A big house in the country: Assessing the biodiversity and ecosystem service values of trees and their management trade-offs in the Harewood Estate parkland

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The candidate confirms that the work submitted is their own and that appropriate credit has been given where reference has been made to the work of others.

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Abstract

Wood-pasture and parkland estates are managed landscapes which represent an important part of UK conservation and cultural heritage. The landscapes have historically been multifunctional, yet their values in meeting 21st century societal needs, such as tackling ecosystem degradation and biodiversity loss, have been overlooked. This thesis aims to value the Harewood Estate parkland to determine its biodiversity and ecosystem service (ES) benefits, identify the characteristics that influence these values, and identify trade-offs between these values to support management decisions. Values were generated using the Tree-related Microhabitats (TreMs) methodology, i-Tree Eco software and CAVAT tool and statistically modelled to determine the main tree and landscape drivers of these values. This thesis found the historic parkland estate is a complex landscape which is valuable due to its dominance of veteran and ancient trees. Despite relatively low levels of diversity within the tree communities themselves, TreM diversity was higher in parkland trees than trees found in broadleaf European forests. Parkland trees currently provide 1,224,030 kg of carbon storage, 14,522 kg/year of gross carbon sequestration, 693 m^3/year avoided stormwater runoff and 412 kg/year air pollution removal regulating ES values and £28,117,919 in cultural amenity ES value. The consistent dominant driver of these values was tree size and age, agreeing with the previous literature that it is the largest and oldest trees that contribute disproportionately to biodiversity and ES. Biodiversity, regulating ES and cultural ES were overwhelmingly synergistic when trade-off analysis was carried out, suggesting the Harewood Estate parkland is a successfully multifunctional landscape. A parkland tree management plan should be created, with future management initially focused on maintaining and retaining veteran and ancient trees, as these trees provide greater benefits the larger and older they become. Wood-pasture and parkland estates have an important part to play in meeting 21st century societal challenges and under the correct sustainable management regime they can help achieve this.

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1 Introduction

There is increasing global recognition that natural ecosystems provide services to life on Earth, and that this makes them valuable (Costanza et al., 2014; Sheng et al., 2019; Dasgupta, 2021). However, natural ecosystems and biodiversity are in decline (IPBES, 2019). In the UK, 71% of ecosystems (quantified as natural capital assets by the Natural Capital Committee) are deteriorating (Natural Capital Committee, 2020), with 41% of species declining in abundance, 27% of species decreasing in distribution and 15% threatened with extinction (Hayhow et al., 2019). These declines have left the UK as one of the most nature-depleted countries on Earth, ranking 189 out of 218 countries for biodiversity intactness (Steffen et al., 2015; Hayhow et al., 2016). In recent years, ecosystem collapse has been listed in the top five global risks to human's way of life (IPBES, 2019; World Economic Forum, 2020). Human activity, including land use change, pollution and climate change, is responsible for these declines (Hayhow et al., 2019; Díaz et al., 2019) and is now the dominant driver of change on Earth (Vitousek et al., 1997; Waters et al., 2016).

1.1 Ecosystem services, biodiversity and their importance within the UK landscape

Declines in ecosystems and biodiversity are reflected in the decline of ecosystem services. Ecosystem services (ES) are the contributions that natural ecosystems make to human well-being (Millennium Ecosystem Assessment, 2005; Haines-Young and Potschin, 2018). Approximately 30% of UK ecosystem services are in decline, with many others reduced or degraded (UK National Ecosystem Assessment, 2011; UK National Ecosystem Assessment, 2014). ES have historically been split in to four categories: provisioning, regulating, cultural, and supporting services (Millennium Ecosystem Assessment, 2005; TEEB, 2010). However, these categories overlap and are interdependent on each other. This, coupled with supporting services having an underpinning role on the three other ES categories, has led to debate and refinement of ES categorisation. The most recent Common International Classification of Ecosystem Services (CICES) now classifies ES in to three categories: provisioning; regulating and maintenance; and cultural services (Haines-Young and Potschin, 2012; Haines-Young and Potschin, 2018).

The importance of biodiversity in underpinning the delivery of ES and human well-being is well recognised (Millennium Ecosystem Assessment, 2005; Díaz et al., 2006; Cardinale et al., 2012), as well as its role as a final ES (e.g. pollination) (Klein et al., 2006; Mace et al., 2012). ES relationships with ecosystems, biodiversity, ecosystem function and human well-being can be illustrated in an ecosystem service cascade (Figure 1). Changes in these biodiversity-ES relationships will have an impact on the flow of ES in this cascade, and therefore impact the benefits and values derived by humans.

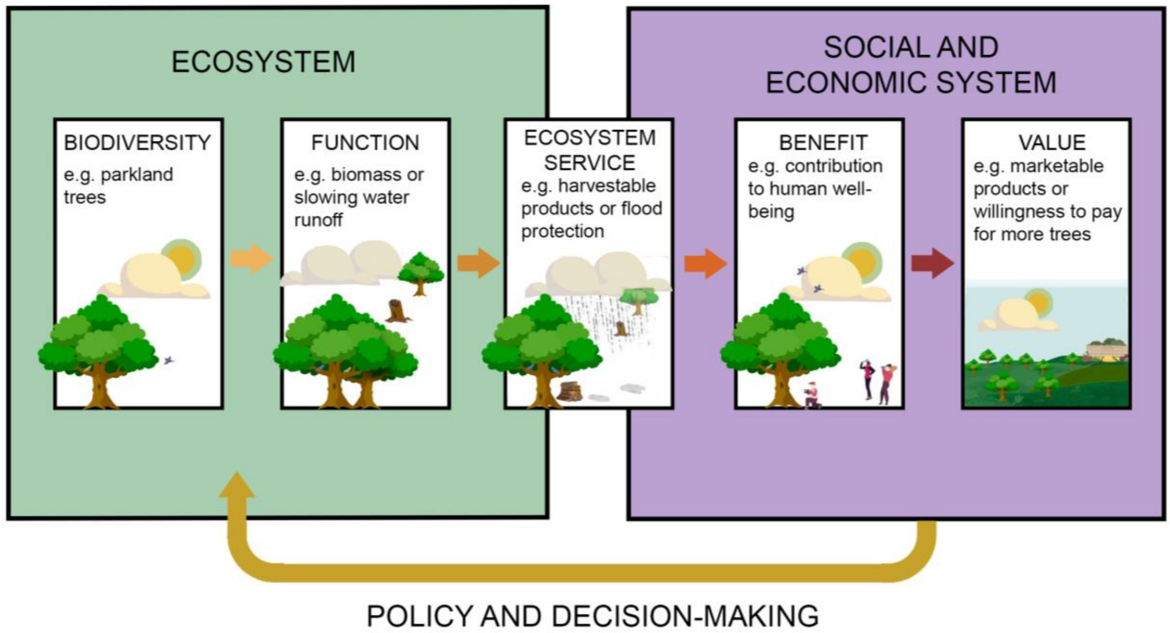

Figure 1. The relationship between biodiversity, ecosystem function and human well-being in the ecosystem service cascade. Policy and decision-making feed back into the cascade and ultimately have control over increases or decreases in human well-being. Adapted from Haines-Young and Potschin (2010) and Martín-López et al. (2014).

The link between declines in biodiversity and declines in ES is increasingly understood (Balvanera et al., 2006; Cardinale et al., 2006), and consequently the link between policy and management decisions on biodiversity and land use and the supplying of these services (Figure 1; Ruckelshaus et al., 2015; Geijzendorffer et al., 2017). It is important then to manage ecosystems and biodiversity through policy and decision-making, to maintain healthy ecosystems and restore previous degradation, and to achieve greater benefits and value for humans. A key requirement to do this is the valuation of localised ES by governments, public and private organisations to understand the range and balance of benefits for people, the environment and the economy (Guerry et al., 2015; Lear et al., 2020; Dasgupta, 2021). The problem remains that such valuation of localised ES is still developing, especially in its coverage of different land use types (Pandeya et al., 2016).

For example, the publication of the 25 Year Environmental Plan by the UK government highlights the need to tackle ecosystem degradation and biodiversity loss, including the need to update the UK National Ecosystem Assessment (2011), to quantify the current ecosystem service values provided by the UK environment, and embed biodiversity net gain within development and land management (Defra, 2018). Without robust data driving policy and land management decisions, decisions will continue to be made that negatively affect biodiversity and ecosystem services and therefore human well-being.

1.2 Wood-pasture and parkland in the UK

An important yet overlooked land use type relevant to biodiversity and ES is wood-pasture and historic parkland. This is one of the oldest land use types in Europe (Vera, 2000; Luick, 2008), and represents an important part of UK conservation and cultural heritage (Read and Bengtsson, 2019).

Wood-pasture and parkland are social–ecological ecosystems that are comprised broadly of human-managed habitat mosaics of open-grown trees with a grazed understorey (Bergmeier et al., 2010; JNCC, 2011; Reid et al., 2021). Rather than being a particular plant community, they represent a vegetation structure. They can vary based on the historic land use and current management (Gillet, 2008; Chételat et al., 2013), with different types including old wooded commons, medieval deer parks, Royal Forests and landscape parks (Rackham, 1986; Hartel et al., 2013; Read and Bengtsson, 2019).

In the UK, wood-pasture and parkland is valuable on a national level, being recognised as a priority habitat of conservation significance in the UK (JNCC, 2019). Wood-pasture and parkland are particularly valued for their trees, especially veteran and ancient trees, with the unique species assemblages they support (Manning et al., 2006; Plieninger, Hartel, et al., 2015) and cultural heritage value (Woodland Trust, 2008a). The high UK concentration of ancient and veteran trees relative to the rest of Europe (Kirby et al., 1995; Butler et al., 2002; Nolan et al., 2020) is also a valued habitat at a European level (Plieninger, Hartel, et al., 2015). It is the importance of its landscape continuity which has created important and unique components of particularly high biodiversity and social–cultural value.

One of the main threats to the habitat is land use change (Plieninger, 2012; Read and Bengtsson, 2019). For example, conversion to more intensive agriculture meaning high-intensity grazing and grassland management and no new tree recruitment into the population (Bergmeier et al., 2010). Therefore quantifying the importance of biodiversity and ES in wood-pasture and parkland is important to put a value on a threatened landscape that is affected by management decisions.

1.3 Veteran trees in wood-pasture and parkland

Of particular significance to the value of wood-pasture and parkland is open-grown veteran and ancient trees: 26% of veteran and ancient trees recorded in the UK are located in wood-pasture and parkland (Reid et al., 2021; Woodland Trust, 2021).

Trees broadly develop in three phases (White, 1998; Read, 2000; Dujesiefken et al., 2016). These are the young phase, from seedling establishment until maturity; the mature phase, when the maximum crown size is reached; and the ancient senescent phase, when crown retrenchment begins whilst trunk girth still increases. Characteristics associated with veteran and ancient trees arise during this final developmental phase, including trunk hollowing, cavities and holes, bark loss, canopy deadwood and the presence of saproxylic organisms (Read, 2000; Rust and Roloff, 2002).

Veteran and ancient trees provide numerous benefits to humans. Like all trees, they contribute to ecosystem services such as carbon storage (Lutz et al., 2018). They are ‘keystone structures’ for biological communities (Lindenmayer et al., 2014), and have been linked with high levels of biodiversity, supporting specialised species including saproxylic invertebrates (Siitonen and Ranius, 2015; Seibold et al., 2018), saprotrophic and mycorrhizal fungi (Boddy, 2001), epiphytes (e.g. lichens and ferns) (Ranius et al., 2008), bats (Kalcounis-Riippell et al., 2005; Tillon et al., 2016) and birds (Loyn and Kennedy, 2009). Veteran and ancient trees are an integral part of many traditional landscapes and sacred sites (Lindenmayer and Laurance, 2017; Nolan et al., 2020), especially wood-pasture and parkland due to their management continuity and open-grown planting regimes. They can also be used as evidence for changes over time. For example, changes in land use practises and tree management such as pollarding and coppicing (Petit and Watkins, 2003), as well as past changes in temperature, water availability and disease outbreaks (Briffa, 2000; Ballesteros et al., 2010; Bengtsson et al., 2021). Veteran and ancient trees are also important safeguards for genetic resources for the future (Read, 2000; Lonsdale, 2013; Paprštein et al., 2015), with the trees potentially harbouring genes for stress tolerance or pest resistance (Major, 1967) which will become important for managing anthropogenic threats such as a changing climate and introduced tree pests and diseases. However veteran and ancient trees are being lost from landscapes, including from wood-pasture and parkland, causing conservation concern, with negative consequences on biodiversity and ecosystem services (Lindenmayer et al., 2012; Miklín and Čížek, 2014; McDowell et al., 2020). Appropriate management of these veteran and ancient trees is therefore important when discussing the management of wood-pasture and parkland (Read, 2000; Lonsdale, 2013; Lindenmayer, 2017).

1.4 Tree valuation

It is clear that taking stock of the biodiversity and ES trees provide is critically important for the management of wood-pasture and parkland and the values these landscapes deliver. Valuation can be monetary or non-monetary (Gómez-Baggethun et al., 2010; Laurila-Pant et al., 2015). Both monetary and non-monetary valuation have been demonstrated to be important to land managers and policy-makers when valuing biodiversity and ES (Ruckelshaus et al., 2015; Handmaker et al., 2021), due to both the importance of monetary valuation in cost-benefit management decisions (de Groot et al., 2012; Dasgupta, 2021), and 'priceless' intrinsic values ascribed to places or species (Daniel et al., 2012; Scholte et al., 2015). Therefore it is important to consider whether to value biodiversity and ES in monetary and/or non-monetary terms depending on the aim of the valuation.

Assessing the biodiversity value of individual trees can either be done by quantifying the microhabitats on trees or quantifying the species associated with these microhabitats (Laurila-Pant et al., 2015; Paillet et al., 2018). Microhabitats could be indicators for the biodiversity that uses them (Lassauce et al., 2011; Gao et al., 2015; Asbeck et al., 2021). Microhabitats are akin to veteran tree features, therefore assessing tree microhabitats also give an insight in to other regulating and cultural ecosystem service values the trees may provide through being older. Continuing on from the initial Veteran Trees Initiative Specialist Survey Methodology, the Tree-related Microhabitats (TreMs) methodology and typology was formalised in 2018 to standardise tree biodiversity monitoring for forest sites (Kraus et al., 2016; Larrieu et al., 2018). This thesis will be the first study to use the TreMs methodology for veteran trees in a parkland habitat.

Beyond biodiversity value, trees are known to provide many ES benefits including timber (UK National Ecosystem Assessment, 2011), carbon sequestration (Nowak and Greenfield, 2018) and human well-being (Salmond et al., 2016). However, ES research on trees has mainly focused on forest (Gamfeldt et al., 2013), agricultural (Barrios et al., 2018) and urban ecosystems (Hall et al., 2018; Turner-Skoff and Cavender, 2019). A habitat under-researched in tree ES literature is wood-pasture and parkland. Veteran trees are also underrepresented in the current ES valuation literature, with studies focused on younger tree populations rather than larger, older trees (Hand et al., 2019). i-Tree Eco is a programme which has been successfully used to quantify forest structure, regulating ecosystem services and the economic value of tree communities (Nowak, 2020).

Capital Asset Value for Amenity Trees (CAVAT) is a tool developed in the UK for public authorities to estimate the replacement value of trees based on their aesthetic qualities and value to the community (Neilan, 2017; Doick et al., 2018). The amenity value of trees is a cultural ES, and despite CAVAT's arguably subjective judgements (Price, 2020), it is still a useful, reproducible tool for indicating how valuable individual trees are based on their aesthetic qualities. Aesthetic cultural value is important in historic wood-pasture and parkland, with landscape design being an crucial feature of their aesthetic appeal and cultural heritage value (Tengberg et al., 2012; Walerzak et al., 2015). The incorporation of cultural value in ES valuation is also important for inclusive valuation, taking in to account social sciences and local practitioners perspectives, to arrive at holistic and sustainable management recommendations (Plieninger, Bieling, et al., 2015; Díaz et al., 2018).

1.5 Managing wood-pasture and parkland in stately homes

Wood-pasture and parkland in the UK is often associated with historic estates, with many estates of stately homes falling within this habitat type. Wood-pasture and parkland covers approximately 278,100 hectares (2.1%) of the total England landcover (data is not available for Wales, Scotland or Northern Ireland) (Natural England, 2020). Approximately 129,539 hectares (46.6%) of this habitat is made up of the estates of stately homes (Roberts, 1995; Historic England, 2018). The estates of stately homes are designed parklands that provide a habitat between urban and intensively managed agricultural landscapes, with many located closer to urban limits due to urban sprawl since the stately homes were originally constructed, making them unique in their ecological placement (Peacock et al., 2018; Šantrůčková et al., 2019).

Previous research on stately homes has mainly focused on their biodiversity values (Liira et al., 2012; Jonsell, 2012; Lõhmus and Liira, 2013; Walerzak et al., 2015; Šantrůčková et al., 2019), yet historic parkland estates have known aesthetic and cultural heritage values. Many in the UK are designated as Registered Parks and Gardens for example (Roberts, 1995), yet their cultural ES values in the UK are still yet to be fully studied (Askwith, 1999; Lawton et al., 2020). They also have other values. Most recently, regulating ES have been the focus of ES valuation study, where the economic value of carbon storage and sequestration, runoff prevention and pollution removal have been studied as additional values provided by trees in the grounds of stately homes (Peacock et al., 2018). However these former studies lack a holistic approach to the valuation of historic parkland which is a key in the management of these multifunctional landscapes, where no value is necessarily more important than the others. Biodiversity, regulating and cultural ES are then important benefits that are provided by parkland estates. Yet in order to make management decisions that provide all these benefits understanding the effects of management decisions, which can change the type, magnitude, and relative mix of ES provided - i.e. trade-off analysis - is needed between them (Carpenter et al., 2009).

In conclusion, stately homes, their parkland and veteran trees are a key land use of local, national and European significance offering a range of services, benefits and values to humans. Yet existing studies have lacked integrated valuation and have overlooked the ES values of parkland estates; there has been only one previous study on the regulating ES of parkland estates, with the cultural ES value understudied in the UK. As an integral part of parkland estates and their value, veteran trees are particularly underrepresented in biodiversity and ES research. There are difficult decisions to be made in terms of the management priority of parkland estate going forward. Despite the importance of biodiversity, regulating and cultural values in the identity of parkland estates, they are faced with other land use challenges such as conversion to intensive agriculture. To inform their management, they need to be able to assess the different values and their combinations offered including potential trade-offs and synergies between different values. Indeed, it is evident that some values are potentially in conflict in decision-making management scenarios, such as cultural heritage values with economic carbon storage values.

1.6 Research aims

It is the overarching aim of this thesis to support understanding in historic parkland estate management scenarios through applications of parkland value assessments based on tree valuation. This will include one of the first ever investigations of some of the potential trade-off relationships between the important values parkland estates provide. The objectives of this thesis are to:

- Advance the understanding of the valuation of parkland landscapes through the valuation of trees,
- Achieve this through the application of different tree valuation approaches, namely biodiversity and ES valuation,
- Use this information to make recommendations for the future management of the parkland estate.

In order to do this, this thesis addresses the following research questions:

1. What are the biodiversity values of parkland estate trees?
2. What are the regulating and cultural ecosystem service values of the parkland estate trees?
3. How can trade-offs or synergies between these values support the management of parkland estates?

1.7 Thesis structure

Following this introductory Chapter 1 setting out the context, aims and objectives, the thesis continues with Chapter 2 which sets out the research design and methodology chosen to undertake the empirical fieldwork. Chapters 3, 4 and 5 provide three separate results chapters, including detailed processes of data analysis. Chapter 3 values the parkland tree community and tree microhabitat diversity, and identifies the driving factors at the tree and landscape scale of the tree microhabitat diversity value. Chapter 4 values the regulating and cultural ES of the parkland trees and identifies the driving factors at the tree and landscape scale of these values. It then goes on to quantify how the parkland and its ES values have changed in 20 years. Chapter 5 identifies whether there are trade-offs or synergies between these tree-scale biodiversity and ES values and why this may be the case. Each of the findings chapters provides an initial discussion. Chapter 6 brings these together to return to the aims, objectives and research questions and to demonstrate how these findings can be used to inform future parkland management.

2 Methodology

2.1 Study site

The study field site is located on the Harewood Estate, West Yorkshire, United Kingdom, approximately 10 km north of Leeds city centre. Harewood House is a Grade I listed building, now over 260 years old, and one of England's foremost stately homes (Treasure Houses of England, 2021). The house is set in a prominent 400 hectare 'Capability' Brown landscape comprising open parkland, ancient and managed woodland, and a large lake. The landscape is designated as a Grade I Registered Park and Garden (Historic England, 1984).

This study focuses on three areas of parkland: Deer Park, North Front and South Park (centroid grid references SE 31161 45378, SE 31165 44815 and SE 31778 44433 respectively) (Figure 2). These were chosen to represent differing parkland management regimes. The Deer Park is a restored medieval deer park, grazed by a managed herd of red deer *Cervus elaphus* and fallow deer *Dama dama.* The North Front is a frequently mown parkland used by visitors and for outdoor events. The South Park is managed by a mixture of mowing, hay cropping and grazing by sheep.

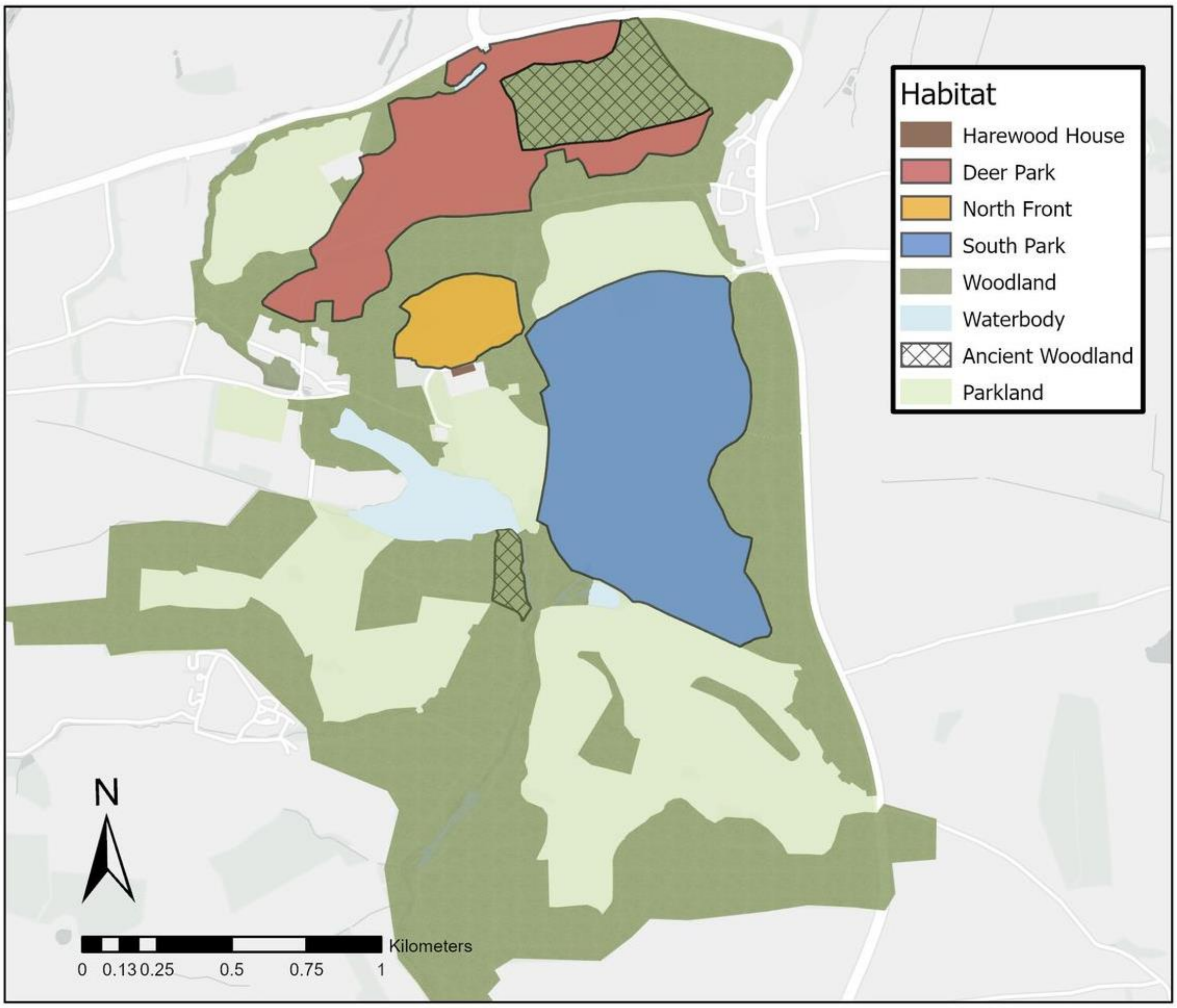

Figure 2. Harewood Estate map of the three parkland study sites: Deer Park (red), North Front (yellow), South Park (blue) with surrounding waterbodies, parkland and woodland habitat.

