Baselining small mammal communities at a rewilding project

By

Oliver Dodd, BSc

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Abstract

As one of the most nature-depleted countries in the world, and one of the European countries with the lowest proportional forest cover, the UK is especially negatively impacted by the consequences of the global biodiversity crisis. Increasingly, rewilding is seen as a method of addressing this issue. The creation of rewilding sites across the UK will inevitably alter the flora and fauna at any given site. Such ecological changes will need to be monitored over time and compared to a baseline to evaluate the success of rewilding projects. This study aims to monitor the changes in small mammal populations across Boothby Wildland, a 617-hectare arable farm on grade 3 land in Lincolnshire that recently began taking steps towards rewilding. The site has been gradually reducing agricultural production since 2022; thus, the site contains fields which are still being actively farmed, and areas which are being subjected to passive rewilding. Small mammals are an ecologically important taxon, that are often overlooked in conservation projects. By studying their responses to spatial and temporal changes in habitat, insight can be gained into how small mammals respond during the early stages of rewilding projects. The main method of monitoring occurred via 6 x 5 grids of Longworth traps across nine months and 70 trapping nights amongst fields withdrawn from agriculture for varying periods. Hedgerows were also monitored. The traps were checked twice a day, and measurements, including species, weight, and sex were taken, before mark and release via fur clippings. Audio equipment was also placed in these trapping grids. This was to test the effectiveness of different survey methods at measuring biodiversity, and as a means of providing a supplementary form of monitoring. The greatest small mammal diversity was found in the hedgerows, confirming their importance as a habitat feature. The majority of catches were dominated by wood mice (Apodemus sylvaticus), which made up 509 of the 524 total catches (97.14%), with the remaining 15 catches consisting of 8 field voles , 5 common shrews, 1 pygmy shrew, and 1 harvest mouse. Small mammal populations appeared to reflect typical annual cycles, i.e. greater abundance in the autumn and considerably fewer individuals in the spring. Audio data corroborated the Longworth trapping with regards to November being the greatest month for activity across all species. The fewest small mammals were found, in both Longworth traps and Audio

recordings in bare fields, demonstrating a shift in small mammal communities as agricultural fields are taken out of production and vegetation is altered in rewilding projects. These data can be used to inform new rewilding projects of the way small mammals may respond to initial management.

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Table of Contents

List of figures

List of tables

1.

Introduction

1.1 Biodiversity status

Globally, there are many threats facing biodiversity, the greatest of which is habitat loss (Hanski, 2011). Recent estimates suggest up to 75% of the Earth's land area has been modified by human activity such as agriculture and urbanisation (Schulte to Bühne et al., 2021). It is widely accepted that habitat loss is the main factor to which the reduction in global biodiversity can be attributed (Schulte to Bühne et al., 2021). Moreover, the functionality of ecosystems is often inextricably linked to their biodiversity (Isbell et al., 2014). Species loss can significantly alter ecosystem services (Banks-Leite et al., 2020). As one of the most nature-depleted countries in the world (worldlandtrust, 2023), and one of the European countries with the lowest proportional forest cover (Pettorelli, Durant and du Toit, 2019), the UK is particularly negatively impacted by habitat loss and the consequences of the global biodiversity crisis. The overall trend shows continued declines in biodiversity and a series of local extinctions. During the 20th century, over 100 species have been lost in the UK (Laycock et al., 2009). Moreover, many of the remaining species have experienced significant declines, including native birds, plants, and bumblebees (Thomas et al., 2004). As well as limited success in halting biodiversity loss, traditional conservation can be costly and labour-intensive (Cardador et al., 2015). If active management is too costly, it may compromise the long-term viability of a project, putting protected species at risk long-term (Cardador et al., 2015).

1.2. Traditional conservation

Traditional conservation management has historically aimed to address the UK biodiversity crisis by targeting specific species or habitats (Volis, 2019). There have been success stories in this field, particularly when it comes to conserving high profile species such as the bittern (*Botaurus stellaris*; Brown, Gilbert, and Wotton., 2012). With regards to the bittern, emergency action was taken via the recovery and creation of reedbeds which led to a significant population increase in the UK. Similarly, the cirl bunting (*Embereza cirlus*) underwent huge decreases in range and abundance in the 20th century, which was remedied by strategic grassland management in its home range (Kleijn and Sutherland, 2003). This led to an exponential increase in numbers as a result of this intervention. Traditional conservation success stories also occurred with black grouse (Lyrurus tetrix), and corncrakes (*Crex crex*) (Kleijn and Sutherland, 2003).

The modification of farming practices to try and address historic lowland agricultural biodiversity loss in the UK have also had some successes, particularly when it comes to increasing field margin size. Increasing the width of margins has benefited numbers of certain birds species such as yellowhammers (*Emberiza citronella*), and whitethroats (*sylvia communis*) (Shore et al., 2005). Creation of 6m margins as part of the UK Environmental Stewardship scheme may also benefit certain small mammals such as bank voles and common shrews (Shore et al., 2005).

However, despite specific examples of certain high profile target species thriving under intensive conservation management in the UK, biodiversity continues to decline (Burns et al., 2016). It has been suggested that one possible way to address these declines could be to alter the approach to conservation in the UK.

1.3. Rewilding

Rewilding has been suggested by many as a means of reducing biodiversity loss in the UK (Pettorelli, Durant and du Toit, 2019). Rewilding comprises multiple concepts, including species reintroduction, land abandonment, and taxon replacement (Carver et al., 2021). Its definition has been the source of much deliberation (Schulte to Bühne et al., 2021), rewilding can be defined as “the reorganisation of biota and ecosystem processes to set an identified social–ecological system on a preferred trajectory, leading to the self-sustaining provision of ecosystem services with minimal ongoing management” (Pettorelli, Durant and du Toit, 2019). Soulé and Noss (1998), however, focus on the idea that large predators require large areas of land, and reserves should feature corridors to promote connectivity. Other definitions have a more anthropocentric focus, with Monbiot (2015) defining rewilding as the mass restoration of ecosystems, but also including the caveat that this restoration has implications for the ecosystem, and the lives of the humans involved. Debate has also ensued over the extent to which rewilding refers to returning ecosystems to their previous historical states or focusing on restoring functionality in a modified state (Jørgensen, 2015).

The reduction in anthropogenic control over the landscape is what separates rewilding from other forms of conservation (Corlett, 2016). Rather than addressing the biological requirements of certain target species or habitats, rewilding aims to allow populations and ecological dynamics to reestablish and self-sustain (Pereira and Navarro, 2015). This self-regulation is often achieved via the translocation of keystone species, i.e. trophic rewilding (Schweiger et al., 2018). One key ecosystem process that can be aided by the translocation of keystone species is grazing. Many rewilding projects replace labour-intensive mowing with grazing by large herbivores (Schou et al., 2020, Lorimer et al., 2015, Schweiger et al., 2018).

In the UK, rewilding has been described as still being in its early stages (Jones and Comfort, 2020). However, in recent years, it has received an increase in uptake (Sandom et al. 2018) and many are seeing its value as a strategy for environmental rejuvenation (Schulte to Bühne et al., 2021). The potential of rewilding to boost biodiversity rapidly at scale, its intuitive appeal to the public and policymakers, and its relatively low cost compared with conventional intensive conservation management, have all led to this approach to land management gaining great popularity in recent years (Perino et al., 2019). Many proponents speak highly of the potential of rewilding to benefit ecosystems and in particular, its ability to enhance trophic complexity, whilst simultaneously creating benefits for societies (Perino et al., 2019). There is reason to suggest that rewilding is a viable option for replenishing biodiversity in the UK (Perino et al., 2019).

Despite rewilding's potential for successful reform within the conservation sector, many criticise this approach for its limitations. Concern has been raised that when rewilding involves species translocations, this could cause negative consequences such as the exacerbation of human-wildlife conflict (Schulte to Bühne et al., 2021), the proliferation of pests in native ecosystems (Nogués-Bravo et al., 2016), and increased risk of disease introduction (Lorimer et al., 2015). Agricultural abandonment, which often constitutes the first step in rewilding, may sometimes be unfavourable to animals that require habitats with an assortment of vegetation densities (Benayas et al., 2007, Gorman, 2018). More broadly, many studies of agricultural abandonment have reported biodiversity loss (Benayas et al., 2007). Also, agricultural land that may be required for rewilding has a long history of anthropogenic modification which can lead to local peoples being very attached to the aesthetic of their landscape. This means they can be resistant to any form of landscape change that rewilding would inevitably bring (Carver, 2007). This can lead to additional scrutiny, especially if people believe they are being excluded from the process of rewilding (Perino et al., 2019). Alongside these challenges, the UK faces the additional difficulties associated with being densely populated (Williams, 2009), making it difficult to establish large areas of land for rewilding and especially difficult to connect them (Lawton et al., 2010).

Another frequently cited criticism of rewilding is the lack of a clear and consistent definition (Sandom et al., 2018, Schulte to Bühne et al., 2021, Perino et al., 2019). The ambiguity when it comes to defining the process can create many policy consequences. Importantly, it can alter peoples' perceptions of the goals of rewilding and if improperly defined, may lead to local opposition (Schulte to Bühne et al., 2021). Finally, further problems stem from the suggestion that there is a lack of evidence supporting the idea that the process of rewilding creates positive ecological outcomes (Jones and Comfort, 2020). This criticism can be further perpetuated by the fact that rewilding is an experimental approach, and thus, many of its long-term benefits remain unproven (Lorimer et al., 2015). We have insufficient data regarding the ecological consequences of large-scale habitat creation and restoration (Fuentes-Montemayor et al., 2019). Long-term monitoring is necessary to address this ecological data deficit and determine the outcomes of rewilding (Hawkins et al., 2022).

1.4. Monitoring

Monitoring plays an important role in conservation management decisions (Hawkins et al., 2022). It is vital to identify the successes and failures of any conservation project and adapt appropriately (Sanders et al., 2019), and increasingly, many new studies focus on the importance of long-term ecological monitoring (Mata, Buitenwerf, and Svenning., 2021, Schulte to Bühne et al., 2021, Carpio et al., 2025).

However, historically, there has been insufficient use of longstanding ecological records in biodiversity conservation in general (Willis et al., 2007). This lack of sufficient data also applies to the majority of current rewilding projects (Schweiger et al., 2018). Many challenges have been identified when it comes to long-term ecological monitoring. For example, progress can be difficult to measure when it is slow or non-linear, (Sanders et al., 2019) and further issues can also arise when reasons for success are not immediately apparent (Sanders et al., 2019).

Another problem with monitoring in rewilding is there is a tendency for taxonomic bias. Much of the current and historic sampling effort focuses on large and charismatic fauna, leaving less visible taxa underrepresented when it comes to sampling (Contos et al., 2021). Examination of IUCN data shows two orders of mammals (artiodactyla, and carnivora) are particularly overrepresented in reintroduction projects (Seddon, Soorae, and Launay, 2005). This largely stems from their frequent classification as umbrella species, and their tendency to invoke public support for conservation projects. However, although successes in the form of improved habitat suitability have been achieved by conserving umbrella species, there are still risks associated with this approach. Not all species are equally influenced by ecological alterations to the landscape, and many species that share a common habitat with umbrella species, have seen population declines, despite simultaneous successes for umbrella species populations (Wang et al., 2021). Thus, whilst large mammals are an important biological group, their effectiveness as umbrella species may sometimes lead to neglect of neighbouring species.

Small mammals are important components of UK ecosystems, and certain species, such as the hazel dormouse, have been the subject of significant public attention (Morris, 2003). However, many more species of small mammals are often neglected with regards to conservation, in favour of larger, more charismatic species (Troudet et al., 2017, Mills, Gordon, and Letnic, 2017). Very little is known about how small mammals respond to reforestation and woodland rejuvenation (Fuentes-Montemayor et al., 2019), which can often occur at the later successional stage of rewilding projects occurring on former agricultural land (Piché and Kelting, 2015). Therefore, the monitoring of small mammals is also necessary when examining the outcomes of rewilding projects. Nevertheless, certain monitoring methods are still relatively early in their development, and field studies employing multiple sampling techniques are required to validate the results obtained from remote audio monitoring.

1.5. Small mammals

Small mammals play important roles within many ecosystems and are likely to form a key part of the rewilding process in UK projects aiming to restore habitats in previously agricultural areas. Of the ecosystem services that rewilding projects aim to promote, small mammals are particularly important with regards to seed dispersal (Gorman, 2018). Their roles as seed dispersers makes small mammals an active part of the process of vegetative succession, and useful biological indicators for monitoring this process (Gorman, 2018). As well as dispersing seeds, small mammals aid in the dispersal of fungal spores (Vašutová et al., 2019). They have also been found to increase plant species diversity (Gorman 2018), and are valuable prey species for several key predators, including the barn owl (*Tyto alba*), kestrels (*Falco tinnunculus*), weasels (*Mustela nivalis*), and stoats (*Mustela erminea*) (Moore, Askew, and Bishop, 2003, Sullivan, Sullivan, and Thistlewood, 2012).

Although many small mammals, such as the field vole (*Microtus agrestis*) and wood mouse (*Apodemus sylvaticus*), are common and widespread in the UK (Middleton, Newson, and Pearce, 2023), other species are in decline, and even those that are not declining may still be vulnerable to anthropogenic pressures (Battersby, 2005). The hazel dormouse (*Muscardinus avellanarius*) has lost around half its range in the UK in the last 100 years due to habitat fragmentation, habitat degradation, and loss of specialised habitat (Bright and Morris, 1996). Harvest mice have specific habitat requirements and may be vulnerable to habitat fragmentation and degradation as a result of agricultural intensification (Smith, 2021).

The water vole (*Arvicola terrestris*) has also undergone population decline this century, mainly due to predation by the invasive American mink (*Neovison vison*) (Gaskin, 2017). The yellow-necked mouse (*Apodemus flavicollis*) has a patchy distribution in the UK, and its dependence on ancient woodland means it may be susceptible to habitat loss and fragmentation (Battersby, 2005). The data on shrews in the UK are limited, and their current status is difficult to assess as there is limited research in recent literature (Mathews et al., 2018). It is however thought that both common shrews (*Sorex Araneus*) and pygmy shrews (*Sorex minutus*) may have been in decline in the UK due to agricultural intensification (Macdonald et al., 2007), and habitat loss, specifically that of ancient grassland and meadows. Although, this decline may have been counteracted by the introduction of new policies such as agricultural set-aside (Brockless and Tapper, 1993)

The response of shrews to agricultural set-aside is an example of how small mammals may respond to conservation measures in agricultural landscapes. Studies of agricultural habitat restoration have determined that small mammals respond positively to increased habitat heterogeneity at the landscape level, resulting in an ecosystem with a mixture of vegetative successional stages (Moro and Gadal, 2006). Also, newly created woodland sites can be rapidly colonised by small mammals, regardless of whether these species are woodland generalists or specialists. The colonisation of new woodland by small mammals will likely create a trophic cascade which positively benefits the ecological community as a whole (Fuentes-Montemayor et al., 2019). More research, and especially long-term monitoring, is required to determine how small mammal species respond to conservation measures, including rewilding.

