		Quality of experience	
	Carbon density (kg m^{-2})	Bird density (birds ha^{-1})	Bird richness	Plant richness	Number of users	Reflection	Attachment	Continuity with past
Year	1890s	1980s	1980s	2000s	1940s	2000s	2000s	1960s
(p) R^2 ^a	0.27	0.18	0.14	0.11	0.37	0.56	0.65	0.64
(Q)AICc ^a	360.95	293.52	550.88	-47.66	-32.69	-2.04	-1.69	6.79
Developed	2.659 (± 1.027)	3.809 (± 1.696)	2.409 (± 0.590)	3.574 (± 0.513)	0.237 (± 3.162)	4.049 (± 0.163)	4.380 (± 0.164)	3.622 (± 0.157)
Open land	1.243 (± 0.187)	4.432 (± 0.182)	2.457 (± 0.060)	3.672 (± 0.084)	1.979 (± 0.466)	4.183 (± 0.062)	4.565 (± 0.063)	3.580 (± 0.080)
Urban greenspace	0.843 (± 0.990)	5.290 (± 0.445)	2.471 (± 0.144)	3.793 (± 0.188)	3.222 (± 0.758)	4.242 (± 0.134)	4.613 (± 0.134)	3.485 (± 0.092)
Woodland	2.189 (± 0.326)	4.596 (± 0.240)	2.404 (± 0.081)	3.568 (± 0.091)	3.761 (± 0.331)	4.170 (± 0.049)	4.549 (± 0.049)	3.573 (± 0.085)
%Urban	-	-	-	-	-	-0.004 (± 0.002)	-0.005 (± 0.002)	-0.007 (± 0.001)
Interaction %Urban:Developed land	-0.033 (± 0.012)	0.002 (± 0.018)	-0.002 (± 0.006)	0.003 (± 0.005)	0.035 (± 0.033)	-	-	-
Interaction %Urban:Open land	-0.016 (± 0.009)	0.012 (± 0.005)	0.002 (± 0.001)	0.004 (± 0.004)	0.000 (± 0.010)	-	-	-
Interaction %Urban:Urban greenspace	0.015 (± 0.022)	-0.004 (± 0.006)	-0.001 (± 0.002)	0.000 (± 0.002)	0.001 (± 0.010)	-	-	-
Interaction %Urban:Woodland	0.019 (± 0.046)	0.000 (± 0.009)	-0.002 (± 0.003)	0.004 (± 0.003)	0.025 (± 0.011)	-	-	-
Error structure	Normal	Normal	Poisson	QuasiPoisson	QuasiPoisson	Normal	Normal	Normal
Time periods offering equally plausible explanations	None	None	1850s	1940s, 1960s, 1980s	None	None	None	None

^a For count data (bird richness, plant richness, number of users), pR^2 was calculated as the proportion of deviance explained (Crawley, 2007); quasiAICc (QAICc) values were calculated for overdispersed models.

relatively modest. Given that the above-ground carbon stocks associated with vegetation are closely correlated with biomass density (i.e., tree size and number) they are more likely to be related to historical management and landcover than present-day conditions (Foster et al., 2003; Gough et al., 2007). For Sheffield's riparian greenspaces, above-ground carbon stocks were strongly predicted by landcover over 100 years ago. However, regardless of which time period we examined between the 1890s and the 2000s, the strength and direction of relationships between above-ground carbon stocks and landcover were similar.

People have multiple motivations for visiting urban greenspaces, which are not limited to the prosaic (e.g., proximity, convenience) and can include reasons that are related to the physical features of, as well as the emotional qualities associated with, the site (Dallimer et al., 2014; Irvine et al., 2013). It is therefore plausible that landcover could influence both the number of visitors a greenspace receives, and the well-being that visitors gain whilst there. Here we found that the number of recreational visitors to riparian greenspaces in Sheffield was more strongly determined by landcover over 60 years previously than by present-day conditions. Likewise, an approximate 40 year temporal lag influenced current levels of one metric of well-being (*continuity with the past*) associated with recreational experience. These time-lags suggest that land-use decisions made many decades ago can have long-lasting implications for the human population, reinforcing the importance of retaining and enhancing existing green infrastructure within cities, rather than simply creating new sites. This is particularly germane given that providing urban greenspaces, in order to encourage their use and thereby deliver individual and societal benefits, is an important policy objective (EEA, 2009).

