Modelling nature-based recreation to inform land management

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Contents

Abstract

Countryside recreation is hugely popular and demand is on the rise. Whilst participation should be encouraged, sensitive management is required to reduce associated environmental impacts. This thesis investigates current and future patterns in countryside recreation at multiple spatial scales, from national to site, to explore the potential impacts on biodiversity and enhance the evidence base for conservation interventions.

A national-level recreation model is developed from a unique and massive data set of georeferenced recreational visits collected over 3 years, which predicts the probability of visitation as a function of land cover composition and accessibility to and within a site, whilst controlling for source population and socio-demographic differences. Land cover types were subdivided into proportion designated and non-designated for high nature value, using Sites of Special Scientific Interest (SSSI) as a proxy. Probability of visitation to preferred land covers, coast and freshwater, decreased when SSSI designated, with no effect for broadleaved woodland. Therefore general recreational use by the public did not represent an important ecosystem service of protected high-nature-value areas. The model was employed to create national- and county-level spatially-explicit predictions of countryside recreation under present and future conditions, the conservation implications of which are discussed.

As species conservation requires knowledge of how recreational pressure is distributed throughout a site, a novel methodology was developed using Thetford Forest as a case study. GIS-based Network Analysis was combined with statistical modelling to predict the number of disturbance events from recreationists for all path sections throughout the site. This tool was able to test the consequences of altering site access on the number of hypothetical new woodlark territories likely to become occupied.

This study contributes to a relatively small body of work on the importance of biodiversity for recreation and provides novel spatial approaches for quantifying demand and testing conservation interventions.

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Chapter 1

Introduction

1.1 The societal role of outdoor recreation

Demand for outdoor recreation is high and ever growing, and with recognised societal benefits for human health and well-being as well as local economies this is to be encouraged; but recreational use of the countryside brings with it environmental impacts, requiring sensitive management based on sound scientific understanding.

Around the world, public perception of nature and landscapes has changed in recent decades from a functional to a more hedonistic image (Buijs, Pedroli & Luginbühl 2006) and global participation in outdoor recreation is on the increase (Jensen & Koch 2004; Balmford *et al.* 2009; Natural England 2011). The scientific study of outdoor recreation has likewise grown, now a large and ever expanding research area, spanning several disciplines from the social sciences, to health, economics and environmental sciences. A large focus has been on the psychological, physiological and social benefits of interacting with nature (Tzoulas *et al.* 2007; Keniger *et al.* 2013); for example recreating in a natural setting can increase self-esteem and enhance mood (Pretty *et al.* 2007). Environmental scientists have found that participation in outdoor recreation can increase pro-environmental behaviour (Dunlap & Heffernan 1975; Theodori, Luloff & Willits 1998; Ardoin *et al.* 2015), and advocate that it gives natural areas amenity value and generates revenues for conservation (Bell *et al.* 2007; Balmford *et al.* 2009).

Such scientific findings filtered through to governments and not-for-profits, who developed strategies and campaigns to engage people in outdoor activities (e.g. Natural England's Access and Engagement strategy (Natural England 2012), the Britain on Foot campaign (www.britainonfoot.co.uk) and the Wildlife Trusts '30 Days Wild' (Richardson *et al.* 2016). Despite campaigns and government access recommendations, however, there is still inequality in access to recreational opportunities; the potential spatial access to neighbourhood parks differs for U.S. residents depending on the state in which they live (Zhang, Lu & Holt 2011), the availability of green spaces within different distance bands from home differs between four Flemish cities (Van Herzele & Wiedemann 2003), approximately half of Seoul residents do not have access to a neighbourhood park (Oh & Jeong 2007), and only 36.5% of households in Sheffield, England, have a green space within 300 m (a government recommended standard) (Barbosa *et al.* 2007). Accessibility is also uneven across socio-economic (Barbosa *et al.* 2007) and ethnic groups (Comber, Brunsdon & Green 2008). Disparities also exist in the uptake of recreational opportunities by different socio-demographic groups; ethnic minorities, young adults (aged 16-25) and less affluent members of society were underrepresented in a sample of visitors (1,095 people) to protected areas in a region of England (Booth, Gaston & Armsworth 2010), and ethnic minorities in Denmark were less likely to regularly visit green spaces than people from a western background (Schipperijn *et al.* 2010). Lack of provision of recreational opportunities and the imbalance in the use of available areas is detrimental to overall public health and well-being, as some sectors of society are not receiving the physical and mental benefits of outdoor recreation.

Research into outdoor recreation covers a wide range of settings (Fig. 1.1) and activities. This thesis focuses on natural or semi-natural spaces (‘nature-based recreation’), with the broad purpose of understanding patterns in recreational use to inform conservation and land management. Specifically, this thesis examines the drivers of recreational demand and the spatial distribution of recreational demand at multiple scales (country, to county to individual site) from a biodiversity conservation perspective, using the UK as a case study. The rest of this chapter summarises the current body of research in these areas, providing the context for the empirical chapters that follow.

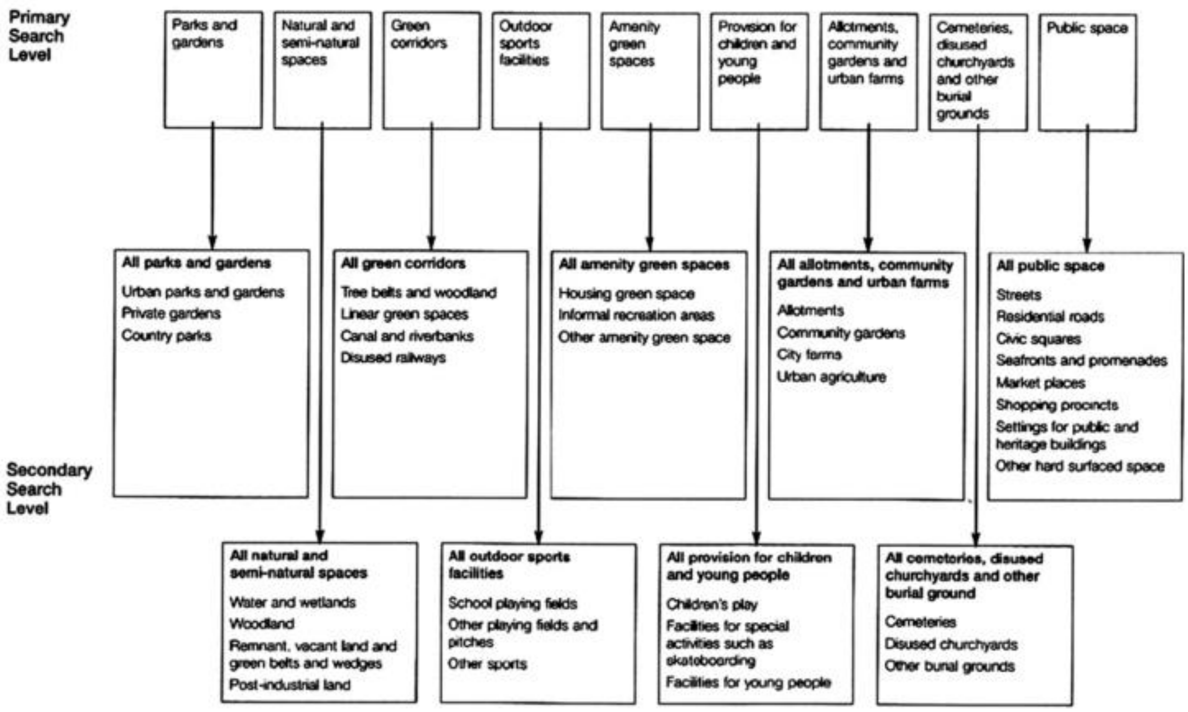

Fig. 1.1 Typology of green and public spaces developed by Bell, Montarzino, et al. (2007)

1.2 A brief history of outdoor recreation in the UK

> *"Man does not live by bread alone. Some men at least need the beauty of nature, even the wilderness of nature, the contact with animals leading their own lives, in their own surroundings."*
>
> J.S. Huxley, chairman of 'Wildlife Conservation Special Committee' 1943-1948

In the 19th century, outdoor recreation in the UK countryside was an upper class pastime, predominantly aristocrats hunting game on their private property (Glytis 1991). James Bryce first attempted to open up the countryside to the general public in 1884 through the Access to the Mountains (Scotland) Bill (Butler 1985), a private member's bill which failed in 1884, 1888 and 1892 (Bathe 2007). In the early 20th century more than four-fifths of the population in England lived in urban areas, with poor living conditions for many and high density terraced housing leaving little space for recreation areas (Glytis 1991). Demand for recreational access started to increase from the working class' desire to escape from town, facilitated by increased ownership of bicycles and the legal right to paid holidays (Glytis 1991). Many outdoor clubs became established and radical groups such as the Ramblers' Rights Movement began lobbying for the public's right to roam (Glytis 1991). Following protests and mass trespasses in the 1920s and early 1930s, the most famous of which was the mass trespass at Kinder Scout in the Peak District in 1932 (Bathe

2007), the Rights of Way Act 1932 was passed which provided for public access to paths in use for at least 20 continuous years (Ramblers 2013). Further progress came in the form of the National Parks and Access to the Countryside Act 1949, which focused on rights of way as well as conserving important areas of the countryside (Seabrooke & Miles 1993). The Act designated ten national parks and established the National Parks Commission to “ensure the preservation and enhancement of natural beauty and provision of opportunities for open air recreation within them” (Glytis 1991). The National Parks Commission was also given the power to designate Sites of Special Scientific Interest (SSSIs) and National Nature Reserves (NNRs), and local authorities to designate Local Nature Reserves (LNRs).

With mass car ownership, demand for recreation grew dramatically as did the need to protect the wider countryside. The Countryside Act 1968 helped to address this by providing guidelines for conserving areas of the countryside outside protected areas (Seabrooke & Miles 1993). This Act replaced the National Parks Commission with the Countryside Commission, which had wider powers, and provided for the designation of country parks – areas primarily intended for recreation. Whilst there were ongoing concerns regarding conflicts between recreation and conservation, government recognition that participation in outdoor recreation is positive and should be encouraged was affirmed by the House of Lords Select Committee on Sport and Leisure in 1973 (Glytis 1991). Further provisions for public rights of way were included in the Wildlife and Countryside Act 1981, which required local authorities to maintain up-to-date definitive maps of rights of way.

In 2000 the UK saw major policy reform with the passing of the Countryside Rights of Way (CRoW) Act by a newly elected Labour Government. Denying access rights over open land had long been considered by the general populace as ‘an assault on liberty by landowning interests’, and historically many Labour politicians supported the access lobby seeking a resolution to the class inequality in public access (Parker 2008). In implementing the CRoW Act, Labour leader Tony Blair stated “I hope and believe that matters can be resolved sensibly, so that people have access to more of our countryside, which they have wanted for many years, without interfering with the proper use of the land by it owners” (Hansard 2004). The CRoW Act provided for the first time a statutory right of access to open country and registered common land in England and Wales, after decades of political battles between private property rights and public access rights (Parker 2008). This was seen as a major triumph for the British people who had been campaigning for this right for almost a century.

The CRoW Act grants a right of access on foot to areas of mountain, moor, heath, down and registered common land, provided that access does not impinge on wildlife (Bathe 2007). Whilst the CRoW Act strengthened legal protection of wildlife from recreational impacts, there was a lot of uncertainty about what the likely impacts would be, particularly in light of the fact that 55% of access land was also designated for conservation as a SSSI (Bathe 2007). Scientific research ensued aided by government funding, and assessment guidelines based on empirical evidence were written by the Wildlife and Access Advisory Group (formed of statutory agencies and conservation NGOs) (Bathe 2007). Rising pressure on the natural environment from changing demand, increasing population, and new breadth in the types of activities people can perform outdoors (e.g. mountain biking, geocaching, gorge walking and mechanised activities such as snowmobiling, jet skiing and ATV riding), as well as other factors such as climate change that may interplay with recreational impacts, requires that research is ongoing to find solutions to the co-existence of recreation and conservation.

1.3 Recreation and biodiversity conservation

1.3.1 Importance of biodiversity to recreationists

It is well-established that exposure to and contact with the natural environment benefits human health and well-being (Keniger *et al.* 2013). Much less is known, however, about the importance of the quality of the environment. Sandifer, Sutton-Grier & Ward (2015) reviewed the current limited literature on the psychological and physiological benefits of biodiversity, and conclude that “taken together these studies suggest that contact with biodiverse environments, or those perceived to be biodiverse, result in positive benefits to human well-being”. However, only two of the studies reviewed investigated recreationists and biodiversity in semi-natural spaces (Fuller *et al.* 2007; Dallimer *et al.* 2012). A more recent study by Carrus *et al.* (2015) found that recreationists reported higher well-being and perceived restorativeness in high compared to low biodiversity areas. However, Qiu, Lindberg & Nielsen (2013) found no positive relationship between preference for, and perception of, biodiversity, with an ornamental park having more ‘liked’ features than a woodland, despite recreationists recognising its comparatively low level of species richness. This mismatch between psychological benefits of, and preferences for, biodiversity clearly requires further investigation.

Preferences for high biodiversity may also be uncovered through empirical research on recreational site use. A comparison of protected areas in Finland found that total annual visits increased with increasing biodiversity (Siikamäki *et al.* 2015) and a comparison of protected area attributes for parks in Namibia showed that number of bird species likely to be seen is a strong driver of park choice (Naidoo & Adamowicz 2005). Hence biodiversity clearly plays a factor in selection of protected sites for nature tourism. However, in Europe, whilst protected areas cover a large proportion of semi-natural habitats, there are many opportunities to pursue recreational activities in the wider countryside. There is therefore a gap in knowledge on the importance of biodiversity for recreational site selection in the wider landscape (i.e. excluding large, statutorily protected areas).

1.3.2 Impacts of recreation on biodiversity

Recreationists can negatively impact the environment during their visit in a multitude of ways (e.g. McClung *et al.* 2004; Mallord *et al.* 2007; Pickering & Hill 2007; Reed & Merenlender 2008; Thompson 2015 and references therein). For decades ecologists have been investigating if, how and to what extent an environment or species is compromised by recreational activities. In the field of population biology, techniques have advanced from simple comparative studies to complex modelling and computer simulations. Gill (2007) describes how recreational disturbance may not always lead to a population-level response in a species, as this depends on the strength of density-dependence within the system. Identification of the mechanism and magnitude of detrimental effects of recreation on biodiversity is key to informing management decisions to obtain a balance between the two. New approaches in spatial analysis in particular could provide vital insights into the complex interactions between recreationists and wildlife, and provide essential tools for conservation and land management.

1.3.3 Managing recreation for biodiversity conservation

Land managers have a variety options available to them to allow coexistence of wildlife and recreation. Knight & Temple (1995) give four categories of restrictions: spatial, temporal, behavioural and visual. Spatial restrictions can involve designating areas important to wildlife, often as a reserve network under legal protection with regulation of recreational activities (e.g. Madsen, Pihl & Clausen 1998). Leung & Marion (1999) describe four spatial strategies for managing visitor impacts that can implemented at various spatial scales (Table 1.1). The particular strategy employed should be based on sound scientific understanding of a species' response to disturbance; Mallord *et al.* (2006) demonstrate how different access strategies and therefore spatial distributions of disturbance can have very different consequences for a species' population.

Temporal restrictions are also strongly informed by knowledge of a species' ecology, as management interventions are applied for 'time periods when wildlife use critical resources' (Knight & Temple 1995). Various temporal scales can be considered, from certain times of the day during, for example, key feeding or sleeping times, to periods lasting many weeks, for example during breeding or hibernation.

Table 1.1 Typology of spatial strategies for managing visitor impacts in national parks and other protected areas developed by Leung & Marion (1999)

	Management Strategy			
	Spatial Segregation	Spatial Containment	Spatial Dispersal	Spatial Configuration
Primary Goal(s)	• Match types and levels of visitation with resource capabilities	• Concentrate visitation pressure on a limited number of established or resistant locations	• Spread visitation pressure across a large area to maintain low frequency of use per unit area	• Reduce unnecessary visitor impacts through spatial arrangement of recreation facilities and resources
	• Shield sensitive areas and resources from visitation pressure		• Increase distance between parties with same number of sites	
Common Forms of Implementation	• Use zoning	• Designated areas	• Linear dispersal	• Campground layout
	• Exclude by buffer area requirements or closures	• Designated sites	• Total dispersal	• Trail networks
Common Spatial Scales of Implementation	• Site, regional, park-wide	• Site, regional	• Site, regional, park-wide	• Site
Common Tools of Implementation	• Park planning and management frameworks	• Regulations	• Regulations	• Site planning and design
	• Regulations	• Provisions of facilities	• Leave No Trace (LNT) education	• Provision of facilities
	• Physical barriers	• Site maintenance		

Behavioural restrictions centre on altering human behaviour, for example noise, speed and type of recreational activity (Knight & Temple 1995). Motorised recreational activities tend to have greater impacts than other (non-consumptive) recreational activities, and are often prohibited in areas of concern (Cole & Landres 1995). Dog walkers may be required to keep dogs on leads in conservation areas, for example where ground nesting birds are present, and swimming and boating may be prohibited in protected lakes to conserve reed beds and associated biodiversity.

Visual buffers can screen recreationists from wildlife to reduce or eliminate adverse impacts. These can include bird hides, vegetation (e.g. Taylor, Green & Perrins 2007), tree lines, fences with a viewing flap etc. Knight & Temple (1995) state that visual buffers can reduce the need for spatial restrictions, as recreationists may be undetected or pose less of a threat at closer distances than without visual interventions.

Placing restrictions on recreational access could be avoided through appropriately designing recreational areas, giving careful consideration to spatial arrangement of facilities (Knight & Temple 1995; Leung & Marion 1999). This affects routes taken within a site (Orellana *et al.* 2012) and the locations where recreational impacts may be concentrated (e.g. in resting/picnic places or campsites), and therefore relative recreational pressure throughout a site. Optimal placement of campsites, trails and attractions in the interest of conservation is generally considered on a site-by-site basis in site-specific management plans. Knight & Temple (1995) argue that more studies are required on wildlife-responses to recreation so that generalisations can be made to inform management plans.

1.4 Current knowledge of recreational demand

Understanding the drivers of recreational demand is key to providing suitable recreational opportunities and increasing participation, whilst knowledge of the distribution of demand is crucial for informing conservation.

Demand is clearly disproportionate across habitat types, with preferences elucidated for coastal and freshwater sites, followed by mountain and woodland sites in England (Sen *et al.* 2014). Similarly in Denmark, parks had the highest frequency of daily visits, followed by beach, sea and lake then forest (Schipperijn *et al.* 2010). Preferences are also evident for certain characteristics within a habitat type; for example Edwards *et al.* (2012) found that phase of development (establishment, young, medium and old) and size of trees positively increased recreational value of forests, and that broadleaved woodlands were slightly preferred over coniferous. Coastal users, however, had weak preferences for specific coastal habitats (sand, rocks, sand dunes, saltmarsh and cliffs), although bird watchers indicated a higher preference for saltmarshes and sand dunes (Coombes & Jones 2010). There is a lack of evidence, however, for how habitat quality affects recreational use, for example whether high conservation value woodlands are preferred over degraded, low biodiversity woodlands.

Importantly, recreationist demands differ with the activity they are performing. For example, horse riders, mountain bikers, joggers and walkers in forests require paths with different surfaces, widths, condition, gradient and features along the route (Roovers, Hermy & Gulinck 2002; Janowsky & Becker 2003), and dog walkers at the coast had a higher preference for a wide and remote beach than other users (Coombes & Jones 2010). It is therefore preferable that when modelling recreational demand, separate models be constructed for different user groups (Brainard, Bateman & Lovett 2001).

Predicting demand for a recreational site is important, as this can provide information on how close a site is to carrying capacity and allow estimation of visit numbers under future scenarios, for example climate change (Coombes, Jones & Sutherland 2009). Visitor management, however, also requires an understanding of the spatio-temporal distribution of visitors throughout a site (Cole & Daniel 2003). Such knowledge not only leads to more effective planning and management to enhance the visitor experience (e.g. developing transportation and facilities Cole & Daniel 2003; Beeco *et al.* 2013), but is also crucial for managing visitors in the interests of conservation. It is, however, very difficult to model or predict visitor distribution within a site; a variety of approaches have been tried, each of which has limitations.

1.4.1 Approaches to mapping and modelling visitor distribution

The site-based interview is a long established method for visitor monitoring and management, but has more recently been used to make spatially-explicit predictions of the levels of recreational use throughout a site. Smallwood, Beckley & Moore (2011) mapped the density of on- and off-shore recreation during peak and off-peak months within a marine national park based on travel pathways of coastal recreationists created by linking start and end locations obtained through questionnaires. Coombes & Jones (2010) used a similar approach, mapping the intensity of recreational use across a coastal area based on route maps sketched by recreationists during surveys. Whilst these methods can provide an assessment of the current relative recreational pressure across the site, it does not provide a statistical model that can be used to predict future patterns arising from management interventions (e.g. closing access points) or change in source population; the mechanisms underlying the current spatio-temporal distribution of recreation have not been identified. Another issue concerns recall bias, as respondents may find it difficult to recall or map their route (Taczanowska, Muhar & Brandenburg 2008). For instance, of 257 visitors at campgrounds in the Flinders Ranges gorges, Australia, that were shown a topographic map with key features marked and given a brief map orientation, over one fifth (22%) were unable to remember where they had camped, 35% unable to recall where they had stopped and 65% for how long they had stopped (Wolf, Hagenloh & Croft 2012); self-reported reliability also indicated uncertainty in responses of those who thought they could remember. Whilst site-based interviews remain important in visitor research, visitor impacts on natural environments result more from what people do than what they say they do (Cessford &Muhar, 2003; Cole & Daniel, 2003).

Wolf, Hagenloh & Croft (2012) therefore recommend the use of Global Positioning System (GPS) technology to significantly increase both reliability and detail of data obtained (e.g. duration of stops, location of stopping places and departures from designated trails; Taczanowska, Muhar & Brandenburg (2008). Reliable data on visitor movements can be obtained by using GPS technology to track movements for the duration of the visit, although data quality may be compromised due to poor GPS signal for example under a forest canopy (Taczanowska, Muhar & Brandenburg 2008). GPS data can be combined with other monitoring methods, such as questionnaire responses to help describe, and potentially explain, travel patterns (e.g. Marwijk, Elands & Lengkeek 2007; Beeco *et al.* 2013). However, retrieving GPS devices may prove difficult, precluding participants not planning to return to a staffed location (Taczanowska, Muhar & Brandenburg 2008), or risking low return rates; Beeco *et al.* 2013) issued GPS receivers and questionnaires and instructed participants to return both in a pre-addressed pre-stamped box at the end of their trip, achieving only a 65% return rate (3.1% of participants did not return the GPS unit, 5.5% returned the GPS with zero data and 26.6% were missing questionnaire data) with the possibility of selection bias. Of the literature in which GPS technology was employed in the study of recreational behaviour, most have focused on describing patterns rather than investigating the mechanisms driving these patterns (Beeco *et al.* 2013).

1.4.2 Combining observational data with statistical modelling

It is possible to develop mechanistic models using observed data on recreational visits if models include information on the factors that influence visitor distribution, and these may be used to make spatially-explicit predictions of visitor numbers at unknown locations within a site. Objective, quantitative data on visitor use and distribution can be obtained by on-site visitor counting, a well-established visitor monitoring technique (Cope, Doxford & Probert 2000). Visitor counts may be recorded manually by observers (e.g. Dolman, Lake & Bertoncelj 2008; Clarke, Sharp & Liley 2010), time-lapse video recording (e.g.

Arnberger & Hinterberger 2003), aerial surveys (e.g. Smallwood *et al.* 2011; Tratalos *et al.* 2013), pre-existing data collected by land management organisations as part of ongoing recreation provision assessments (Cope, Doxford & Probert 2000) and various other methods.

Dolman, Lake & Bertoncelj (2008) modelled the distribution of recreational visits within an extensive site (94,000 ha) using a large sample of visitor counts collected by observers from 308 locations (path intersections) throughout the site. Predictions of number of disturbance events (recreational passes) per hour per path section could then be made for the entire path network and alternative scenarios of house building were explored to predict future levels of recreational activity. Predictions were used to inform stone-curlew (*Burhinus oedicnemus*) conservation. Whilst the collection of data for this approach is time-consuming requiring many observer hours (e.g. 1,110 hours in the above study), it requires no special equipment or training and can be collected by a team of volunteers. The main advantage is that it allows for hypothetical experimental manipulation to investigate potential changes in recreational demand (e.g. population increase) and possible mitigation strategies (e.g. closing access points). It is based on empirical data, and does not therefore have any of the uncertainties or errors associated with questionnaires and recall bias. With such an extensive site it would also be very difficult to issue and reclaim GPS devices, and there would be potential for poor signal in forested areas. The approach could however be improved by taking into account path types along the whole route from an access point to a surveyed path, rather than just the path type of surveyed paths (which was included as a categorical variable in the model), as recreationists may deviate from the shortest linear distance along the path network if they favour particular path types.

Coppes & Braunisch (2013) develop a statistical model based on observations of visitors to predict the probability of winter recreationists leaving marked trails in the southern Black Forest, Germany. Six locations of potential conflict with red deer and 56 with capercaillie were identified by intersecting high probability sites with species' habitat maps. Similarly, Braunisch, Patthey & Arlettaz (2011) mapped the probability of presence of black grouse and snow-sport recreationists in the Swiss Alps and found that both are likely to occur in 67% of suitable black grouse wintering habitat. The resulting conflict maps can be used to identify areas with intense human-wildlife conflict in need of designation as wildlife reserves with controls on human access (Braunisch, Patthey & Arlettaz 2011). Whilst these models can predict areas of likely co-occurrence, they do not give estimates of recreationist numbers, which is an important consideration in evaluating recreational impacts.

1.4.3 Methodological approach used in this thesis

This thesis uses a statistical modelling approach based on observational records of recreationists. A massive data set of visited locations is modelled to identify the drivers of recreational site selection, and observations of recreationists within an extensive and complex site are used to model within-site distribution. Whilst this approach does not incorporate social psychological constructs such as place attachment, which is known to affect recreational demand (Hailu, Boxall & McFarlane 2005), it identifies overarching population-level trends (Coombes 2007) and therefore allows examination of population and demographic change.

1.5 Thesis aims and structure

The broad aim of this thesis is to improve mechanistic understanding of the distribution of recreational demand at a variety of spatial scales from national to individual site. It then aims to use these statistical models to propose solutions for reducing conflict between recreation and biodiversity conservation.

This broad aim can be subdivided into four more detailed aims, each of which are addressed in four empirical chapters. In Chapter 2 a model is developed to investigate the drivers of recreational site selection on a national scale, specifically how land cover type affects the likelihood of site selection and how this is modified by high nature value. This enables us to examine whether recreational use is an important ecosystem service of biodiversity and whether the creation of (or improvement of access to) low biodiversity areas can relieve pressure from vulnerable biodiverse areas.

Chapter 3 investigates spatial patterns of recreational demand across England through application of the model developed in Chapter 2. Fine-scale visit estimates were mapped across the country to reveal the current pressures on habitats and conservation areas, and future population projections were used to investigate the potential changes in distribution and volume of visits.

Chapter 4 builds on the work of Chapter 3 for a case study area (Norfolk), in which potential changes in recreational pressure on natural and semi-natural land covers arising from proposed new housing (changes in land use and localised population increase) and road developments is explored.

Chapter 5 uses a novel, large data set of on-site visitor counts to develop an innovative methodology for prediction of within-site (footpath level) spatial distribution of recreational pressure capable of exploring the effects of micro-level site changes. Spatial patterns of recreational disturbance are overlaid with proposed woodlark territories to estimate the number of territories potentially lost due to disturbance. The effects of house building on disturbance levels and therefore number of territories lost is also investigated along with possible mitigation through closing access points. Finally, Chapter 6 presents a discussion of how this thesis has furthered our knowledge of nature-based recreation and contributed vital insights for conservation and land management. Possible directions for future research are also discussed.

All empirical chapters (Chapter 2 – 5) are written in the format of manuscripts for peer-reviewed journals, thus each chapter has a separate reference list. Chapter 2 (in a slightly condensed form) has been accepted for publication in PLoS ONE pending revisions (see Appendix A), and Chapters 4 and 5 are intended to be submitted to relevant journals imminently. I. Lake and P. Dolman made important contributions to the conception of the study, experimental design and interpretations of the results. Their contributions will be recognised through co-authorship of future manuscripts arising from Chapters 2-5. Chapter 5 is based in large part on visitor records obtained via field surveys conducted in 2007 coordinated by I. Bertoncelj (Dolman, Lake & Bertoncelj 2008) and 2008-9 (Dolman 2010), which were augmented with additional records collected by K. Hornigold via fieldwork conducted in 2013 and 2014 during the current PhD study. Also, the forest path network was modified and updated from an original layer created by I. Bertoncelj. The contribution of I. Bertoncelj to Chapter 5 will be recognised through co-authorship of a manuscript for publication in a peer-reviewed journal.

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Chapter 2

Recreational use of the countryside: no evidence that high nature value enhances a key ecosystem service

Abstract

In Western Europe, recreational amenity is presented as an important ecosystem service of protected high-nature-value areas, but whether recreational use by the general public is enhanced by a site's conservation value is unknown. The efficacy of mitigation of recreational impacts on conservation areas by provision of alternative 'green infrastructure' also requires evidence. We address both issues with the first study to model recreationists' habitat preferences at a national scale including statutory designation as an indicator of conservation importance. Recreationists preferred areas of coast, freshwater, broadleaved woodland and higher densities of footpaths and avoided arable cover, coniferous woodland and lowland heath. Although sites with conservation designation had similar or greater public access than undesignated areas of the same habitat, statutory designation *decreased* the probability of visitation to coastal and freshwater sites and gave no effect for broadleaved woodland. Thus general recreational use by the wider public did not represent an important ecosystem service of protected high-nature-value areas. Management of less important conservation areas to enhance provision of footpaths and desirable habitats can therefore mitigate recreational impacts on nearby valuable conservation areas.

2.1 Introduction

Nature-based recreation is presented as an important cultural ecosystem service (MEA 2005; Mace, Norris & Fitter 2012) that is frequently used to support investment in biodiversity conservation (Balmford *et al.* 2009; Defra 2011). Supporting evidence is, however, surprisingly scarce (Sandifer, Sutton-Grier & Ward 2015). Interacting with nature benefits physical health (e.g. reducing stress levels and mortality), cognitive performance (e.g. reducing mental fatigue) and psychological well-being (e.g. elevated mood and self-esteem; Keniger *et al.* 2013; Sandifer, Sutton-Grier & Ward 2015). On a global scale, visits to protected areas (PAs) are on the increase (Balmford *et al.* 2009) and there is evidence that PAs holding greater levels of biodiversity are preferentially visited by nature-based tourists (Naidoo & Adamowicz 2005; Siikamäki *et al.* 2015). However, whether the general public places greater recreational value on 'high nature value' areas (i.e. areas designated for rare or characteristic species and/or high biodiversity) versus the wider countryside for everyday recreational visits is unknown. This is especially important in Europe where there are many opportunities to pursue recreational activities in other types of 'green space'.

Recreationists can have undesirable effects on high nature value areas (Pickering & Hill 2007; Reed & Merenlender 2008), yet public use gives such areas amenity value with potential to generate conservation revenues (Balmford *et al.* 2009). It is therefore important to strategically manage provision of recreational opportunities, where possible re-distributing recreational pressure to areas of lower nature value. An understanding of the relative importance of high nature value areas versus the wider countryside for recreationists would aid in such provision to support conservation of potentially impacted areas.

Eigenbrod *et al.* (2009) found that PAs across England delivered ecosystem service benefits for biodiversity but fewer recreational visits occurred within PAs than was predicted from their land area. However, the recreational benefits could not be fully evaluated as local population density was not included in the analysis; an important omission as population density was twice as high in the vicinity of visits to the wider countryside than visits to PAs in this study, and considering that the majority of recreational visits take place within a few kilometres from home (Dolman, Lake & Bertoncelj 2008; Schipperijn *et al.* 2010; Sen *et al.* 2014). Access networks should also be considered as recreational visits occur within a limited distance from roads, and recreationists may have a preference for land cover type; both of which were also not included in the study by Eigenbrod *et al.* (2009). Sen *et al.* (2014) modelled recreational visits in Britain based on land cover class, travel distance, socio-demographics and population; but as this work assessed the economic value of recreation several ecologically distinct land cover types were merged (e.g. mountain, moorland and lowland heathland), visits within built-up areas were included and conservation status was omitted.

Here we use a modelling approach to identify the drivers of recreational site selection from an ecological perspective, using a nationwide sample of over 30,000 spatially referenced visits to the countryside. We model the influence of site characteristics including land cover on the probability of site visitation, controlling for accessibility, population density and local behavioural differences. We examined how high nature value affected probability of site visitation, using statutory designation as Sites of Special Scientific Interest (SSSIs) as a proxy. SSSIs represent the UKs most important sites for biodiversity conservation (Ratcliffe 1977), are designated using objective criteria and include all National Nature Reserves (NNRs) and Natura 2000 sites designated under European Directives. We aim to determine whether high nature value areas are under more recreational pressure and whether recreational use is an important ecosystem service of such areas. This would inform whether the creation of nature recreation opportunities in places of lower conservation concern can relieve pressure from more vulnerable conservation areas. Furthermore we aim to enhance the evidence-base for mitigation of recreational impacts, identifying which habitats are in greatest demand and thus where creation of alternative green infrastructure is most required and what this should comprise.

2.2 Methods

2.2.1 Study design

A case-control design (Manly *et al.* 2002) was employed comparing recreational visit sites with randomly selected countryside sites (controls). Point visit locations were taken from the Monitor of Engagement with the Natural Environment (MENE) survey (2009-2012) of recreational activity by English households (Natural England 2012a). Face-to-face in-home interviews were conducted every week of the year, with a nationally representative sample obtained every week. Respondents were asked the number of recreational trips undertaken during the seven days preceding the interview, type of place visited and activities undertaken. One recreational visit was randomly selected during the interview for which the location was obtained, resulting in 44,485 visits. These were provided as a grid reference (Ordnance Survey National Grid) and mapped as points in ArcGIS 10.1 (Copyright © ESRI, USA); hereafter 'visit points'. We excluded visits in predominantly built-up areas to focus on countryside visits (see Appendix S2.1 in Supplementary Information), resulting in a final data set of 31,502 visit points for modelling. Controls (hereafter 'control points') were generated using Geospatial Modelling Environment (GME; Beyer 2012), which were randomly located within the boundaries of England but constrained to be at least 25 m from visit points so that control points could not be placed in a known visit location. Twice as many control points were generated (63,000) to maximise statistical robustness within the bounds of computational limitations. Controls within predominantly built-up areas were excluded in the same way as for visits (Fig. 2.1). A quasi-experimental design was tested where control points were stratified by distance-weighted population, a combined measure of travel cost and population density surrounding visit points (see section 2.2.2.3). Random controls however were considered superior as an explicit measure of population and travel cost could be included in models (see Appendix S2.1). As recreationists generally visit an area not just one point, visit and control points were buffered by a 400 m radius, informed by empirical evidence of visitor countryside access patterns (see Appendix S2.1). Buffered visit and control points are hereafter referred to as 'visit sites' and 'control sites', or jointly as 'sites'. Due to the heterogeneous representation of land cover within each visit site, statistical matching to pair protected locations with sites having similar characteristics were infeasible.