1.6. Small mammal monitoring

Historically, the main method of small mammal monitoring in the UK has been live trapping, with the Longworth trap being the most widely used trap type (Flowerdew et al., 2004). The main benefits of the Longworth trap are its portability and reliability (Flowerdew et al., 2004), and it has long been used to gather ecological data on small mammal communities such as species richness and composition (Torre, Arrizabalaga, and Flaquer, 2004). However, conventional live trapping of small mammals is not without its limitations. Many methods of live trapping can be time and labour-intensive (Yang et al., 2022). The chance of trapping each species may also be disproportional for many reasons. For example, small mammals often actively defend their territories, giving rarer species limited opportunity to encounter live traps (Yang et al., 2022). It has also been questioned whether live trapping reflects the true behavioural dynamics of small mammals (Montgomery, 1989). There is often a discrepancy between movement patterns of small mammals determined by live trapping and radio tracking, and live trapping may not accurately represent an individual's use of space (Desy, Batzli, and Liu, 1989). Nest searching can be used as a method to confirm the presence of harvest mice, however, harvest mice may be found from live trapping in sites where no nests are located (Kettel, Perrow, and Reader, 2016). Comparison studies between owl pellet analysis and live trapping have found live trapping to underestimate the small mammal community assemblages, with owl pellet analysis detecting a significantly higher proportion of a small mammal community (Torre, Arrizabalaga, and Flaquer, 2004). The ideal spacing between traps in a trapping grid also differs between species and habitats, making trapping bias inevitable (Jensen and Honess, 1995).

Newer technologies employed in small mammal monitoring may compensate for the tendencies of Longworth traps to underrepresent small mammal communities. Camera traps may improve efficiency of monitoring and are less invasive than live trapping methods (Glen et al., 2013). They can also reduce observer bias in field studies as images can be recorded and verified by multiple observers (Caravaggi et al., 2017). Likewise, audio monitoring is useful when it comes to recording soundscapes of multiple species, particularly those that are more cryptic and may therefore be missed by conventional trapping methods (Hill et al., 2018).

The collection of acoustic data as a means of monitoring wildlife has increased massively in recent years, mainly due to technological advances making it more affordable (Jarrett et al., 2025). As well as being cost effective, methods of monitoring using audio equipment can reduce survey effort (Teixeira, Maron, and Rensburg, 2019) and enabling monitoring over greater spatial and temporal scales (Jarrett et al., 2025). Also, unlike live trapping, the placement of audio equipment causes no known stress on local fauna (Vilalta, 2024). Passive acoustic monitoring is largely used to study calls in certain taxonomic groups, research in this area tends to focus on bats, birds, and cetaceans (Penar, Magiera, and Klocek, 2020, Gibb et al., 2018, Sugai et al., 2019). However, its use as a method of monitoring is being increasingly used to study a broader range of taxa, including small mammals (Vilalta, 2024).

Machine learning can be trained using recordings of known species and used to characterise calls based on frequency and call patterns. Increasingly, automated bioacoustics is being used as a tool for environmental monitoring (Mutanu et al., 2022). Where small mammals are concerned this monitoring often targets ultrasonic calls. Ultrasonic calls have a variety of uses, including in courtship and interspecific aggression (Newson, Middleton, and Pearce, 2020). The ultrasonic calls of many small mammals represent a vital component of their communication, and allows identification and the collection of population data, with minimal disturbance to the target species (Middleton, Newson, and Pearce, 2023). The utilisation of newer monitoring technologies alongside traditional methods may be an important step in broadening the availability of data in rewilding projects.

1.7. Study questions

This project aims to provide baseline monitoring data at a new rewilding project called Boothby Wildland. The project focused on mammals, in particular, small mammals. As this is a new rewilding project, there are very few data available about the small mammal populations on site. These data presented here can therefore be used to track future progress and to determine if the site-specific conservation goals are being met. The project had several distinct objectives. Firstly, to establish the nature of the small mammal communities present at Boothby (i.e. species abundance and richness). Secondly, to establish how these communities differ seasonally. Thirdly, by comparing habitats with different management histories at Boothby, to determine whether and how the small mammal community has responded in the early phase of the project to the withdrawal of arable farming. Finally, to establish the differences in biodiversity metrics describing the mammal community obtained from traditional (live trapping) and newer (audio recording) monitoring methods.

The main motivation behind the use of audio recorders was to survey a wider range of taxa than can be expected from Longworth trapping alone. A further aim was to compare the effectiveness of Audiomoths and Song Meter Mini Bat 2 devices with regards to their effectiveness at bioacoustics recording. These two devices differ in their technical specifications, hardware, and cost per unit (Audiomoths cost approximately £60 per unit, Song Meter Mini Bat 2 cost approximately £563 per unit) (OpenAcousticDevices, WildlifeAcoustics). The Mini Bat is specifically designed with high frequency calls in mind and hence might be anticipated to be more effective at monitoring ultrasonic calls from small mammals. It is therefore expected that by using both types of audio recorders simultaneously, a difference in their ability to monitor small mammal calls may be revealed.

The aim of the camera traps was to assess the large mammal presence at Boothby. The small mammal population was being monitored by the use of Longworth traps, as well as audio traps, but this gives no indication of the large mammal abundance and distribution throughout the site. The camera traps provide an idea of the large mammal populations during the early stages of the rewilding process at Boothby. If continuous monitoring is carried out, then the temporal change in large mammal distribution and abundance at Boothby can be measured.

2.

Study site and methodology

2.1. Study site

The study was carried out at Boothby Wildland, a 617-hectare arable farm on grade 3 land (Nattergal.co.uk, 2024) in Lincolnshire (52.86 °N, -0.54°E), purchased by Nattergal in 2021 (figure 1). Nattergal is a company that buys, leases, or manages large areas of ecologically degraded land with an aim to restore biodiversity and ecosystem services (Nattergal.co.uk, 2024). The Boothby site consists mainly of arable fields of variable soil types, hedgerows, and an adjacent ancient woodland, as well as several smaller patches of woodland. The West Glen River runs through the site from north to south. Since the site was purchased, the fields have been gradually taken out of agricultural production and rewilding will occur at the site in the following years. Since the data collection for this study has ended, a control field at Boothby has been made available and will continue to be farmed into the future. During the retreat from arable farming, two years of baselining was carried out. Data were collected on multiple species groups such as bats, birds, and butterflies. The baselining enabled a comparison between regions that were still being farmed, and those that have been designated as rewilding areas.

In the previous decade the farm has been part of an agri-environment scheme. This means there are regions of the farm that are less ecologically degraded. The field management was dictated by using ancient field margins to create multiple parcels within each modern field. Saplings have begun to colonise the fields where cultivation has been halted. Once natural vegetation has reestablished in these regions, there are plans to introduce free-roaming herbivores. Pond creation is also occurring to try and entice colonisation by great-crested newts (*Triturus cristatus)*. The section of the river that runs through Boothby is also being widened, and pools and meanders will be created to establish a more natural flow. In the year of the current research project, roughly two thirds of the parcels had been removed from farming, whilst one third were in their final year growing arable crops. Little is known about the small mammal populations in this site prior to this research.

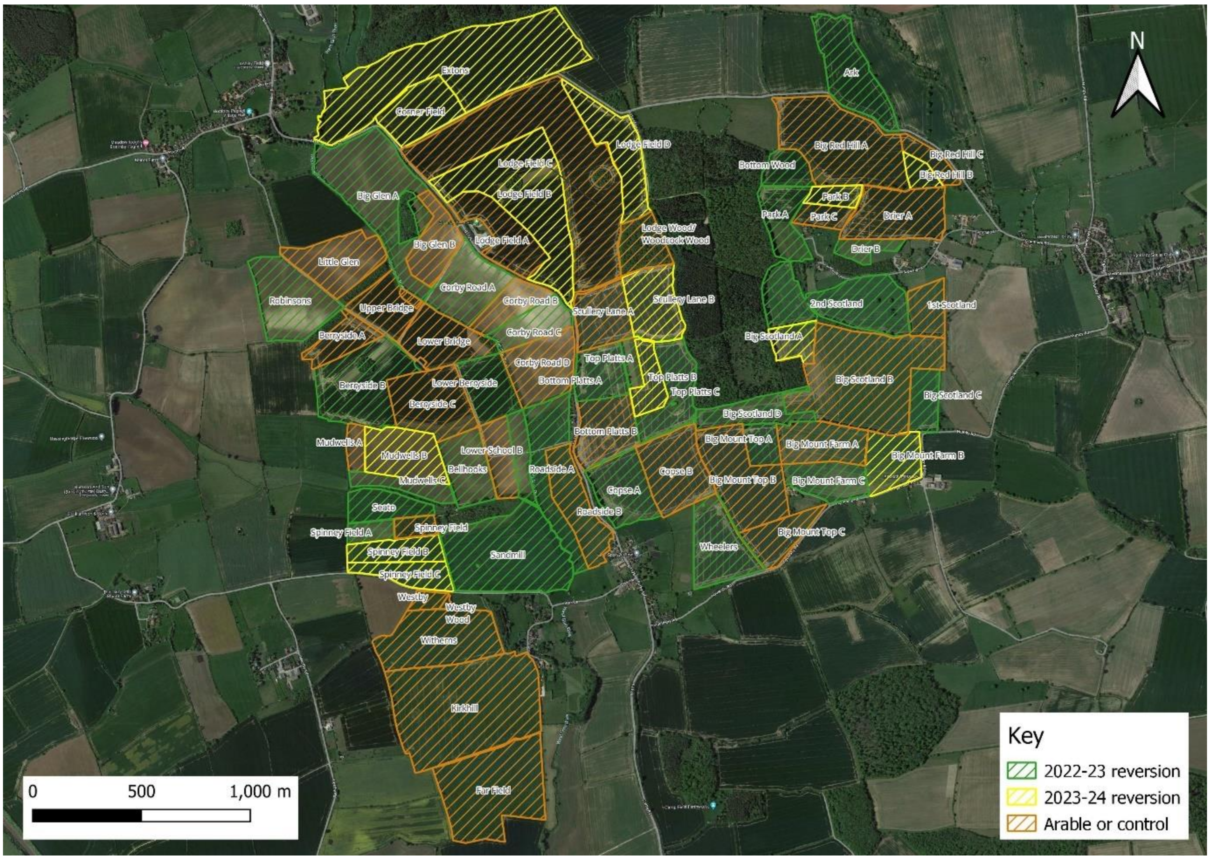

Figure 1. Map of Boothby Wildland showing field management status (QGIS satellite image).

2.2. Methodology

2.2.1. Experimental design

Data were collected for 9 months between October 2023 and June 2024. During each month of data collection four sites representing four different habitat types were surveyed: a 1-year reverted field, a 2-year reverted field, a crop field, and a hedgerow. One trapping grid was placed per field. A total of 12 sites were covered; this allowed for sampling of each site to be repeated three times throughout the 9-month period, with replicates roughly 3 months apart, therefore each treatment (e.g. 1-year reverted field) had 9 replicates. At the time of study, the local control site was not readily available, however, the most recently reverted fields which had just been cropped, started the study in a state close to that which would have been found in any control site. The purpose of the design was to see how the small mammal community at Boothby varied in the early stages of rewilding across different habitat types that can be characterised by differences in vegetation. The surveying took place over a 9-month period so that seasonal variation in the small mammal community at

Boothby could be measured. The replicates of the same field types in different seasons allow a temporal comparison, and the differences in field types allowed a spatial comparison.

Table 1. Trapping dates and site locations where Longworth trapping occurred

Date	Site name	GPS coordinates
16/10/23 – 20/10/23	Big Scotland	52.853549, -0.524625
16/10/23 – 20/10/23	Scullery Lane Hedgerow	52.858786, -0.535871
23/10/23 – 27/10/23	Lodge Field 2	52.863423, -0.533849
23/10/23 – 27/10/23	Lodge Field 1	52.860165, -0.536239
13/11/23 – 17/11/23	Big Glen	52.860284, -0.547444
13/11/23 – 17/11/23	Corby Road Hedgerow	52.859082, -0.541428
20/11/23 – 24/11/23	Little Glen	52.857859, -0.550606
20/11/23 – 24/11/23	Berryside	52.854775, -0.551644
11/12/23 – 15/12/23	Sandmill	52.847685, -0.544143
11/12/23 – 15/12/23	Spinney Field Hedgerow	52.846786, -0.550548
18/12/23 – 22/12/23	Big Red Hill	52.864471, -0.518762
18/12/23 – 22/12/23	Park	52.861917, -0.520248
15/01/24 – 19/01/24	Big Scotland	52.853549, -0.524625
15/01/24 – 19/01/24	Scullery Lane Hedgerow	52.858786, -0.535871
22/01/24 - 26/01/24	Lodge Field 2	52.863423, -0.533849
22/01/24 - 26/01/24	Lodge Field 1	52.860165, -0.536239
05/02/24 – 09/02/24	Big Glen	52.860284, -0.547444
05/02/24 – 09/02/24	Corby Road Hedgerow	52.859082, -0.541428
12/02/24 – 16/02/24	Little Glen	52.857859, -0.550606
12/02/24 – 16/02/24	Berryside	52.854775, -0.551644
04/03/24 – 08/03/24	Sandmill	52.847685, -0.544143
04/03/24 – 08/03/24	Spinney Field Hedgerow	52.846786, -0.550548
11/03/24 – 15/03/24	Big Red Hill	52.864471, -0.518762
11/03/24 – 15/03/24	Park	52.861917, -0.520248
01/04/24 – 05/04/24	Big Scotland	52.853549, -0.524625

01/04/24 – 05/04/24	Scullery Lane Hedgerow	52.858786, -0.535871
08/04/24 – 12/04/24	Lodge Field 2	52.863423, -0.533849
08/04/24 – 12/04/24	Lodge Field 1	52.860165, -0.536239
29/04/24 – 03/05/24	Big Glen	52.860284, -0.547444
29/04/24 – 03/05/24	Corby Road Hedgerow	52.859082, -0.541428
06/05/24 – 10/05/24	Little Glen	52.857859, -0.550606
06/05/24 – 10/05/24	Berryside	52.854775, -0.551644
27/05/24 – 31/05/24	Sandmill	52.847685, -0.544143
27/05/24 – 31/05/24	Spinney Field Hedgerow	52.846786, -0.550548
03/06/24 – 07/06/24	Big Red Hill	52.864471, -0.518762
03/06/24 – 07/06/24	Park	52.861917, -0.520248

2.2.2. Longworth Trapping

Small mammal trapping occurred using Longworth traps. A Longworth trap is an aluminium trap used to collect small mammals without harming them. It consists of a tunnel, leading to a nestbox. When an animal enters, it trips a door which shuts and traps the animal inside. The ingress point of the tunnel is 4.5cm by 4.5cm, meaning small mammals such as mice, voles, and shrews can enter the tunnel with ease, yet larger mammals are excluded from entry. Each trap was given a generous amount of hay inside the nest chamber for use as bedding. Wild bird seed mix (Pets at Home) and castors of the common bluebottle fly (*Calliphora vomitoria*) were used as bait.