2.2 1999 parkland tree survey

A comprehensive parkland tree survey was carried out on the Harewood Estate in summer 1999 (Hutton Forestry, 1999). Information was recorded on the species; diameter at breast height (DBH) (cm), measured at 1.3 m above ground level; height (m); age classification (young growing phase, semi-mature, mature); and an individual ID number for each trees. Notes were also made on the trees' current condition, with particular reference to any major defects that were apparent and recommended tree removals.

The 1999 survey data was digitised and georeferenced (Supplementary Appendix S1). For quality assurance, the grid reference coordinates and descriptions of the trees in 1999 were aligned during 2020 fieldwork, to confirm the trees had been correctly georeferenced.

2.3 Field data collection

566 trees were surveyed as part of this study (Supplementary Appendix S2). Information on the tree species and location was recorded, along with detailed field measurements to assess the size and condition of each tree.

Field data collection combined methodologies from the Veteran Tree Specialist Survey Methodology (Fay and De Berker, 1997), i-Tree Eco (i-Tree, 2020a), Capital Asset Value for Amenity Trees (CAVAT) (Neilan, 2017; Doick et al., 2018), the catalogue of Tree-related Microhabitats (TreMs) (Kraus et al., 2016; Larrieu et al., 2018) and signs and symptoms of notable tree pests and diseases (OPAL, 2013; Forest Research, 2020) (Supplementary Appendix S2).

Fieldwork was carried out during the tree leaf-on season in August-October 2020 for trees in the South Park and Deer Park. In the North Front, 64 trees were assessed in July-August 2017 as part of the Peacock et al. (2018) study, with an additional 5 trees assessed in 2020 to correspond with all those surveyed in 1999.

For this study, a tree was defined as a woody plant species with a DBH of greater than or equal to 7 cm (Larrieu et al., 2018; Gugan et al., 2019), or at least 2 m in height (Rural Payments Agency and Natural England, 2016). Trees that were initially surveyed in the South Park, Deer Park and North Front in the 1999 parkland tree survey (Hutton Forestry, 1999) were re-surveyed, as well as all the protected sapling trees planted since 1999. Tree species identification was reconfirmed to confirm the initial 1999 identification. Shrubs and other herbaceous plants were not surveyed in detail as part of the study.

DBH was measured at a height of 1.3 m (Hutton Forestry, 1999) unless swellings, burrs, branches or other irregular features occurred at 1.3 m height, then measurements were taken at the nearest point where the trunk was more regular (Fay and De Berker, 1997; White, 1998; i-Tree, 2020a).

2.4 TreM

In the UK, the standard methodology for surveying veteran trees is the Veteran Trees Initiative Specialist Survey Methodology (Fay and De Berker, 1997). This was developed to record veteran trees and their veteran tree features in a standardised way. However the aim of the methodology was not to produce data for scientific analysis and thus the methodology is unsuitable for this (Natural England, 2015), with subsequent research taking elements of the methodology and amending it to suit specific research aims (Read et al., 2010; Hall and Bunce, 2011). Therefore this loses the standardisation and ability to compare research carried out using the same methodology.

The Tree-related Microhabitats (TreMs) methodology and typology was formalised in 2018 to standardise tree biodiversity monitoring for forest sites (Kraus et al., 2016; Larrieu et al., 2018). Although not specifically designed for surveying veteran trees, there is much crossover between the TreMs and Specialist Survey methodologies, making it useful for studying veteran trees. The TreMs methodology has also been used to predict the drivers of TreM occurrence (Asbeck et al., 2019), assess TreM suitability as biodiversity indicators (Basile et al., 2020), and inform the management of commercial forests (Asbeck et al., 2021). However up until this study, it has predominantly been used in managed forestry habitats and the typically younger trees associated with these forestry sites. Therefore this thesis will be the first study to use the TreMs methodology for veteran trees in a parkland habitat.

The TreM catalogue includes 64 individual TreM types which can be grouped in to 8 forms (Appendix A). The additional category of fallen deadwood units was added to the fieldwork collection methodology from the Veteran Tree Specialist Survey Method (Fay and De Berker, 1997), to further quantify important deadwood habitat which is not possible in closed-canopy forest habitats.

A sub-set of 249 trees were fully assessed for TreMs in South Park and Deer Park as part of the study. Planted sapling trees were found to have no TreMs and were therefore excluded from further analysis. The North Front was not sampled for any tree microhabitats due to time constraints.

In the literature, the terms ‘veteran’, ‘ancient’, ‘large old’, ‘notable’, ‘heritage’ and ‘champion’ are often used interchangeably (Read, 2000; Fay, 2002; Pautasso and Chiarucci, 2008; Lindenmayer and Laurance, 2017) despite subtle distinctions in their definitions, which are important to clarify when classifying trees with reference to size, age or other characteristics (Woodland Trust, 2008b; Lonsdale, 2013; Nolan et al., 2020). For the purpose of this thesis, a veteran tree is defined as a tree showing ‘veteran’ characteristics e.g. crown retrenchment, trunk hollowing, fissured bark, and the presence of saproxylic organisms (Fay and De Berker, 1997; Dujesiefken et al., 2016; VETcert, 2019). An ancient tree is defined as a tree showing ‘veteran’ characteristics and is of great chronological age for the species. Age is primarily based on trunk DBH (White, 1998), and approximate age-diameter relationships have been calculated for most common UK tree species

(Mitchell, 1974; Defra, 2007; Hand et al., 2019). All ancient trees are veteran trees, therefore any use of the term ‘veteran’ in this thesis refers to both veteran and ancient trees unless specified otherwise.

2.5 i-Tree Eco

The i-Tree suite of software was initially developed for use in urban areas by the United States Department of Agriculture (USDA) Forest Service, and has now been adapted for use in other countries, including the UK (with Forest Research), using location-specific inputs. Previous regulating ES valuation of trees has involved published formulas to work out specific ecosystem service values, for example allometric equations to work out carbon storage values (Chave et al., 2005), whereas i-Tree Eco combines this in to one programme. i-Tree also allowed the valuation of ES on an individual tree-scale, being more appropriate for open-grown individual parkland trees, as opposed to previous methods which analysed ES on a landscape-scale (Torralba et al., 2016).

This study used i-Tree Eco version 6.0.22 (i-Tree, 2020c) to quantify the forest structure, regulating ES and economic value of the parkland trees. A complete tree inventory software configuration was used for each parkland site. Pollution and weather data from the Church Fenton weather station near Leeds in 2013 was used for modelling to correspond with the i-Tree project configuration used by Peacock et. al. (2018) and Gugan et al. (2019).

i-Tree Eco can be run using the minimum mandatory data of tree species and DBH. The addition of recommended field data allows for greater accuracy of the model outputs and includes: height of tree, height of canopy, size of canopy, light exposure, crown percentage missing, crown percentage dieback, and land use type. Tree measurements are then combined with local pollution and weather data, and tree reference data sets to model tree structure (including leaf area and biomass, canopy cover), structural ‘tree replacement’ value as well as estimated carbon storage, carbon sequestration, hydrology effects (avoided run-off, interception, transpiration), and air pollution removal ecosystem service benefits and monetary valuations of these services (Appendix B; i-Tree, 2020b). i-Tree Eco analysis on 2017 and 2020 field data was run with the full recommended field data set. To allow for comparison in outputs of tree services in 1999 and 2020, trees species, DBH, height of tree, light exposure, and land use type were used in analysis as this information was present in both datasets.

2.6 CAVAT

This study used CAVAT (Capital Asset Value for Amenity Trees) to quantify the amenity value of parkland trees by estimating their replacement cost. This study surveyed 497 trees in the South Park and Deer Park using the Full Method (Neilan, 2017; Doick et al., 2018). The North Front was not surveyed for amenity value due to time constraints.

Field data was collected on: crown percentage condition, reduced if canopy functionality was impaired by poor tree health; amenity value, factors taken in to account that increase the benefit of the tree to the community; appropriateness of the tree to the location, factors that decrease the benefits of the tree to the community; and life expectancy, based on 80 years; in addition to DBH and crown percentage completeness, which was based on the inverse of i-Tree crown percentage missing values rounded to the nearest 10% (Neilan, 2017; Doick et al., 2018).

Amenity values were adjusted based on tree characteristics specified in the CAVAT methodology (Neilan, 2017; Doick et al., 2018). All trees surveyed were automatically assigned an amenity value of 10%, as they are an integral part of a designed, protected parkland landscape (Firth, 1980; Historic England, 1984). Trees that were observed to be of particular importance to other wildlife (barn owl *Tyto alba* nest site, bat roost, woodpecker nest holes), or had local or commemorative importance ('two brothers' trees, wedding anniversary planting) were given an additional amenity value of 10%. Trees classified as veteran or ancient were given an additional amenity value of 30%. Veteran tree categorisation was estimated based on the presence of multiple ancient and veteran tree features (Fay and De Berker, 1997; Dujesiefken et al., 2016; VETcert, 2019), in particular:

- Whether the tree had been previously recorded in the Woodland Trust's Ancient Tree Inventory (Woodland Trust, 2021),
- Large DBH or of great chronological age for the species, established by Appendix 10 in the UK Hedgerow Survey Handbook (Defra, 2007),
- Trunk hollowing and the loss of a continuous functional sapwood 'outer shell' (Ranius et al., 2009),
- Crown retrenchment,
- Formation of a secondary tree crown,
- Reiterative and epicormic growth (Drénou et al., 2015),
- Visible fungal activity.

Tree appropriateness to location was decreased by 10% for trees that were causing obstruction, inconvenience or were noticeably unsightly (potential branch failures over public footpaths and roads, severe recent storm damage). Inappropriateness to location was still taken in to account in South Park despite the trees being only visible from private land, as visitors to the estate could still encounter them (Doick et al., 2018).

Tree life expectancy was estimated for all trees based on categories up to 80 years. Less than 80 years life expectancy and life expectancy value was reduced, more than 80 years life expectancy and life expectancy value was not reduced. Life expectancy was defined as the timescale that a tree can be reasonably retained in its location and substantially in its present form (British Standards Institution, 2012; Doick et al., 2018). No reduction was made for condition, such as a structural weakness, where life expectancy was not shortened and the tree was judged to be safe.

Ash *Fraxinus excelsior* trees were estimated to have a life expectancy of 10 years if they showed signs of significant ash dieback *Hymenoscyphus fraxineus* (Coker et al., 2019; Bengtsson et al., 2021). Life expectancy was also decreased if any tree showed signs of hollowing, major structural damage (natural pollarding, major trunk hollowing), and the presence of certain wood-decay fungi (honey fungus *Armillaria mellea*, beefsteak fungus *Fistulina hepatica*, chicken of the woods *Laetiporus sulphureus*, hen of the woods *Grifola frondosa*). Presence and location of epicormic growth increased the estimated life expectancy of a tree as it can indicate vitality of different regions of a tree (Fay and De Berker, 1997; Dujesiefken et al., 2016).

Field measurements were then combined with data on: the unit value factor (UVF), an annually adjusted price to represent the cost of a newly planted tree in £ per cm^2, currently set at £16.26 (London Tree Officers Association, 2021); community tree index (CTI) factor, which accounts for population density, adjusted to Leeds (100%) as the Harewood Estate falls within the Leeds conurbation (London Tree Officers Association, 2021); and location or 'public accessibility factor', defined as how visible each tree is from a public place (Doick et al., 2018; London Tree Officers Association, 2021). The location factor was set at 25% for South Park as it is only visible from private land; Deer Park was set at 100% as there are multiple public rights of way footpaths within the parkland, meaning all surveyed trees were visible from a public place.

However, CAVAT valuation is arguably a subjective judgement due to the need for previous arboricultural knowledge as well as differing opinions on positive or negative characteristics, despite the reproducible methodology (Doick et al., 2018; Price, 2020). Regardless, many i-Tree studies done in the UK have also supplemented their studies with CAVAT to address a lack of cultural ecosystem service considerations by the i-Tree software – including studies in London (London i-Tree Eco Project, 2015), Wrexham (Rumble et al., 2015), Leeds (Gugan et al., 2019; Mooney et al., 2020), and Hyde Park in London (Rogers et al., 2017). i-Tree Eco and CAVAT field measurements overlap, so this is easily done without adding significant additional time to surveying.

2.7 GIS analysis

ArcGIS Pro mapping and analysis software version 2.7.1 (ESRI, 2020) was used to measure the distance from an individual tree to the next nearest tree and to the nearest woodland. Datasets of woodland (Forestry Commission, 2018) and ancient woodland (Natural England, 2021) land cover were downloaded from data.gov.uk and combined in order to calculate the distance to any type of woodland in the vicinity.

2.8 Statistical analysis

Statistical analysis and modelling was carried out using R statistical software package version 3.6.3 (R Core Team, 2020).

The trees were grouped broadly into DBH size classes for analysis: ≤35 cm young phase, all the planted sapling trees fall within this size class; 36-65 cm semi-mature phase; 66-95 cm mature phase; 96-125 cm mature phase; ≥126 cm over-mature and veteran phase based on the previous 1999 parkland tree survey (Hutton Forestry, 1999), BS 5837:2012 tree categorisation assessment (British Standards Institution, 2012) and Hand et al. (2019) tree age classifications. This was for a better understanding of the relationship between tree size and the dependent variables. Trees were further classified in to veteran tree ages based on a large DBH or great chronological age for the specific species, established by Appendix 10 in the UK Hedgerow Survey Handbook (Defra, 2007), and age classification ranges based on the maximum achievable size and age recorded for tree species in Great Britain (Hand et al., 2019). Veteran trees were either classed as 'not veteran', 'notable', 'valuable' or 'ancient'. This further classification allowed for small stature trees (e.g. Crab Apple *Malus sylvestris* with a DBH of 64 cm), which would be classed as veteran based on their relative size for the species, to be correctly analysed as veteran to account for their old age which would be incorrectly analysed as young or semi-mature if grouped in to a low DBH size class.

Cedar *Cedrus* species (Cedar of Lebanon *Cedrus libani*, Atlas Cedar *Cedrus atlantica*, Blue Atlas Cedar *Cedrus atlantica* 'Glauca') were pooled for analysis as they are phenotypically similar (Pijut, 2000; Jasińska et al., 2013). Beech *Fagus sylvatica* and Copper Beech *Fagus sylvatica* 'Purpurea' were analysed together as they are genetically similar (Gallois et al., 1998; Packham et al., 2012). Species with less than six individuals were grouped for analysis into the category 'Other species' (Field Maple *Acer campestre*, Norway Maple *Acer platanoides*, Walnut *Juglans regia*, Crab Apple *Malus sylvestris*, Scots Pine *Pinus sylvestris*, Wild Cherry *Prunus avium*, Scarlet Oak *Quercus coccinea*, Daimyo Oak *Quercus dentata*, White Willow *Salix alba*, Crack Willow *Salix fragilis*) to prevent over-parameterisation. The species analysed were Sycamore *Acer pseudoplatanus*, Horse Chestnut *Aesculus hippocastanum*, Alder *Alnus glutinosa*, Sweet Chestnut *Castanea sativa*, *Cedrus* spp., *Fagus sylvatica*, Ash *Fraxinus excelsior*, London Plane *Platanus* x *hispanica*, Pedunculate Oak *Quercus robur,* Small-leaved Lime *Tilia cordata* (Appendix C).

Percentage tree crown dieback was used as a proxy for tree condition, as specified in i-Tree Eco for estimating the compensatory value of trees (Nowak, 2020):

- Excellent (< 1% dieback)
- Good (1-10%)
- Fair (11-25%)
- Poor (26-50%)
- Critical (51-75%)
- Dying (76-99%)
- Dead (100%)

Dead trees were kept in analysis as a tree condition category. Further details of statistical analysis methods are given in the relevant chapters.

The input for the statistical analysis and models was largely based on human collected field data, which is subject to unavoidable errors and biases. However, human collected data also benefitted analysis by allowing a large number of trees to be assessed, the same individual carrying out both data collection and analyses, and resampling of previously assessed trees for temporal analysis.

3 The biodiversity values of parkland estate trees

3.1 Introduction

The parkland estates of stately homes have been shown to be refugia for biodiversity (Liira et al., 2012; Šantrůčková et al., 2017; Liira et al., 2020), acting as habitat 'islands' whilst urbanisation and rural intensification occur around them (Glendell and Vaughan, 2002; Fornal-Pieniak et al., 2018; Šantrůčková et al., 2019). Trees are important for this biodiversity, both in terms of their individual contribution to biodiversity and in the role they play in providing habitats for other species. Their value can be quantified at a community and individual tree level. Tree community diversity is important as it affects the dynamics of wider biodiversity. Canopy cover and tree species diversity are well used in forestry as simple valuation metrics that can be a good proxy for the benefits provided by tree communities and their climate change and pest resilience (del Río et al., 2016; Doick et al., 2020; Trees and Design Action Group, 2021). Tree density and age structure, including the number of veteran trees, also give an indication of how valuable a tree community is for biodiversity (Kowarik et al., 2016; Schall et al., 2018), and these are particularly important to quantify for parkland estates where the trees are open-grown and often of an old age (Latham et al., 2018).

Open-grown trees (also known as scattered trees and trees outside woodland) have been shown to be keystone structures, having a disproportionate positive effect on biodiversity and ecosystem function relative to the small area they occupy (Manning et al., 2006; Gibbons et al., 2008; Prevedello et al., 2018). Specifically, they act as habitat 'islands': providing shelter, feeding, breeding resources and distinct microclimates and microhabitats (Paltto et al., 2011; Horák and Rébl, 2013; Barth et al., 2019); and increasing landscape connectivity, by acting as stepping stones across intensively managed landscapes (Fischer et al., 2010; Frey-Ehrenbold et al., 2013). They could also help mitigate biodiversity loss by acting as biological legacies in disturbed landscapes: increasing genetic connectivity of tree populations (Lander et al., 2010) and functioning as centres for ecosystem regeneration (Derroire et al., 2016), as well as helping species adapt to climate change by facilitating movement through landscapes (Manning et al., 2009). The spatial configuration of parkland trees is also important aesthetically and culturally, for example in creating the 'green and pleasant land' planting regimes of 'Capability' Brown parkland landscapes (Alcock et al., 2015; Handley and Rotherham, 2017). However, these trees have been previously overlooked by researchers and their importance for biodiversity still remains unclear (Prevedello et al., 2018).

Different individual trees support different levels and types of biodiversity (Asbeck et al., 2019). This is largely determined by the types and amount of microhabitats present on the trees, as trees support other species largely through the provisioning of microhabitats. Studying tree microhabitats (TreMs) is therefore a useful approach for understanding what and how much diversity a tree can support. The TreMs typology was developed to formally quantify tree microhabitats (Larrieu et al., 2018). Research on TreMs using this typology has primarily focused on forestry habitats, with no studies in wood-pasture and parkland (Asbeck et al., 2021). They have also been limited in their range, predominantly within Central-Europe and the Mediterranean, with no studies looking at trees in the UK (Asbeck et al., 2021). Veteran open-grown trees are missing from the literature on TreMs, which are found in wood-pasture and parkland landscapes. Most TreM studies have used DBH as a proxy for tree age, yet the relationship between tree diameter and tree age can be extremely variable, particularly in uneven-aged forests (del Río et al., 2016; Asbeck et al., 2021). Veteran trees are underrepresented in the current TreM literature, so the use of veteran trees as a tree age category would allow the analysis of tree size and tree age to be combined.

There are a number of variables that can potentially influence the type and number of TreMs provided by a tree. For example, tree characteristics such as size, age, species and condition; as well as characteristics of the landscape in which they are found, such as isolation and land management (Asbeck et al., 2021).

The amount and types of diversity supported and provided by open-grown veteran parkland trees in these estates is currently unclear, as are the controlling factors of this diversity. The aims for this chapter are to quantify the parkland tree community diversity and tree-related microhabitat (TreM) diversity, and identify the driving factors at the tree and landscape scale of diversity (richness and community composition) of TreM structures for individual trees. Multiple tree-level and landscape-level parkland attributes were investigated which represent different features of a parkland stand which can be altered through management. It was hypothesised that:

- the richness of TreMs can be explained by tree characteristics including: DBH; veteran tree age category, as a proxy for tree age; tree species; tree condition, expressed as tree canopy condition and whether the tree is alive or dead; and leaf area.
- the richness of TreMs can be explained by the landscape management including: parkland site, as a proxy for differing management regimes; distance to the nearest area of woodland, as a proxy for landscape connectivity; and parkland structural complexity, expressed as crown light exposure and distance to the next nearest parkland tree.
- Individual TreM community composition is influenced by a combination of tree and landscape characteristics.

3.2 Methods

Field data was collected using the methodology outlined in Chapter 2.

3.2.1 Tree community diversity analysis

To quantify tree community diversity, field data and i-Tree Eco output data was analysed. Specifically, tree canopy cover and tree leaf area were calculated using the i-Tree Eco tool to show tree community diversity and tree size in relation to tree canopy size. The Simpson's diversity index (1-*D*) was calculated to summarise the diversity of parkland trees in each parkland site.

3.2.2 TreM richness analysis

To test the hypothesis that TreM richness is influenced by a combination of tree and landscape characteristics, generalised linear models (GLMs) were used to analyse the relationship between the response variable TreM richness and the predictor variables DBH size class, age category, species, tree condition, leaf area, parkland site, distance to nearest woodland, distance to nearest tree and crown light exposure of each individual tree. TreM richness was calculated as the sum of each different TreM group per tree (TreM diversity). DBH was added as a categorical variable to get a more specific result for DBH size from the main model and also act as a proxy for tree age class, applicable to the majority of broadleaf, large stature trees in the sample. Trees were also classified in to age categories to encompass trees which were of a large DBH for their species but not large in comparison to other species (e.g. Crab Apple *Malus sylvestris*).

Seven species (*Acer pseudoplatanus*, *Castanea sativa, Fagus sylvatica*, *Fraxinus excelsior*, *Platanus* x *hispanica*, *Quercus robur*, *Tilia cordata*) were analysed in the models. Species with less than six individuals were grouped for modelling into an eighth category 'Other species' (*Acer campestre*, *Acer platanoides*, *Aesculus hippocastanum, Juglans regia*, *Malus sylvestris*, *Prunus avium*, *Salix alba*, *Salix fragilis*) to prevent over-parameterisation of the models.

3.2.2.1 Model selection

In order to prevent overfitting of the final main model, two sub-models were produced to test the two TreM richness hypotheses. The first sub-model identified the most important tree-level characteristics that explain the TreM richness, whilst the second sub-model identified the most important landscape-level characteristics. The most suitable models were selected using Akaike's information criterion (AIC) values (Thomas et al., 2017). The model with the lowest AIC was chosen, except if simpler models had an AIC less than 2 units higher (Burnham and Anderson, 2004).

DBH size class, age class, tree species and parkland site were identified as being the most important predictor variables in determining TreM richness (p<0.05), so were included in the combined main model. TreM richness was count data therefore a normal error distribution was not appropriate. The analyses for both sub-models and the main model were performed with the glm function using a Poisson error distribution, as no under- or over-dispersion was detected, with a log link function.

The main model was further refined to identify why parkland site was found to be important for TreM richness. Parkland site was analysed in the main model with interaction effects for all the other variables. Leaf area was found to be the most important interaction variable with parkland site as the model had to lowest AIC score. The final main model consisted of these predictors:

$$\text{TreM richness}_{\text{per tree}} \sim \text{parkland*leaf area} + \text{DBH size class} + \text{age category} + \text{species}$$

Where * indicates both a main effect and an interaction effect calculation.

The glht function in the multcomp package was used to carry out Tukey post-hoc tests on each modelled categorical variable, in order to identify which categorical groups were significantly different from each other that were included in the final model (Hothorn et al., 2008).

3.2.3 TreM community composition analysis

To test the hypothesis that TreM composition is influenced by a combination of tree and landscape characteristics, multivariate modelling was used. The effects were analysed of DBH (as a continuous variable), age category, species, tree condition, leaf area, parkland site, distance to nearest woodland, distance to nearest tree and crown light exposure on the abundance of specific TreM types per tree (TreM density). The mvabund package was adapted to predict TreM community abundances for the different tree and landscape variables (David et al., 2017). The package allowed both the full TreM community as well as the occurrence of specific TreMs per tree to be tested for the effects of the stated variables (Wang et al., 2012).

The analyses for both multivariate and univariate TreM abundance were performed with the anova.manyglm function using a negative binomial error distribution. The significance of the tree and landscape variables was examined with the log-likelihood ratio test (LRT) statistic; *P*-values were calculated using 999 resampling iterations via Probability Integral Transform residual bootstrap (PIT-trap) resampling (Warton et al., 2017). The model was simplified by removing non-significant ($p>0.05$) predictor variables: age class, crown light exposure and distance to the nearest tree and then repeating analysis of deviance, with all the remaining predictors found to be highly significant ($p<0.01$). The final model consisted of these predictors:

TreM community $_{\text{per tree}}$ ~ DBH + species + distance to nearest woodland + condition + leaf area + parkland site

19 out of the initial 22 TreM groups surveyed were used in the community composition analysis. Only TreM groups with more than 10 observations were included in analysis, therefore the groups "myxomycetes" (EP2), "epiphytes" (EP3), and "sap and resin runs" (OT1) were excluded (Appendix D). Vertebrate nests (NE11 and NE12) and invertebrate nests (NE21) were grouped together for analysis due to their low number of observations when separated. The direction of the effect of the predictor variables on the continuous predictors was ascertained from the model coefficients.

