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Quantifying cultural ecosystem services: Disentangling the effects of management from landscape features

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Abstract

1. Cultural ecosystem services are undeniably important, yet are typically neglected in land management decisions due to a suite of intractable challenges: they are highly complex, localised, and inextricably associated with landscape features. However, to incorporate the ecosystem services framework into land management, decision-makers need the tools to disentangle the effects of land use from other factors. This is a major challenge for ecosystem services research.
2. Forestry is a widespread land use that has considerable potential to deliver a broad range of ecosystem services, although this requires careful management planning. Additionally, modern production forestry is undergoing a period of rapid change in the face of a plethora of challenges, such as climate change and disease. To increase cultural ecosystem services delivery from forests, managers need tools to understand the implications of different management options.
3. In this paper, we directly test how land use affects cultural ecosystem services. We use a new approach that recognises the underlying complexity of cultural ecosystem services but produces easily interpretable results that are locally relevant and directly applicable to land management. By combining participatory geographic information systems (GIS) and a novel site matching technique, we relate cultural values explicitly to land management, while accounting for the influence of landscape features.
4. Applying this new method to a major UK forest site, we conducted a large survey to gather participatory GIS data points. We showed that land management significantly affected cultural ecosystem service values and were able to make a series of practical forest management recommendations. Notably, a greater diversity of tree species would improve cultural value, and open space is important within the forest landscape.
5. This approach is highly flexible and can be applied to any type of landscape. It allows cultural ecosystem services to be fully integrated into land management decisions to formulate the best management strategy to maximise ecosystem service delivery.

KEYWORDS
cultural values, ecosystem services, forestry, forests, land management decisions, participatory GIS, site matching

1 | INTRODUCTION

People derive a range of goods and benefits from ecosystem services, which are produced by processes in the natural environment (Mace, Norris, & Fitter, 2012; Millennium Ecosystem Assessment, 2005a). Ecosystem services are commonly grouped into provisioning (such as food, fibre and timber), regulating (such as climate regulation and water purification) and cultural (such as aesthetics and recreation). They are underpinned by supporting ecosystem services (such as primary production and nutrient cycling) (Millennium Ecosystem Assessment, 2005a). There has been a dramatic increase in ecosystem service research over the past decade, with good progress in incorporating the results into policy and practice (Guerry et al., 2015; West, 2015). However, the majority of this research has focussed on provisioning and regulating services (Martínez-Harms & Balvanera, 2012), while cultural services have been relatively neglected (Baveye, 2017; Boerema, Rebelo, Bodi, Esler, & Meire, 2016). This may be because cultural ecosystem services are widely considered to be inherently difficult to quantify (Daniel et al., 2012; Dickinson & Hobbs, 2017; Willcock, Camp, & Peh, 2017): while many ecosystem services relate to easily measured biophysical processes or changes (Bagstad, Semmens, Ancona, & Sherrouse, 2017; Satz et al., 2013), cultural services include intangible concepts such as aesthetic value (Daniel et al., 2012; Milcu, Hanspach, Abson, & Fischer, 2013). Furthermore, people value cultural services in different ways, and these values can change over time (Gould, Coleman, & Gluck, 2018; Plieninger et al., 2015). Therefore, despite recognition of their importance (Chan, Guerry, et al., 2012; Daniel et al., 2012), cultural ecosystem services are frequently ignored or play a minimal role in valuation exercises (Small, Munday, & Durance, 2017).

In recent years, people are interacting less with nature. This change in behaviour has been attributed to urbanisation, biodiversity loss, technological changes and safety concerns (Gaston et al., 2018; Soga & Gaston, 2016). However, there is a large body of evidence that demonstrates that exposure and relatedness to nature is beneficial for physical and mental health (Dean et al., 2018; Franco, Shanahan, & Fuller, 2017; Wood, Hooper, Foster, & Bull, 2017). Additionally, poor connectedness to nature can reduce pro-environmental behaviour and drive unsustainable attitudes to resource use, and so re-connecting people with nature will have an important role to play in responding to global ecological challenges (Ives et al., 2018; Klaniecki, Leventon, & Abson, 2018). In this context, cultural ecosystem services—which are broadly defined as the non-material benefits from ecosystems (Chan, Satterfield, & Goldstein, 2012)—has clear potential to help address this challenge. By quantifying how people engage with and value the natural environment, we can find ways of encouraging exposure and maximising the positive benefits.

People and the natural environment are intimately linked in the production of ecosystem services and benefits. Ecological processes generate ecosystem services but, often, people manage the environment to influence this process (Mace et al., 2012). Equally, with the addition of other inputs, people convert flows of services into benefits and goods that are of use. Therefore, ecosystem services are 'co-produced' by both nature and people (Fischer & Eastwood, 2016; Palomo, Felipe-Lucia, Bennett, Martín-López, & Pascual, 2016). Furthermore, relational values—which are derived from relationships and interactions, such as between humans and nature—are now widely recognised to be an important additional perspective to more traditional intrinsic and instrumental value framings (Chan et al., 2016; Díaz et al., 2015; Klain, Olmsted, Chan, & Satterfield, 2017). Fish, Church, and Winter (2016) proposed a conceptual framework for cultural ecosystem services, which considers them in terms of cultural practices and environmental spaces. The framework links cultural ecosystem services to their geographical context (environmental spaces enable cultural practices). It also explicitly incorporates the relational values of cultural ecosystem services, following the work of Chan et al. (Chan, Satterfield, et al., 2012; Chan et al., 2011), which recognises that cultural values 'arise from human–ecosystem relationships'; the environment both shapes and is shaped by human actions.

A common goal of ecosystem services research is to understand how we can increase the overall delivery and diversity of ecosystem services produced from different landscapes (environmental spaces). In particular, for the ecosystem services framework to translate into practical land management, decision-makers must have the tools to understand how land use affects the delivery of different ecosystem services in order to decide what to prioritise or how to achieve the best compromise (De Groot, Alkemade, Braat, Hein, & Willemen, 2010; Martinez-Harms et al., 2015; Maseyk, Mackay, Possingham, Dominati, & Buckley, 2017). For cultural ecosystem services, this is complicated by the fact that they are influenced by many factors, such as natural landscape features, heritage and history, current land management practices, and how people interact with the environment (Church et al., 2011). Disentangling the effects of current land management from factors that are relatively fixed (such as the location of natural features) remains a major challenge for ecosystem services research.

The investigation of cultural ecosystem services is particularly well suited to spatial analysis. A range of sociocultural phenomena influence how people value ecosystem services: preferences are the result of how an individual perceives nature's benefits, which in turn is influenced by a variety of internal and external factors (such as core values and social structures) (van Riper et al., 2017). Such preferences can be measured as cultural values that are assigned to particular environmental spaces, that is, places (Garcia-Martin et al., 2017; van Riper & Kyle, 2014; van Riper et al., 2017). In particular, participatory geographic information systems (GIS) is increasingly used as a method to engage stakeholders in the mapping of ecosystem services (Brown & Fagerholm, 2015; Reilly, Adamowski, & John, 2018). Often, the results from such exercises are descriptive, focussing on the spatial distribution of ecosystem services across the landscape, frequently involving the creation of density—or 'hotspot'—maps (Brown & Fagerholm, 2015). Various studies have related cultural values to land use (Brown, 2013; Fagerholm et al., 2016; Garcia-Martin et al., 2017). However, the places that people visit or value are influenced by a wide range of factors in addition to land management, such as ease of access or location of visitor centres (Garcia-Martin et al., 2017). In this paper, we present a novel methodology that aims to relate spatially assigned cultural values directly to management by accounting for these other features in the environmental space.

More than half of the world's forests are production or multipurpose forests (FAO, 2016), and when managed carefully and sustainably they have significant potential for the provision of a wide range of ecosystem services (Quine et al., 2011; Triviño et al., 2017). However, forestry in general is under increasing threat from various factors including disease (Freer-Smith & Webber, 2015; Potter & Urquhart, 2017) and climate change (Ray, Morison, & Broadmeadow, 2010; Seidl et al., 2017). For example, Corsican pine (*Pinus nigra*), Japanese larch (*Larvis kaempferi*) and ash (*Fraxinus excelsior*), are all important species in British forestry that are currently undergoing major declines or being rendered unviable as a result of pathogen outbreaks (Freer-Smith & Webber, 2015). Forest management worldwide urgently needs to be rethought to increase forest resilience (Cavers & Cottrell, 2015; Jacobsen, Jensen, & Thorsen, 2018; Seidl, 2014). At the same time, forest owners are increasingly motivated and influenced by cultural ecosystem services and seek ways to maximise multifunctionality (Hendee & Flint, 2014; Plieninger et al., 2015). Therefore, this is an opportune time to consider how forest management decisions affect the delivery of cultural ecosystem services.