When seeking to understand the relationship between people and the environment, our findings highlight the importance of considering the dynamic interplay among the spatial and temporal aspects of the biophysical alongside the cognitive and emotional processes of individuals (e.g., di Castri et al., 1981; Ittelson et al., 1974; Kaplan and Kaplan, 1989). What is intriguing is that this relationship is evident even though most of our participants are unlikely to have experienced the study locations directly between 40 to 60 years ago. The results therefore point to the need to consider the socio-cultural context within which the person-environment interaction occurs (e.g., Bonnes and Secchiarioli, 1995; Nassauer, 1997) and the potential influence of long-standing features (e.g., the presence of older trees (O'Brien, 2004) and/or historical built infrastructure (Hull et al., 1994). For example, it may be that well-established greenspaces are more widely valued by city residents, who are therefore more likely to visit them. In the context of our case study city, the relatively recent focus on the preservation and appreciation of cultural heritage (e.g., Griffiths, 1999; Kendall, 2005) may encourage residents (either consciously or subconsciously) to visit more established greenspaces. Indeed, heritage preservation is undertaken for a range of reasons, but can include enhancing a location's character, identity or sense of place (Hull et al., 1994) thus maximising its contribution to the creation of a liveable community (e.g., Timothy and Nyaupane, 2009). Given the theoretical grounding of our *continuity with past* metric in the sense of place literature (Table 1) (e.g., Proshansky et al., 1983), the effect of past land-use on present-day well-being is what might be expected.

Although we found no legacy effect of historical landcover for the two other aspects of well-being (*reflection* and *attachment*), landcover was a strong predictor of all three measures of psychological wellbeing. Both theory (Kaplan and Kaplan, 1989; Kaplan, 1993, 1995) and empirical research (e.g., Gulwadi, 2006; Herzog et al., 1997; Staats et al., 2003; Talbot and Kaplan, 1984) support the idea that the natural environment can facilitate thinking through, and reflecting on, issues and suggest that people actively select certain types of settings for such purposes that are often of a more natural configuration. It may be, therefore, that landcover acts as an

objective measure of some environmental qualities that the people involved in our study were seeking. Within the sense of place literature, the connection or bond between a person and a specific place, as measured by our *attachment* metric, has been widely discussed (for reviews see Altman and Low, 1992; Manzo, 2003; Scannell and Gifford, 2010). Perhaps most relevant in relation to this study is the emphasis on direct experience (e.g., Manzo, 2005 “experience-in-place” concept) and the physical characteristics of the place itself (e.g., Stedman, 2003).

5. Conclusions

The concept that historical information can help in understanding the present-day properties of ecosystems is increasingly being recognised. Nevertheless, land-use time-lags do not routinely feature in predicting ecosystem service provision. This is a potential weakness of their application. Indeed, some historians argue that historical elements are fundamental to conservation biology and that the discipline will continue to be incomplete if history is neglected (Meine, 1999; Newell et al., 2005; Szabo, 2010). No field-derived measures of ecosystem services provision contemporary with the historical maps were available, precluding us from undertaking any form of time series analyses and therefore addressing issues of causality. Nevertheless, we have demonstrated possible links between past landcover (covering periods 20 to 100 years ago) and the present provision of some ecosystem services. This emphasises that historical dimension to biodiversity and ecosystem services management is essential, especially in fast-changing urban ecosystems. Examining other metrics of landcover at different spatial scales (cf., Dallimer et al., 2010) and/or landcover change trajectory and stability (e.g., Watson et al., 2014) offer informative avenues for future research. Although our findings add to the complexity of predicting how ecosystem service delivery may respond to scenarios of LU/LC change, ecologists may find that they can link the biodiversity conservation agenda to the preservation of cultural heritage, and that certain courses of action provide win–win outcomes across multiple environmental and cultural goods.