3.3 Results

3.3.1 Parkland tree community diversity

The density of parkland trees was similar between the Deer Park (7.5 trees/ha) and the North Front (6.8 trees/ha), with a lower density of trees in the South Park (2.6 trees/ha). The average density of parkland trees for the total parkland area surveyed was 4.6 trees/ha. Tree canopy cover was again similar between the Deer Park (10.4%) and the North Front (12.3%), with a lower canopy cover in the South Park (3.6%). The North Front had a higher canopy cover despite having less trees per hectare. The average canopy cover of parkland trees for the total parkland area surveyed was 6.7%.

Despite the species richness of the three parkland areas being the same (*S*=11), the North Front had the greater species diversity, followed by the South Park and Deer Park respectively (Table 1). All the parkland areas were dominated by *Quercus robur,* with it constituting 84% of all trees in the Deer Park, 80% in the South Park, and 55% in the North Front (Figure 3). The majority of these trees were identified as *Quercus robur*, with some suspected hybrids between *Quercus robur* and Sessile Oak *Quercus petraea*. For ease of interpretation, all suspected *Quercus robur x petraea* hybrids are referred to as *Quercus robur.* The average relative abundance of *Quercus robur* for the total parkland area surveyed was 79%. The *Cedrus* genus also made up a large proportion of the North Front trees, constituting 23% of all trees. The remaining parkland tree population was made of up of low numbers of other tree species (Figure 3). Only *Quercus robur*, *Acer pseudoplatanus* and *Fagus sylvatica* were present in all of the parkland areas. Native trees made up 89%, non-native trees made up 11% of the parkland trees. The Deer Park is made up of 94% native species, compared to 92% for the South Park and 62% for the North Front.

Table 1. Diversity data comparing the South Park, Deer Park, North Front and total study area.

Parkland site	**Number of trees**	**Species richness**	**Simpson's diversity (1-*D*)**
South Park	181	11	0.362
Deer Park	316	11	0.282
North Front	69	11	0.664
Study area	566	23	0.367

Size classes of DBH followed a similar trend regardless of parkland site. The parkland tree age structure consisted of higher numbers of young, planted sapling trees (≤35 cm) (24% of all trees) and mature trees (66-125 cm) (61% of all trees), with lower numbers of semi-mature (66-95 cm) (8% of all trees) and over-mature and veteran trees (≥126) (7% of all trees) in the population (Figure *4*).

The density of veteran trees was 0.6 trees/ha (77 veteran trees), with the Deer Park having the highest number of veteran trees per hectare at 1.2 trees/ha (51 veteran trees) followed by the South Park at 0.4 trees/ha (25 veteran trees) and the North Front at 0.1 trees/ha (1 veteran tree). Of these veteran trees, 14 trees were classed as ancient with a density of 0.1 trees/ha. All the parkland sites had a comparable number of ancient trees per hectare at 0.1 trees/ha (6 ancient trees in the Deer Park; 7 ancient trees in the South Park; 1 ancient tree in the North Front).

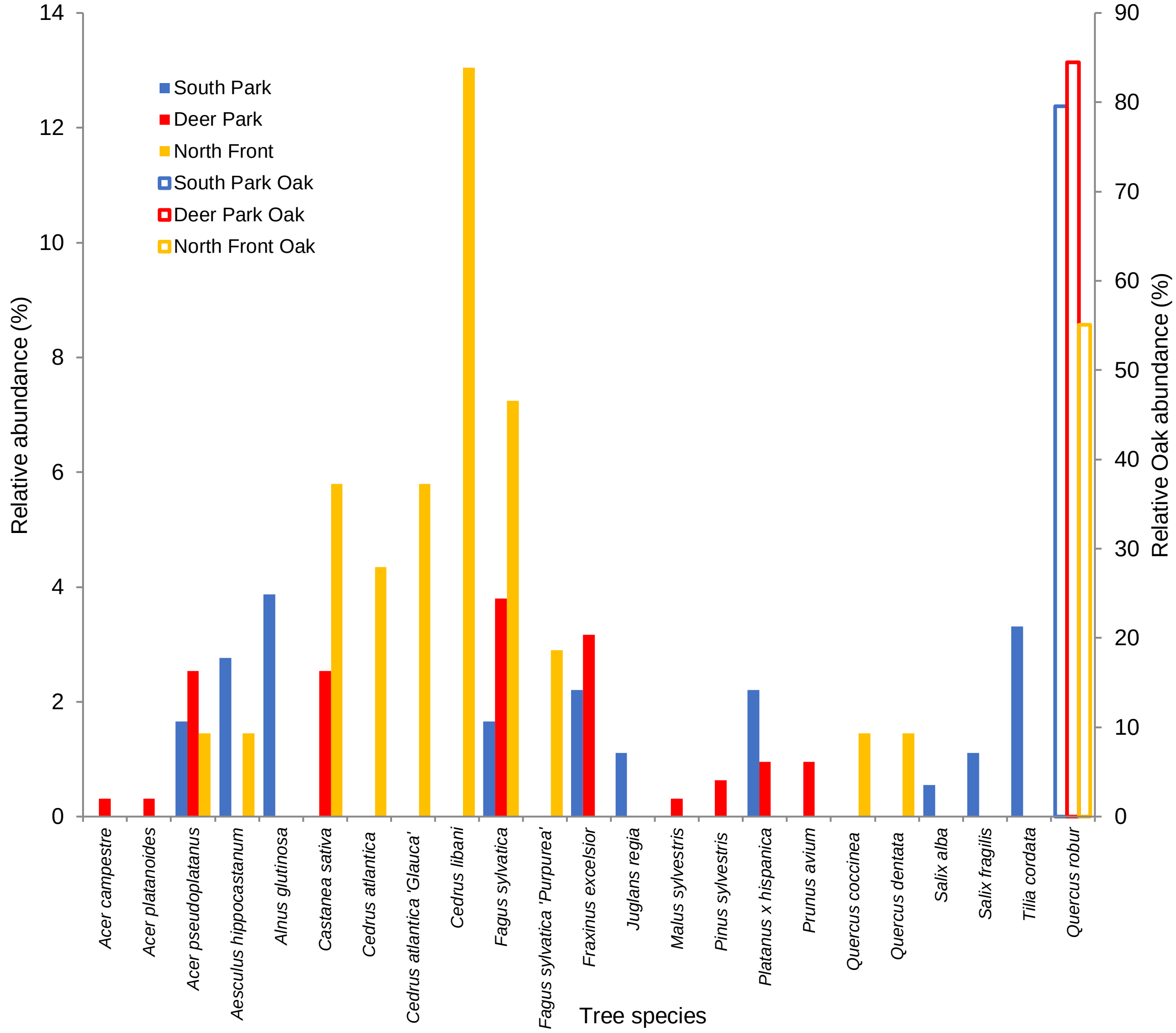

Figure 3. Relative abundance (%) of tree species in the South Park (blue), Deer Park (red) and North Front (yellow).

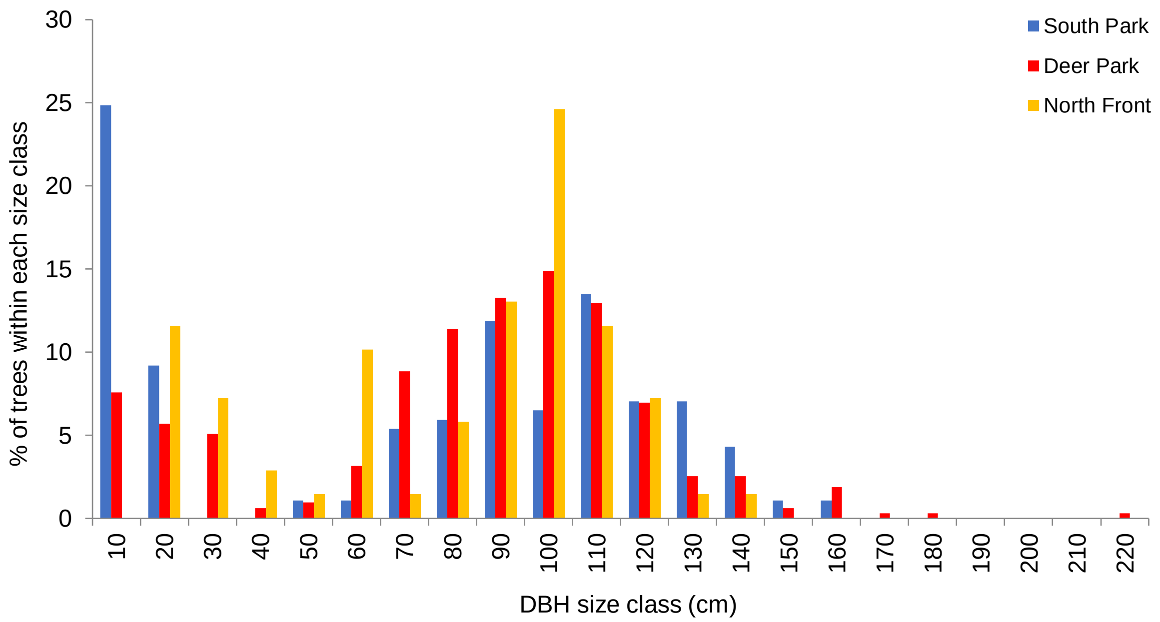

Figure 4. Tree size class distribution based on diameter at breast height (DBH) of trees in the South Park (blue), Deer Park (red), and North Front (yellow).

3.3.2 Parkland tree microhabitat diversity

3.3.2.1 TreM richness

The mean TreM richness per tree was 10, ranging between 1 and 21 different TreM types on a single tree. None of the surveyed trees (n=249) had zero TreMs (Appendix D). The results of the model selection and multi-model inference of the GLM is shown in Table 2.

Table 2. Generalised linear model (GLM) results for the prediction of TreM richness for individual trees after multi-model inference selection using scaled predictors. GLM with Poisson distribution and log link function.

Variable	**Levels**	**Estimate**	**SE**	***p*-value**	**Sign.**[a]
Intercept		1.5202	0.1517	< 2e - 16	***
DBH size class	66-95 cm	0.5364	0.1011	1.11e - 07	***
	96-125 cm	0.7362	0.1114	3.94e - 11	***
	≥126 cm	0.8849	0.1331	3.01e - 11	***
Age class	Ancient	0.4247	0.1607	0.00822	**
Tree species	*Fagus sylvatica*	-0.3306	0.1579	0.03621	*
	Fraxinus excelsior	0.3329	0.1356	0.01406	*
Parkland site	South Park	-0.1098	0.0432	0.01109	*
Leaf area (m^2)		-0.0505	0.0294	0.08589	.
Interaction	South Park* Leaf area (m^2)	0.1132	0.0507	0.02569	*

[a] *** p<0.001, ** p<0.01, * p<0.05, . p<0.1.

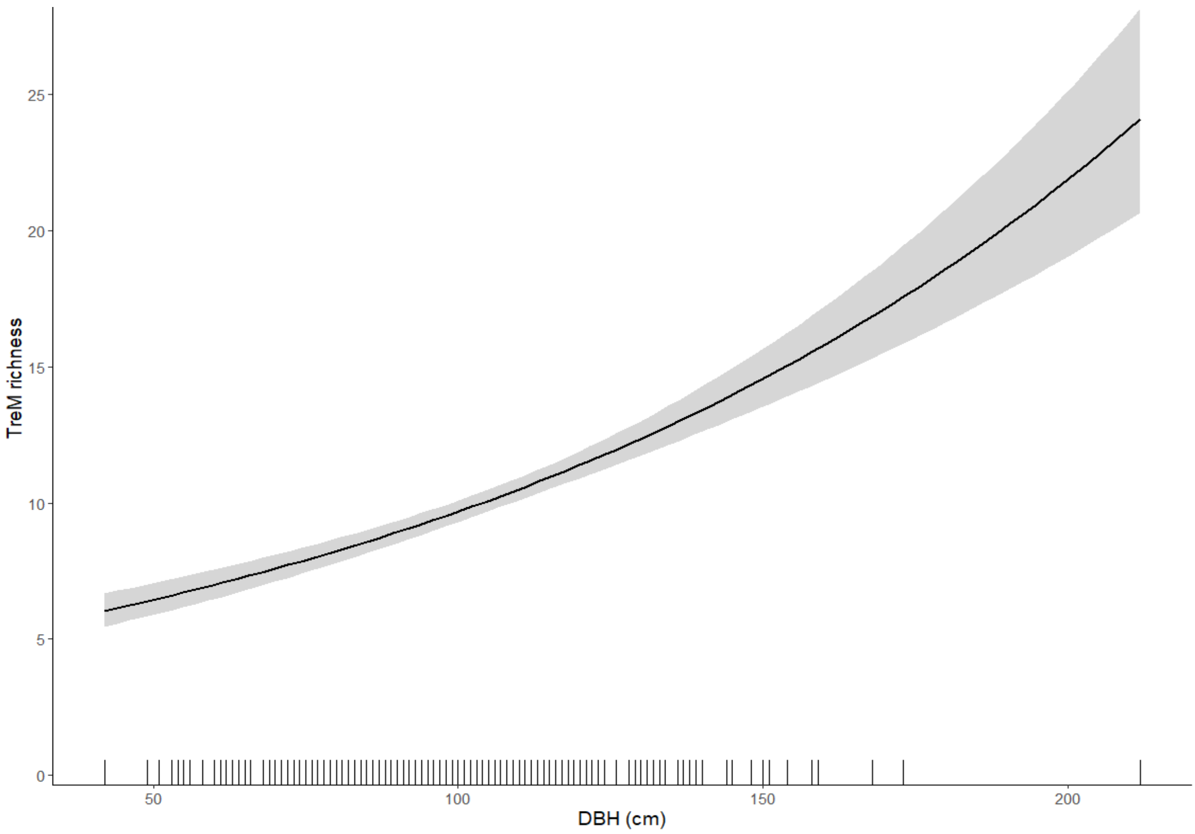

Figure 5. Relationship between tree microhabitat (TreM) richness and diameter at breast height (DBH), modelled with a generalised linear model with a Poisson distribution. Grey bands represent the 95% confidence intervals, the rug plot underneath shows the DBH distribution of the individual inventoried trees.

Tree DBH had the strongest influence on TreM richness (Table 2), suggesting is it the best predictor for TreM richness. The richness of TreMs increased significantly ($p<0.001$) with increasing DBH (Figure 5). Trees classified as ancient also had a strong influence on TreM richness (Table 2). They were a significant predictor of TreM richness and provided a significantly higher ($p<0.01$) richness of TreMs than other veteran and non-veteran tree age classes (Table 2; Figure 6).

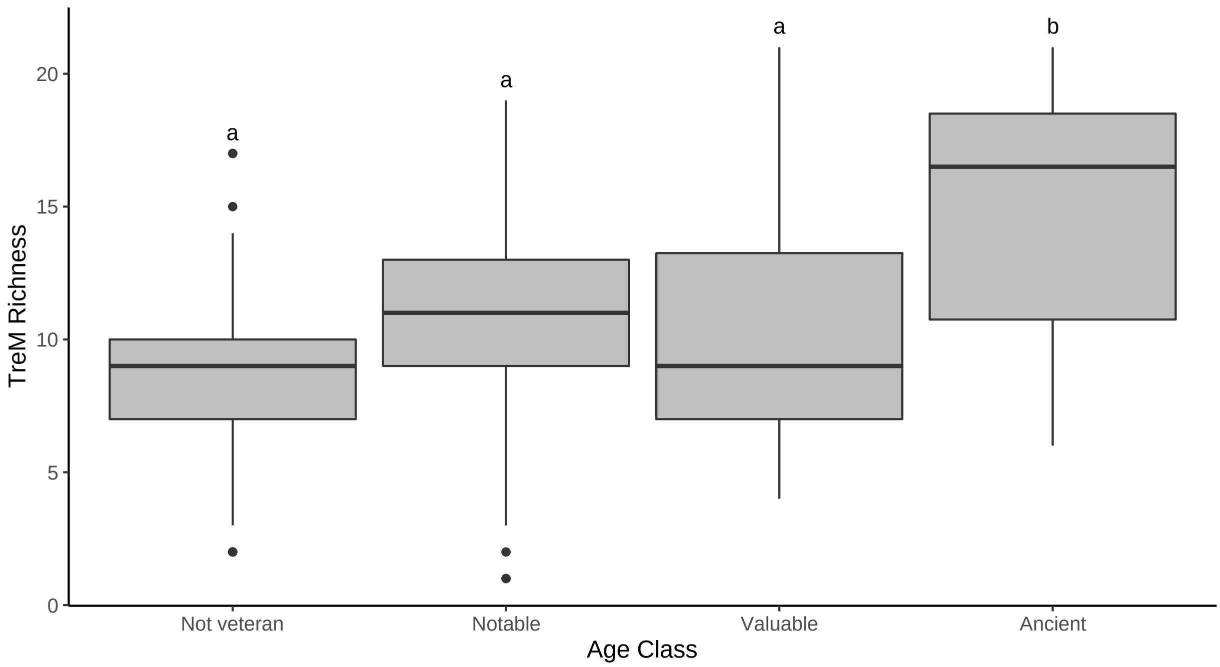

Figure 6. Tree microhabitat (TreM) richness by diameter at breast height (DBH) veteran age class. Significant differences amongst groups denoted by different letters (*p*<0. 05).

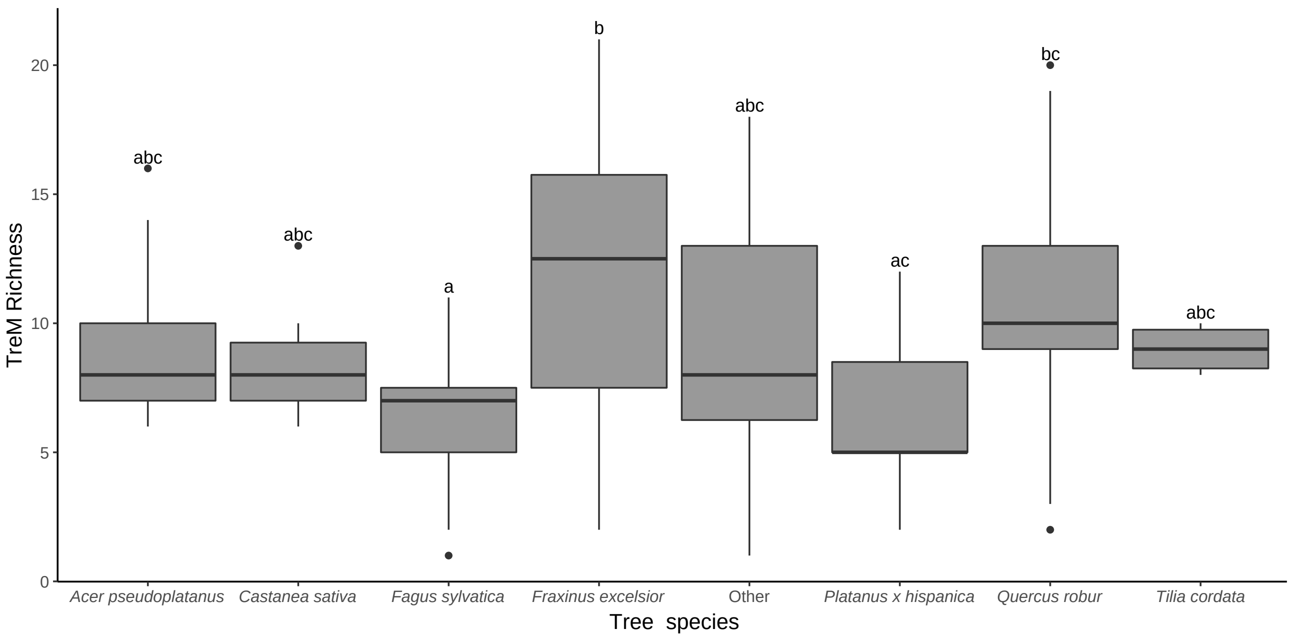

Figure 7. Tree microhabitat (TreM) richness by tree species. Significant differences amongst groups denoted by different letters (*p*<0.05). 'Other' species includes: *Acer campestre, Acer platanoides, Aesculus hippocastanum, Juglans regia, Malus sylvestris, Prunus avium, Salix alba, Salix fragilis.*

Only *Fraxinus excelsior* and *Fagus sylvatica* were significant predictors of TreM richness (Table 2). *Fraxinus excelsior* was found to be a significant ($p<0.05$) predictor of high TreM richness on an individual tree, with *Fagus sylvatica* was found to be a significant ($p<0.05$) predictor of low TreM richness on an individual tree compared to other species. *Fraxinus excelsior* and *Quercus robur* had higher TreM richness on average, and *Fagus sylvatica* and *Platanus* x *hispanica* had lower TreM richness on average than other species (Figure 7). No relationships emerged when modelling two-way species interactions with other tree and landscape characteristics by TreM richness, suggesting species alone could be a good predictor of TreM richness.

The Deer Park was found to have a significantly ($p<0.05$) higher average TreM richness than the South Park. Modelling the interaction between parkland site with other effect variables to explain this result found that South Park had a significant negative relationship ($p<0.05$) with TreM richness when leaf area was taken in to account compared to Deer Park (Table 2). Independently, leaf area was found to have no significant relationship ($p>0.05$) with TreM richness. The modelled two-way interaction between parkland site and leaf area by TreM richness illustrated TreM richness increased with increasing leaf area in the South Park, whereas the opposite was true for the Deer Park (Figure *8*). However, the large overlap of confidence intervals suggested there is a lot of similarity between the TreM richness of the two parkland sites, especially between 1000 m^2 and 2000 m^2 leaf area.

No relationships were observed for tree condition, distance to the nearest woodland, distance to the nearest tree and crown light exposure on TreM richness. None of these predictor variables were selected in the final TreM richness model when multi-model inference selection was performed.

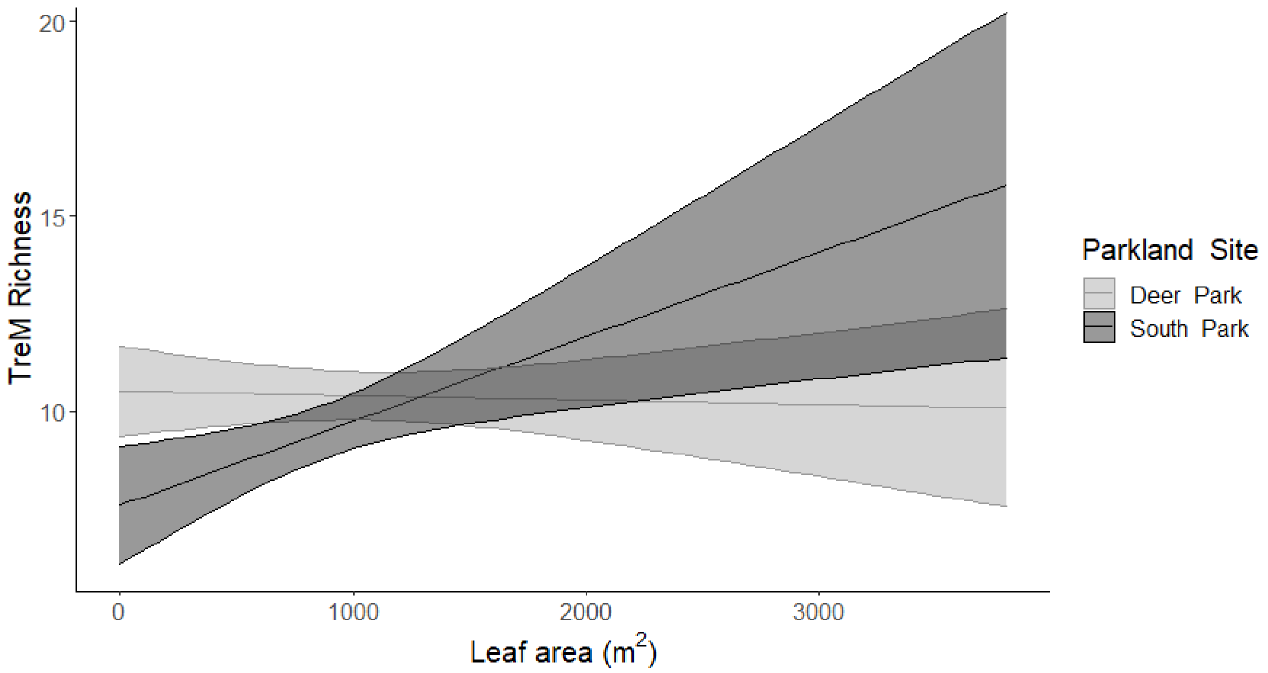

Figure 8. Two-way interaction between Deer Park (light grey) and South Park (dark grey) and leaf area by tree microhabitat (TreM) richness modelled with a generalised linear model with a Poisson distribution. Grey bands represent the 95% confidence intervals.

3.3.2.2 TreM community composition

The mean TreM abundance per tree was 20, ranging between 1 and 89 individual TreMs per tree. Dead branches had the highest average abundance per tree, followed by root buttress cavities and fallen deadwood (Figure 9; Table 4). The results of the multivariate model indicated the majority of the predictor variables were significantly related to the abundance of TreMs. DBH, species, tree condition, leaf area, parkland site and distance to the nearest woodland were highly significant predictors of TreM community composition (Table 3).