There is a large body of literature exploring the aesthetic and recreational values of forested landscapes. For example, many studies have shown that people generally prefer naturalistic forests and larger trees (Blasco et al., 2009; Gundersen & Frivold, 2008; Irvine & Herrett, 2018; Tyrväinen, Silvennoinen, & Hallikainen, 2017). However, deadwood is often viewed unfavourably, and the size of clear-cuts correlates negatively with recreational value (Edwards et al., 2012a; Gundersen, Clarke, Dramstad, & Fjellstad, 2016; Gundersen & Frivold, 2011). In Finland, seasonality has also been shown to be important, with snow cover increasing the suitability of commercial forest stands for recreation (Tyrväinen et al., 2017). Generally, people seem to prefer broadleaved to conifer forests, and mixtures to monocultures (Almeida, Rösch, & Saha, 2018; Felton et al., 2016; Jensen & Koch, 2004; Quine et al., 2011; Schraml & Volz, 2009; Termansen, McClean, & Jensen, 2013). However, results are variable and seem to be highly context-specific; familiarity appears to be important, as do factors such as openness (Edwards et al., 2012a; Felton et al., 2016; Gundersen & Frivold, 2008; Nielsen, Olsen, & Lundhede, 2007). In a pan-European study of the effects of forest structural attributes on recreational values, Edwards et al. (2012a) found general consensus regarding the importance of many attributes, but also identified key regional differences in preferences. For example, left-over residues from forest management operations or the structural diversity of forest stands had differential importance across Europe, attributed to potentially complex people–place relationships (Edwards et al., 2012a), mirroring concepts proposed for cultural ecosystem services (Chan et al., 2016; Fish, Church, Willis, et al., 2016).

Although it is helpful to identify broad patterns in public perceptions towards forest attributes, particularly for the development of policy, existing research has repeatedly shown the importance of local contexts. Currently, to tailor management decisions regionally, forest managers largely use feedback on forest plans, conversations with visitors, or complaints, as their basis for identifying the types of forestry land management that visitors prefer. Additionally, most studies to date have focused on broad preferences, rather than specific cultural ecosystem services, and there is an overall bias towards recreation and aesthetic cultural values (Almeida et al., 2018; Irvine & Herrett, 2018). Similarly, much research has been based on hypothetical scenarios and uses stated preference methods (such as choice experiments) (Elsasser, Meyerhoff, & Weller, 2016; Irvine & Herrett, 2018). The potential limitations of stated preference methods are well documented, as people's behaviour and actions often differ from their statements (Gosal, Newton, & Gillingham, 2018). If forest managers are to more effectively incorporate planning for ecosystem services into forest design, they need the tools to understand how real forest management alternatives are valued in their local contexts. This includes the potential trade-offs and synergies between the full range of cultural services.

In this paper, we address these research gaps using a novel quantitative methodology that relates cultural ecosystem services directly to the management of the landscape. We first use a large participatory GIS survey to map the distribution of different cultural values, then implement a site matching technique to control for the effects of landscape features. We apply our methodology to Thetford Forest—a large commercial plantation in East Anglia, England—to explore how forest management affects cultural values. We test four hypotheses. First, that cultural ecosystem service values vary with land management (hypothesis one). Then, three hypotheses based on existing literature, forest managers' impressions of visitor preferences for land management, and knowledge of the regional context. The wider East Anglian landscape is predominantly agricultural, yet Thetford Forest is the largest lowland forest in England; it is an important regional feature and provides significant recreational amenity (Natural England, 2015). We therefore hypothesise that visitors prefer forest to open landscapes (hypothesis two). Finally, we test the general, although locally variable, findings that visitors prefer broadleaved species to conifers (hypothesis three), and prefer mixtures to monocultures (hypothesis four).

2 | MATERIALS AND METHODS

We followed the framework proposed by Fish, Church, & Winter (2016) by using environmental spaces as an indicator for cultural ecosystem services, while recognising that these spaces will in turn be shaped by cultural practices. This allows us to explore why certain environmental spaces may be more important than others for cultural ecosystem services (Fish, Church, & Winter 2016).

2.1 | Study site

Thetford Forest is a large commercial plantation (18,719 ha), spanning the Norfolk and Suffolk border in the Breckland region of East Anglia, England (Figure 1). Over 50% of the forest landscape is Corsican pine (*P. nigra*) or Scots pine (*Pinus sylvestris*) monoculture. The main forest block lies between and around the towns of Thetford and Brandon, with a number of smaller blocks around the periphery. It is part of the UK public forest estate, managed by the Forestry Commission. Thetford Forest is designated as a Site of Special Scientific Interest for its plant and invertebrate assemblage and breeding populations of woodlark (*Lullula arborea*) and nightjar (*Caprimulgus europaeus*) (Natural England, 2000). It is also part of the Breckland Special Protection Area and Special Area of Conservation (JNCC, n.d; Natural England, 2006). It additionally provides significant recreational amenity and is largely open access (Natural England, 2015).

2.2 | Survey design

Cultural ecosystem services were defined by four categories: outdoors recreation, wildlife, scenic beauty and tranquillity, and heritage or educational value. These were based on commonly used categorisations of cultural ecosystem services (Boerema et al., 2016; Church et al., 2011; Millennium Ecosystem Assessment, 2005b; Quine et al., 2011). To encourage participation and avoid respondent fatigue, it was important to limit the overall number of categories and corresponding survey length (in addition to creating an engaging survey experience; Lavrakas, 2008; O'Reilly-Shah, 2017). We therefore gave careful consideration to the type of landscape and existing knowledge about the key stakeholders and interests in the forest to inform our final categorisations.

An online survey was designed using participatory GIS. This technique normally involves marking points or shapes on a map (Brown & Fagerholm, 2015); however, many people struggle to communicate the value of landscapes using such constrained spatial techniques (Carver et al., 2009; Huck, Whyatt, & Coulton, 2014). We therefore used the participatory GIS spraycan tool Map-Me (Huck et al., 2014), which has been designed to overcome this. The spraycan generates a series of random points within a specified radius of the computer mouse pointer for as long as the mouse button is held down. The respondents can control the location of the points by moving the mouse pointer and can indicate strength of preference by 'spraying' for longer at a specific location, which generates a higher density of points (Huck et al., 2014). It is easy to control, logical and intuitive for the user, and offers flexibility as respondents can decide freely what patterns to mark (Huck et al., 2014). This is especially useful in contexts such as cultural ecosystem services when questions are asked about values, which can be vague. The mapping components of the survey were prefaced by a series of optional demographic and opinion questions. These questions covered respondents' main use of the forest, familiarity with the area, perceived importance of the four cultural ecosystem services (a five-point Likert scale: very unimportant, quite unimportant, neither important nor unimportant, quite important, very important), and demography (age group, gender). Having considered the questions, the survey respondents moved on to the maps. After familiarising themselves with the spraycan technique using a test and practice map, which included full instructions for how to use the spraycan tool and a description of the key features, respondents were asked to mark separate maps with the areas of the landscape that they valued for each ecosystem service. Respondents were able to easily reset each map (erasing the data points and starting again) if they made a mistake or if they were unhappy with the location of points. They could do this multiple times until they were happy that the points they had sprayed accurately reflected their viewpoint. Each map carried a brief definition of the ecosystem service, as detailed in Table 1. The definitions were carefully considered to ensure that respondents could interpret them widely in line with their personal experiences; for example, heritage or educational value could encompass traditional land management practices such as extensive grazing on heathland.

The study was granted ethical approval by the University of Cambridge Psychology Research Ethics Committee (PRE.2016.058). Respondents were fully aware of the consent procedures, and their participation confirmed consent.