Acknowledgements

We thank the people of Sheffield who took part in the study. Andrew Skinner, James Rouquette, Gryphon Felski, Joseph Moore, Grant Bramall, Chris Duffy and Ruth Hallam helped in the field. Maps were obtained from Digimap Ordnance Survey and Historical Collections (www.edina.ac.uk). K.N.I. was supported by the Scottish Government’s Rural and Environmental Science and Analytical Services Division (RESAS). Research was supported by the UK government’s EPSRC (grant EP/F007388/1 to the URSULA consortium) and an EU-FP7 Marie Curie Fellowship (grant 273547) to M.D.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at [doi:10.1016/j.gloenvcha.2015.01.015](https://doi.org/10.1016/j.gloenvcha.2015.01.015).

References

- Abercrombie, P., 1924. *Sheffield: A Civic Survey and Suggestions Towards a Development Plan*. University of Liverpool, Liverpool.
- Altman, I., Low, S.M., 1992. *Place Attachment: Human Behavior and Environment*. Plenum Press, New York.
- Armsworth, P.R., Cantú-Salazar, L., Parnell, M., Booth, J.E., Stoneman, R., Davies, Z.G., 2013. Opportunities for cost-sharing in conservation: variation in volunteering effort across protected areas. *PLoS ONE* 8, e55395.
- Ashworth, G., 2008. In search of the Place-identity Dividend: Using Heritage Landscapes to Create Place Identity. In: Eyles, J., Williams, A. (Eds.), *Sense of Place, Health and Quality of Life*. Ashgate books. <http://www.ashgate.com/isbn/9780754673323>.
- Austin, M.P., Heyligers, P.C., 1989. Vegetation survey design for conservation: Gradient sampling of forests in north-eastern New South Wales. *Biol. Conserv.* 50, 13.
- Balvanera, P., Pfisterer, A.B., Buchmann, N., He, J.S., Nakashizuka, T., Raffaelli, D., Schmid, B., 2006. Quantifying the evidence for biodiversity effects on ecosystem functioning and services. *Ecol. Lett.* 9, 1146–1156.
- Barau, A., Ludin, A.N.M., Said, I., 2013. Socio-ecological systems and biodiversity conservation in African city: insights from Kano Emir’s palace gardens. *Urban Ecosyst.* 16, 783–800.
- Berman, M.G., Jonides, J., Kaplan, S., 2008. The cognitive benefits of interacting with nature. *Psychol. Sci.* 19, 1207–1212.
- Bonnes, M., Secchiarioli, G., 1995. *Environmental Psychology: A Psycho-social Introduction*. Sage Publications, London.
- Bowler, D.E., Buyung-Ali, L.M., Knight, T.M., Pullin, A.S., 2010. A systematic review of evidence for the added benefits to health of exposure to natural environments. *BMC Public Health* 10, 456.
- Burnham, K.P., Anderson, D.R., 2002. *Model Selection and Multimodel Inference: A Practical Information-theoretic Approach*. Springer, New York.
- Carpenter, S.R., Mooney, H.A., Agard, J., Capistrano, D., Defries, R.S., Diaz, S., Dietz, T., Duraipappah, A.K., Oteng-Yeboah, A., Pereira, H.M., Perrings, C., Reid, W.V., Sarukhan, J., Scholes, R.J., Whyte, A., 2009. Science for managing ecosystem services: beyond the millennium ecosystem assessment. *Proc. Natl. Acad. Sci. U. S. A.* 106, 1305–1312.
- Chan, K.M.A., Shaw, M.R., Cameron, D.R., Underwood, E.C., Daily, G.C., 2006. Conservation planning for ecosystem services. *PLoS Biol.* 4, 2138–2152.