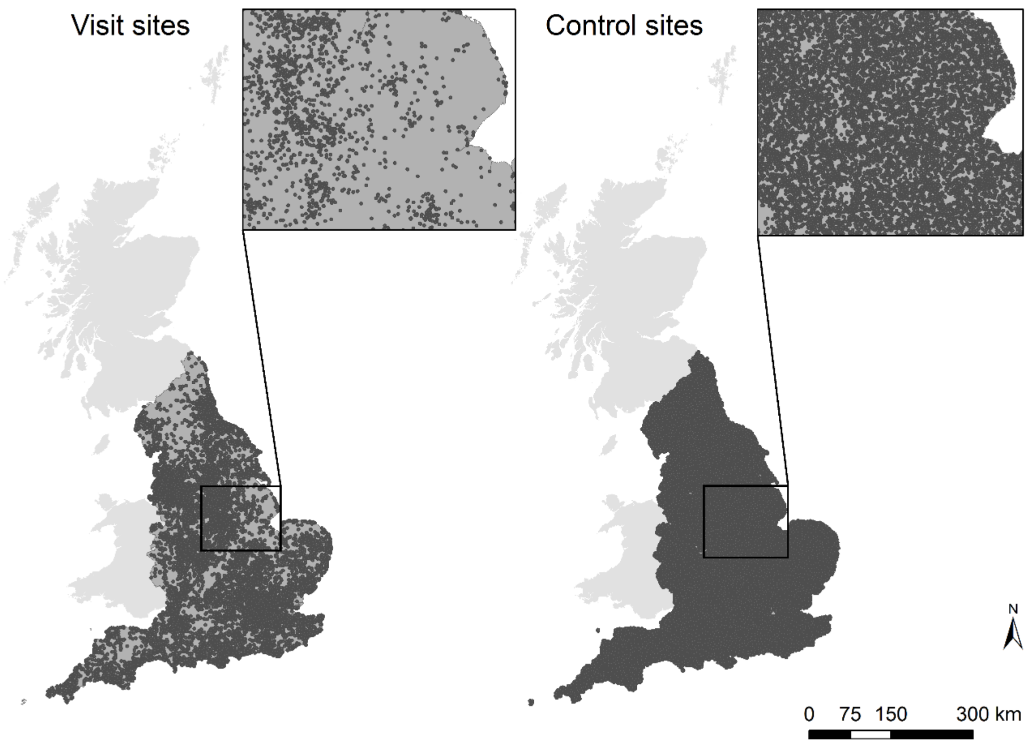

Fig. 2.1. Distribution within England of visit and control points used in this study

2.2.2 Predictors of visitation

2.2.2.1 Land cover data

The proportion of each land cover was extracted for all sites from the 25 m resolution Land Cover Map 2007 (LCM2007; Morton *et al.* 2011). The 22 LCM2007 land cover classes were aggregated into 11 broad classes as some were not distinguished reliably using spectral signature (e.g. neutral, calcareous, acid and rough grassland, therefore grouped as semi-natural grassland; Morton *et al.* 2011) and others were limited in area (e.g. supra-littoral rock, that was grouped with littoral rock). Land cover classes appearing in fewer than 10% of visit sites were excluded from analysis due to insufficient power (following Boughey et al. 2011), so that 9 classes remained (Table 2.1).

To examine the influence of nature value upon visitation, land cover classes were further divided into areas inside or outside Sites of Special Scientific Interest (SSSI), as mapped by Natural England (2012b). Land cover classes not subdivided included built-up and improved grasslands as these are not designated as SSSIs, and arable land which is rarely designated. SSSIs cover more than 8% of the country and the majority (98% of total SSSI area) are designated for biodiversity (Natural England 2011) although a small number have also been designated for their geological interest.

Given that we are examining recreational visits between SSSIs and non-SSSIs it is important to examine access constraints between these two areas. To determine ‘open access’ we combined areas designated as either having a statutory right of access under the Countryside Rights of Way Act (CRoW), or as Country Parks or Local Nature Reserves which are open to the public. Together these cover 8.5% of England. We then calculated the proportion of broadleaved woodland, freshwater, coast and lowland heath that was ‘open access’ within and outside SSSIs. For land covers with SSSI designation, 6-20 times as much area was open access (coast, 6.2%; freshwater, 17.1%; broadleaved woodland, 29.7%; lowland heath, 86.0%) than for equivalent land cover not designated as SSSI (0.3%, 1.4%, 5.2% and 4.1% respectively) so that, all else being equal, a much greater visitation rate would be expected. A further subsidiary analysis was also carried out, in which we divided land cover classes into areas inside or outside National Nature Reserves (NNRs). NNRs are high quality SSSIs used to showcase management of SSSIs and engage the public; thus are areas of high nature value with recreational access encouraged. The results from this analysis are compared with those from the original SSSI analysis.

Table 2.1. Candidate variables used to model probability of site visitation by recreationists.

Code	Predictor	Units	Description
Comp	Arable*	Proportion	Proportion of annual and perennial crops and freshly ploughed land
	Coast*	Proportion	Proportion of sand dunes, shingle, littoral mud and littoral sand
	Broadleaved*	Proportion	Proportion of broadleaved woodland with >20% tree cover or >30% scrub cover
	Built-up*	Proportion	Proportion of urban and suburban areas including towns, cities (and residential gardens), car parks and industrial estates
	Coniferous*	Proportion	Proportion of coniferous woodland with >20% cover
	Freshwater*	Proportion	Proportion of lakes, canals, rivers and streams
	Improved*	Proportion	Proportion of grassland modified by fertiliser and reseeding typically managed as pasture or mown
	Lowland*	Proportion	Proportion of heather and dwarf shrub, gorse and dry heath below 300 m a.s.l. as defined by Gimingham (1972), delimited according to the digital terrain model OS Terrain 50 (OS Terrain 50 2013)
	Semi-natural*	Proportion	Proportion of neutral, calcareous, acid and rough grassland
Pop	Weight.pop.2†	No. people	Total number of people residing within 10 km of the site, inverse-weighted by straight-line distance from visit and control points
Cty	County‡	94 levels	County in which the site is located
Path	Path.length§	m	Total length of path network within site
Elev	Mean.elev¶	m	Mean of all Digital Terrain Model 50 m cells within site
Road	Dist.Aroad**	m	Distance from visit and control points to nearest major road

* LCM2007

†1km resolution population raster created from 2011 ONS census data

‡ Assigned according to county boundaries downloaded from http://www.gadm.org/

§ OpenStreetMap

¶ OS Terrain 50

** OS Meridian

2.2.2.2 Additional within site variables

Outdoor recreation in the UK is enabled by a network of public rights of way. A path network layer encompassing bridleways, cycleways, footpaths, paths and tracks was extracted from OpenStreetMap (2013), the only digital data set delineating public rights of way with a national coverage. These data were collected by contributors using GPS devices, aerial imagery and field maps, which we validated against OS maps (see Appendix S2.1). Site accessibility was indexed as the path network length within sites and additionally as the straight line distance from visit and control points to the nearest major road (A Road; OS Meridian 2 2013).

Mean elevation of sites (extracted from OST50 2013) was selected *a priori* as a predictor variable, as sites with lower mean elevation were expected to have a greater probability of visitation, as those engaging in arduous activity are a subset of recreationists. Elevation gain within sites was explored but provided less explanatory power.

2.2.2.3 Distance-weighted population

The larger the resident population in the vicinity of a site the more likely it is to be visited, diminishing with distance. Population data from the 2011 census of households provided by the Office for National Statistics (2011) (England and Wales) and National Records of Scotland (2011) were linked to coordinates, using the UK Postcode Directory (UKDS 2013), and aggregated into 1 km grid cells to create a UK population raster (some visits close to the borders may originate from Wales or Scotland). Of three population density functions tested, weight.pop.2 (population weighted by inverse of distance squared) best distinguished visit from control sites and was included in all subsequent models (see Appendix S2.1).

2.2.2.4 Analysis

Generalised linear mixed models (GLMMs) with binomial error and logit link function predicted $P(\text{Visit}_i)$, the probability of a recreational visit to site i, as a function of the proportions of site land cover classes (Comp_i), mean site elevation, distance from nearest major road and path density (fixed effects, Table 2.1), controlling for distance-weighted population (weight.pop.2) and county (random effects). Counties vary in area from 28 km^2 to 7965 km^2 with a mean population of 644,944 (categorical, 85 levels, from database of Global Administrative Areas (GADM 2013)). The interaction between weight.pop.2 and county captured unobserved heterogeneity between counties and allowed for potential differences in per capita frequency of recreational activity due to socioeconomic or cultural effects. Predictor variables were centred and scaled with zero mean and unit standard deviation for comparability of coefficients (Schielzeth 2010). Two GLMMs were fitted; in model 1 all land cover classes were included once, in model 2 selected land cover classes were divided into areas designated and non-designated as SSSIs:

$$P(\text{Visit}_i) = \text{f}(\text{Comp}_i, \text{Elev}_i, \text{Road}_i, \text{Path}_i, \text{Pop}_i, \text{Cty}_i) \qquad \text{(model 1)}$$

$$P(\text{Visit}_i) = \text{f}(\text{Comp.non-des}_i, \text{Comp.des}_i, \text{Elev}_i, \text{Road}_i, \text{Path}_i, \text{Pop}_i, \text{Cty}_i) \qquad \text{(model 2)}$$

Differences between equivalent non-designated and designated (model 2) land cover coefficients, were evaluated by Z tests.

GLMMs were fitted using the lme4 package (Bates *et al.* 2013). Inspection of correlograms established that spatial autocorrelation was negligible in both models (see Appendix S2.1). Predictive performance of the two models was evaluated against independent data from the subsequent 2012-2013 MENE survey (n = 10,622) and additional random controls (n = 10,622). For each model, AUC - the area under the receiver operating characteristic (ROC) curve - was calculated using the pROC package in R (Robin *et al.* 2011); AUC ranges from 0.5 for models that perform no better than random, to 1 for models with perfect discrimination (Pearce & Ferrier 2000). Whether AUC values (and thus model prediction accuracy) differed significantly among models was tested (following DeLong *et al.* 1988) within the pROC package.

2.3 Results

2.3.1 Recreationists' preferences for site characteristics

The initial model examining the effects of site characteristics upon visitation probability without considering designation status indicated that the strongest effect was the positive influence of path density (mean length within visit sites 2055 m ± 1916 SD; within control sites 604 m ± 865 SD, Table 2.2). Visitation probability was strongly reduced for sites at higher elevation or far from a major road. Intercepts for each county ranged from -1.45 ± 0.16 95% CI to 1.14 ± 0.24 95% CI and weight.pop.2 coefficients from -1.67 ± 0.95 95% CI to 2.26 ± 0.61 95% CI (Fig. S2.3), showing variation in per capita visitation probability between counties and supporting inclusion of random effect terms.

Table 2.2 Generalised linear mixed model predicting recreational demand in the countryside, controlling for population and county. Dependent variable: probability of visitation. *P*<0.001 '***', *P*<0.01 '**'

	Standardised Coefficient	Std. Error	z	P	Lower CI	Upper CI
Non-land cover variables						
Path length	0.826	0.014	59.96	***	0.799	0.853
Elevation	-0.370	0.017	-22.22	***	-0.403	-0.338
Distance to major road	-0.132	0.013	-9.83	***	-0.158	-0.106
Land cover classes with positive effect						
Built-up	0.631	0.022	29.14	***	0.589	0.673
Coast	0.287	0.016	18.49	***	0.256	0.317
Freshwater	0.161	0.010	16.26	***	0.142	0.181
Broadleaved	0.158	0.015	10.37	***	0.128	0.188
Land cover classes with negative effect						
Arable	-0.645	0.031	-20.70	***	-0.706	-0.584
Improved	-0.129	0.022	-5.80	***	-0.172	-0.085
Lowland heath	-0.080	0.012	-6.64	***	-0.103	-0.056
Coniferous	-0.078	0.013	-6.18	***	-0.103	-0.053
Semi-natural	-0.043	0.016	-2.73	**	-0.074	-0.012
Constant	-0.697	0.077	-9.06	***	-0.847	-0.545

Of the semi-natural land cover classes, coast had the strongest positive effect on the probability of visitation, followed by freshwater and broadleaved woodland. Probability of visitation was 50% at proportionate covers of coast and freshwater of 0.11 and 0.15 respectively (Figs 2.2a & 2.2b), whereas a greater cover of broadleaved woodland (approximately 0.43 and above; Fig. 2.2c) was required before a site was more likely to be visited than not.

The land cover with the strongest negative effect on visitation probability was arable land, which had a large negative coefficient relative to other land cover classes (Fig. 2.2d). Recreationists were less likely to visit sites comprising of a greater proportion of lowland heathland, improved and semi-natural grassland or coniferous woodland.

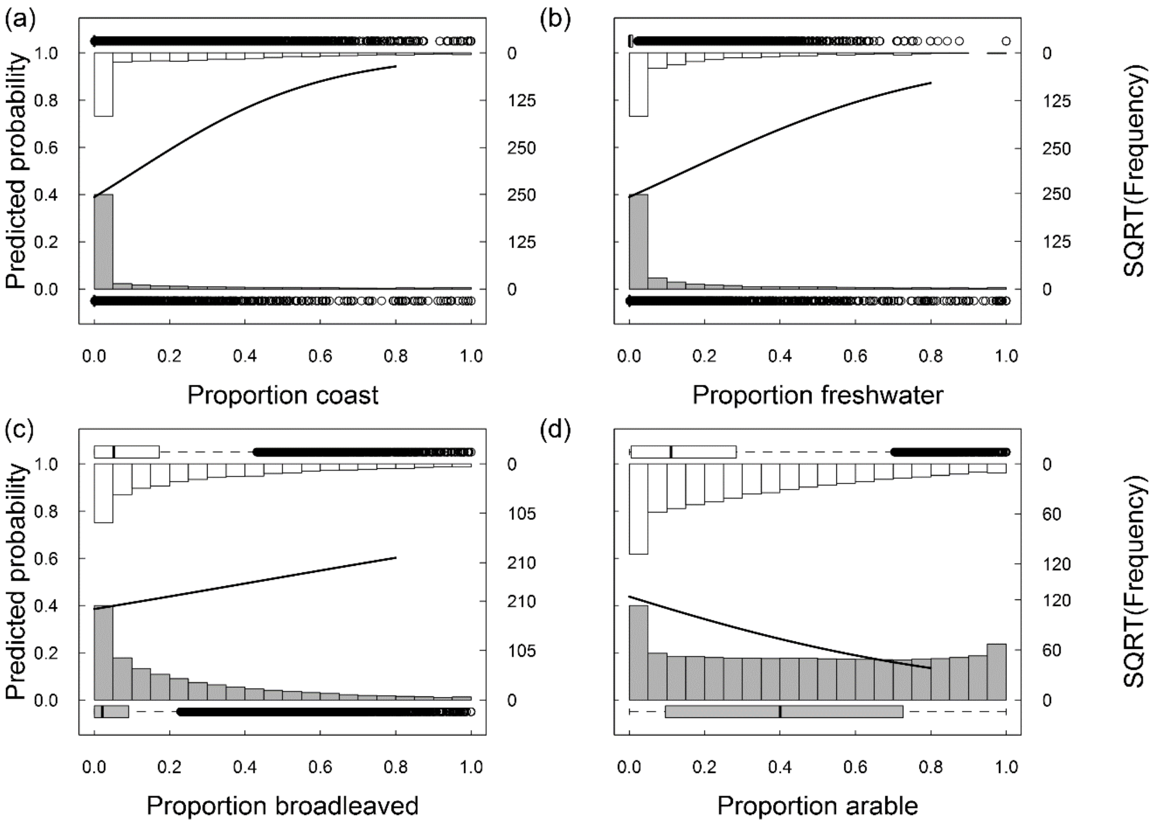

Fig. 2.2. Predicted influence on visitation probability of coast, freshwater, broadleaved and arable cover, from model 1 (see section 2.2.2.4) controlling for path length, elevation, distance to nearest major road, weighted population and county. Bars show the frequency distribution (square root scaled) within visit (unfilled) and control (grey) sites. Predictions were obtained by varying the proportionate cover of the land cover class shown between 0-0.8. All other land cover classes were held proportional to their mean such that they sum to 0.2 (so that total land cover proportion did not exceed 1). Control variables were held at their mean. Horizontal box and whisker plots show median, quartiles and outliers of land cover proportions in visit (unfilled) and control (grey) sites

2.3.2 Effect of habitat designation

We then examined preferences for land cover classes of potential conservation importance separately, according to whether they occurred inside or outside an SSSI. Explanatory power increased (Δ AIC = -186), with a slight increase in predictive ability (Model 1: AUC = 0.8425 ± 0.005 95% CI; Model 2: AUC = 0.8430 ± 0.005 95% CI; $Z = 2.44$, $P < 0.05$).

The appeal of broadleaved woodland was similar irrespective of whether it was within an SSSI ($Z = 0.7$, $P = 0.47$; Fig. 2.3) with the coefficient similar between these models (Δ = -0.013 ± 0.018 SE), whereas the attractiveness of coast and freshwater was significantly greater when not designated ($Z = 7.8$, $P < 0.001$ and $Z = 6.2$, $P < 0.001$ respectively; Fig. 2.3). Non-designated coast and freshwater coefficients were close to the original (model 1) coefficient error bounds, but SSSI-designated coefficients were lower (designated versus non-designated: coast $\Delta = -0.188 \pm 0.024$ SE; freshwater $\Delta = -0.079 \pm 0.013$ SE). Effects of designating the freshwater or coastal area within a site was examined separately for low (20%) and high (80%) overall cover, holding remaining land cover classes constant in proportion to their national mean. Freshwater designation minimally affected visitation probability at low cover (0.559 non-designated, 0.518 designated) but at high cover visitation probability decreased with designation (0.900 non-designated, 0.813 designated; Fig. S2.4a). Coastal designation substantially reduced visitation probability (low cover: 0.748 non-designated, 0.544 designated; high cover: 0.996 non-designated, 0.865 designated; Fig. S2.4b). The negative effect of semi-natural grassland did not differ with designation, whereas coniferous woodland and lowland heath were significantly more negatively associated with visitation probability when designated ($Z = 4.45$, $P < 0.001$ and $Z = 2.20$, $P < 0.05$ respectively). Results from the subsidiary analysis on land covers divided into areas inside or outside NNRs were consistent with SSSI results for broadleaf woodland, coast, freshwater and semi-natural grassland, but lowland heath and coniferous woodland SSSIs had no significant effect on visitation probability (Fig. S2.5).

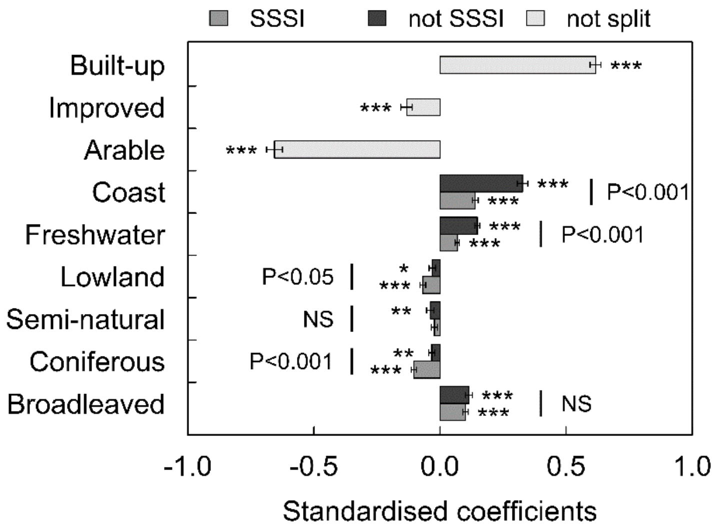

Fig. 2.3. Effects on visitation probability of land covers separately outside or inside an SSSI, showing standardised coefficients from a GLMM controlling for path length, elevation, distance to nearest major road, weighted population and county. Bars denote standard error. For each land cover, P values of Z-tests compare pairs of coefficients between outside/inside SSSI (P<0.001 '***', P<0.01 '**', P<0.05 '*')

2.4 Discussion

This study found that although broadleaved woodland had clear appeal to recreationists, some other land covers of conservation importance such as lowland heath were not preferentially selected. Most importantly, for preferred land cover classes we found no evidence that SSSI designated high nature value sites had greater appeal, despite these having much greater levels of permitted open access. This is an important development of previous work (e.g. Sen *et al.* 2014) which did not consider ecologically distinct land uses or importance of nature value. Recreation was previously found to be underrepresented by protected areas in England, but in analyses that did not control for source population density, road access or footpath density (Eigenbrod *et al.* 2009). Controlling for these factors, we now provide clear evidence that high nature value (inferred by statutory designation as SSSI) does not confer additional recreational value for the general public. This has important implications for provision of recreational infrastructure and justifications of biodiversity conservation. Provision of recreational opportunities in low nature value sites of preferred habitat types can reduce pressure on vulnerable conservation areas, assuming that the number of visits in a recreational catchment remains constant.

When a land cover was of elevated conservation importance (i.e. designated as a SSSI or NNR) recreational use by the wider public was not enhanced and in the case of coasts and freshwater it was less likely to be visited. Dallimer *et al.* (2012) found no consistent relationship between species richness and human well-being in a survey of visitors to riparian greenspaces, but a positive effect of *perceived* richness. We conclude that conservation importance does not strengthen the cultural service of recreational opportunities obtained from ecosystems because this is not recognised or sought by most recreationists. Whilst biodiversity is an important factor for nature tourists visiting national parks in Finland (Siikamäki *et al.* 2015) and protected areas in Uganda (Naidoo & Adamowicz 2005) this is based on a self-selected sample of nature enthusiasts; here we use a representative nationwide sample and find no evidence that conservation value plays a role in recreational site selection for day-to-day use. As SSSI status did not add to the appeal of sites for most recreationists, the public expenditure on these highly valued conservation areas (£85.4 million in England in 2008-09; Committee of Public Accounts 2009) whilst benefiting conservation does not bring benefits in terms of recreation. Most public benefits are likely expressed through non-use values (Dolman 2000).

This study showed that there is a higher likelihood of recreationists visiting broadleaved over coniferous woodland, a distinction not made in previous studies of forest recreation in Britain (Hill & Courtney 2006; Jones *et al.* 2010). Recreation value of coniferous woodlands can therefore be enhanced by planting or retention of broadleaved species along paths. Recreational activities in woodlands can impact sensitive ground flora and cause disturbance to wildlife and animal behaviour change (Marzano & Dandy 2012), which could compromise the effectiveness of woodlands designated for conservation. That designated and non-designated broadleaved woodlands were of equal appeal to recreationists has important implications for conservation woodlands, as provision of low biodiversity broadleaved woodland would be a suitable substitute to draw recreationists away from potentially vulnerable conservation areas. Lowland heaths support species and habitats of European conservation importance that are sensitive to recreational impacts; consequently there has been much research on visitation patterns within heathland (e.g. Clarke *et al.* 2006; Liley *et al.* 2005). Surprisingly, our study indicated that on a national level, in spite of their conservation importance, lowland heathland was not favoured by recreationists. It may be that lowland heaths are visited when local to recreationists although more desirable land covers may still be preferred.

The strong effect of path density emphasises the importance of public rights of way, both as a means to attract recreationists and to manage the intensity of recreational pressure through restriction of access to and within conservation areas. The Countryside Rights of Way Act (CRoW) 2000, provides a statutory right of access to mountains, moors, heaths, downs and registered common land in England and Wales (Bathe 2001) but allows mitigation by closure of access to protect breeding wildlife (Murison *et al.* 2007). Access restriction in deciduous forests aided vegetation recovery (Roovers *et al.* 2005) and closure to recreation was predicted to increase coastal ringed plover (*Charadrius hiaticula*) by 85% (Liley & Sutherland 2007). However, as it is not always possible to restrict access completely, an alternative is to create or re-route trails to divert recreationists to areas where they will have a lower impact on biodiversity (Tomczyk & Ewertowski 2013). Whether restriction or re-routing of paths is chosen, access management is an important tool for balancing recreation and conservation interests.

2.5 Conclusions

Understanding the mechanisms driving countryside recreationists' choice of visit location supports management of the countryside for both recreation and conservation. The relationships we identify were derived from a nationally representative sample of English households, and are likely to be relevant to other developed, urban based countries. Further studies are required however to gain a better understanding of cultural differences in the importance of nature value for general recreation, as the global picture may highlight differing trends as with nature-based tourism (Balmford *et al.* 2009). This analysis shows that, in spite of recreational use being frequently presented as an important ecosystem service and used to support investment in conservation, there is no ecosystem service gain from higher nature value in terms of recreational value by the general public. Protected areas benefit the wider public through non-use values and reconciliation of conservation and recreation remains pertinent.

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Supplementary Information

Appendix S2.1. Additional methodological details

Methodology for defining 'countryside' visits i.e. exclusion of visits to urban green spaces

As the focus of this study is to examine the effects of outdoor recreation on semi-natural habitats, we excluded visits in built-up areas. The data set included all recreational visits to the 'natural environment' (including for example urban recreation grounds, parks and playgrounds), only excluding shopping trips and visiting allotments or personal gardens. To selectively remove 'urban green spaces' the proportion of built-up area (extracted from Land Cover Map 2007 (Morton et al. 2011)) within buffers surrounding visit points was examined and a cut-off point applied to define a visit location as built-up or countryside. Cut-off points of 50%, 60%, 70%, 80% and 90% built-up area were examined at buffer sizes of 200, 400, 600, 800m radii (see Table S2.1).

The visit localities that were retained were examined spatially overlaid on the land cover map (LCM2007) for East Anglia to see where these were in relation to built-up areas. Visit localities retained under each percentage cut-off at each buffer radius in Table S2.1 were examined in turn and it was determined that the 70% cut-off within 400m buffers performed best (highlighted in bold) as visit localities close to large areas of open space within built-up areas and on the edge of cities were retained, whilst visit localities within large homogenous built-up areas were removed. At the 70% cut-off approximately 25% of observed visits were excluded due to being within built-up areas (Table S2.1).

Table S2.1. Number of visits excluded from observed visit (MENE) data set (n = 44,495) at different exclusion thresholds for built-up area within buffers and at different buffer radii

% built-up area	**Buffer radius**			
	200m	**400m**	**600m**	**800m**
>50	19303	19617	19146	18444
>60	15968	15776	15225	14565
>70	12851	**12301**	11796	10903
>80	9848	8579	7769	7049
>90	6721	4730	3704	2990

Methodology for creation of stratified controls, analysis using stratified control sample and comparison of results from models with stratified controls and random controls

We tried an alternative method of controlling for the effects of local source population density and travel cost. Instead of including distance-weighted source population (weight.pop) as a random effect in the model, we stratified the controls (i.e. used a new set of controls generated by stratified random sampling) to have a similar frequency histogram of distance-weighted source population as visit localities; thus we controlled for population and travel distance effects through locating controls in areas with a similar source population within the 10 km travel distance as the visit sample, making the weight.pop variable redundant. By allocating a similar ratio of control to visit localities within intervals of distance weighed population (see below) this allowed conclusions to be made regarding the relative of pull of different land cover types without confounding effects of local population density or proximity to residential areas.

The stratified controls were generated as follows. The frequency histogram of distance-weighted population for visit localities gave the number that fell within 22 bands of 5000 increments between the minimum weight.pop (zero) and maximum (110,000). The counts were then doubled to get the sample sizes for controls within the same bands. To generate control points, random points were placed within the boundaries of England and weight.pop was calculated until the sample sizes for each weight.pop band was achieved. These points were then buffered by 400m to create the final control localities. Model 2 (see section 2.3 in main paper) was run using the existing visit localities and the new stratified control localities and resulting coefficients were compared to original model coefficients (Fig. S2.1). The effect of significantly positive land covers were very similar to the original model. Non-significant and significantly negative land cover coefficients show some differences but their effect sizes remain trivial.

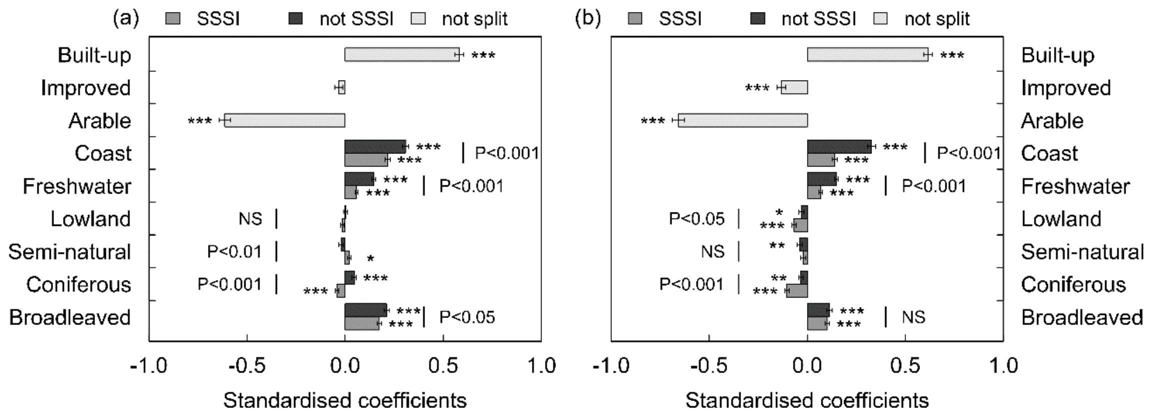

Fig. S2.1. Effects on visitation probability of land covers separately outside or inside an SSSI, showing standardised coefficients from (a) stratified control model (b) random control models (controlling for path length, elevation, distance to nearest major road, weighted population and county in both); bars denote standard error. P<0.001 '***', P<0.01 '**', P<0.05 '*'. Z-tests comparing pairs of coefficients of the same land cover type are shown next to each pair (NS denotes non-significant)

Justification for choice of buffer radius within which to extract characteristics of the area surrounding visit and control points

Mean 'penetration distances', defined as the 'linear distance from the mid-point of the route to the access point' (Liley et al. 2005) were 777m for dog walkers (n=3), and 795m for walkers (n=2), for three heathland complexes in southern England. As these are large protected areas (mean size 5421 ha, range 2742-8400 ha), they are not likely to be representative of the visit behaviour captured in this study as the majority of visits will take place locally (over 80% of respondents reported travelling less than 5 miles to their destination). Mean penetration distance reduces consistently with a reduction in site area (Clarke et al. 2006) which lends support for a smaller buffer size for this study. Hence a buffer of 400m radius was chosen with which to extract characteristics of the area visited. Buffering the point representing visit locations will also account for any uncertainty in the initial mapping by surveyors, as although precise (1m) grid references were provided, often respondents could only give a description of where they went.

Validation of path network layer

For validation of the path network layer (OpenStreetMap n.d.), 1 km grid squares were randomly placed in East Anglia (n=100) and the North East (n=100) and the OpenStreetMap path network was visually compared to paths mapped by Ordnance Survey (OS) VectorMap® (definitive but not available in a useable digital format) within each grid square. In East Anglia 87% of grid squares had fully corresponding paths and in the North East 72%, thus it was concluded that the path network layer was a good representation of reality.

Determining best fitting distance-weighted population function

To obtain inverse distance-weighted population per site, 1 km resolution Euclidean distance rasters with a 10 km radius around visit and control points were generated and multiplied with the population raster as per eqn 1-3. A 10 km radius was chosen as 82% of respondents reported travelling less than 5-8 km. Three different distance weightings were applied to determine the best function for these data (equations 1-3):

weight.pop.1 = pop * (1/d) AIC = 104,393 (eqn 1)

weight.pop.2 = pop * (1/d^2) Δ AIC from eqn 1 = -4,973.9 (eqn 2)

weight.pop.3 = pop * (1/$d^{2.5}$) Δ AIC from eqn 1 = 9,760.5 (eqn 3)

Where pop = number of people per km^2

d = distance from focal 1 km x 1 km cell that the visit or control point lies within, calculated as d = (centroid distance from focal cell centroid (m) + 1000)/1000 so that the maximum weighting of 1 was assigned to the focal cell

To determine the best fitting function these three distance-weighted population metrics were entered separately into univariate generalised linear models (GLMs) with logit link and binomial error structure, to estimate visitation probability (using visit and control localities as a binomial response variable) as a function of surrounding distance-weighted population. Weight.pop.2 provided the best fit to the data based on AIC and thus was included in all subsequent multivariate models.

To check if built-up area shared some explanatory power with weight.pop.2, the proportion of built-up area within localities (i.e. 400m buffers) was added to a model with only weight.pop.2. Upon addition, the weight.pop.2 coefficient was reduced from 1.97 ± 0.02 SE to 0.99 ± 0.02 SE, suggesting built-up area partly acted as a proxy for source population. However, they were only moderately correlated (r = 0.48, df = 94500, P < 0.001) and thus built-up was retained in subsequent models.

Checking for spatial autocorrelation

Correlograms of the spatial dependence between residuals from GLMMs (as estimated by Moran's I) were examined to check for spatial autocorrelation. Correlograms were plotted using the ncf package in R (Bjornstad 2013) (Fig. S2.2). Due to computational limitations, random samples of 10% of the data were used to generate correlograms (repeated three times per model for robustness). Inspection of correlograms led us to conclude that spatial autocorrelation was negligible and supported the use of non-spatial GLMMs.

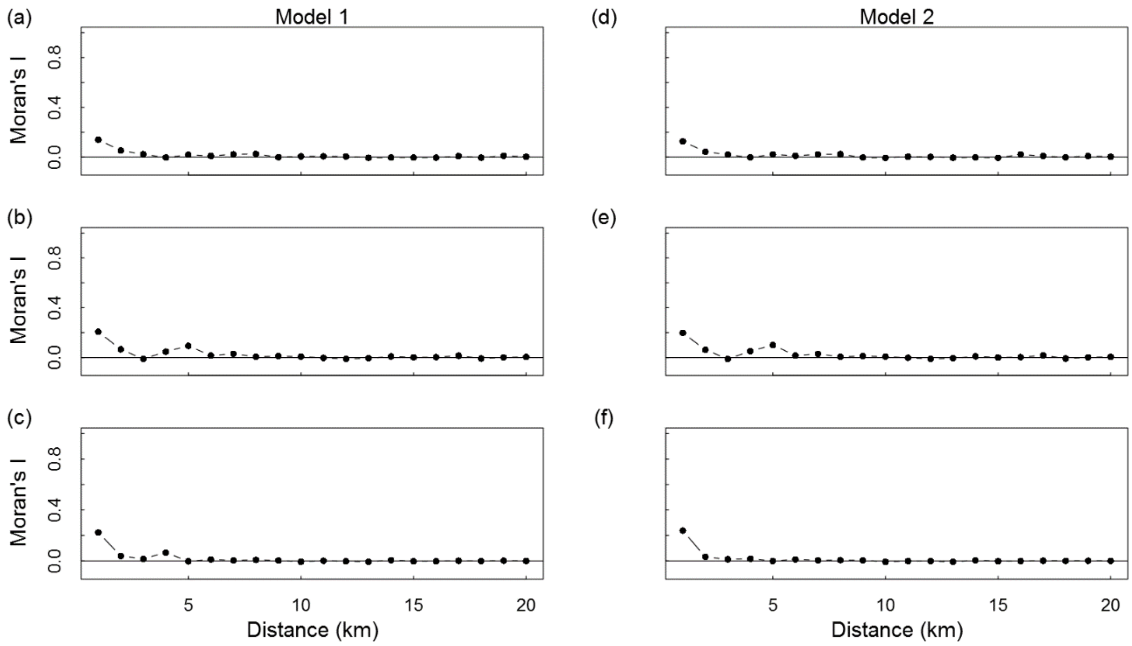

Fig. S2.2. Correlograms from residuals from (a-c) model 1 which does not consider designation status and (d-f) model 2 where land cover classes are divided into SSSI and non-SSSI (created using three random samples of 10% of the data i.e. 9465 observations due to computational intensity)

Supplementary references

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Appendix S2.2. Supplementary figures

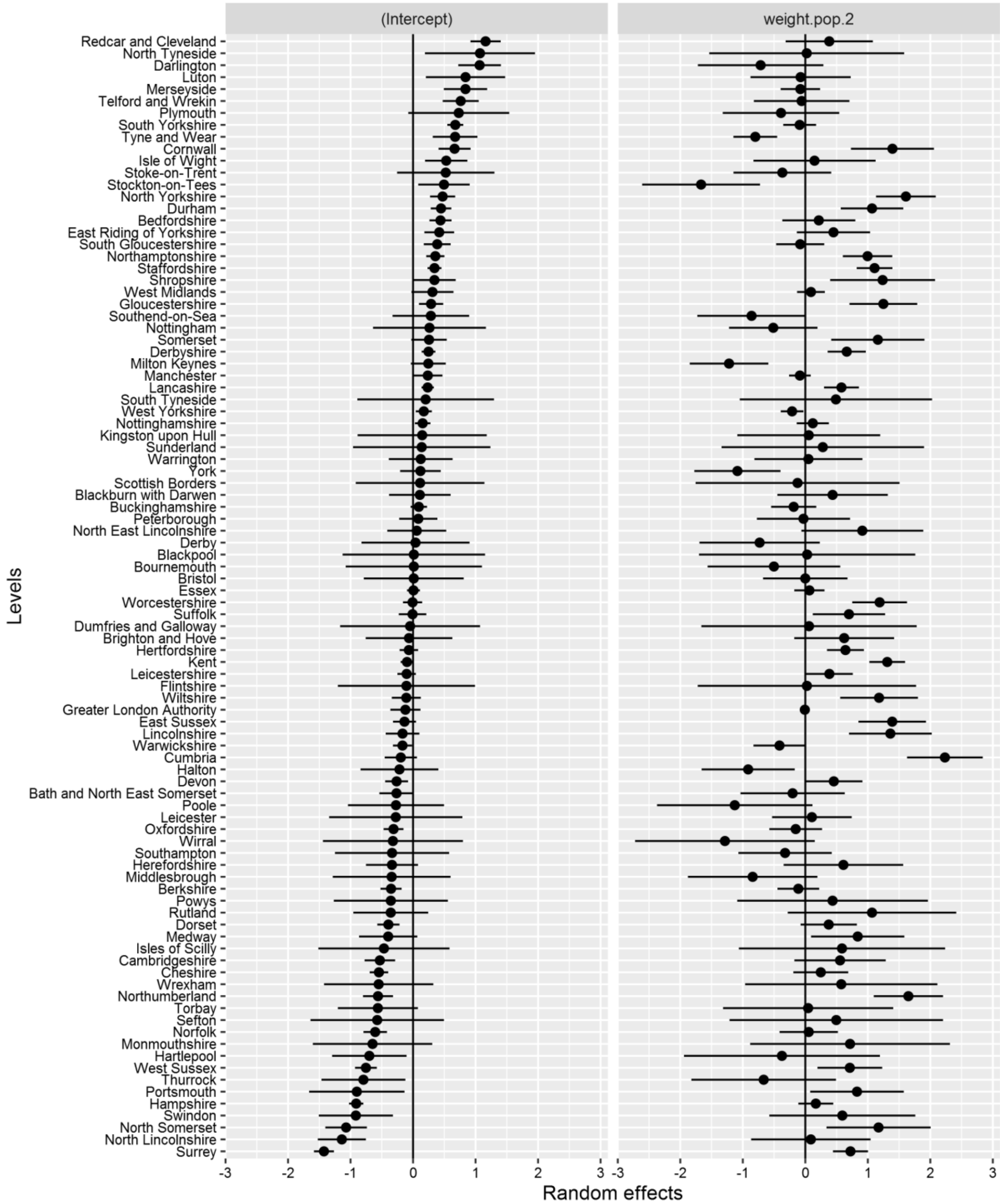

Fig. S2.3. Point estimates (filled circles) for random effect parameters (intercept and weight.pop.2) with 95% confidence intervals (bars) per county. Intercept = additive effect of county, weight.pop.2 = coefficient of effect of distance-weighted population. Intercepts are taken to represent probability of visitation by visitors outside the 10km radius of sites (for which ‘local population’ i.e. weight.pop.2 was measured).