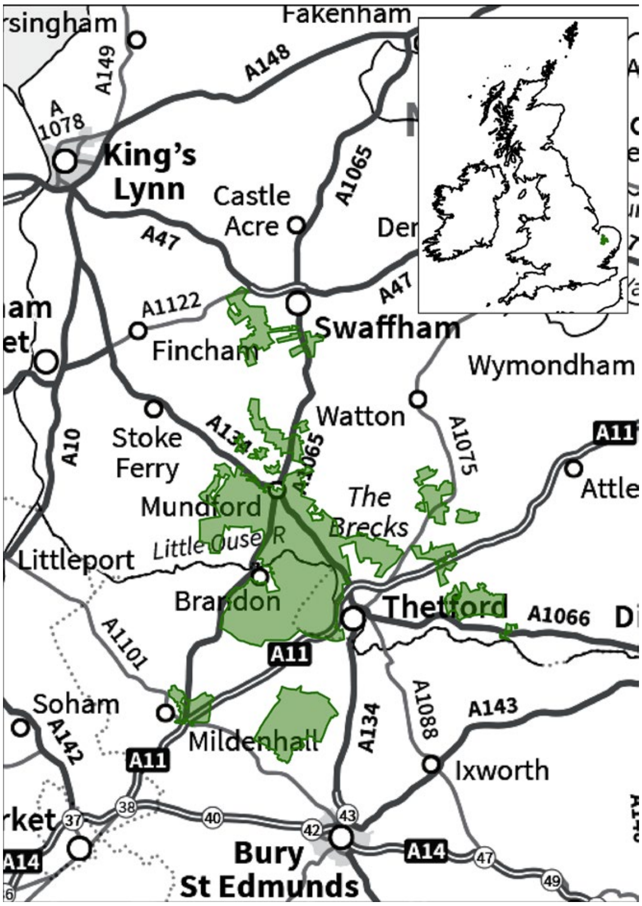

FIGURE 1 Map of Thetford Forest and surrounding region. Inset: national context map. Background map using Ordnance Survey data © Crown copyright and database right (2018)

TABLE 1 Definitions of each of the four ecosystem services

Ecosystem service	Definition
Outdoors recreation	Includes any activity that you undertake in the forest for pleasure or exercise
Wildlife	Includes all aspects of nature, such as plants, animals or natural history in general
Heritage or educational value	Encompasses local history, archaeology, opportunities for learning about the environment, or a sense of place and belonging in the landscape and time
Scenic beauty and tranquillity	Includes landscapes that you think are attractive, or places where you might go for peace and quiet

2.3 | Survey distribution

A detailed outreach plan was formulated in partnership with the Forestry Commission to identify stakeholders and methods of reaching them. We categorised target audiences who use or have an interest in the forest. For example, these included local residents, people with specialist interests or hobbies (such as natural history, walking, mountain biking), forest visitors, and people who work in the forest (for further details see Table S1). We identified relevant organisations or groups for each target audience and contacted them directly via email to ask them to circulate information about the survey to their members or interested individuals and to invite their participation. We also circulated information on social media accounts and distributed posters and flyers around main car parks and noticeboards inviting participation. The survey ran online for 6 months from August 2016 to February 2017.

2.4 | Analysis of survey results:

2.4.1 | Point weightings

In total, 1,037,447 points were sprayed on the four ecosystem service maps by 172 respondents. One of the great advantages of the Map-Me spraycan tool is the ease with which survey respondents can generate a high number of points intuitively, quickly and efficiently. The spraycan generates points continually as the computer mouse is held down, reflecting strength of preference for different areas. Points sprayed outside the forest boundary were excluded, as detailed management information was only available for the forest itself. The forest boundary outline was clearly marked on the map and respondents were made aware that points outside the boundary would be discounted from the analysis. We were able to identify a small number of duplicate answers (where the same person had completed part or all of the demographic questions more than once) and retained only the most recent version. If a respondent had sprayed 30 points or fewer on a map, these were removed from the analysis [a single click of the mouse gives an average of 4.18 ± 1.21 points (mean ± standard deviation, n = 50)] as visual inspection of these points suggested that they were mistakes. After removing these individuals and duplicates, we were left with a total of 168 respondents and 984,149 points. We visually inspected the spray pattern for each respondent and ecosystem service map to ensure that there were no obviously anomalous results (e.g. words or pictures drawn). The number of points sprayed varied between respondents. Across all ecosystem service maps, each respondent sprayed an average of 2,389 points (SD = 6,465). We weighted points to make different respondents comparable such that point weights for each respondent summed to 100 for each map. For example, if a respondent had sprayed 5,000 points, each point was weighted to be 0.02, so the total was 100. We then further weighted the points according to the respondent's preference for each ecosystem service ranked on the Likert scale and given in their answers to general questions in the first part of the survey. Points were multiplied by a number from one to five (one if the respondent thought that the service was very unimportant, five if very important). Where an answer to this question was not given, they were treated as neutral, which had a weighting multiple of three. This secondary weighting prioritised areas where respondents deemed a service as important rather than unimportant.

For illustrative purposes of the weighted point distributions, we generated heat maps using quartic (biweight) kernel density functions.

2.4.2 | Matching

As the layout of the forest was not designed as an experiment, it was necessary to use matching techniques to account for the effects of covariates (Table 2) (Stuart, 2010). For example, an open space area close to a car park might be marked on the maps as highly preferred, but this is likely to be at least partly due to the proximity of the car park itself and ease of access, rather than just the open space management. Site matching is commonly used to account for this situation in ecological studies (Carranza, Balmford, Kapos, & Manica, 2013; Joppa & Pfaff, 2010). It is based on the principle of comparing apples to apples, rather than apples to pears. In our example, to determine whether open space is preferred to a conifer monoculture, we would want to compare the open site to a conifer monoculture site that was equally close to the car park, rather than to one that was a long way from the car park. In

TABLE 2 Definition of terms used in methodology

Term	Definition
Class	Boundaries used for each covariate to divide up the forest; mostly distance groups, some categorical (e.g. soil type)
Compartment	An area within the forest, assigned to a class for each covariate; also defined by their current land management
Covariate	Landscape features that might influence reasons for visiting or valuing an area
Management option	The type of land management for the forest compartment
Subclass	A group of one of more compartments that have the same classes for each of the 21 covariates

doing so, we attempt to account for the effect of the distance from the car park, to determine whether there is a difference between open and conifer monoculture management. Given that there are many such covariates in natural landscapes, site matching works by balancing the distribution of covariates between treatment and control groups as far as possible (Stuart, 2010). However, to our knowledge, it has never been applied either to cultural ecosystem services or when using participatory GIS. Additionally, the matching techniques developed to date divide data into a treatment and a control group (Iacus, King, & Porro, 2011), which is inappropriate for a comparison between multiple land management treatments.

In developing this method to account for covariates, we took inspiration from coarsened exact matching (Iacus et al., 2011). Coarsened exact matching sets boundaries for the maximum imbalance tolerated for each covariate; this is particularly advantageous where there are large numbers of covariates as it removes the possibility of the imbalance on certain covariates being compromised in order to minimise overall imbalance (such as in distance matching; Iacus, King, & Porro, 2012). Through discussion with Forestry Commission staff, we identified 21 features (covariates) that could potentially influence the reasons a respondent valued or visited an area; these included features such as rivers, roads, heritage features and recreation routes. For each of these covariates, we divided the Thetford Forest landscape into different regions according to distance classes from the feature (or categorical classes if appropriate, such as soil type; Table 2). See Table S2 for full details. For example, Figure 2a shows a section of the Thetford Forest landscape divided into regions according to distance from a main river and a heritage feature. Areas of the same colour/shading in Figure 2a are comparable to each other across the forest landscape, because they fall into the same distance class for that covariate.

We then overlaid these regions for each covariate (Figure 2b) and overlaid again with the Forestry Commission sub-compartment database of the forest (Figure 2c), which contains detailed information about the land management option, tree species composition and planting date. Internal areas not owned by the Forestry Commission (and therefore not always freely accessible to visitors) were classified as non-Forestry Commission land. This resulted in a total of 76,158 compartments across the forest landscape (Figure 2d).

Compartments were classified by their current land management option (Table 2). We conducted analysis in two tiers of land management: the first tier gave an overview of the twelve main land management options, such as conifer monoculture, broadleaved mixture, and open space. The second tier added finer detail allowing individual options to be analysed in more detail, such as Douglas fir (*Pseudotsuga menziesii*) monoculture within conifer monoculture. There are various land management types that only cover a small area of the forest, so we set a limit of a minimum of 25 compartments in the forest and at least 7.5 ha of that option across the landscape in order for a land management option to be defined as its own category. Otherwise, the land management option was grouped into other options (e.g, Serbian spruce (*Picea omorika*) monoculture was grouped into 'other conifer' monoculture). This was to reduce the likelihood of the production of significant results due to random variation. See Table S3 for full details of the options used in this case study. We additionally grouped all forested options [conifer monoculture, conifer mixture, broadleaved monoculture, broadleaved mixture, mixture (broadleaved and conifer)] together into one option, which allowed us to test our hypothesis that visitors prefer forest to open landscapes.