- Crawley, M.J., 2007. *The R book*. Wiley, Oxford.
- Crossley, D., Cass, J., 1989. *Water Power on the Sheffield Rivers*. Sheffield Trades Historical Association and University of Sheffield, Sheffield.
- Dallimer, M., Davies, Z., Irvine, K., Maltby, L., Warren, P., Gaston, K., Armsworth, P., 2014. What personal and environmental factors determine frequency of urban greenspace use? *Int. J. Environ. Res. Public Health* 11, 7977–7992.
- Dallimer, M., Irvine, K.N., Skinner, A.M.J., Davies, Z.G., Rouquette, J.R., Maltby, L.L., Warren, P.H., Armsworth, P.R., Gaston, K.J., 2012a. Biodiversity and the feel-good factor: understanding associations between self-reported human well-being and species richness. *Bioscience* 62, 47–55.
- Dallimer, M., Marini, L., Skinner, A.M.J., Hanley, N., Armsworth, P.R., Gaston, K.J., 2010. Agricultural land-use in the surrounding landscape affects moorland bird diversity. *Agric. Ecosyst. Environ.* 139, 578–583.
- Dallimer, M., Rouquette, J.R., Skinner, A.M.J., Armsworth, P.R., Maltby, L.M., Warren, P.H., Gaston, K.J., 2012b. Contrasting patterns in species richness of birds, butterflies and plants along riparian corridors in an urban landscape. *Divers. Distrib.* 18, 742–753.
- Dallimer, M., Tang, Z.Y., Bibby, P.R., Brindley, P., Gaston, K.J., Davies, Z.G., 2011. Temporal changes in greenspace in a highly urbanized region. *Biol. Lett.* 7, 763–766.
- Davies, Z.G., Edmondson, J.L., Heinemeyer, A., Leake, J.R., Gaston, K.J., 2011. Mapping an urban ecosystem service: quantifying above-ground carbon storage at a city-wide scale. *J. Appl. Ecol.* 48, 1125–1134.
- Dearing, J.A., Yang, X.D., Dong, X.H., Zhang, E.L., Chen, X., Langdon, P.G., Zhang, K., Zhang, W.G., Dawson, T.P., 2012. Extending the timescale and range of ecosystem services through paleoenvironmental analyses, exemplified in the lower Yangtze basin. *Proc. Natl. Acad. Sci. U. S. A.* 109, E1111–E1120.
- di Castri, F., Hadley, M., Damalian, J., 1981. MAB—The Man and Biosphere Program as an evolving system. *Ambio* 10, 52–57.
- Diaz Porras, D.F., 2013. *Historical Urban Ecology, Green Spaces, and Biodiversity*. Department of Animal and Plant Sciences, University of Sheffield, Sheffield.
- EEA, 2009. *Ensuring Quality of Life in Europe’s Cities and Towns*. European Environment Agency, Copenhagen.
- 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. *J. Appl. Ecol.* 47, 377–385.
- Eriksson, O., 1996. Regional dynamics of plants: a review of evidence for remnant, source-sink and metapopulations. *Oikos* 77, 248–258.
- Firth, C., 1997. *Domesday to the Dawn of the New Millennium: 900 Years of the Don Fishery*. Environment Agency, Bristol.
- Ford, H.A., Walters, J.R., Cooper, C.B., Debus, S.J.S., Doerr, V.A.J., 2009. Extinction debt or habitat change?—ongoing losses of woodland birds in north-eastern New South Wales, Australia. *Biol. Conserv.* 142, 3182–3190.
- Foster, D., Swanson, F., Aber, J., Burke, I., Brokaw, N., Tilman, D., Knapp, A., 2003. The importance of land-use legacies to ecology and conservation. *Bioscience* 53, 77–88.
- Goddard, M.A., Dougill, A.J., Benton, T.G., 2010. Scaling up from gardens: biodiversity conservation in urban environments. *Trends Ecol. Evol.* 25, 90–98.