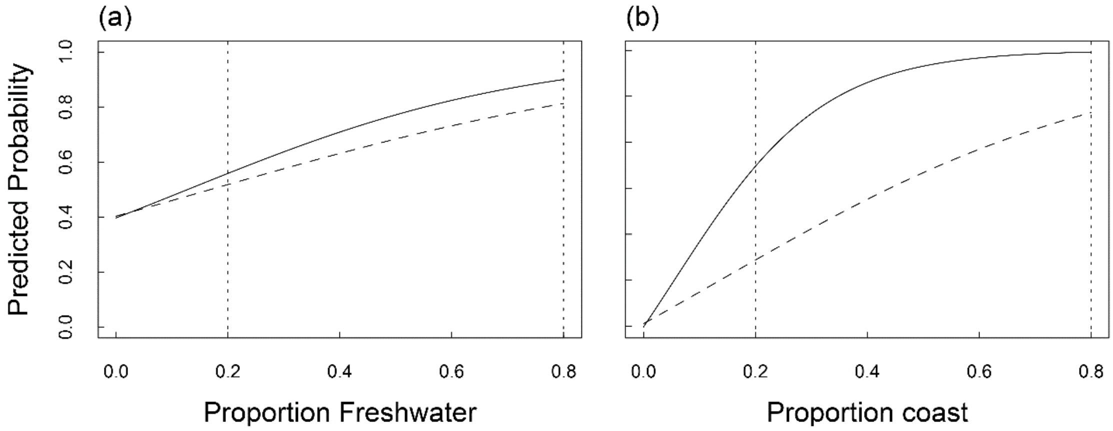

Fig. S2.4. Predicted influence on visitation probability of (a) freshwater and (b) coast when SSSI designated (dashed line) and non-designated (solid line), from model 2 (see text) that controls for path length, elevation, distance to nearest major road, weighted population and county. Predictions were obtained by varying the proportion of the land cover type shown either as designated or non-designated, setting the alternative designation status at zero. All other land cover classes were held proportional to their mean such that they sum to 0.2 (so that total land cover proportion did not exceed 1). Control variables were held at their mean.

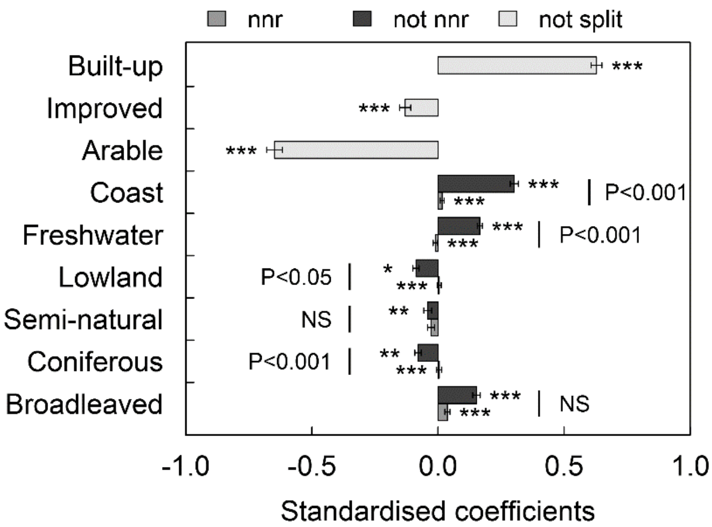

Fig. S2.5. Effects on visitation probability of non-NNR-designated or NNR-designated land covers, showing standardised coefficients from a GLMM controlling for path length, elevation, distance to nearest major road, weighted population and county. Bars denote standard error. For each land cover, P values of Z-tests compare pairs of coefficients between non-NNR-designated/NNR-designated (P<0.001 '***', P<0.01 '**', P<0.05 '*')

Chapter 3

Mapping current and future recreational demand in rural England

Abstract

Across Europe recreational use of the natural environment is increasing in popularity and together with increasing population the volume of recreational visits is likely to rise. England is a case in point, with a varied landscape receiving an estimated 2.5 billion recreational visits per annum and experiencing the fifth largest population increase in Europe. As recreational activities can negatively impact the four major landscape components (soils, vegetation, animals and water) it is important to understand how recreationists are dispersed across the landscape, whether they coincide with important reserves for native habitats and wildlife, and how such patterns may change in the future. The aim of this study was to investigate spatial patterns of recreational demand for different habitat types and protected areas using a model to predict probability of visitation based on land cover composition and statutory designation. We mapped the estimated volume of annual recreational visits at over 800,000 sites covering the entire English countryside, scaling up from probabilities using Natural England's regional visit estimates. Using local authority population projections for the year 2030 and regional per capita annual visit rates we then estimated how the spatial pattern and volume of visits would change in the future. The map of current visits conformed to prior expectations, however, attractive protected landscapes were not attributed high numbers of visits, as 'nature tourism' was underrepresented in the calibration data. Furthermore, the resolution of the visit map proved too course to garner reliable summary statistics for Sites of Special Scientific Interest. The future visit map produced from population projections did not reveal fine scale changes in the distribution of recreational visits, however land use change (urban expansion) was not incorporated. Future work could involve obtaining strategic growth locations at a smaller scale (e.g. county) to investigate a scenario of urban growth and associated land use change.

3.1. Introduction

Across Europe public appreciation of natural landscapes has become commonplace (de Vries, Klein-Lankhorst & Buijs 2007; Vizzari 2011). Assessing the provision, accessibility and quality of natural areas or 'green space' for recreational use has grown as a research area in recent years (Barbosa *et al.* 2007; Zhang, Lu & Holt 2011; Kienast *et al.* 2012), reflecting the growing importance of nature-based recreation to society. Recreational use of the natural environment is ever changing, developing and increasing in popularity, leading to complex issues and conflicts of interest (Bell *et al.* 2007). With an upsurge in demand for outdoor recreation as well as increasing population in 15 out of the 28 EU member states (Eurostat 2014) the volume of recreational visits will rise in many European countries. Within a country, areas with a higher relative increase in population or where there is already a greater propensity to partake in outdoor recreation may experience greater proportional increases in recreational visits.

The English landscape is varied and offers outdoor enthusiasts many opportunities to partake in recreational activities. The annual visit rate to the natural environment in England was estimated at 68.7 visits per capita, resulting in around 2.5 billion recreational visits per year (Natural England 2012a). England is a densely populated country and in addition is experiencing the fifth largest population increase in Europe (Eurostat 2014). England is also home to many species and habitats of international and European importance, with 7.6% of the country designated as terrestrial Sites of Special Scientific Interest, 9% as National Parks and 15% as Areas of Outstanding Natural Beauty (Natural England 2012b). Recreational activities can negatively impact the four major landscape components, namely soils, vegetation, animals and water (Cole 1993). It is therefore important to understand how recreationists are dispersed across the landscape and how such patterns may change in the future. This would enable the identification of areas with high concentrations of recreational visitors and whether they coincide with important reserves for native habitats and wildlife. Such areas would require careful planning and management to reconcile the needs of recreationists with conservation objectives.

Previous research has focused on mapping hotspots of highly attractive natural places (de Vries *et al.* 2013), recreation opportunity (Joyce & Sutton 2009) and spatial distribution of recreationists around towns (Kienast *et al.* 2012). However, there has been no attempt at large scale mapping of recreational pressure, linking the distribution of recreationists across the landscape with the consequent implications for conservation. A conservation-focused, spatially-explicit, national-level study on recreational use of the natural environment would be beneficial for planners, revealing the current pressures on habitats and conservation areas. A spatially explicit model of recreational pressure would provide a means to assess the conservation implications of changes in policy, population growth or large scale land use change.

Sen *et al.* (2014) examined how the distribution of recreational visits in Great Britain would change under future scenarios of land use and changing socioeconomic and demographic characteristics. They predicted very different changes resulting from a range of scenarios that differed their relative emphasis on conservation, economic growth and self-sufficiency as driving forces for change (Haines-Young, Paterson & Potschin 2011). However Sen *et al.* (2014) present and discuss the consequences in economic terms, mapping the change in monetary values of visits, rather than national recreational pressure, with no discussion of the potential significance for conservation.

The aim of this study was to map the estimated volume of annual recreational visits across England according to empirically derived preferences for land cover composition to investigate spatial patterns of recreational demand for different habitat types and protected areas. We applied a national-level recreation model that includes land cover types and controls for transport and footpath networks, source population density and regional behavioural differences to predict the probability of a recreational visit occurring at over 800,000 sites covering the entire English countryside. Using these site probabilities we redistributed an estimate of the total volume of recreational visits in England in 2011 amongst cells in a 400 m resolution raster to map fine scale visit estimates across the country. We then estimated how the spatial pattern and volume of visits would change under a population projection for the year 2030.

3.2. Methods

3.2.1. Recreation model

The forces driving recreationists' choice to visit particular areas of the countryside over other available areas are numerous. To explore this, in Chapter 2 we developed a recreation model comprising of various site characteristics *a priori* selected as having an influence on where recreationists choose to go as well as site land cover composition in order to determine relative preference for certain land cover types. Briefly, observed countryside visit sites were compared with control sites placed randomly in the available countryside (hereafter collectively termed 'sites'). Visit sites were acquired from Natural England's Monitor of Engagement with the Natural Environment (MENE) household survey (Natural England 2012a), in which a visit was defined as any time spent outdoors close to home, work or while on holiday in England, in the week preceding a household interview, excluding for shopping or time in your own garden. Sites were point locations that were buffered by 400 m to obtain '(potential) visited area'; informed by empirical evidence on visitor 'penetration distances' at various locations in England (see Appendix S2.1). A generalised linear mixed model (GLMM) with binomial error distribution and logit link function predicts the probability of visitation to sites across England as a function of land cover composition and accessibility (fixed effects) and source population and county (random effects). Source population was measured as the number of residents within 10 km of a site, inverse weighted by squared distance to account for the exponential decay in the number of people visiting a site the further away they live. Source population, hereafter termed distance-weighted population, was entered as a random effect in the model, along with county, to allow for variations in per capita visit rate between counties thereby accounting for socioeconomic and demographic differences. There was no issue of multicollinearity between predictors (see Table S3.1 in Supplementary Information). The explanatory variables used are nationally available so that the model can be used to predict the probability of a recreational visit to all sites across England under current and future population estimates. A more detailed description of these methods can be found in Chapter 2.

The recreation model in Chapter 2 was adapted slightly with the addition of a 'scenic destination' variable. This is the proportion of a site's area designated as Areas of Outstanding Natural Beauty (AONB) and National Parks (NP), which was *a priori* considered to be an important predictor of a recreational visit; protected areas (PAs) are popular tourism and recreation destinations as demonstrated by the global rising trend in numbers of PA visits (Balmford *et al.* 2009). Also some land covers (broadleaved woodland and semi-natural grassland) that were split into proportion inside and outside conservation areas (Sites of Special Scientific Interest; SSSI) in Chapter 2 were not split here as there was no significant difference in their probability of visitation. The model was evaluated using AUC (the area under the receiver operating curve) in order to assess predictive ability. Recreational visits from the subsequent MENE survey (2012-2013; n = 10,622) and a new set of random controls (n = 10,622) were used to calculate AUC using the pROC package in R (Robin *et al.* 2011).

3.2.2. Extrapolation across England

To map recreational pressure across England we extracted the explanatory variables for all possible sites across England. These sites were represented as points placed within the boundaries of England in a regular grid pattern spaced 400 m apart, resulting in 813,405 points. The 400 m spacing was chosen as once buffered by 400 m (to obtain 'visitable area') there would be some overlap (preferable to gaps created by non-overlapping circular buffers), and the data set was manageable computationally. The sites in our model calibration sample were subset to include only sites in the countryside - we classed countryside sites as sites containing less than 70% built-up area in the 400 m buffer. Consequently, sites were removed from the England dataset that contained greater than 70% built up area within buffers, resulting in 773,028 sites. The recreation model was then used to predict the probability of visitation using the predict function in the lme4 package (Bates *et al.* 2013).

3.2.3. Converting to actual visits

To scale up from probabilities of visitation to actual numbers of visits, we used Natural England's estimate of the total volume of visits taken to the natural environment by the English adult population in 2011 (Natural England 2012a). The estimate was generated by taking into account the number of trips reported by each respondent (taken in the seven days preceding the interview), multiplied by the respondent's demographic weight (applied to take into account varying response rates among different population groups) and multiplied by a Calendar Month Factor (total days in the reporting month divided by seven; Natural England (2012a). This gave an annual per capita visit rate to multiply by total population. The total volume of visits is estimated for all 9 regions in England and given in Appendix 4 of the MENE technical report.

As mentioned, the MENE visit estimates are for all visits to the natural environment, whereas we exclude visits in predominantly urban areas (see section 3.2.2). To determine the estimated number of 'countryside' visits we calculated the proportion of visits in the MENE visit data remaining after the filtering by built-up area for each region. This gave a range of proportions from 0.46 of all visits in London being countryside visits, to 0.78 of all visits in the East Midlands. The MENE regional annual visit estimates were multiplied by their respective regional proportion of countryside visits to obtain regional estimates of the annual number of countryside visits. To map the within-region spatial distribution of countryside visits, visit estimates were applied to our site-level predicted probabilities (P) in 2011 as per equation 1.

$$\frac{P\mathrm{ij}}{\sum P\mathrm{ij}} * \text{Total countryside visit estimate for region}_{\mathrm{j}} \qquad \text{(eqn 1)}$$

Where i = site (n = 773,028 across all regions)

j = region (n = 9)

The resulting estimated numbers of visits per site were mapped in ArcGIS represented as a 400 m resolution raster. Although the sites were originally 400 m spaced points that were buffered by 400 m and therefore adjacent points' buffers overlapped, it is not possible to display this in raster format. Hence conversion of points to a 400 m raster was considered a suitable representation.

3.2.4. Relative pressure on habitats

The map of current visit estimates was subdivided into total volume of visits by habitat type using the 25 m resolution land cover map LCM2007. The 400 m resolution visit map was resampled to 25 x 25 m to overlay with LCM2007 using nearest neighbour assignment i.e. each of the 256 25 x 25 m cells resampled from a 400 x 400 m cell were assigned the value of their parent cell, and were then divided by 256 to maintain the original total. This method distributed visit estimates from 400 m cells evenly among the 25 m land cover cells within them, as we had no information on where within a 400 m cell a recreationist would visit. The zonal statistics tool was then used to obtain the sum of visits per habitat using LCM2007 to summarise the resampled 25 m resolution visit raster.

3.2.5. Predictions for future population projection

We re-estimated the probabilities of visitation for all sites across England using an updated distance-weighted population variable, calculated using the 2012 based subnational population projections for 2030. These were obtained from the Office for National Statistics (England), the Welsh Assembly Government and the General Register Office for Scotland covering the 326 local authority districts in England, 22 local authorities in Wales and 32 council areas in Scotland. They represent the highest resolution data available for population projections in each of the countries, which does not take into account within-district movement to peri-urban areas or between towns and villages due to changes in working patterns, transport costs and facilities, etc. The cohort component method is used to produce these projections, which takes an existing estimate of the population, ages on the population by one year and adjusts for births, deaths and migration to calculate the population at the end of the year (Office for National Statistics 2010). Differences arise in the assumptions per country (Office for National Statistics 2010), thus there are some minor inconsistencies near the borders. Also, as projections are produced at the local authority level there are abrupt differences in population change at county boundaries. These projections assume that recent demographic trends will continue and do not take into account potential effects of housing developments, policy change or wider socio-economic change (Office for National Statistics 2010). Therefore the projections are used here to explore the possible consequences on the volume of recreational visits of an increased population following the demographic assumptions made by the constituent countries of Great Britain.

A raster of UK population in 2030 was created with which to extract the distance-weighted population surrounding each site for making future predictions. The subnational population projections database was joined to a shapefile delineating UK district, borough and unitary boundaries (OS boundary line) and the percentage population change from 2011 to 2030 per area unit was calculated with which it was converted to a 1 km raster (Fig. 3.1). The per 1 km cell change in population was calculated by multiplying the 2011 population raster by the percentage population change raster, giving a range from 773 fewer, to 4,388 additional people per 1 km cell. To create the final 2030 population raster, the per cell population change raster was added to the 2011 population raster. The distance-weighted population surrounding each site in England was then re-calculated as for the sample data set and used with all other input variables unchanged to re-estimate probability of visitation based on local authority level projected population in 2030. This assumes that land cover will remain unchanged over this period.

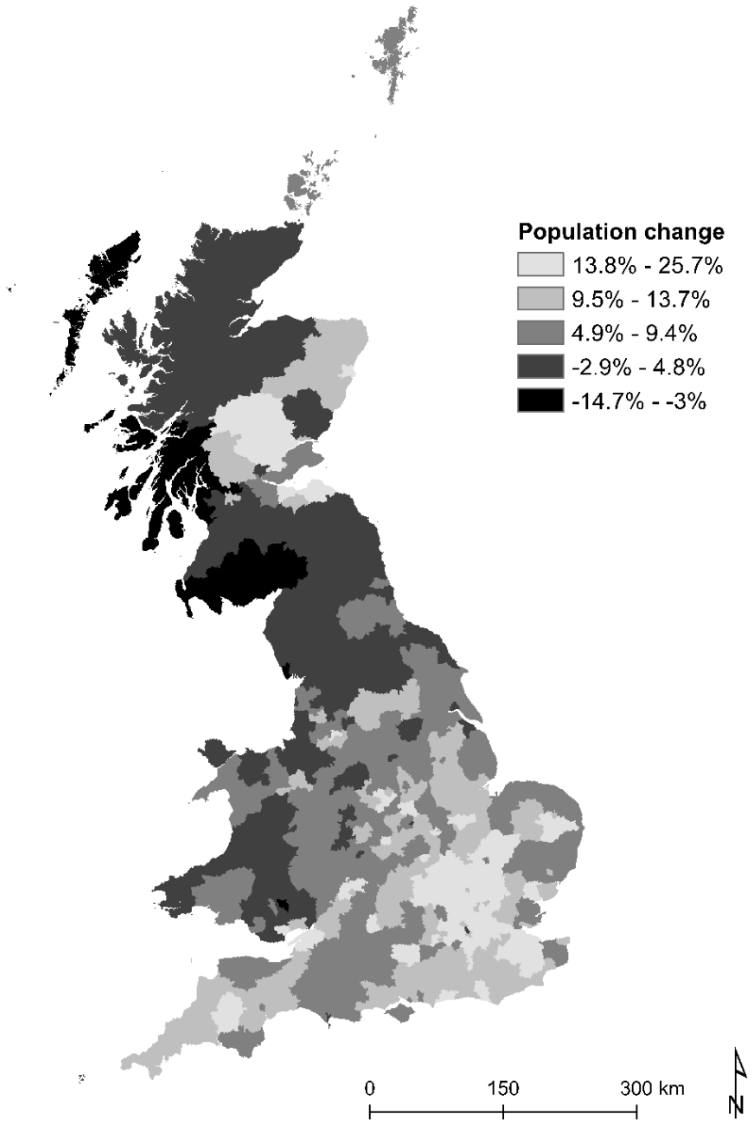

Fig. 3.1. Percentage projected population change per local authority in Great Britain from 2011 to 2030. Used to adjust the 2011 1 km population raster to create a raster of population in 2030.

To obtain future countryside visit estimates for 2030 (i.e. to update the MENE visit estimates according to future population) we multiplied the MENE annual per capita regional visit rates by projected 2030 regional population and proportion of visits per region classed as countryside visits. This assumes that visit rates remain the same in 2030. To obtain and map the estimated visits per site in 2030 we followed equation 1 above, substituting the predicted probabilities for 2011 with predicted probabilities for 2030 (obtained using the updated distance-weighted population variable), and multiplying by the 2030 regional countryside visit estimates.

3.3. Results

3.3.1. Recreation model

The recreation model has good predictive ability, with an AUC score of 0.84 obtained using independent data. This was considered satisfactory to predict probabilities of visitation to out of sample countryside sites across England. All fixed effect variables were significant except for scenic destination which was however retained, as *a priori* these areas were considered attractive recreation destinations (Table 3.1). Intercepts for each county ranged from -1.41 ± 0.16 95% CI to 1.11 ± 0.24 95% CI and distance-weighted population coefficients from -1.68 ± 0.94 95% CI to 2.14 ± 0.61 95% CI, showing variation in per capita probability of visitation between counties.

As we are investigating the effect of changing population, it is important to explore possible collinearity between distance-weighted population and other variables. The proportion of built-up area was in part accounting for local source population as there is moderate correlation between distance-weighted population and built-up area (Pearson's $r = 0.48$, df = 94,600, $P < 0.001$). To further test whether built-up area shared some explanatory power with distance-weighted population we looked at coefficients in models with distance-weighted population as a fixed effect with and without built-up in the model, and obtained predictions from a GLMM both with and without built-up area in the model to see what difference this made to predictions (see Appendix S3.1 in Supplementary Information). There was negligible difference between predictions from these models and the original model that incorporated distance-weighted population as a random effect, hence the model presented in Table 3.1 is used for all subsequent predictions.

Table 3.1. Fixed effect coefficients from the recreation model predicting recreational demand in the countryside (random effects: distance-weighted population and county). Dependent variable: probability of visitation. P<0.001 '***', P<0.01 '**'

	Standardised Coefficient	Std. Error	z	P
Non-land cover variables				
Path length	0.825	0.014	59.51	***
Elevation	-0.385	0.017	-22.11	***
Distance to major road	-0.134	0.013	-9.98	***
Scenic destination	0.023	0.013	-1.86	
Land cover classes with positive effect				
Built-up	0.617	0.022	28.35	***
Coast (non-SSSI)	0.327	0.021	15.66	***
Broadleaved	0.150	0.015	9.79	***
Freshwater (non-SSSI)	0.147	0.010	14.95	***
Coast (SSSI)	0.140	0.012	11.94	***
Freshwater (SSSI)	0.069	0.008	8.56	***
Land cover classes with negative effect				
Arable	-0.656	0.031	-20.9	***
Improved	-0.134	0.022	-5.98	***
Coniferous (SSSI)	-0.099	0.010	-9.77	***
Lowland heath (SSSI)	-0.068	0.011	-6.04	***
Semi-natural	-0.045	0.016	-2.84	**
Coniferous (non-SSSI)	-0.036	0.012	-2.97	**
Lowland heath (non-SSSI)	-0.033	0.013	-2.6	**
Constant	-0.683	0.078	-8.86	***

3.3.2. Nationwide distribution of current recreational demand

The recreation model and Natural England's regional visit estimates (totalling 1,924,123,785 visits per annum after excluding visits to predominantly urban areas) were used to predict the annual number of visits to all 400 x 400 m cells across the English countryside. The number of visits per cell ranged from 11 to 38,381 per annum, with a mean of 2,404 (Fig. 3.2a). There is an obvious pattern of high visit numbers radiating out from large urban centres (see labelled examples in Fig. 3.2a). There are clear influences of habitat effects (for example higher probabilities of visiting the coast; Fig. 3.2b) and also an elevation effect, with upland areas in the north of England having lower visit numbers (Fig. 3.2c). However, the area in Fig. 3.2c is a national park (the Lake District National Park), a popular recreation destination, but the lack of significance for the scenic destination variable coupled with lower local population density and higher mean elevation causes the model to produce lower visit numbers than the area should have in reality.

Two areas were chosen to inspect estimated visit patterns at a finer scale, an urban fringe and a village with surrounding countryside (Figs 3.3a and b). Estimated visit numbers were lower on agricultural land compared to the woods and green spaces closer to the city in Fig. 3.3a and woodlands and lakes clearly elevated visit numbers in Fig. 3.3b, hence the national-scale model performed well at predicting fine-scale distribution of visits within local landscapes. There are some anomalies in visit estimates, however; for example, high visit numbers were predicted along an agricultural drainage ditch due to the presence of water, which has a positive association with probability of visitation (Fig. 3.4a), abrupt changes in visit numbers at county boundaries due to county specific intercepts and distance-weighted population coefficients allowing for differences in recreational propensity (Fig. 3.4b) and obvious effects of the digital elevation model (Fig. 3.4c).

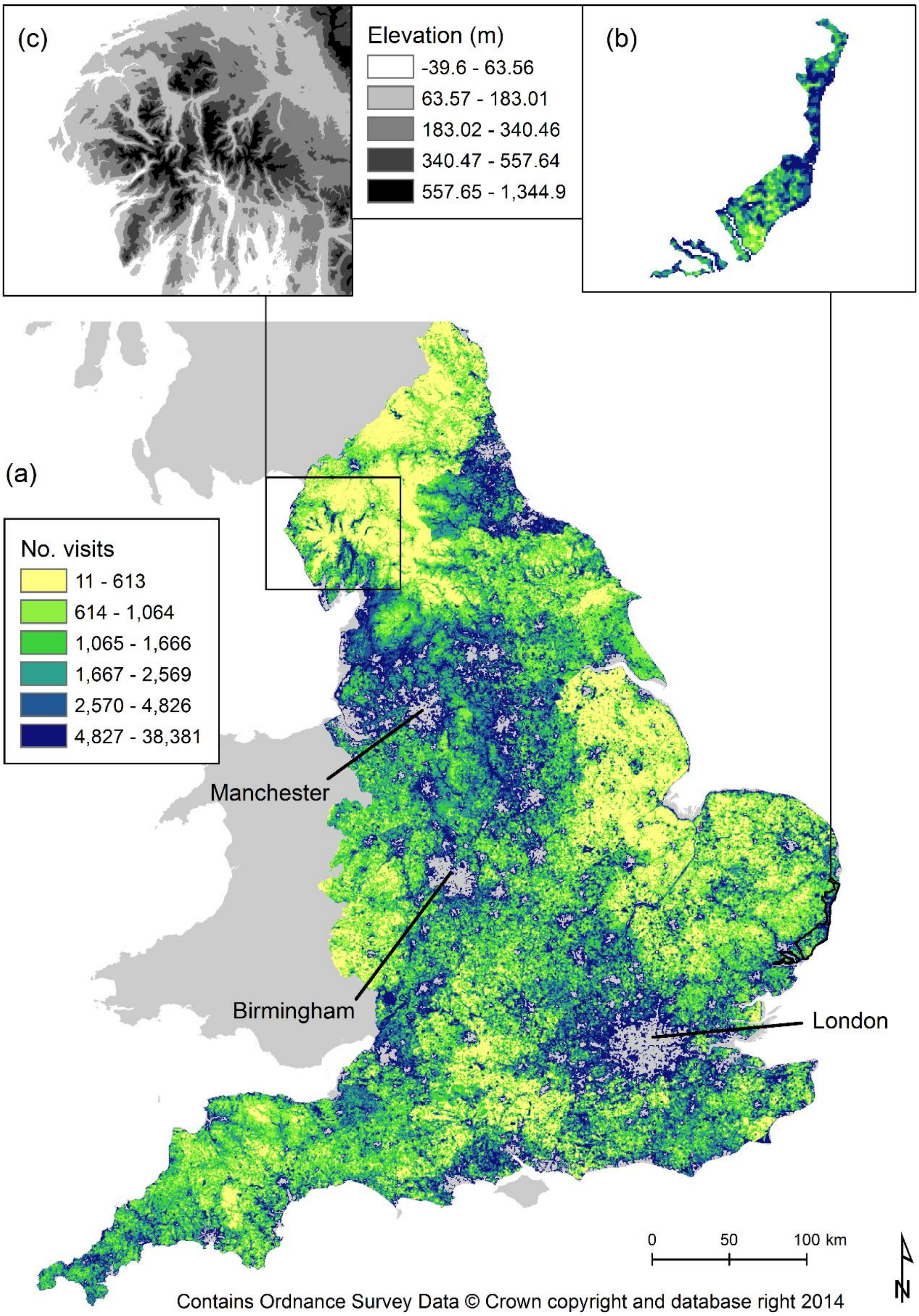

Fig. 3.2. Spatial distribution of predicted annual number of visits in 2011 to 400 m grid cells (a) in England (b) at the Suffolk Coast, and (c) an example of an upland (high elevation) area with predicted low numbers of visits.

(a) (b)

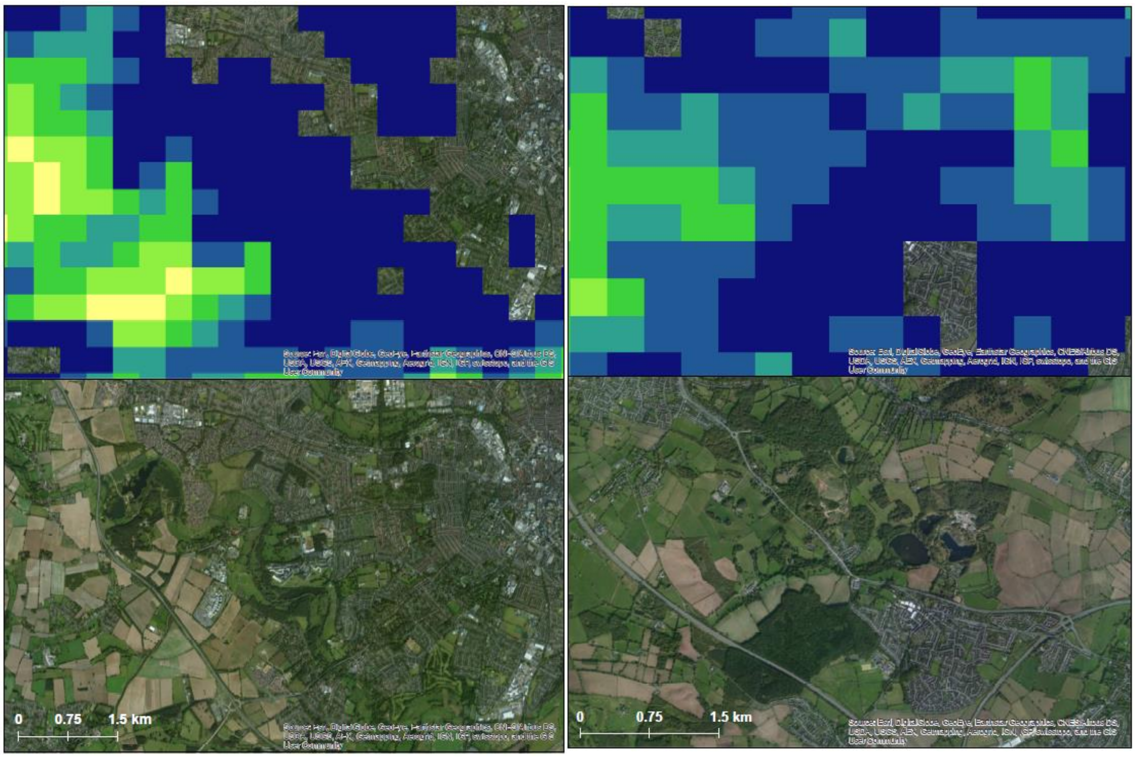

Fig. 3.3. Visit estimates (upper images) and Google Earth imagery (lower images) for two example areas (a) an urban fringe (b) a small town. See Fig. 3.2 legend for estimated visit numbers.

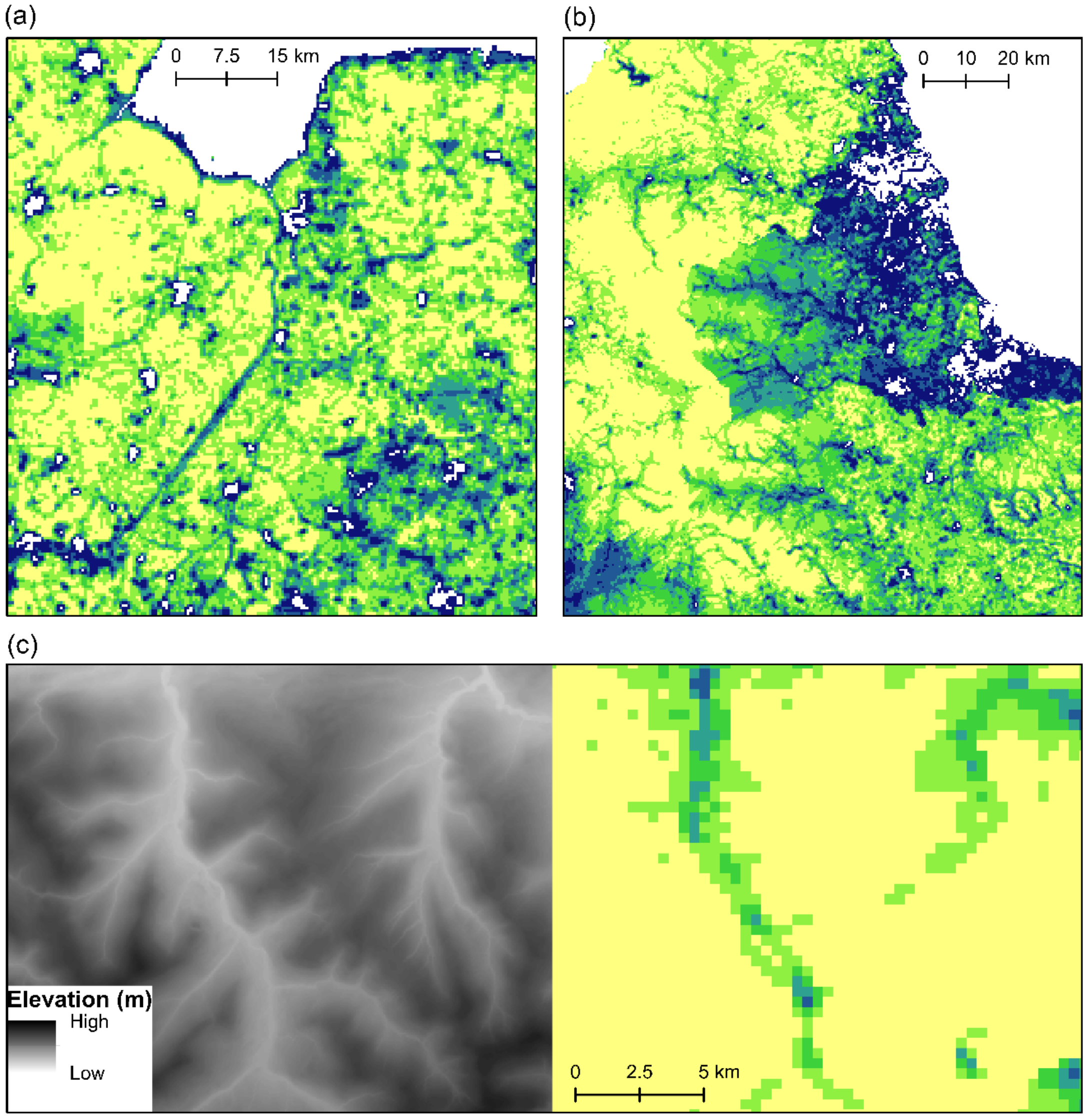

Fig. 3.4. Example of (a) an agricultural drainage ditch (blue line across the centre), (b) abrupt changes in visit estimates at county boundaries (c) influence of digital elevation model (right) on visit estimates (left). See Fig. 3.2 legend for estimated visit numbers.

