The Effects of Conservation Grazing Management on Habitat Structure and Reptile Assemblage of Complex Grassland-Heathland Systems

A technical report prepared from an MRes Wildlife Conservation project, Marwell Wildlife & University of Southampton

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i. Summary for the Layman

The way land is used is continually changing, leading to loss of important wildlife habitat, resulting in declines in wildlife populations and impairing the environment's ability to function. Lowland heath has been particularly badly affected, declining 84% since the 1800's. This is partially due to incorrect management, allowing more dominant vegetation to take over which has affected many species relying on grass-heathland habitat, such as reptiles. Reptiles have very specific vegetation requirements, such as needing open areas to bask in and dense areas to shelter. One method of maintaining heathland systems and providing this habitat for reptiles is through grazing management.

This study aimed to investigate how the level of grazing intensity changes habitat and thereby determines the species of reptiles present in the surrounding area. This was achieved by carrying out reptile surveys across three sites in Northern Hampshire and Surrey, in south-east England, and measuring vegetation across the sites to ascertain the grazing intensity. The variation in grazing intensity across the sites was shown, and relationships between the number of reptiles and the intensity of grazing were identified. The sites were divided into equal patches, and the presence of reptiles within each patch was estimated based on vegetation measurements.

The study found that implementing grazing as a management tool reduces vegetation height and increases the range of vegetation types, thereby creating suitable habitat for many reptile species. Grazing systems that rotate cattle between fields did not seem to be beneficial, neither for reptiles nor heathland, as it results in some areas being very heavily grazed and some left over-grown. However, having a low density of free-roaming grazers was beneficial for both reptiles and heathland. Recommendations for management systems were made for each site. It is important to note that implementing grazing as a management tool is not suitable for all situations, and any decisions should be carefully considered and planned out before action is taken. Future work should focus on restoring landscapes in between sites, which contain little suitable habitat. Grazing could help to achieve this goal, allowing reptiles and other animals to move between areas and enhance their populations.

ii. Abstract

Land-use change is the leading cause of habitat loss, resulting in loss of biodiversity and ecosystem function. Lowland heath in particular has declined 84% since the 1800’s, partially due to inappropriate management not preserving the intermediate successional stage required for heathland to thrive. The loss of grassland-heathland systems has greatly affected species relying on this habitat, such as reptiles. Reptiles rely on a number of important micro-habitat features, which require a high level of variation in structural complexity. One method of preventing succession to woody vegetation and increasing heterogeneity is through controlled grazing management.

The aim of this study was to assess how grazing intensity drives patch-level reptile assemblage by modifying habitat structure, by undertaking reptile surveys across three sites in Northern Hampshire and Surrey, in south-east England, and measuring various vegetation parameters as a proxy for grazing intensity. Principal component analysis showed patterns and variation in grazing intensity between the sites, and relationships between abundance and grazing intensity were identified using generalised linear models. Occupancy models were used to estimate patch occupancy based on these parameters.

The study found that conservation grazing reduces sward height and increases structural complexity of vegetation, creating suitable habitat for many reptile species. Rotational grazing systems do not seem beneficial for heathland restoration or reptile abundance due to high disparity between grazed and un-grazed areas, however free-roaming low-intensity stocking systems are suitable. Management recommendations are made, though it is important to note grazing is not applicable for all situations and decisions should be carefully considered and modelled before any action is taken. Future work should focus on restoring the fragmented landscape to increase connectivity, possibly by implementing grazing.

Key Words: reptile, assemblage, grazing, management, heathland, grassland, patch occupancy

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iii. Abbreviations

AIC	Akaike’s Information Criterion
ARC	Amphibian and Reptile Conservation Group
EEL	Eelmoor Marsh
FOX	Foxlease & Ancells Meadows
FLA	Frensham Flashes
GIS	Geographic Information System
GLM	Generalised Linear Model
GLMM	Generalised Linear Mixed-Effect Model
HIWWT	Hampshire and Isle of Wight Wildlife Trust
HLS	Higher Level Stewardship
MoD	Ministry of Defence
NARRS	National Amphibian and Reptile Recording Scheme
PCA	Principal Component Analysis
RDI	Raw data, Descriptive and Inferential Statistic Plot
SARG	Surrey Amphibian and Reptile Group
SSSI	Site of Special Scientific Interest
SINC	Special Importance for Nature Conservation
SPA	Special Protection Area

1. INTRODUCTION

1.1 Habitat Loss

Land-use change leading to habitat destruction is the main driver of species extinction (Tilman *et al.*, 2001), leading to biodiversity loss (Dobson *et al.*, 2006), reduction in functional diversity and therefore ecosystem function (Flynn *et al.*, 2009). Habitat conversion is 8 times higher than protection in temperate areas (Hoekstra *et al.*, 2005) largely for agriculture, leading to massive habitat loss across Europe (Krauss *et al.*, 2010). Should this dependence on agriculture continue, 10^9 hectares of natural ecosystems will be converted by 2050 (Tilman *et al.*, 2001). By 2100, vegetation cover could decrease by 58%, causing up to 21,000 species extinctions (Jantz *et al.*, 2015).

Habitat destruction leads to fragmentation, often defined as “landscape-scale processes involving habitat loss and breaking apart of habitat” (Fahrig, 2003), creating biodiversity ‘hotspots’ of clumped distributions containing threatened species (Brooks *et al.*, 2002). For example, more than 50% of endangered species are found in less than 2% of US land area (Dobson *et al.*, 1997). These fragments can be considered ‘islands’ within larger, unsuitable habitat (Pimm & Raven, 2000), allowing the application of Island Biogeography Theory; larger fragments sustain higher species diversity and populations (MacArthur & Wilson, 1967). Fragments are therefore heavily reliant on size, habitat quality and connectivity to support colonisation and reduce extinction (Jantz *et al.*, 2015) by allowing movement of metapopulations (Hanski, 1998). Habitat destruction could be reduced with wide-scale preservation and lessening the impact of land-use activities (Jantz *et al.*, 2015).

Heathlands in particular have been hugely affected by increased agriculture, increased timber production and growing human populations (Moore, 1962; Critchley *et al.*, 2013). British heathlands are better developed than in Europe due to the oceanic climate and higher humidity, with three main types: lowland, upland and wet heath. Lowland heath is especially important and has declined rapidly, an estimated 84% since the 1800’s (Jackson and McLeod 2008) and is highly fragmented with multiple small patches instead of the original continuous habitat (Moore, 1962). Hampshire, in southern England, has the largest proportion of the country’s lowland heath and 14% of that in Europe (Groves *et al.*, 2012).

One cause of heathland decline is the lack of management to prevent succession into woody vegetation (Harrison, 1976). Although a strong coloniser, when left unmanaged heather gives way to shrubs and other vegetation (Gimingham, 1992), increasing homogeneity and reducing biodiversity of many taxa (Fitch, 2006). If intermediate vegetation is maintained, more species can thrive (Pielou, 1966). Furthermore, heathland restoration is facing many constraints, including increased nitrogen deposits and fertilisers causing *Calluna vulgaris*, one of the main heather species along with *Erica tetaralix* (Farrell, 1989), to give way to grass species such as *Molinia caerulea* (Purple moor grass) (Heil and Diemont 1983; Heil and Bruggink 1987) which is difficult to control (Ross et al. 2003; Marrs et al. 2004). This has hindered development of semi-natural habitat from agricultural land, with heathland development hugely decreasing throughout the 20th century (Ejrnæs *et al.*, 2008).

1.2 Grazing Management

This intermediate successional state is important for many ground-dwelling species, including reptiles, and was previously maintained by herbivores that are no longer present. It now requires human intervention (Webb, 1998) such as introducing livestock as grazers. This balance of wildlife and livestock is ever more important as agriculture intensifies, especially as the introduction of grazing on heathland is set-aback by the lack of information on methods and effects (Bullock & Pakeman, 1997).

Controlled conservation grazing increases soil quality and allows colonisation of new plants (Read *et al.*, 2016), as well as aiding nutrient cycling, increasing spatial heterogeneity and plant resilience (Wilkie, 2013). However, it requires long-term planning and monitoring with a deep understanding of the ecosystem and how populations will be affected by change (Gordon *et al.*, 2004). High stocking density can lead to over-grazing, degrading grasslands (Mysterud, 2006), increasing the amount of bare soil and reducing the variation in vegetation across the landscape (Augustine *et al.*, 2012), leading to biodiversity loss (Kay *et al.*, 2017). Moderate grazing is considered most beneficial for conservation (Howland *et al.*, 2014), with free-roaming grazers maintaining patches of varying vegetation and habitat (Bullock & Pakeman, 1997).

1.3 Reptiles

1.3.1 Population Decline

Reptiles are area-sensitive, declining with habitat size (Hager, 1998) and land-use change, such as agriculture (Atauri & de Lucio, 2001). This, along with increased urbanisation and disturbance, is resulting in habitat loss, the main driver of reptile decline (Cox & Temple, 2009) and creating fragmented populations. This sensitivity can be used to measure environmental conditions, and as reptiles show long-term population declines (Cox & Temple, 2009) they are suitable bioindicators of habitat change (Morris *et al.*, 2008; Lindenmayer & Likens, 2011). Habitat restoration through management is required, however general land management does not consider microhabitat requirements (Edgar *et al.,* 2010). There is little research on wide-scale management for reptile communities, as ideal habitat likely includes a huge variety of different factors that are largely unknown (Marum, 2006, *unpublished*). In the UK, areas with high reptile abundance can be marked as ‘interest features’ for protection, such as sites of special scientific interest (SSSI). More than 50 SSSIs are designated due to reptile populations, however even in protected areas care is not always taken to ensure management techniques are suitable (Edgar *et al.*, 2010).

Globally, 35% of reptile species are threatened by extinction (IUCN, 2017) and in Europe, 41.7% of species are declining, with the most threatened families being *Viperidae* and *Lacertidae*. There are six native UK species; slow worm (*Anguis fragilis*), common lizard (*Zootoca vivipara*), sand lizard (*Lacerta agilis*), grass snake (*Natrix natrix*), European adder (*Vipera berus*) and smooth snake (*Coronella austriaca*). Most are found across a range of habitat including heathland, open woodland and grassland (Edgar *et al.*, 2010; Stumpel & Van der Werf, 2012), though heathlands are particularly important for species relying on mature heather, particularly when combined with *M. caerulea.* There are few instances of all six species occurring together in other habitats (Spellerberg, 1989). Although extinction risks are ‘Least Concern’ (Agasyan *et al.*, 2009; Agasyan *et al.*, 2010a; b; European Reptile and Amphibian Specialist Group, 1996; Crnobrnja Isailovic *et al.*, 2009a; b), distributions are varied and all suffer from habitat loss (Edgar *et al.*, 2010; Inns, 2011). All are protected under the 1981 Wildlife and Countryside Act against killing, injuring, taking and trading (JNCC, 1981), and are part of the ‘Post-2010 Biodiversity Framework’ (JNCC & DEFRA, 2012), focusing on integrating species with habitat restoration.

1.3.2 General Habitat Requirements

Although some have specific needs, UK reptiles require similar microhabitat features (Webb *et al.*, 2010). As ectothermic animals, all require open habitat to bask. South-facing slopes are preferred, with a mix of open ground and dense cover to alternate and control body heat. Nearby shelter from predators and adverse weather is required, provided by deep vegetation pockets such as prickly gorses and deep mosses and lichen to cool down (Edgar *et al.*, 2010). If vegetation is too short or sparse, predation is higher and reptile populations decrease (Kreulen, 1979; Sato *et al.*, 2014), however highly dense vegetation inhibits basking. Vegetation tussocks also provide hibernation sites. Habitat edges of woodland, hedgerows and grass-scrub interfaces create ‘transitional zones’, allowing reptiles to move between and across sites, increasing connectivity. As most species have short dispersal, they require large areas of continuous habitat or patches linked by favourable habitat to move through. For species with separate hibernation and breeding sites, connectivity is especially important to form larger meta-populations which reduce inbreeding, retain genetic diversity and allow re-colonisation of locally extinct areas (Edgar *et al.*, 2010).

Across all habitats, vegetation drives reptile assemblage. Mowing, cutting and grazing prevent natural succession and maintain varying degrees of intermediate vegetation, providing the mosaic of ages, heights and types, required by reptiles (Edgar *et al.*, 2010). Growth of trees and shrubs, agricultural abandonment and succession removes open land and leads to over-shading (Cox & Temple, 2009), leading to degradation of key micro-habitat features. Wide-scale habitat management for reptile communities must incorporate these general needs, which also benefits other taxa saving time and costs (Edgar *et al.*, 2010). Specific management aimed at rarer reptile species will also benefit those relying on similar features (Inns, 2011).

Grazing is a widely applicable way of retaining intermediate vegetation (Gimingham, 1992), however there is contradictory research on responses of reptile populations and which treatments are most successful. Reptile densities vary with litter cover, sward height and percentage cover (Marum, 2006, *unpublished*), increasing with habitat complexity (Santos *et al.*, 2016). This suggests they benefit from conservation grazing, as abundance increases with livestock density. They are not affected by rotational grazing systems (Dorrough *et al.*, 2012), but free-roaming grazers are thought to be beneficial as they create ideal heterogeneous conditions through high and low intensity grazing to suit many species (Howland *et al.*, 2014). A range of grazing approaches should be applied to suit as many species as possible (Kay *et al.*, 2017). Conversely, research shows even light grazing can increase local extinctions (Larson, 2014), with over-grazing causing fragmentation and conversion from heathland to grassland (Hester & Baillie, 1998) and cattle grazing reducing reproductive rate of many UK reptiles (Reading and Jofre 2015; 2016). Other studies show no impacts of grazing on abundance (Beever & Brussard, 2004).

1.3.3 Species Specific Habitat Requirements

1.3.3.1 Slow Worms (Anguis fragilis)

Slow worms are generalists, tolerating lower diversity of vegetation than other species (Edgar *et al.*, 2010). They bask less openly than other lizards, especially in very warm weather (Inns, 2011), and are mostly fossorial, staying underground and within vegetation. They require loose soil for burrowing, and avoid very wet or very dry areas (Edgar *et al.*, 2010). *A. fragilis* are mainly diurnal, but sometimes forage on warm evenings for prey such as slugs and worms (Inns, 2011). They have small home ranges of several hundred square metres (Edgar *et al.*, 2010).

1.3.3.2 Common Lizards (Zootoca vivipara)

Common lizards are also generalists, thriving in habitats with sufficient sunlight, structure and cover. They tolerate a range of conditions, preferring damp environments but are largely absent from countryside dominated with agriculture, dense woodland or heavily grazed grassland due to structural uniformity and lack of prey. Their wide habitat tolerance enables a large range of prey (Edgar *et al.*, 2010) such as spiders and small insects. *Z. vivipara* operate at lower temperatures, spending less time basking than sand lizards and often basking with other individuals (Inns, 2011). Ranges are limited to a few tens of meters, however juveniles can rapidly colonise nearby habitat if available (Edgar *et al.*, 2010).

1.3.3.3 Sand Lizards (Lacerta agilis)

Sand lizards are rare in the UK and confined to limited sites, so are strictly protected with licensing required for some activities (ARC Trust, n.d.). They have highly specific microhabitat requirements, only found on dry, lowland heath (holding 95% of the UK population) and coastal sand dunes. Key requirements can be met by many habitats, suggesting plant species are less important than the structural conditions they provide, but these conditions are lacking. As the species is on the edge of its European range, they require especially warm habitats. Breeding adults require mature dry, sandy heathland and good ground cover of bryophytes and lichens. Warm, sandy ground with nearby dense cover is essential for egg-laying, however will be avoided if heavily grazed (Edgar *et al.*, 2010). *L. agilis* bask in open areas with nearby mire and water in hot weather, returning to the same site for several years. They hunt on a variety of invertebrates, including spiders, grasshoppers and even bees. They are the only UK reptile species to dig their own holes for hibernation (Inns, 2011). Home ranges are only a few hundred square meters and can overlap; while not territorial, males have dominance hierarchies when competing for females. In favourable habitat, individuals can be highly sedentary and rarely venture into unsuitable habitat (Edgar *et al.*, 2010). Suitable management involves controlling vegetation to reduce shading, increasing bare sand and preserving structural diversity of vegetation (Edgar & Bird, 2006).