- Gomez-Baggethun, E., Barton, D.N., 2013. Classifying and valuing ecosystem services for urban planning. *Ecol. Econ.* 86, 235–245.
- Gough, C.M., Vogel, C.S., Harrold, K.H., George, K., Curtis, P.S., 2007. The legacy of harvest and fire on ecosystem carbon storage in a north temperate forest. *Global Change Biol.* 13, 1935–1949.
- Griffiths, P., 1999. *The Five Weirs Walk*. Hallamshire Publications Ltd, Sheffield.
- Gulwadi, G.B., 2006. Seeking restorative experiences—elementary school teachers’ choices for places that enable coping with stress. *Environ. Behav.* 38, 503–520.
- Güneralp, B., McDonald, R., Fragkias, M., Goodness, J., Marcotullio, P., Seto, K., 2013. Urbanization forecasts, effects on land use, biodiversity, and ecosystem services. In: Elmqvist, T., Fragkias, M., Goodness, J., Güneralp, B., Marcotullio, P.J., McDonald, R.I., Parnell, S., Schewenius, M., Sendstad, M., Seto, K.C., Wilkinson, C. (Eds.), *Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities*. Springer, Netherlands, pp. 437–452.
- Gustavsson, E., Lennartsson, T., Emanuelsson, M., 2007. Land use more than 200 years ago explains current grassland plant diversity in a Swedish agricultural landscape. *Biol. Conserv.* 138, 47–59.

- Herzog, T.R., Black, A.M., Fountaine, K.A., Knotts, D.J., 1997. Reflection and attentional recovery as distinctive benefits of restorative environments. *J. Environ. Psychol.* 17, 165–170.
- Hey, D., 2005. *A History of Sheffield*. Carnegie Publishing Ltd, Lancaster.
- Holland, R.A., Eigenbrod, F., Armsworth, P.R., Anderson, B.J., Thomas, C.D., Gaston, K.J., 2011. The influence of temporal variation on relationships between ecosystem services. *Biodivers. Conserv.* 20, 3285–3294.
- Hope, D., Gries, C., Zhu, W.X., Fagan, W.F., Redman, C.L., Grimm, N.B., Nelson, A.L., Martin, C., Kinzig, A., 2003. Socioeconomics drive urban plant diversity. *Proc. Natl. Acad. Sci. U. S. A.* 100, 8788–8792.
- Hull, R.B., Lam, M., Vigo, G., 1994. Place identity—symbols of self in the urban fabric. *Landscape Urban Plann.* 28, 109–120.
- Irvine, K.N., Warber, S.L., Devine-Wright, P., Gaston, K.J., 2013. Understanding urban green space as a health resource: a qualitative comparison of visit motivation and derived effects among park users in Sheffield, UK. *Int. J. Environ. Res. Public Health* 10, 417–442.
- Ittelson, W.H., Rivlin, A., Winkel, G., 1974. *An Introduction to Environmental Psychology*. Holt, Rinehart and Winston, New York.
- Jenks, M., Jones, C., 2010. *Dimensions of the Sustainable City*. Springer, New York.
- Jiang, M., Bullock, J.M., Hooftman, D.A.P., 2013. Mapping ecosystem service and biodiversity changes over 70 years in a rural English county. *J. Appl. Ecol.* 50, 841–850.
- Kamp, J., Urazaliev, R., Donald, P.F., Holz, N., 2011. Post-Soviet agricultural change predicts future declines after recent recovery in Eurasian steppe bird populations. *Biol. Conserv.* 144, 2607–2614.
- Kaplan, R., Kaplan, S., 1989. *The Experience of Nature: a Psychological Perspective*. Cambridge University Press, New York.
- Kaplan, S., 1993. The role of natural environment aesthetics in the restorative experience. In: Gobster, P.H. (Ed.), *Managing Urban and High-use Recreation Settings*, USDA Forest Service, St. Paul, MN, pp. 46–49, General Technical Report NC-163.