2.2.3. Arrangement of traps

The traps were laid in 6 x 5 grids with 10m spacing between each trap. Each trap was given a code relating to its position within the grid (i.e. trap 5.2 was situated on the 5th column and the second row of the trapping grid). This configuration was broadly in-line with the recommendations of Gurnell and Flowerdew 2019, but with the grid size being designed to maximise the number of traps which could be set, given time constraints. Where applicable (i.e. where the vegetation was tall enough) every other trap was paired with an aerial trap to target harvest mice; aerial traps were Longworth traps raised approximately 1 metre above the ground by mounting them to either a wooden stake or bamboo cane. Due to the linear nature of the habitat, when surveying hedgerows 30 traps were laid out in a line within 3m of the habitat, as opposed to the 6 x 5 grids. The traps in the hedgerows were coded 1 to 30, 1 being the trap placed at the beginning of the hedge, and 30 being the trap laid at the end.

2.2.4. Trapping procedure

The following procedure was used each week of data collection. On day 1 food and bedding were added to the traps, and they were placed in the appropriate position in the field. The traps were left in the "pre-bait" position to help familiarise mammals with the traps and improve the catch rate. On day 2 in the morning the "pre-bait" setting was removed from the traps so that they would be available to catch small mammals for the evening. Bedding and food were replenished where necessary. From the evening of day 2, traps were checked for captured animals. When a trap was sprung the contents of the trap were emptied into a plastic bag. The animal in the bag was then weighed using a Pesola scale, with body mass calculated by deducting the known mass of the bag. The mammal was then held by the scruff of the neck (using gloves). The species and sex were determined and the hindleg was measured from bend to end. A fur clipping was then taken using scissors to cut a patch of outer fur on the left hind leg of the animal and exposing the darker fur beneath. This gave the individual a mark that could identify an animal upon recapture, however, the clip was not unique to the individual, so it was not possible to run analyses which require individual identity to be known. Once all measurements were taken the animal was released. Once released the trap was replenished and reset for capture. On days 3, 4 and 5 the steps described for day 2 regarding capture of an individual were carried out. In the evening of day 5 all traps were collected and cleaned out with warm soapy water.

2.2.5. Habitat surveying

The vegetation surrounding the trapping grids was surveyed each visit to establish height and coverage. The vegetation was measured to give a general impression of the variation amongst fields, however detailed surveying at all trap locations and over time was not feasible, hence vegetation is not included as a predictor in the analysis. A metre stick was used to measure one metre square around each Longworth trap. Plant coverage was estimated by viewing what percentage of the square featured vegetative growth. Plant height was determined by measuring the highest point of vegetation within the square metre surrounding each Longworth trap. The measurements were taken so that each field type could be characterised, providing evidence of a difference in the vegetation between each field type.

2.2.6. Audio

During several months of data collection, Longworth trapping was supplemented with audio recording. Due to constraints with obtaining equipment, not all audio devices could be used at all times. Therefore, throughout November, December, and January eight Audiomoths were used (4 per field). In April, two Mini Bats were used (one per site). In early May, one Mini Bat was used. Finally, in late May/early June, one Mini Bat and two Audiomoths were used, meaning one site had both types of audio recorder, and the other site had one Audiomoth.

The Audiomoths recorded at a sample rate of 192kHz and were scheduled to record during each night throughout the trapping grids. The Mini Bats recorded at a sample rate of 256kHZ. The audio devices were always scheduled at nighttime as most small mammals in the UK are predominantly nocturnal (Jensen and Honess, 1995), and limited storage space precluded constant 24-hour recording. Recording windows were adjusted as the sunset time changed throughout the year. In November, December, and January the Audiomoths were scheduled to record between 18:00-21:00, and 01:00-04:00. In April the Mini Bat devices were recording from 19:00-07:00. In early May the Mini Bat devices were recording from 20:00-06:00. In late May and Early June, during the period when both Audiomoths and Mini Bat devices were recording concurrently, all devices were scheduled to record from 21:00-05:00. When both Audiomoth and Mini Bat 2 were placed in the same position, they were set to the same recording schedule to ensure that any difference in small mammals recorded was due to the ability of each respective device to detect ultrasonic calls, and not a difference in scheduling. The Audiomoths were placed in the fields/ hedgerows along with the trapping grids in positions that provided optimal coverage. In the fields this meant placing an Audiomoth at grid positions 2.2, 2.4, 5.2, and 5.4. This meant that data were collected from all corners of the field. In the spring, additional audio recorders were added. These recorders were two song meter Mini Bats (Wildlife Acoustics). All audio recordings were uploaded to the British Trust for Ornithology (BTO) pipeline for analysis (BTO.org (2024)).

The data used to instruct the ultrasonic section of the BTO pipeline consisted of over 90,000 reference recordings of known species. The pipeline is a third party that provides the identifications without the user having direct control of the process. The species identified include bats, but also ultrasonic calls that are frequently misidentified as bat calls such as the calls of small mammals and bush crickets (BTO.org (2025)). This pipeline involves a third party that provides the identifications without the user having direct control of the process.

Table 2. Dates and locations of audio devices

Date	Site	Location	Device Type
13/11/23 – 17/11/23	Big Glen Corby Road Hedgerow	52.860284, -0.547444 52.859082, -0.541428	Audiomoth
20/11/23 – 24/11/23	Little Glen Berryside	52.857859, -0.550606 52.854775, -0.551644	Audiomoth
11/12/23 – 15/11/23	Sandmill Spinney Field Hedgerow	52.847685, -0.544143 52.846786, -0.550548	Audiomoth
18/12/23 – 22/11/23	Park Big Red Hill	52.861917, -0.520248 52.864471, -0.518762	Audiomoth
15/01/24 – 19/01/24	Scullery Lane Hedgerow Big Scotland	52.858786, -0.535871 52.853549, -0.524625	Audiomoth
22/01/24 – 26/01/24	Lodge Field 2 Lodge Field 1	52.863423, -0.533849 52.860165, -0.536239	Audiomoth
02/04/24 – 05/04/24	Scullery Lane Hedgerow Big Scotland	52.858786, -0.535871 52.853549, -0.524625	Mini Bat
08/04/24 – 12/04/24	Lodge Field 2 Lodge Field 1	52.863423, -0.533849 52.860165, -0.536239	Mini Bat
29/04/24 – 03/04/24	Big Glen	52.860284, -0.547444	Mini bat
06/05/24 – 10/05/24	Berryside	52.854775, -0.551644	Mini bat
27/05/24 – 31/05/24	Sandmill Spinney Field Hedgerow	52.847685, -0.544143 52.846786, -0.550548	Audiomoth (mini bat in Sandmill only)
03/06/24 – 07/06/24	Park Big Red Hill	52.861917, -0.520248 52.864471, -0.518762	Audiomoth (mini bat 2 in Park only)

2.2.7. Camera trapping

Four camera traps were placed per grid/transect. The cameras were attached to wooden stakes at approximately 1 metre height. The camera traps were first installed at 11/12/24 alongside the December trapping grids and were used in all following trapping grids until the research end date in June. Once placed, the cameras recorded continuously. Three images were taken in quick succession when the camera detected movement in its field of view. Still images were used as opposed to video recording to preserve memory so that the cameras could be left recording for long periods of time without interference. Images were photo stamped with date, time, and temperature at the moment the image was captured. Two types of camera trap were used: the Crenova RD1000 Trial Camera, and the Stealth Cam STC-G30. One was placed in each corner of the trapping grid to provide good coverage of the surroundings, as well as a view of the grid itself. The setup at the hedgerows consisted of four cameras placed 5 m from the hedgerow and framed to ensure coverage of the hedge and the adjacent field. One camera was placed to cover each quarter of the section of the hedge that was located within the line of Longworth traps. Their positions were also adjusted slightly if mammal tracks/ pathways were found. For example, observational data in the field confirmed the use of several routes by deer herds to commute between fields. These were covered by cameras wherever possible.

2.3.1. Data analysis: Longworth trapping

Data were analysed using Rstudio version 4.3.1. The analysis was carried out on wood mice (*Apodemus sylvaticus*) only because they represented the vast majority of catches (509/524 individuals, 97.14%). A generalised linear mixed model was used. The data were assumed to follow a binomial distribution with 1 representing the presence of a wood mouse in a Longworth trap and 0 representing the absence. The presence or absence of wood mice in each trap location was the dependent variable. Backwards stepwise model selection was used, with trap location nested within grid location as random effects. Independent variables included in the model were field type (i.e. habitat), time of day (morning or evening), trapping day (with the first day when traps were activated being day 1) and trapping month. Interactions between field type and trapping day, field type and time of day, and field type and month were also included. Terms were tested using likelihood ratio (LR) tests comparing models with and without each fixed effect. These models were run on data from all captured wood mice regardless of whether they were marked or unmarked. These models were stable and gave consistent results with the new simplified analysis i.e. total captures per hour and minimum number alive models.

2.3.2. Data analysis: trap hours

The total number of captures per trap hour was estimated per grid by dividing the total captures per grid by the number of hours that the traps in each grid were open. These data were then analysed in RStudio version 4.3.1. A linear model was used were total trap hours divided by trap hours was the response variable. Fixed variables were habitat and month, which were tested for main effects, and for interactions.

2.3.3. Data analysis: minimum number alive

The minimum number of small mammals alive in each trapping grid was calculated by calculating the number of animals caught during the first trapping day, and adding the number of unmarked animals caught in the following trapping days. These data were then analysed in RStudio version 4.3.1. A negative binomial generalised linear model was carried out where fixed variables were habitat and month, which were tested for main effects, and for interactions.

2.3.4. Data analysis: Audio

Data that were gathered from the three-month period (13/11/23 – 26/01/24) where each field featured 4 Audiomoths were analysed using Rstudio version 4.3.1. A generalised linear mixed model was used. Backwards stepwise model selection was used. The data were assumed to follow a Poisson distribution. The audio data were filtered for high confidence levels before analysis. The model was run with all species recordings, and again with just small mammal recordings. Two different models were run, one model was run with total number of species as the dependent variable, and the other ran for total number of calls as the dependent variable. Independent variables included in the model were field type, and month, and an interaction was tested between field type and month. The specific field surveyed was added as a random effect. Terms were tested using likelihood ratio (LR) tests comparing models with and without each fixed effect.

2.4.1. Spectrogram analysis

The spectrograms of certain recordings were investigated to verify the validity of the BTO pipeline as a method of providing accurate identification of species from audio recordings. The spectrograms were analysed using Audacity version 3.6.4. Audio files were downloaded and loaded into Audacity and switched to spectrogram view. The spectrogram was then viewed in full screen, and the vertical column was expanded so that it covered 90KHz. Clip speed was slowed down to 10% in the pitch and speed menu as small mammal calls are best played 10 times slower than real time (Newson, Middleton, and Pearce, 2020). The Hanning window size was changed in spectrogram settings to 1024 (as seen in Sound Identification of Terrestrial Mammals of Britain and Ireland (Middleton, Newson, and Pearce, 2023). The calls on the spectrograms were then visually compared to existing known small mammal calls from a reference textbook (Sound Identification of Terrestrial Mammals of Britain & Ireland, Middleton, Newson, and Pearce, 2023) to determine the accuracy of the identification.

3.

Results

3.1. Vegetation

In October when surveying began, the crop fields had just been drilled so these fields were bare, with very little vegetative growth (October mean crop cover 0%, October mean crop height 0cm) (Figure 2, figure 3). As time progressed throughout the trapping seasons the crop fields began to be characterised by uniform wheat crop growth which increased in height between October and June (June mean crop cover 100%, June mean crop height 75.33cm) (Figure 2, figure 3)**.** The 1-year reverted fields were characterised by short and patchy vegetation (mean cover 36.19 %, mean height 20.23cm), with mean vegetation cover gradually increasing from 19.53 % in November to 75.33 % in June (figure 2), and with mean vegetation height increasing from 9.83cm in November to 43cm in June (figure 3)**.** Fields in this successional stage consisted of a large proportion of herbaceous plants such as curly dock (*Rumex crispus*), creeping thistle (*Cirsium arvense*), common ragwort (*Jacobaea vulgaris*), and bristly oxtongue (*Helminthotheca echioides*). The 2-year reverted fields tended to have tall vegetation with much less exposed ground (mean cover 86.45 %, mean height 30.64cm). The mean cover for 2-year reverted fields was 70.17 % in November and 91% in May (Figure 2), and the mean vegetation length was 24.13cm in November and 43.9cm in May (figure 3). The 2-year reverted fields were mainly dominated by long grass. The hedgerows were characterised by tall woody flora, and the adjacent field margins also consisted of lengthy, continuous vegetation (mean margin cover 99.87%, mean margin height 46.9cm).

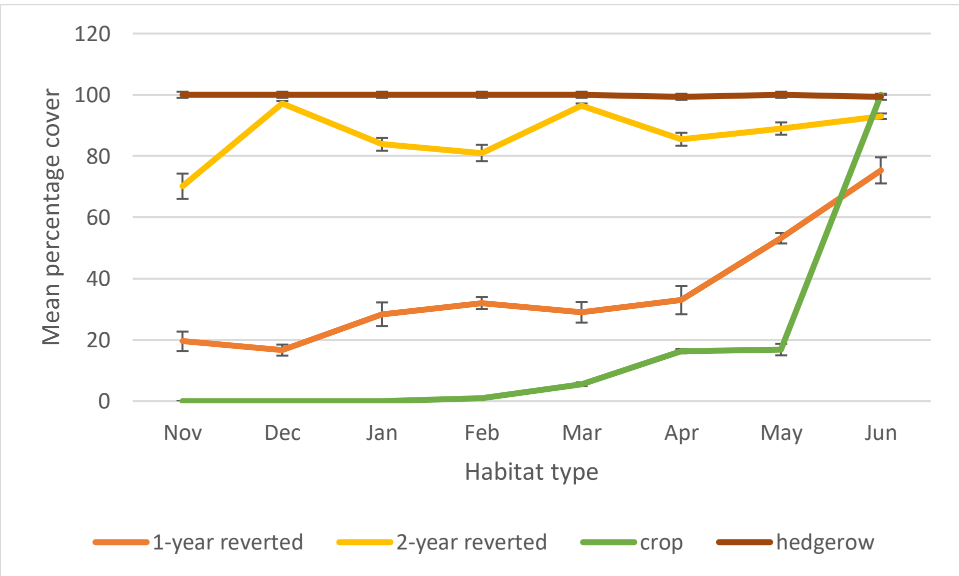

Figure 2. Mean percentage vegetation ground cover by month in the three different surveyed field types.