All compartments therefore were defined by their current land management and the covariate class they fit into. Different compartments across the forest could be assigned the same combination of classes for all the covariates. For example, in Figure 2d, all the compartments that are the same colour belong to the same classes for each covariate, but their management might differ. Overall, there were 27,878 unique combinations of the 21 covariate classes. Each unique combination (assigned a different colour in Figure 2d) was labelled as a different subclass; the 76,158 total compartments were then grouped by the subclass they belonged to (Table 2).

It is important to note that management options as defined in this paper could also be referred to as landscape features. Given that cultural ecosystem services are co-produced by the interactions between people and nature, we are not directly testing the effects of management practices, but rather the combination of management and the natural environment. In using the terminology 'covariate' and 'management option', we seek to distinguish between landscape features that are relatively fixed or static (such as rivers and roads) and landscape features that can be readily influenced through changing habitat management.

2.4.3 | Simulations

To test our hypotheses that cultural ecosystem service values vary with land management, and that respondents show preferences for different land management options, for each ecosystem service, we generated 10,000 simulations of random spray patterns of 'preferred points' as a null comparison to the empirical data. To account for matching, we generated the random points separately for each subclass of compartments. This meant that areas that received higher numbers of respondents due to, for example, being close to a car park were treated separately from areas with low respondent numbers. Points were generated using the following steps:

1. A subclass, x, was selected (Figure 2e).
2. The number of points from the empirical data across all the compartments in subclass x was defined as y.
3. The probability of a point being allocated to each compartment was in proportion to the area of the compartment (as a function of the total area in subclass x).
4. y points were randomly distributed across all compartments in subclass x, according to each compartment's probability (Figure 2f).
5. The weightings from the empirical data points were attached randomly to these simulated points.

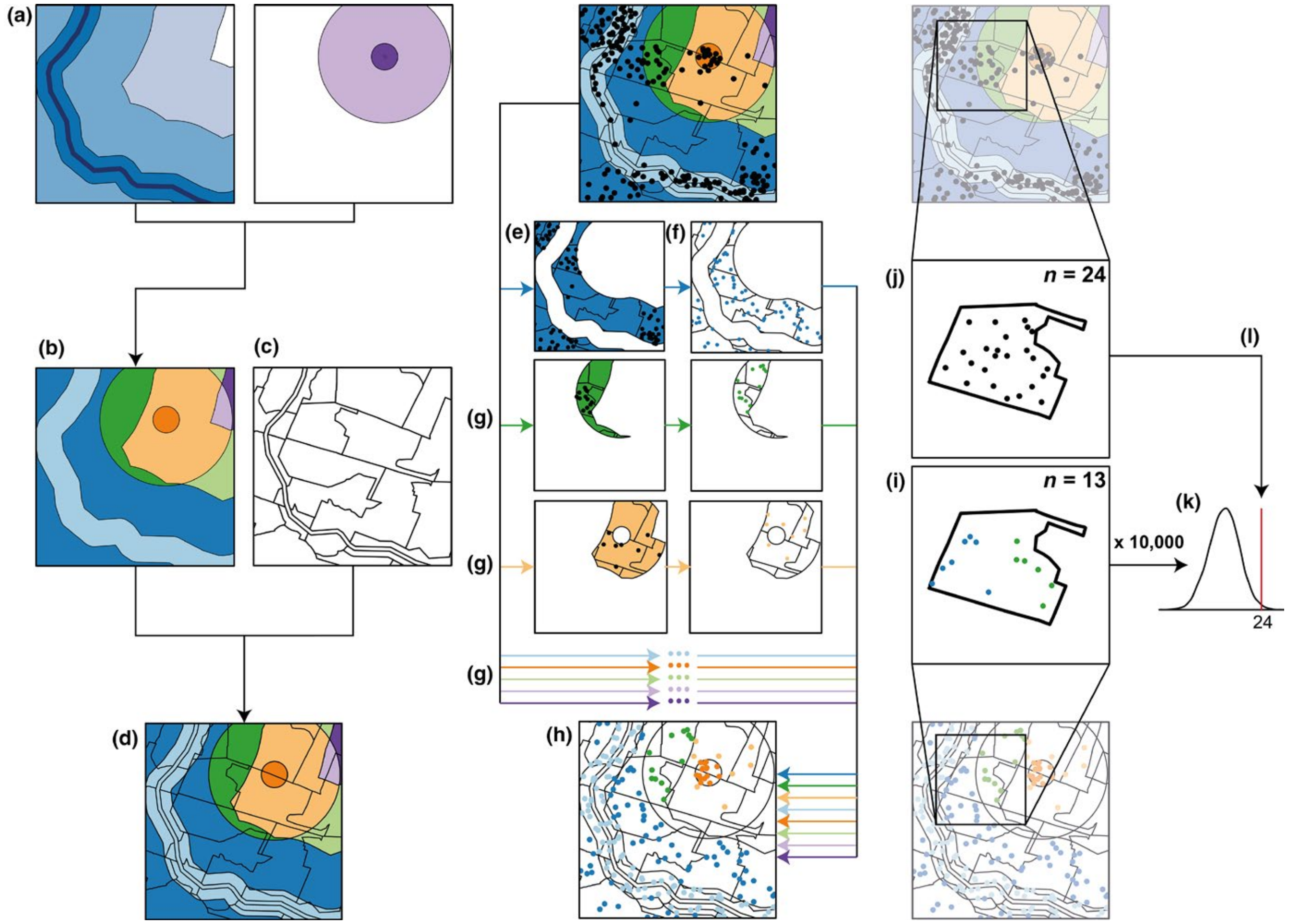

FIGURE 2 Flow diagram of the methodology. (a) The landscape is divided into regions according to classes for each covariate feature (left panel shows buffers for a river, right panel shows buffers for a heritage feature). (b) The regions for each covariate are overlain on top of each other. The different colours represent different subclasses, which are unique combinations of classes for each covariate (e.g. the dark blue regions on either side of the river belong to the same subclass because they are in the same covariate classes). (c) A management database of the landscape specifies the management options for the land cover. (d) The management database is overlain on top of the covariate subclasses, resulting in individual compartments that can be defined by their subclass (the class they belong to for every covariate) and their management option. (e) A subclass is selected from the landscape, and the number of empirical points counted. (f) The same number of points are randomly distributed across the subclass area, and the same point weightings from the empirical points are randomly assigned. (g) Steps (e) and (f) are repeated for every subclass in the landscape. (h) The random points from every subclass are combined to form a random distribution across the entire landscape, which takes into account the different numbers of points in different areas due to the presence of a covariate feature. (i) The random points are aggregated and totalled for each management option (in the figure this is shown for just one management area). (j) The empirical points in the same land management option are also aggregated and totalled. (k) The whole randomisation process is repeated 10,000 times to generate a null frequency distribution of values. (l) The total number of empirical points is compared to the null distribution to see if it is higher or lower than random

6. Steps 4–5 were repeated 10,000 times to generate 10,000 point simulations for subclass *x*.
7. Steps 1–6 were repeated for each of the 27,878 subclasses (Figure 2g).
8. One of the 10,000 simulations from every subclass (selected in the order in which they were generated) was combined to form a null point distribution for the entire forest (when re-attached to one another the compartments from each subclass cover the whole forest; Figure 2h). This resulted in 10,000 separate point simulations for the forest landscape.
9. For each simulation, we aggregated the weighted points by land management option across the forest (Figure 2i).

2.4.4 | Statistics

The empirical data were also aggregated by land management option across the forest (Figure 2j). Each of the sets of 10,000 simulations generated a separate null distribution of values for each management option (Figure 2k), which were compared to the empirical data (Figure 2l). *p* values were calculated as the proportion of random simulated values that were lower than or equal to the empirical value (Ruxton & Neuhäuser, 2013).

