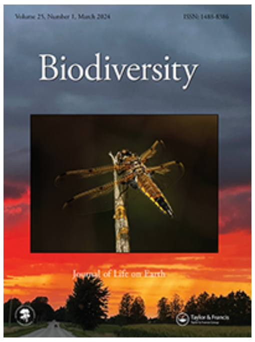

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The effects of recreational footpaths on terrestrial invertebrate communities in a UK ancient woodland: a case study from Blean Woods, Kent, UK

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ABSTRACT
Globally, terrestrial invertebrates are in decline, in part due to habitat fragmentation. Footpaths provide nature-based recreation to the public but can present small-scale spatially continuous changes in forest dynamics. However, their effects on terrestrial invertebrate communities are unknown. Pitfall trapping was undertaken to identify whether terrestrial invertebrate communities were disrupted by a popular recreational footpath in Blean Woods, an ancient UK woodland. The study identified 720 invertebrates across 36 taxa from 20 footpath edge and forest interior traps. It was found that footpaths did not significantly affect terrestrial invertebrate communities. There was no difference in the taxonomic abundance, richness, and diversity; invertebrate trait abundance and richness; or invertebrate community composition between the footpath edge and woodland interior traps. Thus, footpaths in Blean Woods do not disturb the terrestrial invertebrate community, and therefore present a sustainable mechanism for facilitating public engagement with conservation in a nationally important protected ancient woodland.

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Conservation; edge effects; fragmentation; terrestrial invertebrate; pitfall trapping

Introduction

Globally, the abundance and diversity of terrestrial invertebrates are under threat from habitat loss, degradation, and fragmentation (Cardoso et al. 2020). Rates of invertebrate extinction are currently eight times greater than those of mammals, birds and reptiles (Carrington 2019), with up to 40% of all insect species facing the threat of extinction by 2050 (Sánchez-Bayo and Wyckhuys 2019). Such declines are likely to have substantial effects beyond invertebrate communities, given the diverse and important array of ecosystem services that they provide (Table 1). For example, invertebrate communities contribute to (1) provisioning services, such as antiviral, antibacterial, and anticancer medicine (Loko et al. 2019); (2) regulating services, such as providing wildflower pollination (Allsopp, De Lange, and Veldtman 2008) and biotic waste management (Ojha, Bußler, and Schlüter 2020); (3) supporting services, such as nutrient cycling and soil formation (Cardoso et al. 2020); and (4) cultural services, such as recreational ecotourism (Huntly, Van Noort, and Hamer 2005), and even culinary traditions, architecture, and fashion (Duffus, Christie, and Morimoto 2021).

The greatest driver of disruption to insect communities is the conversion of habitat to intensive agricultural land (Sánchez-Bayo and Wyckhuys 2019). This is particularly important in the context of the UK, where >70% of land cover has been converted for agricultural purposes, with forests and woodlands accounting for just 12% of UK land cover (Kilpatrick and Rosemaund 2008). Centuries of farming, building, and industry have led to the UK being one of the most nature-depleted countries in Europe (Davis 2020), which has seen a 30–60% invertebrate population decline since the 1980s (Dirzo et al. 2014). As such, remaining woodlands and forests hold an inflated conservation importance due to the rarity of high-quality habitats. In particular, ancient woodlands provide rich and complex ecosystems, support rare and threatened species, and hold cultural historical significance (Razzaque and Lester 2021; Jones and Rotherham 2012; Smith 2018). However, ancient woodland habitat is rapidly diminishing, and now covers less than 2% of UK land area (Woodland Trust 2018). Whilst attempts to recover woodlands in the UK have led to a small increase in woodland cover over the past 100 years, much of this growth is through non-native trees and small isolated communities, which

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Table 1. Ecosystem services provided by terrestrial invertebrates, and observed UK trends to their populations.

Taxonomic group	Ecosystem services	Observed UK trends
Ants	Organic matter decomposition, pollination services, community regulation, biological indicators (Del Toro, Ribbons, and Pelini 2012).	Red wood ants near threatened at a global level, but few species have been quantitatively assessed (Balzani et al. 2022).
Beetles	Pest control, organic matter decomposition, pollination services, community regulation, population control (Woodcock et al. 2014).	Significant decline (75%) of species in the UK since 1994 (Brooks et al. 2012).
Spiders	Pest control (Cross et al. 2015).	7% decline in the UK over the past 50 years (Outhwaite et al. 2020). 43 spiders widely distributed in the UK in decline (Harvey et al. 2017).
Flies and wasps	Pest control, pollinators, decomposition, food for other invertebrates (Dunn et al. 2020).	33% decline between 1980 and 2013 (Powney et al. 2019). Since 1909, 20 bee and wasp species have gone extinct in the UK (Ollerton et al. 2014).
Woodlice	Organic matter decomposition, nutrient cycling, soil formation, essential ecosystem engineers (De Smedt et al. 2015).	Significant decline due to reduced habitat availability (Purse et al. 2012).
Springtails	Regulation of soil quality, stability of soil (Hopkin 1997; Hoeffner et al. 2021).	Too little research on the distribution of species in Britain to fully evaluate endangerment (Frampton and Hopkin 2001).

do not provide the same valuable services as ancient woodlands (Goldsmith 1988). The remaining ancient woodlands contain a disproportionately high level of biodiversity – and therefore hold a disproportionately high conservation value – for their spatial coverage.