Table 3. Multivariate generalised linear model (GLM) results for the predictors of tree microhabitat (TreM) community composition of parkland trees. GLM with negative binomial distribution.

Variable	***df***	**Dev**	***p*-value**	**Sign.**[a]
DBH	1	208.8	0.001	***
Species	7	467.3	0.001	***
Distance to nearest woodland	1	45.2	0.001	***
Tree condition	5	142.5	0.001	***
Leaf area	1	42.9	0.003	**
Parkland site	1	58.8	0.001	***

[a] *** p<0.001, ** p<0.01, * p<0.05, . p<0.1.

The results of the univariate model identified 6 predictor variables that explained the abundance of 16 specific TreM groups (Table 4). TreM groups differed in their response and direction of change to the predictor variables. DBH was a significant predictor of TreM abundance, with the abundance of CV2, CV3, IN2, IN3, DE1, GR1, OT2 increasing with increasing DBH.

Tree species was a significant predictor of TreM abundance for CV3, CV4, IN2, BA2, DE1, FDU, GR1, GR2, EP1, OT2 (Table 4), with certain species having greater relative abundance of certain TreM groups compared to other species. All species had a highest relative TreM abundance of DE1 and/or FDU, except *Fagus sylvatica* which had a highest relative TreM abundance of GR1 (Figure 9). However overall, *Fraxinus excelsior* had the greatest diversity of TreMs (1-*D*=0.857), followed by *Quercus robur* (1-*D*=0.825), *Acer pseudoplatanus* (1-*D*= 0.819), *Platanus* x *hispanica* (1-*D*=0.776), *Castanea sativa* (1-*D*=0.771), *Tilia cordata* (1-*D*=0.756), *Fagus sylvatica* (1-*D*=0.576).

Distance to the nearest woodland was a significant predictor of TreM abundance, with the abundance of CV5 and DE1 increasing with decreasing distance to the nearest woodland. Tree condition was a significant predictor of TreM abundance for IN1 and DE1. Parkland site was a significant predictor of TreM abundance for DE1 and FDU, with both TreMs occurring more frequently in the Deer Park than the South Park. Leaf area was not found to explain the abundance of specific TreM types, with no significance with any individual TreMs. The abundance of woodpecker cavities (CV1), bark space (BA1), cankers and burrs (GR3), and nests (NE) was not explained by any of the predictor variables.

Table 4. Univariate generalised linear model results. Significance[a] of the effect of the predictors DBH, tree species, distance to the nearest woodland, tree condition, tree crown leaf area and parkland site on the abundance of individual tree microhabitat (TreM) types, including the direction of change for continuous predictors[b] or comparison of average abundance for categorical predictors.

TreM group	**Average abundance per tree**	**DBH**	**Species[c]**	**Distance to nearest woodland**	**Tree condition[d]**	**Leaf area**	**Parkland site**
Trunk cavities (CV2)	0.86	↑**	*Pxh* < *Tc* < Fs < *Cs* < *Qr* < Ot < *Ap* < *Fe* .				
Branch holes (CV3)	0.98	↑***	*Cs* < *Tc* < *Qr* < *Fs* < *Pxh* < *Ap* < Ot < *Fe* *				
Dendrotelms (CV4)	0.38		*Pxh* < *Cs* < *Qr* < *Fe* < Ot < *Fs* < *Tc* < *Ap* ***				
Insect galleries and bore holes (CV5)	0.30			↓*		↓.	
Bark loss (IN1)	0.05		*Cs* = *Fe* = *Pxh* < *Qr* < Ot < *Fs* < *Ap* < *Tc* .		CR = FA = EX < GO < PO < DE *		
Exposed heartwood (IN2)	0.90	↑***	*Pxh* = *Tc* < *Fs* < *Cs* < *Ap* < *Fe* < Ot < *Qr* ***				
Cracks and scars (IN3)	0.29	↑**					
Bark texture (BA2)	0.90		*Fs* < *Pxh* < Ot < *Fe* < *Tc* < *Ap* < *Cs* = *Qr* **				
Dead branches (DE1)	6.00	↑***	*Ap* = *Fs* < Ot < *Pxh* < *Fe* < *Tc* < *Qr* < *Cs* ***	↓*	EX < DE < GO < PO < FA < CR ***		DP > SP *
Fallen deadwood (FDU)	3.08		*Pxh* = *Tc* < *Fs* < *Cs* < *Qr* < Ot < *Fe* < *Ap* *				DP > SP **
Root buttress cavities (GR1)	3.77	↑***	*Cs* < Ot < *Pxh* < *Fe* < *Qr* < *Ap* < *Tc* < *Fs* ***				
Twig tangles (GR2)	1.37		*Fs* < *Pxh* < *Ap* < Ot < *Fe* < *Qr* < *Cs* < *Tc* ***				
Burrs and cankers (GR3)	0.25					↓.	
Fruiting bodies fungi (EP1)	0.24		*Tc* < Ot < *Cs* < *Pxh* < *Qr* < *Fs* < *Ap* < *Fe* *				
Nests (NE)	0.14		*Fe* = Ot = *Pxh* < *Fs* < *Ap* < *Qr* < *Cs* = *Tc* .				
Microsoil (OT2)	0.37	↑***	*Cs* = *Tc* < *Qr* < *Fs* < *Fe* < Ot < *Ap* < *Pxh* **				

[a] *** p<0.001, ** p<0.01, * p<0.05, . p<0.1.
[b] ↑: increase in TreM abundance with increasing predictor, ↓: decrease in TreM abundance with increasing predictor.
[c] *Ap*: *Acer pseudoplatanus*, *Cs*: *Castanea sativa*, *Fs*: *Fagus sylvatica*, *Fe*: *Fraxinus excelsior*, *Pxh*: *Platanus* x *hispanica*, *Qr*: *Quercus robur*, *Tc*: *Tilia cordata*, Ot: Other species.
[d] DE: Dead, CR: Critical, PO: Poor, FA: Fair, GO: Good, EX: Excellent.

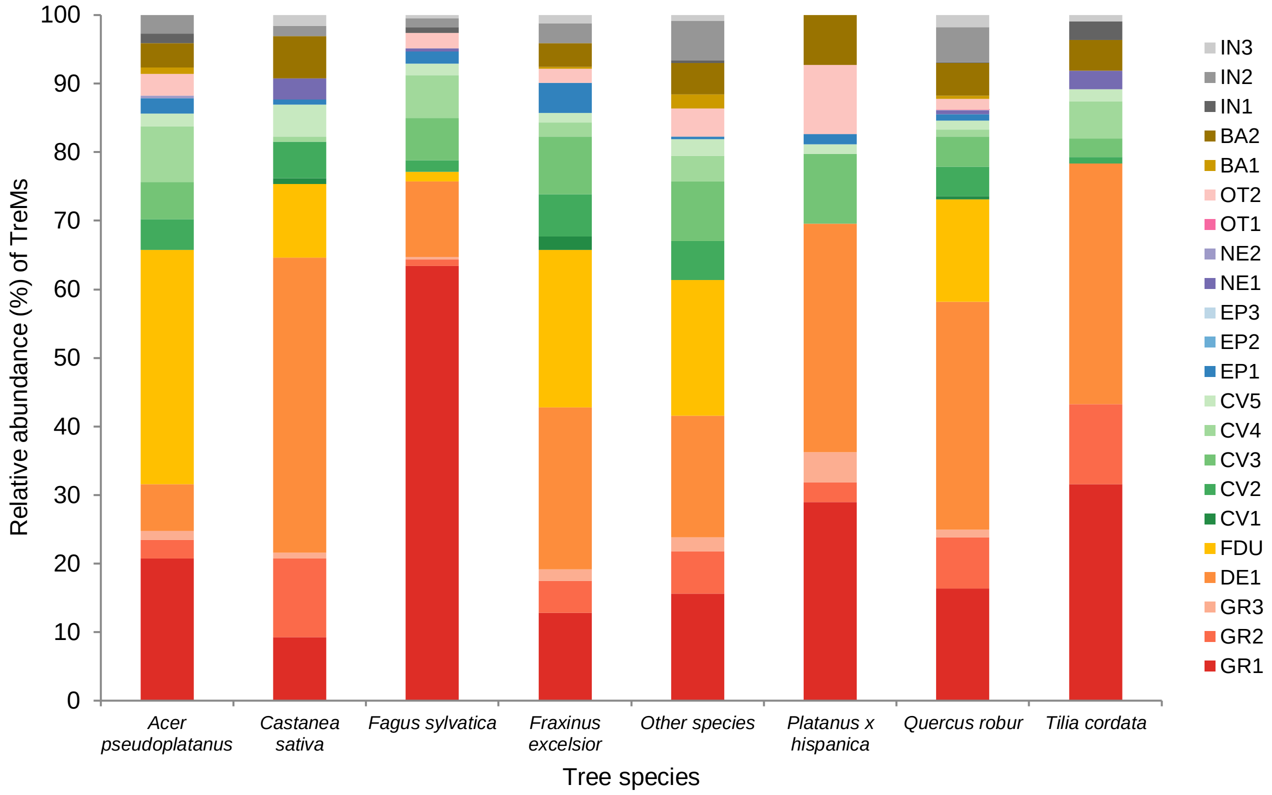

Figure 9. Relative abundance (%) of tree microhabitat (TreM) groups by tree species. Simpson's evenness ($E_{1/D}$) of TreMs for species: *Acer pseudoplatanus* ($E_{1/D}$=0.345); *Castanea sativa* ($E_{1/D}$=0.311); *Fagus sylvatica* ($E_{1/D}$=0.157); *Fraxinus excelsior* ($E_{1/D}$=0.436); Other species ($E_{1/D}$=0.524); *Platanus* x *hispanica* ($E_{1/D}$=0.496); *Quercus robur* ($E_{1/D}$=0.287); *Tilia cordata* ($E_{1/D}$=0.372). 'Other' species includes: *Acer campestre, Acer platanoides, Aesculus hippocastanum, Juglans regia, Malus sylvestris, Prunus avium, Salix alba, Salix fragilis.*

3.4 Discussion

3.4.1 Quantified tree diversity

At a parkland community level, tree species richness was 23. However 79% of all trees were *Quercus robur* with 68% of all trees mature or veteran age class. The similarity of species and age of the majority of parkland trees suggests relatively low levels of diversity within the tree communities themselves. However, parkland trees had a higher TreM diversity than trees found in broadleaf European forests, likely due to the dominance of larger, older trees.

3.4.1.1 Oaks dominate parkland estates

Over three quarters (79%) of parkland trees in the Harewood Estate were *Quercus robur* which could have implications for the future persistence of these parkland tree communities. This dominance of *Quercus robur* could be due to the original planting regime set out by 'Capability' Brown favouring oaks (Gregory et al., 2013), as well as the incorporation of them from the previous agricultural landscape of hedgerows into the parkland (Firth, 1980). This could also be due to oak being the natural tree cover of the majority of the site (Bennett, 1989), coupled with the preference for oaks in historic silviculture (Hartel et al., 2013; Williamson et al., 2017; Latham et al., 2018). In a review of the UK Ancient Tree Inventory by Nolan et al. (2020), *Quercus* spp. (*Q. robur* and *Q. petraea*) were found to be the most common genus recorded across the UK, representing 44.2% of all records (Nolan et al., 2020). This also suggests the longevity of oaks has allowed them to persist in a once more diverse parkland landscape. There is evidence that the majority of *Quercus robur* in the parkland derives from local seed sources (Firth, 1980; Nocchi et al., 2021), suggesting they have a similar genotype, have very local regeneration and are replanted from UK crop. The skew towards a single species with similar genotypes in the parkland could leave the community more vulnerable to pests, diseases and future climate change scenarios (Alfaro et al., 2014; Ekroth et al., 2019). The dominance of oak trees in Harewood Estate parkland could therefore pose a threat to the long-term health and survival of parkland landscapes.

3.4.1.2 Tree generation gaps may threaten the persistence of veteran tree biodiversity

Mature, over-mature and veteran trees are more common in the Harewood Estate parkland compared to commercial and urban forests. Compared to 68% of trees in the parkland being classified as mature, over-mature and veteran, Hand et al. (2019) found only 14% of trees in British urban forests fell within these age classifications. Hartel et al. (2013) found large, old trees are more common in wood-pastures in Eastern Europe than in forests sites, with Nolan et al. (2020) finding ~50% of ancient trees in England are located in wood-pasture or parkland. Therefore the Harewood Estate parkland is an important landscape for these older trees.

The number of veteran and ancient trees per hectare suggests the Harewood Estate parkland is of medium value under SSSI (Site of Special Scientific Interest) criteria (Latham et al., 2018). The density of veteran trees was 0.6 trees/ha (77 veteran trees); 14 of these trees were classed as ancient. The density of veteran trees is particularly important for supporting key saproxylic invertebrate species (Bergman et al., 2012). Bergman (2006) proposed an average density of 2.8 ancient or hollow oak trees/ha within a 57 ha area – a total of 160 trees - to maintain a sustainable population of veteran parkland trees and ensure a rich saproxylic fauna. The amount of fallen and canopy deadwood is also an important consideration for supporting different saproxylic invertebrate assemblages (Milberg et al., 2016; Seibold et al., 2017; Seibold et al., 2018). Dead branches and fallen deadwood had the highest average TreM abundance per tree in the parkland (Table 4). This suggests suitable habitat is available for saproxylic species despite the moderate density of veteran trees, although the diversity of tree species and microclimates provided by this deadwood is also important for high saproxylic species diversity (Vogel et al., 2020).

However, the parkland tree age structure does not follow the tree diameter curve seen in the most sustainable continuous cover forestry management practises (Mason, 2007; Barsoum et al., 2016), alluding to a threat to the long-term persistence of the veteran tree community. The Harewood Estate parkland has an unequal tree population age, lacking semi-mature trees. This generation gap was previously identified both 20 years ago (Hutton Forestry, 1999) and 40 years ago (Firth, 1980) with the gap shifting with the ageing trees. Natural tree regeneration was and is still restricted by grazing and mowing management regimes on the estate, with major tree planting efforts only within the last 15 years (Harewood Estate, 2021). This has created a time lag in the replacement of mature and veteran trees following the death of the current mature and veteran cohort, meaning there is a gap in time where there will be very few veteran tree habitats to support veteran tree biodiversity on site. Lag effects have serious implications for biodiversity associated with these trees, especially rare and threatened species which require specific TreMs only found in these larger, older, open-grown trees such as saproxylic invertebrates (Parmain and Bouget, 2018) and hollow-breeding species (Manning et al., 2013). In order to help mitigate the local absence of veteran tree-associated biodiversity, management should focus on protecting and extending the lives of the current tree stock to reduce the age gap between tree cohorts.

3.4.1.3 Parkland trees support higher diversity than other land use types

At a parkland tree level, average TreM richness was 10 per tree and average TreM abundance was 20 per tree. Whilst studies using the same TreM methodology as this study (as set out by Kraus et al. (2016)) are limited, results from those that do reveal that the larger trees found in the Harewood Estate parkland are likely able to support higher biodiversity as individuals than trees found in broadleaf European forests, due to a more diverse and greater number of TreMs present. Microhabitat richness is lower in French forest reserves, with an average TreM richness of 3 per tree for all Oak *Quercus* spp. trees and an average TreM abundance of 5 per tree for mature oaks over 66 cm DBH (Paillet et al., 2019). TreM richness and abundance is lower in the Black Forest, south-west Germany, with an average TreM richness of 2.6 per *Fagus sylvatica* tree with an average DBH of 51.5 cm and an average TreM abundance of 3.6 per *Fagus sylvatica* tree. Broadleaf species overall average TreM richness is 1.9 per tree and average TreM abundance 2.1 for an average DBH 45 cm tree (Asbeck et al., 2019). This suggests that the larger, older trees found in the Harewood Estate parkland are able to support more biodiversity than trees found in broadleaf European forests due to a more diverse and greater number of TreMs present. Thus, despite the relatively low levels of diversity within the tree communities themselves, parkland trees may actually support a greater diversity of other species than trees in more ‘natural’ landscapes, although these benefits could be offset by the relative paucity of trees in parklands compared with in broadleaf forests.

3.4.2 Large, old trees are most important for TreM diversity

Tree DBH was the strongest predictor of TreM richness and a strong predictor of TreM abundance in 37% of individual microhabitats. Many previous studies have found DBH to be the most significant driver of TreM diversity (Vuidot et al., 2011; Regnery et al., 2013; Paillet et al., 2017; Großmann et al., 2018; Paillet et al., 2019; Jahed et al., 2020), and large trees in general support a greater richness and abundance of TreMs (Paillet et al., 2017; Paillet et al., 2019; Jahed et al., 2020). This could be explained by the species-area relationship, with larger trees accumulating more (and a more diverse mosaic of) TreMs because they have the surface area to do so (Brändle and Brandl, 2001; Le Roux et al., 2015). This could also be explained by tree age, as older trees have had more time to accumulate damage and develop decay features associated with a greater diversity of TreMs, and therefore greater biodiversity (Lindenmayer and Laurance, 2017). In this study, trees classified as ancient were found to be a good predictor of TreM richness compared to younger trees, which confirms the link between larger and older trees having the greatest TreM diversity.

All the trees surveyed had at least one TreM present, with other studies of smaller, younger and evergreen trees recording a large proportion of trees without any TreMs (Larrieu and Cabanettes, 2012; Asbeck et al., 2019). The presence of TreMs could be linked to the trees in this study being older than other forest-orientated studies. The relationship between size and number of TreMs is used to formally classify trees as ancient or veteran, with trees either having to be large for that species, exhibit multiple TreMs (especially large decay features) or both (Fay and De Berker, 1997; Dujesiefken et al., 2016; VETcert, 2019). TreMs were also consistently found across size classes in parklands compared to forest trees (Asbeck et al., 2019). Therefore veteran, open-grown trees may be particularly good indicators of high biodiversity value compared to younger, healthier, crowded trees.

Few studies have investigated veteran tree TreM diversity therefore it is difficult to compare the levels of diversity amongst these parkland trees with those in other locations. However, many studies have examined standing dead trees in relation to TreM diversity. Previous studies in forests have found that TreM richness and abundance is higher on standing dead trees than live trees with comparable dimensions (Vuidot et al., 2011; Larrieu and Cabanettes, 2012; Paillet et al., 2017; Paillet et al., 2019). A small number of standing dead trees were sampled as part of this study, however many old, veteran trees were sampled. Similarities can be drawn between veteran trees and standing dead trees as they have both been found to have higher TreM diversity than younger, smaller trees and are classified by the presence of multiple TreM decay features. Indeed in this study the high volumes of dead branches and fallen deadwood suggest a similarity of microhabitats provided between the two. Trees typically enter a veteran phase before death (Dujesiefken et al., 2016), so similarities in TreM diversity are perhaps unsurprising. This suggests that if the population of veteran trees in the Harewood Estate parkland die, they will still support high levels of TreM diversity if left on site as standing dead trees. They could be used as a stop-gap to close the parkland tree population gap in the future, however species which cannot be supported by dead trees will likely still be lost.

3.4.3 Greater species diversity could lead to greater TreM diversity

Certain tree species were more important than others for TreM diversity. *Fraxinus excelsior* and *Quercus robur* had higher TreM richness on average, and *Fagus sylvatica* and *Platanus* x *hispanica* had lower TreM richness on average than other species, however only *Fraxinus excelsior* and *Fagus sylvatica* were significant predictors of TreM richness. This relationship of TreM richness to species was also found in the Chillingham Park parkland estate in northern England, with *Fraxinus excelsior* found to have a high TreM richness and *Fagus sylvatica* found to have a low TreM richness compared to other broadleaf species (Hall and Bunce, 2011). Further research in to the differences in parkland tree species' TreM diversity is needed to confirm this trend.

Tree species was a significant predictor of TreM abundance in 53% of TreM groups, with certain species having a greater relative abundance of certain TreM groups compared to other species. For example, *Fagus sylvatica* had the highest relative TreM abundance of root buttress cavities. However for the most part many parkland tree species had a low sample size (Figure 3), meaning key species important for certain TreMs were difficult to identify. Previous studies looking at species effect on TreM diversity have focused on the comparison between broadleaf and coniferous species, with no studies explicitly comparing between broadleaf species (Asbeck et al., 2021). However overall, *Fraxinus excelsior* had the greatest diversity of TreMs, followed by *Quercus robur*, *Acer pseudoplatanus*, *Platanus* x *hispanica*, *Castanea sativa*, *Tilia cordata* and *Fagus sylvatica*. *Quercus robur* has previously been linked to supporting a high diversity of species compared to other tree species (Southwood, 1961; Mitchell et al., 2019), which this study supports.

The high TreM diversity of *Fraxinus excelsior* could be attributed to the majority of *Fraxinus excelsior* surveyed within the Harewood Estate exhibiting Ash Dieback *Hymenoscyphus fraxineus* symptoms that are comparable to the old-growth characteristics exhibited by older, decaying and dying trees (Bengtsson et al., 2021). Non-native and ornamental species *Acer pseudoplatanus*, *Platanus* x *hispanica*, *Castanea sativa* had a greater diversity of TreMs than native *Tilia cordata* and *Fagus sylvatica*. However non-native species support fewer native species as there has been less time for them to form associations (Kennedy and Southwood, 1984; Brändle and Brandl, 2001), therefore their contribution to the overall parkland biodiversity is likely lower than for native trees. Different tree species provided different TreM community compositions — a diverse mosaic of microhabitats — which in turn may support slightly different niches and increase the overall biodiversity of the parkland (Alexander et al., 2006; Sjöman et al., 2016). However to further clarify the effect of tree species on TreM diversity, it may be more important to focus on tree functional groups that are related to the occurrence of certain TreMs as opposed to species (Asbeck, et al., 2020).

3.4.4 Landscape management can affect TreM diversity but is not a main driver

Landscape characteristics had a limited effect on TreM diversity compared to tree size, age and species despite management being important for the continuity of the parkland landscape. The Deer Park was more important for TreM diversity than the South Park and the Deer Park had a significantly higher TreM richness than the South Park. The interaction between parkland site and leaf area illustrated that TreM richness increased with decreasing leaf area in the Deer Park whereas the opposite was true for leaf area in the South Park. Leaf area has previously been suggested to act a proxy for tree size, canopy health and microclimate (e.g. trunk light exposure). The Deer Park was also found to be a significant predictor of higher abundance of dead branches and fallen deadwood. This suggests that the Deer Park is more important for TreM diversity due to the higher TreM richness, lower leaf area suggesting crown retrenchment, and higher abundance of canopy and fallen deadwood. This could be linked to a higher number of veteran trees per hectare found in the Deer Park than the South Park. Management continuity of the Deer Park could have led to these differences, with Johann and Schaich (2016) finding a relationship between private forest ownership and higher TreM diversity. The South Park has been more intensively managed, being previously ploughed and grazed and frequently cut compared to the Deer Park (Firth, 1980).

Other landscape characteristics were found to have a limited effect on TreM diversity. The spatial layout of trees in parkland is important aesthetically (Walerzak et al., 2015), however only decreasing distance to the nearest woodland was found to have a significant effect, increasing TreM abundance of insect galleries and bore holes as well as dead branches. These findings agree with Asbeck et al. (2020) who found no relationship between the spatial distribution of trees and their TreM diversity. The canopy cover of the Deer Park and North Front falls within the definition of wood-pasture set out by Plieninger et al. (2015) (tree canopy density >10%), however the canopy cover in the South Park does not meet this definition. Despite open-grown trees being an important feature of wood-pasture and parkland, providing distinct microclimate and microhabitats, parkland structural complexity was not a driver of TreM diversity. This is surprising as a relationship has previously been found between lower tree density in forest plots and higher TreM richness (Regnery et al., 2013). This suggests that tree size, age and species are more important drivers of TreM diversity and further study is needed on parkland structural complexity, including environmental drivers, to fully explore this relationship.

Tree management may have positively contributed to TreM diversity. Wood-pasture and parkland trees have historically undergone pollarding management (Kirby et al., 1995; Read and Bengtsson, 2019), and despite no evidence of this occurring on the parkland trees surveyed as part of this study, evidence of the removal of dead canopy branches was widespread. Pollarding has been shown to extend the lives of trees (Nolan et al., 2020) as well as increasing TreMs (specifically tree hollows) formation rate (Sebek et al., 2013), with a recent study finding that intensive tree maintenance stimulating tree microhabitat development in urban forests (Großmann et al., 2020). This suggests that trees that have undergone management via dead branch removal in the parkland could have higher TreM diversity compared to forest trees which do not undergo this management regime.