As there were multiple comparisons for each of the different sets of simulations, a Benjamini–Hochberg correction (Benjamini & Hochberg, 1995; Pike, 2011) was applied to find an appropriate level of significance for the p value (Table 3). There were 23 comparisons in total (12 broad management options: Figure 4, 1 forested/open: Figure 5 [the open plots in Figure 5 are a repeat of the open plots from Figure 4] and 10 finer management options: Figure 6). We used a conservative false discovery rate of 5%. All tests were two-tailed. Where p_e is the empirical data value and p_α is the Benjamini–Hochberg corrected p value, if $p_{e \le (}p_{\alpha/2)}$ the empirical value was significantly lower than random, whereas if $p_{e \ge (1 - }p_{\alpha/2)}$ the empirical value was significantly higher than random. This allowed us to test whether any land management was valued significantly more or less than expected from random and also to distinguish between different land management options. When the p value was 0 or 1, we reported this in the text as $p < 0.0001$ or $p > 0.9999$ (as there were 10,000 simulations), and in figures as 0 or 1, respectively.

All data analyses were performed using R (R Core Team, 2017), ESRI ArcGIS 10.4 software (ESRI, 2016) and QGIS software (QGIS Development Team, 2018).

3 | RESULTS

In total, 431 people submitted usable responses to the survey, of which 168 completed map components. The number of responses differed between each of the four ecosystem service maps; outdoors recreation had the highest number of respondents whereas heritage or educational value had the fewest (Table 4). Hereafter, we refer to outdoors recreation as 'recreation', scenic beauty and tranquillity as 'scenic', and heritage or educational value as 'heritage'.

Around 60% of respondents perceived recreation, wildlife and scenic to be very important in the landscape, whereas 36% thought that heritage was very important. For each ecosystem service, fewer than 6% perceived them to be quite unimportant or very important.

TABLE 3 Benjamini–Hochberg adjusted p values

Ecosystem service	Adjusted p value (p_α)
Outdoors recreation	0.0074
Wildlife	0.0292
Heritage or educational value	0.0285
Scenic beauty and tranquillity	0.0078

TABLE 4 Number of respondents for each ecosystem service map

Ecosystem service map	Number of respondents
Outdoors recreation	162
Wildlife	101
Scenic beauty and tranquillity	88
Heritage or educational value	61

There was an even split between male and female respondents (48% each, 4% not given) and a spread of age groups. See Table S4 for more details of survey responses.

3.1 | Heat maps

For demonstration purposes, we generated heat maps of the weighted point density distributions (Figure 3). As expected, for all four ecosystem services, there was extremely high point density over the main visitor centre area (point A marked on the wildlife map). For recreation, there was also high point density over the nearby car park and river area further to the north (point B). For heritage, the Grime's Graves heritage site was also a hotspot (point C).

3.2 | Relationship between cultural ecosystem service values and land management (hypothesis one)

We first analysed our results by dividing the landscape into 12 broad land management options (the first tier, Figure 4). For a third (16 out of 48) of land management options and ecosystem service combinations, land uses were valued significantly higher or lower than expected from random, even with a Benjamini–Hochberg correction factor. This confirms our first hypothesis that cultural ecosystem service values vary with land management.

3.3 | Preferences for forests compared to open space (hypothesis two)

When all the different forested areas were combined and compared to open space, the results showed that open space was valued more positively by respondents than forested areas (Figure 5). For recreation, wildlife and heritage, forested areas were significantly negative, indicating that respondents valued these areas less than random (all $p < 0.01$). Open space was positive for recreation, and significant when using an unadjusted p value of 0.05 ($p = 0.99$). Open space was positive, but not significant, for both wildlife and for heritage (94% and 96%, respectively, of random simulations were lower than the empirical value). Neither forested nor open space was significant for scenic. However, as shown subsequently, monocultures and conifers were viewed negatively compared to mixtures and broadleaved species, and as the majority of the forested area is monoculture (65.5%; of which 61.7% is conifer monoculture and 3.8% is broadleaved monoculture), the current composition of Thetford Forest may be negatively skewing perceptions of forest in relation to open space. These results lead us to reject our second hypothesis that forested areas are preferred to open space.

3.4 | Preferences for broadleaved species compared to conifers (hypothesis three)

Comparing broadleaved monocultures with conifer monocultures shows that, while conifers were universally very negative

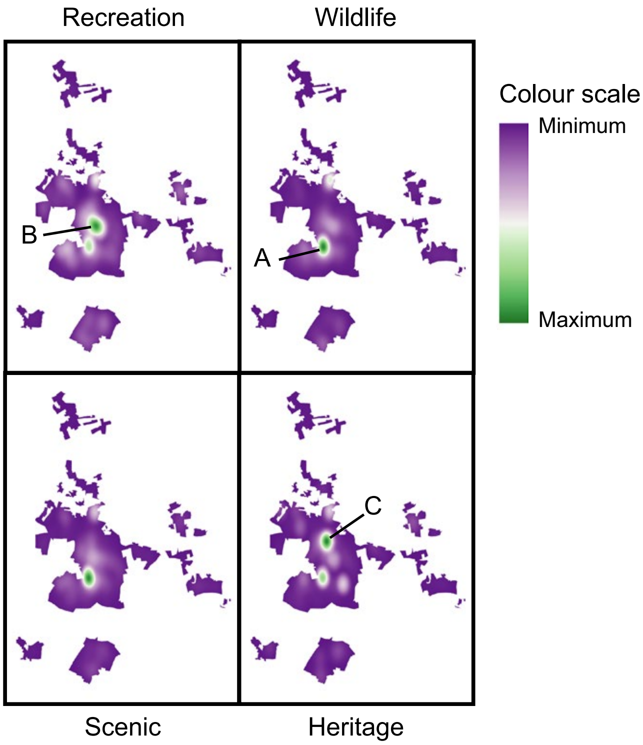

FIGURE 3 Heat maps of the weighted point density distributions for each ecosystem service. The colour scale is adjusted to show the minimum and maximum value for each map. Labelled points: (a) Visitor centre; (b) Main river and car park area; (c) Grime's Graves heritage site. [Correction added after online publication on 5 April 2019: 'Maximum' and 'Minimum' have been reversed on the scale]

(recreation and scenic $p = 0.06$, wildlife and heritage both significant at $p < 0.0001$), broadleaves were only significantly negative for heritage (heritage $p < 0.0001$), and positive for recreation and wildlife ($p = 0.89$ and 0.97 respectively; Figure 4). For tree species mixtures, broadleaves were not significantly negative for any ecosystem service, whereas conifers were significantly negative for recreation ($p < 0.0001$). Whereas conifer mixtures were significantly positive for scenic ($p > 0.9999$), broadleaved mixtures were significantly positive for heritage ($p > 0.99$). Overall, these results support our third hypothesis that broadleaves are preferred to conifers.

We explored these relationships further by dividing the forested land uses into finer management options (the second tier, Figure 6). Breaking down conifer monoculture into different species' components shows trade-offs between species (Figure 6a). Whereas Corsican pine was very negative across all ecosystem services (recreation $p = 0.04$, wildlife and heritage both significant at $p < 0.0001$, scenic $p = 0.08$), all other species (with the exception of other conifer) were significantly positive for at least one ecosystem service. Of particular note, larch (*Larix* × *marschlinsii* and *L. kaempferi*) and Douglas fir were significantly positive for recreation and wildlife (recreation: $p > 0.99$ for both; wildlife: $p > 0.9999$ for both). This is important given the different percentage compositions of these different species monocultures across the forest. Corsican pine accounts for 77.8% of conifer monoculture, whereas larch and Douglas fir monoculture combined comprise just 3.3% of conifer monoculture. As with the previous comparison between forested areas and open space, the dominance of negatively valued Corsican pine seems to have skewed the overall valuation of conifer monocultures. Similarly, the majority of broadleaved monoculture is other broadleaved species (83%). Increasing the proportion of sweet chestnut (*Castanea sativa*) within the forest, which was significantly positive for scenic ($p > 0.9999$), may have increased the overall valuation of broadleaved monoculture. Birch (*Betula spp.*), however, was significantly negative for recreation and heritage (recreation $p < 0.01$, heritage $p = 0.01$).

Deconstructing conifer and broadleaved mixtures into separate options according to whether the largest component (i.e. the species with the greatest percentage of the total species composition) was conifer or broadleaved revealed interesting results (Figure 6c): for heritage and scenic, mixtures dominated by a conifer were valued more positively, whereas the opposite was true for recreation and wildlife.

3.5 | Preferences for mixtures compared to monocultures (hypothesis four)

Within conifers, mixtures were valued more positively than monocultures for wildlife, heritage and scenic (monoculture was significantly negative and mixture positive, though not significant, for wildlife and heritage; mixture was significantly positive and monoculture negative, though not significant, for scenic; Figure 4). Interestingly, conifer mixture was negatively significant for recreation whereas conifer monoculture was not, although arguably conifer monoculture is approaching significance ($p = 0.06$) (Figure 4). For broadleaved species, mixtures were valued significantly positive and monocultures significantly negative for heritage (mixtures $p > 0.99$; monocultures $p < 0.0001$). Generally, mixtures were more positively valued than monocultures, supporting our fourth hypothesis, but the preference is not strong.