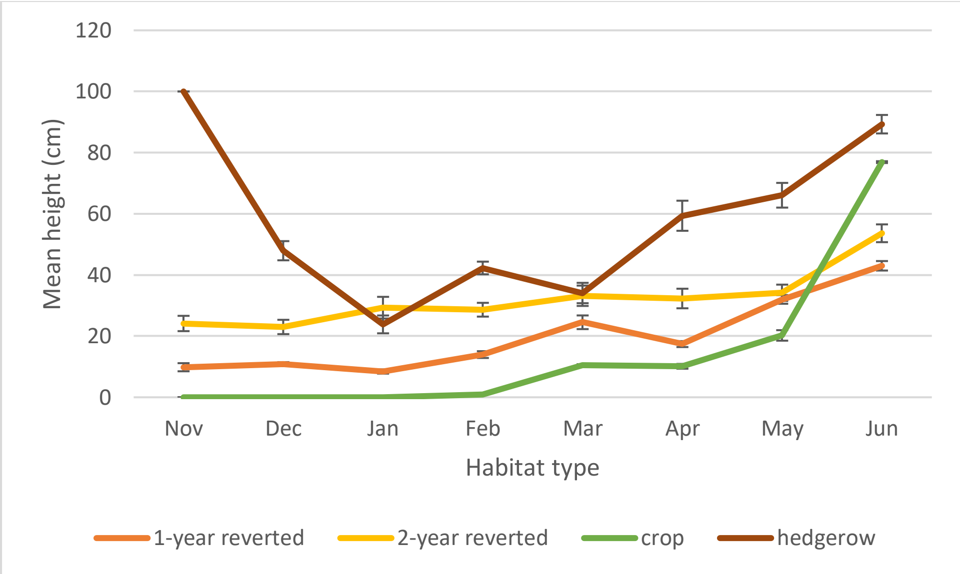

Figure 3. Mean vegetation height by month in the three different surveyed field types.

3.2.1 Longworth trapping

Of the 7080 total trap checks, a total of 524 caught small mammals, giving a catch rate of 7.4 %. The most commonly caught species caught by far was the wood mouse (*Apodemus sylvaticus*): 509 individuals (97.14 %) were wood mice. The remaining 15 individuals included 8 (1.53 %) field voles (*Microtus agrestus)*, 5 (0.95 %) common shrews (*Sorex araneus*), 1 (0.19 %) Pygmy shrew (*Sorex minutus*), and 1 (0.19 %) harvest mouse (*Micromys minutus*). Of the total small mammals caught, 293 were marked (56 %), meaning a total of 231 unique individuals were captured. No small mammals were caught in the elevated traps.

There was significant seasonal variation in the probability of capturing a wood mouse, with much higher capture rates in autumn months than in late winter and spring (figure 4; LR test: chi squared = 248.65, degrees of freedom = 1, $P < 0.001$). No significant difference occurred between field types in the probability of catching a wood mouse (figure 4; chi squared =5.8316, degrees of freedom = 3, P=0.1201). There was, however, a significant interaction between the effects of field type and trapping month on the probability of catching a wood mouse (figure 4; chi squared =85.974, degrees of freedom = 3, $P<0.001$). This interaction is a result of the fact that between October and December, wood mice were much less likely to be captured in the crop fields than the other habitat types. After these months, the differences in catch rates amongst habitat types become smaller, although wood mice tended to be most likely to be caught in the 1-year reverted fields in March and April.

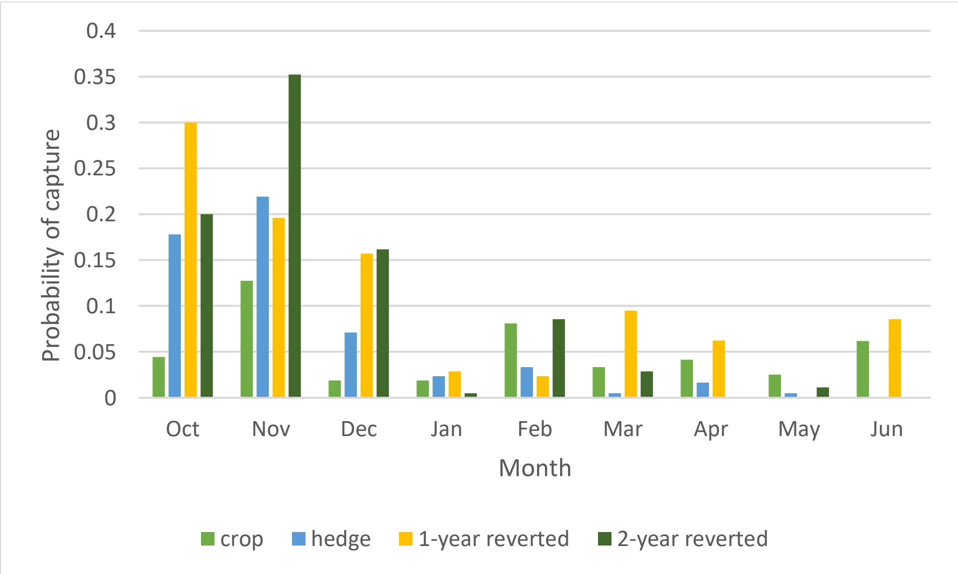

Figure 4. Probability that a wood mouse (*Apodemus sylvaticus*) was caught in any given trap during each month and in each of the four surveyed habitat types. i.e. the proportion of available trap sessions in which an animal was caught.

The probability that wood mice would be caught in the morning was significantly higher than the probability that they would be caught in the evening (figure 5; chi squared =195.04, degrees of freedom = 1, $P<0.001$).

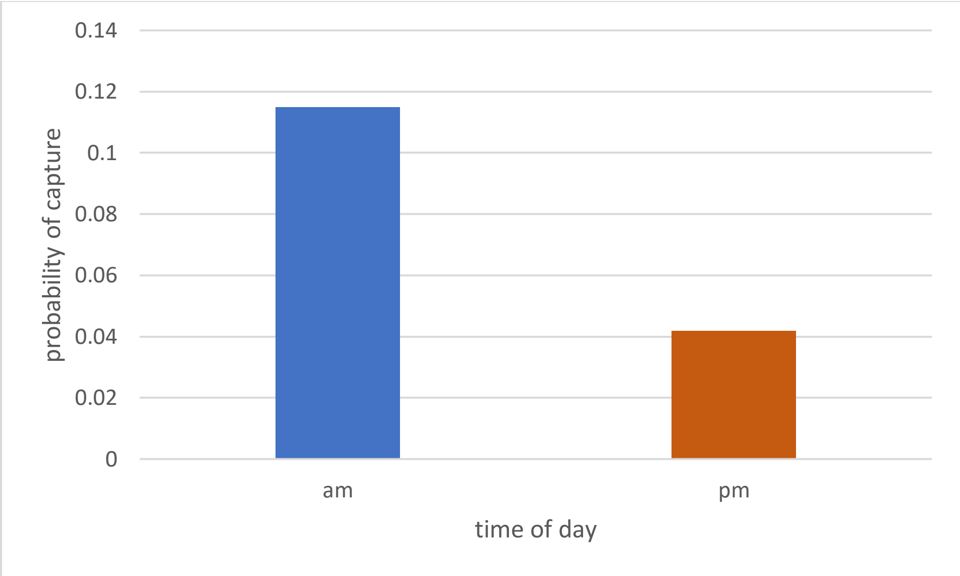

Figure 5. Probability that a wood mouse (*Apodemus sylvaticus*) was caught in traps checked in the morning and evening across all months and field types.

There was no significant effect of the trapping day on the probability of catching a wood mouse (chi squared =1.2292, degrees of freedom = 1, P=0.2676). There was also no significant interaction between trapping day and field type on the probability of catching a wood mouse (chi squared =0.1295, degrees of freedom = 3, P<0.9881). Finally, there was no significant interaction between the effects of field type and time of day on the probability of catching a wood mouse (chi squared =3.9881, degrees of freedom = 3, P=0.2628).

3.2.2. Captures per trap hour

When examined in isolation, habitat type had no significant impact on captures per trap hour (degrees of freedom 3,31, p=0.2376). However, captures per trap hour did change significantly depending on the trapping month (degrees of freedom 1,34, p<0.001). There was also a significant interaction between month and field type regarding the total captures per trap hour (degrees of freedom =3,28, p=0.0354), meaning that as the months went by, the total captures per trap hours decreased in all habitat types, with the exception of crop fields where this remained low throughout the entire study period.

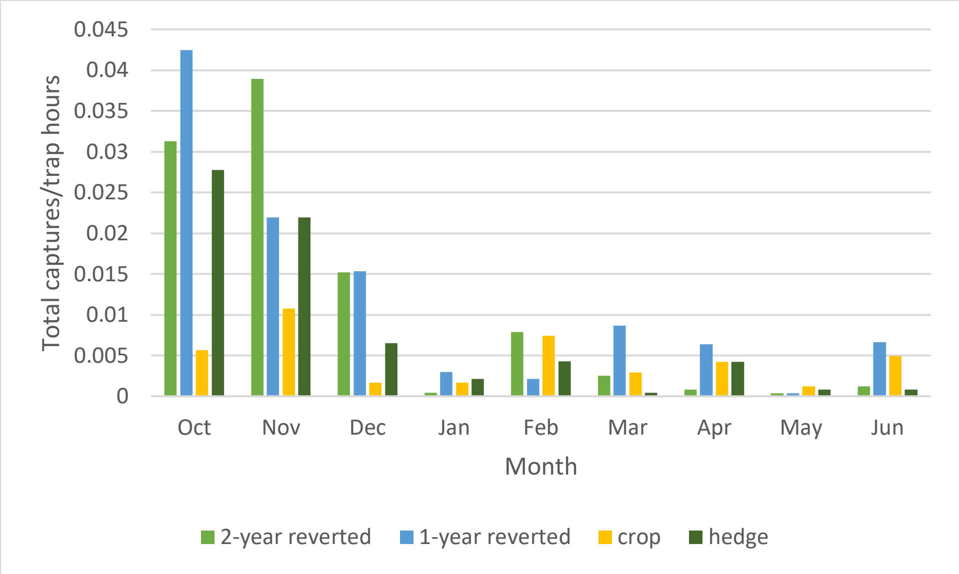

Figure 6. total captures divided by trap hours in each trapping grid across all trapping months and habitat types

3.2.3. Minimum number alive

Habitat type alone did not have a significant effect on the minimum number of small mammals alive per trapping grid (degrees of freedom=3, likelihood ratio=2.3518, P=0.5027). The trapping month, however, did have a significant impact on the minimum number alive in each trapping grid (degrees of freedom =1, likelihood ratio 11.8933, P<0.001). There was also a significant interaction between month and field type with regards to the minimum number of small mammals alive in each trapping grid (degrees of freedom=3, likelihood ratio=10.8007, P=0.0129). These data, along with the captures per trap hour data, concur with the pattern displayed in figure 4, where more wood mice are being caught in Longworth traps placed in non-crop fields in the autumn months.

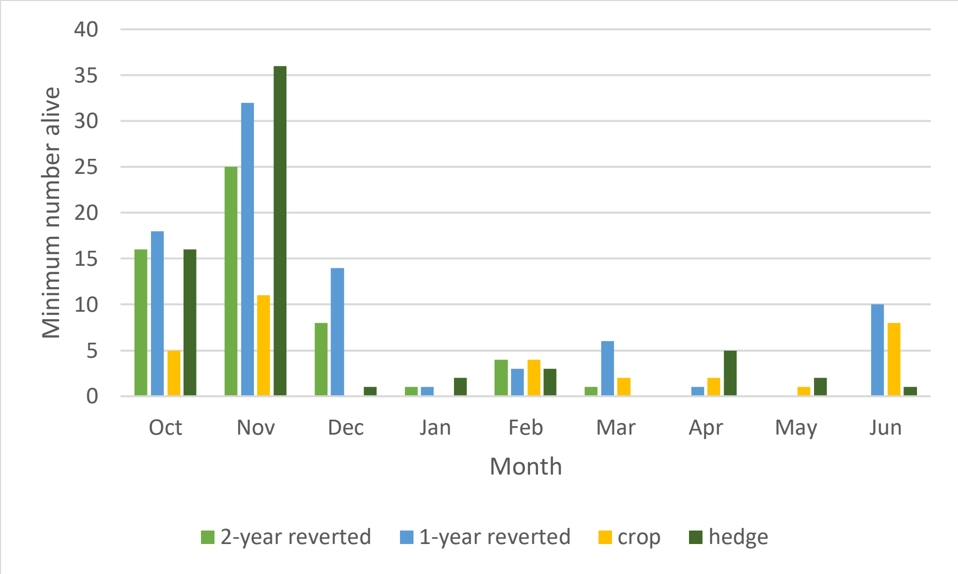

Figure 7. Minimum number of small mammals alive in each trapping grid in each habitat type across all trapping months

3.3. Audio

3.3.1. Totals and confidence levels

The results from the BTO pipeline throughout the whole duration of the study identified nine species of small terrestrial mammals, along with 13 species of bats, and five species of bush crickets (table 3). The total number of recordings throughout the study was 133,565. Of this total, 76,747 (57.46 %), were bats, 28,044 (21.00 %) were crickets, 27,895 (20.89 %) were birds, and 879 (0.66 %) were small terrestrial mammals. When filtered for only identifications made with confidence levels of 0.5 or above, there were a total of 50,240 recordings: 28,708 (57.14 %) bats, 13,854 (27.58 %) crickets, 7,566 (15.06 %) birds, and 112 (0.22 %) small mammals (table 3). If filtered for identifications made at an even greater confidence level of 0.8 or above, the proportions of calls identified per species group remains similar to when the data were filtered at 0.5 confidence level. There were a total of 21,363 recordings: 11,253 (52.68%) bats, 6466 (30.27%) crickets, 3598 birds (16.84%), and 46 (0.22%) small mammals.