3.3.3. Recreational demand on habitats

The annual number of visits per hectare of each habitat type largely conforms to expectations based on the identified preferences from the recreation model (Fig. 3.5). Arable and improved grassland in particular received far fewer visits ha^{-1} than other land covers despite having a greater coverage across England. The attractiveness of coast and freshwater is evident, as these have the highest estimated visit densities.

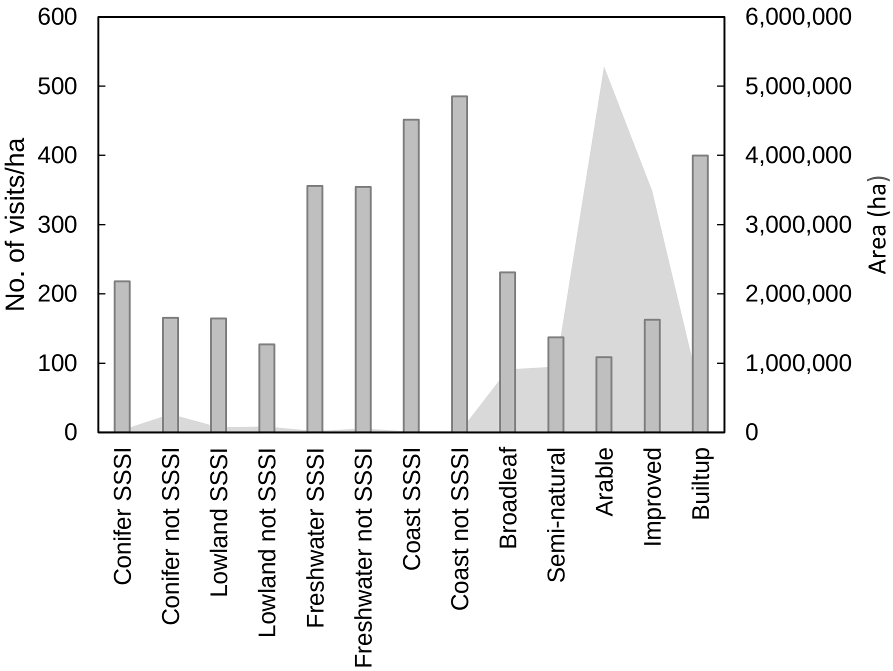

Fig. 3.5. Predicted number of visits per hectare (bars) per land cover type in 2011 and area (ha) of each land cover type in England (light grey fill).

3.3.4. Future distribution of recreational demand

We first calculated regional countryside visit estimates in 2030, based on Natural England's visit rate per capita, proportion of visits to the countryside and predicted population in 2030 (Table 3.2). The proportion of visits to the countryside is similar between regions at around 70%, except for London which is lower at 46%, whilst the per capita visit rate is more variable between regions (Table 3.2). There was an estimated 50% increase in total visits, from 1,924,123,785 in 2011 to 2,890,335,003 visits in 2030 predicted as a consequence of a 13.4% increase in total population, accounting for the visit rate per capita and proportion of visits that are to the countryside. The increase in visits was unevenly distributed amongst regions; London had the highest increase (107.7%), whilst the North East had the lowest (18.5%; Table 3.2). As London has the lowest visit rate per capita and lowest proportion of countryside visits, the large increase in visits is driven by the large initial population and large proportionate increase in population. The percentage increase in visits across all regions largely corresponds to the percentage increases in population, with visit estimates only modified slightly by per capita visit rates (Table 3.2).

We then redistributed these visit estimates throughout their respective regions using the predicted probability of visitation in 2030 per 400 m cell obtained using updated distance-weighted population from local authority level population projections. We predicted an increase of between 4 and 41,286 (mean 1,162) recreational visits per 400 m cell across the English countryside (Fig. 3.6a). The regional visit estimates clearly had the strongest effect on predicted number of visits in 2030, as the mapped per cell changes in visits from 2011 to 2030 (Fig. 3.6b) match very closely the overall regional change (Table 3.2; regional boundaries shown in Fig. 3.6c). The finer-scale local authority level population projections used in re-estimating probability of visitation from the recreation model are less evident and the spatial distribution of visits in 2030 are almost the same as 2011 within each region, just the absolute number is higher.

Table 3.2. Estimated regional countryside visits in 2030 (the product of per capita visit rate, proportion of countryside visits and population in 2030 per region), proportionate increase in population and proportionate increase in number of visits from 2011.

Region	Visit rate per capita	Proportion of visits to countryside	Population in 2030	Estimated countryside visits in 2030	Increase in population from 2011	Increase in visits from 2011
London	44.2	0.46	10,101,000	206,947,384	0.24	107.7
East Anglia	81.7	0.77	6,801,000	427,542,820	0.16	67.8
South East	76.6	0.71	9,922,000	540,956,208	0.15	66.4
East Midlands	74.1	0.78	5,075,000	293,881,618	0.12	43.2
South West	97.8	0.74	5,992,000	435,634,857	0.13	40.8
North West	55.8	0.71	7,524,000	299,407,843	0.07	39.8
Yorkshire	69	0.74	5,774,000	293,915,455	0.09	39
West Midlands	55.1	0.7	6,184,000	238,756,837	0.10	36.5
North East	75.6	0.74	2,733,000	153,291,980	0.05	18.5

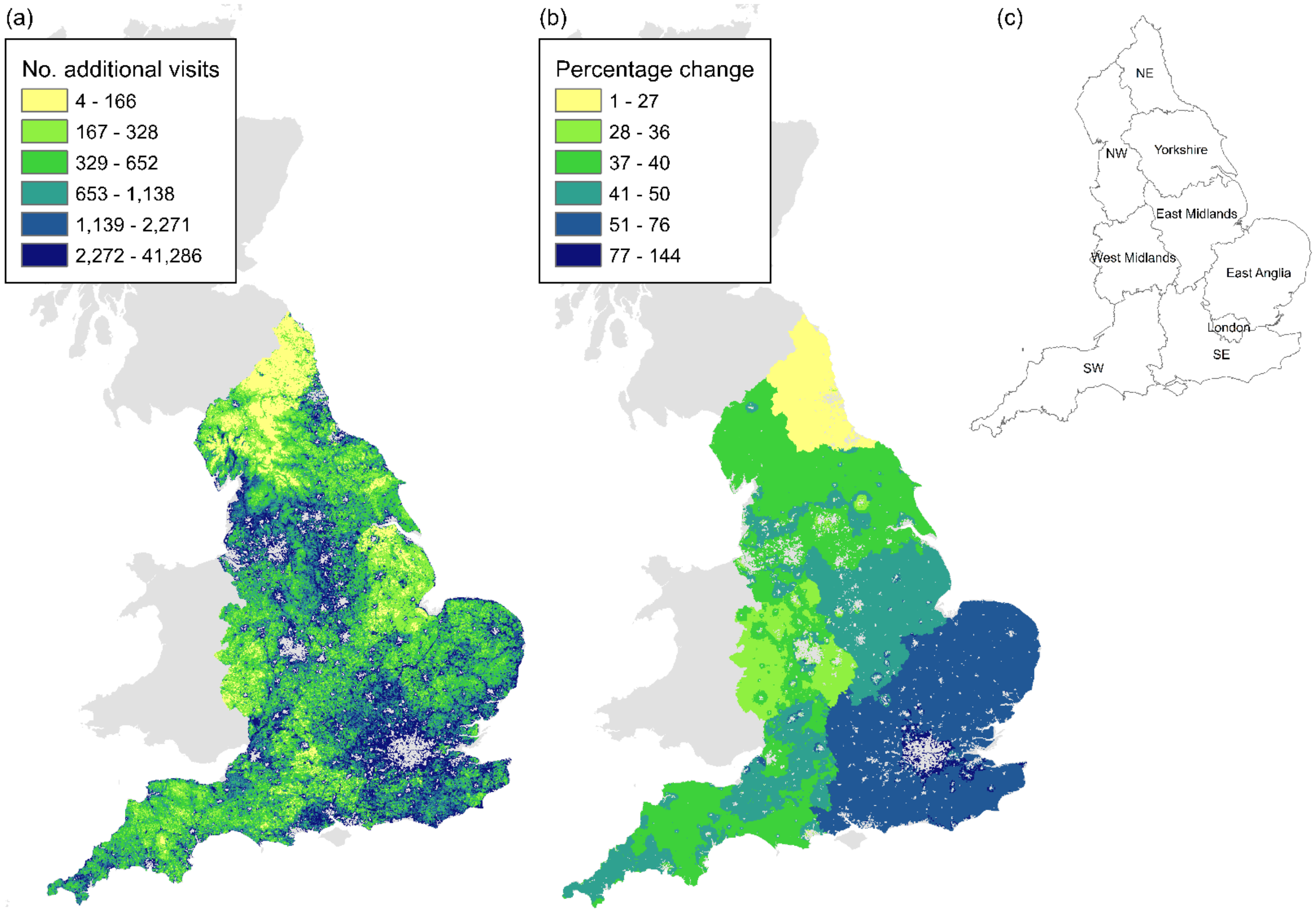

Fig. 3.6. (a) Predicted number of additional visits and (b) percentage change in visits per 400 m grid cell in 2030 from 2011, (c) regional boundaries in England.

3.4. Discussion

Using the recreation model developed here that identified relationships between site characteristics and probability of visitation we mapped current estimates of recreational visits to the English countryside. The spatial distribution of visits conforms to prior expectations, as visit estimates are higher close to urban areas and more densely populated cities, which is consistent with recreation studies from the field of environmental economics (Brainard, Bateman & Lovett 2001; Hill & Courtney 2006; Jones *et al.* 2010; Sen *et al.* 2014). In these studies, size of local population and travel time from origin to site were significant predictors of recreational demand. Preferable habitats such as beaches and woodlands received higher visits than less preferred habitats, as expected. However, attractive landscapes, such as the Lake District National Park, were not attributed high numbers of visits, yet in 2015 the Lake District National Park was estimated to have received 17.32 million tourists over 25.21 million tourist days (Cumbria Tourism 2016). It is likely that landscape designation (‘scenic destination’) was not significant in the recreation model because the calibration sample was based on visits undertaken within the seven days preceding the survey (Natural England 2012a) and therefore longer-distance trips to touristic destinations were underrepresented. The Lake District, for example, is too far to drive for a day trip even from the nearby cities.

There were also some anomalies in visit estimates that arose due to spectral signatures indicating a land use (e.g. water body) on the land cover map that is not a suitable recreation site (e.g. agricultural ditch). Furthermore, county boundaries are evident on the visit map in places, where neighbouring counties have a large difference in initial propensity to undertake a recreational visit (i.e. different county intercepts and distance-weighted population coefficients), which is an artificial boundary that is in reality less sharply defined.

The relative distribution of recreational pressure (visit density) on habitats coincides with expectations according to the strength of coefficients in the recreation model, but also provides additional information regarding habitat use by recreationists. Although broadleaved woodland and freshwater coefficients are similar, visit density is higher for freshwater (54% more visits h^{-1}), and despite the negative coefficient for coniferous woodland SSSI, visit density is similar to broadleaved woodland. This may be due to broadleaved woodland covering a much greater extent (909,406 ha compared to 35,044 ha of coniferous SSSI woodland), causing visit density to be lower. Results regarding habitat use should however be interpreted with caution, as visit estimates from 400 m cells were distributed evenly among the 25 m land cover cells within them, as we had no information on where within a 400 m cell a recreationist would visit. We contemplated weighting land covers using the coefficients from the model to assign greater numbers of visits to more desirable habitats, but as estimates were based on the mix of habitats in a 400 m cell we chose to distribute visits uniformly. Recreationists may, for example, walk along the edge of a woodland, in which case it would not be accurate to assign greater visit numbers to the woodland. One intended application of this work was to investigate recreational pressure on Sites of Special Scientific Interest (SSSIs) and how it may change in the future. However, the 400 m resolution of the visit raster proved too course to garner reliable summary statistics for many SSSIs, which have irregular shapes and may overlap small portions of one or more cells, and as for land cover types, we could not infer where within a 400 m cell recreationists would visit.

Using a future population scenario and annual visit estimates we were not able to reveal meaningful, fine scale changes in the distribution of recreational visits in 2030. Local authority level projected changes in population that were used to recalculate site-level distance-weighted population did not result in substantial changes to predicted probabilities of visitation. The mapped change in estimated visit numbers therefore simply reflected the scaled-up regional visit estimates. Distance-weighted population did not have a large effect size, and even a doubling of distance-weighted population in a subsidiary analysis did not substantially increase predicted probabilities (see Appendix S3.2).

The future scenario was not realistic, as we increased future population without increasing built-up area to accommodate the additional people. Hypothetical scenarios could have been developed to investigate the effects of land use change, such as those used for the National Ecosystem Assessment (Haines-Young, Paterson & Potschin 2011). Alternatively, cellular automata (CA) models could be used to simulate urban growth, which assume that “past urban development affects future patterns through local interactions among land uses” (Santé *et al.* 2010). CA models are complex (see explanation in Santé *et al.* 2010) and choosing the most appropriate model is difficult, but they have been used extensively for urban simulation and urban planning in recent years (e.g. Clarke & Gaydos 1998; Barredo & Kasanko 2003; Guan *et al.* 2011).

Simulations are however inherently subject to errors stemming from uncertainties regarding future conditions and changes in current trends. Forecasting for actual development plans is therefore preferable. Unfortunately, this was not possible on a national scale due to the devolved planning system in England and associated difficulties in obtaining comparable housing plans from all unitary authorities. It could be possible to obtain such plans at a smaller scale (e.g. county) to update built-up area in the land cover map and population according to number of new households, providing much finer scale population predictions and future land cover.

As fine-scale visit estimates using the future scenario here was not possible we did not examine relative distribution of recreational pressure on habitats in the future, as we had intended.

3.5. Conclusions

We have developed a predictive model using a large representative sample of the English adult population and nationally available predictor variables, which we used to map recreational visits across England. This baseline map can serve as a useful comparison to future scenarios of recreational demand based on extensive land use change.

The future population scenario investigated here was unrealistic, as land use change in terms of urban growth that is associated with increasing populations was not incorporated. Future work could involve obtaining strategic growth locations at a smaller scale (e.g. county) to investigate a scenario of urban growth and associated land use change, applying the model developed here.

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Supplementary Information

Table S3.1. Pearson correlation coefficients between all pairs of predictor variables

	Broadleaved	Coniferous (non-SSSI)	Semi-natural	Lowland heath (non-SSSI)	Freshwater (non-SSSI)	Coast (non-SSSI)	Coniferous (SSSI)	Lowland heath (SSSI)	Freshwater (SSSI)
Broadleaved									
Coniferous (non-SSSI)	0.105								
Semi-natural	-0.065	-0.016							
Lowland heath (non-SSSI)	-0.001	0.099	0.143						
Freshwater (non-SSSI)	0.013	0.001	-0.033	-0.007					
Coast (non-SSSI)	-0.076	-0.031	-0.045	-0.013	-0.008				
Coniferous (SSSI)	0.063	0.022	-0.007	-0.002	-0.014	-0.011			
Lowland heath (SSSI)	-0.007	0.012	0.034	0.038	-0.017	-0.013	0.094		
Freshwater (SSSI)	0.006	-0.012	-0.009	-0.008	0.017	-0.004	0.012	0.000	
Coniferous (SSSI)	-0.061	-0.028	-0.045	-0.019	-0.011	0.127	0.002	-0.006	0.018
Arable	-0.270	-0.157	-0.254	-0.131	-0.109	-0.131	-0.072	-0.097	-0.050
Improved	-0.078	-0.127	-0.052	-0.037	-0.060	-0.120	-0.071	-0.082	-0.048
Built-up	-0.081	-0.090	-0.172	-0.055	0.063	0.179	-0.037	-0.053	-0.022
Path length	0.255	0.047	-0.062	-0.012	0.168	0.095	0.073	0.044	0.011
Elevation	-0.004	0.155	0.380	0.153	-0.076	-0.134	-0.014	0.074	-0.029
Distance to major road	-0.087	0.184	0.199	0.093	-0.067	-0.053	0.006	0.066	0.003
Scenic destination	0.118	0.097	0.261	0.104	-0.055	-0.048	0.028	0.142	0.005
Distance-weighted pop	0.072	-0.063	-0.135	-0.045	0.157	0.042	-0.019	-0.033	-0.001

Table S3.1 cont.

	Coniferous (SSSI)	Arable	Improved	Built-up	Path length	Elevation	Distance to major road	Scenic destination	Distance-weighted pop
Arable	-0.110								
Improved	-0.093	-0.398							
Built-up	0.056	-0.382	-0.131						
Path length	0.012	-0.325	-0.021	0.345					
Elevation	-0.117	-0.303	0.046	-0.239	-0.119				
Distance to major road	0.013	0.006	-0.065	-0.302	-0.191	0.370			
Scenic destination	-0.013	-0.197	0.025	-0.239	-0.052	0.470	0.277		
Distance-weighted pop	-0.005	-0.267	0.043	0.476	0.565	-0.183	-0.257	-0.209	

Appendix S3.1

Two alternative GLMMs were specified with distance-weighted population as a fixed effect with and without the built-up variable in the model to examine its effect on the strength of the distance-weighted population coefficient. The coefficient of distance-weighted population with built-up in the model was 0.18 (SD ± 0.03) compared to 0.35 (SD ± 0.03) without. The land cover fixed effects in the GLMM without the built-up variable changed drastically; the only land covers that positively influenced visitation were coast and freshwater non-SSSI. Despite the lack of interpretability and apparent breakdown of meaningful relationships this model was used for comparative purposes. As the distance-weighted population variable had greater explanatory power without built-up we obtained predictions from both GLMMs to see what difference this made.

Predicted probabilities did not change much when removing built-up from the model (mean difference in predictions from the two GLMMs: 0.004 ± 0.04 SD; Fig. S3.1). Therefore, as removal of built-up from the model caused substantial reduction in explanatory power (Δ AIC = 975) and drastic changes in other land cover effects, built-up was retained in the model.

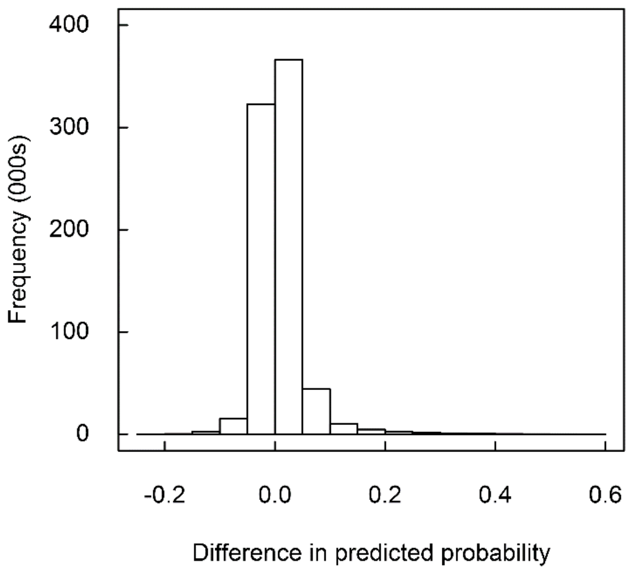

Fig. S3.1. Histogram of the difference in predicted probability to sites across England from a GLMM without and with proportion of built-up, with distance-weighted population as a fixed effect. Predictions based on 2011 population.

Appendix S3.2

Probability of visitation to all sites across England was estimated under a hypothetical scenario, doubling the source population to a site by multiplying distance-weighted population by two. This represents a much greater increase in distance-weighted population than the increase from the population projections for 2030 (mean distance-weighted population in 2030: 3030 ± 5164 SD, compared to 5505 ± 9173 SD from doubled 2011 distance-weighted population). Despite this huge artificial increase in population, the mean difference in predicted probabilities was only 0.01 ± 0.04 SD (Fig. S3.2).

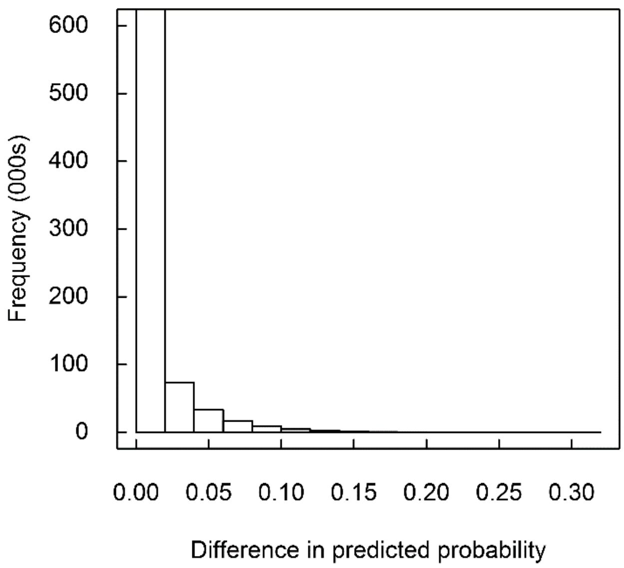

Fig. S3.2. Histogram of the difference in predicted probability to sites across England using distance-weighted population based on 2011 population, and doubled distance-weighted population.

Chapter 4

Large increase in peri-urban recreational visits predicted around new housing developments

Abstract

Population and urban growth are ongoing threats to biodiversity conservation and with rising trends, sustainable development is a pertinent issue. The UK currently mitigates impacts for the sites being developed but does not consider the increase in recreational pressure, due to the larger resident population, that is likely to be placed on the surrounding countryside or urban green spaces (collectively ‘Green Infrastructure’; GI). This may lead to adverse effects on biodiversity – a consequence not currently accounted for by biodiversity offsetting policies in the UK for sites outside EU protection. Using government plans for new housing developments over a 15 year horizon (totalling 40,467 dwellings) for an administrative area in England (5,509 km^2), we quantify conversion of land cover and then examine recreational use of GI, predicting the potential increase in visits arising from the urban growth scenario. To do this, we make spatially-explicit predictions of the change in numbers of recreational visits across the area to map increases in recreational pressure on GI, using a model that incorporates future land cover composition, road network and localised increases in population density. Developments are largely planned on arable land which has a low biodiversity offsetting requirement, but as recreationists are predicted to visit habitats of higher biodiversity value this indicates a major shortcoming in the UK’s current biodiversity offsetting policy. We predict that the increase in visits from future developments and population rise will be extremely local, but emphasise that this represents the day-to-day use of the natural environment rather than visits to honey pot sites. The resultant maps can be used as part of a comprehensive pre-development biodiversity impact assessment. Future work is needed to further our knowledge on the concepts and mechanisms proposed here to better inform the sustainable development agenda, for which we make several recommendations.

4.1. Introduction

In densely populated areas of the world the landscape has been highly modified to support the human population, most notably in terms of land conversion to urban area and farming. In 2012, cropland accounted for 24.7% of the total area across 27 EU member states, whilst grassland covered 19.5% and built-up areas 4.6% (Eurostat 2015). With the world's population continuing to expand, urban growth is a significant driver of ecological change. Whilst urban growth is considered in some cases to pose a threat to biodiversity (e.g. Pauchard *et al.* 2006; Mcdonald, Kareiva & Forman 2008), many studies have shown for some taxa that species richness or population density increases with moderate to intense urbanisation (Tratalos *et al.* 2007; McKinney 2008) and urban areas can harbour species of conservation concern (Fuller, Tratalos & Gaston 2009). Where loss of native species in and around urban areas does occur, this may be due to habitat loss, degradation and fragmentation, physical changes (e.g. air and soil pollution, increase in temperature), introduction of non-native species and human disturbance, including that associated with recreation (McKinney 2002). Recreational activities in the surrounding countryside can reduce populations of species of conservation concern (Mallord *et al.* 2006; Liley & Sutherland 2007).

In populous, urban areas, much of the remaining biodiversity persists in rural areas (McKinney 2002) or within urban areas in undeveloped green spaces (Kong *et al.* 2010) (hereafter collectively termed Green Infrastructure; GI). In England, the government is committed to sustainable development and places strict controls on planning, with conservation of biodiversity a high priority (Defra 2005; DCLG 2012). Brownfield (i.e. previously developed) land is developed first with high densities of new housing to reduce development of greenfield (undeveloped, i.e. semi-natural or agricultural) land (Defra 2005). However, under this policy subsequent integrity of both undeveloped and newly created GI may still be compromised as it must meet the ecosystem service needs of the larger resident population (Niemelä *et al.* 2010). A case in point concerns recreational ecosystem services; there will be a larger source population seeking recreational opportunities in remaining GI, which could place sensitive species under stress for example from increased trampling (Liddle 1991), eutrophication associated with domestic dog waste (Shaw, Lankey & Hollingham 1995), or disturbance (Gill 2007).

Much published work emphasises the need to integrate ecosystem services with land use planning (e.g. Jansson 2013; Bateman *et al.* 2013) but competing interests and multiple stakeholder involvement means that planning practitioners face difficulties in implementation. Thus the increase in recreation-mediated impacts on the environment arising from urban growth may not be accounted for in local development plans. European wildlife sites (i.e. Natura 2000 sites) are an exception, however, as environmental impact assessments are required by law (Article 6 of the ‘Habitats’ Directive 92/43/EEC) when there could be adverse impacts from developments and in such cases planning authorities have undertaken assessments of likely recreational impact. For example, a 5% increase in visits to Breckland Special Protection Area in the East of England was predicted based on 5000 new houses in the nearby town of Thetford, which raised concerns for the Annex I bird species breeding not far from the town (Liley & Tyldesley 2011). There appears, however, to be a paucity of academic research quantifying the conservation implications, which is surprising due to the current rate and extent of urban growth.

Where adverse impacts on European wildlife sites are anticipated, the development must either be relocated or mitigation strategies incorporated in accordance with Article 6 of the Council Directive 92/43/EEC on the Conservation of natural habitats and of wild fauna and flora, known as the Habitats Directive. This states that any plan likely to have a significant effect on the site shall be subject to appropriate assessment of its implications in view of the site’s conservation objectives and will only be approved if it will not adversely affect the site’s integrity, or if it is unavoidable all compensatory measures must be taken. However, in the UK, sites outside the protection of the EC Habitats Directive are only considered for biodiversity mitigation according to the actual site developed. This mitigation strategy uses biodiversity offsets, where potential biodiversity losses arising from a development are quantified and compensated for by restoration or creation of habitats (Defra 2012). Under this system the potential recreational impacts on sites with biodiversity value outside the restrictive focus on the development land would not be considered for biodiversity offsetting.

Here we first quantify expected land cover and population change across an administrative area in England from a baseline to future scenario based on government plans for new housing developments over a 15 year horizon. We then examine recreational use of GI, predicting the potential increase in visits arising from the urban growth scenario. To do this, we make spatially-explicit predictions of the change in numbers of recreational visits across the area from the baseline to future scenario to examine potential increases in recreational pressure on GI, based on a model that incorporates future land cover composition, road network and localised increases in population density. The methodology that we develop can be used to evaluate potential conflicts with biodiversity conservation at the planning stage so that a more comprehensive biodiversity impact assessment can be made and appropriate adjustments or mitigation measures implemented.

4.2. Study area

England is a densely populated nation representative of many western European countries in terms of population density, land cover and land planning policies. England's population is projected to increase by around 7 million by 2030, to over 60 million inhabitants (Office for National Statistics 2010) and the government is tasked with meeting subsequent housing needs. We focus on one county in England, Norfolk, as a case study for developing our methodology (Fig. 4.1). Counties in England may be split into several smaller areas (unitary authorities or District Councils) that are responsible for housing and planning applications. Norfolk is comprised of seven District Councils.

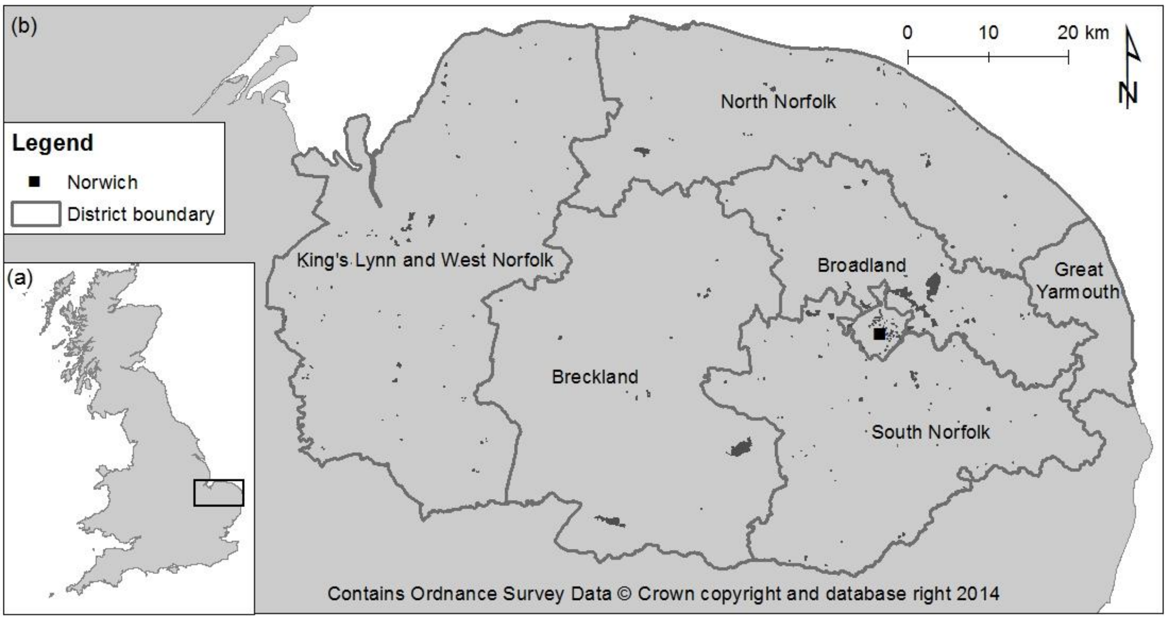

Fig. 4.1. (a) The location of Norfolk within the UK and (b) District Council boundaries. Dark grey areas represent sites allocated for new housing developments.

Norfolk is important from an ecological perspective, with 11.4% of the 5,509 km^2 total area designated as areas of significant conservation importance (Site of Special Scientific Interest or County Wildlife Site). In addition the Broads National Park, a large protected wetland covering 305 km^2, is almost wholly within Norfolk, and the Norfolk Coast Area of Outstanding Natural Beauty (AONB) covers over 450 km^2 of undeveloped coast. These sites are highly popular destinations for outdoor recreationists, receiving millions of visits per year. Outdoor recreation is popular in this region, with a higher visit rate than the national average (estimated at 81.7 visits per adult per annum compared to 68.7; Natural England 2012). Norfolk is projected to experience a 13% increase in population from 841,042 in 2011 (Office for National Statistics 2011) to approximately 950,000 by the year 2026 (Office for National Statistics 2014), slightly higher than the 11.4% national increase. Thus Norfolk is an ideal case study for investigating the impacts of population and urban growth on the number and distribution of recreational visits and the subsequent conservation implications.

4.3. Methodology

4.3.1. Recreation model

Several investigations have modelled recreational use of the natural environment at the site scale, often using on-site surveys and site-scale predictors (e.g. Brainard, Bateman & Lovett 2001; Jones *et al.* 2010). These models are limited in predictive ability to the types of sites the sample came from. In Chapter 3, a model is presented that describes the drivers of recreational site selection by the general public based on an extensive and massive country-wide sample of recreational visits (Natural England 2012). Characteristics of these visit sites were compared with available (not reported as visited) control sites. Visit sites and control sites were point locations across England, thus to obtain the characteristics of these sites, they were buffered by 400 m (based on likely area visited according to empirical studies). The buffered visit and control sites are hereafter termed ‘sample sites’. The explanatory variables used are nationally available so that the model can be used to predict the probability of a recreational visit to any site across England.

The model was formulated as a binomial generalised linear mixed model that predicts the probability of a recreational visit as a function of land cover composition, accessibility and source population. Source population was measured as the total population within a 10 km buffer of sample site centroids (as the majority (82%) of respondents in Natural England’s survey reported travelling less than 5-8 km). The 10 km cut-off distance is further supported by other studies of recreational use (Table S3.1). The percentage travelling from beyond 8 km is slightly greater in some of these studies than from the NE national survey, which may be because these surveys were conducted in large protected areas which attract visitors from further away. The use of 10 km in this study implies that the 18% of recreational visits in the national survey used to calibrate the model will not be accurately modelled. Within the 10 km buffer different distance weightings were tested in univariate generalised linear models predicting visit probability as a function of source population within 10 km, but Chapter 3 demonstrated that inverse squared distance (hereafter 'distance-weighted population') provided the best fit to the data according to change in AIC.

As we were testing a scenario involving future changes in population, the distance weighting determining the scale over which the population from new developments exerts an effect was a vital component of our predictive model. We therefore conducted a sensitivity analysis in which we specified models using different distance weightings of population and re-predicted probability of visits to GI for comparison. Population weighted by inverse squared distance (the measure that gave the lowest AIC) gave the population within 1 km a weight of 1, within 2 km a weight of 0.5, 4 km a weight of 0.25, etc., thus those living closest to a site contributed the most to the site's source population. Two alternative functions were used for making predictions, all still measuring population within 10 km of a site:

$$\sum(pop * (1/d)) \qquad \text{(eqn 1)}$$

$$\sum(pop * (1/\sqrt{d})) \qquad \text{(eqn 2)}$$

We also examined total unweighted population (i.e. total number of people within 10 km radius) and the proportion of built-up area in a 10 km radius around sites (as a proxy for population). Our hypothesis was that some sites would experience a larger change in probability of visits in the future, as an increase in population further away from a site will contribute more to the site's source population than for population weighted by inverse squared distance.

Accessibility to sample sites was measured as the distance from the centre point to the nearest major road, which had a negative relationship with visit probability. Thus if a new road is built, accessibility of nearby sites and thus their visitation probability may increase. Land cover, distance to major road and distance weighted population were significant predictors of visit probability, therefore to model the future we must test scenarios involving changes in these.

The model was evaluated using AUC (the area under the receiver operating curve) in order to assess predictive ability. Recreational visits from an independent data set using the same survey methods (n = 10,622) and a new set of random controls (n = 10,622) were used to calculate AUC using the pROC package in R (Robin *et al.* 2011).

4.3.2. Baseline and future scenarios

To investigate the change in number and distribution of recreational visits across Norfolk in the future we first need a baseline. We predict the probability of a visit to all sites across Norfolk using baseline data (see section 4.3.3), then re-run the predictions using updated data for expected change in land cover, population and road network, i.e. the 'future scenario'. The baseline year is 2011, the year of the latest UK population census. For all other variables, age of data sets vary due to availability but were created in 2007 or later.

The future scenario tested here is based on explicit Local Development Plans (LDPs), which specify land allocations for new housing developments and proposed major roads. Based on these, we made projections for future land cover, population and accessibility via major roads. LDPs are a legal requirement for all Local Planning Authorities (i.e. District Councils) under the National Planning Policy Framework (DCLG 2012). Six of the seven District Councils in Norfolk have LDPs in place with detailed Site Allocation Plans (Norfolk County Council 2014) that map locations for new housing developments (totalling 40,467 dwellings) up to the year 2026 (Table 4.1, Fig. 4.1), which is therefore the year of our future scenario. Site Allocation Plans provide estimated numbers of houses and considerations regarding green space requirements, access, flood risk and other factors potentially affected by the development. Housing densities vary according to local character (mean 31 houses per hectare); numbers per site are provided in Site Allocation Plans. Development boundaries were obtained from each of the District Councils as ESRI shapefiles and imported into ArcGIS 10.1 (ESRI, Redlands, California).