4 | DISCUSSION

Our results from the Thetford Forest landscape showed that respondents had strong preferences for certain land management options even though they were not asked to consider this in the survey. A third of all broad land management and ecosystem service combinations were valued significantly positively or negatively, confirming our first hypothesis that cultural ecosystem service values vary with land management. This demonstrates the importance of land management for cultural ecosystem services and underlines the great potential to increase the delivery of cultural values from landscapes through management decisions.

Given that Thetford Forest is recognised as being an important site within the wider region, particularly for recreation, we hypothesised that visitors would prefer the forest to open landscapes. However, we discovered that the current forest composition, with

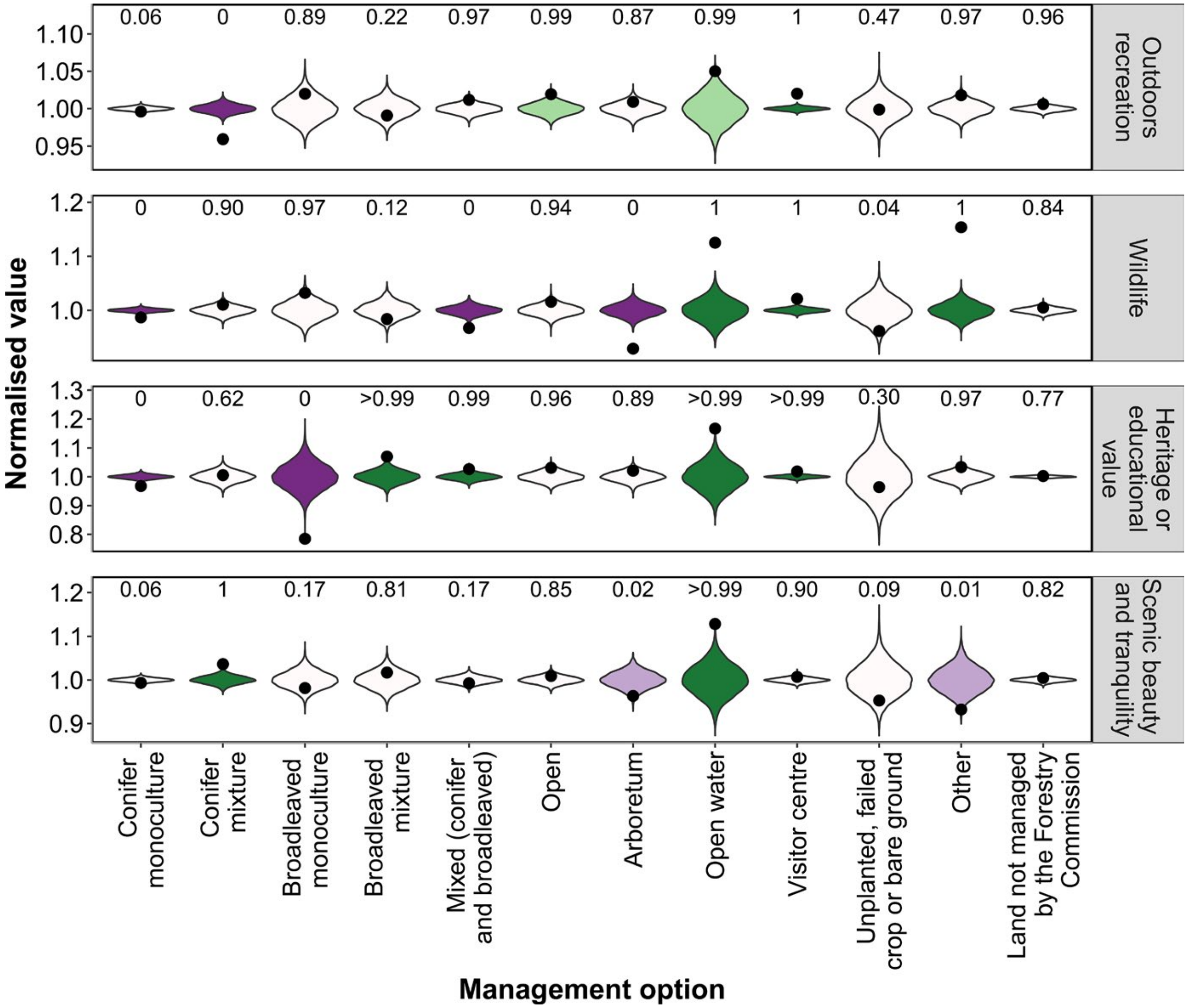

FIGURE 4 Cultural values in relation to broad management options. Values from 10,000 random simulations of point data are shown as violin plots. Empirical data values are marked in relation to the violin plots by a black point. For each management option and ecosystem service combination, all values are normalised by being plotted as a proportion of the mean of the random simulations. Violin plots are scaled to have equal width across management options. Violin plots are coloured dark green if the empirical value significantly exceeds random and dark purple if significantly lower (two-tailed); a Benjamini–Hochberg correction is calculated for each ecosystem service (see Table 3, methods). Plots are coloured light green or purple if the empirical values are significantly different using an unadjusted p value of 0.05 (but not with the Benjamini–Hochberg correction factor). Numbers above the plots indicate p values (the proportion of random simulations that have values lower than the value of the empirical data). Number of respondents for each ecosystem service were as follows, outdoors recreation: n = 162; wildlife: n = 101; heritage or educational value: n = 61; scenic beauty and tranquillity: n = 88

a heavy bias towards monocultures (particularly Corsican pine), made the forest area as a whole valued negatively compared to open spaces (Figure 5). On the other hand, other species monocultures (such as Douglas fir, larch, Scots pine and sweet chestnut) and species mixtures were valued positively in different ways, with all ecosystem services valued significantly positively by at least one of these different management options (Figures 4 and 6). Open space was positively valued across all ecosystem services, so it seems unlikely that forested areas will ever be preferred overall to open space within the Thetford Forest landscape. Nevertheless, these results suggest that changing the forest composition to include a greater proportion of mixtures and different species compositions could greatly increase the overall cultural value of forested areas.

In the UK, substantial areas of open habitat were afforested during the 20th century, and now a reversion of forested land back to priority open habitats is recognised as appropriate in some circumstances for biodiversity conservation (Forestry Commission, 2010). The open space network within Thetford Forest supports rare plant and invertebrate assemblages, with several large heathland areas, and is designated accordingly (Natural England, 2000).

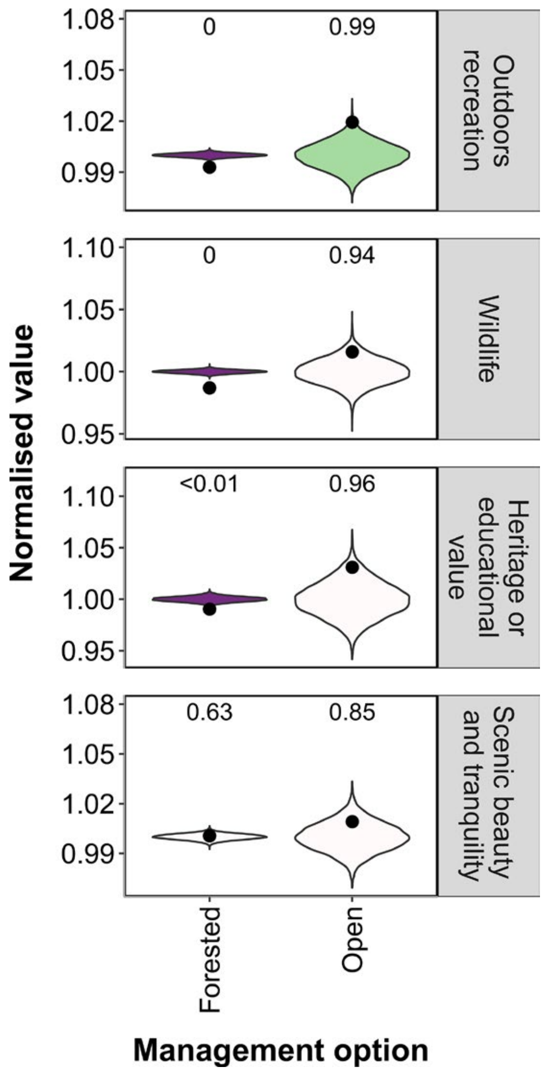

FIGURE 5 Comparison of forested areas to open space. All forested management options (conifer monocultures, conifer mixture, broadleaved monocultures, broadleaved mixture, mixed) are grouped together as 'forested'. Values from 10,000 random simulations of point data are shown as violin plots. Empirical data values are marked in relation to the violin plots by a black point. For each management option and ecosystem service combination, all values are normalised by being plotted as a proportion of the mean of the random simulations. Violin plots are scaled to have equal width across management options. As for Figure 4, violin plots are coloured dark green if the empirical value significantly exceeds random and dark purple if significantly lower (two-tailed); a Benjamini–Hochberg correction is calculated for each ecosystem service (see Table 3, methods). Plots are coloured light green or purple if the empirical values are significantly different using an unadjusted p value of 0.05 (but not with the Benjamini–Hochberg correction factor). Numbers above the plots indicate p values (the proportion of random simulations that have values lower than the value of the empirical data). Number of respondents for each ecosystem service were as follows, outdoors recreation: n = 162; wildlife: n = 101; heritage or educational value: n = 61; scenic beauty and tranquillity: n = 88