Five species were only recorded at low confidence levels. These species were as follows: brown rat (*Rattus norvegicus*), bank vole (*Myodes glareolus*), serotine bat (*Eptesicus serotinus*), short-winged bush cricket (*Conocephalus dorsalis*), and yellow-necked mouse (*Apodemus flavicollis*). The total number of recordings for many species was much greater when low confidence recordings were included. This was particularly the case for the alcathoe bat (*Myotis alcathoe*), common shrew (*Sorex Araneus*), Nathusius' pipistrelle (*Pipistrellus nathusii*), and hazel dormouse (*Muscardinus avellanarius*) (table 3).

Table 3. total number of calls for each species detected throughout the entire period of data collection across all months, field types, and recording devices, with and without a high confidence filter.

Species	all	Relative proportion (%)	High confidence	Relative proportion (%)
Bird spp.	27895	20.88	7566	15.06
Alcathoe Bat (*Myotis alcathoe*)	10897	8.16	530	1.055
Barbastelle (*Barbastella barbastellus*)	187	0.14	161	0.32
Brown Long-eared Bat (*Plecotus auratus*)	25	0.019	15	0.030
Common Pipistrelle (*Pipistrellus pipistrellus*)	8405	6.29	6990	13.91
Daubenton's Bat (*Myotis daubentoniid*)	343	0.26	50	0.1
Greater Horseshoe Bat (*Rhinolophus ferrumequinum*)	8	0.006	5	0.01
Leisler's Bat (*Nyctalus leisleri*)	155	0.12	89	0.18
Nathusius' Pipistrelle	639		28	

(*Pipistrellus nathusii*)		0.48		0.056
Natterer's Bat (*Myotis nattereri*)	178	0.13	106	0.21
Noctule (*Nyctalus noctula*)	51	0.038	34	0.068
Serotine (*Eptesicus serotinus*)	38	0.028	0	0
Soprano Pipistrelle (*Pipistrellus pygmaeus*)	55820	41.79	20699	41.2
Whiskered Bat (*Myotis mystacinus*)	1	0.00075	1	0.002
Bank Vole (*Myodes glareolus*)	9	0.0067	0	0
Brown Rat (*Rattus norvegicus*)	3	0.0022	0	0
Common Shrew (*Sorex Araneus*)	499	0.37	15	0.030
Eurasian Harvest Mouse (*Micromys minutus*)	18	0.01	2	0.004
Eurasian Pygmy Shrew (*Sorex minutus*)	60	0.045	10	0.02
European Water Vole (*Arvicola amphibius*)	21	0.016	7	0.014
Hazel Dormouse (*Muscardinus avellanarius*)	139	0.1	3	0.006

Wood Mouse (Apodemus sylvaticus)	115	0.086	75	0.15
Yellow-necked Mouse (*Apodemus flavicollis*)	15	0.011	0	0
Dark Bush-cricket (*Pholidoptera griseoaptera*)	15832	11.85	9363	18.64
Great Green Bush-cricket (*Tettigonia viridissima*)	8141	6.095	2987	5.95
Grey Bush-cricket (*Platycleis albopunctata*)	17	0.013	2	0.004
Short-winged Bush-cricket (*Conocephalus dorsalis*)	22	0.016	0	0
Speckled Bush-cricket (*Leptophyes punctatissima*)	4032	3.019	1502	2.99
Grand Total	133565		50240	

Mean confidence levels from the BTO pipeline varied widely between species (table 4). These mean confidence levels also varied across taxa. The average confidence level of all bat species was 0.5. This was 0.4 for birds, 0.36 for crickets, and 0.29 for mammals. The small terrestrial mammal species with the lowest mean confidence was the hazel dormouse, and the small terrestrial mammal species that had the highest average confidence level was the wood mouse (table 4).

Table 4. Mean confidence level applied to each species detection by the BTO pipeline across all months and recording devices.

Species	Mean confidence level	Standard error
bird spp.	0.40	0.001
alcathoe (*Myotis alcathoe*)	0.21	0.001
barbastelle (*Barbastella barbastellus*)	0.86	0.02
brown long-eared (*Plecotus auritus*)	0.63	0.067
common pipistrelle (Pipistrellus pipistrellus)	0.67	0.0018
Daubentons bat (*Myotis daubentonii*)	0.29	0.01
greater horseshoe (Rhinolophus ferrumequinum)	0.57	0.087
Leislers (*Nyctalus leisleri*)	0.56	0.022
nathusius pipistrelle (*Pipistrellus nathusii*)	0.22	0.0052
Natterer's bat (*Myotis nattereri*)	0.58	0.023
noctule (*Nyctalus noctula*)	0.68	0.044
serotine (*Eptesicus serotinus*)	0.13	0.0097
soprano pipistrelle (*Pipistrellus pygmaeus*)	0.44	0.001
whiskered bat (*Myotis mystacinus*)	0.60	n/a
bank vole (*Myodes glareolus*)	0.18	0.017
brown rat (*Rattus norvegicus*)	0.27	0.057
common shrew (*Sorex araneus*)	0.18	0.0049
harvest mouse (*Micromys minutus*)	0.21	0.036
pygmy shrew (*Sorex minutus*)	0.28	0.029
water vole (*Arvicola amphibius*)	0.37	0.053
hazel dormouse (M*uscardinus avellanarius*)	0.15	0.0092
wood mouse (*Apodemus sylvaticus*)	0.65	0.024
yellow necked mouse (*Apodemus flavicollis*)	0.17	0.021
dark bush cricket (*Pholidoptera griseoaptera*)	0.57	0.0022

great green bush cricket (*Tettigonia viridissima*)	0.43	0.0028
grey bush cricket (*Platycleis albopunctata*)	0.24	0.05
short winged bush cricket (*conocephalus dorsalis*)	0.15	0.015
speckled bush cricket (*Lectophyes punctatissima*)	0.43	0.0044

3.3.2 Device comparison

When Audiomoth and Mini Bat recording devices were placed in the exact same location, with identical settings (2.2.6. audio), many species were recorded the same number of times on both devices (table 5). However, there were a few differences. For example, Daubentons, noctule, soprano pipistrelle, and common pipistrelle bats, were recorded more on the Mini Bat than the Audiomoth. Also, three species: brown long-eared bat, Leisler's bat, and wood mouse, were only detected on the Mini Bat. Throughout this period a total of 953 calls were recorded on the Audiomoth, and 999 calls were detected on the Mini Bat. No bush crickets were detected on either recording device during this period. Birds were excluded from the comparison as the focus was on the difference in ultrasonic recording capabilities and birds call mainly in the audible frequency range.

Table 5. Comparison of total calls for each species detected when both Audiomoth (AM) and Mini Bat (MB) detectors were placed in the same location across all confidence levels.

	29th May	29th May	30th May	30th May	31st May	31st May	3rd June	3rd June
	AM	MB	AM	MB	AM	MB	AM	MB
Alcathoe Bat	28	31	51	51	277	277	374	374
Barbastelle	7	18	0	0	0	0	0	0
Brown long-eared bat	0	0	0	3	0	1	0	0
Common Pipistrelle	21	29	20	20	61	61	83	83
Common Shrew	3	3	6	6	0	0	5	5

Daubenton's Bat	1	5	0	0	0	0	0	0
Hazel dormouse	4	4	4	4	0	0	0	0
Leisler's bat	0	6	0	1	0	1	0	0
Noctule	2	6	0	0	0	0	0	0
Serotine	1	1	0	0	0	0	0	0
Soprano pipistrelle	5	8	0	0	0	0	0	0
Wood mouse	0	0	0	0	0	1	0	0
Total	72	111	81	85	338	341	462	462

3.3.3 Month and field type

When looking at the effect that month had on just wood mice, it was revealed that during this period a total of 75 wood mice were detected via audio recording, 67 of which were recorded in November, which was also the month with the greatest number of wood mice caught in Longworth traps. A correlation wasn't possible between audio and trap data because due to recording failures and a limited window where audio equipment was available, the audio data was limited to a shorter period of time than the Longworth traps. During this time a small number of small mammals were detected by the Audio equipment.

In the three-month period where each field had four Audiomoths, November had a far greater number of total recordings (84,838), followed by December (20,054), and then January (12,145). This trend remains the same when filtered for only high confidence recordings, with November at 33,142, December at 6,202, and January at 4,047. There was a significant difference in mean calls per day detected between these months (chi squared = 37.013, degrees of freedom = 2, $P < 0.001$). There was also a significant difference in mean calls detected in different field types (chi squared = 23.255, degrees of freedom = 3, $P < 0.001$), and there was a significant interaction between the effects of month and field type on the mean calls detected (chi squared = 71.644, degrees of freedom = 6, $P < 0.001$). (figure 6). Crop fields had very low numbers of total recordings across all months. This effect was particularly prevalent in November when the other habitat types had much higher numbers of recordings.

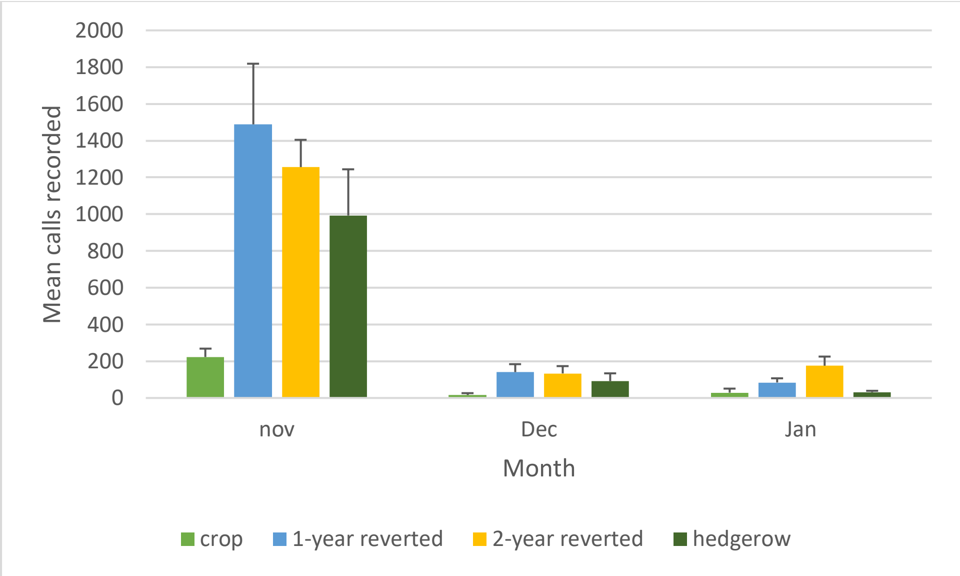

Figure 8. Mean number of high confidence calls per day detected in audio recordings from Audiomoth devices across all field types (one field type per month) in autumn/winter months. In all cases, devices were recording for six hours per day (dawn and dusk), and for four days per month per field.

When testing all recorded species at a high confidence threshold there was a significant difference in the mean number of species detected per day in different months, with the vast majority of species being detected in November (LR test: chi squared =37.013, degrees of freedom = 2, P < 0.001). There was also a significant difference in the mean species detected in different field types, with fewer species being detected in the crop fields (chi squared =15.079, degrees of freedom = 3, P = 0.00175). There was no significant interaction between the effects of field type and month on the mean number of species detected (chi squared =2.4276, degrees of freedom = 6, P = 0.8765), meaning the effect of field type on mean species detected was consistent as time progressed (figure 7).

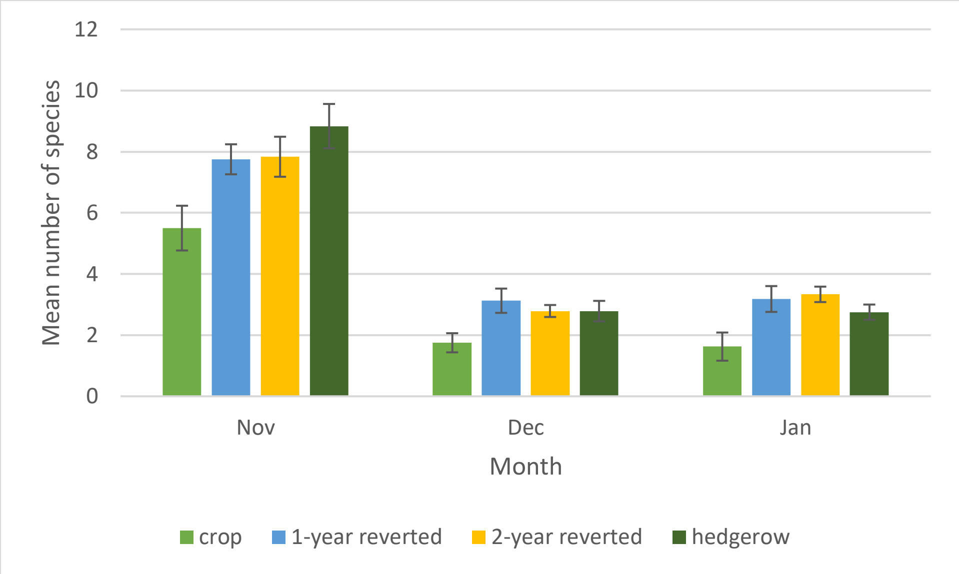

Figure 9. Mean number of species detected per day in high confidence audio recordings from Audiomoth devices across all field types (one field type per month) in autumn/winter months. In all cases, devices were recording for six hours per day (dawn and dusk), and for four days per month per field.

When testing for just small mammal calls there was no significant difference between mean number of calls per day detected in different months (chi squared = 5.5269, degrees of freedom = 2, $P = 0.06307$), although there were generally more calls detected in November. The difference between field types also did not significantly alter the mean number of small mammal calls detected (chi squared = 5.1626, degrees of freedom = 3, $P = 0.1603$), although no calls were detected in the crop fields. There was, however, a significant interaction between the effects of month and field type on the mean number of calls detected (chi squared = 30.444, degrees of freedom = 6, $P < 0.001$). Many more calls were detected in 2-year reverted fields than other habitat types in November, but in December and January very few calls were recorded in any habitat.