The only new major road planned during the period of the Local Development Plans for Norfolk is the Northern Distributor Road (NDR), a *c.*20 km long dual-carriageway that will be constructed within Broadland, north of the city of Norwich by around 2017. We first examine the predicted number of visits arising solely from new housing and source population (scenario 1), then those predicted from new housing and also the construction of the NDR (scenario 2) to quantify the separate effects on recreational patterns of improved GI accessibility.

Table 4.1. Details of proposed housing developments in District Council Site Allocation Plans and estimated number of residents

District	No. of housing development sites¥	Mean size (ha) of development (± SD) ¥	No. of houses*	Estimated no. of residents€
Breckland	46	10.4 (± 43.0)	10,064	24,153.6
Broadland	51	17.2 (± 45.4)	12,566	30,158.4
Kings Lynn and West Norfolk	89	3.1 (± 6.2)	4,889	11,733.6
North Norfolk	46	4.4 (± 12.8)	3,086	7,406.4
Norwich	62	1.8 (± 4.1)	5,877	14,104.8
South Norfolk	62	3.5 (± 6.2)	3,985	9,564
Total	**356**		**40,467**	**97,120.8**

¥ Provided as shapefiles by the respective District Council.

* Taken from District Council site allocation documents

€ Calculated by multiplying number of houses by mean number of residents per house (2.4; Office for National Statistics 2011)

4.3.3. Land cover, population, and infrastructure projections

Investigating the impacts of the future scenario required updating: i) the land cover composition to reflect the new housing developments; ii) the source population, reflecting the localised increases in number of residents, and iii) the road network to incorporate increased site accessibility from the proposed Northern Distributor Road.

Land cover data used for building the recreation model came from the 25 m resolution Land Cover Map 2007 (LCM2007; Morton *et al.* 2011). The proportion of each land cover within sample sites was extracted and entered as separate variables into the model. LCM2007 was used to obtain land cover variables for all sites in Norfolk to predict the distribution of recreational visits across Norfolk in the baseline scenario (see section 4.3.4), but needed to be updated for the future scenario according to locations of new housing developments. Housing developments may contain a mix of land covers due to the retention of tree lines, creation of GI such as parks and recreation grounds etc., therefore the areas allocated for development could not simply be classified as 100% built-up (the most obvious land cover classification for housing in LCM2007). Thus, to estimate the land cover composition of the proposed housing developments, as represented in LCM2007, nineteen recently completed developments across Norfolk were identified in Google Earth using the historical imagery tool. Developments were chosen that were completed after 1999, and therefore have a similar housing density and housing type as proposed developments. Furthermore, the chosen completed developments were constructed by 2006, before the date of satellite images used to create LCM2007, so represent land cover composition in the early phase (1-8 years) after building. The nineteen completed development boundaries were digitised in Google Earth and imported into ArcGIS (mean area (ha) = 7.2 ± 5.8 SD). Land cover composition was extracted from LCM2007 and the mean proportionate cover calculated (84% built-up, 2% broadleaf, 7% arable, 7% improved grassland and 1% semi-natural grassland). This represents the average remote-sense classification of land cover for a modern housing development in this region. This was then used to update 25 m raster cells in LCM2007 in areas allocated for future developments in Site Allocation Plans by randomly recoding cells to revised land covers proportionate to the mean composition of recent developments.

To calculate source population as an input for the recreation model, a 1 km resolution baseline raster was created from population data from the 2011 census for England (Office for National Statistics 2011). The 2011 population raster was used for predictions under the baseline scenario. To create our 2026 future population raster, population projections made by the ONS are too course even at the smallest administrative unit, the District Council level (n = 7, mean area = 826 km^2; Fig. 4.1), which does not reflect localised increases resulting from housing developments, nor does it allow fine scale predictions of changes in spatial distribution of recreational visits resulting from population increase. Thus we used the allocations for new housing developments to update our 2011 population raster, multiplying the estimated numbers of houses per development by the average household size in England (2.4 persons; Office for National Statistics 2013) to estimate the additional population per development in 2026 (Table 4.1). The additional population was assumed to be dispersed evenly across the development; thus cells of the 1 km population raster received additional people according to the proportion of the new development overlapping with each 1 km population grid cell. Developments were only obtained for the county of Norfolk, but there may be developments planned in neighbouring counties close to the Norfolk borders that could increase the source population to sites in Norfolk within 5 km of the border. Therefore numbers of visits to some sites within 5 km of the borders may be underestimated. For validation, the 1 km^2 population predictions were summed per district and compared with ONS district level predictions

The proposed route for the NDR, provided by Broadland District Council, was merged with the existing road network (A roads; OS Meridian 2 2013) for the future scenario.

4.3.4. Predicting visits

To extrapolate from the sample to the whole of Norfolk, points were placed across Norfolk in a regular grid pattern spaced 400 m apart (n = 33,498) and buffered by 400 m as this was the nature of the sample sites used in model calibration. The 400 m spacing was chosen to ensure complete coverage of Norfolk as 400 m buffers overlapped slightly. For each buffer (site) the explanatory variables were extracted using the baseline data, and the recreation model (calibrated on national data) was used to predict probability of a recreational visit to all sites across Norfolk under the baseline scenario. Then variables for proportion of land covers within each site, source population and distance from nearest major road were re-extracted from the updated layers to re-estimate the probability of visitation under scenario 1 and 2. None of the sites had a predicted visitation probability of 1 in either the baseline or future scenarios so there was no issue of saturating the upper threshold (that would curtail the predicted increase in recreation).

As site visitation probabilities (per week) are not intuitive we scaled up to estimate total numbers of visits per site per year. Firstly, the annual number of visits in Norfolk in 2011 were calculated as per Eqn 3, whereby *Norfolk pop* is the total population of Norfolk from the 2011 population raster, *visit rate* is Natural England's estimated population visit rate for Eastern England (87.1 visits $person^{-1}$ $year^{-1}$; Natural England 2012) and *proportion of countryside visits* is the proportion of visits that were deemed countryside visits (0.77) in Natural England's sample for Eastern England. A 'countryside visit' is that to a site containing less than 70% urban area. The visit estimate from Eqn 3 was divided by $\sum P_i$, the sum across all sites of visitation probabilities, to get a visitation rate (Vr) per unit of probability (Eqn 4). We then multiplied P_i, the probability of a recreational visit to site i, by this visitation rate to obtain a visit estimate for each site in Norfolk (Eqn 5).

Total countryside Norfolk visit estimate = Norfolk pop * visit rate * proportion of countryside visits (eqn 3)

Vr = Total countryside Norfolk visit estimate / $\sum P_i$ (eqn 4)

No. visits.yr^{-1} cell^{-1} = $Vr * P_i$ (eqn 5)

To predict the new visitation rates for sites in 2026, we substituted P_i in Eqn 4 and 5 with visitation probabilities from the two future scenarios (scenario 1: incorporates increases arising from increased population; scenario 2: increased population and updated distance from major road incorporating the NDR) but retained the same Vr. The predicted number of visits per site were subsequently represented as a continuous (400 m resolution) raster surface with each site represented as a cell for ease of interpretation.

4.4. Results

4.4.1. Changes in land cover and population

New developments are mainly planned on arable land and improved grassland, which are therefore predicted to experience the largest loss of cover within Norfolk relative to other LCM2007 land cover types (*c.*1360 ha and 170 ha respectively).

The population of Norfolk in 2026 predicted from estimated numbers of additional residents in new houses planned by District Councils was 953,985, in close agreement with the ONS projection of an increase from 857,888 to *c.*950,000 (summed across districts), validating the approach taken in this chapter. Predictions derived from planned housing developments *within* districts are presented in Fig. 4.2. The ONS district level population projections are also shown in Fig. 4.2 alongside our district level predictions. Discrepancies arose because ONS projections used data on past demographic trends while our predictions are based on land availability and government selected growth points, and developments planned after constraints have been considered in the LDP process. We therefore suggest that our district level estimates are likely to be more accurate. Most notably we predicted an increase of 23% from the baseline in Broadland and 7.6% in South Norfolk (Fig. 4.2), whilst the ONS predicted an 8.3 and 19.3% increase respectively. The large proportionate increase in Broadland from our predictions is due to the North East Growth Triangle, an area of *c.*680 ha allocated for *c.*8700 houses to the north-east of Norwich.

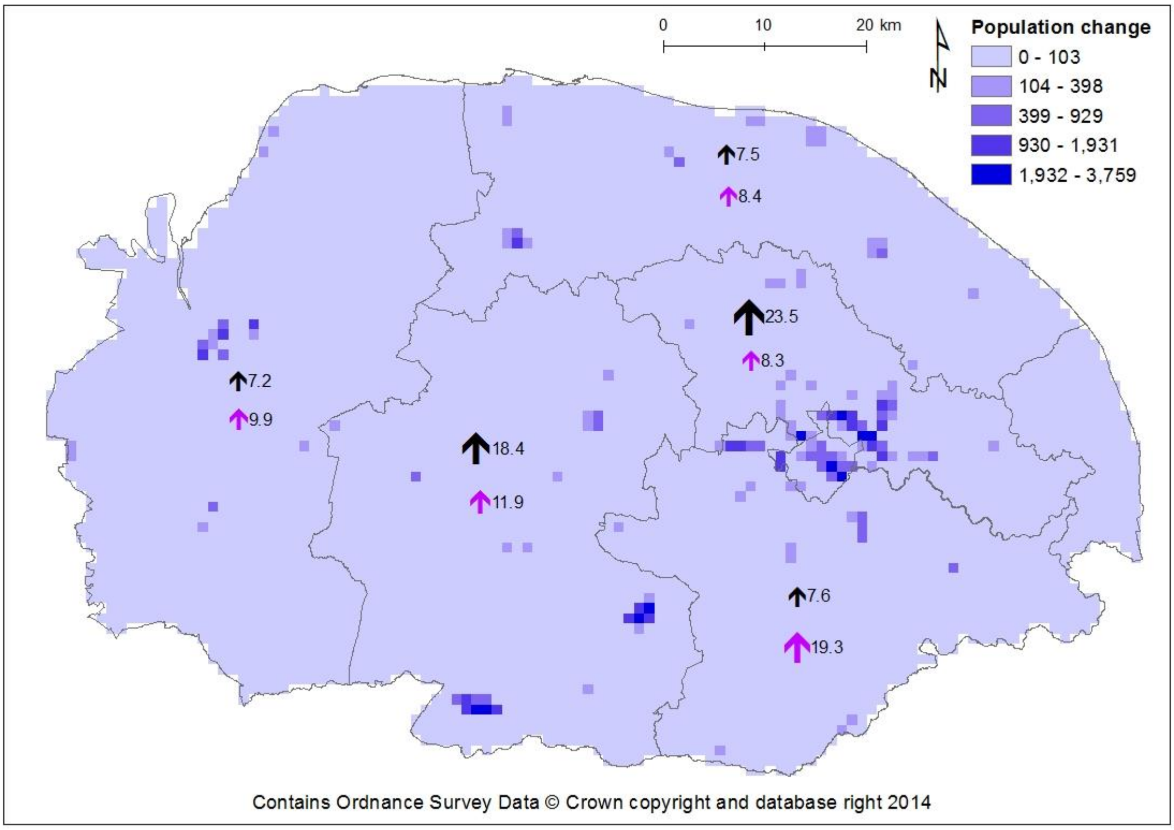

Fig. 4.2. Estimated population change from 2011 to 2026 per 1 km grid square based on location and density of new housing. Proportional increase per district predicted by the models generated in this paper is shown by purple arrows, and from ONS projections by black arrows.

4.4.2. Predicted increase in recreational visits

The sensitivity analysis showed that there was very little difference in predictions using the different distance weightings of, or proxies for, population; using any of the four alternative methods made very little difference to the change in probability of visit from baseline to future scenario (Fig. S3.1). Thus we were confident in using population weighted by inverse square distance, which was used for all subsequent predictions.

We estimated the total annual number of countryside visits as 57,544,693 within Norfolk in 2011, rising by 1.9% to 58,646,529 in 2026 based on scenario 1. Of the 1,101,836 additional predicted visits, 91% were within 1 km of new housing developments, with a further 4% within 1-3km (Fig. 4.3a). Considerable increases were predicted around and in the urban area extensions of Thetford and Attleborough and the North East Growth Triangle (Figs 4.3b – d). There is a minimal increase in the annual number of visits to cells across the wider Norfolk countryside. This is expected, both from the formulation of the distance-weighted population predictor, and potentially also due to the positive coefficient of ‘built-up’ land cover, so that house building within the 400 m buffer increases the potential for site visitation. Built-up area increased in 1,128 sites, which had a mean change in annual visits of 892 (± 1,654 SD). The mean proportion of land converted to built-up in the 125 sites that were predicted to experience more than 2,500 additional visits per year was 0.33 (± 0.14 SD). Of the rest of the visitation sites considered across Norfolk, 31,795 were affected only in terms of an increase in source population, and experienced a very low increase in annual number of visits (mean 3 ± 8 SD). Only 188 sites did not experience any change in proportion of built-up or source population and therefore had no change in number of visits.

Addition of the NDR (scenario 2) resulted in only a slight (1%) increase in the total number of visits relative to that predicted for scenario 1. With the addition of the NDR, 2,057 sites were then closer to a major road and therefore increased in predicted numbers of visits from scenario 1 (mean change in annual visits 21 ± 68 SD). Of these, only 240 were predicted to receive more than 50 additional visits per year (Fig. 4.4). Under the two scenarios the same percentage of visits were within 3 km of developments (95%), but with the addition of the NDR the percentage within 1 km decreased slightly from 91% to 89%, with 6% within 1-3 km.

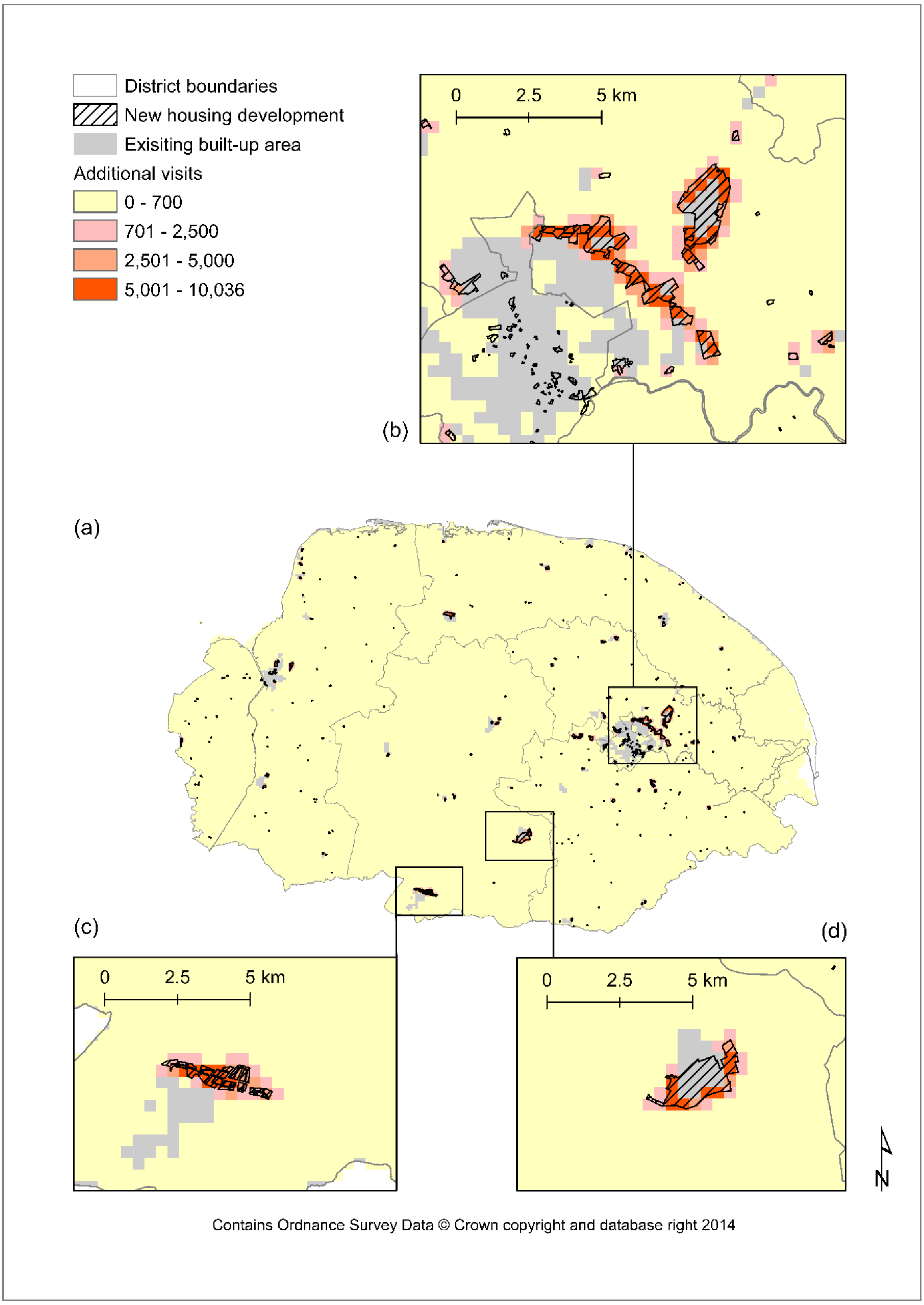

Fig. 4.3. (a) Predicted change from 2011 to 2026 in annual number of recreational visits across Norfolk per 400 m grid square based on future scenario 1 (house building only; see text). Insets show the effects of planned urban extensions in (b) the Norwich Growth Triangle, (c) Thetford and (d) Attleborough. Class intervals calculated using Jenks natural breaks.

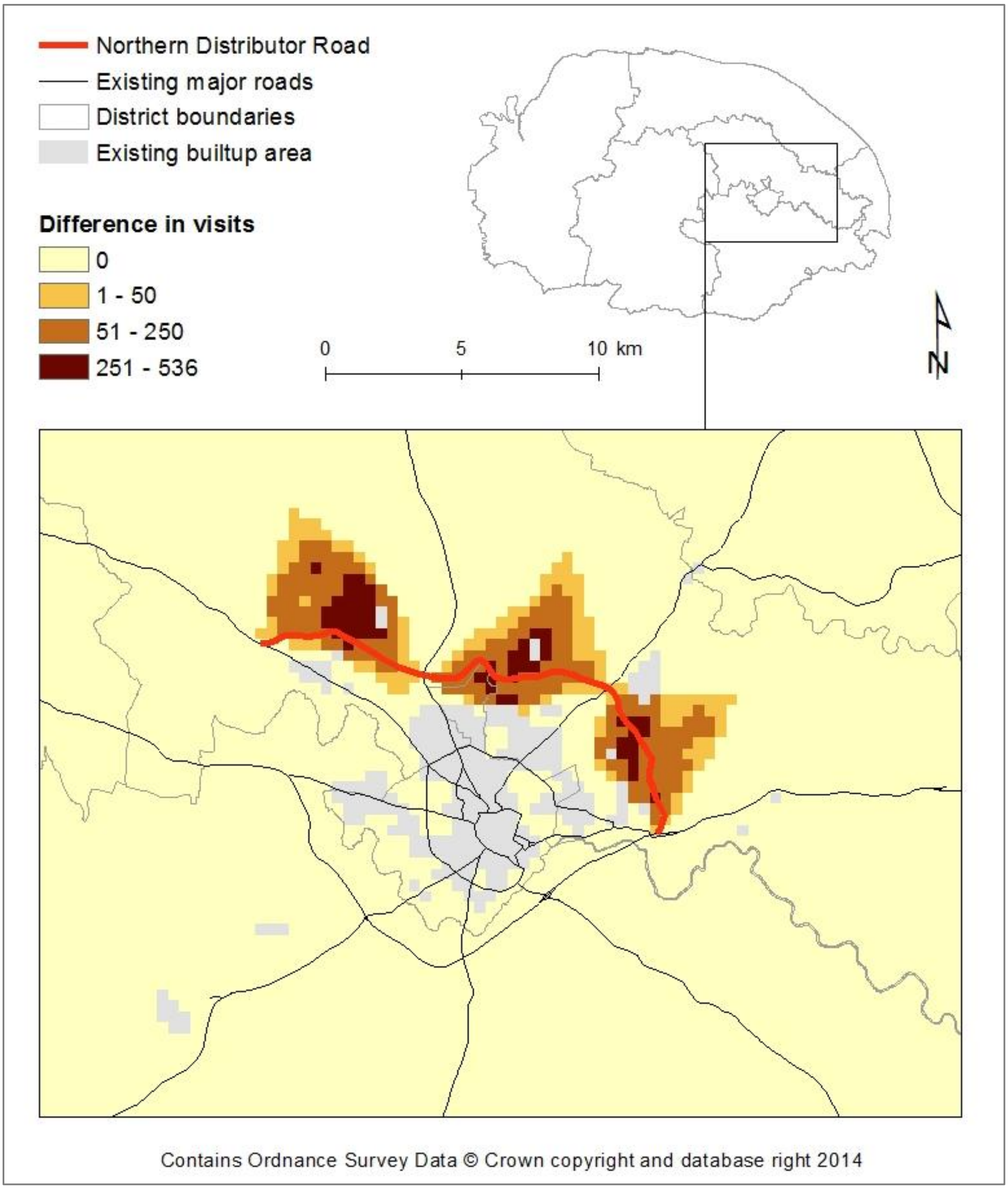

Fig. 4.4. Difference in predicted number of visits upon construction of the Northern Distributor road, from that predicted from house building only.

4.5. Discussion

We proposed that consideration of how developments will affect recreational patterns should be taken pre-construction as part of an environmental impact assessment. Specifically, we suggest that biodiversity offsetting should account for the potential impacts on GI outside development areas in addition to considering impacts within the developed site. We show that developments are mainly planned on agricultural land, which is classed as low in the habitat type band and distinctiveness category in England's current biodiversity offsetting framework (Defra 2012), that gives agricultural land a lower number of 'biodiversity units' per hectare (depending on condition) and consequently a lower offsetting requirement. The new inhabitants of the development, however, are likely to visit habitats of higher biodiversity units such as woodland, according to studies of recreationists' preferences (Hornigold et al., unpublished data; Sen *et al.* 2014). Thus mitigation for developments in terms of biodiversity offsetting is likely to be inadequate.

Across Norfolk, we predicted high increases in visits close to developments, thus recreational pressure will greatly increase in peri-urban GI in the future. Conversely, the wider countryside is not predicted to experience large increases in visits. As the sample used to calibrate the predictive model captured mainly local visits this is not surprising. Trips further from home (e.g. greater than 5 km) tend to be more infrequent (monthly to seldom or never; Schipperijn *et al.* 2010) but may tend to aggregate in fewer highly desirable sites (National Parks, AONBs, etc.). Therefore, the predictions presented here may be considered to represent the recreational pressure arising from 'day to day use of the natural environment'. Indeed, over 85% of daily users of green space in a Danish national survey travelled less than 300 m, and cumulatively 98% travelled less than 1 km (Schipperijn *et al.* 2010), corroborating our findings.

Van Herzele & Wiedemann (2003) propose that there are different 'functional levels' of GI, for example a regularly visited local urban park compared to a large peri-urban forest visited on occasional weekends. In order to capture visits in different functional levels, additional questions would be required in the MENE survey perhaps specifying a minimum travel distance or travel time. Visits occurring in different distance and frequency bands would have to be modelled separately to identify relationships driving site selection at each functional level, and separate predictions made for changes in visits within each functional level.

The tool developed here used nationally-available predictors to predict changes in recreational use, and as such does not contain sufficient information for a complete biodiversity impact assessment. Rather it can highlight areas expected to receive high numbers of additional visits for which further investigation of potential biodiversity impacts is required. We recommend that decision makers acquire further details for such areas, such as presence of priority, vulnerable, or sensitive species and habitats, and expert opinion on carrying-capacity for recreationists and management strategies. One simple option for assessing conservation status of these areas is to overlay boundaries of UK statutorily protected areas including both SSSIs and County Wildlife Sites (CWS). We could not make specific predictions for change in number of visits to SSSIs and CWS due to the resolution of the predicted visits raster (400 m) and the nature of the SSSI and CWS data (irregular shaped polygons of much smaller area). The predicted visits raster could have been resampled to a finer resolution, but this would be unrepresentative as the predictions were made for the characteristics found within the cell, with no information on where recreationists go within the cell. Nonetheless if a SSSI or CWS is close to, overlapping or within a raster cell with a high predicted increase in visits this would indicate a site in need of further assessment.

Alternative distance weightings or proxies of population made negligible difference to predicted visitation probability. Furthermore, sites where the only change in the future scenario was in population (i.e. no land use change or new roads) were predicted very slight increases in numbers of visits. The construction of a new major road increased visits to nearby sites and also led to a slightly greater proportion of visits further away from developments. Site land cover, however, was the biggest determinant of recreational visits, as all of the sites that experienced large increases in recreational visits in the future had some proportion of land converted to built-up, a strong positive predictor in the model.

The change in annual visits under a scenario of land use change was estimated by Sen *et al.* (2014) whereby 100 ha of intensively farmed agricultural land was converted into open access woodland. Their model, based on a single year sample of the same survey used here, included site land cover composition, substitute availability (percentage of each land cover around outset areas) and the travel time from outset areas to sites. An increase of over 200,000 visits per annum to the new woodland was predicted, with a reduction in visits to other local sites comprising urban, farmland, floodplain and grasslands. This reduction was due to the new woodland acting as a substitute site, drawing visitors away from less attractive sites. As our approach is location-centred (on site characteristics), rather than people-centred (e.g. no explicit measure of travel from outset areas to sites), we were unable to investigate substitute effects. Providing insights into the efficacy of substitute sites at reducing visitor pressure on conservation sites should be a high priority for the sustainable development agenda. Furthermore, as 'honey pot' sites further from home were underrepresented and we did not measure amenity provision at sites we could not test the substitutive effect of creating a new honey pot site, which we acknowledge requires further investigation. With appropriate real life case studies it would be possible to test whether creation of a new honey pot site would draw visitors away from existing vulnerable sites (i.e. total number of visits remains unchanged), or whether it would create new demand (i.e. increase total number of visits).

A further consideration related to substitute sites concerns site attachment (emotional ties) and continuity with the past (Dallimer *et al.* 2014). If there is a high level of attachment to a site then creation of an objectively desirable substitute site may not draw as many visitors as predicted by a national-level model. These cultural elements to site selection can only be measured on a site-by-site basis through surveys of the source population and therefore cannot be included in a national model.

Finally, an extension to this work could involve deriving visit rates specific to certain socio-demographic groups. Socio-demographic characteristics have been shown to influence propensity for undertaking recreational activities (Jones *et al.* 2010; Sen *et al.* 2014; but see Dallimer *et al.* 2014). So in terms of predicting the effect of house building and change in population on recreational use of the surrounding area it is important to consider the demographic make-up of the new resident population. A different demographic would be accommodated if, for example, the housing composition was apartments, starter homes or large detached houses. Using county-specific intercepts in the model, as here, does not capture this finer level of detail.

4.6. Conclusions

We have made the first attempt at mapping changes in recreational ecosystem services arising from urban and population growth. Using a Geographical Information System and readily available digital spatial databases, including explicit government plans for new housing, maps can be generated with relative ease. We predicted that the increase in visits will be extremely local, which accords with previous work on visitor behaviour.

Importantly, we identified a possible mechanism that would indicate a major shortcoming in the UK's current biodiversity offsetting policy.

There is, however, more work needed to better inform the sustainable development agenda. We propose the following: 1) map visits at different 'functional groups' of GI to acquire a more complete picture of the recreational use of the natural environment; 2) investigate substitute sites, especially honey pot sites, to elucidate their effect on overall recreational levels as well as visitor distribution; 3) use demographic-specific visit rates with detailed population projections to obtain greater accuracy in predicted numbers and distribution of visits. Once these issues are addressed, we would have a powerful tool capable of testing alternative development scenarios pre-construction to allow for strategic planning at the most optimal stage of decision-making (Mandle *et al.* 2016).

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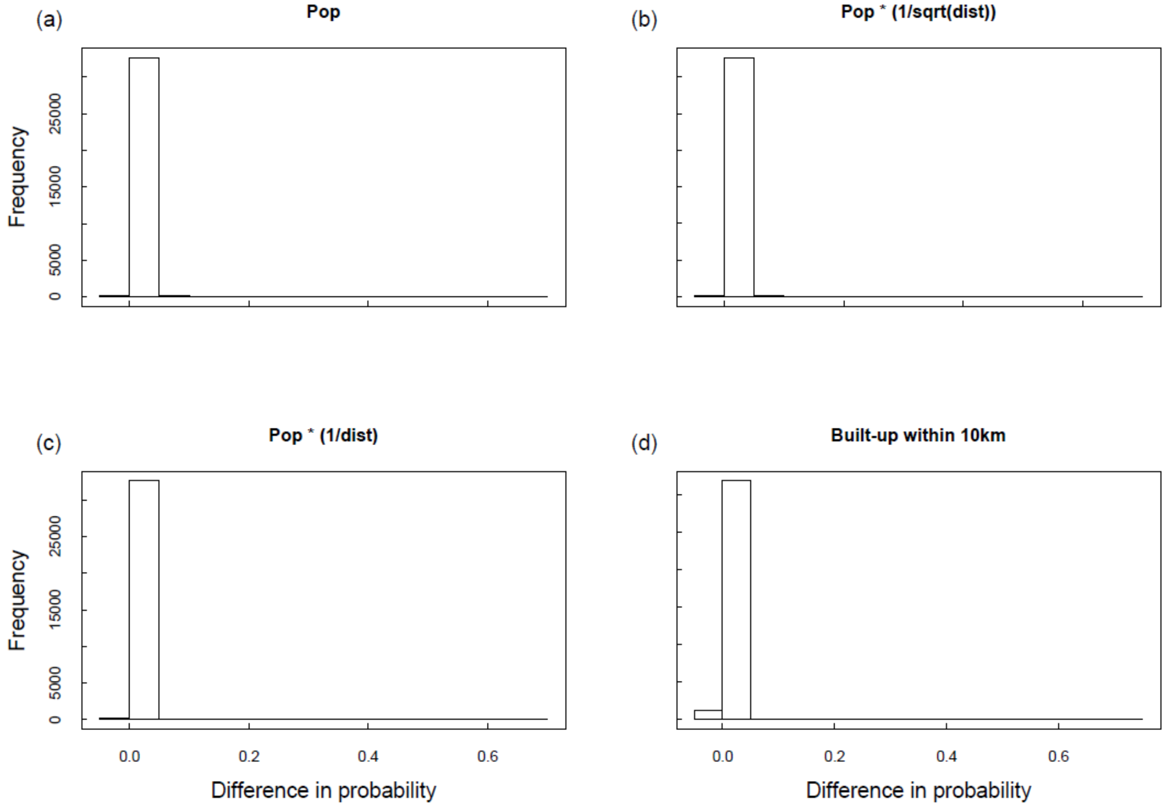
Supplementary Information

Table S3.1. Distances travelled by visitors from home to reach a recreation destination, according to questionnaire surveys

Study area	Size of study area	Travel distance	Source
Denmark	Country-wide	98.1% of daily visitors travelled less than 1 km	Schipperijn et al. (2010)
Breckland SPA	940 km^2	63% travelled less than 10 km	Dolman et al. (2008)
Thames Basin Heaths	82.8 km^2	70%, 100% and 93% arriving by car, on foot or by bike respectively travelled less than 5 km	Liley et al. (2005)
Dorset Heathlands	81.7 km^2	90% arriving by car or on foot travelled less than 8.8 and 1.1 km respectively	Clarke et al. (2006)
Ashdown Forest	30 km^2	76% travelled less than 10 km	Clarke et al. (2010)
South Sandlings	30 km^2	63%, 94% and 89% arriving by car, on foot or by bike respectively travelled less than 10 km	Cruickshanks et al. (2010)
New Forest National Park	280 km^2	35% live within National Park or within 8 km of its boundary	Sharp et al. (2008)
Heverlee-Meerdal	18.9 km^2	69% of visitors live within a 10 km radius	Roovers et al. (2002)

Fig. S3.1. Frequency histogram of the difference in predicted probabilities from the baseline to future scenario using different measures of population: (a) raw population (pop), (b) population weighted by inverse square root distance (pop * (1/sqrt(dist)), (c) population weighted by linear distance (Pop * (1/dist)), and (d) proportion of built-up area within a 10 km buffer (built-up within 10 km)



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Chapter 5

Modelling use of forest recreation routes: an application for woodlark (*Lullula arborea*) conservation

Abstract

Forests are multifunctional landscapes, managed for commercial timber, recreational use, nature conservation and other ecosystem services. To reconcile recreation and conservation interests, knowledge of the spatial distribution of visitors within a site is necessary, and a means of testing the effectiveness of interventions prior to their implementation can enhance conservation effectiveness. We present a novel methodology for temporally- and spatially-explicit prediction of recreational disturbance within a large (187 km^2) protected forest with an extensive path network and nearby residential areas. By combining statistical modelling and Geographical Information System (GIS) based Network Analysis (NA) we predict the number of disturbance events on each path section in the network arising from dog walkers and walkers. The model was parameterised using a large sample of 1 hour visitor counts at *c.* 180 sampling points on the path network collected over 5 years, totalling 1,713 survey hours. Controlling for variability due to time, day, month, school holidays and potential source of recreationists from nearby households, the number of disturbance events per hour was modelled as a function of a GIS-derived network distance from the lowest-impedance access point. The impedance caused by path type and car park capacity at access points (known to affect recreational use of a site) was determined by selecting NA weightings that provided the best model fit as judged by lowest AIC. We evaluated the performance of the final GLMMs using 10-fold cross validation. Interesting patterns in spatio-temporal distribution of dog walkers and walkers were revealed that can be used to inform decision-making. An application for woodlark (*Lullula arborea*), a disturbance sensitive species of conservation concern, is presented. Using our methodology we estimate the number of potential woodlark territories lost under a range of potential disturbance thresholds and examine a scenario of future housing with possible mitigation through access point closure.

5.1 Introduction

Forests are multifunctional landscapes, managed as a timber reserve (Peterken 1993, pg. 85) and also providing various ecosystem services (MEA 2005) including those of conservation habitat and recreation destinations (Marzano & Dandy 2012). In developed countries, forests are often enhanced for recreation through provision of extensive path networks, picnic sites and car parks. However, it may be necessary to manage recreational use if species sensitive to disturbance are present (Marzano & Dandy 2012), as disturbance by recreationists can significantly reduce animal populations (Mallord *et al.* 2006; Liley & Sutherland 2007). If the effectiveness of conservation interventions can be tested prior to implementation, such as those intended to enhance habitat provision or mitigate recreational disturbance (e.g. Mallord *et al.* 2006), practical management will likely be more successful. Therefore, effective recreation management requires knowledge of the spatial distribution of visitors within a site but also an understanding of the factors affecting these distributions, without which predictive modelling of alternative scenarios is not possible. The intended contribution of this paper is to provide such a tool, describing a novel approach to using on-site survey data to build a mechanistic model for predicting the spatial pattern of recreational disturbance throughout an extensive site.