1.3.3.4 Grass Snakes (Natrix natrix)

Although generalists, grass snakes require ponds and lakes for fish and amphibian prey but can travel far from water. Amphibian decline has affected populations (Inns, 2011), however lack of monitoring and high mobility makes it difficult to determine their status. They require decomposing matter for egg-laying, in communal areas that females return to every year. The loss of these sites has caused population declines, so conservation efforts are focused on pond management to increase prey and restoring breeding sites. *N. natrix* are mobile, migrating several kilometres between breeding and hibernation sites, so are less dependent on structural diversity at single sites and may migrate through poor quality habitat to reach favoured areas (Edgar *et al.*, 2010). They hibernate underground and in deep litter, sometimes sharing with adders, and disperse rapidly after emergence (Inns, 2011).

1.3.3.5 European Adders (Vipera berus)

Adders are found in most open habitats, however require ‘Adder banks’, which are underground hibernacula that need to be dry, with dense vegetation cover and nearby basking sites (McInerny, 2014). They often utilise disused mammal burrows to hibernate communally, returning every year; if these are damaged or removed, for example through unsuitable land management, it can hugely affect populations (Edgar *et al.*, 2010). They prefer chalk or sandy soil and are rarely found in areas with intense agriculture. With more restricted habitat preferences, adders are less resistant to human-induced changes leading to local extinctions (Inns, 2011), so habitat connectivity is particularly important. *V. berus* feed mainly on small mammals. They have distinct seasonal migrations between spring breeding and wetter, summer foraging sites, which can be up to two kilometres apart, requiring high connectivity with adequate vegetation cover (Edgar *et al.*, 2010).

1.3.3.6 Smooth Snakes (Coronella austriaca)

Smooth snakes are a highly localised species, confined to specific habitats and are therefore strictly protected by law with licensing required for some activities (ARC Trust n.d.). They are secretive, staying in mature heathland with deep layers of bryophytes and lichens. They prefer a lower body temperature than other reptiles, so are less likely to bask in the open, making population estimates difficult (Edgar *et al.*, 2010). They often occur with sand lizards and benefit from similar management regimes. They feed mainly on lizards, including slow worms, and sometimes small mammals (Inns, 2011). *C. austriaca* do not undertake long seasonal migrations, with ranges rarely exceeding 100m, making them especially sensitive to habitat loss (Edgar *et al.*, 2010).

1.4 Aim and Objectives

Aim

Assess how various grazing regimes implementing different grazing intensities modify habitat structure and drive patch-level reptile assemblage within complex grassland-heathland systems.

Objectives

1. Compare reptile assemblage and vegetation parameters as a proxy for grazing intensity between management systems.
2. Determine how grazing intensity drives reptile assemblage through modification of habitat structure.
3. Quantify patch occupancy of reptiles based on grazing intensity parameters.

1.5 Hypotheses

Null hypotheses:

1. Different grazing management systems will support equal reptile abundance
2. Different grazing management systems will support equal species proportions.
3. Proportion of reptiles in each habitat type will not vary from proportion of tins.
4. There will be no variation in vegetation height, percentage ground cover, vegetation structure or canopy cover between grazing management systems.
5. Different grazing management systems will support equal reptile prey abundance.
6. There will be no relationship between reptile abundance and grazing intensity or weather conditions.
7. All areas, regardless of grazing intensity, will show equal patch occupancy.

Alternate hypotheses:

1. Reptile abundance will vary with grazing management systems.
2. Species proportions will vary with grazing management systems.
3. Proportion of reptiles in each habitat type will vary from proportion of tins.
4. Vegetation height, percentage ground cover, vegetation structure and canopy cover will vary with grazing management systems.
5. Reptile prey abundance will vary with grazing management systems.
6. There will be a relationship between reptile abundance and grazing intensity and weather conditions.
7. Areas with varying grazing intensity will show different patch occupancy.

2. METHODOLOGY

2.1 Study Area

The study was carried out across three sites in Northern Hampshire and Surrey (*fig 1*). All are SSSI's designated under Natural England.

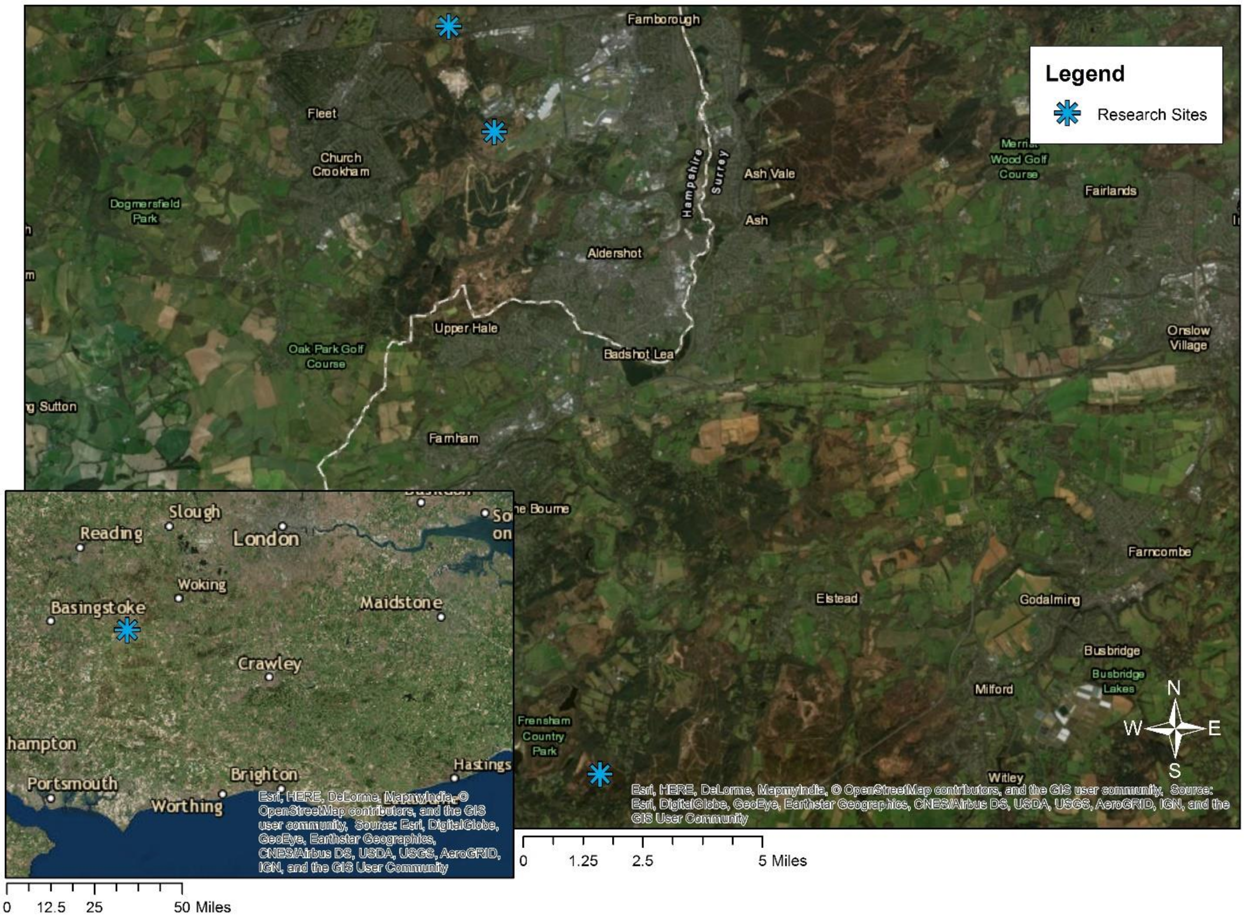

Figure 1: locations of three sites visited for this study: Eelmoor Marsh (central), Foxlease and Ancells Meadows (top) and Frensham Flashes (bottom). The inset map shows their location in south-east England, UK. Made using ArcMap 10.4.1.

2.1.1 Eelmoor Marsh

Eelmoor Marsh SSSI (78.72 ha) was designated in 1978 due to rich flora species and high invertebrate diversity present in the rare grass-heath system (Natural England, 1978). It is split into three units, which are all classed as ‘favourable’ condition by Natural England (Natural England, 2014). The latest national vegetation classification survey results (*see appendix 1, fig. 1*) can be broadly categorised into five types; grassland (acid, neutral, and *M. caerulea* dominated), dry heathland, wet heathland, woodland and mire, along with some aquatic features. The heathland, mire and grassland are considered valued habitats (Hall *et al.*, 2015). It has been part of the Thames Basin Heaths Special Protection Area (SPA) since 2005 and contains the Farnborough Airfield Site of Importance for Nature Conservation (SINC) established in 2000 (Rushmoor Borough Council, 2017). The land has no public access and is within a securely fenced area, however staff from the nearby technology park are permitted during working hours on weekdays.

The site is managed by Marwell Wildlife and QinetiQ estates, and is split into 24 management compartments, separated by physical features and vegetation communities (*see appendix 1, fig. 2*). Vegetation is managed by free-roaming grazers, with four Przewalski’s horses and six Highland Cattle on the south side, and three cattle on the north.

Eelmoor contains five of the six native UK reptile species (*table 1*). The four most common species are regularly found across most compartments (*see appendix 1, table 1*), and *L. agilis* populations were reintroduced in 2017 (expected to be found mostly in compartments one and two). All are expected to be breeding and hibernating on site (Hutchins, 2004). Although reptiles likely benefit from general habitat maintenance (Edgar & Bird, 2006), there is no specific management here for valued species. Systematic surveys are carried out, and extensive monitoring has been proposed for 2020 (Hall *et al.*, 2015). Populations were found to be declining, however this was based on small sample sizes so further study is required. All habitats present are suitable for the reptile species (Langham, 2015).

2.1.2 Foxlease and Ancells Meadows

Foxlease and Ancells Meadows SSSI (70.54 ha) was designated in 1988 due to the range of habitat forming a mosaic of plant diversity rarely seen in southern England, and the extensive network of ponds and ditches harbouring many rare species (Natural England, 1988). Foxlease and Ancells Meadows are separate sites but are considered one designation under Natural England. They consist of nine units, all classed as ‘Unfavourable – Recovering’ by Natural England apart from one, classed as ‘Favourable’ (Natural England, 2013), but this unit is not included in the study as there is no grazing present.

Foxlease and Ancells Meadows are owned by the Ministry of Defence (MoD), used as training areas and managed by Hampshire and Isle of Wight Wildlife Trust (HIWWT). Public are permitted onto the land for recreational use, such as dog walking. The area is fenced into many separate fields (*appendix 2, figure 3*) enabling rotational grazing under the higher-level stewardship (HLS) scheme, aiming increase habitat condition to ‘Favourable’ or ‘Recovering’ (Natural England 2011a; 2009). This study focused only on the fields where grazing is implemented (*appendix 2, figure 3*). Foxlease north contains two herds, the first consisting of four British White and one Hereford cow, and the second of 27 British White cows, one Hereford cow and one Hereford Bull. Foxlease south contains one herd of 17 Sussex Red cows, and Ancells Meadows contains one herd of 19 Shetland cows and one British White bull (Richard Hennessey, pers. comm., 2018).

Although this SSSI contains the four most common British native reptile species (*table 1*), there is no mention of reptiles as indicators of success in the HLS prescription (Natural England 2011a; 2009) and no specific management aimed at their conservation. There is little data on reptile abundance as previous to this study there were no reptile tins present, however adders have been seen around Field F, and grass snakes around Ancells Pond (Richard Hennessey, pers. comm. 2018).

2.1.3 Frensham Flashes

Frensham Flashes (110.48 ha) is part of the SSSI Thursley, Hankley and Frensham Commons designated in 1955. There are two units covering the Flashes, both classified as ‘Favourable’. The commons were designated due to their incredibly high-quality heathland habitat, providing an area of national importance for birds, all six native reptile species, four amphibian species and many invertebrates. There are also many areas of open water and bog (Natural England 2011b).

The land is owned and managed by Waverley Borough Council. Historically, the western area was grazed (Darren Hill, pers. comm. 2018), but was abandoned in 2001 after foot and mouth outbreak. Currently there is no grazing management in place, instead it is rotationally scraped and turf is removed to create bare soil important for many species to the standard set by the HLS scheme. This work is carried out by the Amphibian and Reptile Conservation Trust (ARC) under an agreement with the council to create and preserve vital habitat for reptiles, that runs until 2020. Heather is also mown to create bare ground, along with bracken removal to minimise damage to ground nesting birds and reptiles. Grazing management has been suggested to remove dense swards of *Molinia caerulea*, to ensure it does not cover more than 60% of wet heath areas. The land is public access, and public engagement with nature is of high importance within the management of this site. Dog walking can have negative impacts on many species including reptiles, so leads are required from March-August (Waverley Borough Council, 2017).

All six native reptile species are present here (*table 1*). It is possible that *C. austriaca* populations were enhanced by reintroductions to an adjacent site (Steve Langham, pers. comm. 2018). Regular surveys are undertaken by Surrey Amphibian and Reptile Group (SARG).

Table 1: summary of reptile species present at each survey site.

Site	Slow Worm	Common Lizard	Sand Lizard	Grass Snake	European Adder	Smooth Snake
Eelmoor Marsh	√	√	√	√	√	
Foxlease and Ancells Meadows	√	√		√	√	
Frensham Flashes	√	√	√	√	√	√

2.2 Study Period

Two pilot surveys were carried out at Eelmoor to ensure reptiles had emerged from hibernation. The study was undertaken during late spring of 2018 (*for schedule, see appendix 3, table 2*). Reptiles are encountered more during these months due to cooler weather leading to longer basking, and as reptiles are actively looking for mates (*fig. 2*) they are more resilient to disturbance (Spence-Bailey *et al*. 2010; Inns 2011). Surveys were carried out weekly for 10 weeks, totalling 8 visits per site.