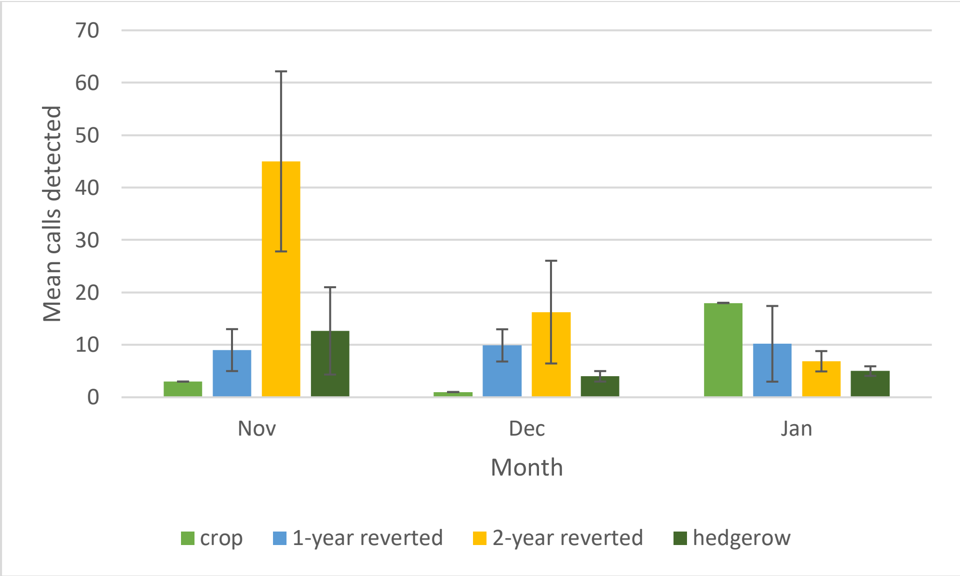

Figure 10. Mean number of small terrestrial mammal calls detected per day in high confidence recordings detected in audio recordings from Audiomoth devices across all field types (one field type per month) in autumn/winter months. In all cases, devices were recording for six hours per day (dawn and dusk), and for four days per month per field

When testing for just small mammal calls there was no significant difference in the mean number of species detected per day in different months (chi squared = 5.9597, degrees of freedom = 2, P = 0.0508). The mean small mammal species also did not differ significantly between field types (chi squared = 6.8597, degrees of freedom = 3, P = 0.07651). However, despite these two factors being non-significant when examined in isolation, there was a significant interaction between the effects of month and field type on mean number of species detected (chi squared = 13.95, degrees of freedom = 6, P = 0.0302), this is due to the greater mean species being detected in 2-year reverted fields in November.

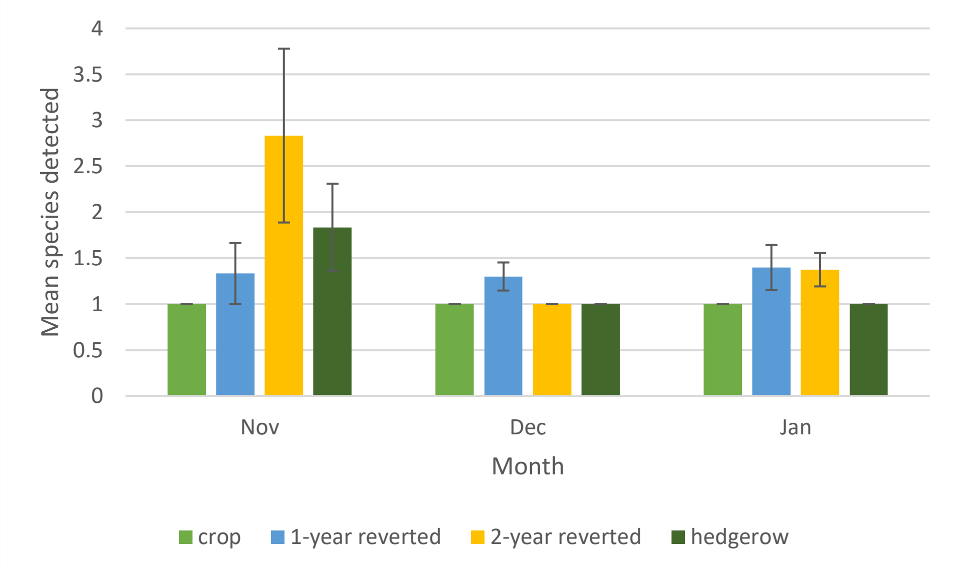

Figure 11. Mean number of small terrestrial mammal species detected per day in high confidence audio recordings from Audiomoth devices across all field types (one field type per month) in autumn/winter months. In all cases, devices were recording for six hours per day (dawn and dusk), and for four days per month per field.

3.4. Spectrograms

The audio recordings of wood mice that were given high confidence levels by the BTO pipeline shared similar ultrasonic call characteristics to reference wood mouse spectrograms (figures 10 and 11). Both the spectrograms of Audiomoth recordings from this study, and reference wood mouse spectrograms from the literature feature calls of a characteristic hockey stick shape between 50KHz and 70KHz. The low confidence wood mouse recordings featured no characteristic call structure (figure 12). Hazel dormouse spectrograms did not resemble those from the literature, with no characteristic horizontal line (figures 13 and 14).

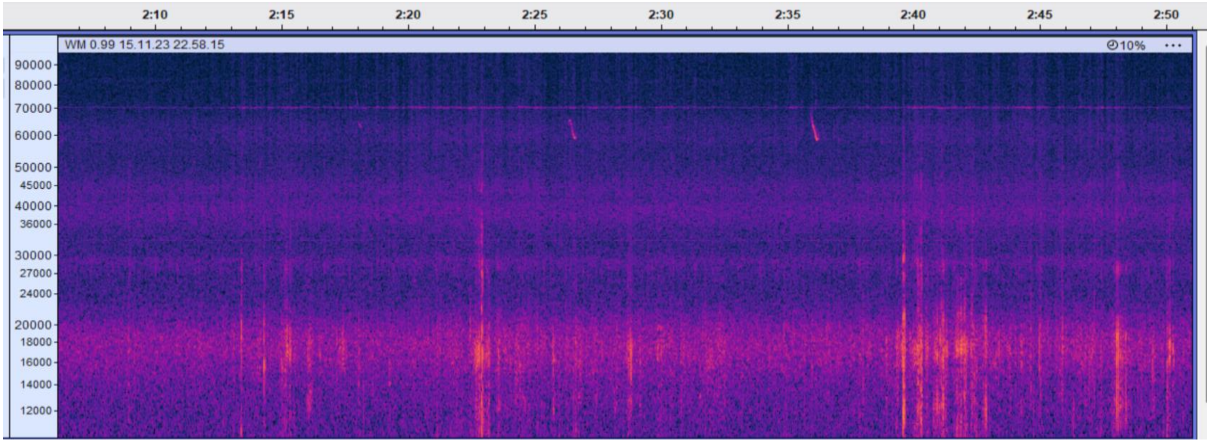

Figure 12. Spectrogram of a high confidence level (0.99) wood mouse spectrogram, fmax >70KHz (frame width: c.0.9 sec, size 1024, Hanning window).

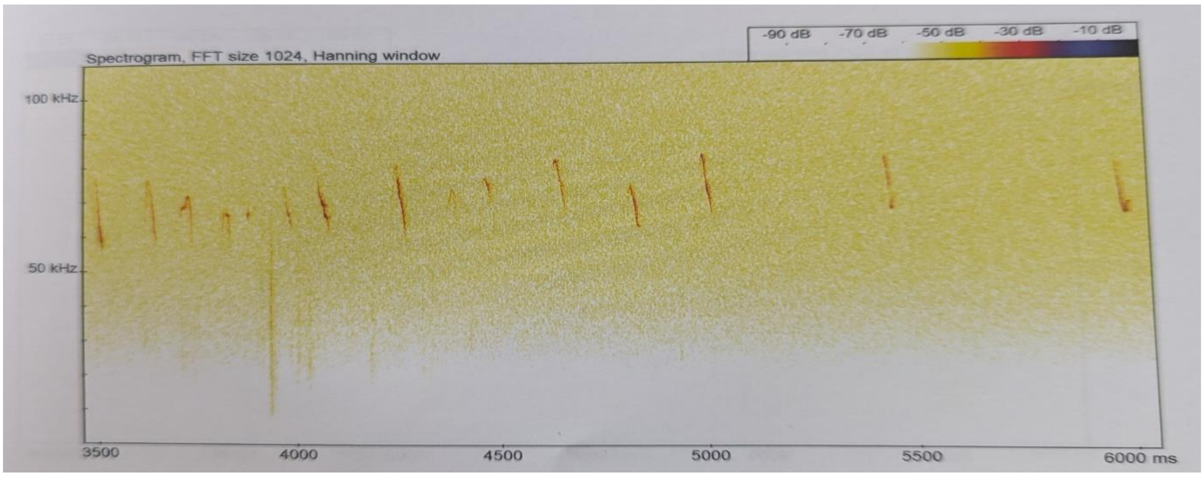

Figure 13. Spectrogram of a wood mouse-a series of higher-frequency calls with fmax >70KHz (frame width: c.2.5 sec) (Middleton, Newton, and Pierce, 2023).

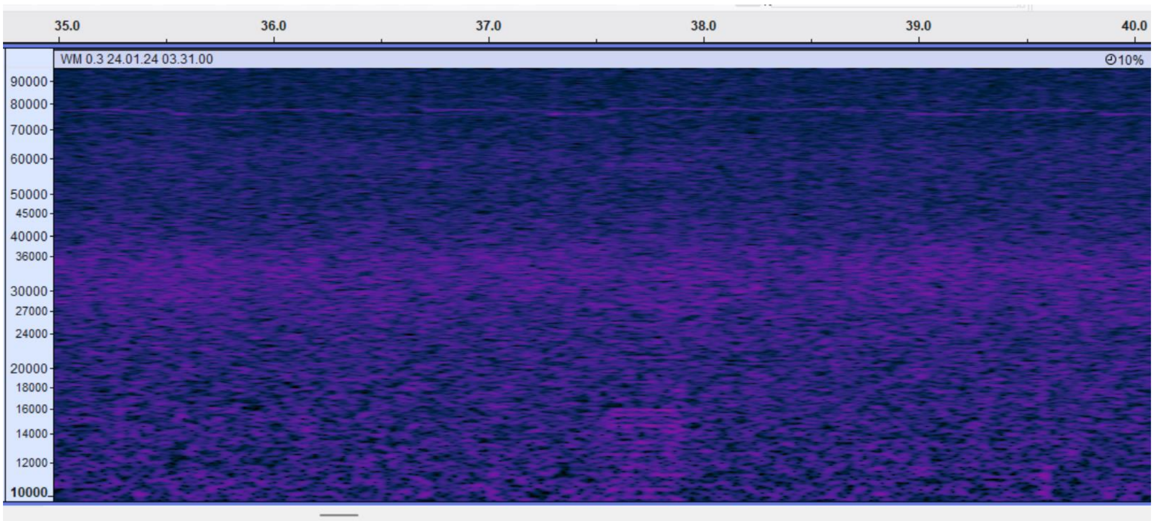

Figure 14. Spectrogram of a low confidence level (0.3) wood mouse call (frame width: c.5 seconds, size 1024, Hanning window).

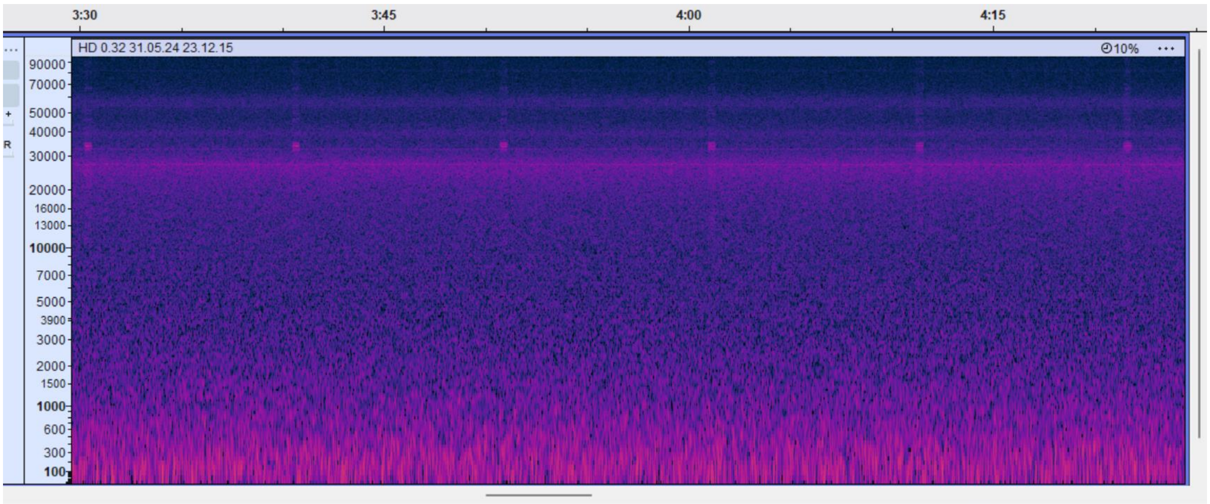

Figure 15. Spectrogram of a low confidence (0.32) Hazel dormouse call (frame width: c.1.25 seconds, size 1024, Hanning window).

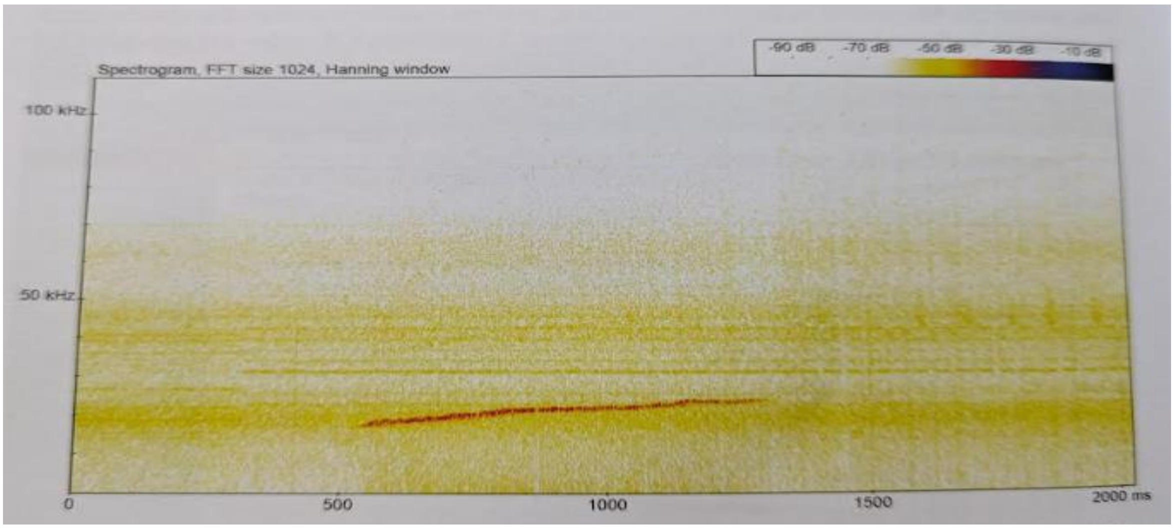

Figure 16. Spectrogram of a hazel dormouse-ascending FM call (lower frequency) (frame width c.2 seconds, size 1024, Hanning window) (Middleton, Newton, and Pierce, 2023).