One of the challenges of landscape-scale recreational modelling is that the spatial distribution of visitors within an extensive landscape depends upon many factors including the population surrounding access points, car park capacity, distance from the nearest access point and the relative quality and appeal of alternative routes (Dolman, Lake & Bertoncelj 2008). Previous studies have estimated visitor numbers to a site (e.g. state owned forests or protected heathland; Brainard, Bateman & Lovett 2001; Liley, Jackson & Underhill-Day 2005; Jones *et al.* 2010), but did not address the issue of within-site distribution i.e. how visitors are dispersed throughout the site. One study that did, by Clarke, Sharp & Liley 2010), surveyed access points to a 3000 ha mixed forest and heathland location to obtain numbers arriving at the site. They then predicted total number of visitors arriving at each access point over a one month period based on car park capacity and number of residents in different distance bands around access points. The resulting visitor estimates for each access point were then distributed across the path network in relation to the frequency distribution of distance travelled within the site taken from routes mapped by visitors in their survey. Limitations to this approach are that a generic path distance function cannot account for potential effects of path characteristics (e.g. overgrown, narrow paths compared to wide, mown paths), context (e.g. locations of attractions, such as a visitor centre or playground), and that the calibration (training) data depend on the accuracy to which visitors are able to recount and map their routes when subsequently interviewed on exit and some visitors may be uncertain of the route they took (e.g. Liley, Jackson & Underhill-Day 2005). This issue of recall bias is particularly prevalent in areas with a dense path network. This latter issue could be addressed by issuing hand-held GPS devices to recreationists for the duration of their visit; GPS tracking has previously been used to map recreationists' routes and look at patterns in their movement (Orellana *et al.* 2012; Beeco *et al.* 2013; Beeco, Hallo & Brownlee 2014). However, in a forest setting satellite signal may be poor and dense path networks present logistical difficulties of device retrieval (Taczanowska, Muhar & Brandenburg 2008).

Here, we develop an approach to model and then extrapolate the spatial distribution of visitors across a large, complex landscape (187 km^2) with extensive path network (*c*. 1,490 km). For model calibration we use a novel, large data set of visitor counts on paths within a forest to obtain objective, empirical evidence on recreational path usage that is not confounded by recall bias or GPS location error. We develop an innovative methodology that brings together statistical modelling and Geographical Information System (GIS)-based Network Analysis (NA) to predict the number of disturbance events (a measure of recreational disturbance) on each path section throughout the forest as a function of source population density and proximity, access points, network distance and path characteristics, accounting for time, day and season effects. Network Analysis has previously been used to assess access to green space via roads for different religious and ethnic groups in an English city (Comber, Brunsdon & Green 2008) and service area of parks in Seoul, South Korea, using pedestrian routes with impedances (additional travel time) for crossings, overpasses and underpasses (Oh & Jeong 2007). Here, we use NA to predict routes taken within a site, rather than to a site, and use impedance to combine a number of important features (road crossings, path type and car park capacity), generating a measure of likely path usage.

We demonstrate the application of this model as a tool for recreation management and conservation of species of concern by estimating recreational disturbance rates on footpaths under a planned programme of open habitat creation within Thetford Forest that could potentially benefit breeding woodlark (*Lullula arborea*), an Annex I species under the EC Birds Directive 1979. Breeding woodlark are sensitive to all types of recreational disturbance; the probability of settling on suitable habitat in Dorset (n = 16 heathland sites) was less than 50% at 8.3 (95% CI: 5.8-10.9) disturbance events hour^{-1} (Mallord *et al.* 2006). The proposed habitat creation will bound sections of the path network, hence estimation of disturbance rates is crucial for decision-making. We test a scenario of future disturbance levels from increased housing, as well as possible mitigation through closure of access points, demonstrating the utility of our model for assessing the effectiveness of management interventions.

5.2 Study site

Thetford Forest is a large (18,730 ha) plantation forest in the East of England, managed by the Forestry Commission (FC) as part of the Public Forest Estate in England. It comprises predominantly coniferous woodland and a range of open habitats that contain rare and threatened species (Armour-Chelu, Riley & Brooke 2014). Consequently, Thetford Forest is of significant conservation interest, with multiple designations under European and national law: notably as part of the Breckland Special Protection Area (SPA), designated for the breeding populations of Woodlark (*Lullula arborea*) and Nightjar (*Caprimulgus europaeus*), and also contributing to the Breckland Special Area of Conservation (SAC), with the majority of the forest designated as a Site of Special Scientific Interest (SSSI). Almost 75% (14,108 ha) of the forest is open access land designated under the Countryside and Rights of Way Act 2000, and is therefore accessible to the public (Armour-Chelu, Riley & Brooke 2014). Public access is forbidden in the Stanford Training Area, a military training base. In relation to other heathland/woodland SPAs in England, recreational visits to this area are relatively low. Thetford Forest receives around 1 million visits per year (1,064 visitors/km^2) (Armour-Chelu, Riley & Brooke 2014) compared to 5 million (61,200 visitors/km^2), 7.5 million (90,580) and 13.3 million (47,500) in the Dorset Heaths, Thames Basin Heaths and New Forest respectively (Sharp, Lowen & Liley 2008). As a managed forest, the site is criss-crossed by a network of paths (Pedley *et al.* 2013; Wäber, Spencer & Dolman 2013) that provide access to recreationists, for rangers to carry out deer management (Wäber, Spencer & Dolman 2013) and provide open-habitat for associated biodiversity (Pedley, Bertoncelj & Dolman 2013). Management for these latter two entails cutting grass verges along some sections of the path network to maintain visibility for rangers or optimal conditions for biodiversity, but may also benefit recreationists as path conditions are generally drier.

5.3 Methods

5.3.1 Visitor surveys

Recreational visit data were collected between April and October to sample the spring and summer seasons over 5 non-consecutive years (2007-2009; 2013-2014). Surveyors spent 1 hour periods between 6 am and 6 pm recording recreationists encountered at path intersections (sampling points) randomly located within the forest site (chosen without prior knowledge of their popularity and at different distances from access points and housing areas; Fig. 5.1). Sampling points were placed at path intersections as this is an efficient way of simultaneously monitoring multiple paths leading to the intersection. The majority of the data were collected in 2007 (number of sampling points = 138; Dolman *et al.* 2008) and 2008-2009 (174; Dolman 2009). Additional (non-random) sampling of points at popular ‘honey pot’ picnic sites (n = 15) was carried out in 2013-2014 to extend surveys to areas where high levels of visitation were expected, with additional previously surveyed points (n = 11) re-surveyed to control for any year effects. For each disturbance event, the number of recreationists, their activity (dog walking, walking or cycling) and the path section they used to approach and leave the intersection were recorded. Annually, each point was surveyed approximately 3 times during April-October so that each point was visited on a different time-day combination each month to capture variability in recreational patterns (including school and national holidays) as this affects the number of visitors to paths (Liley, Jackson & Underhill-Day 2005; Cruickshanks, Liley & Hoskin 2010). This allowed us to model three temporal scales (time, day and month), so that predictions can be averaged over the periods of interest for any conservation intervention (for example during a species’ breeding season as for woodlark in this study). For additional methodological details see (Dolman, Lake & Bertoncelj 2008; Dolman 2009).

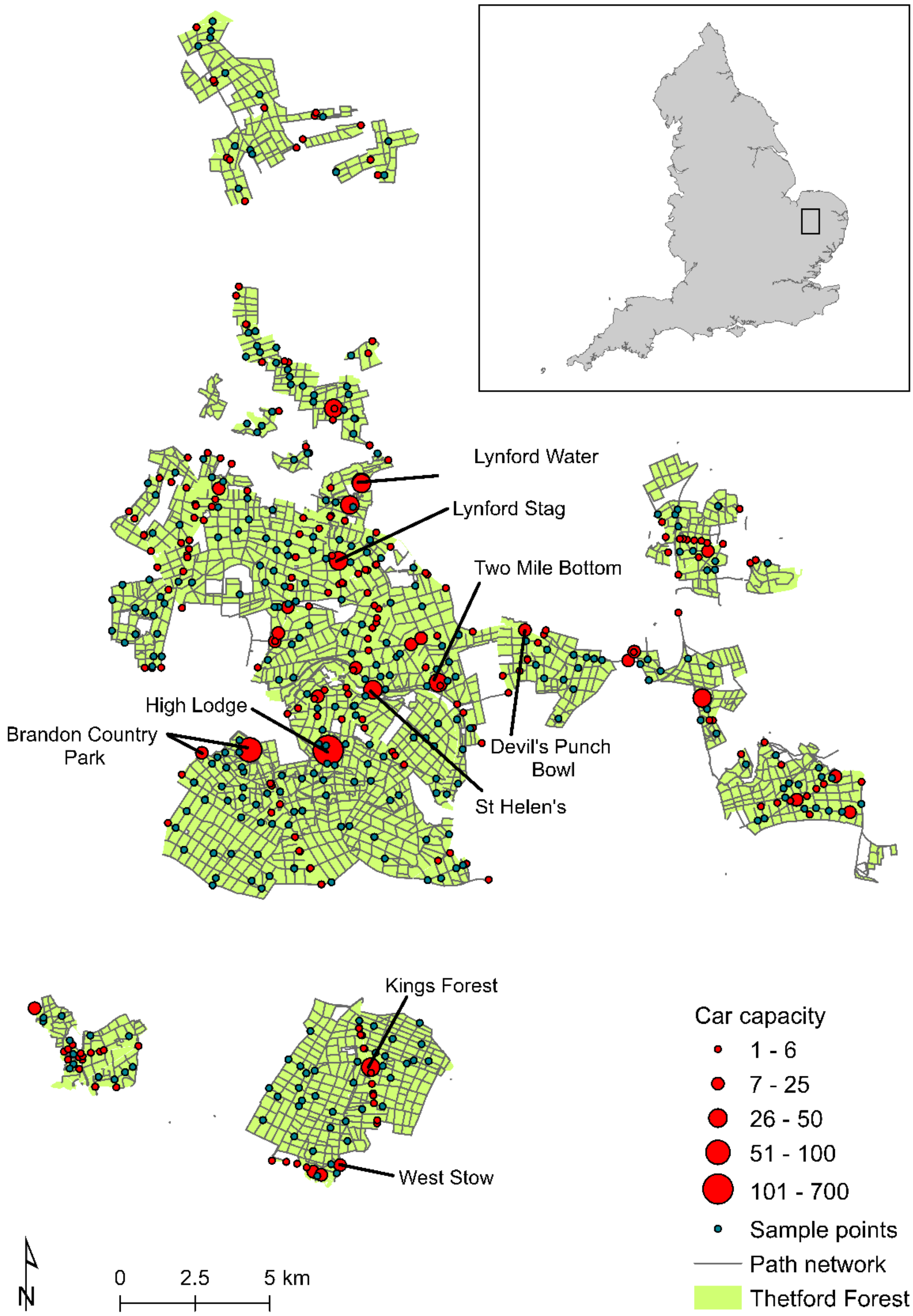

Fig. 5.1. Thetford Forest boundary, path network, access points with parking (red circles) proportionate to car capacity and sampling points used in model calibration. Locations of major car parks labelled. Inset shows location of Thetford Forest within England.

5.3.2 Generation of spatial explanatory variables

To generate potential explanatory variables for these recreational visits, several spatial layers were imported into ArcGIS 10.3 (Copyright © ESRI, USA). These included a point feature class delineating sampling points and a line feature class delineating the path network (initially derived from forest compartment boundaries and validated by ecological fieldworkers from UEA and FC staff including rangers with detailed knowledge of the entire forest path network). Path sections (path network segments delimited by intersections) were classified into path types according to the FC database (Forestry Commission, 2016; Table 5.1). We hypothesised that wide, well maintained forest roads (median verge width according to Wäber & Dolman (2015) is 6 m, range = 1-45 m, n = 459) and fire routes (3 m (1-9 m), n = 321) would be preferred by visitors to narrower (potentially overgrown or muddy) tracks (0.6 m (0.1-1 m), n = 205). Furthermore, as tracks are narrower, we hypothesised that tracks with cut vegetation along verges would be preferred over uncut, thus we further classified tracks as cut or uncut in each year of the survey (cutting regime feature classes were provided by FC East of England Office). Where roads intersect the forest separating it into blocks, links (line features) were added to join paths end-to-end across the roads. Links across ‘A’ roads (major arterial roads that comprise the top tier of the roads classification system in the UK) were *a priori* excluded as too dangerous to cross. Links were added across ‘B’ roads (the second tier of the classified roads system that connect A roads and smaller roads) and minor roads (smaller roads connecting residential areas to the road network). We hypothesised that B roads would present a greater barrier for recreationists to cross than minor roads.

A point feature class of access points with space for car parking was generated through visual assessment of all path-road intersections on Google Earth, determining whether or not this intersection provided an access point, and categorising car capacity through visual estimation after calibration through field visits. Access points were either formal or informal car parks (areas available for parking not within the bell mouth of a gateway or track; n = 34, mean capacity = 42.7 cars ± 116.13 SD), gateways where fire routes meet roads (n = 121, mean capacity = 2.8 ± 1.7 SD), or where tracks meet roads (n = 91, mean capacity = 2.2 ± 1.4 SD). Capacity estimates for large car parks were obtained from FC (Victoria Tustian, *pers. comm.*). The square root of number of cars was used to account for the diminishing effect of increasing car park capacity (i.e. the larger the car park, the less likely it is to reach full capacity and thereby limit the number of recreationists to the area). Access points with no space for parking were not retained as due to the size of the forest, distance from urban areas and barriers to pedestrian access created by major roads, most users arrive at Thetford Forest by car (84.9% of recreationists in a 2007 survey, n = 172; Dolman, Lake & Bertoncelj 2008).

Table 5.1. Path types used in classifying the Thetford Forest path network

Path Type	Definition
Forest road	Well-maintained, hard surface road
Fire route	All-weather, hard surface maintained for fire truck access
Track cut	Less well-maintained, usually grass or mud surface. Grass verges cut during survey year
Track uncut	Less well-maintained, usually grass or mud surface. Grass verges not cut during survey year

Recreationists travel from their home to access points where they can enter the forest; thus the larger the local source population surrounding access points, the more recreational activity is expected in nearby parts of the network. Number of households, taken from the 2011 census of households (Office for National Statistics 2011), which were linked to coordinates using the UK Postcode Directory (UKDS 2013), was summed within buffers around access points using 500 m radius increments, from 0 m to 5000 m, and wider 1,000 m radius increments from 5000 m to 10,000 m (15 buffers total). Due to co-linearity, several adjoining household buffers were merged based on a Spearman correlation coefficient above 0.6 providing 9 modified distance bands, that could be used as independent predictors in models of recreational frequency. To model effects of a scenario of increased housing, current household data were updated with new housing allocations (14,614 units) within 10 km of the Thetford Forest boundary planned up to 2026. Areas designated for new housing were provided as ESRI shapefiles by district councils and the number of proposed houses per development (polygon) were obtained from Site Allocation Plans. Points were generated randomly within each polygon to represent new postcodes, and the number of allocated houses were distributed evenly across points per polygon. Under this scenario there will be an increase in occupied households within 10 km of the Thetford Forest boundary of 15%. The sum of households within buffers around access points was then recalculated and used with other predictors unchanged to estimate future recreational frequency.

5.3.3 Modelling disturbance events

From observed visit data we summed the number of groups of walkers (≥1 person) to obtain a separate count of disturbance events (DE) for dog walkers and other walkers, on each path section joining the sampling point during the 1 hour survey. Records of cyclists were excluded, as they are more mobile (so network distance is unlikely to be a good predictor of counts), and sample size was low (only 6% of observations). The number of dog walker or walker DE per path section (dependent variable) was modelled separately in relation to day of week ('day'), time of day ('time'), month ('month'), a binary variable for school holiday, weighted network distance (net. dist.) from access point and the number of households in buffers around access points. School holiday dates were obtained from Norfolk County Council's online archive of school term dates (Norfolk County Council 2016). We built up models sequentially, first determining the appropriate model structure (in terms of coding and numbers of parameter levels within each control variable class) for what were expected to be the most influential and important effects, before going on to incorporate more subtle effects.

Day and time and month were expected to strongly influence number of DE, hence we first modelled counts of dog walkers and walkers in response to these, controlling for spatial pseudoreplication of sampling points by including point ID as a random effect in the model. The final specification was a Generalised Linear Mixed Effects Model, with Poisson error (fitted in R 3.2.3 using the lme4 package; Bates *et al.* 2013). An overdispersion function developed for mixed effect models (http://glmm.wikidot.com/faq) indicated that no overdispersion was present (theta < 2). For parsimony, day (categorical, 8 levels: Mon-Sun and national holiday) and time (categorical, 13 levels: 1 hour intervals between 06:00 and 18:00 coded as start of hour) were recoded, iteratively merging into fewer categories (based on examination of parameter coefficient similarity) if model fit was not significantly reduced. First modelling day with all levels, checking coefficients and standard errors of each category and sequentially merging those categories with large overlap in standard error; if there was no significant increase ($P > 0.05$) in residual deviance (tested by -2 x Log-Likelihood, -2 LLR) and also if Δ AIC did not exceed +2 or decreased, the merged category was retained. This resulted in 5 day categories for both dog walker and walker models: Mon (reference level), Tue/Fri, Wed/Thu, Sat, Sun/national holiday. Using the simplified coding for day, this process was then repeated merging time categories. Final coding of categorical time variables differed for dog walker and walker models, reflecting different patterns of recreational behaviour (dog walkers: 6 (reference level), 8/18, 9, 11/13/14/16, 12/17, 7/10/15, walkers: 6 (reference level), 7/8/18, 11/14/15/17, 13, 9/10/12/16). Month categories were also merged in this way, resulting in 5 levels for dog walkers (April (reference), May/June, July/October, August, September) and 4 levels for walkers (April (reference), May, June-September, October). After accounting for day, time and month no support was found for an effect of year when 'honey pot' sites were excluded (categorical variable, 5 levels: addition resulting in an increase in AIC of 5.6 and 3.8 for dog walkers and walkers respectively), hence recreational behaviour was consistent between survey years and year could be excluded from subsequent models. Furthermore, mean counts of dog walkers and walkers surveyed between 2007-2009 did not differ from matched (i.e. resampled) path sections in 2013-2014 surveys (dog walkers: Wilcoxon signed rank test, V = 348, $P = 0.17$, n = 63; walkers: Wilcoxon signed rank test, V = 213, $P = 0.15$, n = 63).

Local source population was expected to provide an important determinant of number of DE. Source population was measured as the number of households surrounding access points. To associate sampling points with an access point for joining household data, we used the 'Closest Facility' tool in ArcGIS Network Analyst to find the closest access point based on linear network distance along the path network. Once the closest access points were known, household count in distance buffers around access points (see section 5.3.2) were added as 9 predictor variables to the basic dog walker and walker models that controlled for day, time and month. Distance bands were then further merged in the interest of parsimony, again based on similarity of coefficients and change in AIC and -2 LLR. This resulted in the same 3 distance bands for both dog walker and walker models of 0-2000 m, 2000-6000 m and 6000-10000 m that were used as predictors (termed 'household number') in all subsequent models.

We hypothesised that path type in combination with distance to nearest access point, and car capacity at access points may also affect number of DE. Therefore a weighted network distance ('net. dist.') between sampling points and access points was calculated using the Closest Facility tool. Applying weightings in Network Analysis is equivalent to increasing or decreasing the linear network distance between sampling points and access points according to the characteristics of that route. This is similar to 'effective geographical distance' in least-cost modelling, which is the Euclidean distance (in meters) 'modified for the effect of landscape and behaviour' (Adriaensen *et al.* 2003). Weightings were applied for 1) car park capacity, 2) road crossings and 3) path type. Different combinations of plausible weightings were tested to explore the most appropriate estimate of impedance resulting from each of these factors, while continuing to control for source population, time, day and month. Road crossings were tested to determine whether roads present a semi-permeable (impedance) barrier to recreationists reducing crossing frequency or as an upper limit, entirely separating paths on opposite sides of the road. Total impedance to reach each sampling point was recalculated a number of times, systematically varying the weightings for the three components and each set was entered as an alternative net. dist. variable in separate walker and dog walker models controlling for day, time, month and household numbers. Data on household numbers was joined to the data set each time new routes were calculated, as the access point serving a sampling point (i.e. the 'closest' access according to that set of impedance weightings) could change under the new weightings. The weighting set that resulted in the lowest AIC (best model fit) was retained (see Appendix S5.1 in Supplementary Information) for the final dog walker and walker models.

5.3.4 Model evaluation and predictions

Correlograms of model residuals revealed that no spatial autocorrelation was present in the dog walker or walker model (see Fig. S5.1 in Supplementary Information). We evaluated the performance of the final GLMMs using 10-fold cross validation, randomly partitioning the data into 10 sets, building the model with 90% of the data (training) and validating with the remaining 10% (test). A different set was excluded from each of 10 model runs so that each observation was used for model validation exactly once. Model fit was evaluated after each run using a pseudo R^2, developed for GLMMs, which gives an estimate of the variance explained by fixed effects (marginal R^2) and both fixed and random effects (conditional R^2) (Nakagawa & Schielzeth 2013). Predictive performance was assessed by linear regressions of predictions on observations from the test datasets.

Extrapolation across the entire Thetford Forest path network required the generation of the net. dist. variable for every path section. The Closest Facility analysis was run using the weightings determined for dog walkers and walkers, to calculate the lowest impedance route to access points from the centre point of every path section. Data on household numbers were joined according to the access point associated with each path section. Predictions for dog walkers and walkers were made separately for each combination of time, day and month (resulting in 588 predictions per path section: for each

1 hour period between 06:00 and 18:00, all 7 days and 7 months), created once with school holiday set as 1 (holiday) and again as 0 (not holiday); resulting predictions were multiplied by the ratio of holiday (or non-holiday) days in the relevant month. The final holiday and non-holiday predictions were summed per combination. The average PDE h^{-1} per path section over the whole season was calculated from this final set of predictions, as well as PDE h^{-1} during the quietest and busiest periods, and presented as maps of dog walker and walker disturbance both separately and combined.

5.4 Results and Discussion

5.4.1 Weighted network analysis

Impedance weightings for car park capacity and path types differed between dog walkers and walkers, but road crossing weightings were the same (Appendix S5.1).

‘B’ roads present a substantial barrier to recreationists as no routes crossed a B road when using the weighting that produced the lowest AIC. There was no impedance, however, for crossing minor roads in the best fit model, so free movement of recreationists was allowed between parts of the forest path network nominally separated by minor roads.

The numerators for the inverse weighted car park capacity (see Appendix S5.1) differed for dog walkers (2,000) and walkers (4,500) such that low capacity car parks had a greater impedance for walkers than dog walkers, with the difference in impedance diminishing with increasing car park capacity (Fig. 5.2). This implies that walkers may be less familiar with informal car park access points (where tracks or fire routes meet roads) and tend to use larger car parks.

Path types differed in impedance weightings for dog walkers (forest road = 1, fire route = 0.9, track = 2) and walkers (forest road = 1, fire route = 0.5, track = 1). This could be interpreted as dog walkers preferring wide paths (forest roads and fire routes *cf.* tracks) to let their dogs run off lead and also they may be more habitual, using the same preferred route regularly, whilst walkers may perhaps ‘wander’ more than dog walkers and therefore use tracks more often. There was no additional impedance for tracks with uncut verges compared to cut verges, which may be due to the ephemeral nature of this management practice; cutting is only done in certain months, vegetation grows back, and not all tracks have their verges cut every year.

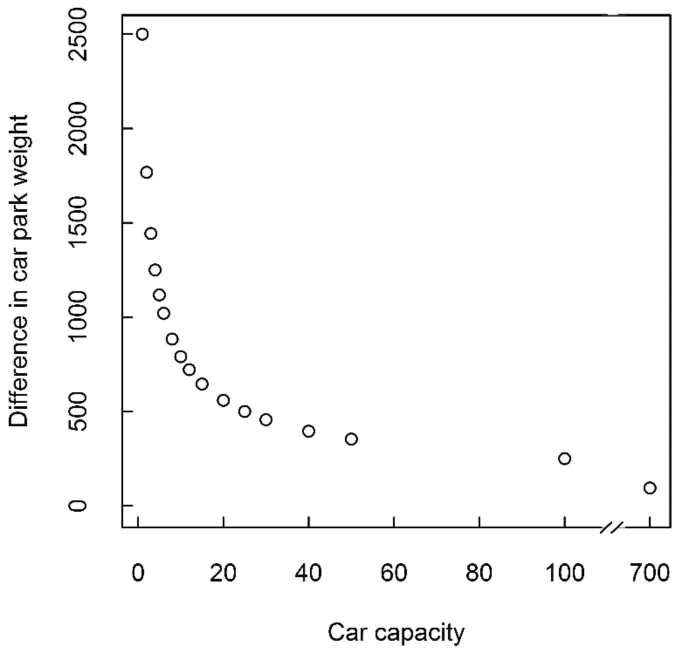

Fig. 5.2. Difference in car park impedance weight (walker car park weight – dog walker car park weight) at all car park capacities. Note the break in the x axis.

5.4.2 Models of recreational disturbance

Dog walker and walker models are shown in Table 5.2. Mean marginal R^2 from 10 fold cross validation was 0.13 for dog walkers and 0.10 for walkers, and mean conditional R^2 was 0.41 and 0.18. The beta coefficient from a regression of observed test data on predictions (both square root transformed) averaged 1.00 (95% CI: 0.86–1.14) (dog walker model) and 1.17 (95% CI: 0.58–1.50) (walker model). These values were used to re-scale all subsequent model predictions to correct for systematic under prediction.

The models reveal seasonal and temporal patterns in recreational use of Thetford Forest. The busiest hour for dog walkers is 09:00, in contrast that for walkers is 13:00, with disturbance from both activities highest on the weekends. More recreational activity occurs during school holidays for both dog walkers and walkers, as expected. 'Effective distance' increases with lower car park capacity at the access point and/or a route traversing tracks rather than fire routes or forest roads. Number of households within 2 km of access points (household numbers (0-2000 m)) is a strong positive predictor of number of disturbance events, but the effect of the number of households in the two more distal bands was non-significant and they were therefore removed. Not surprisingly, the greater the 'effective distance' (net. dist.) between sampling points and their associated access point, the fewer

Potential Disturbance Events (PDEs) h^{-1} occur on that path section. Considering the forest-wide extrapolation, the predicted number of PDEs h^{-1} decrease from a median of 0.5 (dog walkers) and 0.29 (walkers) with a net. dist. ranging between 0-1, to 0.25 and 0.14 with a net. dist. of 2.5 – 3 km (Fig. 5.3). To quantify the extent to which the localised distribution of housing around different parts of the landscape affects the spatial pattern of recreation within the forest, predictions for all path sections using the actual number of households per distance band, were contrasted with those predicted with the mean number of households per band (Fig. 5.4). When mean household numbers was used, PDEs h^{-1} for dog walkers did not exceeded 0.6 and walkers 0.35, whilst with actual household numbers, PDEs h^{-1} reached 1.0 and 0.5 respectively, 67% and 43% greater. Thus although net. dist. has a stronger influence on number of disturbance events than household number according to standardised coefficients, there is still a non-trivial effect of household distribution that locally enhances the density of PDEs.

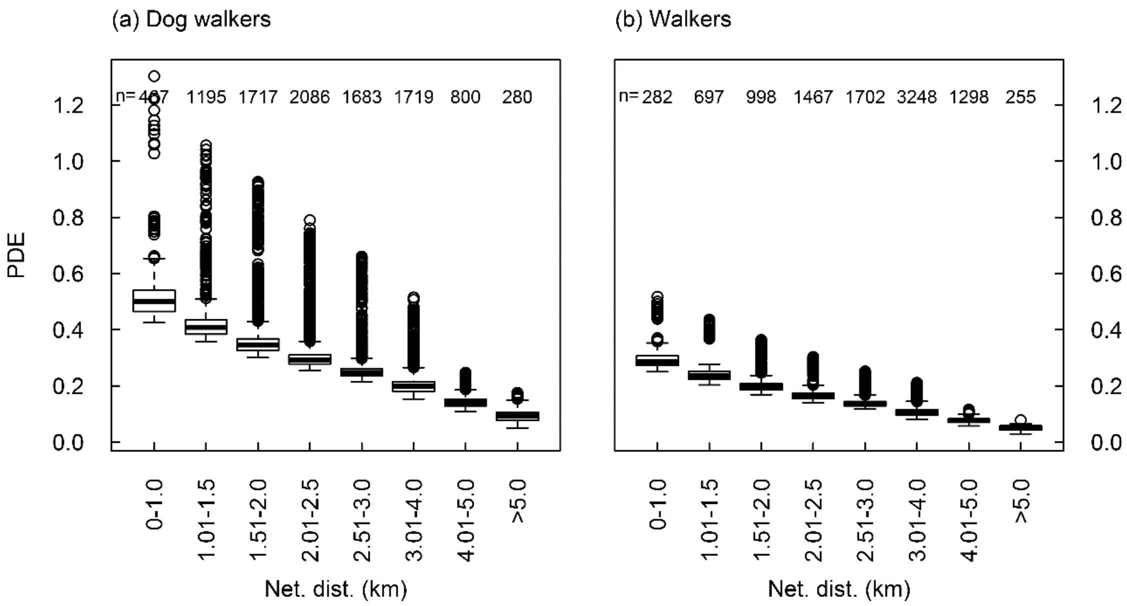

Fig. 5.3. Box and whisker plots showing median, quartiles and outliers of mean PDEs h^{-1} from (a) dog walkers and (b) walkers for different ranges of weighted network distance (net. dist.)

Table 5.2. Model parameter estimates

	Dog walker model				Walker model			
Variable	Category	Standardised Coefficient (Std. Error)	z	P	Category	Standardised Coefficient (Std. Error)	z	P
Time cat. (coded as time at the start of one hour survey period)	6	-	-	-	6	-	-	-
	7/10/15	0.328 (0.122)	2.68	**	7/8/18	0.466 (0.308)	1.514	
	8/18	0.518 (0.149)	3.469	***	9/10/12/16	1.126 (0.281)	4.009	***
	9	0.666 (0.132)	5.042	***	11/14/15/17	1.496 (0.28)	5.335	***
	11/13/14/16	0.17 (0.124)	1.366		13	1.981 (0.307)	6.457	***
	12/17	-0.323 (0.15)	-2.154	*				
Week cat.	Mon	-	-	-	Mon	-	-	-
	Tue/Fri	-0.006 (0.107)	-0.053		Tue/Fri	0.445 (0.189)	2.357	*
	Wed/Thu	-0.15 (0.108)	-1.39		Wed/Thu	0.589 (0.184)	3.207	**
	Sat	0.345 (0.12)	2.86	**	Sat	0.94 (0.209)	4.501	***
	Sun/national holiday	0.637 (0.111)	5.749	***	Sun/national holiday	1.951 (0.187)	10.426	***

Table 5.2. Continued over the page

Month cat.	April	-	-	-	April	-	-	-
	May/June	0.004 (0.08)	0.047		May	-0.746 (0.096)	-7.754	***
	July/Oct	-0.364 (0.101)	-3.59	***	JuneToSept	-0.485 (0.093)	-5.196	***
	August	-0.443 (0.119)	-3.731	***	October	-0.104 (0.33)	-0.314	
	September	-0.129 (0.113)	-1.139					
School holiday	No	-	-	-	No	-	-	-
	Yes	0.279 (0.082)	3.411	***	Yes	0.248 (0.082)	3.034	**
Net. dist. (m)		-0.655 (0.101)	-6.484	***		-0.732 (0.12)	-6.126	***
Household numbers								
(0-2000 m)		0.382 (0.088)	4.336	***		0.336 (0.109)	3.084	**
Constant		-2.87 (0.188)	-15.256	***		-5.35 (0.363)	-14.755	***

Dependent variable: Number of disturbance events in an hour from dog walkers or walkers. *P*<0.001 '***', *P*<0.01 '**', *P*<0.05 '*'

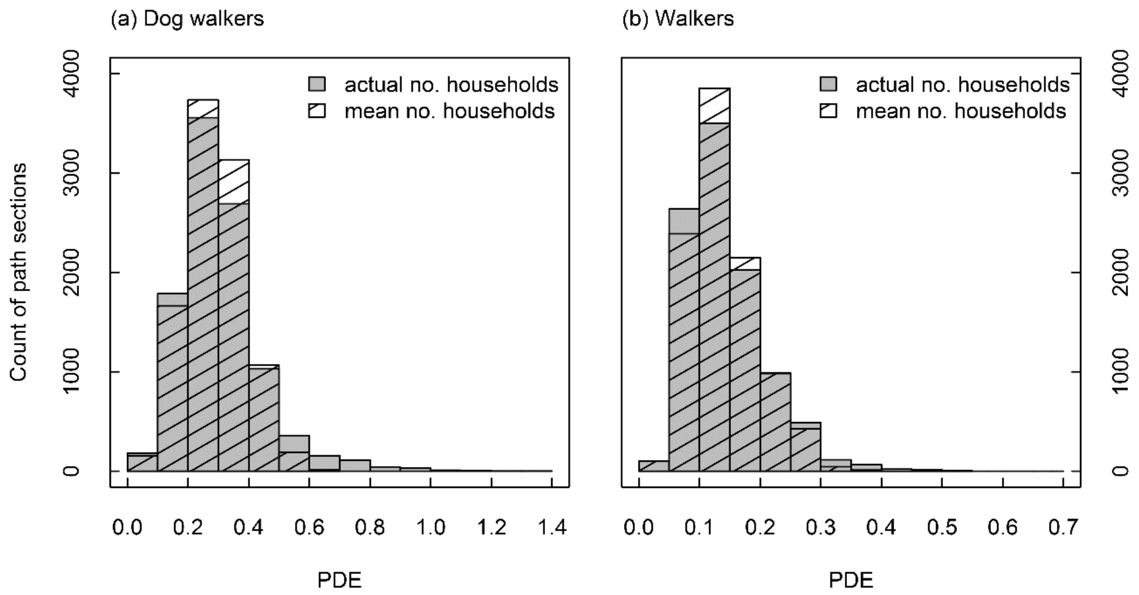

Fig. 5.4. Frequency histogram of PDE h^{-1} from (a) dog walkers and (b) walkers for all path sections in Thetford Forest, based on predictions using the actual number of households in buffers surrounding access points compared to predictions with the number of households held at the mean for each buffer.

5.4.3 Spatial distribution of recreational disturbance throughout Thetford Forest

The mean number of PDEs h^{-1} per path section are low, with few greater than 1, averaged across the week. The spatial distribution of dog walkers and walkers is similar; the mean number of PDEs h^{-1} from both is highly correlated (Pearson's r = 0.76, n = 9945, P<0.001), with the highest levels of disturbance from both in Brandon Country Park (Figs 5.5a and b). Mean PDEs h^{-1} are higher from dog walkers than walkers throughout the whole path network, particularly outside the 'honey pot' sites (areas that tend to have larger car parks), with an average of 2.2 ± 0.6 SD times as many dog walkers to walkers (Fig. 5.6a). However, as penetration distance into the forest increases, the ratio of dog walkers to walkers decreases (Fig. 5.6b) probably as dog walkers make more regular but shorter visits. During the quietest period (walkers: Tuesday and Friday at 7:00, 8:00 and 18:00; dog walkers: Wednesday and Thursday at 12:00 and 17:00), most path sections are expected to receive less than 1 disturbance event in 3 hours from dog walkers or walkers (Figs 5.7a and c). During peak times (walkers: Sunday at 13:00; dog walkers: Sunday at 9:00), walkers are still mainly distributed around honey pot sites, possibly due to the larger car parks, whilst dog walker PDEs h^{-1} exceed 0.5 for much of the forest (Figs 5.7b and d). Dog walker and walker PDEs h^{-1} were combined to obtain the total PDE h^{-1} per path section over the whole week and on Sunday, the peak day for both dog walkers and walkers (Fig. 5.8).