3.4.5 Conclusions

As a social–ecological system, the parkland tree community has been shaped by the parkland creation and subsequent management regime, with little input from natural factors. The similarity of species and age of parkland trees suggests relatively low levels of diversity within the tree communities themselves, which poses a threat to parkland continuity. However due to the dominance of large, old trees, the microhabitat biodiversity value of individual trees in the parkland was higher than in broadleaf forests. Tree characteristics influenced TreM diversity more than landscape management characteristics, with the mosaic of microhabitats created by larger, older and a more diverse species community of trees being important for a higher biodiversity value of parkland trees. Yet, historical tree management may have positively contributed to TreM diversity. This is one of the first studies to quantify parkland TreM diversity and identify that certain tree and landscape characteristics drive this diversity. This study supports existing research that suggests tree size, age and live status are the main drivers of TreM diversity. Understanding the biodiversity of parkland trees and what drives this can be used to inform future long-term management plans for the parkland estate. Future management should focus on individual tree-level management such as protecting and extending the lives of the current tree stock and leaving dead trees and deadwood on site, as well as increasing the number and species diversity of trees in the parkland estate.

4 The ecosystem service values provided by parkland trees

4.1 Introduction

Ecosystem services (ES) can be classified in to three categories: provisioning, regulating and maintenance, and cultural services (Haines-Young and Potschin, 2018). Provisioning services comprises the products humans obtain from ecosystems, for example food and raw materials. Regulating and maintenance services regulate and maintain ecosystem processes, for example nutrient cycling and regulating water flow. Cultural services are the non-material benefits humans obtain from ecosystems through, for example, recreation, aesthetic appreciation and a sense of place.

Trees are known to provide many ecosystem service benefits including timber (UK National Ecosystem Assessment, 2011), carbon sequestration (Nowak and Greenfield, 2018) and human well-being (Salmond et al., 2016), and ES research on them has mainly focused on forest (Gamfeldt et al., 2013), agricultural (Barrios et al., 2018) and urban ecosystems (Hall et al., 2018; Turner-Skoff and Cavender, 2019). One land use type that is under-researched in the ES literature is wood-pasture and parkland. Wood-pasture and parkland are social–ecological systems and the trees there have historically provided multiple services, including the provisioning of forest products (e.g. timber, fruit, animal fodder) and grazing land (Rackham, 1976; Bergmeier et al., 2010; Read and Bengtsson, 2019). Trees in current UK parkland estates for the most part are no longer used for their products, with their value coming from the other ES benefits and public goods they provide. Therefore this chapter will focus on the regulating and cultural service values trees provide in parkland estates, with the aim of informing estates how to best manage land for multiple ES benefits.

It is important to understand the changes in ES values over time. When considering future sustainability and resilience, long-term scales are more relevant to decision-makers (Rodríguez et al., 2006; Boyd et al., 2013). Temporal scale is also relevant to veteran trees, with many living for centuries (Lindenmayer et al., 2014). The best and most informed management decisions need to take this in to account and require data-driven decisions to guide policy and land management.

There are different ways to assess the benefits delivered by trees, which could include a monetary valuation of the asset value and the benefits the trees in question provide (Gómez-Baggethun et al., 2010; Laurila-Pant et al., 2015). Both monetary and non-monetary valuation have been demonstrated to be important to land managers and policy-makers when valuing biodiversity and ES in different scenarios (Ruckelshaus et al., 2015; Handmaker et al., 2021), therefore this chapter will provide monetary and non-monetary ES values.

This chapter builds upon the work of Peacock et al. (2018) to quantify both the regulating and cultural ES benefits of trees on different parkland sites in the Harewood Estate, investigate how different tree (size, age, species, health) and landscape (planting regime ,land management) characteristics affect ecosystem services delivery, and make use of historical parkland tree survey data to quantify changes in the trees and their ES over time. The close proximity of the Harewood Estate parkland to the Leeds urban conurbation, whilst still within a rural agricultural area, makes it an ideal setting to investigate the ES values of trees in an underrepresented yet important habitat.

Specifically, this chapter sets out to answer the following research questions:

- What are the regulating and cultural ES values provided by parkland trees?
- What differences in tree and landscape characteristics can explain differences in ES values provided by parkland trees?
- How have the parkland trees and the ES benefits they provide changed in 20 years between 1999 and 2017/2020?

4.2 Methods

The fieldwork and ES valuation methodologies are as detailed in Chapter 2.

4.2.1 Regulating and cultural ES valuation

Parkland trees were assessed for carbon storage, carbon sequestration, avoided stormwater runoff and air pollution removal regulating ES values using the i-Tree Eco tool, with tree amenity value cultural ES assessed using the CAVAT tool. Both monetary and non-monetary ES valuation was carried out for parkland trees, as detailed in Section 2.5 (regulating ES) and Section 2.6 (cultural amenity ES). Dead trees were kept in for carbon storage analysis but taken out for other regulating ES as they did not have a value. Dead trees were kept in for amenity cultural value analysis despite having a value of £0.

4.2.2 Tree and landscape characteristics analysis

To test the hypothesis that ES value is influenced by a combination of tree and landscape characteristics, statistical analysis was carried out to confirm the relative strengths of the input tree and landscape variables (Appendix B) on the i-Tree and CAVAT valuation outputs, as well as for input variables not directly used by either model.

DBH was analysed as a categorical variable (DBH size class) to get a more comprehensive result for DBH size from analysis as well as act as a proxy for tree age class, applicable to the majority of broadleaf, large stature trees in the study. Trees were also classified in to ‘veteran’ and ‘non-veteran’ categories to encompass trees which were of a large DBH for their species but not large in comparison to other species (e.g. Crab Apple *Malus sylvestris*). Veteran categorisation status was estimated based on the presence of multiple veteran tree features as detailed in Section 2.6.

The ES variables were found to be non-normal regardless of whether they were statistically transformed or not, therefore non-parametric statistics were employed to analyse these variables. Spearman’s Rank correlation was used to analyse the relationship between the ES response variables and continuous tree and landscape characteristics (leaf area, distance to nearest tree, distance to nearest woodland). Kruskal-Wallis with post-hoc Dunn tests were used to analyse the differences between categorical tree and landscape characteristics (DBH size class, veteran status, species, condition, parkland site, crown light exposure) and ES values, and *p*-values were adjusted with the Benjamini-Hochberg method to reduce the probability of type I errors occurring.

4.2.3 20 year changes analysis

To quantify the changes in parkland trees between 1999 and 2017/2020, both the growth rate and changes in regulating ES were quantified. Mann-Whitney *U* tests were used to test for differences between carbon storage, carbon sequestration, avoided runoff and pollution removal of 1999 and 2017/2020 trees.

For quality control in data processing, annual growth rates of individual trees were categorised as NA and removed from growth rate calculations for trees that were dead (n=7), not surveyed in 1999 (n=21) or the DBH was not measured in 1999 (n=4), and sapling trees planted after the 1999 parkland survey was undertaken (n=125) (Supplementary Appendix S1; Supplementary Appendix S2). Non-comparable DBH measurements taken under 1 m and above 1.6 m were removed from growth rate calculations (+-0.3 m from 1.3 m), because they differed too greatly from the 1.3 m standard DBH measurement height. Any data outliers that appeared possibly incorrect were rechecked mathematically and, if necessary, remeasured. Negative growth rate values that were likely due to sampling error and not tree decomposition or trunk hollowing were removed from growth rate calculations (n=6). Very high growth rates that were likely due to sampling error were removed from growth rate calculations (n=1). In total 400 trees were analysed to quantify annual growth rate.

4.3 Results

4.3.1 Regulating and cultural ES values provided by parkland trees

4.3.1.1 Regulating ES

Parkland trees currently provide 1,224,030 kg of carbon storage, 14,522 kg/year of gross carbon sequestration, 693 m^3/year avoided stormwater runoff and 412 kg/year air pollution removal regulating ES benefits (Table 5). This has been valued at a total of £73,442 carbon storage value and £4,055 total annual benefits value. The parkland trees in the Deer Park were found to provide the greatest total regulating ES value in terms of carbon storage (711,893 kg), gross carbon sequestration (7,729 kg/year), avoided stormwater runoff (373 m^3/year) and air pollution removal (222 kg/year) than the South Park or North Front. However, per hectare, the North Front provided the greatest values in terms of carbon storage (17,550 kg/ha), avoided runoff (16 m^3/ha/year) and pollution removal (10 kg/ha/year), with the Deer Park still providing the greatest carbon sequestration benefit per hectare (183 kg/ha/year) (Table 6). Both the North Front and the Deer Park have approximately 7 trees/hectare compared to 2.6 trees/hectare for the South Park, suggesting an increase of the number of trees on a parkland site increases the regulating ES of that site.

Table 5. Total regulating ecosystem service values of the parkland trees in the South Park, Deer Park, North Front and total study area. Total annual benefits calculated as the sum of gross carbon sequestration, avoided stormwater runoff and air pollution removal.

Parkland site	**Number of trees**	**Carbon storage**		**Gross carbon sequestration**		**Avoided stormwater runoff**		**Air pollution removal**		**Total annual benefits**
		(kg)	**(£)**	**(kg/yr)**	**(£/yr)**	**(m^3/yr)**	**(£/yr)**	**(kg/yr)**	**(£/yr)**	**(£/yr)**
South Park	181	334,704.80	20,082.33	5,736.30	344.16	156.80	238.54	93.29	482.42	1,065.32
Deer Park	314	711,892.90	42,713.47	7,728.60	463.65	372.80	566.84	221.70	1,146.56	2,177.17
North Front	69	177,431.90	10,645.90	1,057.40	63.39	163.70	247.95	96.97	501.49	812.91
Study area	**564**	**1,224,029.60**	**73,441.70**	**14,522.30**	**871.20**	**693.30**	**1,053.33**	**411.96**	**2,130.47**	**4,055.40**

Table 6. Per hectare regulating ecosystem service values of the parkland trees in the South Park, Deer Park, North Front and total study area. Total annual benefits per hectare calculated as the sum of gross carbon sequestration, avoided stormwater runoff and pollution removal.

Parkland site	**Number of trees**	**Number of trees/ha**	**Carbon storage/ha**		**Gross carbon sequestration /ha**		**Avoided stormwater runoff/ha**		**Air pollution removal/ha**		**Total annual benefits /ha**
			(kg)	**(£)**	**(kg/yr)**	**(£/yr)**	**(m^3/yr)**	**(£/yr)**	**(kg/yr)**	**(£/yr)**	**(£/yr)**
South Park	181	2.6	4,802.08	288.13	82.30	4.94	2.25	3.42	1.34	6.92	15.28
Deer Park	314	7.5	16,893.52	1,013.61	183.40	11.00	8.85	13.45	5.26	27.21	51.67
North Front	69	6.8	17,550.14	1,053.01	104.59	6.27	16.19	24.53	9.59	49.60	80.41
Study area	**564**	**4.6**	**10,037.14**	**602.23**	**119.08**	**7.14**	**5.69**	**8.64**	**3.38**	**17.47**	**33.25**

4.3.1.2 Cultural ES

Parkland trees currently provide £28,117,919 in amenity ES value (Table 7). Trees in the Deer Park were found to provide a greater total and per hectare amenity value than the South Park. The Deer Park provided £24,958,590 and £592,278/ha amenity value compared to the South Park providing £3,159,329 and £45,328/ha. This also suggests that an increase in the number of trees on a parkland site increases the cultural ES of that site.

Table 7. Total and per hectare cultural amenity ecosystem service values of the parkland trees in the South Park and Deer Park.

Parkland site	**Number of trees**	**Number of trees/ha**	**Amenity value (£)**	**Amenity value/ha (£)**
South Park	181	2.6	3,159,328.78	45,327.53
Deer Park	314	7.5	24,958,590.15	592,277.87
Study area	**495**	**4.4**	**28,117,918.94**	**637,605.42**

4.3.2 Tree and landscape characteristics driving ES values in parkland trees

The majority of ES values provided by parkland trees could be explained by differences in tree and landscape characteristics (Table 8) and the initial parameters used in the i-Tree Eco and CAVAT tools to quantify the ES (Appendix B). However, further analysis of tree and landscape characteristics revealed the relative strengths of these variables on ES valuation outputs. Tree size (DBH size class and leaf area) was the strongest driver of ES values, whereas landscape characteristics (specifically distance to nearest woodland and distance to nearest tree) were weak drivers. Correlation between other tree and landscape characteristics (species, veteran status, condition, parkland site, crown light exposure) with tree size could not be excluded, indicating these were weaker drivers of ES values. Similar relationships were identified for avoided runoff and pollution removal due to both ES being calculated using the same input variables (Appendix B).

Table 8. Significant[a] results of Kruskal-Wallis and Spearman's Rank correlation analysis confirming which ecosystem service (ES) values provided by parkland trees can be explained by different tree and landscape characteristics. Where correlation analysis was able to be carried out for continuous variables, the direction of the relationship is confirmed (+ denotes a positive relationship, - denotes a negative relationship). Test statistics given in full in Appendix E.

	Carbon storage	**Gross carbon sequestration**	**Avoided stormwater runoff**	**Air pollution removal**	**Amenity value**
DBH size class	***	***	***	***	***
Leaf area	*** +	* +	*** +	*** +	*** +
Veteran status	***	***			***
Species	***	***	***	***	**
Condition	***	***	***	***	***
Parkland site	**	***	***	***	***
Distance to nearest woodland		** +	*** -	*** -	*** -
Distance to nearest tree		** +	** -	** -	** -
Crown light exposure	***	***	***	***	***

[a] *** p<0.001, ** p<0.01, * p<0.05, . p<0.1.

4.3.2.1 Tree size

Both DBH size class and tree canopy leaf area were significant predictors of all ES values (Figure *10*). Carbon storage, gross carbon sequestration, avoided runoff, pollution removal and amenity value increased with increasing DBH. ES values were significantly lower in the young and semi-mature age classifications, increasing significantly between mature age classes, but were only significantly higher in the over-mature and veteran age classifications for carbon storage and amenity value (Figure 10.A; Figure 10.E). Leaf area was also significantly positively correlated with all ES values (Figure 10). Leaf area was used directly to calculate both avoided runoff (r_s=1.00, p<0.001) and pollution removal (r_s=1.00, p<0.001) values and therefore exhibited a strong significant positive correlation.

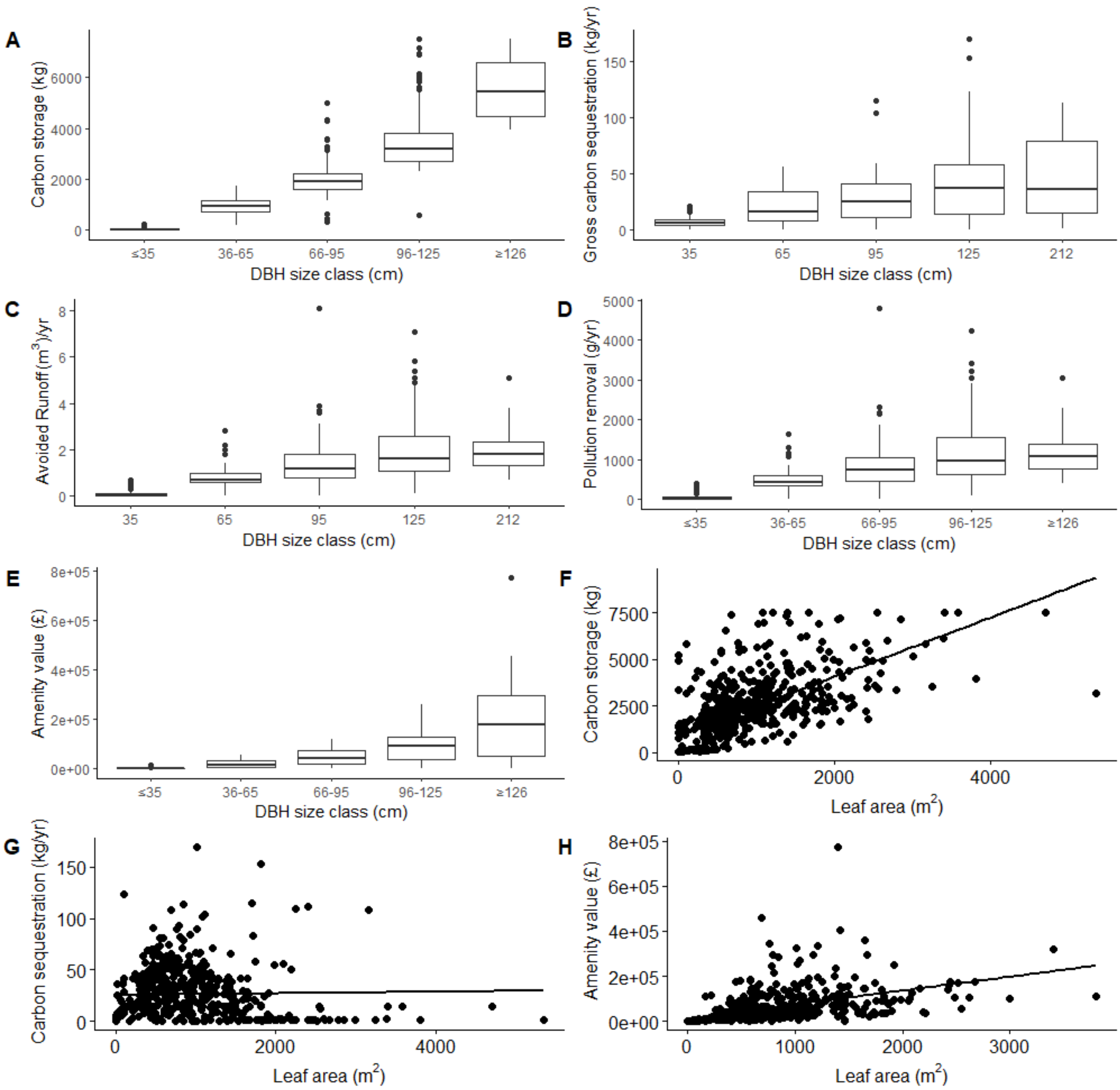

Figure 10. Relationships between tree size (DBH size class and leaf area) and ES values. DBH size class: A) Carbon storage ($\chi^2(4)$=489.360, p<0.001); B) Gross carbon sequestration DBH size class ($\chi^2(4)$=124.180, p<0.001); C) Avoided stormwater runoff ($\chi^2(4)$=318.84, p<0.001); D) Air pollution removal ($\chi^2(4)$=317.76, p<0.001); E) Amenity value ($\chi^2(4)$=305.82, p<0.001). Leaf area: F) Carbon storage (r_s=0.75, p<0.001); G) Gross carbon storage (r_s=0.09, p<0.05); H) Amenity value (r_s=0.77, p<0.001). Trendline added based on the formula '*y ~ x*'.

4.3.2.2 Tree species

Tree species ES values differed and were a strong driver of ES values despite their relationship with tree size. Carbon storage values increased with increasing DBH for all species modelled, greater in some species (e.g. *Cedrus* spp., *Fagus sylvatica*, *Platanus* x *hispanica*) and smaller in other species (e.g. *Quercus robur*, *Alnus glutinosa*) (Figure 11). The carbon storage values modelled by i-Tree Eco are capped at 7,500 kg to prevent overestimation of the value for very large trees. Eight trees had their carbon storage values capped (*Cedrus libani* n=3, *Fagus sylvatica* n=1, *Fraxinus excelsior* n=1, *Platanus* x *hispanica* n=2, *Quercus robur* n=1), suggesting these trees had the greatest carbon storage values.

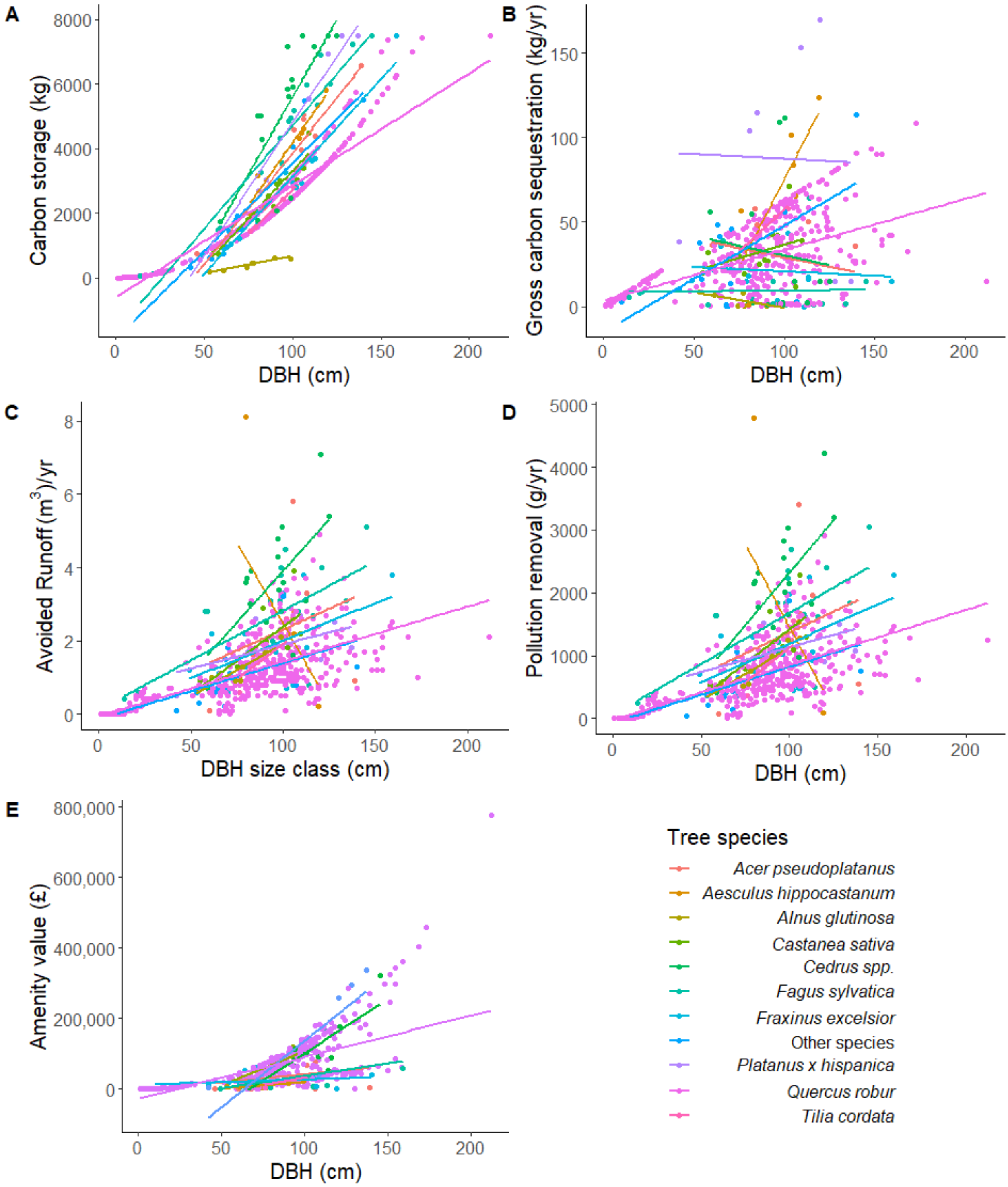

Figure 11. Relationships between DBH (cm) and carbon storage (kg), gross carbon sequestration (kg/yr), avoided stormwater runoff (m^3/yr), air pollution removal (g/yr) and amenity value (£) of individual parkland trees by species. Trendlines fit a linear regression.

Gross carbon sequestration values both increased and decreased for species depending on whether a tree was considered to have reached maturity. Tree maturity is defined by i-Tree Eco as when a tree species is reaching its maximum growth height (Nowak, 2020). The Eco model decreases the annual growth rate when a tree is reaching maturity, causing a decline in gross carbon sequestration values. The DBH at which a species was considered to have reached maximum growth height differed between species. Avoided runoff and pollution removal species relationships were analogous, reflecting an increase in leaf area with increasing DBH. *Cedrus* spp. showed the highest increases in avoided runoff and pollution removal relative to their size due to being evergreen, as opposed to broadleaf trees which have very limited avoided runoff and pollution removal values in the leaf-off season (Nowak, 2020). *Aesculus hippocastanum* showed a negative relationship in avoided runoff and pollution removal relative to their size likely due to the small sample size (n=6). Amenity value also increased with increasing DBH, although the influence of other tree and landscape characteristics likely better explain species ES values.

4.3.3 20 year parkland tree changes

4.3.3.1 Growth rate

The maximum annual growth rate recorded was 2.00 cm per year, with the minimum annual growth rate recorded at 0.03 cm per year. Annual growth rate increased with increasing tree size; on average significantly higher for the largest parkland trees than for smaller trees (Figure 12). Annual growth rate increased significantly between mature (66-95 cm and 96-125 cm) and over-mature and veteran (≥126 cm) size classes.