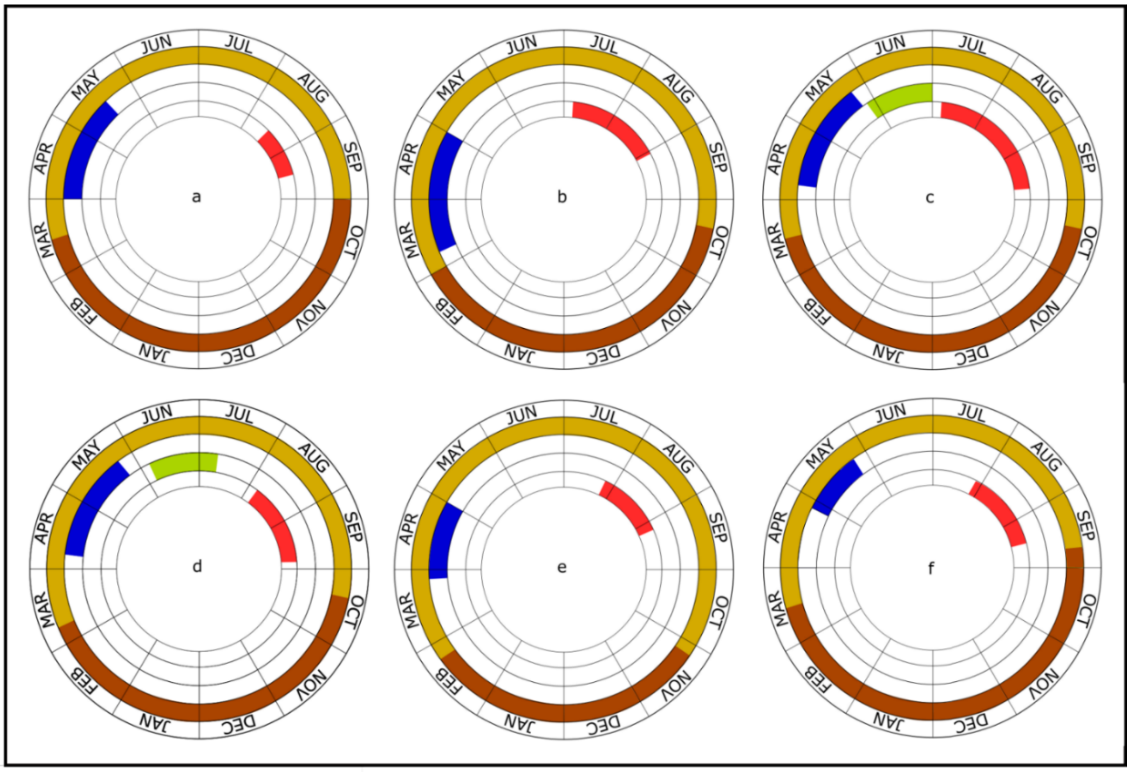

Figure 2: annual activity clocks of six native UK reptile species, showing active periods when main life-cycle stages occur, hibernation period, time of mating, egg laying and birth. a) *A. fragilis* b) *Z. vivipara* c) *L. agilis* d) *N. natrix* e) *V. berum* f) *C. austriaca.* Adapted from Inns, 2011. N.B: some species are viviparous and give birth to live young.

2.3 Reptile Surveys

The National Amphibian and Reptile Recording Scheme (NARRS) provides guidelines on suitable surveying methods. Using metal reptile tins is a widely accepted method, as reptiles bask on top or shelter underneath since they are warmer than the surrounding environment (Sewell *et al.*, 2013). Using this method requires little maintenance, is low cost and provides means to quantify reptile abundance and distribution (Grant *et al.*, 1992).

Prior to this study, there were 51 tins (at least 50cm x 50cm) present at Eelmoor Marsh (*fig. 3)* and 33 at Frensham Flashes (*fig 4*). 54 tins were laid out at Foxlease and Ancells Meadows (*fig. 5*), and left for two weeks to establish before surveying began (Grant *et al.*, 1992). The tins are spread between the various habitat types at each site (*table 2*).

Table 2: distribution of reptile tins across five broad habitat types.

Site	Habitat Type	Number of Tins	Percentage of tins in each habitat
Eelmoor Marsh	Grassland	5	9.8
	Dry Heath	21	38.9
	Wet Heath	11	21.6
	Mire	8	15.7
	Woodland	6	11.8
Foxlease and Ancells Meadows	Grassland	28	51.9
	Dry Heath	0	0
	Wet Heath	12	22.2
	Mire	13	24.1
	Woodland	1	1.9
Frensham Flashes	Grassland	1	3
	Dry Heath	15	45.5
	Wet Heath	14	42.4
	Mire	2	6.1
	Woodland	1	3

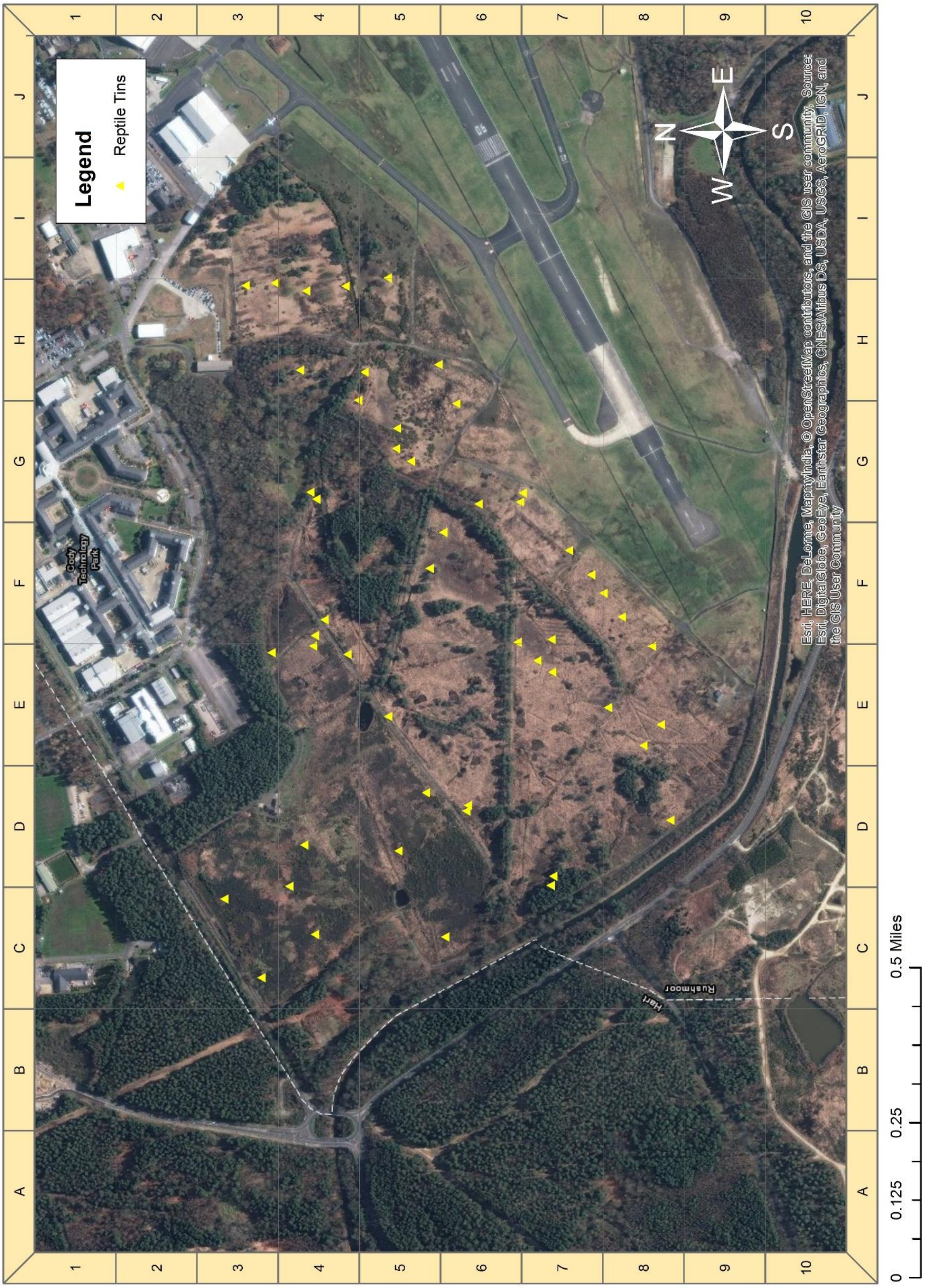

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Figure 3: distribution of 51 tins across Eelmoor Marsh. Created using ArcMap 10.4.1.

Figure 4: distribution of 54 tins across Foxlease and Ancells Meadows. Created using ArcMap 10.4.1.

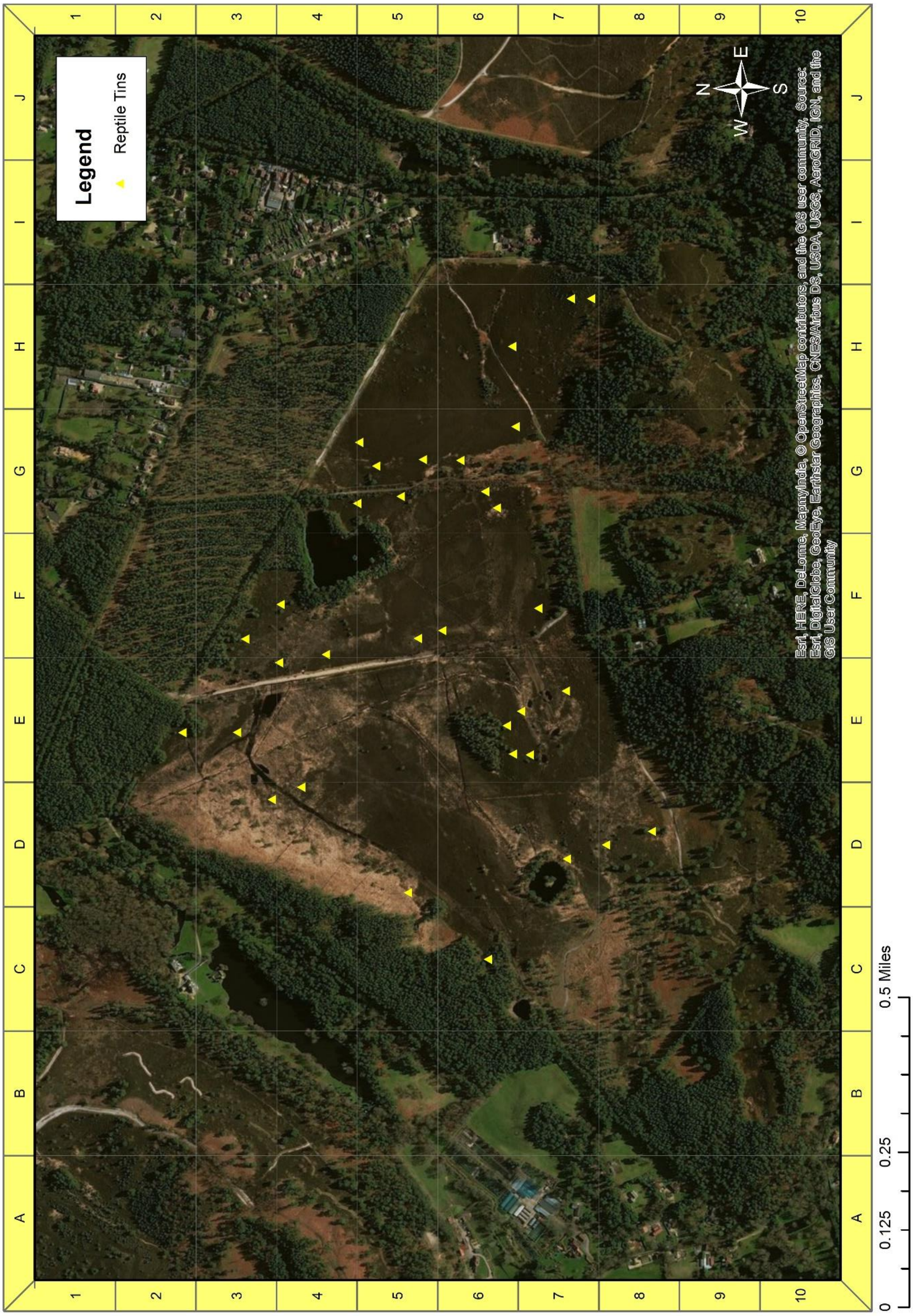

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Figure 5: distribution of 33 tins across Frensham Flashes. Created using ArcMap 10.4.1.

At each site visit, all reptile refugia were checked once following the same route, which was reversed every visit to avoid bias due to time of day. Each tin was carefully monitored before approaching as *Z. vivipara* and *L. agilis* tend to bask on top and are often missed as they move away before being seen. Tins were lifted to identify individuals underneath. Vegetation edges were visually monitored while moving around sites (Sewell *et al.*, 2013). Each reptile found was identified to species, gender and age class (juvenile, sub-adult or adult), along with the time and the tin number. Any reptiles seen while walking between tins were logged on a GPS unit and if possible, species, gender and age also recorded.

The location of any amphibians or small mammals found were noted. On each surveying day, temperature, wind strength (Beaufort scale) and percentage cloud cover were recorded.

2.4 Vegetation Surveys

Vegetation characteristics were recorded as a proxy of grazing intensity, to measure the grass resource reptiles are in direct contact with and provide a quantitative index of grazing intensity to compare across sites. Methodology was adapted from Howland *et al.* (2014) and Howard and Hailey (1999). Vegetation surveys were carried out across weeks six and seven.

At each tin, four 1m x 1m quadrats were placed around it (top, bottom, left, right) and percentage cover of vegetation estimated by eye. Mosses and lichens were included as these provide important habitat for reptiles, but leaf litter was not. At the mid-point of each quadrat, a meter ruler was used to measure vegetation height (Howland *et al.*, 2014). Complexity of surrounding vegetation (~20m radius) was recorded using a scale adapted from Howard and Hailey (1999) (*see appendix 4, table 3*). As areas with lower grazing intensity may have increased shading (Edgar *et al.*, 2010), canopy cover above the tin was also measured using a densiometer.

2.5 Analysis

Analyses were carried out and figures created using statistical software R v3.5.0.

2.5.1 Management Approaches

Differences between grazing management approaches were measured by comparing reptile abundance, proportion of species, distribution across habitat types and prey abundance between sites. These were analysed using chi-squared goodness-of-fit tests, comparing observed and expected counts assuming random sampling and independent data points. Post-hoc testing identified where differences were seen by calculating standard residual values (Sharpe, 2015). Differences in average vegetation between sites were determined with Kruskal-Wallis tests, which do not assume normality as Shapiro-Wilk tests showed data were not normally distributed and transformations were not successful. Furthermore, each site had unequal numbers of data points so samples were unbalanced. R package '*yarrr*' was used to create raw data, descriptive and inferential statistic (RDI) plots which show data distribution to detect skew or variance, the mean and range of the data and 95% confidence limits (Phillips, 2015).

2.5.2 Principal Component Analysis (PCA)

PCA is a dimension reduction tool which condenses predictor variables into smaller sets with lower autocorrelation while retaining most of the information (Perez, 2017) by combining various matrices to capture patterns within the data (Wold *et al*., 1987). Using vegetation height, percentage ground cover and vegetation structure to visually represent grazing intensity showed patterns and variation between each site and guided further analysis. To aid interpretation of results, Spearman's Rank correlation coefficient tests were carried out to investigate relationships between each parameter and PC value (Howland *et al*., 2014).

2.5.3 Generalised Linear Models (GLM)

GLMs identified relationships between reptile abundance and grazing intensity. Canopy cover is included as a covariate as it may affect abundance but is not a measure of grazing intensity. The model for this analysis is:

$$\text{Reptile Abundance} \sim \text{Vegetation Height} + \text{Percentage Cover} + \text{Vegetation Structure} + \text{Canopy Cover} + \varepsilon$$

The data were over-dispersed, due to correlations between variables identified with a Spearman’s Rank correlation co-efficient test. To account for this, negative binomial error structure was used with a logit link function. A chi-squared was used to compare the residual deviance with the residual degrees of freedom to show the negative binomial structure fits the data (Hinkelman, 2012). This method was also used to investigate the effects of temperature, wind speed and cloud cover on reptile abundance, as these could have affected the results. Akaike’s Information Criterion (AIC) values are reported for each analysis, used to assess and select among ecological models (Richards, 2008). Pseudo R-squared values were calculated, allowing relative measures among similar models to indicate how well each model explains variance in the response variable. The *nagelkerke* function of R package ‘*rcompanion*’ was used, which adjusts the result to a maximum value of 1 (all variation explained) (Mangiafico, 2016).

2.5.4 Occupancy Modelling

When surveying, it is unlikely all present individuals will be found, but this does not imply absence if detection probability is lower than 1 (Mackenzie *et al.*, 2002). Occupancy is the probability that a randomly sampled plot is occupied by a species. Detection is the probability an animal is captured, given its presence (Larson, 2014).