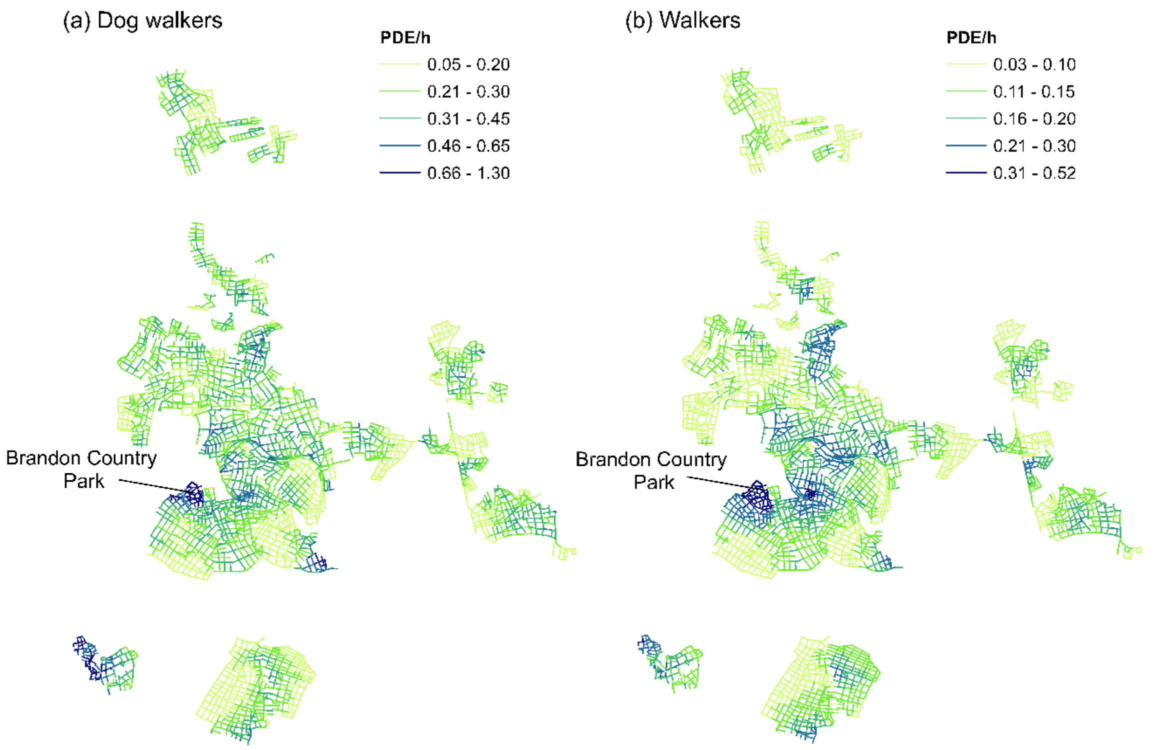

Fig. 5.5. Mean Potential Disturbance Events (PDE) h^{-1} in Thetford Forest from (a) dog walkers and (b) walkers.

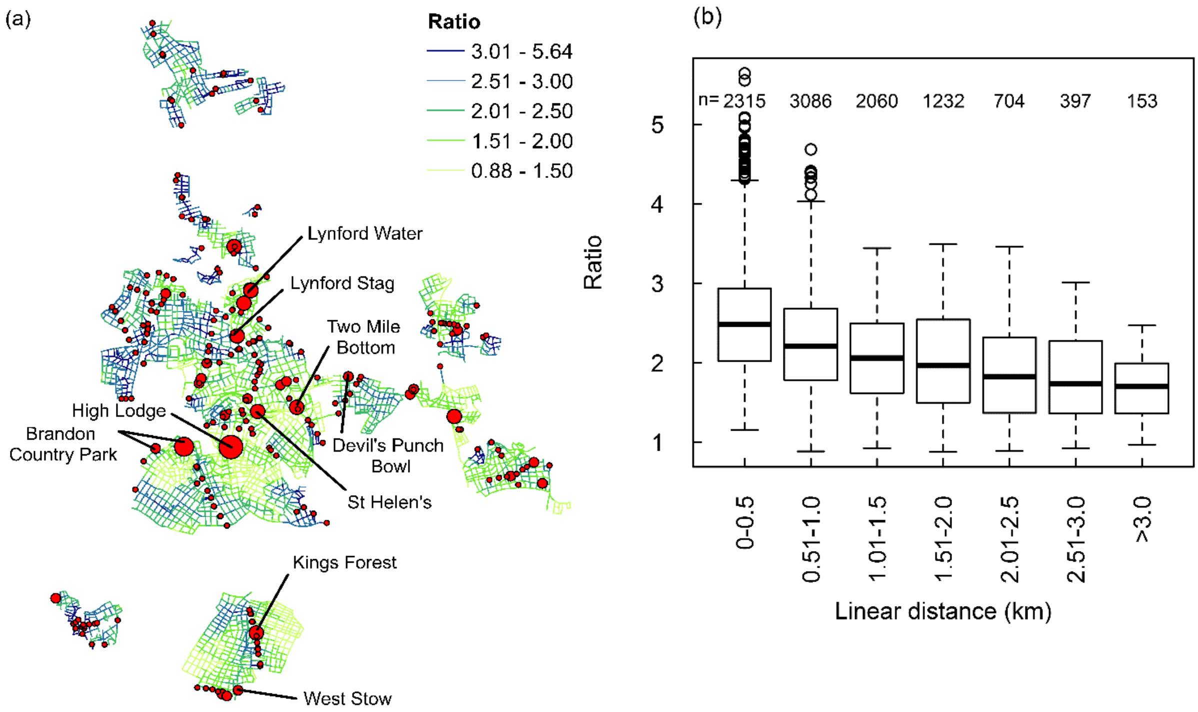

Fig. 5.6. (a) Ratio of dog walker to walker PDE h^{-1} in Thetford Forest. A positive value indicates more PDEs from dog walkers than walkers, (b) box and whisker plots showing median, quartiles and outliers of the ratio of dog walkers to walkers at different distances from the nearest access point.

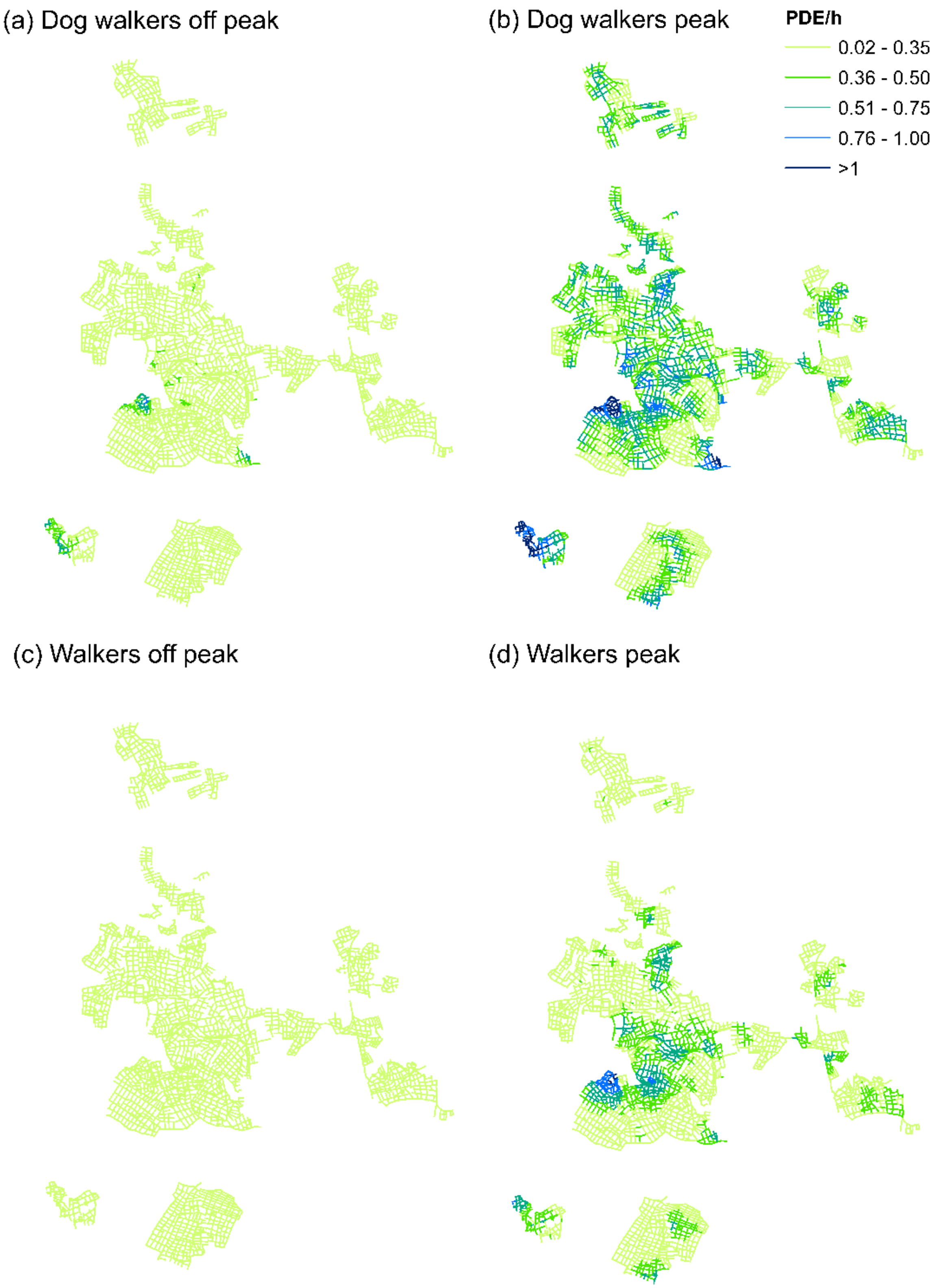

Fig. 5.7. PDE h^{-1} from dog walkers and walkers during the quietest and busiest periods of the week (mean across all months).

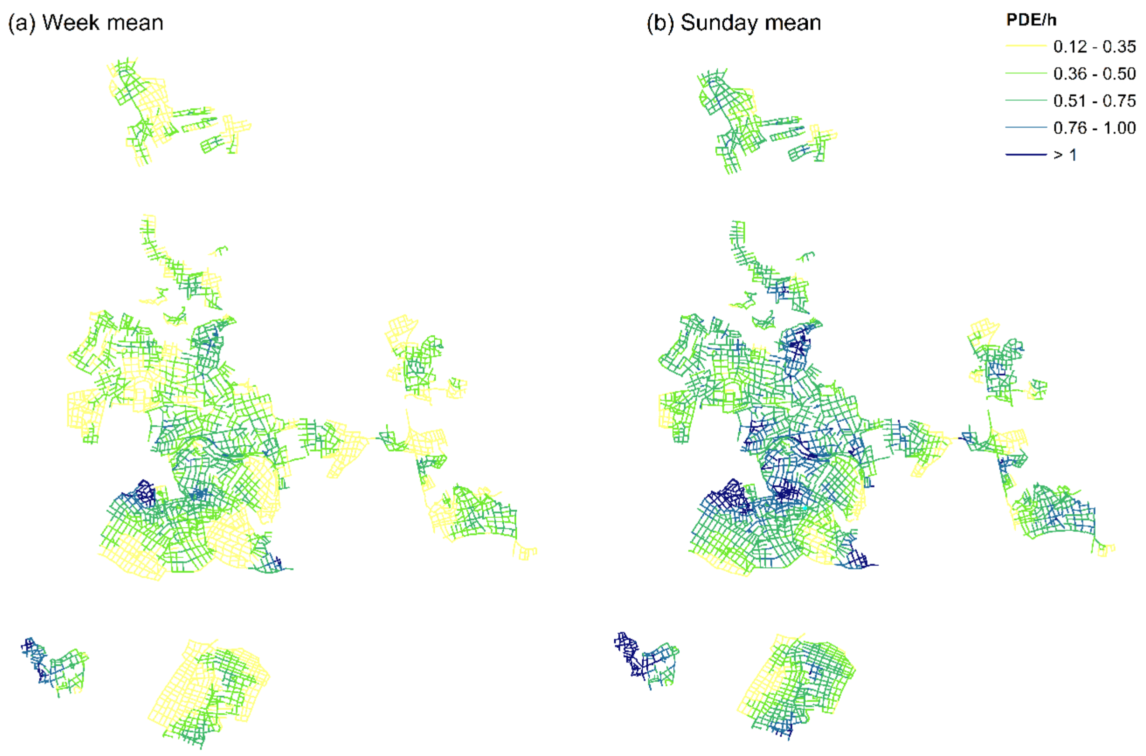

Fig. 5.8. PDEs h^{-1} from dog walkers and walkers combined (a) averaged over the whole week and all months and (b) averaged for Sunday (peak day) over all months.

5.5 Application of recreation model as a conservation management tool

Here we demonstrate the utility of our methodology as a planning tool, examining the potential consequences of forest management on territory settlement and size of a woodlark (*Lullula arborea*) population.

5.5.1 Background

The woodlark is a ground nesting passerine that forages in short vegetation with areas of bare or disturbed ground, and nests in patches of bracken, heather and long grass in heathland and within areas of clear fell and young re-stocks in plantation forests (Wright *et al.* 2007). In Europe, the woodlark is designated as a species of conservation concern following widespread declines in numbers and extent (Wright *et al.* 2007). The Thetford Forest SSSI population has been in decline since 1999 (Wright *et al.* 2009) and is below target according to Natural England’s latest assessment in 2010 (209 male song territories recorded *cf.* target for ‘appropriate condition’ of 253; Natural England 2010). This represents a 54% decline since the designation of the Breckland Forest SPA in 2000

(Dolman & Morrison 2012). In 2014, the population had declined further to 145 singing males according the Forestry Commission's (FC) annual survey (provided by FC East of England Office).

The FC is currently planning to increase the area of open habitat in Thetford Forest, which has the potential to provide suitable habitat for breeding woodlark. At present open habitats are under-represented in Thetford Forest at 7.12% of the total area, less than the 10% recommendation by the UK Forestry Standard (Forestry Commission 2011) and the UK Woodland Assurance Standard (UKWAS 2012). The FC plans to create an Open Habitat Network (OHN) through a novel approach of widening existing paths (*c.*278 km total length) to 40 m, which will create 751 ha of open space improving both quality of open habitats and connectivity of isolated populations (Armour-Chelu, Riley & Brooke 2014). However, providing open habitat in a linear configuration as part of a path network may reduce its potential for woodlark due to disturbance by recreationists preventing settlement so that potential territories remain unoccupied. Using our methodology we quantify the number of PDEs h^{-1} on path sections of the OHN, estimate the number of potential woodlark territories per path section lost under different disturbance thresholds and examine a scenario of future housing with possible mitigation through access point closure.

5.5.2 Predicting disturbance on the Open Habitat Network

Path sections (n = 1150) in the current path network designated for the OHN (OHN layer provided by FC East of England Office) were re-coded as fire routes to reflect their increased width and altered recreational potential. Then we re-applied recreation models to create predictions using this updated path network, obtaining PDEs h^{-1} for each OHN path section. Predictions were averaged only for April and May, the key woodlark breeding period (Brambilla & Rubolini 2009).

The OHN will provide permanent open habitat for woodlark, as following clearfelling the organic top layer (turf) will be removed to expose the mineral soil surface and periodic intervention will prevent succession to later stages (Pedley *et al.* 2013; Armour-Chelu, Riley & Brooke 2014). This is opposed to the open habitat created as part of the six year felling cycle, which is replanted with commercial timber species. Therefore it is important to assess its suitability under future conditions, for example the proposed increase in housing around Thetford Forest (scenario described in section 5.3.2). To examine the possible impact on woodlark we compared the number of potential territories lost under current disturbance levels with future levels that were obtained by substituting current household number predictors with future household numbers. Three different disturbance thresholds were tested, as the level of disturbance tolerated by woodlark in linear habitat patches is currently unknown (see section 5.5.3).

5.5.3 Woodlark disturbance

We first estimated the number of potential woodlark territories by dividing the length of each OHN path section by 500 m and rounding to the nearest whole number, so that with the target width of 40 m each territory was at least 2 ha (following Dolman & Morrison 2012). In complete absence of recreational disturbance and providing habitat conditions for settlement are optimal, the OHN has the potential to provide 554 woodlark territories, providing a huge opportunity to increase the woodlark population to the 253 pairs required to meet the SSSI target. As Thetford Forest is managed for commercial timber, there will be periods when the habitat adjoining the OHN path section will be clear felled or recently restocked. As forest coups are currently felled on a 60 year rotation (Eycott, Watkinson & Dolman 2006), and restock coups between 0-4 years are optimal for woodlark (Dolman & Morrison 2012), 6.7% of the OHN will adjoin restock coups all times. The number of territories created by the OHN was adjusted (decreased) by 6.7%, to 517 territories, as widened paths adjoining clear fell/young restock would not create additional territories, rather they will supplement habitat within existing potential territories in these coupes.

We quantified the number of potential territories lost under three theoretical disturbance thresholds (an average of 1 PDE in 1 hour; 1 in 2 hours; 1 in 3 hours), each of which lower than the estimate of 8.3 (95% CI: 5.8-10.9) disturbance events hour^{-1} determined by Mallord *et al.* (2006). This is because in the 2006 study, during a disturbance event birds were likely able to relocate within their territory by a short flight to seek refuge away from the path but still be in sight of their nest. For linear habitat bounding or spanning a path, as in this study, lack of a 'refuge' may substantially increase avoidance costs (requiring escape flight across the adjacent trees), which could result in greater demographic costs (Gill 2007). For each disturbance threshold we evaluated the potential impact on territory settlement, based on estimated PDEs h^{-1} on OHN path sections under current and future distribution of housing around access points (Table 5.3).

There is a huge difference in number of territories affected at different frequencies of disturbance (Table 5.3), but a considerable impact even at the moderate sensitivity threshold (41.1% unsettled). For both the moderate and low sensitivity thresholds, the total number of occupied woodlark territories would exceed the SSSI target of 253, but the high sensitivity threshold results in 72 territories below target under present household numbers and 74 below when including the impact of future housing. Determining the actual threshold is therefore crucial, which requires a systematic field study on woodlark nesting in linear open habitat with adjoining mature trees. The impact of future housing is trivial in comparison to disturbance thresholds, with a small proportionate increase in the number of territories affected (Table 5.3).

Table 5.3. Number (and percentage) of 517 potential woodlark territories not settled under current and future housing if sensitive to 1 PDE (Potential Disturbance Event) every hour, every 2 hours, every 3 hours, and total number of woodlark territories in Thetford Forest (OHN territories plus the 145 territories already settled in 2014).

Disturbance sensitivity	Frequency of PDEs	Current housing		Future housing	
		Territories not settled	Total no. territories	Territories not settled	Total no. territories
High	1 in 3 hours	481 (93.1)	181	483 (93.5)	179
Moderate	1 in 2 hours	213 (41.1)	449	227 (43.9)	435
Low	1 in 1 hour	3 (0.5)	659	5 (0.9)	657

As the high sensitivity threshold resulted in fewer total woodlark territories than the SSSI target of 253, we experimented with closing access points to investigate the number of closures required to reach the target. A total of 120 access points serve the OHN path sections with predicted disturbance levels exceeding the high sensitivity threshold. These were ordered according to the most territories impacted and sequentially removed from the GIS network to re-run the network analysis and re-predict PDEs h^{-1} following their 'closure'. One car park (High Lodge) supplied high numbers of path sections, but was not practical to close as this is a popular recreational area with visitor attractions (cafes and playgrounds etc.). Upon closure of the 35 access points (excluding High Lodge) serving the most territories (summarised in Table 5.4), the total number of woodlark territories would increase to 262, meeting (and slightly exceeding) the SSSI target.

Table 5.4. Number of each access point type that are closed due to serving the most path sections and number of territories affected pre-closure.

Access type	Total	Number of territories
Car park	7	101
Fire route gateway	20	186
Track gateway	3	31

5.6 Conclusions

This paper describes a novel methodology for predicting within-site spatial distribution of different recreational users within an extensive and complex site, which can be used for recreation management and conservation decision-making. The Network Analysis (NA) functionality of a GIS was combined with statistical modelling to incorporate impedances for important features (road crossings, path type and car park capacity) to obtain a measure of likely path usage driven by observations of recreational users on paths throughout the site. The mechanistic basis of the model then permitted extrapolation of predictions for all path sections across the 187 km^2 forest extent and provided a tool that could be used to explore consequences for the distribution of recreational activity under future scenarios, including housing development and access closure.

Conforming to prior expectations, disturbance rates were higher from dog walkers than walkers throughout the forest. However, the NA revealed some important differences in their recreational behaviour. Access points with low car park capacity represented a greater impedance to walkers than dog walkers; consequently as dog walkers more readily used informal parking at gateways which are more numerous than larger car parks, they were more evenly dispersed throughout the forest. This suggests that walkers could be dispersed more evenly throughout the site by providing more sign-posted larger capacity car parks in areas of the forest currently lacking such provision, although this may create new demand and uptake will depend on travel distance from recreationists' point of origin. Furthermore, B roads were a barrier to the movement of both dog walkers and walkers, thus access provision and distribution of recreationists can be managed independently in adjacent forest blocks where these are separate by a major (A) or B road. As expected, recreational activity was higher at weekends and during school holidays, but peak times differed between dog walkers and walkers. Disturbance from dog walkers was highest in the mornings in comparison to walkers, which peaked at mid-day; thus dog walkers are more likely to impact on the activity of ground nesting birds that tend to be more active earlier in the day. Furthermore, dogs are a potential nest predator (Dolman 2009) and may disturb nesting birds more than walkers without dogs, therefore knowledge of the different patterns in their activity is important for managing negative impacts.

The utility of our methodology as a planning tool was demonstrated through an application for woodlark conservation in relation to proposed habitat enhancement and creation. This showed how estimates of recreational disturbance can be generated under current and future conditions, and possible mitigation measures tested to aid conservation management. In this case study, recreational use of the forest post open habitat creation would only supress the woodlark population below the SSSI target if woodlark inhabiting linear habitat patches bounding a path are sensitive to one disturbance event every three hours. Determining the actual disturbance threshold through empirical research is crucial as this is a huge uncertainty at present. The future housing scenario did not result in a great increase in disturbance, perhaps because of the local effect of housing (i.e. households within 2000 m of an access point). In order to mitigate the effects of disturbance to increase the number of occupied territories in the high sensitivity threshold, 35 access points would need to be closed during the breeding season (April and May), seven of which are car parks. This may cause conflict with recreational users, especially walkers who are either less familiar with informal parking or prefer larger car parks (perhaps due to other facilities such as toilets, picnic benches etc. that are present). It is possible, however, to close a different combination of access points after consultation with site managers or rangers to achieve the optimal result for both wildlife and recreationists.

Modelling recreational use of routes presents many challenges, including the effort required to sample a large path network at the landscape scale (1,713 person-hours in this study, although the use of volunteers may make this practicable), and to generate a classified path network for use in the network analysis. Some path layers were available from FC (such as forest roads, fire routes, paths in the cutting regime) but the remainder of the paths had to be digitised and coded. Assessment of appropriate weightings for components of the network analysis was also time consuming. However, once the model is calibrated, predictions are simple and quick for extrapolations to larger areas and repeat runs for scenario testing. If similar recreation data is available for other forest sites to calibrate the model, this approach can be easily followed to determine disturbance levels in those sites and test the effects of management interventions.

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Supplementary Information

Appendix S5.1. Creation of weighted network distance variable

A ‘Closest Facility’ analysis was carried out in ArcGIS Network Analyst using access points as ‘facilities’, sampling points as ‘incidents’ and the forest path network as the ‘edge’ feature. Weightings were used for access points, road crossings and path type, so that sampling points were connected via the route of lowest impedance to an access point. The total impedance for reaching a sampling point is termed ‘weighted network distance’, abbreviated to ‘net. dist’. (Eqn 1).

Net. dist = access point weighting + road crossing weighting + ∑(type weighting for path section$_i$ * length of path section$_i$) (Eqn 1)

access point weighting = x / SQRT(car capacity) (Eqn 2)

where x is the empirically determined weight based on model fit (see below).

We determined weightings for access points, road crossings and path type in turn using a three step process, fixing the weighting for a component at each step, starting with access points. Access points had an estimated capacity of 1-700 cars. We considered high car capacity to be less of an impedance to accessing a particular part of the forest, due to lower chances of reaching full capacity. The majority of access points had a capacity of 10 cars or less (n = 226), a few had moderate capacity (>10 but ≤ 50, n = 18) and just 2 had a capacity > 50 cars (Brandon Country Park: 100, and High Lodge: 700). Therefore we took the square root of car capacity. As access point weighting is an additive term in Eqn 1, but high car capacity should have a lower impedance, we took the inverse of SQRT(car capacity) (Eqn 2), varying the numerator to generate different weightings for access points.

A ‘weight’ field was added to the path layer in the GIS to contain the weighted values for path section type. These values were initially set to linear distance (m). No road crossings were initially included in the network. When loading the access points as ‘Facilities’ in the Closest Facility analysis, the access point weighting field was selected for the attribute ‘weight’. Sample sites were loaded as ‘Incidents’ with no weight attribute. Accumulation attributes were set to both ‘length’ and ‘weight’ to store the linear distance between access points and sampling points, as well as the total impedance (i.e. net. dist.; Eqn 1) per route. The Closest Facility analysis was repeated using the different weightings for access points (Table S5.1), with a new layer of routes produced each time. Net. dist. from the route layers was entered into models that included time, day and number of households in 3 distance bands around access points. The net. dist. that resulted in the best fit model (lowest AIC) for dog walkers and walkers separately, was retained (Table S5.1).

Table S5.1. Numerators tested in inverse weighted car capacity (Eqn 2)

AIC	Numerator							
	1000	1500	2000	2500	3000	4000	4500	5000
Dog walker	6718.2	6716.1	$\boxed{6714.6}$	6718.6	6717.8	6717.9	NA	NA
Walker	NA	NA	NA	NA	4086.9	4081.7	$\boxed{4081.1}$	4084.3

The second step was to determine the impedance caused by crossing roads that run through the forest. Links (line features) in the path network were added so that path sections were joined across roads, classified as A roads, B roads or minor roads. We *a priori* excluded links across A roads as being too dangerous to cross. We set various combinations of weightings for B roads and minor roads by updating the weight attribute for links in the path layer (Table S5.2). We repeated the Closest Facility analysis, this time using the access point weighting determined in step 1 and varying road crossing weightings in each run, again generating alternatives for the net .dist variable. We re-ran dog walker and walker models replacing the net. dist. variable each time, finding that the best fit weightings did not allow crossing of B roads, but minor roads had no impedance (Table S5.2).

Table S5.2. Weightings for road crossings

	Weighting 1	Weighting 2	Weighting 3	Weighting 4	Weighting 5
A road	NA	NA	NA	NA	NA
B road	10,000	10,000	10,000	1000	5000
Minor road	0	100	200	0	0
AIC (dog walkers)	$\boxed{6706.8}$[a]	6707.2	6707.7	6707.3	6706.8[a]
AIC(walkers)	$\boxed{4071.4}$[a]	4071.9	4072.4	4070.7[b]	4071.4[a]

[a] No routes crossed B roads with a weighting of 5000 or above

[b] At a weighting ≥ 1000, a single route crossed over a B road from a large car park (in King's Forest). From personal observations, recreationists go into the forest from the side of the road that the car park is located, without crossing the road, therefore weighting 1 was chosen over weighting 4.

The final step was to determine path type weightings. We hypothesised that recreational appeal of fire routes > forest roads > tracks. Path section length was multiplied by a weighting according to its type - a number of weighting combinations were tested, first varying the track weighting (Table S5.3) and then the fire route weighting (Table S5.4). The Closest Facility analysis was repeated for each combination (including access point and road crossing weightings) to get an updated total impedance for modelling. Tracks had a higher impedance for dog walkers than walkers according to AIC, but fire routes had a slightly lower impedance (Tables S5.3 and S5.4). We then went on to test whether tracks with cut verges were preferred by recreationists by increasing the weighting for uncut tracks. This involved running the Closest Facility analysis five times (once for each year surveyed) using different track weightings each time according to the tracks cut in that year. The resulting net. dist. data was joined with other variables by sampling year for modelling. However, this lead to an increase in AIC in both dog walker and walker models, thus tracks were not split by cut/uncut in the final path type weightings.

Table S5.3. Weightings for tracks

Class	Comb. 1	Comb. 2	Comb. 3	Comb. 4	Comb. 5	Comb. 6
Forest road	1	1	1	1	1	1
Fire route	1	1	1	1	1	1
Track	1	1.25	1.5	1.75	2	2.25
AIC (dog walkers)	6706.8	NA	6701.1	6699.2	$\boxed{6698.1}$	6698.4
AIC (walkers)	$\boxed{4071.4}$	4076.0	4077.2	4080.7	4081.5	NA

Table S5.4. Weightings for fire routes

Class	Comb. 6	Comb. 7	Comb. 8	Comb. 9	Comb. 10	Comb. 11
Forest road	1	1	1	1	1	1
Fire route	0.9	0.8	0.7	0.6	0.5	0.4
Track	Best fit (Table C)	Best fit (Table C)	Best fit (Table C)	Best fit (Table C)	Best fit (Table C)	Best fit (Table C)
AIC(dog walkers)	$\boxed{6697.9}$	6698.4	6698.8	6698.8	NA	NA
AIC (walkers)	4070.7	4070.2	4069.7	4069.1	$\boxed{4069.0}$	4069.1

Once the weightings were fixed, the net. dist. to the access points with the second and third lowest impedance was generated in the Closest Facility analysis. However, these were significantly correlated (Table S5.5) and thus not included in models.

Table S5.5. Pearson correlation of weighted network distance ('net. dist') from sample sites to access points with the lowest, second lowest and third lowest impedance

	Dog walker net. dist.		Walker net. dist.	
	Lowest	Second lowest	Lowest	Second lowest
Second lowest	0.71		0.68	
Third lowest	0.54	0.71	0.50	0.79

Appendix S5.2. Supplementary figures

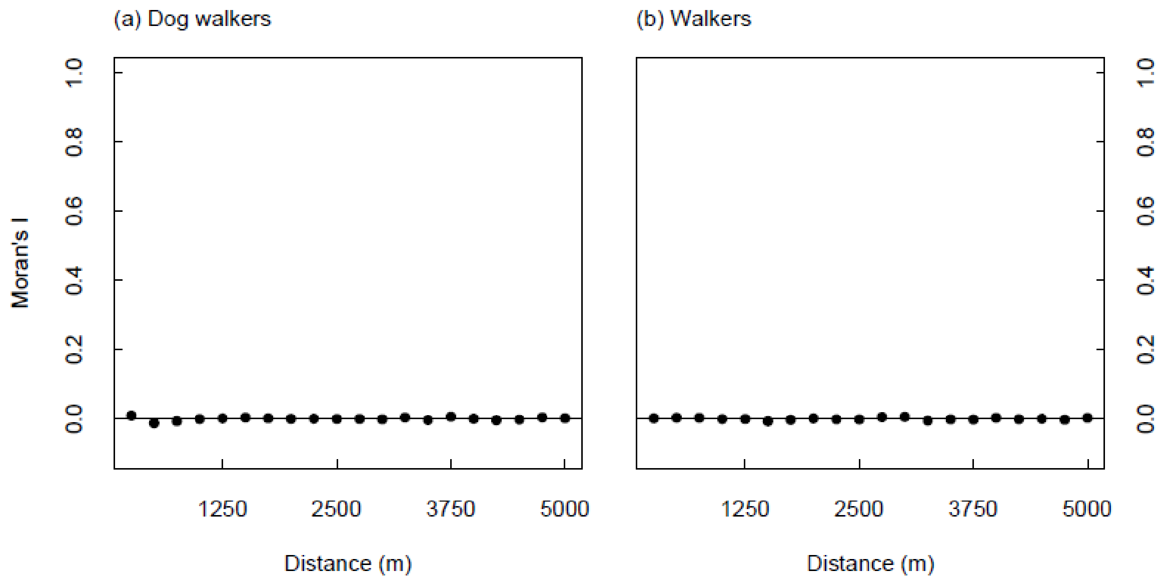

Fig. S5.1. Correlograms from residuals from (a) dog walker model, (b) walker model

Chapter 6:

Concluding remarks

Recreational use of protected and (semi-)natural areas presents one of the greatest challenges for countryside management, which must address the needs of both recreationists and nature. Whilst participation in outdoor recreation is important for human health and wellbeing (Tzoulas *et al.* 2007; Keniger *et al.* 2013) and raising public awareness of conservation issues (Thompson 2015), recreationists can negatively impact the very environment they seek to enjoy (Pickering & Hill 2007; Reed & Merenlender 2008). In Europe, this is compounded by high population densities and high demand for countryside access. This thesis explored recreational use of the countryside from a conservation perspective to improve mechanistic understanding of the distribution of recreational demand at a variety of spatial scales from national to site, and propose solutions for reconciling recreation and biodiversity conservation. This was achieved by modelling a national coverage data set of recreational visits to the countryside, which identified causal mechanisms for recreational site selection (Chapter 2). This model was used to predict recreational pressure across the English countryside (Chapter 3) and under a future scenario of localised population increase and house building in Norfolk (Chapter 4) to identify if protected areas or particular habitats are, or will be, subjected to high levels of recreational pressure and how this may be relieved. A site scale study then explored the distribution of recreational use within an extensive forest landscape (Chapter 5), providing a mechanistic model capable of testing the effectiveness of experimental manipulation of recreational pressure for endangered species recovery.

6.1 Key findings and conservation implications

6.1.1 Importance of nature value for recreation

Chapter 2 provides no evidence that high nature value is of importance to recreationists for everyday recreational visits. This is the first time this has been documented through empirical research on recreational site use, using a national-level model calibrated from a massive and nationally representative data set of over 30,000 visits collected over 3 years (Natural England 2012). This study contributes to a relatively small body of work investigating the extent to which nature value is important for recreation. There is conflicting evidence regarding whether people find biodiversity attractive (Lindemann-Matthies, Junge & Matthies 2010; Qiu, Lindberg & Nielsen 2013) and whether biodiversity enhances wellbeing (Fuller *et al.* 2007; Dallimer *et al.* 2012; Carrus *et al.* 2015), but until this current work, the sparse evidence on recreational use and biodiversity all indicated that there is a positive association (Naidoo & Adamowicz 2005; Siikamäki *et al.* 2015). The disparity between these findings and those presented in Chapter 2 can be explained in terms of the types of recreational sites and users studied. Whilst the aforementioned studies focused on visits to large protected areas, effectively capturing ‘nature tourism’, Chapter 2 was based on all recreational visits to the wider countryside of which nature tourism is likely to be a small subset. It therefore included visits such as regular dog walking, short family walks, picnicking etc., which are the vast proportion of visits to semi-natural areas in the UK. Many of these visits were local as 82% of respondents surveyed reported travelling less than 5-8 km, therefore the influence of longer distance, nature tourism trips was greatly diluted. Furthermore, the aforementioned studies were conducted in Finland and Uganda, which are very different to the UK in terms of recreational behaviour and provision of recreational opportunities; the UK does not have such vast areas of wilderness, but rather a matrix of ‘green infrastructure’ within close proximity to residential areas. This study, therefore, does not contradict these findings, but rather addresses a gap in knowledge regarding the importance of biodiversity for day-to-day recreation. The implications are that biodiversity loss may negatively impact nature tourism, but not everyday recreation, and the general public does not obtain benefits from conservation efforts in the form of recreational opportunities. To gain public support and justify investment in biodiversity conservation, non-use values of biodiversity should be promoted; non-use values remain important, as demonstrated by the general public’s willingness to pay to increase the area of sites managed to enhance biodiversity (Garrod & Willis 1997).

A positive conservation implication is that pressure on vulnerable conservation areas may be relieved through provision of recreational opportunities in low nature value sites. By encouraging recreational use of sites that have the capacity to absorb recreational pressure, recreational site provision can be maintained, whilst redirecting pressure away from conservation areas. Preferences were revealed for beaches, lakes and rivers, and broadleaved woodland, in accordance with Sen *et al.* (2014), which can be targeted for recreational site provision either through increasing accessibility or creation of new sites. It was not within the scope of this thesis, however, to quantify the effects of new recreational opportunities.

6.1.2 National-level predictions reveal spatial patterns in recreation but are poor at investigating change

Application of the model developed in Chapter 2 to create the first conservation-focused nationwide map of recreational pressure on the countryside revealed that sites close to urban areas are under the most intense recreational pressure, with relatively fewer visits to the wider countryside (Chapter 3). This is due to the data used in model calibration, which captured local day-to-day recreational visits, not nature tourism; 82% of respondents surveyed reported travelling less than 5-8 km. Similarly, in a national survey conducted in Denmark, 98.1% of daily green space visitors travelled less than 1 km (Schipperijn *et al.* 2010). It is not surprising therefore that nationally, peri-urban green space receives the greatest numbers of day-to-day recreational visits. As urban and peri-urban green spaces offer refuges to many species in highly urbanised areas and contribute significantly to biodiversity (Kong *et al.* 2010), conservation of such sites is important, particularly in light of the high levels of recreational pressure shown here.

Future population projections at the local authority level failed to reveal meaningful changes in patterns of recreational visits across the country, due to the course resolution of the future population data and the inability to forecast the specific locations of for example new housing developments. This could not be obtained nationally due to the devolved planning system in England and associated difficulties in obtaining housing plans from all composite unitary authorities. A simulation approach using cellular automata (CA) models (Santé *et al.* 2010) could have been employed to predict land use change associated with urban growth as used by Clarke & Gaydos (1998), Barredo & Kasanko (2003) and Guan et al. (2011) amongst others (see review by Santé *et al.* 2010). CA models are however complex (Santé *et al.* 2010) and simulations are subject to errors due to uncertainties regarding future conditions and changes in current trends; therefore the solution used in this thesis involved a smaller scale study for which strategic growth locations for the entire area could be obtained (Chapter 4).