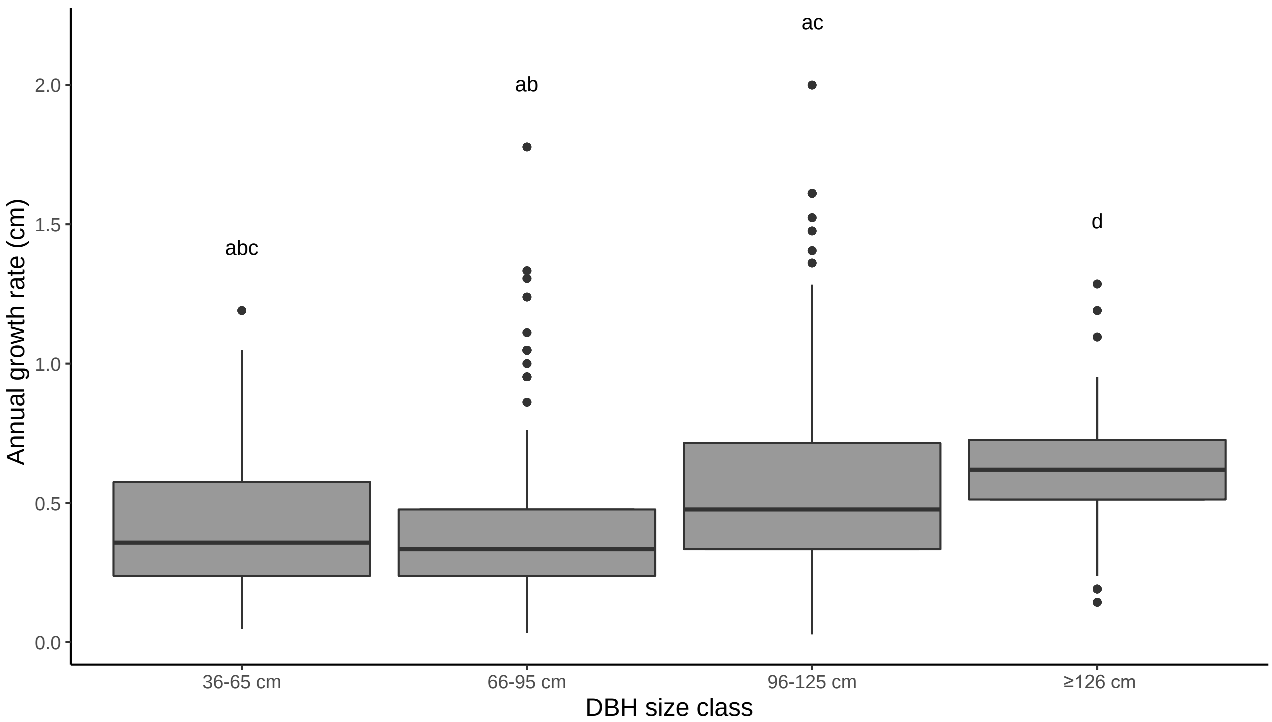

Figure 12. Annual trunk growth rate by diameter at breast height (DBH) size class of parkland trees. Significant differences amongst groups denoted by different letters (*p*<0.05), achieved using Tukey post-hoc tests based on the model formula *x* ~ *y*.

Cedrus spp. and *Platanus* x *hispanica* had significantly higher growth rates compared to the majority of other parkland species (Figure 13). The top 10% (n=40) growth rates of different species 35% were *Quercus robur,* 22.5% *Cedrus spp.*, 15% *Platanus* x *hispanica*, 7.5% *Fraxinus excelsior*, 5% *Fagus sylvatica* and *Castanea sativa* and *Tilia cordata*, 2.5% other species (*Quercus coccinea* and *Acer platanoides*). Despite growth rates differing between tree species, no significant interactions were found between species and DBH or leaf area or location in order to explain the relationship between growth rate and DBH.

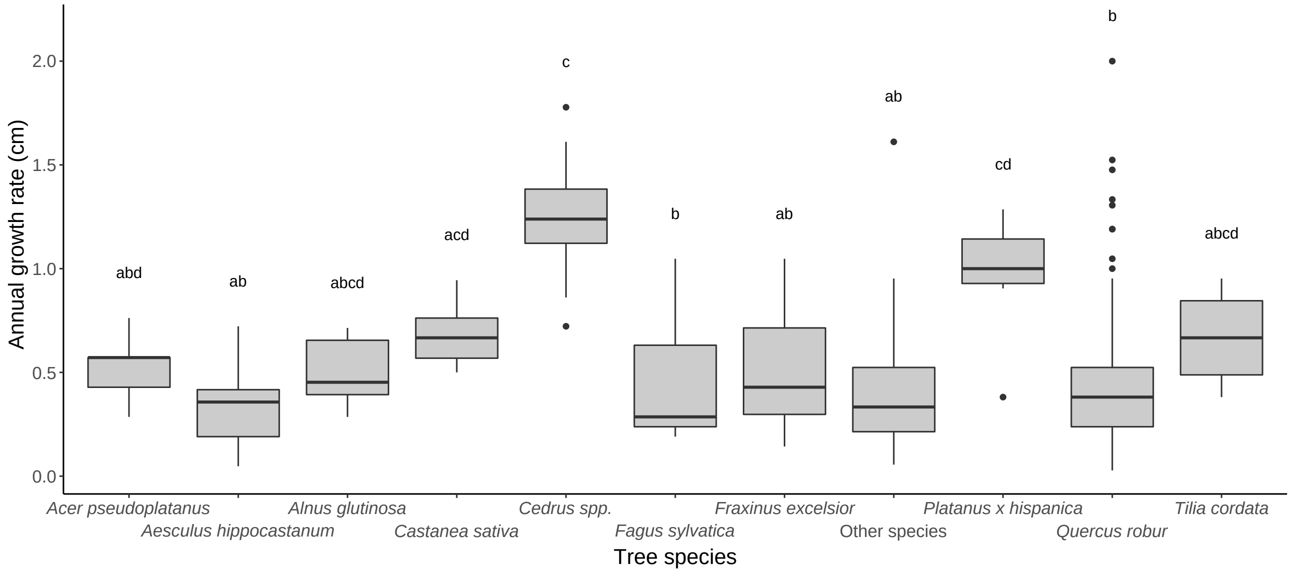

Figure 13. Annual trunk growth rate (cm) of parkland trees by species. Significant differences amongst groups denoted by different letters (p<0.05), achieved using Tukey post-hoc tests based on the model formula $x \sim y$.

4.3.3.2 Historical ES changes

Since 1999, 14 trees have been removed from the parkland: 9 trees in the Deer Park, 4 trees in the North Front, and 1 tree in the South Park. Of these trees, 57% were *Quercus robur,* 14% were *Cedrus libani* and the remaining individuals were *Aesculus hippocastanum*, *Acer platanoides*, *Fraxinus excelsior* and *Fagus sylvatica*. This is calculated at an annual mortality rate of 0.12%, or 2% in 20 years. All but two of these trees were classed as a mature age (Hutton Forestry, 1999), with an estimated 71% of the trees being classified as veteran based on having a large DBH or great chronological age for the species (Defra, 2007), and tree condition comments from the 1999 parkland tree survey (Hutton Forestry, 1999). 7 trees have died and been left as standing deadwood, which were able to be surveyed and valued for carbon storage ES benefits.

The carbon storage value of trees in the parkland has significantly increased between 1999 and 2017/2020 (z=2.33, p<0.05) (Figure 14). The gross carbon sequestration value of trees in the parkland has significantly decreased between 1999 and 2017/2020 (z=6.24, p<0.001). The avoided runoff and pollution removal values have not changed significantly between 1999 and 2017/2020.

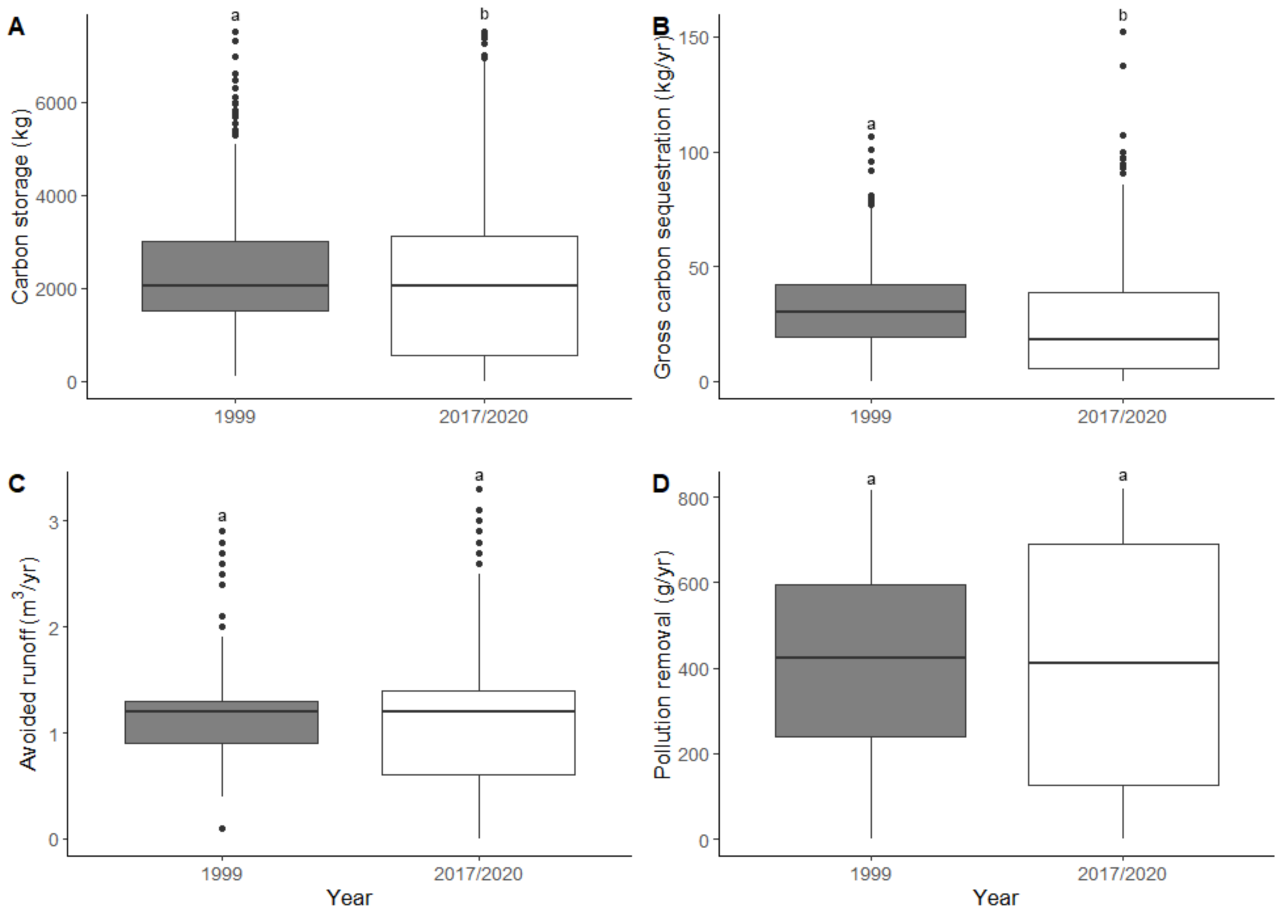

Figure 14. Changes in A) carbon storage (z=2.33, p<0.05), B) gross carbon sequestration (z=6.24, p<0.001), C) avoided stormwater runoff (z=0.05, p>0.05) and D) air pollution removal (z=-0.37, p>0.05) ecosystem service (ES) values of trees in the Harewood Estate parkland between 1999 and 2017/2020. Significant differences amongst groups denoted by different letters (p<0.05).

4.4 Discussion

4.4.1 Quantified ES benefits

Parkland trees currently provide an estimated 1,224,030 kg of carbon storage, 14,522 kg/year of gross carbon sequestration, 693 m^3/year avoided stormwater runoff and 412 kg/year air pollution removal regulating benefits for three parkland sites within the Harewood Estate. This has been valued at a total of £73,442 carbon storage value and £4,055 total annual benefits value. Parkland trees currently provide £28,117,919 in amenity value.

4.4.1.1 Parkland ES values are low compared to similar urban landscapes

Compared to similar urban landscapes, the Harewood Estate parkland provides less ES value. When compared to Hyde Park, an urban park with open-grown trees in central London, the ES values are less than half the value that Hyde Park provides per hectare (Rogers et al., 2017). The University of Leeds campus, located 10 km south of the Harewood Estate, also provides more ES values per hectare, although the gap is narrower (Gugan et al., 2019). However, this is likely due to both Hyde Park and the University of Leeds having more trees per hectare: 22.4 trees/ha and 36.3 trees/ha respectively compared to only 4.6 trees/ha for the Harewood Estate. Some difference in values could also be attributed to this study using updated, more accurate i-Tree Eco algorithms since the studies by Rogers et al. and Gugan et al. (Nowak, 2020). From a simplistic point of view, this suggests that in order to increase the ES values of the parkland estate, more trees should be planted. However this outcome does not take in to account the potential trade-offs that this management decision may have on other ES values (Mitchell et al., 2015), such as the cultural heritage value of the open vistas of the 'Capability' Brown parkland landscape (Natural England, 2016), or biodiversity (Chapter 3).

4.4.1.2 Parkland estates are more valuable per tree than other land use types

On an individual tree level, trees in Hyde Park and the University of Leeds with a DBH greater than 100 cm represent less than 1% of the population, compared to 43% of trees in the Harewood Estate. Veteran trees have been previously shown to provide more ES than younger trees, including storing larger quantities of carbon both above- (Clark and Clark, 1996; Stephenson et al., 2014) and below-ground (Dean et al., 2020), and greater cultural and aesthetic values (Blicharska and Mikusiński, 2014). Veteran trees also have the most cumulative benefits due to their relatively long lifespan (Nowak, 2004; Hand et al., 2019). It could therefore be suggested that Harewood Estate trees provide more value per tree than those in Hyde Park and the University of Leeds due to typically being larger and older, despite having less trees per hectare. Thus stately homes have an unusual size class of tree that is particularly important for ES.

4.4.1.3 Cultural ES valuation method is important

The cultural amenity ES value of the Harewood Estate parkland trees was estimated at £28 million, a higher value than their regulating ES. However, the CAVAT methodology used has limitations for estimating cultural ES values, particularly for veteran trees. The amenity value produced by CAVAT captures the aesthetic value of parkland trees, which is an important feature of a designed landscape (Kümmerling and Müller, 2012; Walerzak et al., 2015). However, this valuation is arguably a subjective judgement despite the reproducible methodology (Price, 2020). The monetary replacement value it represents is pragmatic yet may not be appropriate for rural, veteran parkland trees which have a range of cultural, historic, heritage values attached to them from both an individual tree and landscape perspective (Blicharska and Mikusiński, 2014). The monetary values produced by CAVAT also vary considerably, for example a difference of £774,846 between the most and least valuable parkland trees surveyed. Trees with veteran tree features are also penalised by CAVAT (unless the tree is formally classified as veteran), with CAVAT values being reduced with increasing age, poor vitality and wounds or damage. Due to this, the methodology has previously been assessed as having a low usefulness for the economic valuation of veteran trees (Östberg and Trädförening, 2019). Alternative methods of cultural ES valuation such as photo elicitation could be better suited for valuing the cultural ES of parkland trees (Plieninger et al., 2015), however many of these methods are on a landscape-scale as opposed to tree-scale. Cultural ES have been generally neglected in ES research (Baveye, 2017; Boerema et al., 2017) because they include intangible concepts, such as aesthetic value or a sense of place, and are considered difficult to quantify (Daniel et al., 2012; Milcu et al., 2013; Willcock et al., 2017), as well as people valuing cultural ES in different ways, and these values changing over time (Plieninger, Bieling, et al., 2015; Gould et al., 2018). The amenity value of trees is an important aspect governing trees retention and management in parkland landscapes, and should not be overlooked despite criticisms of valuation methodologies and suitability of use for veteran tree valuation.

4.4.2 Tree size is most important for high ES values

Tree size was the most important driver of ES values. Both DBH and leaf area were found to be the dominant drivers of ES values. Tree size measurements were consistently important input variables in to all ES valuation models: carbon storage, gross carbon sequestration and amenity value use DBH and avoided runoff and pollution removal use leaf area (Appendix B). Correlations between significant tree and landscape characteristics (species, veteran status, condition, parkland site, crown light exposure) and tree size further suggest tree size are the most important variables to be included in ES valuation models. However including some of the initial parameters (e.g. leaf area) used to quantify some ES variables (e.g. pollution removal) in analysis could lead to circular reasoning. The further analysis of tree and landscape characteristics was carried out as sensitivity analysis to reveal the relative strengths of these variables on ES valuation outputs. Sensitivity analysis on the i-Tree Eco model carried out by Lin et al. (2020) agrees with these findings, with DBH playing the most important role in carbon storage and sequestration estimation and leaf area index being one of the most important variables in air pollution removal. This suggests management should focus on protecting the current tree stock to reach maturity, as tree size is the most important driver of ES values.

Veteran status was an important driver of ES values, however it was only important where tree size was also important for regulating ES. This suggests tree area (e.g. woody biomass, leaf area, canopy size), and its link with tree age, was in fact the main important variable driving regulating ES values rather than veteran tree microhabitat features. Amenity value takes veteran status directly in to account, making it an important driver of amenity ES value. This link between tree area and higher ES values could be affected by tree health and therefore whether a tree is veteran or not. For example, trunk hollowing could lead to an overestimation of trunk wood biomass in i-Tree allometric equations (Zheng et al., 2016). Therefore trees exhibiting certain veteran tree characteristics could lead to inaccurate ES estimates.

The annual growth rate was on average significantly higher for the largest parkland trees than for smaller trees. This contradicts the classical idea of ‘rise-and-fall’ unimodal patterns of tree size-growth relationships: tree biomass growth declining the larger and older a tree becomes (Weiskittel et al., 2011), however studies providing clear evidence of these relationships are limited (Sheil et al., 2017). More recent studies agree with the results of this study that annual growth rate increased with increasing DBH (Stephenson et al., 2014; Sheil et al., 2017; Forrester, 2021), yet suggest this result may not be unequivocal. Measurement errors and data control procedures can bias estimates of biomass growth (Sheil et al., 2017), with an overall tree volume change important to measure as opposed to primarily changes in trunk biomass (Forrester, 2021). A lack of replicated measurements, with this study only comparing measurements between 20 years, could also lead to biased estimates. Parkland tree characteristics could also explain this result. Open-grown parkland trees could be released from tree-tree competition, allowing a larger canopy to develop and subsequently increasing growth rate (Lennon, 2009; Thorpe et al., 2010). Therefore despite many of the Harewood Estate parkland trees exhibiting decay features associated with tree death, many of these trees may have not reached an age of senescence yet, as growth rates of large trees still show decline in the last period of life (Forrester, 2021). This result alludes that the modelled carbon sequestration is inaccurate and actual parkland tree gross carbon storage should be greater than what was estimated by i-Tree Eco. The Eco model decreases the annual growth rate when a tree is reaching maturity, causing a decline in gross carbon sequestration values, however the calculated annual growth rate of trees between 1999 and 2017/2020 suggested these trees are still growing and in fact are increasing in volume more than smaller, younger trees. Consequently, large, old trees are not simply senescent carbon reservoirs but actively sequester large amounts of carbon and provide large canopy areas compared to smaller trees.

4.4.3 Both native and non-native species are important for ES values

Both native and non-native tree species were important drivers of ES values, yet tree size was still the most important driver of ES values. Tree species was a consistent input variables in to all ES valuation models, yet different species were important for different ES values, agreeing with previous research on large-stature trees (Hand et al., 2019). Sensitivity analysis on the i-Tree Eco model carried out by Lin et al. (2020) also agrees with these findings, with species being a minor driver in carbon storage and sequestration estimation compared to DBH. Differences in species morphologies such as wood density, leaf size and area, and growth habit are the most likely explanation for differences in species ES values (McPherson et al., 2016; Nowak, 2020), but this was beyond the scope of the current study to investigate.

Non-native, ornamental tree species are an important part of parkland estates and could be important in a changing future. Native and non-native species had the highest growth rates of individual trees, however only *Cedrus spp.* and *Platanus* x *hispanica* had significantly higher growth rates than the other species. Growth rate varies across tree species (Vaz Monteiro et al., 2017), so species that have higher growth rates will reach maturity faster and will be able to provide greater ES values. This also suggests that longer-lived species such as *Quercus robur* and *Platanus* x *hispanica* (Hand et al., 2019) can provide more ES values over their lifetimes compared to shorter-lived species such as *Alnus glutinosa. Platanus* x *hispanica* and *Cedrus libani* are a defining feature of 'Capability' Brown landscapes (Gregory et al., 2013). Historic parks and gardens such as the Harewood Estate parkland act as a living collection of rare and ornamental species (Prigioniero et al., 2020). Some, such as *Cedrus atlantica* and *Cedrus libani* are now under threat in their native habitat (Gardner, 2013; Thomas, 2013), making the Harewood Estate an important resource for ex-situ conservation and research (Dosmann, 2006; Westwood et al., 2020). These non-native, ornamental species are therefore important for cultural ES and conservation. The inclusion of non-native tree species could be important for the future resilience of the parkland estate. Tree pests and diseases are predicted to decrease the ES provided by trees in the future (Boyd et al., 2013). Non-native species made up 11% of the Harewood Estate parkland trees (Figure 3). Non-native tree species have fewer pests and diseases associated with them as they have been removed from their native range where specialists herbivores and diseases are present (Connor et al., 1980). Alternative, potentially non-native species, will be important to consider planting when thinking about a more resilient future parkland tree community less vulnerable to pests, diseases and future climate change scenarios (Mitchell et al., 2021). Utilising the strengths of different species and their functional differences can help increase ES values and improve resilience of parkland estates in the future.

4.4.4 ES values increased and decreased in 20 years

The mortality rate of the Harewood Estate parkland trees was 0.12% per annum. This is lower than expected when compared with other studies which have found an ancient tree natural mortality rate of between 1-2% (Drobyshev et al., 2008; Bengtsson et al., 2012). Open-grown *Quercus robur* trees have been found to have a lower mortality than those in high density forests (Drobyshev et al., 2008), and these results suggest that the majority of parkland trees have not yet reached the end of their lives despite many being classified as veteran.

Different ES increased and decreased in value over 20 years, likely due to the retention of standing dead trees, higher annual growth rate in larger trees and the planting of new sapling trees. The low tree mortalities in the parkland since 1999 have contributed to the carbon storage value of the parkland trees increasing significantly in 20 years. Continued annual growth of the mature and veteran parkland trees along with the important carbon storage value of standing dead trees left on site (Keith et al., 2009; Pfeifer et al., 2015) both contributed to this carbon storage increase, as well as the planting of new sapling trees. The gross carbon sequestration value of parkland trees has significantly decreased in 20 years. This could be attributed to more trees reaching their mature height than 20 years ago and growth rates levelling off, parameterised by i-Tree Eco model used to calculate it. However, the calculated annual growth rate in this study alludes this is incorrect, as larger trees had significantly higher annual growth rates than smaller trees.

The avoided runoff and pollution removal values have not changed significantly in 20 years, suggesting there has been a negligible change in the net leaf area of the parkland trees in 20 years. Tree mortalities will have decreased the net leaf area, however these could have been offset by the planting of new sapling trees. As trees age the canopy starts to retrench and reduce in volume

(Dujesiefken et al., 2016), which decreases the overall leaf area, yet the continued growth rate in larger trees could be offsetting any veteran tree canopy declines. Monitored ES changes in UK farmland, forestry and upland sites found similar trends over 20 years that there was generally no net loss in regulating ES, with regulating ES generally increasing (Dick et al., 2016). In order to draw specific conclusions on the temporal changes in ES, more information on the specific drivers and the landscape history is needed (Tomscha and Gergel, 2016).

4.4.5 Conclusions

Parkland trees are valuable to the Harewood Estate and this has been shown through the regulating and cultural ES benefits they provide. The low number of parkland trees per hectare generated lower ES values compared to comparable urban landscapes, but parkland estates are more valuable per tree due to the large population of veteran trees compared to other land use types. Cultural ES are important to include in ES valuations, particularly in historic parkland estates with known aesthetic and cultural heritage values, but the choice of valuation methodology is important, as monetary replacement valuation may be too simplistic to encompass the many intangible cultural ES. Tree size was the most important driver of ES values, suggesting management should focus on protecting the current tree stock to reach maturity to increase parkland ES values. The annual growth rate was significantly higher for the largest parkland trees than for smaller trees, contradicting classical ideas about tree size-growth relationships. Calculation discrepancies and tree characteristics could potentially explain this, however more recent research agrees with these findings. Along with the mortality rate of the Harewood Estate parkland trees being lower than expected, this suggests the majority of parkland trees have not reached an age of senescence yet, further confirming large, old trees are important for ES values. Both native and non-native tree species were important drivers of ES values and longer-lived species could provide more ES values over their lifetimes compared to shorter-lived species. Different ES increased and decreased in value over 20 years, likely due to the retention of standing dead trees, higher annual growth rate in larger trees and the planting of new sapling trees. Understanding the ES values parkland trees provide and what drives this can be used to inform future long-term management plans for the parkland estate.

5 The trade-offs and synergies between biodiversity, regulating ecosystem services and cultural ecosystem services in parkland trees

5.1 Introduction

One of the main challenges for decision-makers and land managers is navigating trade-offs and synergies between biodiversity and multiple ecosystem services (ES) (UK National Ecosystem Assessment, 2011; Fu et al., 2015). Trade-offs occur when the provision of one set of variables or outcomes can only be achieved at the expense of others (Rodríguez et al., 2006; Howe et al., 2014; Tomscha and Gergel, 2016), whereas synergies occur when two or more variables or outcomes change in the same direction (Bennett et al., 2009; Howe et al., 2014). In some scenarios, a trade-off may be an explicit choice; but in others, trade-offs arise without premeditation or awareness that they are taking place (Rodríguez et al., 2006). Land use decisions inevitably come with trade-offs, therefore understanding biodiversity and ES relationships is essential to support the minimisation trade-offs and the maximisation of potential synergies.