Each site was divided into equally-sized patches (*fig. 3, 4, 5*) with 24 at Eelmoor, 22 at Foxlease and 20 at Frensham Flashes. Patch occupancy models were used to analyse dynamics of species ranges while accounting for poor detection probability, using vegetation parameters and canopy cover as predictors of abundance. For patches with more than one tin, averages were taken. The model assumes occupancy is closed within seasons, and there are no false positives (Kery *et al.*, 2013). It also follows the basic assumption that local dynamics are difficult to model, so reduces this down to presence and absence data, necessary in large scale ecological studies (Steinberg & Kareiva, 1997). The R package '*unmarked*' was used, providing a simple framework for occupancy analysis (Fiske & Chandler, 2012). Empirical Bayes-Estimate of occupancy was calculated to give the actual proportion of occupied patches, with a maximum of 1 (all patches occupied) (Rahman & Han, 2018). Post-hoc chi-squared analysis was used to compare occupancy estimates to 1. Estimated detection probabilities were also given.

2.5.5 Heat Mapping

The observed abundance of reptile species was mapped using opensource 'heatmapper' software (Babicki *et al.*, 2016) to create maps of each site, showing 'hotspots' of high reptile abundance. These were compared to patch occupancy estimates and used to infer which areas may be occupied.

3. RESULTS

3.1 Management Approaches

3.1.1 Reptile Abundance

Figure 6 shows the difference in observed and expected reptile abundance between the three sites. As each site had a different number of tins, total abundance was divided by the number of tins per site to acquire an abundance estimate per tin. Expected abundance was calculated as equal abundance at each site, as expected by null hypothesis 1.

There is a difference between observed and expected raw abundance (chi-squared= 143.5, d.f.= 2, $p<2.2^{-16}$) as Eelmoor contained the most reptiles and Foxlease the least. However, when considering the number of tins present, there is no difference between the observed and expected values (chi-squared= 3.1, d.f.= 2, p= 0.2163). Null hypothesis 1 cannot be rejected as there is no difference between sites when considering variation in number of tins.

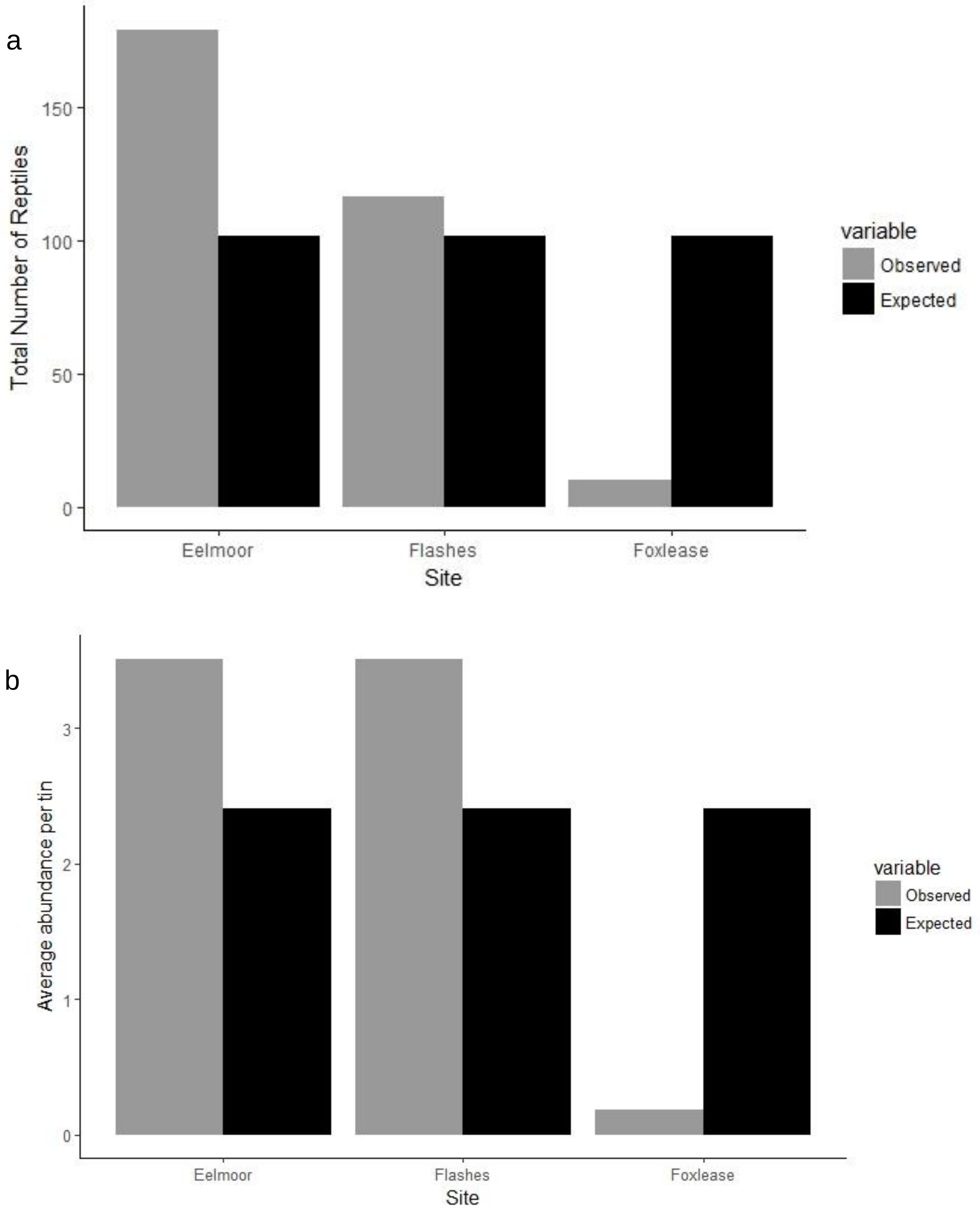

Figure 6: a) reptile abundance comparison across sites.

b) reptile abundance per tin comparison across sites.

3.1.2 Species Proportions

Figure 7 shows observed proportions of each species at each site. Expected abundance is calculated as equal proportions of each species at each site, as expected by null hypothesis 2.

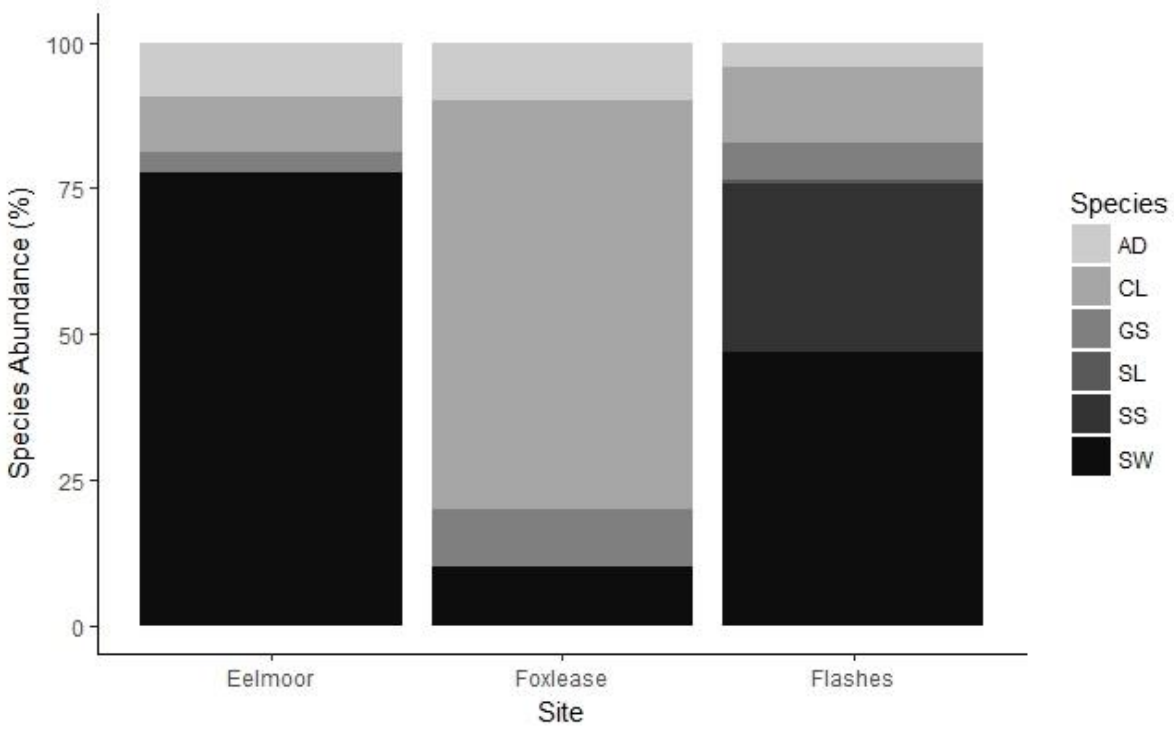

Figure 7: proportions of species found at each site. AD: Adder. CL: common lizard. GS: grass snake. SL: sand lizard. SS: smooth snake. SW: slow worm.

There is a difference in the observed and expected frequencies of reptile species found at each site (chi-squared= 191.5, df= 10, $p<2.2^{-16}$). Post-hoc testing showed fewer slow worms and more common lizards at Foxlease, more slow worms at Eelmoor and more smooth snakes at Frensham Flashes. Null hypothesis 2 can be rejected as each site supports varying abundance of each species.

3.1.3 Habitat Differences

Proportions of tins in each habitat type was calculated as the expected proportion of reptile abundance, as expected by null hypothesis 3. The proportion of reptiles in each habitat type is the observed abundance. Figure 8 shows this comparison.

At Eelmoor, there was no difference in the observed and expected abundance of reptiles (chi-squared= 6, d.f.= 4, p= 0.20024488). However, at Foxlease and Frensham Flashes there was a difference (chi-squared= 25.7, d.f.= 4, p= 3.633^{-5}; chi-squared= 43.8, d.f.= 4, p= 1^{-8}) with more reptiles in mire and fewer in wet heath than expected, shown by post-hoc testing. Null hypothesis 3 can be rejected regarding Foxlease and Frensham Flashes as differing habitat types support different reptile abundance, but cannot be rejected regarding Eelmoor.

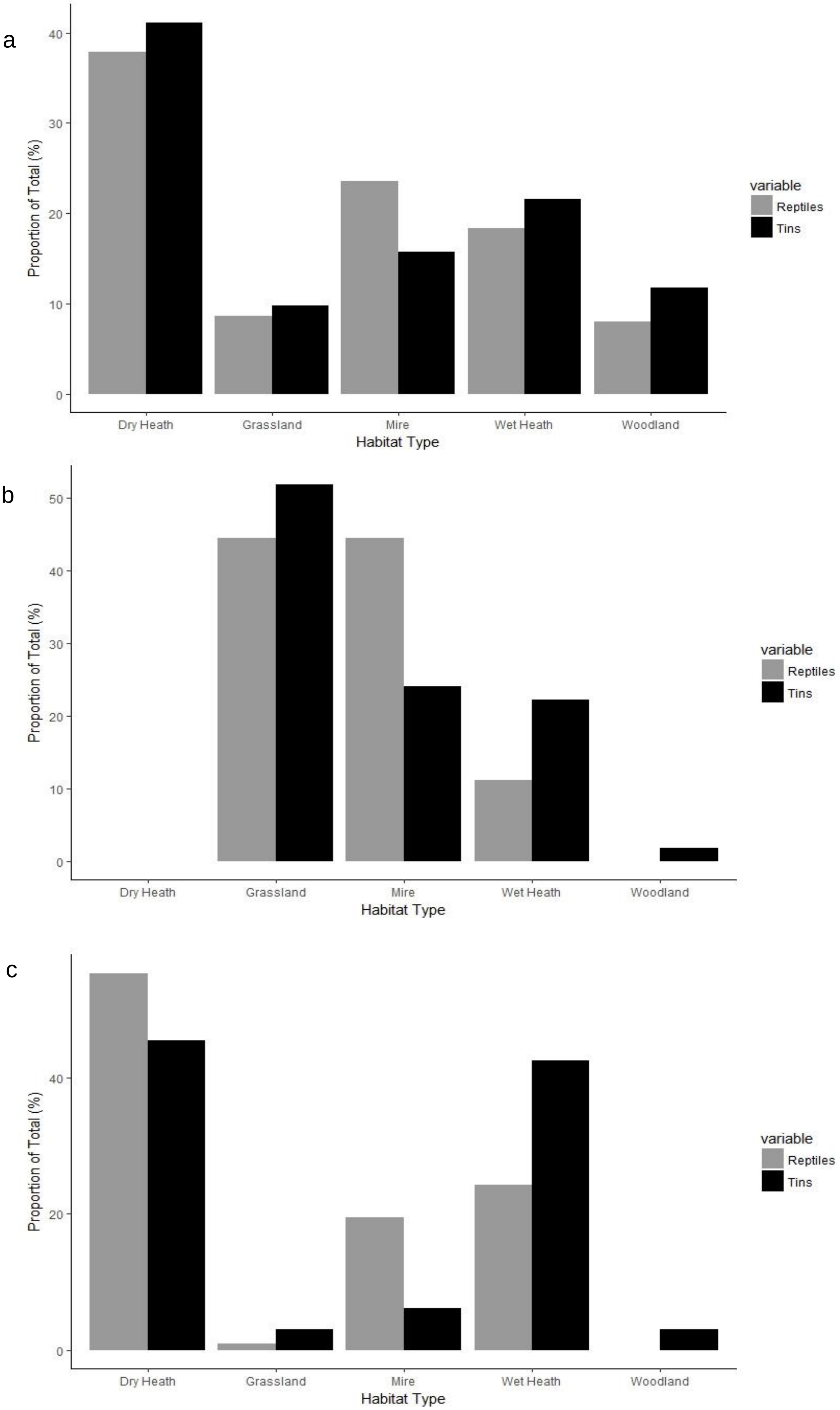

Figure 8: reptile abundance (observed) and number of tins (expected abundance) within each habitat type. a) Eelmoor Marsh, b) Foxlease and Ancells Meadows, c) Frensham Flashes.

3.1.4 Vegetation Differences

Data were not normally distributed (W= 0.98, p= 0.02207; W= 0.72, p= 5.539-^{15}; W= 0.96, p= 2.879^{-4}; W= 0.94, p= 1.796^{-5}) and could not be transformed requiring non-parametric Kruskal-Wallis tests. Figure 9 shows differences in vegetation parameters between sites.

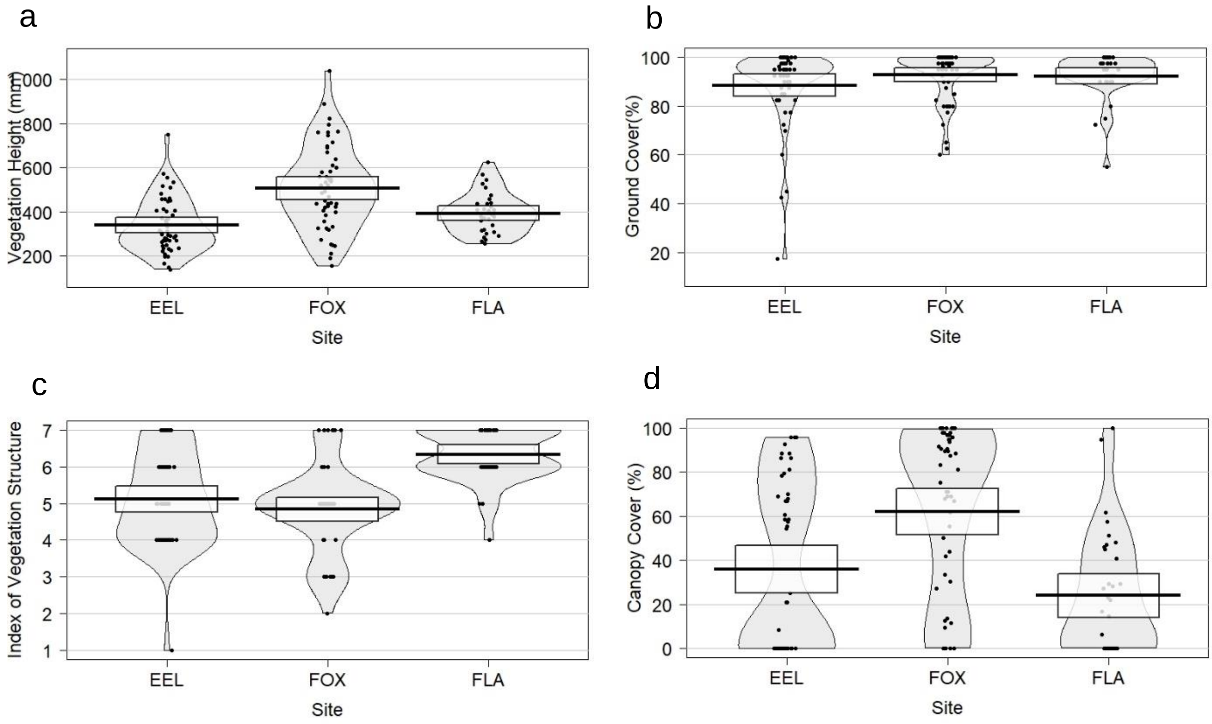

Figure 9: Raw data, descriptive and inferential statistic plots showing differences in a) vegetation height b) percentage ground cover c) vegetation structure and d) canopy cover between Eelmoor Marsh (EEL), Foxlease and Ancells Meadows (FOX) and Frensham Flashes (FLA). These show distribution, mean and range of the data, and inference bands showing 95% confidence limits.