6.1.3 Housing development plans must include mitigation for nearby green spaces that are predicted to experience an associated increase in recreational pressure

In Chapter 4, large increases in recreational pressure were predicted around planned housing developments in peri-urban green spaces, and it was revealed that developments are predominantly planned on agricultural land. Agricultural land is classed as low in the habitat type band and distinctiveness category in England's current biodiversity offsetting framework, thus it has a lower number of 'biodiversity units' per hectare and a lower offsetting requirement (Defra 2012). However, this chapter demonstrates that as the new inhabitants of the development are likely to visit habitats of higher biodiversity units nearby, such as woodland, mitigation for developments based only on the land cover being developed is likely to be inadequate. Consideration of the wider impacts of developments in terms of increases in recreational pressure on surrounding areas appears to only have been considered in the context of European protected areas (e.g. Liley & Tyldesley 2011); it is vital that this issue receives more attention from planners, practitioners and governments to ensure that due consideration is given to the further reaching impacts of housing developments on green spaces and appropriate mitigation measures are implemented. With the current rate of urbanisation (almost 40,500 dwellings planned in Norfolk alone up to the year 2026) this should be made a research priority.

A limitation of the recreation data used for the national-level model on which these predictions were based, is that only a single location (visit point) was provided. In order to determine the likely area visited, the point was buffered; however, we still had no information on where recreationists would go within this 'site'. To investigate within-site recreational distribution, finer resolution data was acquired at the site scale for the study reported in Chapter 5.

6.1.4 Network Analysis can be combined with statistical modelling of on-site observational data to predict within-site recreational distribution

In Chapter 5 a novel methodology is developed for predicting spatial distribution of different recreational users at the site level, which combines the Network Analysis (NA) functionality of a GIS with statistical modelling. This is the first study to incorporate many different factors that influence likely path usage and make spatially-explicit predictions using models based on observations of recreational users throughout a site. The Network Analysis combined multiple factors that determine the most likely route taken by recreationists into a single measure, with impedances for different characteristics of that route determined by model fit. The methodology developed, using Thetford Forest as a case study, overcame the challenges presented by an extensive and complex site with dense path network of different quality paths and over two hundred access points of varying car park capacity. The resulting model and predictions have applications for both recreation management and conservation decision-making.

The mechanistic basis of the model provided a tool that could be used to explore consequences for the distribution of recreational activity under future scenarios, including housing development and access closure. Application of this tool for woodlark (*Lullula arborea*) provided several insights relevant to woodlark conservation and site management in general. Predictions showed that recreational disturbance in Thetford Forest is unlikely to limit the potential of proposed habitat creation for providing additional woodlark territories; only if woodlark are highly sensitive to disturbance (1 disturbance event every 3 hours) in the newly created habitat patches (which will bound paths in a linear strip) would mitigation be required. In which case, the model was able to re-predict disturbance levels to test selective closure of access points. Successful redistribution of recreational pressure was achieved upon closure of 35 access points, reducing disturbance at potential territories below the hypothetical critical threshold and proving the capacity of the model as a management tool. Thus in areas where paths are already in existence and designing a path network that will have minimal impact on wildlife (as suggested by Thompson 2015) is not possible, closing access points can be a subtle yet effective way of redistributing recreationists, whilst maintaining visitor levels. This approach is also flexible, as access points may be closed only during important life stages when disturbance is most harmful (Knight & Temple 1995). The tool has a great capacity for future development and additional conservation applications.

6.2 Contribution to methodological development

6.2.1 National-level recreation model

This thesis has contributed to an understanding of how spatial recreational visit data can be used to develop mechanistic models to facilitate understanding of the drivers of recreational site selection (Chapter 2). Furthermore, the function developed is transferrable, capable of making spatially-explicit predictions of recreational visits at out of sample locations (Chapters 3 and 4).

The approach developed in this thesis is different to that employed in the environmental economics literature, but retains the underlying principles. In economic valuations of recreation, the travel cost method is widely used (Milcu *et al.* 2013), which takes into account the actual expense incurred by recreationists to travel between start and destination locations, as well as the opportunity cost of time spent travelling, to construct a consumer demand curve (Brainard, Lovett & Bateman 1999). Travel cost, along with visitor and site characteristics have been modelled to produce 'trip generate functions' capable of predicting recreational demand (visit counts) for sampled (and potentially also non-sampled) sites (Bateman, Lovett & Brainard 1999; Brainard, Bateman & Lovett 2001; Hill & Courtney 2006; Jones *et al.* 2010; Sen *et al.* 2014). Visit counts are then converted to a monetary value (Bateman, Lovett & Brainard 1999; Jones *et al.* 2010; Sen *et al.* 2014). The travel cost approach makes generalisations in the calculation of these variables by using census blocks, measuring from the centroid when calculating travel distance and using the socio-demographic characteristics of each census unit, rather than an individual's home postcode or specific socio-demographic data in order to make predictions to out of sample areas (Brainard, Lovett & Bateman 1999).

This thesis took a site focused approach employing a used-available design of visited versus available recreation sites, as the principle aim was to determine site characteristics important for recreational site selection. Nevertheless, travel cost was accounted for in the distance-weighted population function, and variation in visitor characteristics through the inclusion of county as a random effect. The purpose here was to control for associated variation, hence the use of a mixed effects model in which socio-demographic variables were not explicitly measured and modelled as fixed effects. This simple approach negated the need for computer processor heavy calculations of road network distance, and modelling of multiple socio-demographic variables, which nevertheless have uncertainties and inaccuracies due to the use of census blocks. The disadvantage of the site-focused approach is that models do not incorporate where recreationists originate from, which the travel cost method does, and knowledge of the demand as well as supply is key to investigating substitute effects.

The method developed here can be transferrable to other countries, subject to good quality, nationally available spatial data sets. Recreation data of comparable geographic extent and long-term fine-scale temporal resolution will be most difficult to obtain. The household survey technique used by Natural England is costly, requiring a great deal of resources (surveyors, hand held computers etc.), which other countries may not be able to invest in. Other data sets may be available, however, such as mobile phone data which can provide large volumes of data on people's locations (Toole *et al.* 2012). Fine scale national-level land cover data may also be difficult to obtain, although technology is advancing in this respect. At present though, even in the United States, which is at the forefront of spatial data development, nationally available land cover data sets are low resolution with poor accuracy for some land cover classes (Troy & Wilson 2006).

6.2.2 Site-level recreation model

A powerful tool for recreation management and conservation decision-making was developed in Chapter 5 capable of predicting within-site spatial distribution of recreationists. The mechanistic basis of the model allows exploration of the consequences of management interventions or future scenarios on the distribution of recreational disturbance, as demonstrated using a real world problem. The use of the Network Analysis functionality of a GIS to determine routes taken overcomes the issues associated with use of GPS units or visitor reported routes (discussed in section 1.4.1), and the difficulties associated with combining multiple factors to determine most likely routes taken.

An alternative approach, known as agent-based or individual-based models, has been used in the conservation literature to predict conflicts (i.e. interactions) between recreationists and wildlife using computer simulation modelling. In agent-based models, autonomous agents move around a simulated environment making decisions on where they go and what they do based on a set of pre-defined behavioural rules (Gimblett *et al.* 2000; Cole & Daniel 2003). These autonomous agents may be virtual people or animals, which are programmed to respond to environmental change in a realistic way (Bennett *et al.* 2009). The characteristics of the study site are represented as a virtual environment using spatial layers created in a Geographical Information System (e.g. maps of habitat types, footpaths, locations of available breeding sites). Agents are parameterised using theoretical and/or empirically derived behavioural rules (e.g. types of paths used for a particular recreational activity or amount of time a species spends foraging in a day) and may be assigned demographic, physiological or habitat-specific characteristics (Bennett *et al.* 2009). Agent-based modelling was applied to assist reserve design through predicting the levels of disturbance-related behaviour in yellow-headed blackbirds (*Xanthocephalus xanthocephalus*) resulting from recreational use on seven different proposed circular path systems (Bennett *et al.* 2009). The same model (SODA: Simulation of Disturbance Activities) was employed to test design and management options for a proposed recreational area with a nesting colony of black-crowned night-herons (*Nycticorax nycticorax*) (Bennett *et al.* 2011).

Whilst agent-based models are excellent at testing a variety of scenarios to elucidate possible disturbance impacts, they are complex, requiring a high level of computer programming expertise to develop (e.g. Itami *et al.* 2003; Bennett *et al.* 2009; Van Kirk & Douglas 2014) and once developed may not be particularly user friendly for stakeholders (Edwards & Smith 2011). They also do not preclude the need for baseline field data, as both the environment and the agents it is populated with need to be simulated and assigned attributes, and where empirical data is lacking, expert judgement is obtained which may be overly subjective or biased. The methodology developed in this thesis provides a statistical approach, for non-computer programmers with knowledge of classical statistical techniques.

6.3 Limitations and future research directions

There are several opportunities for building upon the work presented in this thesis. Firstly, in Chapter 2 we suggest that provision of substitute sites could relieve pressure from conservation sites. However, it was not possible to test this as only land cover within a site (400 m circumference circle) contributed to predicted visit probability – if land cover outside the site changed, the site's probability would not be affected. This issue of substitute sites has been addressed in the economic valuation literature through inclusion of variables for percentage of different habitats occurring in the outset area (Jones *et al.* 2010; Sen *et al.* 2014) or travel time to other attractions, such as National Trust sites (Jones *et al.* 2010). For this current study, an attempt was made to model substitute sites by measuring the proportion of different land covers in 10 km buffers around sites (i.e. the wider landscape), but positive site-level predictors (Fig. 6.1a) were non-significant at the landscape scale (Fig. 6.1b) whilst negative site-level predictors were positive at the landscape scale. As the addition of landscape scale variables did not significantly affect site-level coefficients (Fig. 6.1a) and did not add much interpretative value, the model excluding landscape variables was used in this thesis. Addition of desirable substitute sites (e.g. a woodland) using this (10 km buffer) approach would not have affected the probability of the focal site being visited due to their non-significant effect (Fig. 6.1b); yet

Sen *et al.* (2014) predicted that creation of a 100 ha woodland would attract over 200,000 more visits per annum and reduce visits from surrounding areas. Hence substitution effects clearly require further investigation, but with emphasis on reducing visits to conservation areas experiencing high levels of visitation.

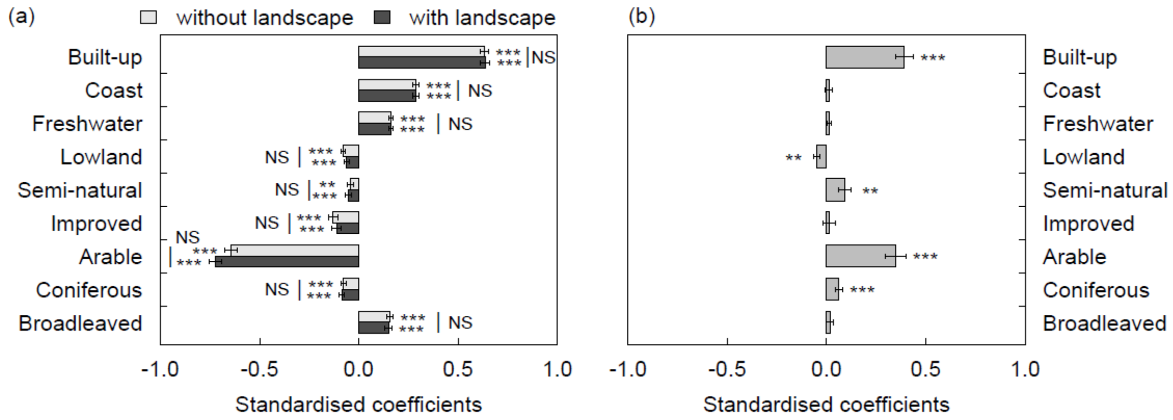

Fig. 6.1 Standardised coefficients from an alternative model specification from Fig. 2.2, which includes landscape variables (proportion of land cover within 10 km buffers around sites). (a) Effects of site-level land covers on site visitation probability with and without surrounding landscape variables included, and (b) effects of landscape-scale land covers on site visitation probability (bars denote standard error; P<0.001 '***', P<0.01 '**').

Secondly, the national-level recreation model was based on visits that recreationists undertook within seven days preceding the household interview in order to reduce recall bias (Natural England 2012). Consequently, the recreational visits that were captured were mainly short-distance, local trips and the subsequent model predictions can be interpreted as 'everyday' recreational visits. By focusing upon local visits, the number of visits to areas that people travel a long way to access were under-predicted, as highlighted in Chapter 3. As discussed by Van Herzele & Wiedemann (2003), green spaces at different spatial levels serve different functions, and those sites used in everyday life are unlikely to be used for weekend recreation or occasional day trips or holidays. A major extension of this thesis would be to model nationwide recreational visits at the different functional levels proposed by Van Herzele & Wiedemann (2003), i.e. green spaces of different minimum land area in different distance bands from urban areas. This may reveal a positive relationship between recreational use and nature value for larger, more remote areas, that would be in line with the findings of Naidoo & Adamowicz (2005) and Siikamäki *et al.* (2015). The methodology may need to be adapted slightly, for example, reconsideration of the buffer size for sites in different functional levels. Development of separate models would give a more holistic view of recreational use of the natural environment in England and provide greater insight into whether high biodiversity areas are under more recreational pressure than low biodiversity areas.

Some improvements could be made to the site-level recreation model presented in Chapter 5 for Thetford Forest. More intensive sampling of honey pot sites is required to better estimate the upper disturbance levels, which may be underestimated for some areas of the forest (although Thetford Forest does receive a much lower volume of visits compared to other heathland/woodland SPAs in England). The initial projects from which the bulk of the data were obtained sampled areas where birds were known to be present (Dolman 2010), but not uninhabited areas (that most likely experience high levels of recreational use). For this thesis these areas were sampled to supplement the existing data, and hotspots of intensive use were identified, but the models systematically under predicted the number of visits when compared with observational test data in cross validation. Furthermore, more investigation could be made into modelling the path verge cutting regime, as this would provide a useful management strategy if an effect is found; paths could be left to become overgrown as a subtle way of reducing recreational use. In the current study, applying weightings for uncut paths in the Network Analysis resulted in poorer model fit, but as vegetation grows back throughout the season there may be an interaction between month and cutting which could not be examined when fitting the network distance variable to the entire year's data.

Further work is required to improve the applicability of the site-level model for real-life management applications. Estimation of the actual disturbance threshold for woodlark inhabiting linear habitat patches bounding recreational paths is required, as well as collaboration with the Forestry Commission to test management scenarios relevant to their objectives, informed by their site-specific knowledge. The Forestry Commission would also be able implement the recommended interventions to test the effect of closing access points to provide validation for the model. Further investigation is also required to determine whether the model may be transferrable to other recreational forests, or if it needs to be recalibrated due to differences in recreational behaviour at these sites.

This thesis provides new methodological frameworks for spatial analysis of recreational data at a variety of spatial scales from national to site level. These contributed to a greater understanding of the factors that influence recreational use of the countryside. Of greatest note is the lack of support for high nature value enhancing everyday recreational use. This has far reaching implications both for justification of biodiversity conservation and recreation management. This thesis also demonstrates how GIS-based spatial approaches can provide tools for practical applications aimed at balancing recreation and conservation interests.

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Appendix A:

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Recreational use of the countryside: no evidence that high nature value enhances a key ecosystem service

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Abstract

In Western Europe, recreational amenity is presented as an important cultural ecosystem service that, along with other values, helps justify policies to conserve biodiversity. However, whether recreational use by the public is enhanced at protected areas designated for nature conservation is unknown. We report the first study to model outdoor recreation at a national scale, examining habitat preferences with statutory designation as an indicator of nature conservation importance. Recreationists preferred areas of coast, freshwater, broadleaved woodland and higher densities of footpaths and avoided arable, coniferous woodland and lowland heath. Although sites with conservation designation had similar or greater public access than undesignated areas of the same habitat, statutory designation *decreased* the probability of visitation to coastal and freshwater sites and gave no effect for broadleaved woodland. Thus general recreational use by the public did not represent an important ecosystem service of protected high-nature-value areas. Intrinsic and existence values remain as primary justifications for conservation of high nature value areas. Management of ‘green infrastructure’ sites of lower conservation value that offer desirable habitats and enhanced provision of footpaths, could mitigate recreational impacts on nearby valuable conservation areas.

Introduction

Nature-based recreation is presented as an important cultural ecosystem service [1,2] that supports investment in biodiversity conservation [3,4]. Evidence is, however, surprisingly scarce [5]. Interacting with nature benefits physical health (reducing stress levels and mortality), cognitive performance (reducing mental fatigue) and well-being (elevated mood and self-esteem) [5,6]. On a global scale, visits to protected natural areas (PAs) are on the increase [4] and there is evidence that PAs holding greater levels of biodiversity are preferentially visited by nature-based tourists [7,8]. However, whether the general public making every-day recreational visits place greater value on areas designated to protect or conserve biodiversity (henceforth high nature value), versus the wider countryside is unknown. This is especially important in Europe where there are many opportunities to pursue recreational activities in other types of 'green space'.

Recreationists can have undesirable effects on high nature value areas [9,10] that may be mitigated by re-distributing recreational pressure to other areas of lower nature value; yet public access to nature is essential to build a constituency for conservation [11,12] and use gives amenity value with potential to generate conservation revenues [4]. Strategic management of recreational provision would be strengthened by better understanding the importance of high nature value areas relative to the wider countryside. PAs across England deliver biodiversity benefits but fewer recreational visits than predicted from their relative area [13]. However, recreational benefits were not evaluated as analyses did not control for local population density, that was twice as high in the vicinity of visits to the wider countryside than visits to PAs, and also did not control for effects of access networks or preferred land cover types. Sen et al. [14] assessed the economic value of recreation, modelling land cover class, travel distance, socio-demographics and population; but did not examine whether conservation status affected visitation preferences.

Here we identify the ecological preferences underlying recreational behaviour, using a nationwide sample of over 30,000 spatially referenced visits to the countryside to model the influence of land cover on the probability of site visitation, controlling for transport and footpath networks, source population density and regional behavioural differences. We then examine how high nature value affects likelihood of site visitation, using statutory designation of Sites of Special Scientific Interest (SSSIs) as a proxy. SSSIs represent the UKs most important sites for biodiversity conservation [15], are designated using objective criteria and include all National Nature Reserves (NNRs) and Natura 2000 sites designated under European Directives. We aim to determine whether recreational use is an important ecosystem service provided by high nature value areas relative to the wider countryside. Irrespective of whether recreation amenity provides additional justification for conservation, understanding which habitats are in greatest demand informs the provision of green infrastructure and recreation opportunities to mitigate recreational pressure on vulnerable conservation areas.

Methods

1.1. Study design

We used a case-control design [16] to compare recreational visit localities with randomly selected countryside localities (controls). Point visit locations were taken from the Monitor of Engagement with the Natural Environment (MENE) survey (2009-2012) of recreational activity by English households [17]. A nationally representative sample of face-to-face, in-home interviews, were conducted each week of the year. During each interview, the location was obtained of one recreational visit selected randomly, providing grid references (Ordnance Survey National Grid) for 44,485 visit locations that we mapped as points in ArcGIS 10.1 (Copyright © ESRI, USA). We excluded visits in predominantly built-up areas (see S1 Appendix in Supporting Information) to focus on the remaining 31,502 countryside visits (hereafter ‘visit points’). Twice as many controls (63,000) were generated (hereafter ‘control points’) using Geospatial Modelling Environment (GME) [18], randomly located within the boundaries of England but constrained to be at least 25m from visit points so that control points could not be placed in a known visit location. Controls within predominantly built-up areas were excluded in the same way as for visits (Fig 1). A quasi-experimental design was tested also, with control points stratified by distance-weighted population, a combined measure of travel cost and population density surrounding visit points (see below); but random controls were considered superior as an explicit measure of population and travel cost could be included in models (see S1 Appendix). As recreationists generally visit an area not just one point, visit and control points were buffered by a 400m radius, informed by empirical evidence of visitor countryside access patterns (see S1 Appendix). Buffered visit and control points are hereafter referred to as 'visit localities' and 'control localities', or jointly as 'localities'. Due to the heterogeneous representation of land cover within each visit locality, statistical matching to pair protected locations with sites having similar characteristics was infeasible.

Fig 1. Distribution within England of visit and control points used in this study

1.2. Predictors of visitation

Land cover was extracted for all localities from the 25m resolution Land Cover Map 2007 (LCM2007) [19]. The 22 LCM2007 land cover classes were aggregated into 11 broad classes as some were not distinguished reliably using spectral signature (e.g. neutral, calcareous, acid and rough grassland, grouped as semi-natural grassland) [19] and others were limited in area (e.g. supra-littoral rock; grouped with littoral rock). Land cover classes appearing in fewer than 10% of visit localities were excluded from analysis due to insufficient power (following Boughey et al. [20]), so that 9 classes remained (Table 1). Land cover classes were further divided by designation as SSSI (e.g. broadleaf woodland SSSI, broadleaf woodland non-SSSI) using SSSI boundaries from Natural England [21], with the exception of built-up land and improved grasslands that are not statutorily designated as SSSIs and arable, which is rarely designated. SSSIs cover more than 8% of the country with the majority (98% of total area) designated for biodiversity (e.g. richness, representativeness) and or nature conservation (e.g. species of national or international conservation concern) [22].

Table 1. Candidate variables used to model likelihood of site visitation by recreationists.

Code	Predictor	Units	Description
Comp	Arable[a]	Proportion	Proportion of annual and perennial crops and freshly ploughed land
	Coast[a]	Proportion	Proportion of sand dunes, shingle, littoral mud and littoral sand
	Broadleaved woodland[a]	Proportion	Proportion of broadleaved woodland with >20% tree cover or >30% scrub cover
	Built-up[a]	Proportion	Proportion of urban and suburban areas including towns, cities (and residential gardens), car parks and industrial estates
	Coniferous woodland[a]	Proportion	Proportion of coniferous woodland with >20% cover

	Freshwater[a]	Proportion	Proportion of lakes, canals, rivers and streams
	Improved grassland[a]	Proportion	Proportion of grassland modified by fertiliser and reseeding typically managed as pasture or mown
	Lowland heath[a]	Proportion	Proportion of heather and dwarf shrub, gorse and dry heath below 300m a.s.l. as defined by Gimingham (1972), delimited according to the digital terrain model OS Terrain 50 [25]
	Semi-natural grassland[a]	Proportion	Proportion of neutral, calcareous, acid and rough grassland
Pop	Weight.pop.2[b]	No. people	Total number of people residing within 10km of the site, inverse-weighted by straight-line distance from visit and control points
Cty	County[c]	85 levels	County in which the site is located
Path	Path.length[d]	m	Total length of path network within site
Elev	Mean.elev[e]	m	Mean of all Digital Terrain Model 50m cells within site
Road	Dist.Aroad[f]	m	Distance from visit and control points to nearest major road

[a]LCM2007

[b]km resolution population raster created from 2011 ONS census data

[c]Assigned according to county boundaries downloaded from http://www.gadm.org/

[d]OpenStreetMap

[e]OS Terrain 50

[f]OS Meridian

Given that we are examining recreational visits between SSSIs and non-SSSIs we examined potential access constraints between these. Areas either having a statutory right of access under the Countryside Rights of Way Act (CRoW), or as Country Parks or Local Nature Reserves, that together cover 8.5% of England were mapped and the proportion cover compared between SSSI and non-SSSI land, while controlling for land cover type. In a further subsidiary analysis we divided land cover classes by designation as National Nature Reserves (NNRs). NNRs are high-quality SSSIs used to showcase conservation management and engage the public and thus are areas of high nature value where recreational access is encouraged. These results are compared with those using SSSI designation.

Outdoor recreation in the UK is enabled by a network of public rights of way. A path network layer encompassing bridleways, cycleways, footpaths, paths and tracks was extracted from OpenStreetMap [23], the only national digital dataset delineating public rights of way. These data, collected by contributors using GPS devices, aerial imagery and field maps, were validated against rights of way shown on OS raster maps (see S1 Appendix). Site accessibility was indexed as the path network length within localities and additionally as the straight line distance from visit and control points to the nearest major road (A Road) [24].

Mean elevation of localities (extracted from OST50 [25]) was selected *a priori* as a predictor variable. Localities with lower mean elevation were expected to have a greater probability of visitation, as those engaging in arduous activity are a subset of recreationists. Elevation gain within localities was explored but provided less explanatory power.

The larger the resident population in the vicinity of a site the more likely it is to be visited, diminishing with distance. Population data from the 2011 census of households provided by the Office for National Statistics [26] (England and Wales) and National Records of Scotland [27] were linked to coordinates, using the UK Postcode Directory [28], and aggregated into 1km grid cells to create a UK population raster (some visits close to the borders may originate from Wales or Scotland). Of three population distance-density functions tested, weight.pop.2 (population weighted by inverse of distance squared) best distinguished visit from control localities and was included in all subsequent models (see S1 Appendix).

1.3. Analysis

Generalised linear mixed models (GLMMs) with binomial error and logit link function predicted $P(\mathrm{Visit}_i)$, the probability of a recreational visit to site *i*, as a function of the proportions of locality land cover classes (Comp_i), mean elevation, distance from nearest major road and path density (fixed effects, Table 1), controlling for distance-weighted population and county (random effects). Counties (categorical, 85 levels, from database of Global Administrative Areas [29]) vary in area from $28\mathrm{km}^2$ to $7965\mathrm{km}^2$ with a mean population of 644,944. The interaction between weight.pop.2 and county allowed for potential differences in *per capita* frequency of recreational activity due to socioeconomic or cultural effects. Predictor variables were centred and scaled with zero mean and unit standard deviation for comparability of coefficients [30]. Two GLMMs were fitted; in model 1 all land cover classes were included once, in model 2 selected land cover classes were divided into areas designated and non-designated as SSSIs:

$$P(\mathrm{Visit}_i) = \mathrm{f}(\mathrm{Comp}_i, \mathrm{Elev}_i, \mathrm{Road}_i, \mathrm{Path}_i, \mathrm{Pop}_i, \mathrm{Cty}_i) \quad \text{(model 1)}$$

$$P(\mathrm{Visit}_i) = \mathrm{f}(\mathrm{Comp.non\text{-}des}_i, \mathrm{Comp.des}_i, \mathrm{Elev}_i, \mathrm{Road}_i, \mathrm{Path}_i, \mathrm{Pop}_i, \mathrm{Cty}_i) \quad \text{(model 2)}$$

Differences between equivalent non-designated and designated (model 2) land cover coefficients, were evaluated by Z tests.

GLMMs were fitted using the lme4 package [31]. Inspection of correlograms established that spatial autocorrelation was negligible in both models (see S1 Appendix). Predictive performance of the two models was evaluated against independent data from the subsequent 2012-2013 MENE survey (n = 10,622) and additional random controls (n = 10,622). For each model, AUC - the area under the receiver operating characteristic (ROC) curve - was calculated using the pROC package in R [32]; AUC ranges from 0.5 for models that perform no better than random, to 1 for models with perfect discrimination [33]. Whether AUC values (and thus model prediction accuracy) differed significantly among models was tested (following DeLong et al. [34]) within the pROC package.

Results

1.4. Recreationists' preferences for site characteristics

Examining effects of locality characteristics upon visitation probability without considering designation status indicated a strong positive influence of path density (mean within visit localities 2055m ± 1916 SD; within control localities 604m ± 865 SD, Table 2). Visitation probability was strongly reduced for localities at higher elevation or far from a major road. Intercepts for each county ranged from -1.45 ± 0.16 95% CI to 1.14 ± 0.24 95% CI and weight.pop.2 coefficients from -1.67 ± 0.95 95% CI to 2.26 ± 0.61 95% CI (S1 Fig), showing variation in *per capita* visitation probability between counties and supporting inclusion of these random effects.

Table 2. Generalised linear mixed model predicting recreational demand in the countryside, controlling for population and county.

	Standardised Coefficient	Std. Error	z	P
Non-land cover variables				
Path length	0.826	0.014	59.96	***
Elevation	-0.370	0.017	-22.22	***
Distance to major road	-0.132	0.013	-9.83	***
Land cover classes with positive effect				
Built-up	0.631	0.022	29.14	***
Coast	0.287	0.016	18.49	***
Freshwater	0.161	0.010	16.26	***
Broadleaved woodland	0.158	0.015	10.37	***
Land cover classes with negative effect				
Arable	-0.645	0.031	-20.70	***
Improved grassland	-0.129	0.022	-5.80	***
Lowland heath	-0.080	0.012	-6.64	***
Coniferous woodland	-0.078	0.013	-6.18	***
Semi-natural grassland	-0.043	0.016	-2.73	**
Constant	-0.697	0.077	-9.06	***

Dependent variable: the likelihood of visitation. $P<0.001$ '***', $P<0.01$ '**'

Of the semi-natural land cover classes, coast had the strongest positive effect on the probability of visitation, followed by freshwater and broadleaved woodland. Probability of visitation was 50% at proportionate covers of coast and freshwater of 0.11 and 0.15 respectively (Figs 2a & 2b), whereas a greater cover of broadleaved woodland (approximately 0.43 and above; Fig 2c) was required before a locality was more likely to be visited than not. Arable had the strongest negative effect on visitation probability, with a large effect size relative to other land cover classes (Fig 2d). Recreationists were less likely to visit localities comprising a greater proportion of lowland heath, improved and semi-natural grassland or coniferous woodland.

Fig 2. Predicted influence on visitation probability of coast, freshwater, broadleaved woodland and arable. From model 1 (see text) controlling for path length, elevation, distance to nearest major road, weighted population and county. Bars show the frequency distribution (square root scaled) within visit (unfilled) and control (grey) sites. Predictions were obtained by varying the proportionate cover of the land cover class shown between 0-0.8. All other land cover classes were held proportional to their mean such that they sum to 0.2 (so that total land cover proportion did not exceed 1). Control variables were held at their mean. Horizontal box and whisker plots show median, quartiles and outliers of land cover proportions in visit (unfilled) and control (grey) sites

1.5. Effect of conservation designation

We then examined preferences for land cover classes of potential conservation importance separately, according to whether they were SSSI designated. Explanatory power increased (Δ AIC = -186), with a slight increase in predictive ability (Model 1: AUC = 0.8425 ± 0.005 95% CI; Model 2: AUC = 0.8430 ± 0.005 95% CI; $Z = 2.44$, $P < 0.05$). For important land cover types 6-20 times as much SSSI area was open access (coast, 6.2%; freshwater, 17.1%; broadleaved woodland, 29.7%; lowland heath, 86.0%) than for equivalent land cover not designated as SSSI (0.3%, 1.4%, 5.2% and 4.1% respectively) so that, all else being equal, a greater visitation rate would be expected.

The appeal of broadleaved woodland was similar irrespective of whether it was designated an SSSI ($Z = 0.7$, $P = 0.47$; Fig 3) with the coefficient similar between these models ($\Delta = -0.013 \pm 0.018$ SE), whereas the attractiveness of coast and freshwater was significantly greater when not designated (Fig 3). While non-designated coast and freshwater coefficients were close to the original (model 1) coefficient error bounds, SSSI-designated coefficients were lower (designated versus non-designated: coast $\Delta = -0.188 \pm 0.024$ SE, $Z = 7.8$, $P < 0.001$; freshwater $\Delta = -0.079 \pm 0.013$ SE, $Z = 6.2$, $P < 0.001$). Effects of designating the freshwater or coastal area within a locality was examined separately for low (20%) and high (80%) overall cover, holding remaining land cover classes constant in proportion to their national mean. Freshwater designation minimally affected visitation probability at low cover (0.559 non-designated, 0.518 designated) but at high cover visitation probability decreased more with designation (0.900 non-designated, 0.813 designated; S2 Fig). Coastal designation substantially reduced visitation probability (low cover: 0.748 non-designated, 0.544 designated; high cover: 0.996 non-designated, 0.865 designated; S2 Fig). The negative effect of semi-natural grassland did not differ with designation, whereas coniferous woodland and lowland heath were significantly more negatively associated with visitation probability when designated ($Z = 4.45$, $P < 0.001$ and $Z = 2.20$, $P < 0.05$ respectively). Despite the encouragement of access within NNRs, subsidiary analysis contrasting land covers designated or non-designated as NNRs was consistent with SSSI results for broadleaf woodland, coast, freshwater and semi-natural grassland, but lowland heath and coniferous woodland NNR-designation had no significant effect on visitation probability (S3 Fig).

Fig 3. Effects on visitation probability of non-SSSI-designated or SSSI-designated land covers. Standardised coefficients from model 2 (see text) controlling for path length, elevation, distance to nearest major road, weighted population and county. Bars denote standard error. For each land cover, *P* values of *Z*-tests compare pairs of coefficients between non-SSSI-designated/SSSI-designated (P<0.001 '***', P<0.01 '**', P<0.05 '*')

Discussion

For preferred land cover classes we found no evidence that high nature value areas had greater appeal despite having greater levels of permitted open access. Recreation was previously found to be under-represented by protected areas in England, but in analyses that did not control for source population density, road access or footpath density [13]. Controlling for these factors, we now provide clear evidence that high nature value (inferred by statutory designation as SSSI) does not confer additional recreational value for the general public. This has important implications for justifications of biodiversity conservation.

When a land cover was of elevated conservation importance recreational use by the wider public was not enhanced and in the case of coasts and freshwater it was less likely to be visited. Thus while the public sought access to the countryside or greenspace, this was independent of the nature conservation quality of these locations. Dallimer et al. [35] found no consistent relationship between species richness and human well-being in a survey of visitors to riparian greenspaces, but a positive effect of *perceived* richness. Conservation importance may not strengthen the broader cultural service of recreational opportunities obtained from ecosystems if this is not recognised or sought by most recreationists or the general public. Whilst biodiversity is an important factor for nature tourists visiting national parks in Finland [8] and protected areas in Uganda [7] this is based on a self-selected sample of nature enthusiasts. Similarly, nature-watching is a popular recreation in the UK [36]. However we found no evidence that high nature value plays a role in recreational site selection for day-to-day use based on representative nationwide sample of the general public. As SSSI designation did not add to the appeal of localities for most recreationists, the public expenditure on these highly valued conservation areas (£85.4 million in England in 2008-09) [37] whilst benefiting conservation does not bring benefits in terms of recreational amenity of the general public. Most public benefits are likely expressed through non-use values [38].

Accepting the importance and necessity of conservation areas, pressure on vulnerable sites may be mitigated by providing recreational opportunities in low nature value sites of preferred habitat types. We found a distinct preference for broadleaved over coniferous woodland, a distinction not made in previous studies of forest recreation in Britain [39,40]. Recreation value of coniferous woodlands can therefore be enhanced by planting or retaining broadleaved species along paths. Although broadleaved woodland had clear appeal to recreationists, some other land covers of conservation importance were not preferentially selected. Lowland heaths support species and habitats of European conservation importance that are sensitive to recreational impacts; consequently there has been much research on visitation patterns within heathland [41,42]. Nevertheless on a national level lowland heath was not favoured by recreationists; it may be therefore that lowland heaths are visited when local to recreationists although more desirable land covers are preferred.

Conclusions

Understanding the mechanisms driving countryside recreationists' choice of visit location supports management of the countryside for both recreation and conservation. The relationships derived from a nationally representative sample of English households are likely to be relevant to other developed, urban based countries. Further studies are required however to gain a better understanding of cultural differences in the importance of nature value for general recreation, as the global picture may highlight differing trends as with nature-based tourism [4]. We found that, in spite of recreational use being frequently presented as an important ecosystem service and used to support investment in conservation, there is no ecosystem service gain from higher nature value in terms of recreational value to the general public. Protected areas benefit the wider public through non-use values and reconciliation of conservation and recreation remains pertinent.

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

S1 Appendix. Additional methodological details

S1 Fig. Point estimates and 95% confidence intervals of random effect parameters

S2 Fig. Predicted influence on visitation probability of freshwater and coast when SSSI designated and non-designated

S3 Fig. Effects on visitation probability of non-NNR-designated or NNR-designated land covers. Standardised coefficients from a GLMM controlling for path length, elevation, distance to nearest major road, weighted population and county. Bars denote standard error. For each land cover, *P* values of *Z*-tests compare pairs of coefficients between non-NNR-designated/NNR-designated ($P<0.001$ '***', $P<0.01$ '**', $P<0.05$ '*')