The remaining UK ancient woodlands face increased pressures from habitat fragmentation. Edge effects are a well-studied ecological phenomenon (Ries et al. 2017; Cardoso et al. 2020; Willmer, Puettker, and Prevedello 2022; Ries and Sisk 2004), where changes to ecological communities occur due to being exposed to a new neighbouring habitat type. Common edge effects in woodlands include altered microclimates, with edges typically being windier, warmer and drier than interiors (Wright et al. 2010; Murcia 1995), causing altered species interactions (Fagan, Cantrell, and Cosner 1999; Ewers and Didham 2006), and increased exposure to species invasions (LeBrun, Plowes, and Gilbert 2012; Holway 2005). Invertebrate taxa are differently affected by such impacts due to their diversity in functional and life history traits (Cardoso et al. 2020). Indeed, a global meta-analysis of 712 anthropogenic-induced edge effects on insects identified that edge effects positively promoted the abundance of flying species and invasive species, and negatively affected soil-foraging and social insects (Cardoso et al. 2020). Thus, edge effects have an important role in structuring local insect communities. Therefore, understanding edge effects at a local scale is important for determining the risk to invertebrate communities and broader ecosystem functioning.

An understudied mechanism of fragmentation is recreational footpaths (Leung, Pickering, and Cole 2012; Gaston 2010). Footpaths represent a small-scale but spatially continuous change in forest dynamics, which could disrupt ecological communities via habitat fragmentation (Leung, Pickering, and Cole 2012; Miller, Knight, and Miller 1998). Further, the trampling of footpaths can lead to soil compaction and erosion, overground water flow and soil percolation, soil nutrient cycling, vegetation damage, and a reduction in vegetated ground cover (Littlemore and Barker 2001; Harden 1992; Nir et al. 2022; Sutherland et al. 2001). As such, local terrestrial invertebrate communities – particularly soil dwelling invertebrates – may be disturbed by recreational footpaths.

Footpaths serve as an important conservation tool; as well as conserving biodiversity, protected areas serve the dual purpose of providing nature-based recreation to the public. Tourism provides a major economic justification for large-scale conservation projects (Hall 2019), and nature-based recreation is increasingly used to support investment in conservation (Balmford et al. 2009). Nature-based tourism provides mental and physical health benefits for human users (Coventry et al. 2021) and provides income to support conservation activities (Admasu 2020; Brightsmith, Stronza, and Holle 2008). However, recreational activities can reduce the effectiveness of protected areas (Garber and Burger 1995; Taylor and Knight 2003; Reed and Merenlender 2008; Papouchis, Singer, and Sloan 2001). The disturbance of wildlife in forests has been well documented (Marzano and Dandy 2012), and there is some evidence that footpath trampling may disrupt carabid beetle communities (Lehvävirta et al. 2006; Kotze et al. 2012).

Given the global and UK negative trends in invertebrate abundance and diversity, the importance of ancient woodlands for conservation, and the importance of footpaths for providing public access to protected areas, it is key to assess whether an appropriate balance is met between providing cultural services to the public and conserving biodiversity. To our knowledge, the only research that has examined the effects of recreational footpaths on terrestrial invertebrates suggests that footpaths may have negative effects on carabid beetles in Scandinavia (Grandchamp, Niemelä, and Kotze 2000; Lehvävirta et al. 2006; Kotze et al. 2012), and may affect the abundance and species richness of fruit-feeding butterflies in tropical forests (Gueratto et al. 2020). However, the effects that footpaths have more broadly on terrestrial invertebrate communities in temperate forests is not known. Therefore, pitfall trapping in a nationally important ancient UK woodland was undertaken to identify whether recreational footpaths disrupted terrestrial invertebrate communities. Specifically, the following research questions were investigated:

(1) Do footpaths affect the taxonomic abundance, richness, and diversity of terrestrial invertebrates in an ancient UK woodland?
(2) Do footpaths affect the terrestrial invertebrate traits in an ancient UK woodland?
(3) Do footpaths affect the community composition of terrestrial invertebrates in an ancient UK woodland?

Materials and methods

Study site

Blean Woods is an ancient UK woodland, covering 1257 acres, approximately 3 km north of Canterbury, Kent. Blean Woods is the second largest area of ancient woodland in southern Britain, with over half of Blean Woods designated as a Site of Special Scientific Interest (SSSI), and one third designated as a Special Area of Conservation (SAC). The climate is a temperate oceanic climate (Cfd in the Koppen climate classification), with an annual average temperature of 11°C and annual precipitation of 730 mm. The vegetation in Blean Woods is heterogeneous, with a high number of different stands, including hornbeam (*Carpinus betulus*), hazel (*Corylus avellana*), common beech (*Fagus sylvatica*), oak (*Quercus* spp.), silver birch (*Betula pendula*), and sweet chestnut (*Castanea sativa*), owing to previous independent management regimes. Paths in Blean Woods range from wide, semi-gravelled tracks for vehicle access (~4 m wide) to narrow, infrequently used trodden footpaths (~0.4 m wide).

The study site comprised a 168 acre area of woodland located on the eastern side of Blean Woods National Nature Reserve (Figure 1), which is bordered by fields (north and east), a main road (south) and a vehicle track (west), and is typified by mixed broadleaf vegetation. The footpath examined was a semi-gravelled footpath for use by recreational walkers, with a mean width of 2.9 m.