There were differences in vegetation height (Kruskal-Wallis chi-squared= 27.17, d.f.= 2, p= 1.258^{-6}), vegetation structure (Kruskal-Wallis chi-squared= 33.93, d.f.= 2, p= 4.283^{-8}) and canopy cover (Kruskal-Wallis chi-squared= 22.99, d.f.= 2, p= 1.018^{-5}) between each site. There was no difference in percentage ground cover (Kruskal-Wallis chi-squared= 5.92, d.f.= 2, p= 0.05174). Null hypothesis 4 can be rejected for three vegetation parameters, but not regarding percentage ground cover.

3.1.5 Prey Abundance

Figure 10 shows total observed abundance of prey species found at each site. Expected abundance was calculated as equal abundance at each site, as expected by null hypothesis 5.

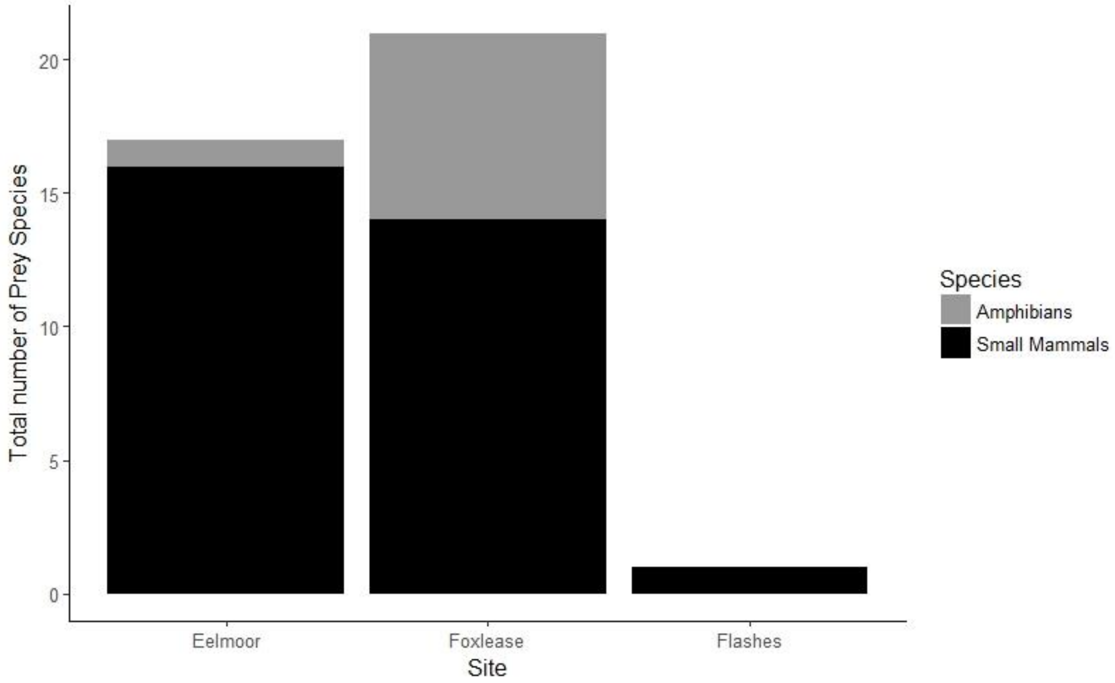

Figure 10: total abundance of amphibian and small mammal prey at each site.

There is a difference in the observed and expected abundance of amphibians at each site (chi-squared= 10.75, d.f.= 2, p=0.004631) and small mammals (chi-squared= 12.8, d.f.= 2, p=0.00163). Post-hoc testing showed more amphibians than expected at Foxlease, and fewer small mammals at Frensham Flashes. Null hypothesis 5 can be rejected as each site supports varying prey abundance.

3.2 Principal Component Analysis

Figure 11 shows the outcome of PCA.

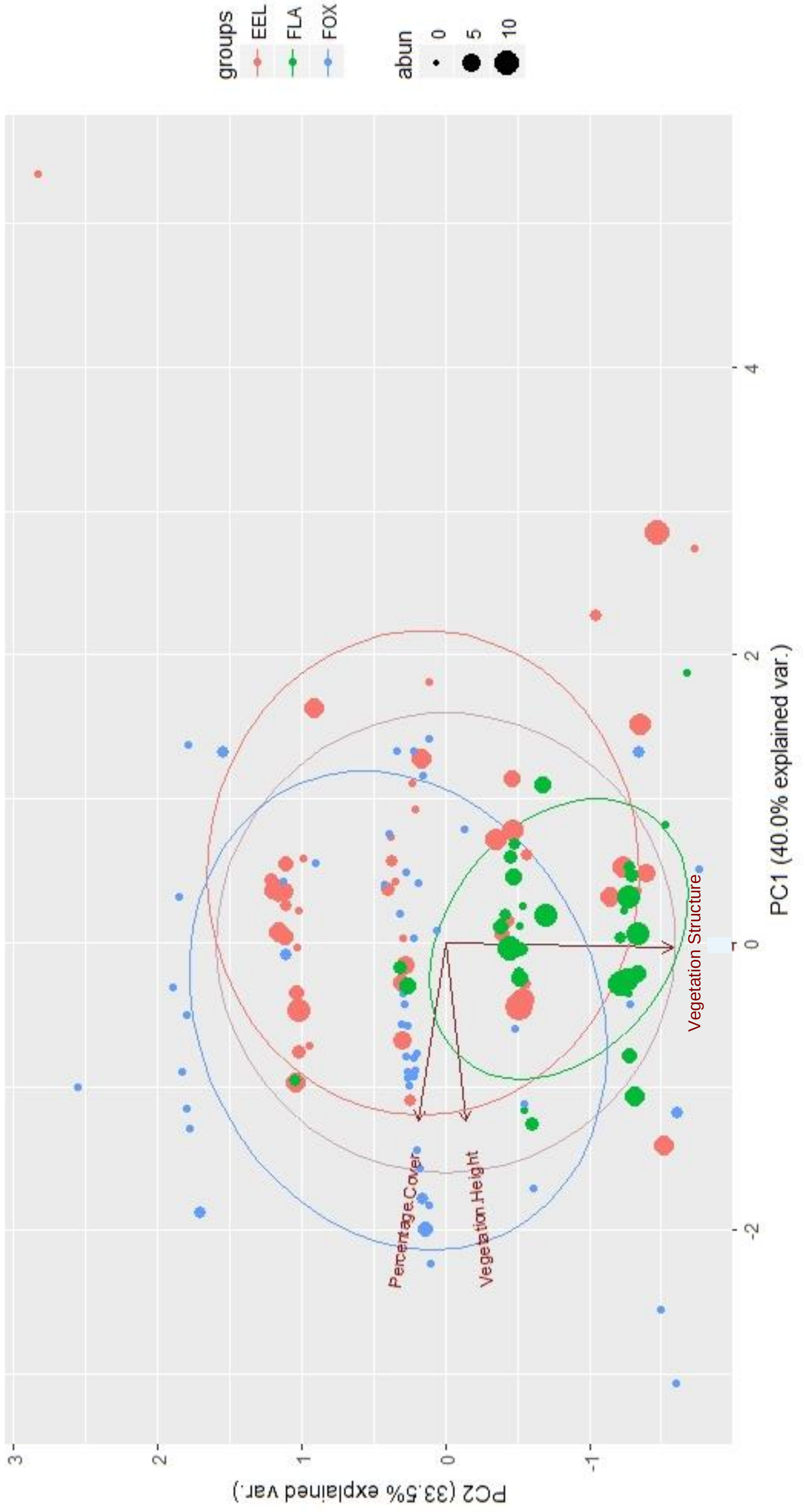

Figure 11: the outcome of PCA of vegetation parameters at each site, Eelmoor Marsh (EEL), Foxlease and Ancells Meadows (FOX) and Frensham Flashes (FLA). The size of the points represents the abundance of reptiles found at that location.

Table 3: correlations between principle components and vegetation parameters.

	Vegetation Height (mm)		Ground Cover (%)		Vegetation Structure	
	S	p-value	S	p-value	S	p-value
PC1	785440	$<2.2^{-16}$	766170	$<2.2^{-16}$	401100	0.3261
PC2	488020	0.1822	325150	0.002283	862090	$<2.2^{-16}$
PC3	748110	$<2.2^{-16}$	324450	0.002139	372010	0.07778

Components 1 and 2 are correlated with all three vegetation parameters (*table 3*), and combined explain 73.5% of the variation within the data. By capturing patterns, these clusters show clear site differences, suggesting lower variation in vegetation parameters at Frensham Flashes than the other two sites. Large point sizes suggest that despite this, Frensham Flashes still retains a large reptile population, while Foxlease contains much smaller populations while having higher heterogeneity. Eelmoor has high heterogeneity supporting large reptile populations, suggesting other factors are affecting abundance.

3.3 Generalised Linear Models

Figure 12 shows relationships between abundance and vegetation parameters across all sites. The data were over-dispersed with a dispersion parameter of 3.43, due to correlations between variables (S= 297750, p= 0.000129; S= 514210, p= 0.04122; S= 303710, p= 0.0002551). When using negative binomial error structure there is no difference between residual deviance and residual degrees of freedom (p=0.47702), showing this error structure fits the data (Hinkelman, 2012).

There is a relationship between abundance and vegetation height ($z_{2,133}$= -1.995, p= 0.04606) and vegetation structure ($z_{2,133}$= 2.985, p= 0.00284). There is also an effect of canopy cover ($z_{2,133}$= -2.839, p= 0.00453). There is no relationship between abundance and percentage cover ($z_{2,133}$= 0.809, p= 0.41846). Null hypothesis 6 can be rejected, as lower vegetation and more complex structures driven by grazing intensity support a higher abundance of reptiles. The model has an AIC value of 496.84, and a pseudo R-squared value of 0.16.

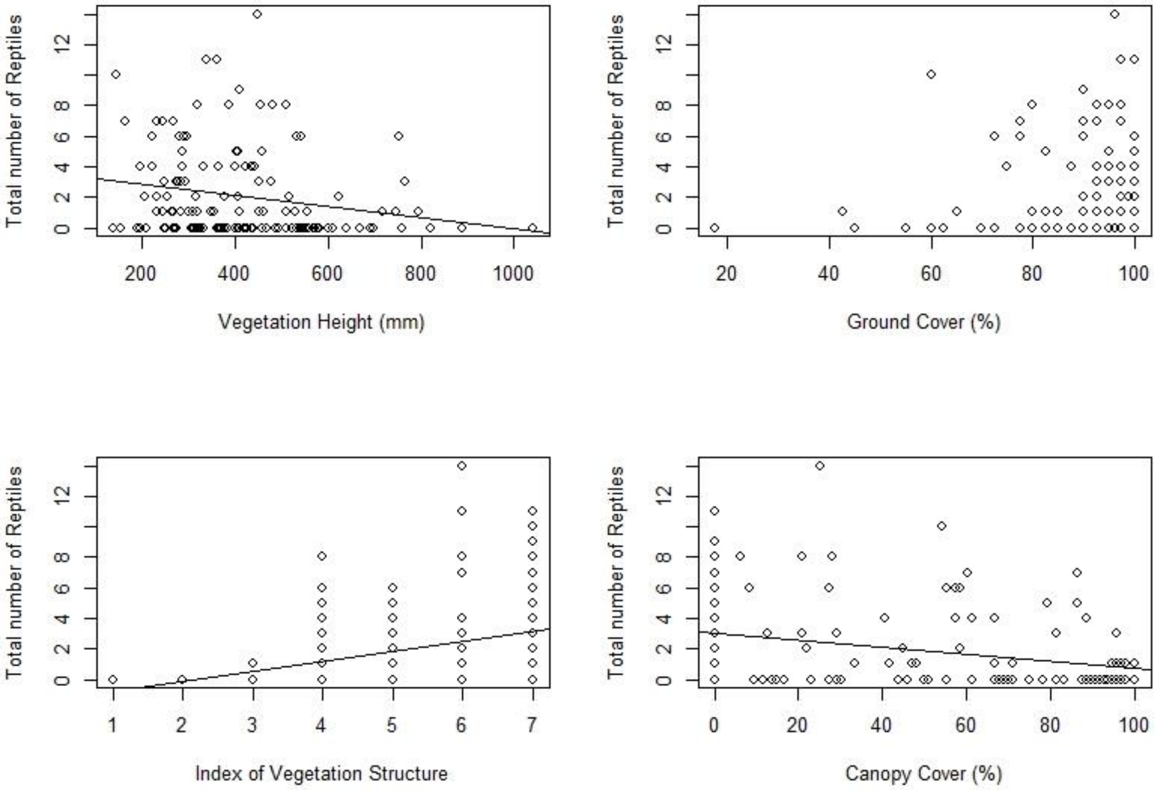

Figure 12: relationships between reptile abundance and vegetation height, percentage cover, vegetation structure and canopy cover. Regression lines show direction of relationship.

Weather parameters were assessed to account for effects of variation in temperature, wind speed and cloud cover. Figure 13 shows their effects across the survey period. Data are over-dispersed with a dispersion parameter of 11.7, likely due to high numbers of zeros as there are no correlations between variables (S= 3079.2, p= 0.1053; S=2647.3, p= 0.4812; S=1538, p= 0.1138). When using negative binomial error structure there is no difference between residual deviance and residual degrees of freedom (p= 0.11156), showing the error structure fits the data (Hinkelman, 2012).

There is no relationship between reptile abundance and temperature ($z_{2,20}$= 1.235, p= 0.21668), wind speed ($z_{2,20}$= -0.701, p= 0.48338) or cloud cover ($z_{2,20}$= 1.26, p= 0.2075). The model has an AIC value of 174.7, and a pseudo R-squared value of 0.1.