3.5. Camera trapping

Camera trapping aimed at capturing large mammal presence at Boothby revealed a total of 59 individuals and 8 known species. Sixteen hares (*Lepus europaeus*), 16 deer that were unidentifiable to species level, 10 fallow deer (*Dama dama*), 4 European badgers (*Meles meles*), 4 unidentifiable animals, 3 muntjac (*Muntiacus reevesi*), 2 owls (unidentifiable to species level), 2 pheasants (*Phasianus colchicus*), 1 roe deer (*Capreolus capreolus*), and 1 carrion crow (*Corvus corone*).

The greatest proportion of animals were captured in the 1-year reverted fields (26/59, 44.07 %), followed by the crop fields (15/59, 25.4 %), 2-year reverted fields (10/59, 16.9 %), and hedgerows (8/59, 13.6 %).

4.

Discussion

4.1. Results summary

Five species of small mammals were identified via Longworth trapping. 27 species were also identified via audio recording, nine of which were small mammals, although many of these species may have been false positive, as confidence levels varied greatly. Eight species were also identified via camera trapping.

Wood mouse was the most prominent species throughout all habitat types, and numbers were greater in the autumn and winter compared with spring and summer. It isn't possible to get a full assessment of seasonal effects of the audio equipment, as it wasn't available for the full length of the study. However, November was the number with the greatest number of trapping events, and small mammal calls.

Audio equipment increased the number of species detected, the mini bat audio detectors recorded more species than the Audiomoths when both were placed side by side and settings were identical.

4.2. Wood mice

The fact that wood mice represented the majority of catches is unsurprising as it is one of the UKs most abundant small mammals (Mammal Society.org, 2024). This is also backed by the literature as many similar trapping studies found the wood mouse to be similarly abundant amongst trapping grids (Tattersall et al., 2002, Churchfield, Hollier, and Brown, 1997, Gorman, 2018, Kotzageorgis and Mason, 2009). Especially given the low regional diversity, it is not unusual in the UK for small mammal communities to be dominated by a small number of species (Tattersall et al., 2002).

Wood mouse numbers being greatest in the autumn (and having more captures per trap hour and having a greater minimum number alive during this time) aligns with food availability for granivorous small mammals as this is when seed production reaches its peak (Green, 1979). In late winter and spring, food availability is much lower. A combination of this lowered food availability, and the colder weather typical of winter, which increases metabolic demands (Hume et al., 2019), typically leads to lower population numbers of wood mice (Gurnell, 1978, Janova et al., 2011). Wood mice may also move to over habitats such as adjacent woodland in the winter (Fitzgibbon, 1997). This explains why the catch rate of wood mice dropped significantly in late winter and spring.

The fact that wood mice representing such a high proportion of the trapped small mammals at Boothby, may be partly due to this species reaching the growth/peak phase of its inter-annual variation (Andreassen et al., 2021, Sunyer et al., 2016, Montgomery et al., 1991, Kotzageorgis and Mason, 2009). Wood mice population numbers, along with other small mammal species, vary significantly between years due to differences in food availability and weather patterns. It is possible that the discrepancy in catch rate between wood mice, and other small mammals, is due to them being at different stages of their inter-annual cycles, with wood mice being at the growth/peak phase, and other small mammals being at their low phase (Andreassen et al., 2021). As such, further study is required to investigate the changes in small mammal population densities and community composition over time at Boothby. The seasonal change in population densities also needs to be accounted for when making assumptions about site level small mammal presence. It is thus important that any future sampling occurs over multiple seasons so a comparison of site-specific seasonal change per year can be implemented.

One factor that has significant impacts on the inter-annual cycles of wood mice is the fluctuations in tree seed supply. These fluctuations can be heavily influenced by masting: the large synchronous drop of seeds (Pearse et al., 2021). It is likely that the high population of wood mice is at least partially impacted by the fact that there have been two consecutive mast years (NationalTrust.org, 2024). Although they are generalists with widespread food preferences, the fact that wood mice are granivores, means they have particular ties to tree seed abundance (Sunyer et al., 2016). Seed supply and wood mice population numbers are often positively correlated in the UK (Montgomery et al., 1991). Masting occurs in many tree species but in the UK it is typically found in European beech (Fagus sylvatica), and oak (Quercus spp.) (Koenig, 2021). Rodents are known to react strongly to masting as they typically have a fast life history and large litters (Zwolak, Bogdziewicz, and Rychlik, 2016). They are therefore able to capitalise on the increased seed availability caused by masting by producing more offspring (Zwolak, Bogdziewicz, and Rychlik, 2016).

4.3. Field voles

In 1995, the field vole UK population was estimated at 75 million, making it the most numerous UK mammal (Harris et al., 1995), and it is still thought to be the most abundant today (WoodlandTrust.org, 2024). Thus, they may have been expected to appear in greater numbers throughout the trapping grids, especially when compared to wood mice. However, many other studies of small mammals in arable land found field voles in very low numbers, especially compared to wood mice, despite their high estimated total population (Kotzageorgis and Mason, 2009, Tattersall et al., 2002).

Field voles were not very abundant amongst trapping grids and only 13 were caught in total. One explanation is that voles are more trap-shy than mice, making voles less likely to be caught in Longworth traps. A long-term mark-recapture study did find that in general, mice are more trappable than voles (Gurnell, 1982).

One additional explanation is that field voles are specialists (Gelling, Macdonald, and Mathews, 2007). Their preferred habitat type is rough grassland, although they do also utilise woodland margins and hedgerows with long grass (Alibhai and Gipps, 1991). Field margins also provide important habitat for field voles (Tattersall,1999). Their use of field margins was corroborated during this study as every field vole trapped was found in the hedgerows adjacent to field margins. Vegetation cover plays a large role in the habitat preference of field voles (Birney, Grant & Baird,1976). Their partiality for tall vegetation explains their association with the hedgerow and field margin regions of Boothby. However, it does not explain their complete absence from 2-year reverted fields, which seemingly provided ideal habitat for field voles. It may be the case that when at low densities, the field vole is confined to its favoured habitat type of regions with unbroken vegetation cover, which in Boothby's case is the hedgerows and margins. They may not be present in the reverted fields because the vegetation in these regions is still patchy in some places, and in order to stay in persistent cover, they need to remain close to the thick vegetation around the hedgerows and field margins.

More importantly, it is likely that voles occurred at a very low density, and their catch rate was indicative of this. Vole populations vary hugely year on year due to factors such as predation and food availability (Andreassen et al., 2021). The findings at Boothby are most likely representative of vole populations during their low phase. Thus, these findings represent a snapshot of the vole populations, and further monitoring is required to provide a more accurate representation of the population over time.

4.4. Bank voles

Bank voles were not caught in any traps for the duration of the study. This could mean they are not present in the region around Boothby, which seems unlikely given that they are widespread throughout the UK (Wilson et al., 2014). Previous research has found large yearly variation in bank vole density, with no bank voles being detected at sites where previously numerous individuals were caught in Longworth traps (Gorman, 2018). Much like field voles and other small mammals, stochastic variation and intrinsically unstable population dynamics can lead to significant changes in population density from year to year.

It is also known that bank voles are more specialist with their habitat requirements than wood mice and therefore their population could have been situated mainly in more preferred habitat, which is woodland (Bergstedt, 1966). Suitable woodland is capable of housing a large proportion of small mammals in agricultural regions (Moore et al., 2003), and Boothby does feature an area of ancient woodland. The home range size of bank voles is significantly smaller than that of wood mice (Bergstedt, 1966). Thus, although both species are often associated with woodland habitats, wood mice at Boothby may venture much further into adjacent fields, while bank voles remain exclusively in the wooded areas. Other studies of small mammals in agricultural regions have found that typically very few bank voles venture into crop fields (Tattersall et al., 2002), although they may do so when their populations are at high density (Kotzageorgis and Mason, 2009).

It could also be possible that inter-specific competition between wood mice and bank voles caused bank voles to avoid regions with high wood mouse density. Many small mammals counteract the impacts of interspecific competition via habitat switching (Schmidt, Manson, and Lewis, 2005). This could explain the low population numbers of vole and shrew species compared to the wood mouse, as the wood mouse was the dominant small mammal species at Boothby.

Whilst wood mice have been suggested to aggressively outcompete some species (notably harvest mice), competitive exclusion has not been found between bank voles and wood mice in similar studies (Wilson et al., 2014, Gelling, Macdonald, and Mathews, 2007, Broughton et al., 2014). There are significant differences in the diets of the two species. Wood mice are mainly granivorous, eating large quantities of seeds, whereas bank voles are mainly herbivorous, eating large quantities of leaves and fruit (Watts 1968). These dietary differences between the two species can lead to facilitation, enabling spatial coexistence (Wilson et al., 2014). The two species may also be able to coexist due to differences in peak activity levels, with wood mice being mainly nocturnal, whereas bank voles are active during the day (Wilson et al., 2014).

It is likely that the lack of bank vole catches, as with the wood mouse, is due to seasonal cycles in population dynamics. Significant variation has been found in bank vole population densities due to factors that do not seem to influence wood mouse populations, such as autumn temperatures and berry production in hedgerows (Featherstone, 2004, Poulton, 1994). Also, the timing of population increase due to masting differs between wood mice and bank voles. Masting can cause high autumn wood mouse densities, whereas the densities of bank voles are often greatest in the summer following a good seed crop (Mallorie and Flowerdew, 1994). Also, autumn temperatures may be another factor influencing bank vole population cycles that has less of an impact on wood mice (Featherstone, 2004). Long-term studies have found significant variations in the proportions of wood mice and bank voles between years (Featherstone., 2004).

4.5. Other small mammals

It is not completely unexpected that no yellow-necked mice were caught. They are typically found mostly in southern parts of England, and whilst there are sightings in the English north and midlands, they are much less common in these regions than they are in the south of England and in Wales (NBNAtlas.org, 2024) Also, they typically rely on mature woodland as a permanent habitat (Marsh and Harris, 2000) and radio-tracking studies have shown that they rarely forage in crop fields; therefore, if they are present in the surroundings at Boothby, they were unlikely to encounter trapping grids frequently (Kotzageorgis and Mason, 2009).

Few pygmy shrews were captured throughout the duration of the study. Annual cycles of pygmy shrew populations are very variable and can lead to very low densities. Therefore, it is possible that pygmy shrews at Boothby do occur in such low densities. It is also likely that pygmy shrews were underrepresented by capture samples. This is due to their small size, and potential to enter traps without triggering the trapping mechanism (Stromgren and Sulliven., 2014). As for common shrews, they are habitat generalists; therefore, it may have been expected that they encountered the trapping grids laid out in the crop fields. However, their home range sizes are much smaller than those of the wood mouse, making them less likely to encounter trapping grids in these regions (Tattersall et al., 2002).

Harvest mice numbers typically peak around autumn, and it is not surprising therefore that the only harvest mouse caught was trapped in November. A combination of the known poor detectability difficulty in being detected using conventional trapping methods, their lower population densities, and their more specialised habitat requirements (they are more commonly associated with reed beds than arable land), probably led to them being absent in most trapping sessions (Occhiuto et al., 2021). Harvest mice typically utilise the stalk-zone and are therefore much less likely to be found in Longworth traps placed on the ground (kettel, Perrow, and Reader, 2016), but they were not detected at all in ariel traps in this study. It is likely that their absence from the ariel traps is due to the stalk zone of the vegetation being insufficiently well developed in the early successional stages of the reverted fields at Boothby.

4.6. Field type

The 1-year reverted fields reflected the pattern of vegetation change that would be expected following agricultural cessation. The species found there are typical of the ruderal stage following one year from abandonment of cultivated fields (Churchfield, Hollier, and Brown, 1997). The 2-year reverted fields also reflected the vegetation that was typical of this stage of succession (Churchfield, Hollier, and Brown, 1997). These fields were largely dominated by grasses, and also featured large amounts of dead biomass. These changes in the vegetation observed in the first two years of the rewilding process at Boothby have the potential to impact on food availability, habitat structure, and predation risk for many small mammal species.

Field type when examined alone did not significantly impact wood mouse populations. Wood mice occupied every habitat type throughout the study, and there is no significant difference in the numbers found in any particular habitat type, when examined without considering seasonal effects in habitat numbers. This concurs with previous findings that wood mice are habitat generalists that can occupy the majority of English lowland habitats (Tattersall et al., 2002). Their distribution is not as closely linked to vegetative cover as some other UK small mammals (Wilson et al., 2014, Green, 1979). Habitat generalists such as wood mice can disperse between woodland patches with ease, regardless of the vegetative qualities of the dispersal habitats. Habitat specialists such as voles, are less inclined to disperse if habitat of suitable quality is not present (Gentili, Sigura, and Bonesi, 2014). This difference is likely explained at least partly by differences in predator avoidance mechanisms between small mammal species. Wood mice often rely on speed to outmanoeuvre predators, whereas voles prefer to remain hidden amongst dense vegetation (Jensen and Honess, 1995). Additionally, wood mice can avoid predators by being more active on dark nights (Sinclair, 1994); therefore, they are more likely than other species to utilise open habitats. Others have found wood mice populations to remain stable in crop fields (Janova et al., 2011).

Although wood mice were found in all habitat types, and there were no significant overall differences in abundance amongst fields at different stages of the reversion, clear differences were evident in the autumn; hence, the interaction between field type and time of year was significant. In October to December, fields which had been taken out of agricultural production for longer had on average larger mouse populations, with traps in active crop fields having the lowest probability of capturing an animal.

However, the interaction between field type and time of year was significant. This indicates that although there was no difference in wood mice presence between field types overall, there was a seasonal effect with regards to field use. During the winter and spring, wood mouse numbers were low across all habitat types, and the habitat associations were not as strong as they were in the late autumn. This could be evidence of low density but mobile populations, with individuals foraging over larger areas to compensate for low food availability. Grodzinski (1962) found movement patterns in rodents can change in response to low food availability. 1-year reverted fields did have slightly higher catch rates in this period, perhaps due to the high availability of seed-bearing herbaceous plants in these fields compared to alternative habitat types.