- Kaplan, S., 1995. The restorative benefits of nature—toward an integrative framework. *J. Environ. Psychol.* 15, 169–182.
- Kareiva, P., Tallis, H., Ricketts, T.H., Daily, G.C., Polasky, S., 2011. *Natural Capital: Theory and Practice of Mapping Ecosystem Services*. Oxford University Press, Oxford.
- Kendall, K., 2005. *Walking the Rivelin*. Rivelin Valley Conservation Group, Sheffield.
- Keniger, L.E., Gaston, K.J., Irvine, K.N., Fuller, R.A., 2013. What are the Benefits of Interacting with Nature? *Int. J. Environ. Res. Public Health* 10, 913–935.
- Kuo, F.E., Sullivan, W.C., 2001. Environment and crime in the inner city - Does vegetation reduce crime? *Environ. Behav.* 33, 343–367.
- Kuusaaari, M., Bommarco, R., Heikkinen, R.K., Helm, A., Krauss, J., Lindborg, R., Ockinger, E., Partel, M., Pino, J., Roda, F., Stefanescu, C., Teder, T., Zobel, M., Steffan-Dewenter, I., 2009. Extinction debt: a challenge for biodiversity conservation. *Trends Ecol. Evol.* 24, 564–571.
- Lautenbach, S., Kugel, C., Lausch, A., Seppelt, R., 2011. Analysis of historic changes in regional ecosystem service provisioning using land use data. *Ecol. Indic.* 11, 676–687.
- Liu, J., Daily, G.C., Ehrlich, P.R., Luck, G.W., 2003. Effects of household dynamics on resource consumption and biodiversity. *Nature* 421, 530–533.
- Luck, G.W., Smallbone, L.T., O'Brien, R., 2009. Socio-economics and vegetation change in urban ecosystems: patterns in space and time. *Ecosystems* 12, 604–620.
- Mace, G.M., Norris, K., Fitter, A.H., 2012. Biodiversity and ecosystem services: a multilayered relationship. *Trends Ecol. Evol.* 27, 19–26.
- Manes, F., Incerti, G., Salvatori, E., Vitale, M., Ricotta, C., Costanza, R., 2012. Urban ecosystem services: tree diversity and stability of tropospheric ozone removal. *Ecol. Appl.* 22, 349–360.
- Manzo, L.C., 2003. Beyond house and haven: toward a revisioning of emotional relationships with places. *J. Environ. Psychol.* 23, 47–61.
- Manzo, L.C., 2005. For better or worse: exploring multiple dimensions of place meaning. *J. Environ. Psychol.* 25, 67–86.
- McDonald, R.L., Forman, R.T.T., Kareiva, P., 2010. Open space loss and land inequality in United States cities, 1990–2000. *PLoS ONE* 5.
- MEA, 2005. *Ecosystems and Human Well-being: Biodiversity Synthesis*. World Resources Institute, Washington, DC.
- Meine, C., 1999. It's about time: conservation biology and history. *Conserv. Biol.* 13, 1–3.
- Mitchell, R., Popham, F., 2007. Greenspace, urbanity and health: relationships in England. *J. Epidemiol. Community Health* 61, 681–683.
- Myint, S.W., Wentz, E.A., Brazel, A.J., Quattrochi, D.A., 2013. The impact of distinct anthropogenic and vegetation features on urban warming. *Landscape Ecol.* 28, 959–978.
- 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. *Proc. Natl. Acad. Sci. U. S. A.* 105, 9495–9500.
- Nassauer, J.I., 1997. Cultural sustainability: aligning aesthetics and ecology. In: Nassauer, J.I. (Ed.), *Placing Nature: Culture and Landscape Ecology*. Island Press, Washington, DC, pp. 65–83.
- Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, D.R., Chan, K.M.A., Daily, G.C., Goldstein, J., Kareiva, P.M., Lonsdorf, E., Naidoo, R., Ricketts, T.H., Shaw, M.R., 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. *Front. Ecol. Environ.* 7, 4–11.