Invertebrate sampling

Pitfall traps were used to sample terrestrial invertebrates. Pitfall traps are one of the most frequently used methods of sampling terrestrial invertebrates (Oliver and Beattie 1996; Woodcock 2005), and provide a standardized, quantitative method for comparing local invertebrate communities. A total of 20 pitfall traps were placed at either end of ten 65 m transects (Standard Deviation (SD) = 14.7 m; min = 47.7 m, max = 89.9 m; variable due to accessibility) throughout Blean Woods, from the prominent footpath into the interior woodland (Figure 1). This length of transect was used in line with recent studies, which have used a median transect length of 38 m (Willmer, Puettker, and Prevedello 2022), with most edge effects observed <90 m from the forest edge (Willmer, Puettker, and Prevedello 2022). Thus, 65 m provided a distance where any differences in invertebrate communities would likely be witnessed. Edge traps were placed in

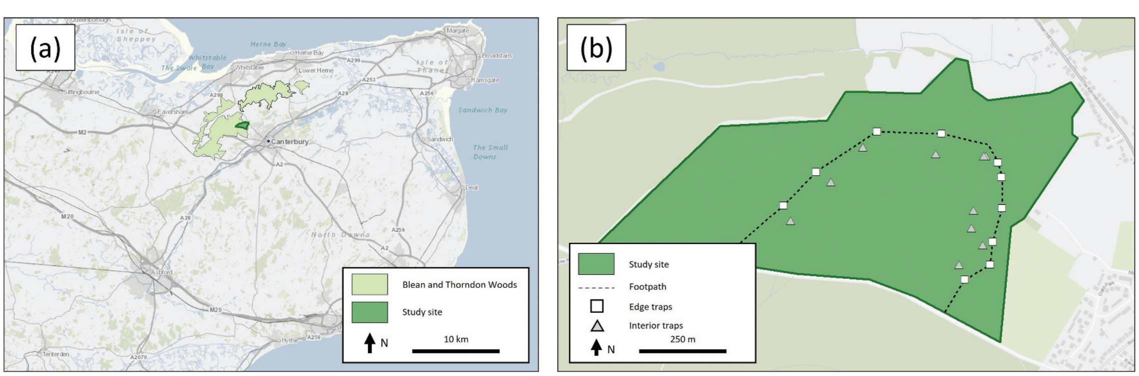

Figure 1. The location of (a) Blean and Thorndon Woods in Kent, UK, and (b) pitfall traps in the study area within Blean Woods.

close proximity (~2 m) to the footpath. Studies of environmental gradients, in particular studies investigating edge effects, risk confounding species patterns caused by the spatial gradient (edge effect) with patterns resulting from the spatial arrangement of the sampling design (Baker and Barmuta 2006). Whilst some spatial autocorrelation may be present given the locations of the pitfall traps (Figure 1), the distance between neighbouring traps (mean = 125.1 m, SD = 61.8 m) was significantly greater than the length of interior–interior transects ($t_{(17)} = 2.990$, $p = 0.004$), and so any differences observed in the dataset likely reflect differences between interior and exterior conditions (mean distance 65 m) rather than any spatial autocorrelation between neighbouring transects (mean distance 125 m).

Pitfall traps were 110 mm in diameter and 100 mm deep, and were buried so that the lip of the pitfall trap was level with the surrounding soil (Figure 2). Traps were constructed using plastic; this is the most common pitfall trap material (82% of traps in 60 studies; Brown and Matthews 2016), as it is lighter, sturdier, and cheaper than alternatives (Brown and Matthews 2016).

The traps were not baited. Traps were filled with a 3:1 solution of water and ethylene glycol with a dash of odourless dish detergent, following Schmidt et al. (2006). The solution is the most effective at preserving invertebrates, with loss of body parts three times lower compared to alternatives (Schmidt et al. 2006).

Rain guards, which have been shown not to affect sampling efficiency (Buchholz and Hannig 2009), were placed over the traps to prevent the traps filling with rain, mud, or leafy debris. Traps and rain guards were green, to camouflage the traps from public interference without interfering with sampling efficiency (Buchholz et al. 2010).

Pitfall traps were left *in situ* for five days during August 2021, commensurate with durations used by other studies (e.g. Prasifka et al. 2007; Ward-Jones et al. 2019; A. Hall, Sage, and Madden 2021; Penariol and Madi-Ravazzi 2013; Marsh 1984; Waage 1985). After five days, traps were recovered, and preserved invertebrates were taken to the laboratory for formal identification.

Taxa were morphologically identified, and classified into broad taxonomic groupings for analysis (ants, beetles; centipedes and millipedes, mites; woodlice, slugs;