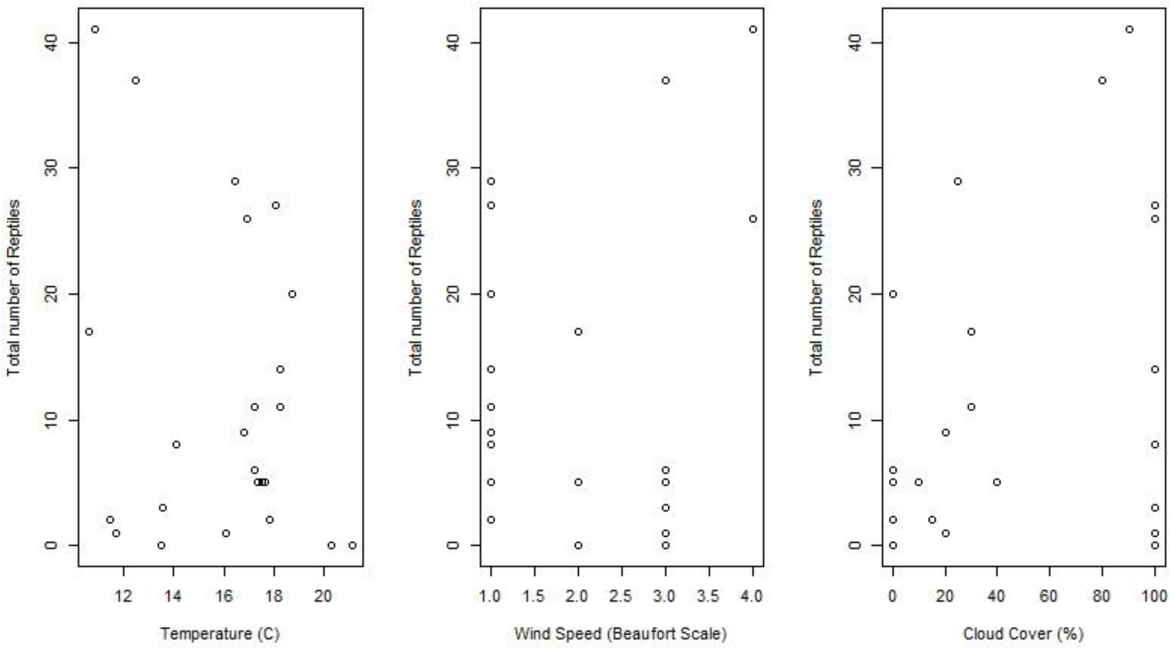

Figure 13: relationships between reptile abundance and temperature, wind speed and cloud cover.

3.4 Occupancy Models

Figure 14 shows occupancy and detection probabilities at each site.

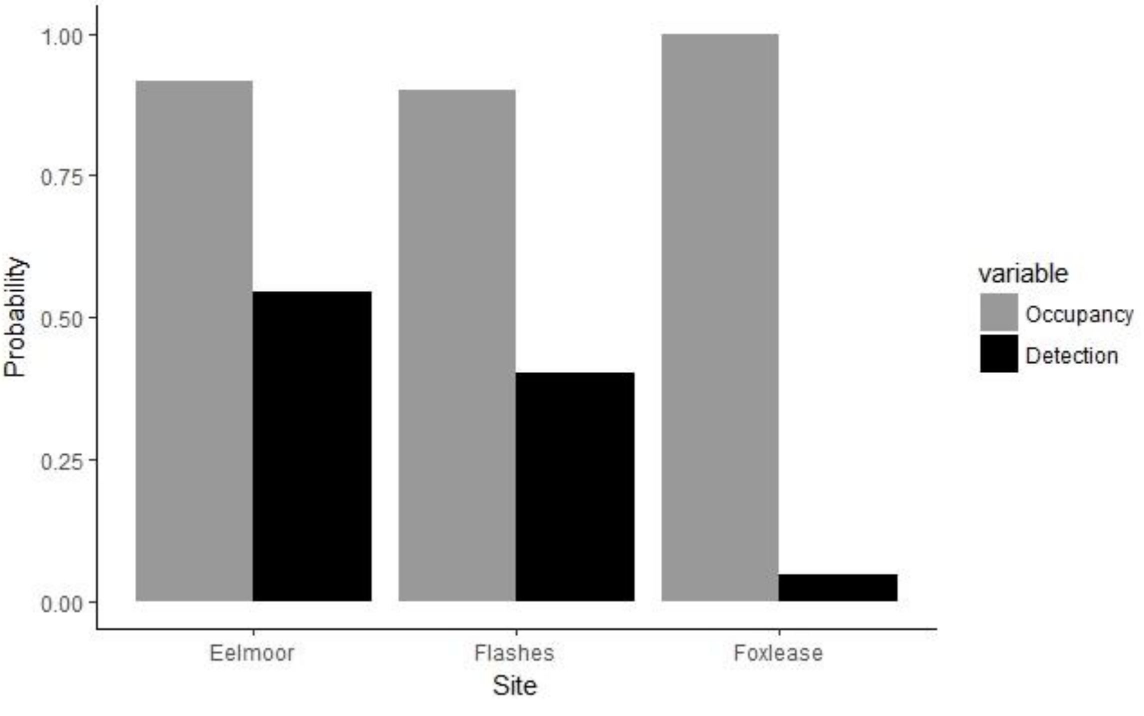

Figure 14: Empirical Bayes-Estimate of occupancy and detection probabilities at each site.

Patch occupancy estimates are very high under all management systems, suggesting almost all patches are occupied. Empirical Bayes-estimates give exact numbers of occupied patches: 22 at Eelmoor and Foxlease and 18 at Frensham Flashes. Chi-squared analysis shows no difference between empirical Bayes-estimates and occupancy of 1, as expected by null hypothesis 7 (chi-squared= 0.02, d.f.= 2, p= 0.99) which must be rejected as all areas show high patch occupancy regardless of grazing intensity. Conversely, detection estimates are low, suggesting many more reptiles are present at all sites than were found during this study.

3.5 Heat Mapping

Figure 15 shows distributions of recorded reptiles. At Eelmoor, ‘hotspots’ of abundance are spread amongst habitat types, as suggested by figure 8a. At Foxlease, highest abundance is within grassland with others spread through mostly mire (*see figure 8b*) and at Frensham Flashes, highest abundances are within dry heath and mire (*see figure 8c*).

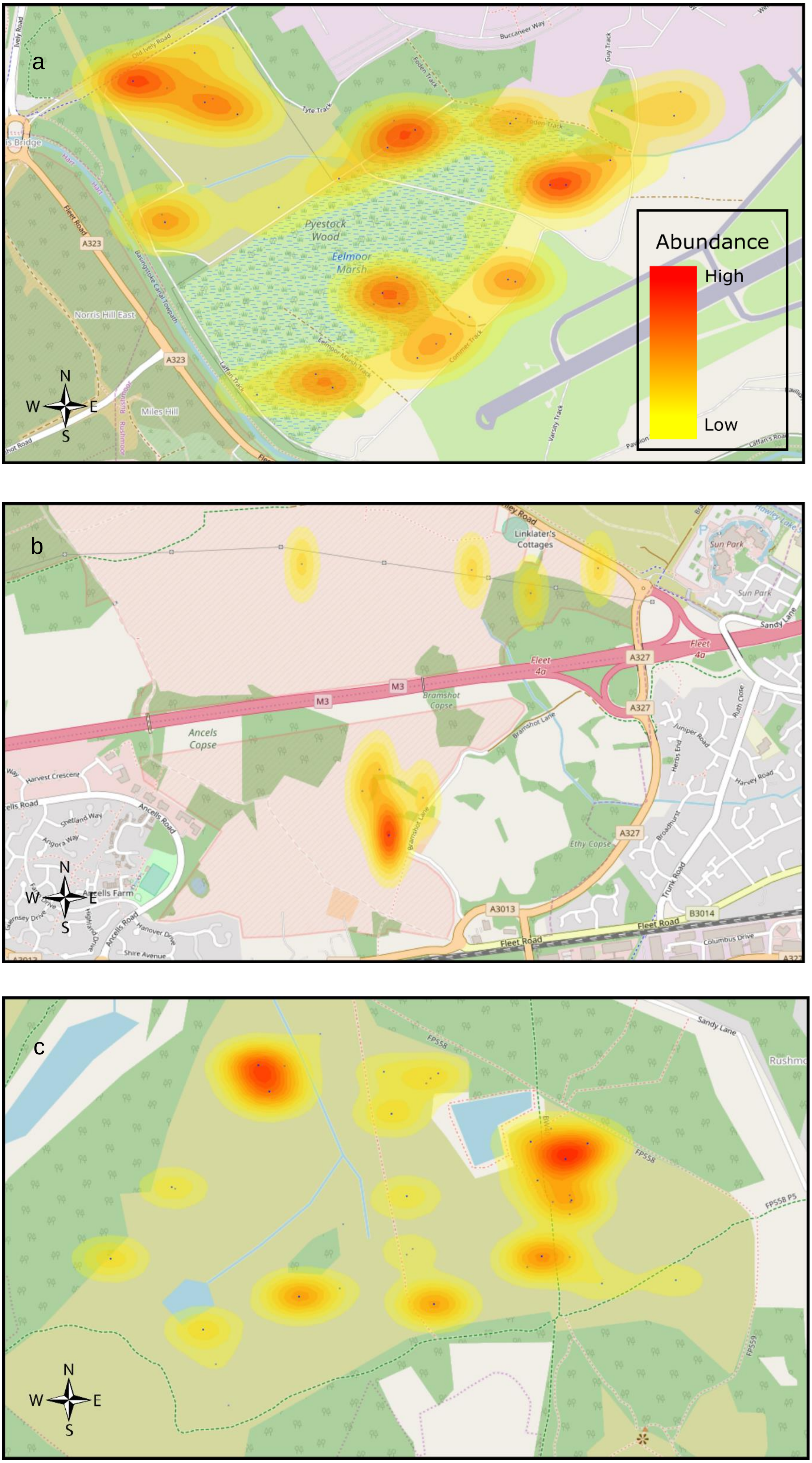

Figure 15: heat maps of observed reptile abundance of a) Eelmoor Marsh, b) Foxlease and Ancells Meadows, c) Frensham Flashes. Areas with no colour show where no reptiles were found. Created using ‘heatmapper’ software (Babicki *et al.*, 2016).

4. DISCUSSION

Research on how reptiles respond to grazing is largely contradictive, with responses driven by intensity and management systems. However, there is little specific research into these differing systems. Three systems under varying intensities were compared to identify how each modifies habitat structure and drives reptile assemblage, and recognise which is most suitable for conservation grazing. The results support research finding benefits of free-roaming grazing, but find new evidence showing rotational grazing is disadvantageous to reptile assemblages.

4.1 Grazing Management Systems

Comparison of management systems showed that low-stocking density of free-roaming grazers supported high numbers of reptiles, almost equal to the non-grazing condition. This system is successful in replicating the more natural management of scraping and mowing, with low-intensity grazing that is driving habitat change, but is not too intense to cause over-grazing. This vegetation provides optimal reptile habitat, with intermediate height, high ground cover, heterogeneric structural complexity and low canopy cover (Edgar *et al.*, 2010). The robust highland cattle are well-adapted to grazing coarse vegetation (Grazing Animals Project, 2001), grazing equally between habitat types (*appendix 5, figure 4*) allowing even reptile distribution, supporting null hypothesis 5. All five reptile species are well-supported, especially slow worms due to available open habitat for basking (Edgar *et al.*, 2010; Inns, 2011). High prey abundance is vital for reproductive success, and is positively correlated with female body weight (Reading, 2004). Adder populations are well-supported by small mammal prey however grass snakes may be hindered by fewer amphibians, although prey estimates are likely inaccurate as surveys were reptile-specific.

Despite finding no differences between reptile abundance per tin, rotational grazing supported far fewer reptiles. Rotations create disparity between grazed and non-grazed fields, with some short vegetation and some overgrown, neither suitable for reptiles (Edgar *et al*. 2010; Sato *et al*. 2014). Higher average vegetation and canopy cover restricts basking sites and increases over-shading causing degradation of important micro-habitat features (Cox & Temple, 2009; Pike *et al.*, 2011). There were more common lizards than expected, possibly as shorter vegetation provided preferred open habitat (Edgar *et al.*, 2010), however the sample is too small to draw conclusions and with more data proportions may change. Thresholds between grassland and young heathland can be difficult to assess (Ejrnæs *et al.*, 2008), and at Foxlease areas classified as ‘wet heath’ were very similar to ‘grassland’. *C. vulgaris* takes 40 years to mature and reaches one metre (Watt, 2018), which Foxlease’s 10-year old vegetation is far from. As reptiles are associated with mature heather, this little amount of heath could explain low abundance (Edgar *et al.*, 2010). Furthermore, these smaller patches are more susceptible to negative effects of grazing and conversion to grassland (Moore, 1962), with development further hindered by public access (Cox & Temple, 2009). High numbers of amphibians and small mammals show potential for sustaining reptiles if habitat quality were improved.

Habitat management through scraping and mowing at Frensham Flashes provided intermediate vegetation height, high ground cover and low canopy cover required by reptiles (Edgar *et al.*, 2010). Vegetation is predominantly uniform dry and wet heathland with little variation in structural complexity, as shown by PCA and is driven by lack of grazing (Pielou, 1966; Howland *et al.*, 2014). The site supports large *C. austriaca* populations preferring denser, less open areas due to lower body temperature requirements (Edgar *et al.*, 2010). If excluded from analysis, total abundance is lower suggesting this habitat does not suit other reptile species despite targeted management to provide habitat for all. However, site comparisons are biased as smooth snakes were not present at all sites and may thrive equally well. Fewer small mammals and amphibians at Frensham Flashes suggests a prey shortage, perhaps as small mammals are also supported by heterogeneity in vegetation structure (Spellerberg, 1989), however populations can be harmed by cattle grazing (Read & Cunningham, 2010). Creating suitable reptile habitat through grazing could also be detrimental to smooth snakes, reducing their reproductive rate (Reading & Jofre, 2015) so increasing heterogeneity through other methods is favoured. Although there are no controlled grazers, wild deer graze across heathland (Hester *et al.*, 1999) with large populations increasing fragmentation and grassland conversion (Hester & Baillie, 1998). Negative effects are enhanced on slopes, with uphill grazing and downhill trampling (Hester & Baillie, 1998) affecting reptiles relying on south-facing slopes. Low-intensity deer grazing may increase vegetation quality at Frensham Flashes, explaining some of the similarities to Eelmoor, however could hinder heathland restoration at Foxlease as cattle grazing of *M. caerulea* promotes spring growth, providing forage for deer and further increasing damage (Gordon, 1988).

Reptile tins are often waterlogged in wetland areas, reducing sampling effectiveness (Mitchell *et al.*, 1993) due to reptile's sensitivity to heat, humidity and soil wetness (Grant *et al.*, 1992). However, at Foxlease and Frensham Flashes there were more reptiles than expected in mire. As the tins in Mitchell *et al*'s study (1993) were in wooded areas, high shading was not accounted for and could have affected this result. Some species also prefer damper habitats (Edgar *et al.*, 2010). In Mediterranean countries, *N. natrix* are found almost exclusively in wet areas (Capula *et al.*, 1994), perhaps as a means of thermoregulation. As spring temperatures were unusually high (Met Office, 2018), reptiles may have sought damper habitats to cool down. There is some suggestion of avoidance behaviour by reptiles, as livestock breeds do not move evenly between habitat types (Rook *et al.*, 2004) and some avoid mire when given the choice (Bullock & Pakeman, 1997). Again, Foxlease's small sample may show an inaccurate depiction of reptile distribution between habitat types and with larger samples proportions may equalise, as expected by tin distribution and supporting null hypothesis 5.