The impact of habitat types on small mammal communities being more prevalent in the autumn and winter months demonstrates the need for careful longitudinal monitoring. Especially when studying small mammals, which exhibit such drastic variation in population size over time.

What further changes in the mammal community are likely to occur during the next phase of rewilding at Boothby remains poorly understood, since projects of this kind are relatively new. But the arrival of woody plants is expected (Benayas et al., 2007). Harvest mice populations in particular may benefit from additional growth of long grasses (Occhiuto et al., 2021). The later successional stages will likely feature different proportions of small mammal species (Churchfield, Hollier, and Brown, 1997), as the landscape change caters to different species-specific niches. It is important to continuously monitor small mammal community compositional changes in response to these landscape alterations.

4.7. Time of day

Wood mice are mainly nocturnal and therefore are far more likely to be active at night (Wilson et al., 2014). This, together with the fact that the interval between checks was longer overnight, explains why far more were caught in the morning trap check than the evening (figure 5). However, wood mice are not completely inactive during the day, and have been observed in the daylight, possibly due to disruption of their regular behaviour due to trapping (Flowerdew, 2000). Hence, it stands to reason that although there were significantly more wood mice caught in the morning, some were caught in the evening. Interestingly, there was no interaction between field type and time of day. It might have been expected that mice in fields with low vegetative cover to be very exposed to diurnal predators, and hence bigger differences among field types could have occurred during the day than at night, but this was not evident during this study.

4.8. Trapping day

The trapping day had no significant impact on the likelihood of catching a wood mouse. There is research to suggest that rodents of any species are attracted to the scents associated with traps (Daly and Behrends, 1984). Therefore, if there were to be any kind of trend it would be expected that there would be a greater proportion of small mammals trapped in the later trapping days. Also, neophobia can occur, meaning that initially only trap-prone individuals will encounter the trapping grids and as the week progresses, trap-shy individuals become accustomed to traps and enter later in the trapping week (Tanton, 1965). However, these effects appear to have been unimportant in the current study, suggesting that the pre-baiting strategy employed was successful in allowing more of the population to become familiar with the traps before any catches were made.

4.9. Hedgerows

Excluding wood mice, the small mammals encountered in this study were almost always found in the hedgerows. Hedgerows have been previously found to be predictors of small mammal biomass in agricultural landscapes, with greater hedgerow area being associated with greater small mammal biomass (Gelling, Macdonald, and Mathews, 2007). This study found they also host greater small mammal diversity. This highlights their importance as a means of providing additional habitat heterogeneity across the site and acting as a refuge for species that are more likely to avoid the open fields. This is unsurprising as many studies have found vertical habitat heterogeneity to be a strong predictor of small mammal presence (Gorman, 2018). Hinsley and Bellamy (2000) found that small mammals in agricultural regions are often restricted to wooded areas and hedgerows. The only harvest mouse caught throughout the study period was in a hedgerow. It is likely that the vegetation in many of the fields was not tall enough to provide easy access to harvest mice which utilise a prehensile tail to move high above ground level (Occhiuto et al., 2021). Where the vegetation was tall enough, it may not have been dense enough for horizontal movement. In fact, one study has suggested that at times the hedgerow becomes the sole habitat for both wood mice and field voles (Boone and Tinklin, 1988). It has also been suggested that small mammals rarely use the open fields and are restricted to the hedgerows and field margins. Whilst this may be the case for other small mammals, this definitely was not the case for wood mice at Boothby.

Had there been greater variety in small mammals species that were frequently caught in the Longworth traps at Boothby, perhaps the contrast between species present within the hedges and fields would have been greater. During several months of trapping, more small mammals were caught in reverted fields than hedgerows (figure 4). The dominance of the wood mouse, which is significantly more of a habitat generalist (Díaz, Santos, and Tellería,

1999) than other small mammals, perhaps meant that the hedges at the time of trapping may have been less necessary as a valuable habitat refuge.

4.10. Audio

The results from the BTO pipeline provided fewer small terrestrial mammal vocalisations than expected. It was initially speculated that because the audio equipment was placed in the trapping grids, this close proximity to the small mammals that approach the trapping grids would guarantee numerous vocalisations recorded. However, small mammals represented a minority of the total species recorded (0.22% of high confidence recordings). There are numerous explanations for the low total number of small mammal calls. One possible explanation is that because the ultrasonic section of the BTO pipeline was initially designed to capture bat calls, the software is more aligned to detect these vocalisations, and less likely to recognise small mammal calls. In fact, the ability of the software to recognise small mammal calls was originally added due to numerous small mammal vocalisations being recorded as by-catch from bat survey data (Newson, Middleton, and Pearce, 2020). The idea of the BTO pipeline being more effective at detecting calls from other taxa is supported by the difference in mean confidence levels between taxa (table 4). The mean confidence level was lowest for small mammals, and further analysis of call variation in this group may be required.

Another explanation is that although they do vocalise at high frequencies, small mammals do so less often compared with other taxa that were detected. Orthoptera for example, are highly dependent on using sound to communicate, and although other forms of communication are used, it is usually as a means of complementing audio signalling (Greenfield, 1997). Birds also call extensively, and audio signalling is often their primary method of communication (Marler, Slabbekoorn, and Kroodsma, 2004). Bats call frequently and may utilise social calls to communicate information to conspecifics, as well as using echolocation to detect prey (Middleton, Froud and French, 2022). This reliance on echolocation to catch prey is likely to have led to high call frequency, which led to bats being detected at a much greater frequency than other taxa. Much like Orthoptera, bats may use visual and olfactory cues to communicate, but these can often be used in conjunction with ultrasonic communication (Chaverri, Ancillotto and Russo, 2018).

In comparison, little is known about small mammal ultrasonic calls, so it is hard to determine the level of importance as a means of communication compared with other methods such as olfaction and calls in the audible frequency range. It is known that voles are likely much more reliant on olfactory cues to communicate, and therefore call less frequently than other UK small mammals (Middleton, Newson and Pearce, 2023). This likely contributes to the low number of vole calls detected in this study; although data from the trapping grids suggested that they were also at low population densities. The frequency of calls is only one factor contributing to detection of small mammal calls. Call volume/amplitude could also be a contributing factor. Burrowing species such as wood mice will communicate more with kin when present in their burrows (Middleton, Newson and Pearce, 2023). These calls are much less likely to be detected by audio devices due to the obvious natural barrier that will dampen sound and prevent it from reaching the surface. More generally, terrestrial mammal communication may typically take place over shorter distances than other groups. When comparisons of detection distance of small mammal vocalisations have been made, the maximum detection distance was the brown rat at 9m (Middleton, Newson, and Pearce, 2023). In comparison, great-green bush cricket (*Tettigonia viridissima*) calls can be heard from over 100m (Christopher, 1997). Further research is needed to determine factors that influence detection rates of small mammal vocalisations.

November being the optimal month for all species calls is likely explained by seasonal effects. Bats in the UK typically hibernate from November; therefore, the drop off in detections past this time was expected (Bat Conservation Trust, 2024). Likewise, adult bush crickets in the UK die off during the winter (Wildlife Trust.org, 2023), leading to an expected drop-off in activity. There was also reduced call activity from all species in crop fields in the autumn/winter months compared with other tested habitat types. This was expected as the crop fields were recently drilled at this time, meaning there was negligible vegetative growth which likely caused a reduction in activity. Bare fields are usually avoided across wide ranging taxa in favour of preferred habitat (Tucker, 1992).

The fact that both the Audiomoths, and the Song Meter Mini Bat 2 were available, allowed for a comparison of their recording abilities. The Audiomoth is a much lower cost device (approximately £500 cheaper per unit than the Mini Bat), and was designed primarily to detect birds at frequencies audible to humans. Therefore, it was expected that it would not function as well at recording high quality high frequency data. Whilst there were differences in the number of detections when Audiomoths and Mini Bat recorders were placed in the same location (table 5), there were not as many differences as expected. Thus, the Audiomoth may be suitable at providing a lower cost method of gathering small mammal audio data, although further like-for-like comparisons in different habitats with different communities are required to be more certain of the differences in recording device capabilities.

The analysis of calls revealed several mammal species that were not caught in Longworth traps. These species are as follows: brown rat, European water vole, yellow-necked mouse, and bank vole. For some of these species the reasons behind their vocalisations in the fields, but absence from the Longworth traps is very apparent. For example, the brown rat is common within the agricultural landscape of the UK (Hasni, 2008). Therefore, their presence within the fields at Boothby is expected. Their absence from the Longworth traps is due to their size. The average size of the brown rat is much larger than that of any of the other small mammal species caught in the study (Natynczuk, Macdonald, and Tattersal., 1995), and therefore it is highly plausible that brown rats were present in the fields but simply unable to fit inside the Longworth traps. Body size may also explain why European water voles were not caught in the Longworth traps. They reach sexual maturity at around 112g (Moorhouse, Gelling, and Macdonald, 2008), which makes them significantly larger than wood mice. These recordings of larger species provide additional motivation for the use of audio recording alongside Longworth traps, as it widens the pool of detectable species within a study. All yellow-necked mouse recordings were given a low confidence level by the BTO pipeline. Thus, it is possible that these calls were actually misidentified wood mice. The two species share similar call characteristics (Newson, Middleton, and Pearce, 2020).

The differences in confidence levels proved a useful tool for questioning unlikely recordings. It is highly probable that filtering for calls of high confidence (greater than 0.5) gives a much more accurate picture of the soundscape at Boothby.

Further confidence in the output of the BTO pipeline species identification results can be gained via analysis of ultrasonic call structure in spectrograms. The wood mouse spectrograms showed a clear distinction between high and low confidence calls, with the higher confidence calls having similar call features to reference wood mouse calls.

Spectrogram data also failed to support the validity of the hazel dormouse audio output. Dormouse detections from the audio pipeline mainly had low confidence scores, and the spectrograms that were examined did not bear enough similarities to reference hazel dormouse spectrograms to verify a positive identification (figure 13 and 14). This corroborates with the fact that the hazel dormouse isn't typically found in the region around Boothby (nbnatlas.org,2025). Further investigation both with live trapping, and audio recording may be necessary to verify the presence of rarer species such as the hazel dormouse at Boothby.

A limitation of the spectrogram analysis is that it is very labour intensive. Due to the time taken to observe each spectrogram, the majority of calls remained unchecked. There were also issues with manually identifying certain species via spectrogram due to call variation. Many small mammals do not have one specific call, but have a range of calls, each producing different visual imagery on the spectrogram. Despite these limitations, spectrogram analysis remains a viable option for corroborating calls detected by rare, or unlikely species.

Further research is required with audio equipment such as Audiomoths, to investigate if there is any difference in audio calls by small mammals and other species over time at Boothby.

4.11. Longworth and Audiomoth comparison

Whilst the seasonal effects were present in the audio data when analysing calls from all species, the trend was not consistent when analysing small terrestrial mammal calls in isolation, where the pattern of greater species detections in the autumn was only found in 2-year reverted fields. The lack of significant seasonal effects with the audio recording data, where there were significant effects with the Longworth trapping data, means there was a misalignment between the two recording methods.

The Longworth trapping data does not give a complete picture of the small mammal community and many differences between species e.g. size differences (Grant, 1970) have been found to bias the capture rate in previous studies. Therefore, the audio data has value in providing evidence of further small mammal species presence that's separate from the live trapping data. However, the confidence levels of the small mammal recordings being the lowest of all species groups presents issues with the tangibility of these data. Calls with a

BTO pipeline confidence level below 0.5 have lower value and cannot be relied upon to give an accurate picture of the local small mammal soundscape. The verification of recordings via spectrogram analysis is one way to remedy this issue, however, due to the laboriousness of this process, it isn't feasible for all recordings.

4.12. Camera traps

The data from the camera traps provided an idea of the baseline levels of the larger animals at Boothby. Although the number of images recorded was small, it appears that deer are the most common mammals present at the site which were not detected in the traps or audio recordings. This may be because deer are large and move in herds and are therefore more easily captured by camera traps than other animals. They are also highly mobile (Barton, 2023), and were frequently seen by eye, crossing the crop fields at all stages of growth, thus it is unlikely that these fields represent the same barrier to dispersal with deer as they do with certain small mammals. The 1-year reverted fields contained the highest number of mammals caught in camera trap images, owing partially due to the large number of European hares captured in these fields. The hares likely require the herbaceous plants growing in these fields as a food source (Blay,1989). It is also likely that their presence, along with the presence of many other mammals, is underrepresented in the 2-year reverted fields due to long vegetation masking their presence.

Aside from the generally low number of detections, the main limitation of the camera trapping was that not every image produced was of a quality high enough to obtain a reasonable species identification. Many images were taken in the dark, which can provide a clear image if the animal in question is close to the camera; however, when the animal is far from the camera, the image only captures eyeshine, with no other distinguishing features. This led to a high number of unidentified animals appearing amongst the camera trap images. Further monitoring with camera traps over time may capture the potential shifts in large mammal communities, especially as new species are introduced to the site.

4.13. Limitations

With regards to the data analysis, despite grid being used as a random effect, there were potential issues with treating each trap as a replicate, as the presence or absence of any given small mammal is not truly independent of the presence or absence of any other small mammal. To remedy this, a less powerful, but more reliable grid level analysis was run on the captures per hour and minimum number alive data which aligned with the results of the capture probability analysis.

5. Conclusion

Long term monitoring in rewilding projects provides valuable data that can be used to inform management decisions on a local scale, as well as providing information that can go some way towards remedying the shortage of long-term conservation monitoring data (Willis et al., 2007). Before this project there were very little data available about small mammal communities at Boothby. The baseline data provided by this study provides an account of the small mammal communities in their current state, and also remains valuable as a reference point for when further successional changes occur at Boothby. The seasonal differences in small mammal abundance, in Longworth traps and audio recording data, provide a greater picture of small mammal communities at Boothby over time, and are an important baseline for future research. The audio data supported the idea that certain species may be underrepresented by Longworth trapping alone, and that newer technologies can be utilised effectively alongside traditional monitoring methods. However, audio recording remains less tangible, and additional time may be needed to verify questionable audio recordings via methods such as spectrogram analysis. Live trapping, and audio data supported the idea that withdrawal from cultivation in agricultural landscapes, can provide benefits to small mammal communities during certain seasons. Over the following years mammals will continue to be monitored at Boothby, and any future data gathered will be compared to these data to give an idea of shifting mammal communities on site.

6. Ethical statement

The work complied with relevant local and national animal welfare legislation and was approved by the University of Nottingham Animal Welfare and Ethical Review Body. The study was covered by a government shrew licence (GL01). Trap checking frequency complied with Government animal welfare legislation.

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