- Newell, B., Crumley, C.L., Hassan, N., Lambin, E.F., Pahl-Wostl, C., Underdal, A., Wasson, R., 2005. A conceptual template for integrative human-environment research. *Glob. Environ. Change-Hum. Policy Dimens.* 15, 299–307.
- Nicholson, E., Mace, G.M., Armsworth, P.R., Atkinson, G., Buckle, S., Clements, T., Ewers, R.M., Fa, J.E., Gardner, T.A., Gibbons, J., Grenyer, R., Metcalfe, R., Mourato, S., Muuls, M., Osborn, D., Reuman, D.C., Watson, C., Milner-Gulland, E.J., 2009. Priority research areas for ecosystem services in a changing world. *J. Appl. Ecol.* 46, 1139–1144.
- Nowak, D.J., Greenfield, E.J., 2012. Tree and impervious cover change in US cities. *Urban Forest. Urban Green.* 11, 21–30.
- O'Brien, L., 2004. *A Sort of Magical Place: People's Experiences of Woodlands in Northwest and Southeast England*. Forest Research, Edinburgh.
- ODPM, 2003. *Living Places: Cleaner, Safer, Greener*. Stationery Office, London.
- ONS, 2012a. *2011 Census—Population and Household Estimates for England and Wales, March 2011*. Office for National Statistics, London.
- ONS, 2012b. *Families and Households 2001 to 2011*. Office for National Statistics, London.
- Park, M., Hagishima, A., Tanimoto, J., Narita, K., 2012. Effect of urban vegetation on outdoor thermal environment: field measurement at a scale model site. *Build. Environ.* 56, 38–46.
- Pärtel, M., Mandla, R., Zobel, M., 1999. Landscape history of a calcareous (alvar) grassland in Hanila, western Estonia, during the last three hundred years. *Landscape Ecol.* 14, 187–196.
- Pickett, S.T.A., Cadenasso, M.L., Grove, J.M., Groffman, P.M., Band, L.E., Boone, C.G., Burch, W.R., Grimmond, C.S.B., Hom, J., Jenkins, J.C., Law, N.L., Nilon, C.H., Pouyat, R.V., Szlavecz, K., Warren, P.S., Wilson, M.A., 2008. Beyond urban legends: an emerging framework of urban ecology, as illustrated by the Baltimore Ecosystem Study. *Bioscience* 58, 139–150.
- Proshansky, H.M., Fabian, A.K., Kaminoff, R., 1983. Place-identity: physical world socialization of the self. *J. Environ. Psychol.* 3, 57–83.
- Pugh, T.A.M., MacKenzie, A.R., Whyatt, J.D., Hewitt, C.N., 2012. Effectiveness of green infrastructure for improvement of air quality in urban street canyons. *Environ. Sci. Technol.* 46, 7692–7699.
- Raudsepp-Hearne, C., Peterson, G.D., Bennett, E.M., 2010a. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. *Proc. Natl. Acad. Sci. U. S. A.* 107, 5242–5247.
- Raudsepp-Hearne, C., Peterson, G.D., Tengö, M., Bennett, E.M., Holland, T., Benessaiah, K., MacDonald, G.K., Pfeifer, L., 2010b. Untangling the environmentalist's paradox: why is human well-being increasing as ecosystem services degrade? *Bioscience* 60, 576–589.
- Rouquette, J.R., Dallimer, M., Armsworth, P.R., Gaston, K.J., Maltby, L., Warren, P.H., 2013. Species turnover and geographic distance in an urban river network. *Divers. Distrib.* 19, 1429–1439.
- Scannell, L., Gifford, R., 2010. Defining place attachment: a tripartite organizing framework. *J. Environ. Psychol.* 30, 1–10.