In this study, we found that open space was rated very positively for wildlife value, although this was not significant, which is perhaps not surprising given the relatively specialist biodiversity interests. Indeed, open space was universally positive across all four cultural ecosystem services, probably reflecting an appreciation of open vistas in what is a relatively uniform forest landscape with little topographical change. Edwards et al. (2012a) found that variation between forest stands was of high importance to recreational value in Central Europe where forest density is high, but of relatively low importance in Great Britain where forest density is low, therefore suggesting that overall landscape structural diversity is key. Our findings that open space was positively valued within a largely continuous, extensive forest landscape add support to this hypothesis. Forest openings, particularly where they are openings of other natural habitats rather than clearfell areas (as is the case with much open space in Thetford Forest), are generally found to be valued, as are forest landscapes that offer views of surroundings (Gundersen & Frivold, 2008; Gundersen et al., 2016). Tyrväinen et al. (2017), however, found that open views containing few trees were of low value for tourism in Finnish Lapland, although clearly expectations and preferences for habitats will vary regionally. In the Thetford Forest region, open space is clearly a highly valued component of the landscape.

Following general findings in the literature, we also hypothesised that broadleaved species would be preferred to conifers (hypothesis three) and that mixtures would be preferred to monoculture (hypothesis four). We found some evidence to support these hypotheses, although the differences were not strong. However, this corroborates the findings of Edwards et al. (2012b) that broadleaves and mixtures are only marginally preferable to conifers and monocultures, respectively. Furthermore, although these preferences are generally found across Europe, results are mixed (Felton et al., 2016; Gundersen & Frivold, 2008; Termansen et al., 2013), so it is perhaps not particularly surprising that there is not a more dramatic distinction between broadleaves and conifers or between mixtures and monocultures.

Additionally, as this methodology uses revealed preferences, these results suggest that respondents' behaviour does not reflect a strong distinction between broadleaves and conifers or between mixtures and monocultures, even if people generally claim to prefer one or the other. This reinforces the general importance of comparing people's behaviour with their stated preferences; part of the strength of this methodology is that it infers the importance of management options by asking about general values but avoids potential biases by not explicitly asking about management. Thetford Forest was established as a predominantly conifer monoculture plantation, and the initial planting of the majority of the forest is now starting to fall beyond living memory. It seems likely that respondents viewed the forest's identity as innately single-species coniferous stands. Familiarity has also been proposed as an important factor in determining people's preferences for forest attributes (Edwards et al., 2012a; Nielsen, Gundersen, & Jensen, 2018); although respondents did not overall value conifers or monocultures more than broadleaves or mixtures, this may partially explain why there was less of a preference for broadleaves or mixtures than might be anticipated.

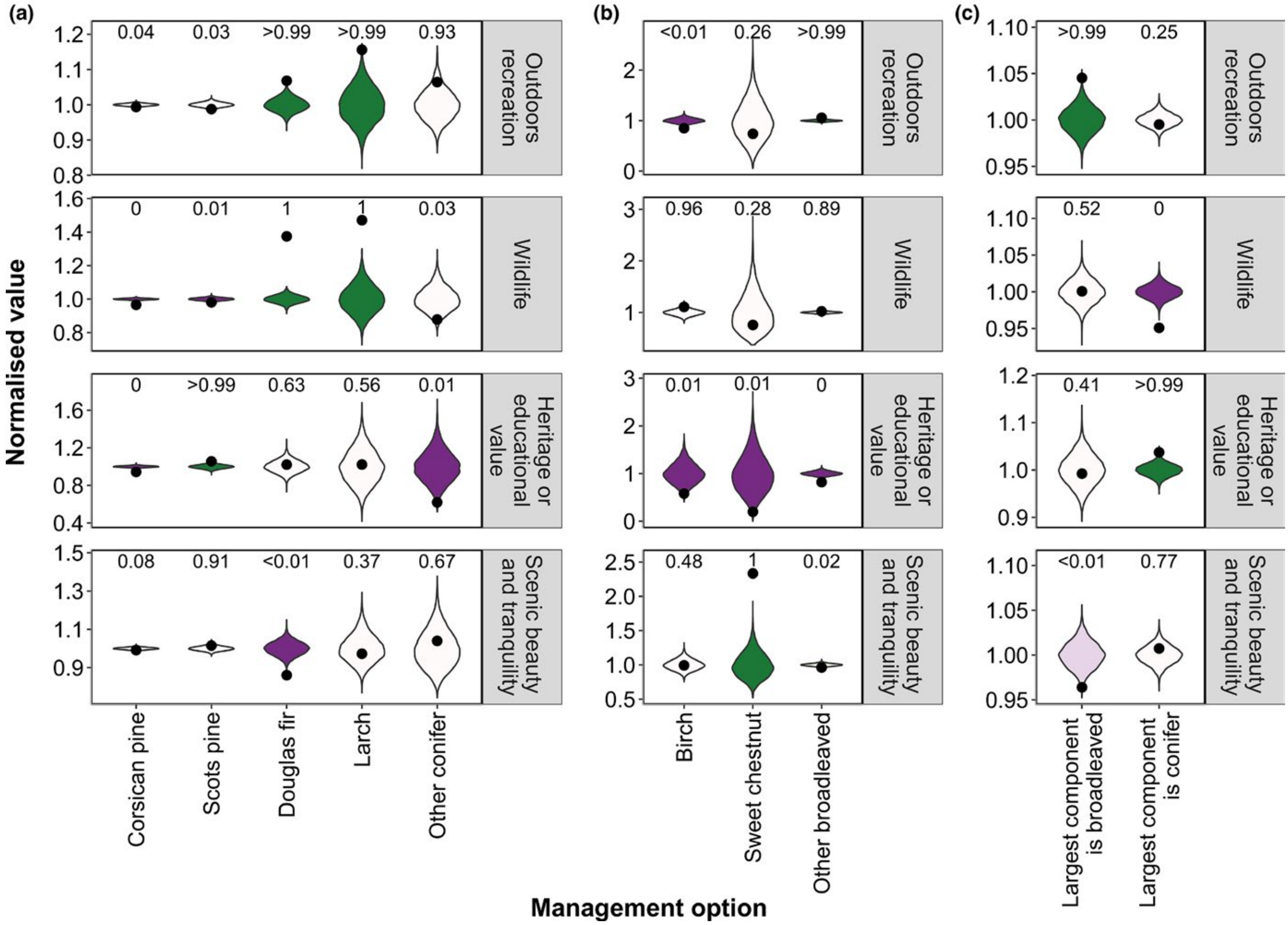

FIGURE 6 Cultural values in relation to finer management options. (a) Conifer monoculture sub-options (b) broadleaved monoculture sub-options (c) conifer and broadleaved mixture sub-options. Values from 10,000 random simulations of point data are shown as violin plots. Empirical data values are marked in relation to the violin plots by a black point. For each management option and ecosystem service combination, all values are normalised by being plotted as a proportion of the mean of the random simulations. Violin plots are scaled to have equal width across management options. As for Figure 4, violin plots are coloured dark green if the empirical value significantly exceeds random and dark purple if significantly lower (two-tailed); a Benjamini–Hochberg correction is calculated for each ecosystem service (see Table 3, methods). Plots are coloured light green or purple if the empirical values are significantly different using an unadjusted p value of 0.05 (but not with the Benjamini–Hochberg correction factor). Numbers above the plots indicate p values (the proportion of random simulations that have values lower than the value of the empirical data). Number of respondents for each ecosystem service were as follows, outdoors recreation: n = 162; wildlife: n = 101; heritage or educational value: n = 61; scenic beauty and tranquillity: n = 88