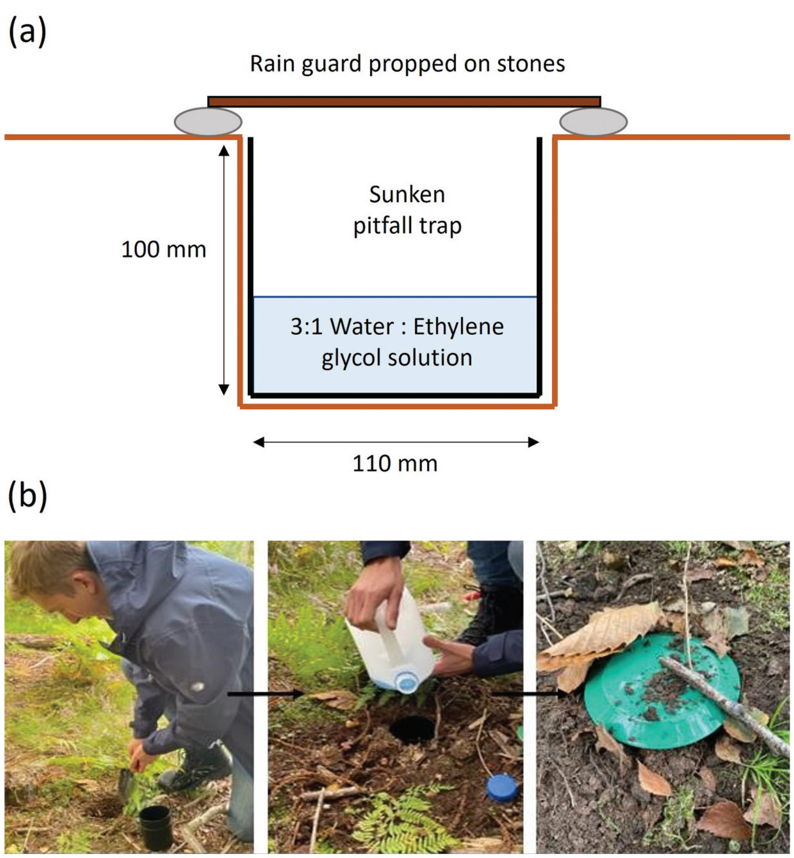

Figure 2. (a) Schematic of pitfall traps used in this study and (b) the installation of pitfall traps.

flies, spiders; springtails, wasps; true bugs; Table S1), commensurate with similar studies (e.g. Corti, Larned, and Datry 2013; De Smedt et al. 2019; Prasifka et al. 2007; Koivula et al. 2003; Hill, Roberts, and Stork 1990).

Data analysis

To analyse whether footpaths disrupted terrestrial invertebrate communities, the difference in taxonomic abundance, richness and diversity of terrestrial invertebrates caught in edge and interior pitfall traps (Q1), the difference in invertebrate traits caught in edge and interior pitfall traps (Q2), and differences in the community composition of terrestrial invertebrates caught in edge and interior pitfall traps (Q3) were examined.

Do footpaths affect the taxonomic abundance, richness, and diversity of terrestrial invertebrates?

The total number of individuals trapped from each taxonomic grouping was compared between edge and interior traps. Species data were pooled into broad taxonomic groupings for analysis (Table S1) due to the high diversity of species identified from the pitfall traps compared to the number of individuals of each species captured, to allow for meaningful comparison. The examination of density plots, Q–Q plots, and Shapiro–Wilk tests indicated that taxonomic group abundance was not normally distributed. Therefore, non-parametric Mann–Whitney U tests were used to compare the abundance of each broad taxonomic group between edge and interior traps.

Taxonomic species richness was normally distributed (Shapiro–Wilk, $p > 0.05$), and so a two-tailed independent sample t-test was used to compare the taxonomic species richness between edge and interior traps.

Invertebrate diversity was examined using Simpson's diversity metric, which was calculated (1) using species data and (2) using pooled taxonomic group data. Invertebrate diversity in the interior traps was not normally distributed (Shapiro–Wilk, $p < 0.05$), and so a Mann–Whitney U test was used to compare the diversity of species and broad taxonomic groups between edge and interior traps.

Do footpaths affect the terrestrial invertebrate traits?

Functional traits were determined for each invertebrate taxon identified (where identification resolution allowed) according to Cardoso et al. (2020), considering (1) social behaviour (eusocial or non-social); (2) invasiveness (invasive or non-invasive); and (3) foraging mode (flying or ground-dwelling; Table S1). Trait abundance (the total number of individuals possessing each trait per sample) and trait richness (the total number of taxa possessing each trait per sample) was compared between edge and interior traps for each trait. A hybrid variable of the difference in each trait score was calculated for each trap:

$$\text{Social difference score} = \text{Non} - \text{social} - \text{Eusocial} \tag{1}$$

$$\text{Invasiveness difference score} = \text{Non} - \text{invasive} - \text{Invasive} \tag{2}$$

$$\text{Foraging difference score} = \text{Ground dwelling} - \text{Flying} \tag{3}$$

Trait abundance data were not normally distributed (Shapiro–Wilk, $p < 0.05$ in all cases), and so Mann–Whitney U tests were used to compare trait abundance between edge and interior traps. Trait richness was normally distributed for eusocial versus non-social invertebrates, but was not normally distributed for invasive versus non-invasive or flying versus ground-dwelling invertebrates. Therefore, a two-tailed independent sample t-test was used to compare eusocial and non-social invertebrates from edge and interior traps, and a Mann–Whitney U test was used to compare invasive and non-invasive invertebrates, and flying and ground-based invertebrates, between edge and interior traps.

Do footpaths affect the community composition of terrestrial invertebrates?

The effect of footpaths on community composition was visualized using non-metric multidimensional scaling (NMDS) ordination plots and tested statistically using permutational multivariate analysis of variance (PERMANOVA).

All data were analysed using SPSS Statistics version 29 (IBM 2023), with the exception of NMDS plots and PERMANOVA analysis, which were undertaken in the software PAST version 4.12.

Results

A total of 720 invertebrates belonging to 36 taxa were identified across the 20 traps. The most commonly occurring taxa across the study were the ants *Lasius niger* Linneaus, 1758 (318 across 18 traps) and *L. acervorum* Fabricus, 1793 (89 across 12 traps), ground beetles *Carabidae* spp. (133 across 18 traps), the springtails *Collembola* sp. (A) (195 across 19 traps. and *Collembola* sp. (B) (70 across 16 traps), and the slug *Arion ater* Linneaus, 1758 (28 across 9 traps). All other identified taxa had a total abundance of < 20 individuals across the study.

Do footpaths affect the taxonomic abundance, richness, and diversity of terrestrial invertebrates?