4.2 Relationships between Reptile Abundance and Grazing Intensity

Low-level disturbance through grazing increases heterogeneity and drives suitable habitat (Bullock & Pakeman, 1997; Wilkie, 2013; Howland *et al.*, 2014), thereby benefitting reptiles (Spellerberg, 1989). There is little research on ideal sward height, but intermediate height is thought optimal as basking opportunities are lost in higher vegetation (Edgar *et al.*, 2010) and low vegetation does not provide adequate shelter from adverse weather and predators (Kreulen, 1979; Sato *et al.*, 2014). This is supported by this study, as abundance decreased in higher vegetation, opposing null hypothesis 6. Abundance also increased with variation in structural complexity, providing micro-habitat structures such as basking sites and shelter required by all reptiles (Garden *et al.*, 2007; Edgar *et al.*, 2010; Santos *et al.*, 2016). Insufficient grazing causes increased shading as vegetation succeeds further towards its woody, dense climax (Edgar *et al.*, 2010), leading to degradation of key micro-habitat features (Cox & Temple, 2009) and lower abundance particularly in open-habitat species (Pike *et al.*, 2011). Rotational grazing systems are not intense enough to halt this succession, creating heavily over-grown areas with high canopy cover and low reptile abundance. Connectivity between and within patches and across landscapes could be restored by implementing low-stocking density grazing systems to drive habitat changes and create transitional zones to allow movement of metapopulations.

Many other factors affect reptile abundance. Warmer ambient temperatures lead to higher occurrence due to increased basking (Read & Moseby, 2001; Spence-Bailey *et al.*, 2010), increasing hatching success, offspring size and performance (Shine & Elphick, 2001). Windspeed affects detection rate of many animals (Anderson, 2001), though there is limited research on reptile responses. In turtles, higher windspeed increases rate of water loss allowing the ectotherms to cool down (Foley & Spotila, 1978) which may be seen in other reptiles. Although this study claims to investigate evaporation in natural conditions, the highly controlled laboratory experiment may not be representative and should be repeated in the field. Cloud cover also affects detection (Anderson, 2001) with increased cover decreasing reptile observations due to reduced sunlight (Spence-Bailey *et al.*, 2010) and prolonged overcast weather having long term effects on body size and reproductive success (Hare & Cree, 2010). Although ambient temperature, windspeed and cloud cover did not affect abundance in this study, recording micro-climate parameters at each tin may show relationships and interactions as they fluctuate. These factors likely have an optimum, and deeper analysis may reveal non-linear relationships with reptile abundance. Furthermore, spring temperatures were unusually high and did not vary greatly (Met Office, 2018) so may have had a lesser effect on abundance.

When modelled separately, canopy cover and grazing intensity had higher AIC values, so the inclusive model was deemed more suitable (Richards, 2008). Combined, vegetation parameters and weather conditions explain 27% of variation in the response variable, indicated by pseudo R-squared values, leaving 73% unexplained. High empirical Bayes-estimates of occupancy suggest vegetation was similarly suitable across patches, however not all supported reptiles. Other factors such as habitat type (Edgar *et al.*, 2010), plant species richness (Spellerberg, 1989), changes to vegetation (Read, 2002) and trophic impacts (Read & Cunningham, 2010) influence tin use. Including these in GLM and occupancy analysis would provide more accurate estimates, accounting for more variation in abundance and showing clear differences between grazing systems. Detection probability is also influenced by many factors, such as climate, vegetation height and habitat type (Anderson, 2001). Low abundance estimates at Foxlease may have been caused by higher vegetation and denser habitat, and in reality, could be much higher than originally thought. This would change interpretation of results, as higher abundance would show rotational systems as an effective conservation tool for reptile assemblages.

Heat maps show reptile abundance was not spread equally across sites, contradicting occupancy estimates. Since occupancy is influenced by many factors, using heat maps as a field tool may provide a more realistic view. Under free-roaming grazing, highest abundance was found in dry heath, mire and grassland, spread equally as suggested by habitat analysis. Rotational grazing drove increases in grassland, where reptiles were mostly found and under the non-grazing system, dry heath and mire contained highest abundance, opposing null hypothesis 5. As habitat type influences assemblage with dry heath, grassland and mire being especially important (Edgar *et al.*, 2010), including dominant habitat as a predictor in patch-occupancy analysis may give results comparable to heat map distributions. However, some areas had fewer tins and lower sampling effort, so heat maps may be biased.

4.3 Recommendations

Conservation grazing is not always suitable as livestock will not feed on highly woody or dense vegetation. Combining a range of methods to reduce density may be most effective, allowing grazers to feed on regrowth (Bullock & Pakeman, 1997).

Low-intensity, free-roaming grazers at Eelmoor are increasing vegetation heterogeneity and reducing sward height, supporting healthy reptile populations which are using all habitat types equally. Potential plans to carry out smooth snake reintroductions are on hold (Langham, 2015), but as *C. austriaca* prefer denser, deeper heathland this thinner, more heterogeneous habitat may not be suitable, with careful planning required as cattle grazing may harm populations (Edgar *et al.*, 2010; Reading & Jofre, 2015). If re-introductions went ahead, the most suitable area is the northern dry heath where sand lizards were introduced as these are often found together (Inns, 2011). Removing grazers could increase heathland density, however may disadvantage other reptiles. Further surveying of prey species is advised, as amphibian populations appear low which may hinder grass snake populations in the future (Edgar *et al.*, 2010; Inns, 2011).

At Foxlease, rotational grazing is creating disparity between grazed and un-grazed areas with neither providing suitable reptile habitat. Reptiles are thought not to benefit from rotational grazing (Dorrough *et al.*, 2012), though this study included sheep grazing which are known to cut vegetation very close to the ground and cause over-grazing if not managed correctly (Grazing Animals Project, 2009). Nonetheless, free-roaming grazers are considered more beneficial (Bullock & Pakeman, 1997) and introducing a low-stocking system similar to Eelmoor may be advantageous as it drives heterogeneity in vegetation complexity and optimal vegetation height. However, Ancells Meadows SSSI has a small constant herd, which does not seem to improve habitat or increase reptile abundance. Since heathland restoration here is relatively recent (Richard Hennessey, pers. comm. 2018), habitat quality may increase in time and support a healthy reptile assemblage, as it has the potential to do. Continued surveys are advised, by deploying well-established tins and regularly returning to investigate trends over time. The site will be recommended to SARG.

Although lower vegetation heterogeneity at Frensham Flashes is suitable for smooth snakes, which are clearly thriving, other reptiles may be suffering as they require higher structural diversity. Free-roaming grazers could be detrimental to smooth snakes (Reading & Jofre, 2015) so are not advised. Continued surveying will further document reptile assemblage across the area, identifying important dry heath areas for smooth snakes. Tailoring management to suit all species through increased scraping and mowing or introducing fenced grazers in other habitat types may support other reptiles and small mammal populations. This could force a trade-off decision: to manage the site for all reptiles, or just for the rarer species. Further prey surveys are advised as small mammal populations appear low.

Protected areas contain disproportionate numbers of threatened species, justifying conservation of these diverse areas (Dobson *et al.*, 1997; Reid, 1998). Although lowland heath is abundant in southern England (Groves *et al.*, 2012), landscapes surrounding these protected areas are highly fragmented with low connectivity. Widescale habitat preservation is required to lessen impacts of habitat destruction (Jantz *et al.*, 2015), by increasing fragment size to sustain higher biodiversity (MacArthur & Wilson, 1967; Pimm & Raven, 2000) and implementing wildlife corridors to increase connectivity, allowing movement of metapopulations and increasing gene flow (Hanski, 1998). Grazing management combined with other methods could create larger, well-connected areas and restore habitat. Although reptiles do not respond to immediate effects of grazing (Read, 2002) and high fragmentation affects indicator quality (Moore, 1962), monitoring abundance across the landscape before, during and after restoration will show long-term population trends and indicate changes to habitat (Morris *et al.*, 2008; Cox & Temple, 2009).

4.4 Limitations and Improvements

Frequent snow storms meant surveying started a month later than planned, requiring pilot surveys to monitor emergence and causing fieldwork to be extended into June. Although spring months have higher survey success as showers cause reptiles to retreat and return in dryer conditions, and mating behaviour increases activity and reduces effects of disturbance (Spence-Bailey et al. 2010; Inns 2011), there was little rain with unusually high spring temperatures (Met Office, 2018) which may have decreased detectability. To increase available time for analysis and writing, surveys were shortened to 10 weeks. Since mating was complete, many individuals had retreated by June (Inns, 2011) so little data was lost. With more time, the proposed use of 'sample-point' software (Booth *et al.*, 2006) to accurately quantify percentage cover may have shown relationships between ground cover and reptile abundance. Using remote sensing software such as '*Fragstats*' (McGarigal *et al.,* 2012) could analyse patch dynamics, giving connectivity indices within and between patches which could be included in occupancy models as predictors of reptile abundance. Although the vegetation complexity scale provided an efficient record of habitat structure (Howard & Hailey, 1999), more comprehensive measures such as taking photographs and using '*Fragstats*' software would give multiple measures of complexity and density (Marsden *et al.*, 2002). Lastly, by using R package '*MetaLandSim*' (Mestre *et al.*, 2016), stochastic patch occupancy models could be created to model possible grazing scenarios at each site and estimate effects on reptile patch occupancy.

Low abundance at Foxlease may be influenced by non-established tins, reducing survey effort. Mechanisms behind establishment are not fully understood, but could be due to chemical balances taking time to form suitable microhabitats with specific heat, humidity and soil requirements (Grant *et al.*, 1992). However, a large sheet of corrugated metal present for many years (Richard Hennessey, pers. comm. 2018) did not harbour reptiles when checked. Repeating the study with well-established tins when heathland has developed may show different relationships between grazing and abundance, and effects of rotational grazing could be clearer. With regular surveys, patterns and trends in population dynamics may be seen. Comparisons were further hindered by unequal tins between sites, which were not distributed evenly between patches. This was partly due to Foxlease being highly waterlogged after the snow melted, meaning some fields could not be accessed and tins could not be spread evenly. Finding sites with equal numbers of well-established tins, or deploying equal amounts in dryer conditions and leaving to establish would rectify this. The latter would allow even distribution between patches and habitat types, equalising patch replication and increasing power of patch-occupancy analyses as well as reducing bias of heat maps.

Many ecological studies do not account for over-dispersion when selecting models (Richards, 2008), which over-estimate the influence of variables often resulting in overly-complex models and poorly drawn conclusions (Anderson *et al.*, 1994). Originally, general linearised mixed-effect models (GLMMs) were created, however high over-dispersion caused by auto-correlation and zero-inflation could not be accounted for, reducing statistical power. Models were simplified to GLMs with suitable negative binomial families to reduce over-dispersion and increase statistical power. Using principal component values in GLMMs may reduce over-dispersion caused by multicollinearity (Perez, 2017), allowing the use of more complex models. Furthermore, weather parameters could not be included in the model as co-variates due to different levels of replication, however measuring micro-climate parameters at each tin would allow this. Incorporating these other explanatory variables into GLMM may increase the R-squared value, explaining more of the variation in the response variable, and give a more powerful model with a lower AIC value.

Due to practicality and licensing, reptiles were not handled or individually identified. Individuals could have been counted twice, leading to pseudo-replication which violates assumptions of many statistical analyses. However, if individuals return to a tin, surrounding habitat is likely highly favourable indicating high occupancy. Individually identification would allow exact population counts through mark-recapture analysis and comparison of diversity indices across sites (Grant *et al.*, 1992).

Vegetation parameters are an indirect proxy of grazing intensity (Howland *et al.*, 2014), however direct measures such as stocking density could be included in GLM analysis. Since grazers do not move equally between habitats (Bullock & Pakeman, 1997), observations of livestock location at Eelmoor would highlight areas frequented more often, and at Foxlease occupied fields under the rotational system could be considered, providing a more accurate measure of grazing intensity.

Red wood ant (*Formica rufa*) nests were often found underneath tins. Although not documented in the UK, reptile predation is seen in fire ants (Flowers & McCallum, 2012) which may affect reptile's use of tins and further affect populations. Nests may bias the results, as tins with suitable habitat are not used by reptiles; one slow worm found by a nest left immediately as ants attacked. Moving tins with nests underneath may reduce this bias, however the tin would require time to re-establish and another nest may develop underneath. Furthermore, *e*xtinction risk of *F. rufa* is classed as 'Near Threatened' (Social Insects Specialist Group, 1996) and moving tins may harm populations. Ant predation also increases with habitat disturbance (Todd *et al.*, 2008), so grazing may exacerbate the problem.

4.5 Further Study

Breeds of grazing livestock impact vegetation differently, for example Shetland cows are robust and graze boggy, un-improved areas (Grazing Animals Project, 2008). Differences are likely due to variations in dental and digestive anatomy and body size, and can be used as mechanisms to meet biodiversity outcomes (Rook *et al.*, 2004). Further study into breeds at Foxlease and Eelmoor may indicate those more suitable for supporting reptile assemblage. There is little research on reptile's responses to livestock, however this study showed some evidence of avoidance behaviour. Reptiles may be disadvantaged by livestock, despite benefitting from vegetation changes they drive, an area that requires further study.

Since high prey abundance is vital for reproductive success (Reading, 2004), determining grazing effects on small mammals and amphibians as a means of supporting reptile abundance is important. Although small mammal populations benefit from structurally complex vegetation (Spellerberg, 1989), they are harmed by cattle grazing (Read & Cunningham, 2010). Using surveying methods more appropriate for these taxa, such as Longworth Traps (Chitty & Kemspon, 1949) or amphibian visual and acoustic surveys (Rödel & Ernst, 2004), relationships between grazing, prey and reptiles could be further investigated.

As *N. natrix* feed on amphibians they are often associated with water sources (Edgar *et al.*, 2010), however there is limited research on reptile responses to proximity to water. Reptiles in damper habitats have higher rates of evaporative water loss than arid-adapted species (Cox & Cox, 2015), suggesting they require closer water sources to replace moisture. In Mediterranean countries, grass snakes are found almost exclusively in wet areas (Capula *et al.*, 1994), possibly for thermoregulation. With rising temperatures, other species may show similar behaviour. If this were true, water availability may be used as another predictor of reptile abundance.

Finally, since nitrogen deposits are causing *C. vulgaris* to give way to grasses such as *Festuca ovina* and *M. caerulea* (Heil & Diemont, 1983; Heil & Bruggink, 1987) soil samples may identify nutrient-rich heathland areas at risk. At Foxlease in particular, nutrient-rich soil might be a factor inhibiting heathland development. Mitigation may aid heathland restoration, thereby increasing reptile abundance.

4.6 Conclusion

This research has provided much needed insight into the consequences of various conservation grazing systems on reptile assemblage within complex grassland-heathland systems, showing relationships between grazing intensity and reptile abundance. Based on the findings of this study, grazing reduces sward height and increases variation in structural complexity, thereby creating suitable habitat for many UK native reptile species and driving reptile assemblage. Free roaming, low-stocking density grazing creates optimum habitat in intermediate successional stage with high heterogeneity, ideal for reptile populations. Rotational grazing systems do not seem beneficial for heathland restoration or reptile populations, leading to high disparity between grazed and un-grazed areas with little optimum habitat and could be causing conversion from heathland to grassland. However, grazing is not always an applicable management tool. Vegetation with lower heterogeneity may be unsuitable for open-habitat species, but denser heathland supports smooth snake populations and introducing grazing may be detrimental. Although beneficial for the conservation of many species, any grazing management decisions should be carefully considered and modelled before action is taken.