Our results also showed key differences between different tree species. Corsican pine was valued very negatively for all ecosystem services, which is unsurprising given that the majority of the Corsican pine trees within the forest are contaminated with *Dothistroma septosporum*, a fungal disease that not only renders the crop unproductive for timber, but also leads to defoliation and tree disfigurement (Brown & Webber, 2008). Larch and Douglas fir were significantly positive for recreation and wildlife, and sweet chestnut for scenic beauty. This may be partially explained by their relative rarity (larch and Douglas fir account for 3.3% of conifer monocultures, sweet chestnut accounts for 1.2% of broadleaved monocultures). However, other conifers in monoculture are also very uncommon (0.9% of conifer monocultures), and these were not significantly positive for any ecosystem services (Figure 6; and were significantly negative for heritage, p = 0.01). There appears to be no particular distinction between native and non-native species, with mixed results for both (e.g. birch and Scots pine are native, Douglas fir is non-native). Further work to identify the exact reasons for these differences is required. Nonetheless, these results, in combination with the fact that respondents also generally valued mixtures more than monocultures, support the proposed diversification of tree species (increasing response diversity, as well as species diversity) away from historically dominant species such as Corsican pine to improve ecosystem resilience (Mori, Furukawa, & Sasaki, 2013; Mori, Lertzman, & Gustafsson, 2016).

Forests provide multiple ecosystem services, and tree species richness has been shown to correlate with delivery of multiple ecosystem services (Gamfeldt et al., 2013; Schuler, Bugmann, & Snell, 2017). Our results generally support this, demonstrating that mixtures deliver more cultural ecosystem service value than monocultures. Additionally, as found in other studies (Gamfeldt et al., 2013), no single tree species or broad management option delivers significantly positive results for all cultural ecosystem services. Trade-offs between ecosystem services are recorded three times more than synergies (Howe, Suich, Vira, & Mace, 2014). Our results underline the importance of understanding the trade-offs between different species and management options. Methods such as ours, which make trade-offs explicit, can be used to make practical management decisions that maximise the delivery of ecosystem services (Costanza et al., 2017).

In landscapes such as Thetford Forest, where there is public access and high visitor use, the method has clear value in helping forest managers understand which management options are valued positively or negatively for cultural ecosystem services. The results from the analysis can contribute to future management strategies that seek to balance visitor needs against silvicultural requirements with the aim of maximising and balancing the delivery of all ecosystem services. We can make recommendations for the management of the Thetford forest landscape to increase cultural ecosystem service values. For example, we recommend the diversification of tree species used in commercial conifer planting (particularly a shift away from the dominance of Corsican pine towards species such as Douglas fir, larch and Scots pine). Open space habitats are also of great cultural importance in the Thetford forest landscape, and should be retained as a complementary management option to forestry.

The methodology described here enables the quantification and inclusion of cultural ecosystem services into land management planning by relating cultural ecosystem services directly to land management decisions. This approach has several key strengths. First, by focussing on environmental spaces as an indicator of cultural ecosystem services, the outputs will equip land managers (who do not have the time or expertise to disentangle complex human–environment relationships) with the information required to incorporate cultural ecosystem services values into practical decision-making. Second, it incorporates opinions from across all stakeholder groups in a fair and unbiased manner, ensuring high legitimacy of the results. Legitimacy has been found to be the most important factor in explaining the impact of ecosystem services science on decision-making (Posner, McKenzie, & Ricketts, 2016). Third, it is the first methodology to our knowledge to develop a spatial matching technique for use with participatory GIS data (and is moreover a practical means of generally comparing multiple treatments with site matching). It allows the user to directly relate cultural values to land management while accounting for the confounding effects of landscape features. The point density maps show hotspots over the visitor centre and main recreation and heritage areas (Figure 3), as would be expected from knowledge of how people use the landscape. However, in our analysis we have been able to distinguish in more detail whether and how the underlying land management affects visitor preferences. This is an important step towards identifying the ecological characteristics of environmental spaces that affect cultural ecosystem services (Fish, Church, Willis, et al., 2016). Finally, the survey is a form of revealed preference evaluation, as respondents were not told or asked about the management of the landscape. This is powerful because it incorporates preferences that respondents may not even be aware of themselves, and avoids biases.

There are a number of limitations to the methodology, which are important to bear in mind. First, we did not supervise individual respondents, so we could not guarantee the quality of all responses. Although we visually checked each response to ensure that there were no obviously anomalous patterns, we could not be certain that all patterns were intentional and accurate. Equally, we did not have a measure of the time that each respondent spent recording. Clearly, facilitation of surveys could decrease this uncertainty, although by using a non-supervised approach we were able to reach a wide variety of stakeholders and achieved a high number of responses from individuals who were free to complete the survey at their own convenience. Second, it is important to note that respondents were asked to mark areas that they positively valued, and were not asked to distinguish areas they did not like. Therefore, significantly negative results are inferred from an absence of points (significantly fewer points than would be expected from random). We believe that an absence of points is sufficient to show areas that are not valued, but future research could address this more formally using the same methodology. Thirdly, we recognise that cultural ecosystem services are co-produced by both nature and people, and therefore management interventions may have different effects on cultural values depending on the local context. Our matching technique aims to tease apart these interactions, but there may be other factors that influence cultural values besides the covariates that we have accounted for. Equally, the value of different management options will vary across different landscapes; given that our study site formed one forest landscape, we deemed it appropriate to consider management options equally across the landscape, but this should be considered in wider contexts. We set the distance class bands for covariates according to sensible thresholds for their likely influence, but this could also affect results. Finally, the localised spatial structure created by the spraycan was not fully replicated in our randomisations. However, given that the point density is so high and our compartments relatively small, and also through our matching technique, there was a high degree of spatial structure in our randomised data. Nevertheless, this could be a consideration for refinement of the technique in future analyses. Despite these limitations and considerations, we feel that the matching technique and overall methodology present a valuable approach to better understand how landscape management affects cultural values.

The flexibility of the methodology allows it to be applied to many scenarios across all types of landscape and management, as the landscape features and management options can be specified freely. Furthermore, as we have demonstrated with our case study, the analysis can be run at different levels of detail to reveal broad trends or to make detailed comparisons. Additionally, the ability to compare multiple treatments (rather than just a treatment and control) has relevance for other types of spatial analysis. For example, the method could be used to compare the effects of multiple conservation interventions on species abundance, while accounting for covariates. Overall, although developed to analyse cultural ecosystem service values, the site matching technique is an improvement to participatory GIS data analysis with broad multidisciplinary potential.

5 | CONCLUSIONS

Human well-being is inextricably linked to natural capital and the provision of ecosystem services (Millennium Ecosystem Assessment, 2005a). The incorporation of these concepts into decision-making processes is essential if we are to achieve future sustainability targets (Guerry et al., 2015). However, we must ensure that we consider all ecosystem services in order to achieve the greatest overall benefits, rather than focussing on a subset of well-understood services (Costanza et al., 2017). To date, cultural ecosystem services have been neglected in valuation frameworks due to their perceived intangibility. The methodology developed here shows that it is possible to ascertain, in a statistically rigorous manner, whether land management (rather than landscape features) affects cultural ecosystem service values, and it provides detailed information about trade-offs between different management options. In our case study, we have been able to use these results to make a series of forest management recommendations to increase cultural ecosystem services values. Additionally, understanding how people value landscapes at this detailed level presents an opportunity to increase engagement and connectedness to nature through changing land management at the site level. The methodology can be applied in any landscape to take local influences and viewpoints into account, and as such, it represents a significant step forward in the quantification of elusive cultural ecosystem service values.

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AUTHOR'S CONTRIBUTIONS

E.R.T. and W.J.S. conceived the study. E.R.T designed the research and analysed the results. B.I.S. assisted with the methodology. All authors contributed to the interpretation and writing of the paper.

DATA ACCESSIBILITY

The point data that support the findings of this study will be available from the University of Cambridge repository service Apollo from April 2019 at the following https://doi.org/10.17863/CAM.35449 (Tew, 2019).

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SUPPORTING INFORMATION

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