Across all studied groups, there was no difference in invertebrate abundance between the edge and interior sites ($p > 0.05$ in all cases; Figure 3a). There was also no difference in the taxonomic richness ($t_{(18)} = -1.187$, $p = 0.250$) and the taxonomic diversity ($U = 76.0$, $p = 0.052$) between edge and interior sites (Figure 3).

Do footpaths affect terrestrial invertebrate traits?

There was no difference in the abundance or richness of any of the examined traits between edge and interior samples (Figure 4). There was also no difference in the abundance and richness for each trait pair between edge and interior samples (Figure 5).

Do footpaths affect the community composition of terrestrial invertebrates?

NMDS plots indicated that there was a high similarly between the edge and interior communities, with a large overlap in the 95% confidence interval ellipses, considering all taxa (Figure 6a) and pooled taxonomic groups (Figure 6b). PERMANOVA analysis confirmed that there was no difference in the centroids of the NMDS plots considering all taxa ($F = 1.114$, $p = 0.348$) and the taxonomic groups ($F = 1.063$, $p = 0.408$).

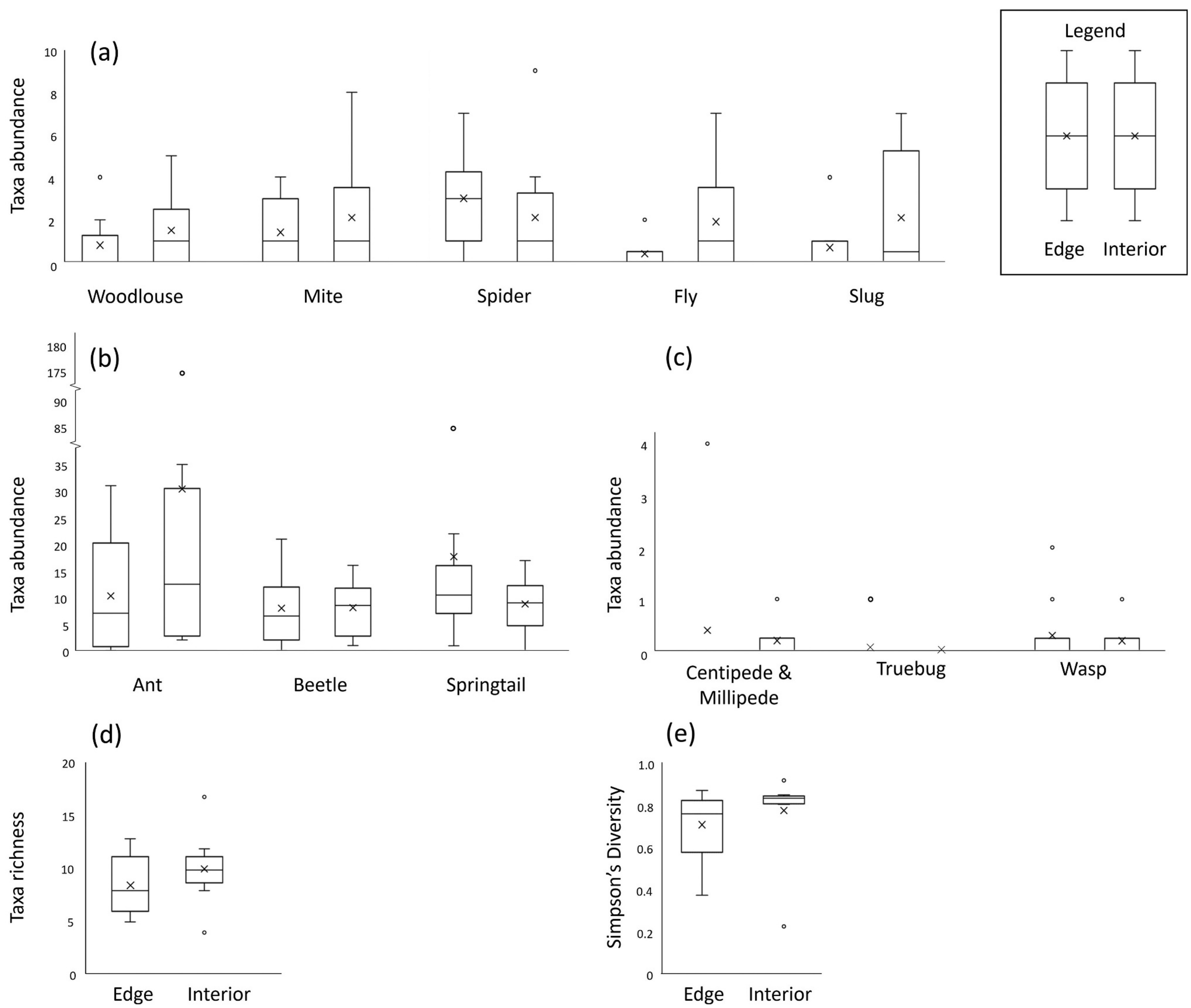

Figure 3. Edge and interior terrestrial invertebrate abundance (a, b, c), richness (d), and diversity (e) in Blean Woods. 'Edge' distributions are shown on the left, and 'interior' distributions are shown on the right for each pair. No significant difference was observed between any pairs.