Trade-off analysis is particularly important for sites like the Harewood Estate, which are multifunctional landscapes. Historic parkland estates are known to be important for their unique biodiversity; some sites in the UK are notified as Sites of Special Scientific Interest (Natural England, 2015), as well as being important for cultural heritage; many including the Harewood Estate are Registered Parks and Gardens (Historic England, 1984). The management requirements of both outcomes may not be compatible. The study and understanding of these trade-offs between biodiversity, regulating ES values and cultural ES values on historic parkland are only just starting to be undertaken.

Specifically, this chapter sets out to answer the following research questions:

- What biodiversity, regulating ES and cultural ES variables show associations in parkland trees?
- Which types of biodiversity and ES are related negatively (trade-off) or positively (synergy) in these trees?

5.2 Methods

The biodiversity value is taken to be the habitat value of individual trees based on tree-related microhabitats (TreMs). This is based on TreM richness, TreM abundance and TreM evenness to account for TreM diversity as a whole. TreM abundance was the sum of all the TreMs recorded per tree (TreM density). Simpson's evenness ($E_{1/D}$) was used to calculate a measure of evenness that was independent of species richness (Magurran, 2013). Regulating ES were taken from i-Tree Eco outputs: carbon storage (kg), gross carbon sequestration (kg/year), avoided stormwater runoff

(m^3/year), air pollution removal (g/year). Cultural ES was taken from the tree amenity value (CAVAT) output (£).

5.2.1 Statistical analysis

The correlation coefficient method was chosen to determine associations between biodiversity and ES, due to its ease of defining no-effect relationships based on the correlation strength as opposed to other methodologies. This makes it a useful method for trade-off analysis compared to descriptive methods, regression analysis or multivariate statistics (Mouchet et al., 2014; Lee and Lautenbach, 2016).

Spearman's rank correlation was chosen to carry out calculations for all relationships due to the distribution of the data being non-normal, despite many data transformations being carried out. This included Tukey's Ladder of Powers transformation which was the closest to normality. In order to compare biodiversity and ES, all variables were standardised to z-scores (scaling the data to mean 0, standard deviation 1).

The relationships between the variables were illustrated in a principal component analysis (PCA) using the imputePCA function in the missMDA package, used to account for missing biodiversity values in the complete dataset (Josse and Husson, 2016).

5.3 Results

The correlation matrix indicated that TreM richness, TreM abundance, carbon storage, gross carbon sequestration, avoided stormwater runoff and air pollution removal were significantly positively correlated with one another, with only TreM evenness showing significant negative correlations with the other variables (Figure 15).

Trees that exhibited higher TreM richness and abundance per tree were found to be less even TreM communities. This relationship is reflected in the associations of biodiversity and ES; TreM richness and TreM abundance were found to be positively correlated with the ES whereas TreM evenness was found to be negatively correlated. This was further illustrated by PCA analysis (Figure 16). Gross carbon sequestration was found to have only a weak positive correlation with TreM richness, with TreM abundance and TreM evenness not significant.

Carbon storage, gross carbon sequestration, avoided runoff, pollution removal and amenity value were all found to be positively correlated with each other. Avoided runoff and pollution removal were found to be perfectly positively correlated, due to both ES being calculated using the same variables (Appendix B). Relationships between avoided runoff and pollution removal with other variables were therefore similar.

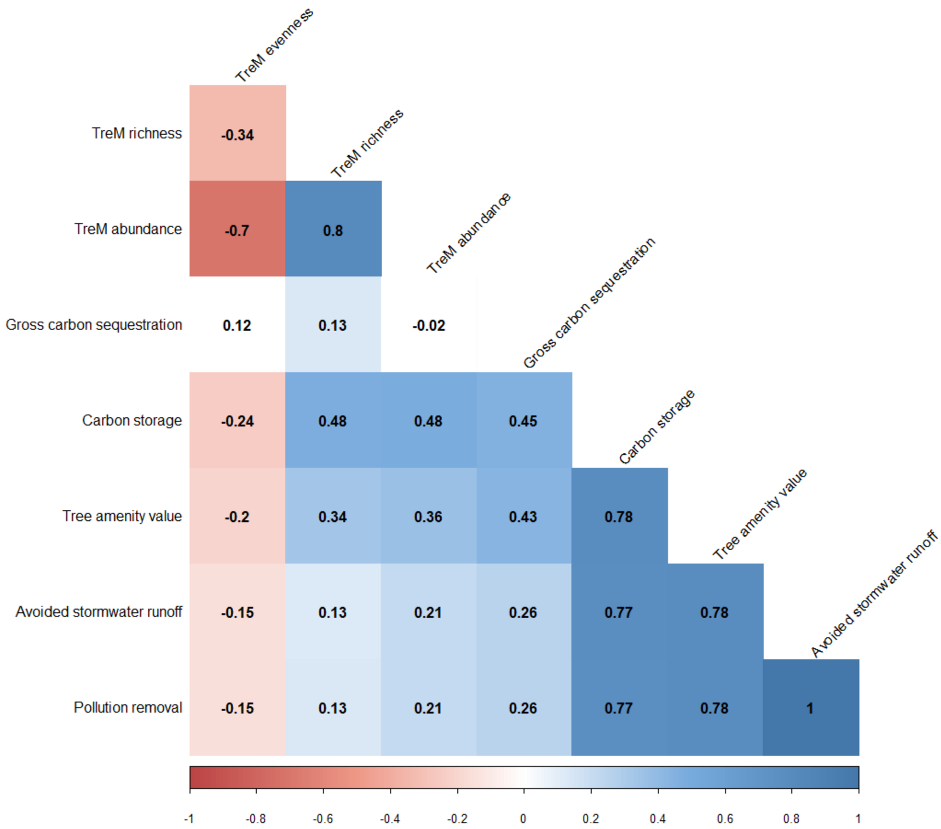

Figure 15. Pairwise Spearman's Rank correlation coefficients (r_s) between biodiversity and ecosystem service values. Blue are positive relationships (synergies) and red are negative relationships (trade-offs) (p<0.05). White indicates relationships with no established inter-relationship (p>0.05).

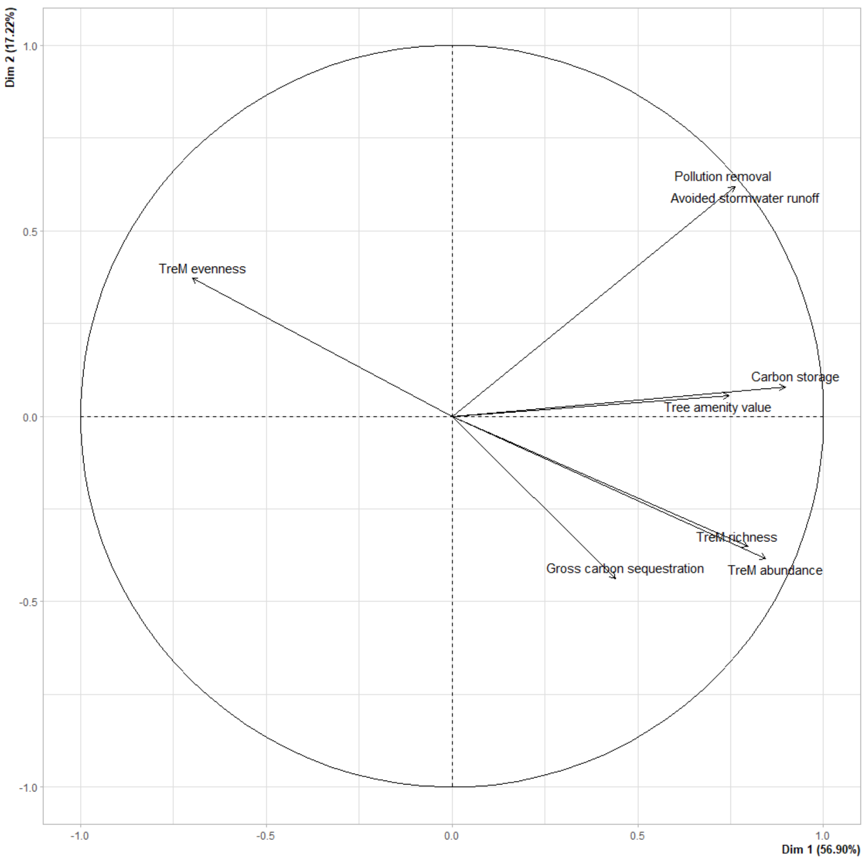

Figure 16. PCA plot highlighting the discrimination between biodiversity (TreM richness, TreM abundance, TreM evenness), regulating ecosystem services (carbon storage, carbon sequestration, avoided stormwater runoff, air pollution removal) and cultural ecosystem services (amenity value) variables. Dim 1 represents 56.90% and Dim 2 represents 17.22% of the variation.

5.4 Discussion

5.4.1 Relationships are synergistic between biodiversity and ES

TreM richness and abundance were found to have synergies with the ES analysed. These results support the hypothesis that there is generally a positive relationship between biodiversity and ES (Haines-Young and Potschin, 2010; Tilman et al., 2014), with biodiversity being shown to be synergistic with most ES in this study, and as with other studies (Balvanera et al., 2006; Cardinale et al., 2012; Harrison et al., 2014), due to the positive relationship between biodiversity and ecosystem functioning (Figure 1, Tilman et al., 2014; Dasgupta, 2021). The only trade-offs identified were between TreM evenness and the other biodiversity values, regulating and cultural ES. This is due to trees exhibiting higher TreM evenness having lower TreM richness and TreM abundance per tree. The unevenness of TreMs could be due to the majority of TreM-bearing trees in the parkland being of mature or veteran age (Figure 4) as well as the majority of species being *Quercus robur* (Figure 3). This could have contributed to many of the same TreM type occurring on the same tree, for example crown deadwood occurring at a high abundance on senescent trees due to age. Gross carbon sequestration was found to have only a weak synergy with TreM richness, with no significant association with TreM abundance and TreM evenness. This could be due to the carbon sequestration rate being similar for mature and veteran trees (Figure 10). In contrast, the biodiversity value of older trees was found to be higher (Figure 5; Figure 6; Table 4) suggesting there is a trade-off between gross carbon sequestration and biodiversity in parkland trees. However, the modelled carbon sequestration rate could be incorrect for mature and veteran parkland trees, as it was shown in Chapter 4 they had a higher annual growth rate the larger the tree was, which was the opposite relationship the i-Tree Eco model predicted to calculate the gross carbon sequestration.

This study found synergies between all regulating and cultural ES (Figure 15). This is comparable to a synthesis of ES relationships carried out by Lee and Lautenbach (2016), finding synergistic relationships between carbon sequestration, storage and bio-remediation by biota and bequest to future generations, as well as between carbon sequestration, storage and accumulation by ecosystems and scientific, educational, heritage, entertainment and aesthetic values. However, overall the majority of relationships between regulating and cultural ES are undecided or found to have no-effect (Lee and Lautenbach, 2016). Lee and Lautenbach note that the types of cultural services analysed were limited, with 69% of cultural case studies analysed focusing on ‘physical and experiential interactions’ with different environments rather than aesthetic, heritage or bequest which are more related to the cultural values the Harewood Estate parkland is known for. This suggests that cultural ES are understudied and there is not enough evidence to gain a clear picture of regulating and cultural ES relationships in the range of different scenarios they present themselves in.

This study has contributed to the evidence and knowledge gap around biodiversity and cultural ES relationships in different land management scenarios (Balvanera et al., 2006). It found a positive relationship between TreM richness, TreM abundance, and amenity value. This supports previous evidence that higher cultural ES can be associated with more biodiverse landscapes (Schneiders et al., 2012; Cáceres et al., 2015; Assandri et al., 2018; Tew et al., 2019). The knowledge gap around biodiversity and cultural ES may be due to many components of biodiversity having a cultural value, for example the appreciation of wildlife and scenic places, spiritual, religious, educational and recreational values (Mace et al., 2012) and therefore there being less need for separation when analysing ES trade-offs.

This study also found synergies between all regulating ES. Regulating services have been previously found to have strong synergistic relationships with each other (Lee and Lautenbach, 2016) due to their association with ecosystem processes and functions (Bennett et al., 2009; de Groot et al., 2010) and general positive relationship with biodiversity (Balvanera et al., 2006; Mace et al., 2012; Harrison et al., 2014).

In summary, this study has found that in the case of the Harewood Estate parkland, the majority of the biodiversity, regulating ES and cultural ES relationships are synergistic. This indicates the parkland estate is a multifunctional landscape. This may also be because none of the parkland trees are known to provide any provisioning services. Provisioning ES have previously found to be more likely to have trade-offs with biodiversity, regulating and cultural ES (Raudsepp-Hearne et al., 2010; Power, 2010; Lee and Lautenbach, 2016), explaining why in this study the majority of the relationships were found to be synergistic.

5.4.2 Parkland is still a multifunctional landscape

PCA illustrated that all but TreM evenness was explained by 56.90% of the variation between variables (Figure 16). This suggests that TreM richness, TreM abundance, carbon storage, carbon sequestration, avoided runoff, pollution removal, amenity value could be grouped together in an 'ES bundle'. This confirms that the Harewood Estate parkland is a multifunctional landscape. Multifunctional landscapes with intermediate human intervention (such as wood-pastures and parklands) have a positive effect on regulating and cultural ES, biodiversity, and are more highly valued compared to intensified systems (García-Llorente et al., 2012; Torralba et al., 2016; Assandri et al., 2018; Kay et al., 2018). Identifying these bundles has been receiving increased attention as they can give a compelling message to decision-makers when working with complex landscapes (Queiroz et al., 2015; Baró et al., 2017; Ament et al., 2017). However these associations would need to appear together repeatedly in order to meet the ES bundle definition set out by Raudsepp-Hearne et al. (2010) and Mouchet et al. (2014). It is beyond the scope of this study to confirm spatial or temporal consistencies.

5.4.3 *Tree size and age underlie synergistic relationships*

Understanding the drivers and mechanisms that underlie the relationships between biodiversity and multiple ES is fundamental for understanding the trade-offs and synergies between them to make land management decisions (Bennett et al., 2009; Landuyt et al., 2016; Dade et al., 2019). Correlation analysis alone cannot explore how various complex tree characteristics (e.g. DBH, species) or landscape factors (e.g. management practises) influence relationships between biodiversity and ES.

Comparing the tree and landscape drivers previously found for biodiversity and ES is able to address this shortfall in methodology. Chapter 3 and Chapter 4 concluded that tree size was a consistent dominant driver of increases in biodiversity and ES value, with larger, older trees providing greater biodiversity and ES values. This concurs with previous morphometric research using PCA which found the DIM1 first principal axis (Figure 16) nearly always counts for size in morphometric analysis (Berner, 2011). However, there could be collinearity between all the ES explanatory variables as they all use tree size as a variable to determine the valuation output (Appendix B). This could have introduced bias into the trade-off calculations, meaning the ES bundle is showing a relationship that was built in to the ES valuation models. Tree size is a covariate and cannot be excluded from analysis due to tree size being an intrinsic characteristic (Zuur et al., 2010). Nevertheless, the TreM results were classified into an ES bundle with ES, as well as being driven by tree size rather than the model relationship being caused by it (Table 2; Table 4; Paillet et al., 2019).

Previous studies on forest attributes concur with this study's findings that larger trees have on average a positive effect on ES (Harrison et al., 2014; Felipe-Lucia et al., 2018) and provide more ES than other younger trees (Lindenmayer and Laurance, 2017). This suggests that the potential collinearity between ES variables should not discount the results of this study, as a mechanistic understanding as well as previous research supports the conclusions of this study. However caution should be taken when making predictions using this method and specific variable combination in other parkland sites with different or unknown collinearity structures, as the underlying drivers and mechanisms may be different (Dormann et al., 2013).

5.4.4 *Conclusions*

This is one of the first studies to quantify the trade-offs and synergies between biodiversity and ES of parkland trees. Overall, there were synergistic associations between biodiversity, regulating ES and cultural ES. This study supports existing research which has broadly found synergies between biodiversity and these ES, and suggests that the Harewood Estate parkland is functioning as a successful multifunctional landscape. However, trade-off analysis results can be context dependent and be influenced by the method used to analyse trade-offs. The study found that biodiversity, regulating and cultural ES of parkland trees could be grouped in to an 'ES bundle' as they appear in a data cluster together. However to confirm this they would need to appear together repeatedly either spatially or temporally. It is likely that this ES bundle is driven by tree size and age, with previous chapters concluding that tree size was a consistent dominant driver in increases in biodiversity and ES values. This suggests however that there was collinearity between ES variables as they all use tree size as a variable to determine a valuation output. Nevertheless, the results concur with previous studies. A cautionary approach should be taken when applying this methodology to different parkland sites as ES bundles may not be comparable.

6 Discussion

6.1 Big trees are best

Throughout this investigation, it has been shown that it is the largest and oldest trees that contribute disproportionately to biodiversity and ES values. Tree size has been the dominant factor in influencing both the biodiversity value and the regulating and cultural ES values compared to other tree and landscape characteristics. This finding is supported by previous work that has shown that tree size can be the dominant influence on TreM diversity (Asbeck et al., 2021), and influence the amount of biodiversity supported by individual trees, with larger trees acting as keystone structures for increasing biodiversity (Lindenmayer et al., 2014). Larger trees were also found to have higher regulating ES values, in line with previous literature (Hand et al., 2019). This investigation found evidence that there was no decline in growth rate that would be expected in larger, older parkland trees and they had a higher growth rate than smaller, younger trees. This implies a continued increase in values as these trees age. The link between tree size and age — the age-diameter relationship — is well known (White, 1998; Lennon, 2009), and this thesis further strengthens the knowledge on veteran trees and the important biodiversity and ES values they provide. This is particularly relevant for cultural ES, as the age of trees, rather than their size, often influences their value through their longevity in human communities (Lindenmayer and Laurance, 2017; Nolan et al., 2020). Larger, older trees are rare in other land use classes (Hand et al., 2019; Asbeck et al., 2021), parkland estates are therefore important strongholds for these veteran and ancient trees. The dominance of larger, older trees in parkland tree community is therefore able to support more biodiversity and greater ES values per tree, although the relative paucity of trees in parklands could reduce these benefits on a landscape-scale.

Tree species was also an important driver of biodiversity and ES values. A more diverse tree species community is important for a higher biodiversity value of parkland trees as they provide a diverse mosaic of microhabitats which in turn supports slightly different habitat niches (Alexander et al., 2006; Sjöman et al., 2016). However species that are valuable for biodiversity may not necessarily be as valuable for regulating or cultural ES. For example, *Fagus sylvatica* had a low TreM diversity, yet had relatively high carbon storage, avoided runoff, pollution removal and amenity values compared to other parkland tree species. Utilising the strengths of different species and their functional differences can help increase biodiversity and ES values, and improve resilience of parkland estates in the future.

Tree characteristics were more important than landscape management characteristics at predicting the value of trees. The spatial layout of trees in the parkland landscape is an important part of its classification and values (Walerzak et al., 2015), yet this study found limited evidence that these were main factors driving biodiversity and ES values. However, this study did not explore the effect of environmental drivers on biodiversity and ES such as solar radiation, precipitation and wind.

Previous studies have found a link between the distinct microclimates of open-grown trees and rare invertebrates (Manning et al., 2006; Sebek et al., 2016). Felipe-Lucia et al. (2018) found that environmental factors play an important role in driving ES in combination with tree characteristics. However the study also found that tree characteristics such as mean DBH and structural heterogeneity were the most important factors explaining ES values.

This thesis has shown the valuation of parkland trees is a useful tool for the valuation of parkland landscapes. However, this study has also highlighted the complexity of these landscapes in that different tree and landscape characteristics impacted the value of parkland trees. In order to apply the methodologies used in this thesis to other parkland estates, the suitability and limitations of the valuation methods used for valuing veteran parkland trees need to be highlighted.

6.2 Parkland is a landscape mosaic

Biodiversity, regulating ES and cultural ES were overwhelmingly synergistic when trade-off analysis was carried out, suggesting that the Harewood Estate parkland is functioning as a successful multifunctional landscape. This study focused on parkland trees as a proxy for historic parkland valuation due to open-grown trees being significant to the classification of the habitat. However, the scrub and grassland understorey not surveyed in this study are also important defining features of wood-pasture and parkland and are therefore valuable. Ecological surveys undertaken in the Harewood Estate parkland in 2020-2021 concluded that the scrub cover is low and the grazed grassland is agriculturally improved species-poor condition due to the dominance of grasses (Penny Anderson Associates, 2021a; Penny Anderson Associates, 2021b). This thesis found limited landscape-scale drivers on parkland tree biodiversity and ES, however the combination of tree, scrub and grassland features in a multifunctional mosaic is what gives historic parkland an important biodiversity (Tscharntke et al., 2005; Bergmeier et al., 2010; Torralba et al., 2016), regulating (Torralba et al., 2016; Kay et al., 2018) and cultural (Plieninger et al., 2013; Fagerholm et al., 2016; Oteros-Rozas et al., 2018) value on a landscape-scale.

Veteran parkland trees have been shown to provide important microhabitats for a range of species, in particular saproxylic insects (Jonsell, 2012). Many of the associated saproxylic insects are also pollinators that require flowers to complete their lifecycle, many with specific requirements in terms of plant species used and the period in which foraging needs to take place (Falk, 2021). The pollen and nectar required by these insects is deficient in the parkland; the majority of parkland trees are *Quercus robur* which is of low value as a pollen and nectar resource (Alexander et al., 2006; Donkersley et al., 2017). This, coupled with the low level of scrub cover (e.g. Hawthorn, *Prunus* spp., *Rubus* spp.), which has been attributed to providing an important floral source for saproxylic pollinators (Falk, 2021), as well as the species-poor grassland which has a low abundance of non-grass flowering plants (Penny Anderson Associates, 2021a), suggests the Harewood Estate parkland may not be fulfilling all the habitat requirements of these rare, threatened specialist saproxylic insects. The low diversity of flowering plants available may also not provide the breadth of flowering period required (March until October). It is therefore important to consider the parkland landscape mosaic when valuing the whole parkland ecosystem.

6.3 Threats to Harewood Estate parkland continuity

Land use change is one of the main drivers of biodiversity and ecosystem service declines in the UK (Foley et al., 2005; UK National Ecosystem Assessment, 2011; Hayhow et al., 2019), therefore how land is managed is important. Threats to the Harewood Estate parkland continuity have been identified in this thesis, including: i) tree generation gap, meaning there is a gap in time where there will be very few veteran tree habitats to support veteran tree biodiversity on site; ii) the low parkland tree diversity due to the dominance of *Quercus robur*, therefore landscape continuity is more sensitive to pests, diseases and climate change; iii) distance between veteran trees is too large, potentially affecting the connectivity of specialist invertebrates; and iv) high-intensity and incorrect grazing and grassland management, leading to a lack of tree recruitment and lower species diversity in the grassland.

Wood-pasture and parkland are multifunctional landscapes of anthropogenic origin, so constant low intensity management is crucial for the maintenance of biodiversity and ES (Bugalho et al., 2011). Conservation of the Harewood Estate parkland is important; the parkland is a 'Capability' Brown Grade I Registered Park and Garden (Historic England, 1984) and a priority habitat of biodiversity conservation significance (JNCC, 2019). However, these important cultural heritage ES can either encourage the maintenance of these valuable landscapes or act as barriers to necessary innovation and development (Plieninger, Bieling, et al., 2015) — conservation should not be done for conservation's sake. Future management of parkland estates should aim to keep the landscape multifunctionality in order to maximise multiple biodiversity and ES benefits, as was concluded by trade-off analysis (Chapter 5). Parkland estates need to be able to adapt to ecosystem degradation and biodiversity loss and future threats of climate change and emerging pests and diseases.

6.4 Parkland management recommendations

The most important parkland management recommendations should initially focus on tree-level management. This thesis concluded tree characteristics – in particular tree size – affect parkland tree biodiversity and ES values more than landscape management characteristics, therefore evidence-based sustainable management recommendations should initially focus on tree-level management as the easiest and most cost effective way of increasing and maintaining parkland values.

A parkland tree management plan should be created, detailing how mature and veteran trees should be managed, with the aim of retaining them to maintain parkland continuity and reducing the gap between tree cohorts. Tree work on veteran trees could involve impeding bark damage by livestock and reducing ground compaction around roots, as well as the prevention of serious mechanical failure through selective pruning and removal of large branches (British Standards Institution, 2010; Lonsdale, 2013). Pollarding has also been shown to extend the lives of trees (Nolan et al., 2020). Retaining all forms of deadwood (crown, fallen, standing) could also be important as a way of bridging the gap between tree generations. Retaining parkland *Fraxinus excelsior* with Ash Dieback after death could be an important part of this (Bengtsson et al., 2021), as this thesis showed *Fraxinus excelsior* had the highest TreM diversity of all parkland tree species likely due to exhibiting old-growth characteristics associated with Ash Dieback disease progression. Veteranisation (deliberate damaging of trees to stimulate infection and decay) could be an ‘engineering’ solution. Examples include techniques to mimic natural damage (Bengtsson et al., 2015) and artificial nest and habitat boxes (Newton, 1994; Carlsson et al., 2016). Previous removal of deadwood from the parkland trees and pollarding act as a type of veteranisation, increasing TreMs, specifically hollows, formation rate (Sebek et al., 2013) and TreM development (Großmann et al., 2020). However this would only be worthwhile on younger trees, yet semi-mature trees are lacking in the Harewood Estate parkland (Figure 4), with all the trees surveyed having at least one TreM already present. Ultimately, parkland tree management needs to be considered on a tree-by-tree basis depending on objectives and local circumstances (Plieninger et al., 2015).