Although Hampshire contains the largest area of lowland heath in England and these sites are supporting high biodiversity of many species, they sit within a highly fragmented landscape. Increasing fragment size and restoring connectivity between them should be top priority for the future to allow metapopulations of reptiles and other taxa to interact. Implementing grazing as a management tool in conjunction with other methods to increase habitat heterogeneity in these areas could be a widely applicable method of achieving this long-term goal.

iv. Appendix

Appendix 1

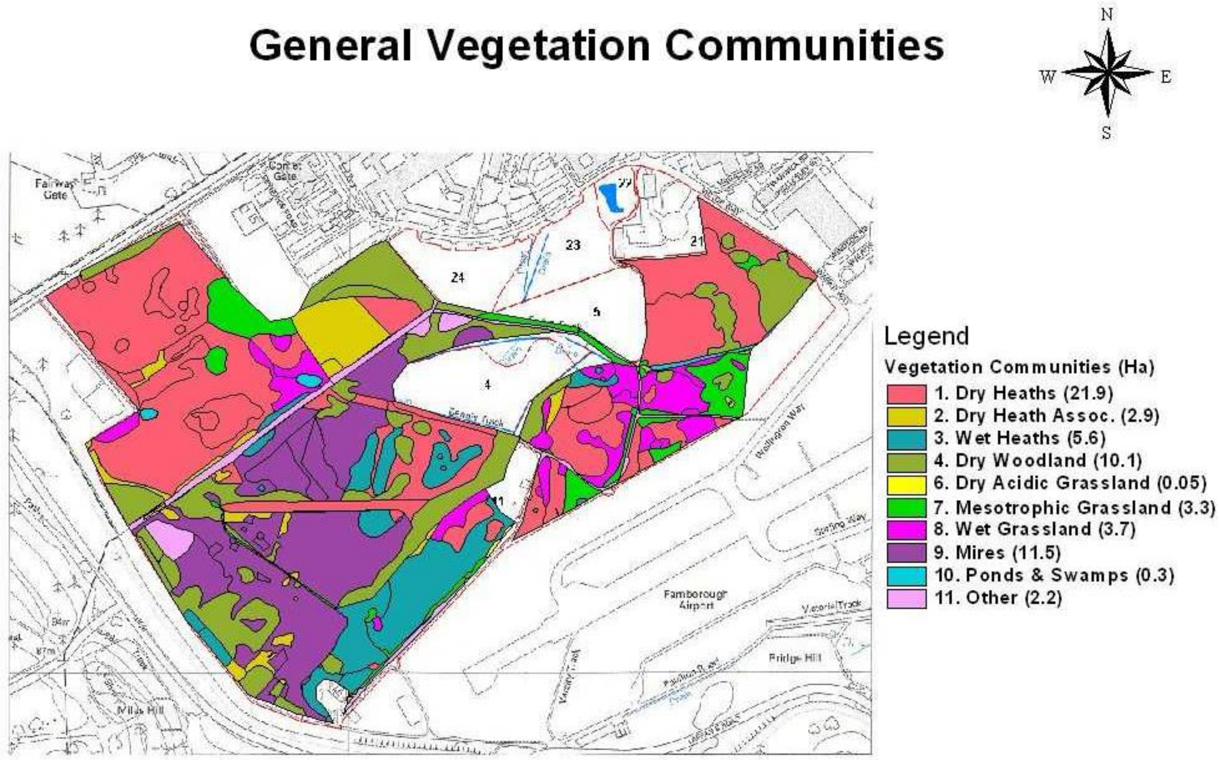

Figure 1: National Vegetation Classification Map of Eelmoor Marsh.

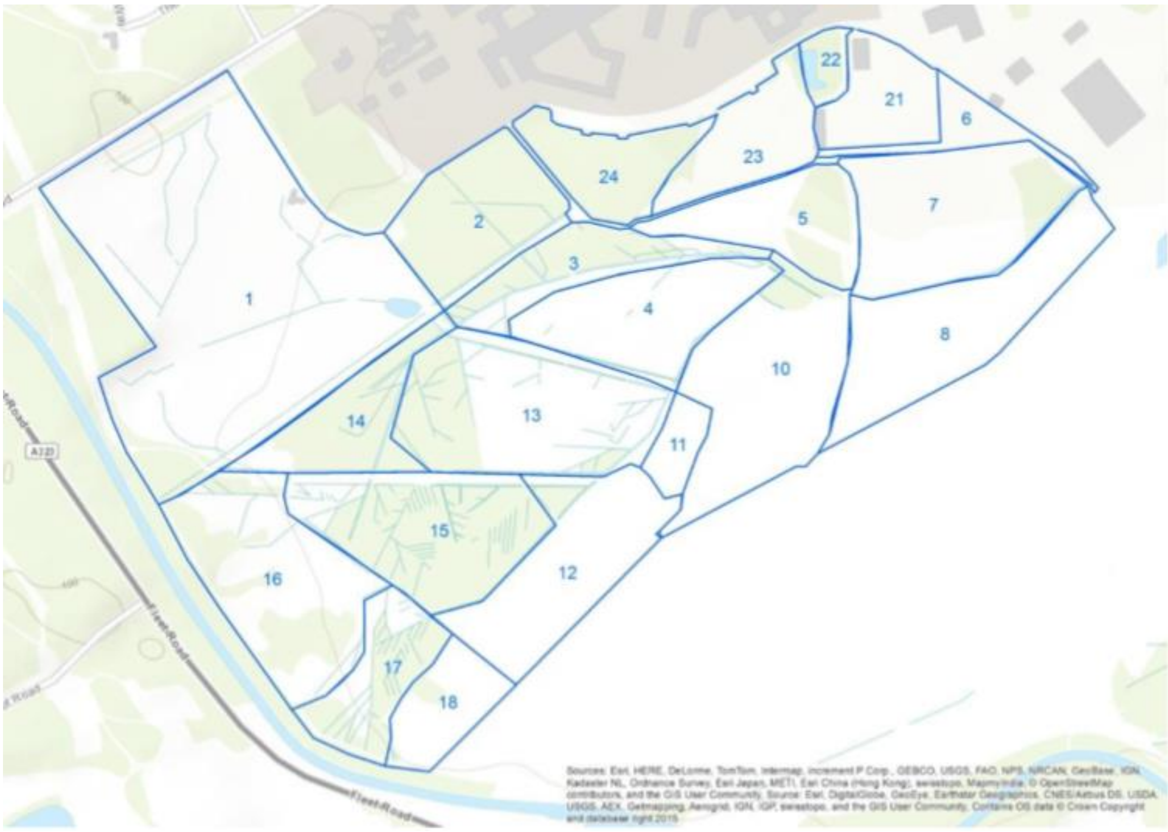

Figure 2: Management Compartments at Eelmoor Marsh. Source: Hall *et al, 2*015.

Table 1: Reptile sightings across Eelmoor Marsh (1996-2003). Source: Hutchins, 2004.

Compartment	Slow Worm	Common Lizard	Grass Snake	Adder
1	√	√	√	√
2			√	
3		√	√	√
4	√	√	√	√
5	√	√	√	√
6				
7			√	√
8		√		√
10	√	√	√	
11	√			√
12		√	√	√
13		√	√	√
14		√		
15	√	√	√	√
16		√		√
17	√	√	√	√
18	√	√	√	√
20			√	
21				
22			√	
23	√		√	√
24	√			

Appendix 2

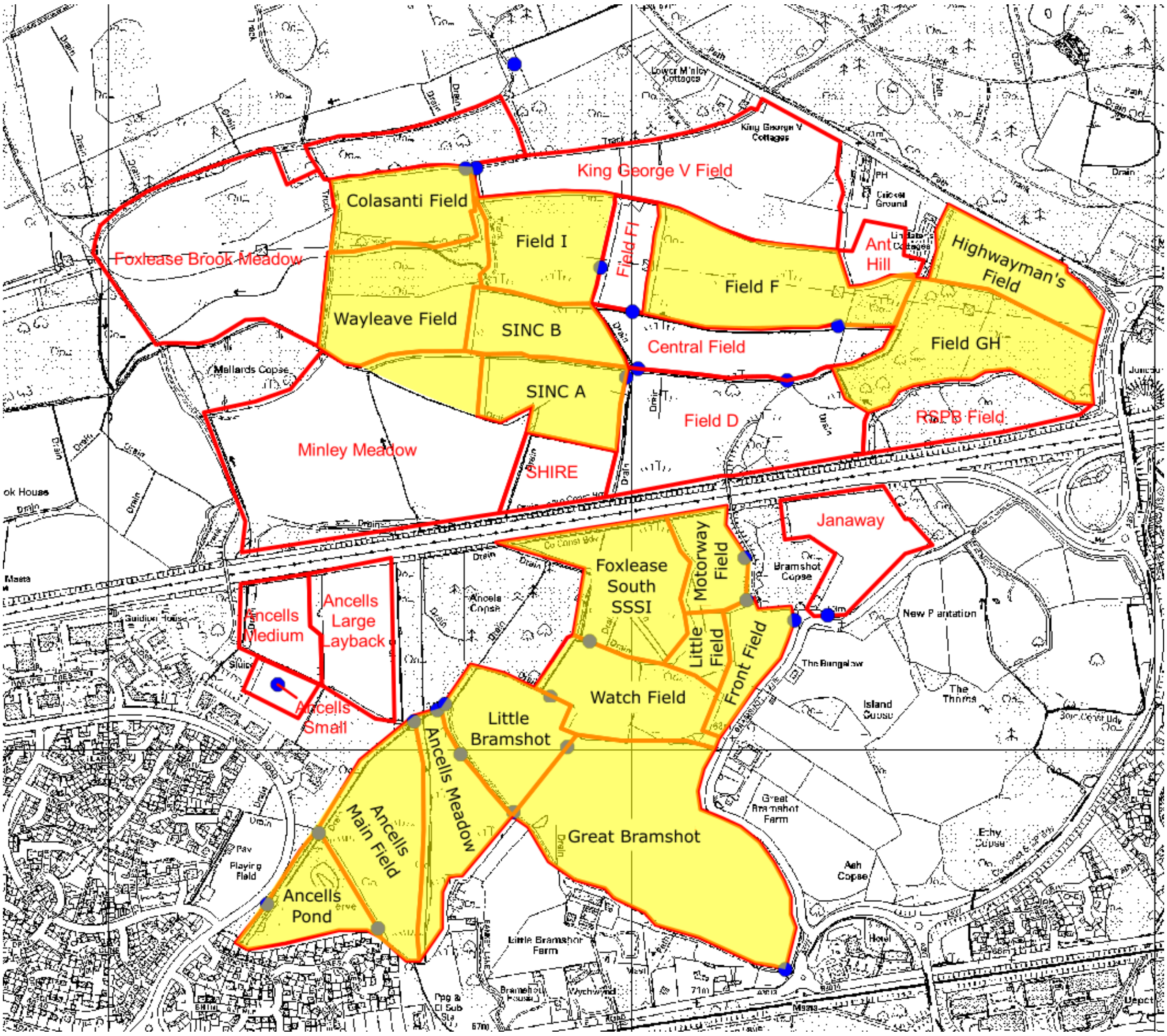

Figure 3: Foxlease and Ancells Meadows SSSI. Red outline: the area contained within the SSSI. Yellow colouring: the fields that are within the rotational grazing management and were surveyed during this study. Source: HIWWT (Richard Hennessey, pers. comm. 2018).

Appendix 3

Table 2: survey schedule

Week	Date beginning	Monday	Tuesday	Wednesday	Thursday	Friday
	12-03-2018		Eelmoor pilot			
	19-03-2018				Eelmoor pilot (visit Foxlease)	
1	16-04-2018	Eelmoor	(tins out at Foxlease)	(tins out at Foxlease)		
2	23-04-2018	Eelmoor	(Visit Flashes)			
3	30-04-2018		Foxlease	Eelmoor		Flashes
4	07-05-2018	Flashes	Fox	Eelmoor		
5	14-05-2018	Eelmoor	Flashes		Foxlease	
6	21-05-2018		Foxlease	Eelmoor		Flashes
7	28-05-2018	Flashes		Foxlease	Eelmoor	
8	04-06-2018	Eelmoor	Foxlease			Flashes
9	11-06-2018			Flashes	Foxlease	
10	18-06-2018		Foxlease			Flashes

Appendix 4

Table 3: classification of vegetation complexity, adapted from Howard and Hailey (1999).

Scale	Description
0	No vegetation
1	Leaf litter
2	Short grassland (<50cm)
3	Long grassland (>50cm)
4	Small sparse shrub (<50%, <1m)
5	Large sparse shrub (<50%, >1m)
6	Small dominant shrub (>50%, <1m)
7	Large dominant shrub (>50%, >1m)

Appendix 5

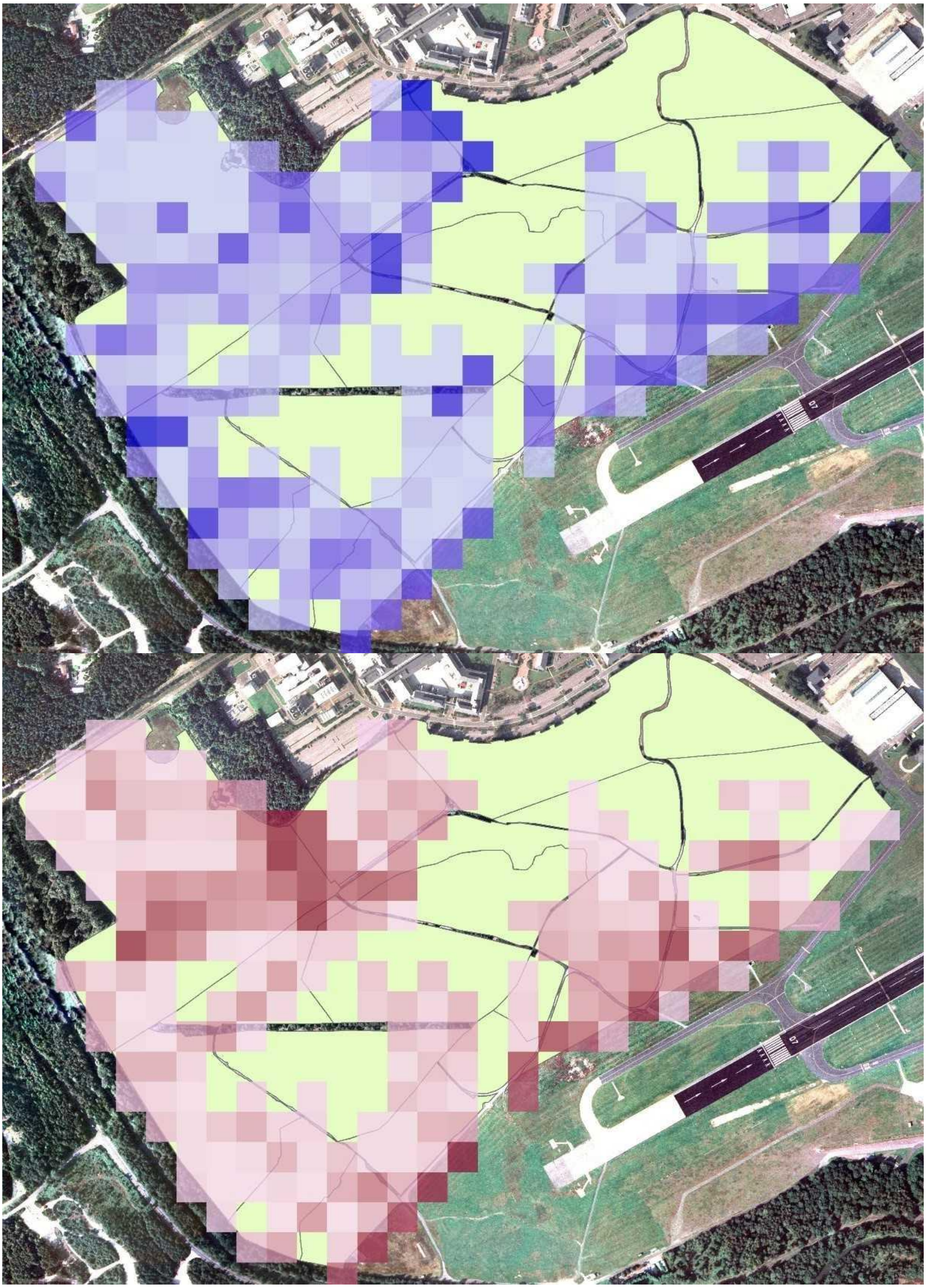

Figure 4: Intensity of cattle (blue) and horse (red) grazing at Eelmoor Marsh (Wilkie 2013).

v. References

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