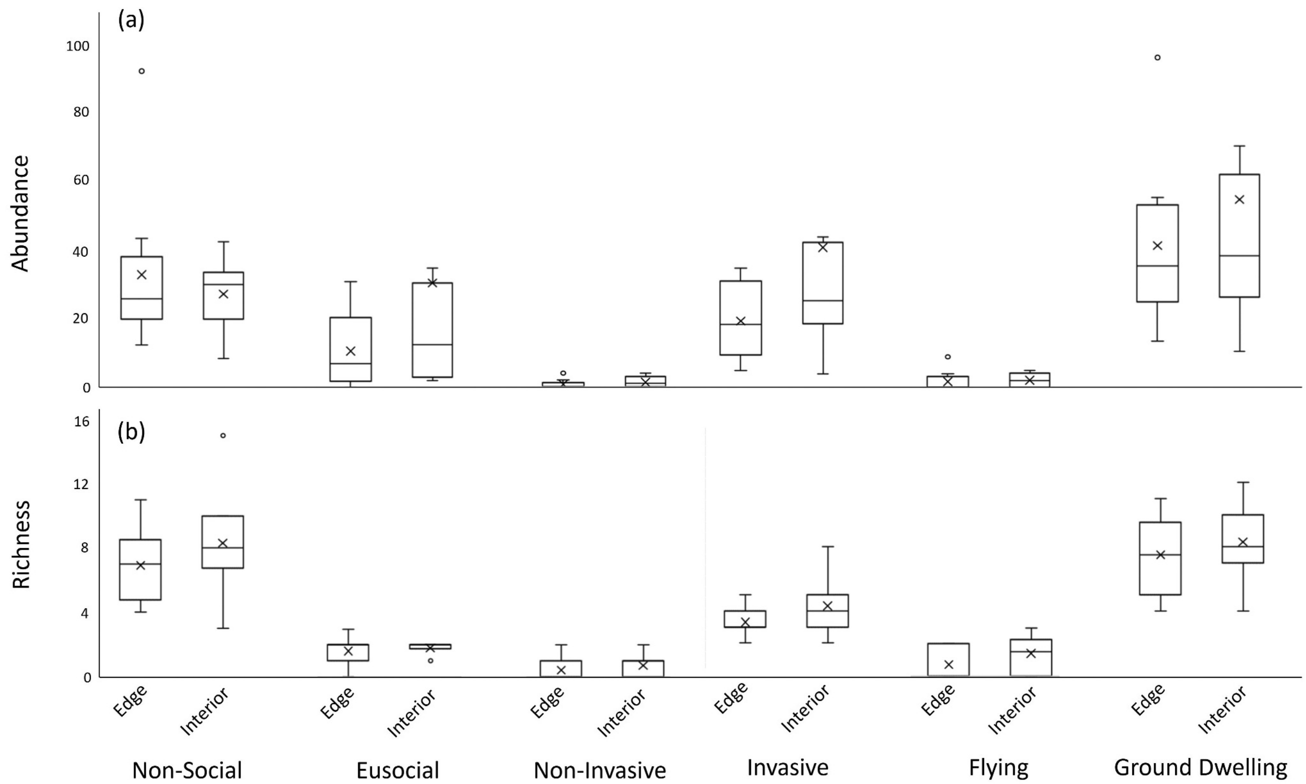

Figure 4. Edge and interior terrestrial invertebrate trait abundance (a) and richness (b). No significant difference was observed between any pairs.

Discussion

Across the study, there were no significant differences observed in terrestrial invertebrate abundance, richness, diversity, or composition between the edge and interior samples, considering both taxonomic and functional classifications. This suggests that footpaths in Blean Woods do not disrupt invertebrate communities whilst providing sustainable tourist access to the woodlands. This is contrary to global patterns of the effects of habitat edges on invertebrate biodiversity, where edge effects have differently affected different taxa (Cardoso et al. 2020), leading to alterations in community richness and composition.

The lack of differences between edge and interior invertebrates observed across the study, in contrast to previous research, may have occurred due to four key reasons. Firstly, the management history of Blean Woods may help explain these results. In a global meta-analysis of 674 studies, Willmer, Puettker, and Prevedello (2022) identified that, across plants and animals, richness decreases at edges were weaker in regions subjected to historical disturbance than in regions without historical disturbance. Blean Woods has previously experienced intensive management, including coppicing, tree thinning, and pollarding. As such, species sensitive to disturbance may have already been lost from ancient woodlands, with the remaining communities being dominated by species resilient to small-scale habitat fragmentation (Balmford 1996).

Secondly, the understanding of edge effects on invertebrates largely come from studies on individual taxa and organisms. Many studies have focussed on charismatic taxa (Gaston 2010), with uncharismatic species, and the community as a whole, frequently overlooked (Eisenhauer et al. 2019). This is true in the case of small-scale disturbances such as fragmentation by footpaths, with footpath trampling having been investigated for carabid beetles (Grandchamp, Niemelä, and Kotze 2000; Lehvävirta et al. 2006; Kotze et al. 2012) but not for the whole invertebrate community. In the case of footpath fragmentation, individual taxa may be widely recorded to decrease in density near forest edges (e.g. Cardoso et al. 2020), but community richness as a whole may be maintained via species turnover and replacement, which is commonly observed in response to disturbance (Chase 2007; Corti and Datry 2016; Tonkin et al. 2016; Hughes et al. 2007; Tockner et al. 1999). The bias towards studying individual taxa may result in meta-analyses identifying strong edge effects on terrestrial invertebrates, whereas community metrics may remain stable via species turnover, which are less well studied and thus represented in large-scale analyses.