It is also important to take into account the surrounding landscape management, as parkland trees are keystone structures, acting as habitat ‘islands’ in unfavourable landscapes (Manning et al., 2006). This thesis has compared parkland trees with trees found in broadleaf European forests, and the tree management principles are analogous (Lonsdale, 2013). However, wood-pasture and parkland is a distinct habitat from intact woodland, with different understorey management. Both are recognised as different priority habitats of biodiversity conservation significance in the UK (JNCC, 2019). Therefore wood-pasture and parkland should not replace woodland to meet conservation goals, but instead enhance the parkland mosaic in estates to support a larger number of species and act as stepping stones between woodland patches. New trees and more sympathetic grassland management are key to improving connectivity. New trees could be introduced in to the parkland through either natural regeneration or tree planting. Individual large old trees can act as nodes of regeneration (Fischer et al., 2009; Mölder et al., 2019) due to contributing a disproportionately large number of germinants (Wenk and Falster, 2015). The success of natural tree regeneration has recently been shown on abandoned farmland in England (Broughton et al., 2021) and indeed during fieldwork for this thesis many seedling *Quercus robur* were observed in the parkland. However wood-pasture and parkland require active management, therefore protection of sapling trees from grazers and mowing, or a reduced intensity, would be required if they are to make it beyond seedling establishment.

The species selection and location of new tree planting is important to consider (Hand et al., 2019) and this thesis has given an insight in to this through the explanation of biodiversity and ES values by different tree and landscape characteristics. Native tree species were better for biodiversity than non-native species, with both native and non-native species important for different ES. In urban environments, it is recommended the species distribution follows the 30/20/10 rule where no family represents more than 30%, no genus more than 20% and no individual species more than 10% of the total tree population (Santamour, 1990; British Standards Institution, 2014; Trees and Design Action Group, 2021), meaning *Quercus robur* exceeds this guideline at 79% of the parkland tree population. Alternative, potentially non-native species, will be important to consider planting when thinking about a more resilient future parkland tree community less vulnerable to pests, diseases and future climate change scenarios, utilising the strengths and functional differences of different species (Mitchell et al., 2016; Mitchell et al., 2021).

The location of new plantings could be based on historic tree layouts set out by 'Capability' Brown or near current dead or dying trees. Planting around the current mature and veteran parkland trees could be beneficial. New trees can act as protection buffers to protect vulnerable veteran trees, for example from windthrow. Trees in these buffers would be recruits into the parkland tree population to eventually replace the existing veteran trees after they have died (Lindenmayer, 2017). The increased density of trees could have an impact on the microclimatic conditions important for creating certain microhabitats, however this thesis found limited effects of the spatial distribution of trees on TreM diversity, agreeing with previous studies (Asbeck, Messier, et al., 2020). However, overcrowding could lead to a reduced canopy size, which was shown to negatively affect ES values. Allowing trees to reach their maximum canopy size whilst also having neighbours could accommodate for both greater biodiversity and ES values.

Sympathetic grassland management could be key to improving landscape connectivity for parkland species, increasing grassland species diversity for saproxylic pollinators, as well as for increasing natural tree recruitment into the parkland. Livestock grazing is an important component of wood-pasture and parkland management (Plieninger, Hartel, et al., 2015). The correct grazing pressure and grazing regime is essential to ensure the parkland habitat mosaic and tree regeneration while halting the encroachment of dense scrub cover (Plieninger, 2007; Bergmeier et al., 2010). For the principal parkland regeneration phases in the New Forest, UK, maximum grazing pressure thresholds constitute 0.3 AU ha^{-1} y^{-1} (grazing animal units per hectare per year) for cattle, 0.15 AU ha^{-1} y^{-1} for ponies, and 0.45 AU ha^{-1} y^{-1} for deer (Mountford and Peterken, 2003), with similar thresholds in former pastures and arable fields in Belgium (Plieninger, Hartel, et al., 2015). Browsing intensity is greater in spring and summer on saplings, when new growth is available, than in autumn or winter (Plieninger, Hartel, et al., 2015). Therefore, tree regeneration is not prevented by grazing outside the main tree growth period. Whilst grazing of wood-pasture and parkland is most common, a combination of grazing and mowing could help promote a greater diversity of non-grass flowering species (The Wildlife Trusts, 2016). Nationally rare waxcap fungi are found in the Harewood Estate parkland grassland (Penny Anderson Associates, 2021c), therefore a mosaic of cut and uncut grassland could be necessary to maintain the waxcaps whilst increasing flowering plant diversity. Parkland grassland structural heterogeneity would allow for many habitat niches for all these management objectives.

6.5 Results in the broader context of parkland estates

The findings of this thesis can be applied to other parkland estates wishing to enhance the biodiversity and ES values of their parkland. Broadly, as with the Harewood estate, other parkland estates should initially focus on tree-level management as the easiest and most cost effective way to increase and maintain parkland values. However, with regards to the successful multifunctionality of other parkland estates, bespoke research will be needed as the specific trade-off analysis done in this thesis is likely not comparable to other estates whose biodiversity, regulating ES and cultural ES values differ. The successful multifunctionality of parkland estates by providing multiple societal benefits fulfils the role many of these landscapes once had (Bergmeier et al., 2010; Plieninger, Hartel, et al., 2015). This thesis has shown they still have an important part to play in meeting 21st century societal challenges such as ecosystem degradation, biodiversity loss, and climate change through providing important habitats and ES, and under the correct sustainable management regime parkland estates can help meet these challenges.

The dissemination of parkland management recommendations is key to helping meet these challenges. Organisations affiliated with parkland estates could be key to this, such as the Treasure Houses of England heritage consortium (Treasure Houses of England, 2021), or charities regularly engaging with parkland estates such as the Ancient Tree Forum conservation charity (Ancient Tree Forum, 2021). Consultations with land managers could help integrate recommendations into long-term parkland estate management plans.

Given this thesis represents one of the first investigations of the trade-off relationships between the important values parkland estates provide, future research should strive to ascertain whether the findings presented here are ubiquitous across parkland estates – whether biodiversity, regulating ES and cultural ES always synergistic. Future studies should also seek to encompass a greater number of tree species into analysis to explain some relationships found with different species' values, which was not possible in this investigation owing to a statistically insignificant number of some tree species. It would also be interesting to explore tree functional groups and how different functional ratios could change biodiversity and ES values.

7 Conclusion

This thesis has demonstrated that historic parklands are complex landscapes which are valuable in many different ways. Parkland trees have been shown to be valuable for biodiversity, regulating ES and cultural ES. TreM diversity was higher in parkland trees than trees found in broadleaf European forests. Parkland trees currently provide 1,224,030 kg of carbon storage, 14,522 kg/year of gross carbon sequestration, 693 m^3/year avoided stormwater runoff and 412 kg/year air pollution removal regulating ES value and £28,117,919 in cultural amenity ES value within the Harewood Estate. This thesis has also shown that the value of a parkland depends on the tree and landscape characteristics. However, the consistent dominant driver of these values is tree-level characteristics, specifically tree size and age, agreeing with the previous literature that it is the largest and oldest trees that contribute disproportionately to biodiversity and ES. Tree species also plays an important role in driving these values; however, species that provide a high value for biodiversity may not necessarily provide a high value for ES. Despite this, biodiversity, regulating ES and cultural ES were overwhelmingly synergistic when trade-off analysis was carried out, suggesting the Harewood Estate parkland is a successful multifunctional landscape. Multifunctional landscapes with intermediate human intervention (such as wood-pastures and parklands) have been shown to have a positive effect on biodiversity, regulating and cultural ES. Although, this study has shown individual parkland trees are valuable, the scrub and grassland understorey are important defining features of wood-pasture and parkland and are therefore valuable. As social-ecological systems, the type of value is dictated by the management regime preference. The main future threats identified in the continuity of the Harewood Estate parkland are the tree generation gap, low parkland tree diversity, large distance between veteran trees, and lack of tree recruitment. Future management should initially focus on maintaining and retaining parkland trees, as these trees provide greater benefits the larger and older they become, makes this the easiest, most cost effective way of increasing and maintaining parkland values. Improving the connectivity is also crucial and can be achieved through new trees and more sympathetic grassland management. Parkland estates have an important part to play in biodiversity and ecosystem recovery, and under the correct management regime they can help achieve this. Further research now needs to take place across other parkland estates to identify whether biodiversity, regulating ES and cultural ES are always synergistic.

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9 Appendices

Appendix A. The Tree-related Microhabitat (TreM) hierarchical typology taken from Kraus et al. (2016) and Asbeck et al. (2019) and summarised. An additional fallen deadwood microhabitat category was added to the TreM typology, taken from Fay and De Berker (1997), highlighted in bold.

Form	Group	TreM type	Code	Description
Cavities	Woodpecker cavities (CV1)	Small woodpecker breeding cavity	CV11	Woodpecker breeding cavity with a round entrance < 4 cm in diameter.
		Medium woodpecker breeding cavity	CV12	Woodpecker breeding cavity with a round entrance 5–6 cm in diameter.
		Large woodpecker breeding cavity	CV13	Woodpecker breeding cavity with an oval entrance > 10 cm in diameter.
		Woodpecker foraging excavation	CV14	Hollows resulting from woodpecker foraging. The excavation is conical and ≥ 10 cm diameter. The entrance is larger than the interior.
		Woodpecker "flute" (breeding cavity string)	CV15	At least three woodpecker nesting cavities on the trunk, with < 2 m distance between two neighbouring cavities.
	Trunk cavities (CV2)	Trunk base cavity ≥ 10 cm diameter (closed top, ground contact)	CV21	Trunk cavity with mould, cavity bottom has ground contact. The cavity is protected from the external microclimate and precipitation (a roof is present).
		Trunk base cavity ≥ 30 cm diameter (closed top, ground contact)	CV22	
		Trunk cavity ≥ 10 cm diameter (closed top, no ground contact)	CV23	Trunk cavity with mould, without ground contact. The cavity is protected from the external microclimate and precipitation.
		Trunk cavity ≥ 30 cm diameter (closed top, no ground contact)	CV24	
		Semi-open trunk cavity	CV25	Semi-open trunk cavity ≥ 30 cm diameter. The cavity is not completely protected from the external microclimate and precipitation can enter.
		Chimney trunk cavity	CV26	The cavity is open at the top, > 30 cm diameter, often due to stem breakage.
	Branch holes (CV3)	Rot hole cavity ≥ 5 cm diameter	CV31	Rot hole originating from branch breakage at trunk, when fungal decay of wood is progressing faster than occlusion of wound.
		Rot hole cavity ≥ 10 cm diameter	CV32	
		Hollow branch	CV33	A rot-hole ≥ 10 cm diameter located on a large broken limb. Often forming a roughly horizontal, tube-shaped shelter.
	Dendrotelms (CV4)	Water-filled hole ≥ 3 cm diameter, at trunk base	CV41	Cup-shaped hollow where precipitation can accumulate and then gradually evaporate.
		Water-filled hole ≥ 15 cm diameter, at trunk base	CV42	
		Water-filled hole ≥ 5 cm diameter, in tree crown	CV43	
		Water-filled hole ≥ 15 cm diameter, in tree crown	CV44	
	Insect galleries and bore holes (CV5)	Insect gallery	CV51	Gallery with single small bore holes.
		Bore holes	CV52	Large bore hole ≥ 2 cm diameter.
Injuries and wounds	Bark loss (IN1)	Bark loss 25-600 cm^2, minor decay	IN11	Loss of trunk bark thus sapwood is exposed.
		Bark loss > 600 cm^2, minor decay	IN12	

		Bark loss 25-600 cm^2, clear signs of decay	IN13	
		Bark loss > 600 cm^2, clear signs of decay	IN14	
	Exposed heartwood (IN2)	Trunk breakage	IN21	The trunk has broken off, exposing the heartwood ≥ 20 cm diameter. The tree is still alive and is developing a secondary crown.
		Crown breakage	IN22	Exposed heartwood ≥ 300 cm^2 through the fork insertion breakage into the trunk.
		Limb breakage	IN23	A 1st order ≥ 20 cm diameter branch has broken off.
		Splintered stem	IN24	Trunk has splintered ≥ 20 cm diameter at broken end, with several long splinters, due to high force wind.
	Cracks and scars (IN3)	Crack ≥ 30 cm long; > 1 cm wide; > 10 cm deep	IN31	Line-shaped crack through the bark and into the underlying sapwood. Not recorded if injury had occluded.
		Crack ≥ 100 cm long; > 1 cm wide; > 10 cm deep	IN32	
		Lightning scar	IN33	Bark loss and crack caused by lightning strike exposing the sapwood. Not recorded if injury had occluded.
		Fire scar	IN34	Fire scar on trunk ≥ 600 cm^2.
Bark	Bark space (BA1)	Bark shelter	BA11	Loose hanging bark > 1 cm wide; > 10 cm deep; > 10 cm length. Open at the bottom, forming a shelter.
		Bark pocket	BA12	Detached bark > 1 cm wide; > 10 cm deep; > 10 cm length. Open at the top, forming a pocket where mould can accumulate.
	Bark texture (BA2)	Coarse bark	BA21	Coarse and fissured bark, sometimes tree species specific.
Deadwood	Dead branches (DE1)	Sun exposed dead branch, 10-20 cm diameter, ≥ 50 cm length	DE11	Dead, decaying limbs in contact with living wood.
		Sun exposed dead branch, > 20 cm diameter, ≥ 50 cm length	DE12	
		Not sun exposed dead branch, 10-20 cm diameter, ≥ 50 cm length	DE13	
		Not sun exposed dead branch, > 20 cm diameter, ≥ 50 cm length	DE14	
		Dead top	DE15	Entire top of the tree has died, ≥ 10 cm diameter base.
	Fallen deadwood	**Fallen deadwood units**	**FDU**	**Detached fallen deadwood units, near the tree within its natural height scope. 1 unit = > 15 cm diameter, ≥ 100 cm length.**
Deformation / growth form	Root buttress cavities (GR1)	Small root buttress cavity ≥ 5 cm diameter	GR11	Natural hollow at the base of the tree trunk formed by the tree roots.
		Large root buttress cavity ≥ 10 cm diameter	GR12	
		Trunk cleavage	GR13	Cleft ≥ 30 cm length formed by tree growth at trunk base.
	Twig tangles (GR2)	Witches broom	GR21	Dense mass of intertwined twigs > 50 cm diameter on a branch.
		Epicormic shoots	GR22	Presence of dense mass of shoots on the base (B), trunk (T) or crown (C) sprouting from dormant buds under the bark.
	Burrs and cankers (GR3)	Burr	GR31	Presence of the proliferation of cells with rough bark but no rotten wood.
		Canker	GR32	Decayed canker with rotten wood and exposed sapwood.
Epiphytes	Fruiting bodies fungi (EP1)	Annual polypore	EP11	Fruiting bodies of annual polypores > 5 cm diameter or group of > 10 fruiting bodies, remaining for several weeks.

		Perennial polypore	EP12	Fruiting bodies (conks) of perennial bracket fungi > 10 cm diameter with a woody texture and several layers of tubes
		Pulpy agaric	EP13	Large, thick, and pulpy or fleshy, gilled sporophores > 5 cm diameter or group of > 10 fruiting bodies, generally remaining for several weeks.
		Large ascomycetes	EP14	Fungi > 5 cm diameter or group covering > 100 cm^2 of large tough hemispheric dark fungus looking like a lump of coal.
	Myxomycetes (EP2)	Slime mould	EP21	Amoeboid slime mould > 5 cm diameter which forms plasmodium, looking like gelatinous mass when fresh.
	Epiphytes (EP3)	Bryophytes	EP31	Trunk covered in > 25 % mosses and liverworts.
		Foliose and fruticose lichens	EP32	Trunk covered in > 25 % foliose or fruticose lichens
		Woody vines	EP33	Climbing plants covering > 25 % of the tree surface.
		Ferns	EP34	Epiphytic ferns > 5 fronds growing directly on the trunk or at the intersection of a branch.
		Mistletoe	EP35	Individual clumps growing in the tree crown.
Nests	Vertebrate nests (NE1)	Large vertebrate nest	NE11	Large bird nest > 80 cm diameter.
		Small vertebrate nest	NE12	Small bird or rodent nest > 10 cm diameter.
	Invertebrate nests (NE2)	Invertebrate nest	NE21	Presence of a nest containing invertebrate larvae.
Other	Exudates (OT1)	Sap flow	OT11	Fresh sap flowing along > 50 cm length of the trunk.
		Resin flow	OT12	Fresh resin flowing along > 50 cm length of the trunk.
	Microsoil (OT2)	Crown microsoil	OT21	Presence of crown microsoil formed from debris and litter originating in the canopy.
		Bark microsoil	OT22	Presence of microsoil on the trunk bark formed from moss, lichen or epiphytic alga residues and old, thick and decaying bark.

Appendix B. Description of input variables used in i-Tree Eco (carbon storage, gross carbon sequestration, avoided stormwater runoff, air pollution removal) and CAVAT (amenity value) to calculate regulating and cultural ecosystem service benefits and values per tree. A detailed description of each calculation is provided in Nowak (2020) and (Doick et al., 2018).

Ecosystem service	**Input variables**	**Calculation description**
Carbon storage	DBH, species, total height, crown base height, crown width, crown light exposure, % crown missing	Annual increase in total tree biomass (excluding leaf area for deciduous species) estimated using species-specific allometric equations and then converted to above and below-ground carbon storage estimates.
Gross carbon sequestration	DBH, species, total height, crown light exposure, % crown missing, crown health, land use	Change in carbon storage from the current year to the next based on estimated annual growth rate. This does not account for carbon lost due to tree death or decay.
Avoided stormwater runoff	Species, total height, crown base height, crown width, % crown missing, % tree cover	Leaf area is calculated from species and crown measures, and combined with local hourly weather data, using accepted methods to estimate rainfall interception.
Air pollution removal	Species, total height, crown base height, crown width, % crown missing, % tree cover	Leaf area is calculated from species and crown measures, and combined with local hourly weather data, using accepted methods to estimate air pollutants interception.
Amenity value	DBH, tree unit value factor, community tree index, public accessibility factor, crown % completeness, crown % condition, amenity value, life expectancy	A monetary value is assigned based on tree health and function and the interaction with people.

Appendix C. Taxonomic details of tree species identified in the Harewood Estate parkland as part of the study[a].

Family	Genus	Species	Variety	Authority	Common name
Betulaceae	*Alnus*	*glutinosa*		(L.) Gaertn.	Alder
Fagaceae	*Castanea*	*sativa*		Mill.	Sweet Chestnut
	Fagus	*sylvatica*		L.	Beech
	Fagus	*sylvatica*	'Purpurea'	Aiton	Copper Beech
	Quercus	*coccinea*		Münchh.	Scarlet Oak
	Quercus	*dentata*		Thunb.	Daimyo Oak
	Quercus	*petraea*		(Matt.) Liebl.	Sessile Oak
	Quercus	*robur*		L.	Pedunculate oak
Juglandaceae	*Juglans*	*regia*		L.	Walnut
Oleaceae	*Fraxinus*	*excelsior*		L.	Ash
Pinaceae	*Cedrus*	*atlantica*		(Endl.) Manetti ex Carrière	Atlas Cedar
	Cedrus	*atlantica*	'Glauca'		Blue Atlas Cedar
	Cedrus	*libani*		A.Rich.	Cedar of Lebanon
	Pinus	*sylvestris*		L.	Scots Pine
Platanaceae	*Platanus*	x *hispanica*		Mill. ex Münchh.	London Plane
Rosaceae	*Malus*	*sylvestris*		(L.) Mill.	Crab Apple
	Prunus	*avium*		(L.) L.	Wild Cherry
Salicaceae	*Salix*	*alba*		L.	White Willow
	Salix	*fragilis*		L.	Crack Willow
Sapindaceae	*Acer*	*campestre*		L.	Field Maple
	Acer	*platanoides*		L.	Norway Maple
	Acer	*pseudoplatanus*		L.	Sycamore
	Aesculus	*hippocastanum*		L.	Horse Chestnut
Malvaceae	*Tilia*	*cordata*		Mill.	Small-leaved Lime

[a] Confirmed by the Collins Tree Guide (Johnson and More, 2006) and International Plant Names Index (IPNI, 2021)

Appendix D. Microhabitat type and total number of observed tree-related microhabitats (TreMs) in the sample of 249 trees.

Group	TreM type code	Number of observations
Woodpecker cavities (CV1)	CV11	18
	CV12	7
	CV13	0
	CV14	0
	CV15	0
Trunk cavities (CV2)	CV21	90
	CV22	45
	CV23	35
	CV24	25
	CV25	5
	CV26	15
Branch holes (CV3)	CV31	116
	CV32	87
	CV33	42
Dendrotelms (CV4)	CV41	69
	CV42	9
	CV43	4
	CV44	13
Insect galleries and bore holes (CV5)	CV51	74
	CV52	1
Bark loss (IN1)	IN11	8
	IN12	1
	IN13	3
	IN14	0
Exposed heartwood (IN2)	IN21	13
	IN22	55
	IN23	154
	IN24	2
Cracks and scars (IN3)	IN31	24
	IN32	44
	IN33	5
	IN34	0
Bark space (BA1)	BA11	10
	BA12	14
Bark texture (BA2)	BA21	225
Dead branches (DE1)	DE11	362
	DE12	223
	DE13	445
	DE14	335
	DE15	130
Fallen deadwood	FDU	767
Root buttress cavities (GR1)	GR11	417
	GR12	237
	GR13	284
Twig tangles (GR2)	GR21	6
	GR22	335
Burrs and cankers (GR3)	GR31	59
	GR32	3
Fruiting bodies fungi (EP1)	EP11	17
	EP12	29
	EP13	10
	EP14	4
Myxomycetes(EP2)	EP21	0

Epiphytes (EP3)	EP31	0
	EP32	0
	EP33	0
	EP34	0
	EP35	0
Vertebrate nests (NE1)	NE11	2
	NE12	26
Invertebrate nests (NE2)	NE21	6
Exudates (OT1)	OT11	1
	OT12	0
Microsoil (OT2)	OT21	50
	OT22	43

Appendix E. Results of Kruskal-Wallis (Table A) and Spearman's Rank correlation (Table B) analysis confirming which ecosystem service (ES) values provided by parkland trees can be explained by different tree and landscape characteristics. The direction of the relationship was able to be confirmed in Table B by the Spearman's rank correlation coefficient

Table A	**Carbon storage**			**Gross carbon sequestration**			**Avoided stormwater runoff**			**Air pollution removal**			**Amenity value**		
	χ^2	**df**	***p*-value**[a]	χ^2	**df**	***p*-value**	χ^2	**df**	***p*-value**	χ^2	**df**	***p*-value**	χ^2	**df**	***p*-value**
DBH size class	489.360	4	0.000***	124.180	4	0.000***	318.840	4	0.000***	317.760	4	0.000***	311.620	4	0.000***
Veteran status	56.644	1	0.000***	26.044	1	0.000***	1.489	1	0.223	1.551	1	0.213	27.383	1	0.000***
Species	64.112	9	0.000***	56.708	9	0.000***	87.280	9	0.000***	87.512	9	0.000***	21.836	8	0.005**
Condition	35.869	6	0.000***	44.158	5	0.000***	20.994	5	0.001***	22.648	5	0.000***	92.069	5	0.000***
Parkland site	12.017	2	0.002**	31.766	2	0.000***	59.125	2	0.000***	58.657	2	0.000***	120.590	1	0.000***
Crown light exposure	78.696	5	0.000***	33.654	4	0.000***	111.27	4	0.000***	110.98	4	0.000***	105.480	4	0.000***

Table B	**Carbon storage**			**Gross carbon sequestration**			**Avoided stormwater runoff**			**Air pollution removal**			**Amenity value**		
	r_s[b]	***p*-value**[a]	**Dir.**[c]	r_s	***p*-value**	**Dir.**	r_s	***p*-value**	**Dir.**	r_s	***p*-value**	**Dir.**	r_s	***p*-value**	**Dir.**
Leaf area	0.750	0.000***	+	0.093	0.031*	+	0.997	0.000***	+	1.000	0.000***	+	0.770	0.000***	+
Distance to nearest woodland	0.027	0.522	+	0.111	0.01**	+	-0.148	0.001***	-	-0.144	0.001***	-	-0.174	0.000***	-
Distance to nearest tree	-0.042	0.314	-	0.137	0.001**	+	-0.140	0.001**	-	-0.134	0.002**	-	-0.149	0.001**	-

[a] **** p<0.001, ** p<0.01, * p<0.05, . p<0.1.*
[b] Spearman's rank correlation coefficient
[c] + denotes a positive relationship, - denotes a negative relationship