- Seppelt, R., Dormann, C.F., Eppink, F.V., Lautenbach, S., Schmidt, S., 2011. A quantitative review of ecosystem service studies: approaches, shortcomings and the road ahead. *J. Appl. Ecol.* 48, 630–636.
- Seto, K.C., Güneralp, B., Hutyra, L.R., 2012. Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. *Proc. Natl. Acad. Sci. U. S. A.* 109, 16083–16088.
- Staats, H., Kieviet, A., Hartig, T., 2003. Where to recover from attentional fatigue: an expectancy-value analysis of environmental preference. *J. Environ. Psychol.* 23, 147–157.
- Stedman, R.C., 2003. Is it really just a social construction? The contribution of the physical environment to sense of place. *Soc. Nat. Resour.* 16, 671–685.
- Sullivan, W.C., Kuo, F.E., DePooter, S.F., 2004. The fruit of urban nature - Vital neighborhood spaces. *Environ. Behav.* 36, 678–700.
- Susca, T., Gaffin, S.R., Dell'Osso, G.R., 2011. Positive effects of vegetation: urban heat island and green roofs. *Environ. Pollut.* 159, 2119–2126.
- Szabo, P., 2010. Why history matters in ecology: an interdisciplinary perspective. *Environ. Conserv.* 37, 380–387.
- Szabo, P., Hedl, R., 2011. Advancing the integration of history and ecology for conservation. *Conserv. Biol.* 25, 680–687.
- Talbot, J.F., Kaplan, R., 1984. Needs and fears: the response to trees and nature in the inner city. *J. Arboricult.* 10, 222–228.
- Thomas, L., Buckland, S.T., Rexstad, E.A., Laake, J.L., Strindberg, S., Hedley, S.L., Bishop, J.R.B., Marques, T.A., Burnham, K.P., 2010. Distance software: design and analysis of distance sampling surveys for estimating population size. *J. Appl. Ecol.* 47, 5–14.
- Tianhong, L., Wenkai, L., Zhenghan, Q., 2010. Variations in ecosystem service value in response to land use changes in Shenzhen. *Ecol. Econ.* 69, 1427–1435.
- Tilman, D., May, R.M., Lehman, C.L., Nowak, M.A., 1994. Habitat destruction and the extinction debt. *Nature* 371, 65–66.
- Timothy, D.J., Nyaupane, G.P., 2009. *Cultural Heritage and Tourism in the Developing World: A Regional Perspective*. Routledge.
- Troy, A.R., Grove, J.M., O'Neil-Dunne, J.P.M., Pickett, S.T.A., Cadenasso, M.L., 2007. Predicting opportunities for greening and patterns of vegetation on private urban lands. *Environ. Manage.* 40, 394–412.
- Tzoulas, K., Korpela, K., Venn, S., Yli-Pelkonen, V., Kazmierczak, A., Niemela, J., James, P., 2007. Promoting ecosystem and human health in urban areas using Green Infrastructure: a literature review. *Landscape Urban Plann.* 81, 167–178.
- UKNEA, 2011. *The UK National Ecosystem Assessment: Technical Report*. UNEP-WCMC, Cambridge.
- United Nations, 2013. *World Urbanization Prospects, the 2012 Revision*. United Nations, Department of Economic and Social Affairs, Population Division, New York.
- Ward-Thompson, C., Roe, J., Aspinall, P., Mitchell, R., Clow, A., Miller, D., 2012. More green space is linked to less stress in deprived communities: Evidence from salivary cortisol patterns. *Landscape Urban Plann.* 105, 221–229.
- Watson, S.J., Luck, G.W., Spooner, P.G., Watson, D.M., 2014. Land-use change: incorporating the frequency, sequence, time span, and magnitude of changes into ecological research. *Front. Ecol. Environ.* 12, 241–249.
- Zhao, S.Q., Da, L.J., Tang, Z.Y., Fang, H.J., Song, K., Fang, J.Y., 2006. Ecological consequences of rapid urban expansion: Shanghai, China. *Front. Ecol. Environ.* 4, 341–346.