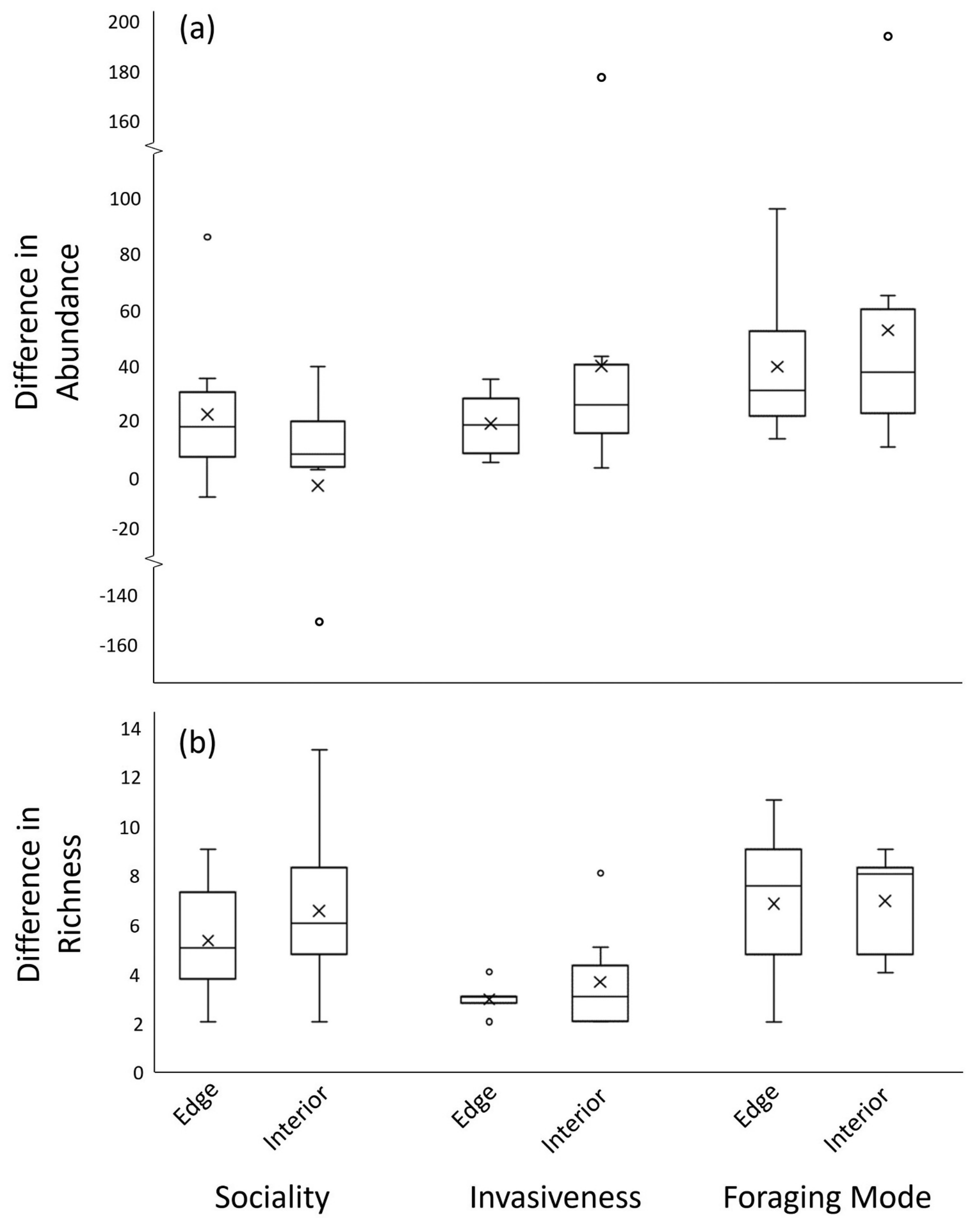

Figure 5. Difference between edge and interior terrestrial invertebrate trait abundance (a) and richness (b). No significant difference was observed between any pairs.

Thirdly, the theories which suggest that the centre of habitats is more abundant and diverse (abundant centre hypothesis) are increasingly seen as outdated (Lomolino 2001; Sagarin and Gaines 2002). Across all plants and animals, Willmer et al. (2022) observed that whilst edge effects had negative impacts on species richness in tropical forests, a greater number of species were supported near the edge of temperate forests than in interior regions. As such, the concept that forest fragmentation may be positive for biodiversity has been hotly debated (see Fahrig 2017; Fahrig et al. 2019; Fletcher et al. 2018).

Lastly, and perhaps most importantly, previous studies have typically focussed on large-scale disturbances, such as roads, pipelines, and agriculture. It may be that the scale of disruption and fragmentation caused by the footpaths in Blean Woods is not sufficient to disturb the taxa present – especially if only hardy taxa remain following continued historical management techniques (Balmford 1996). During the summer, there is continuous forest canopy above many of the footpaths of Blean Woods, and the footpaths thus may not present the magnitude of disruption required to affect the mobility of many invertebrates, particularly flying insects. Indeed, previous studies investigating disruptions to invertebrates by footpaths have also identified no effects or weak effects on carabid beetles in Scandinavia (Grandchamp, Niemelä, and Kotze 2000; Lehvävirta et al. 2006; Kotze et al. 2012), and no disruption to the

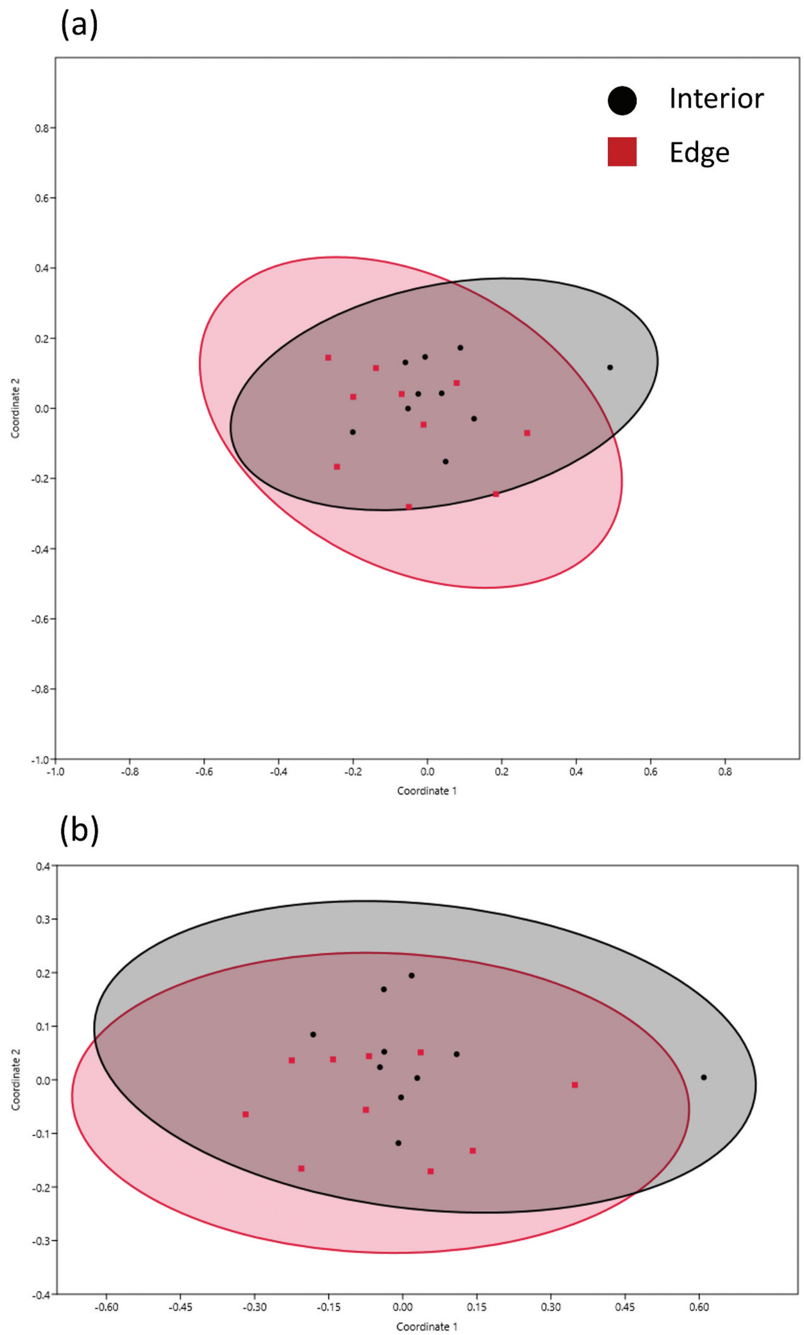

Figure 6. Non-metric multidimensional scaling (NMDS) ordination plots of interior and edge communities considering species (a) and pooled taxonomic groups (b). Ellipses show a 95% confidence interval. No significant difference was observed across either group.

vertical stratification pattern or species richness of fruit-feeding butterflies in tropical forests (Gueratto et al. 2020). Therefore, identifying what size a footpath must reach before any edge effects on invertebrate composition are witnessed will be important research to identify the maximum size that sustainable footpaths can be in temperate ancient woodlands.

This study provides an important insight into an under-researched area in identifying that small footpath trails for walking do not significantly disrupt terrestrial invertebrate communities. However, it is important to recognize that these results are site-specific. As a result of a number of historical and cultural factors, the ecology of British woodlands differs from that in other countries (Razzaque and Lester 2021). Whilst this study has observed no differences in terrestrial invertebrate abundance, richness, diversity, or composition between edge and interior sites in Blean Woods, site-specific monitoring will be important when installing and evaluating footpaths in other woodlands. This will be important, as recreational uses of woodlands is increasing rapidly worldwide, and so studying a greater number of small-scale disruptions and fragmentation is required to fully understand whether footpaths cause disturbance to invertebrate communities.

These data are also valuable beyond examining forest fragmentation. Ancient woodlands face many pressures beyond fragmentation, such as climatic impacts from climate change (Ellis 2015; Milad et al. 2011), the introduction of invasive non-native species (Jones and Rotherham 2012), the introduction of novel diseases (Rackham 2008), and ownership and management conflicts (Razzaque and Lester 2020). As such, this study provides important baseline data for continued monitoring of a threatened and important habitat.

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Disclosure statement

No potential conflict of interest was reported by the author(s).

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Catherine H. Sanders holds a PhD in river science and is currently a Lecturer in Physical Geography at the University of Lincoln. Catherine's main interests are the impacts of invasive species, rewilding, and novel restoration strategies.

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Data availability statement

Data are attached as supplementary material.

Author contributions

Samuel Kennett: Conceptualization (lead), investigation (lead), formal analysis (equal), writing – original draft preparation (lead), writing – reviewing and editing (equal). Naomi Rintoul-Hynes: Conceptualization (supporting), supervision (equal), formal analysis (equal), writing – reviewing and editing (equal). Catherine Sanders: Supervision (equal), formal analysis (equal), visualization (lead), writing – reviewing and editing (lead). This work was undertaken as part of Samuel Kennett's honours dissertation.

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