Conservation value of constructed reedbeds

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Thesis submitted to the University of Nottingham for the degree of Doctor of Philosophy

September 2017

Abstract

Natural reedbed habitat in the UK has suffered severe declines in the last few centuries. With only 5,000 ha remaining and much of what remains is highly fragmented and degraded. The loss and fragmentation of reedbed habitat in the UK has had a profound effect on the species associated with it. Constructed reedbeds are artificial wetlands dominated by *Phragmites australis* designed to treat wastewater. They have been widespread in the UK because they are perceived to be a green solution, but their conservation value has never been assessed. This project aims to determine how they contribute to biodiversity. This study surveyed 24 sites, 12 constructed sites paired to 12 natural sites, for small mammals, moths and stem-dwelling invertebrates. Results show that there is no difference between constructed and natural sites. The indication is that constructed reedbeds, despite their small size, young age and contamination, are just as diverse as natural reedbeds. The implications of this for reedbed and wetland conservation are potentially substantial. There are over 1,000 constructed reedbeds in the UK that are just as diverse as their natural equivalents, which could provide connectivity across unsuitable agricultural and urban landscapes.

Acknowledgements

A sincere and special thank you to my supervisor, Tom Reader for his unwavering support throughout my PhD. Your patience and understanding has been exceptional and your encouragement and positivity is greatly appreciated and has made my PhD a thoroughly enjoyable experience. Also thank you to Rachel Gomes for her positivity and support via coffee and cake!

Special thanks go to Clodagh Murphy at ARM and as part of the Constructed Wetlands Association for all her efforts in helping me get started with my project and introducing me to companies throughout the UK. Thank you to all those companies that gave me site access and to their site managers for their support on site: ARM, Anglian Water, Caerphilly Council, The Coal Authority, Derbyshire County Council, Forestry Commission, the International Fire Training Centre, Nottinghamshire County Council, Sheepdrove Organic Farm, Wigan Council, Wildfowl and Wetlands Trust (Slimbridge, Martin Mere and Welney), the Wildlife Trusts (BBO, BCN, Derbyshire, Lincolnshire, Teeside).

Thank you to all the volunteers that assisted me during fieldwork: especially those students at WWT centres Slimbrdge and Martin Mere and to those from the University of Nottingham.

Thank you to those in the research group, especially those in B109 and in the DTP cohort for their support.

Thanks go to my funders in BBSRC and the Doctoral Training Program at the University of Nottingham.

Last but certainly not least a huge thank you to my parents, sister, grandparents and extended family for their incredible support and encouragement throughout my whole PhD.

Contents page

Abstract …… 3

Acknowledgements …….. 4

Chapter 1: Introduction …….. 9

1.1 Reedbed conservation ……..…10

1.1.1 Wetlands ……...…10

1.1.2 Reedbeds …….. 11

1.1.3 Species associated with reedbeds ……..… 12

1.1.4 Small mammals and invertebrates in reedbeds…….....14

1.2 State of a nation: constructed reedbeds ……... 16

1.2.1 Constructed reedbed history and design in the UK ……16

1.2.2 Expected differences between constructed and natural reedbeds ……18

1.2.3 Contamination and its implications for biodiversity …..22

1.2.4 Biodiversity in Constructed Reedbeds ……..24

1.3 Project aims ……..…25

1.4 Thesis outline ……. 26

Chapter 2: Methods …… 27

2.1 Introduction …… 28

2.2 Methods …….. 30

2.2.1 Site selection …… 30

2.2.2 Survey methods ……... 32

2.2.3 Comparison of habitats in constructed and natural sites .. 33

2.3 Results ……40

2.4 Discussion …….. 45

Chapter 3: Small mammals in constructed and natural reedbeds in the UK …… 47

3.1 Introduction …… 48

3.2 Materials and methods …… 51

3.2.1 Field surveys …… 51

3.2.2 Statistical analysis ……... 52

3.3 Results …… 54
3.4 Discussion …….. 65
3.4.1 Abundance and diversity of small mammals in UK reedbeds ……... 66
3.4.2 Species composition …… 67
3.4.2.1 Harvest mice ……. 68
3.4.3.2 Other species …… 70
3.4.3 Size and age of constructed reedbeds …….. 72
3.4.4 Other factors affecting small mammals in UK reedbeds ……... 74
3.5 Conclusions …… 74

Chapter 4: Moths in constructed and natural reedbeds in the UK …… 77
4.1 Introduction …… 78
4.2 Materials and methods …… 82
4.2.1 Field surveys …… 82
4.2.2 Statistical analyses ……..… 84
4.3 Results …… 85
4.4 Discussion …….. 96
4.4.1 Species composition …… 96
4.4.2 Age and size …… 98
4.4.3 Environmental effects on moth diversity, species richness and abundance ……. 99
4.4.4 Other factors affecting moth abundance, species richness and diversity ……... 100
4.5 Conclusions …….. 100

Chapter 5: Stem-dwelling invertebrates in constructed and natural reedbeds in the UK …….. 103
5.1 Introduction …….. 104
5.2 Materials and methods ……. 107
5.2.1 Field surveys ……. 107
5.2.2 Statistical analyses ……. 107
5.3 Results …….. 108

5.4 Discussion …… 117
5.4.1 Species composition ……..…. 117
5.4.2 Abundance of stem-dwelling invertebrates in reedbeds in the UK …… 118
5.4.3 Age and size ……... 119
5.4.4 Habitat associations ……...… 120
5.5 Conclusions …….. 120

Chapter 6: Constructed and natural reedbed biodiversity in the UK ….. 121
6.1 Introduction ……...… 122
6.2 Methods ……..... 123
6.2.1 Field surveys …….. 123
6.2.2 Statistical analyses ……. 124
6.3 Results …….. 125
6.4 Discussion …… 135
6.4.1 Age and size …….. 135
6.4.2 Community composition ……... 136
6.4.3 Indicator taxa ……. 136
6.5 Conclusions: the conservation value of constructed reedbeds ….. 137

References ……. 142

Appendix 1 Species list **…… 153**

Appendix 2 BBSRC Doctoral Training Programme **…… 159**

Chapter 1: General introduction

1.1 Reedbed conservation

1.1.1 Wetlands

Wetlands are one of the most threatened biomes in the world. It is estimated that half of the world's wetlands have been lost since 1990, with 20% of the world's freshwater species being extinct, threatened or endangered in that time (Lee, 2015). Freshwater habitats are thought to sustain over a third of all vertebrate species yet they only cover 0.8 % of the Earth's surface (Dudgeon et al., 2006). Wetlands overall, including brackish and saltwater habitats, cover less than 9 % of the world's surface and much of what remains is extremely degraded (Zedler and Kercher, 2005).

Wetlands are one of the most valuable natural habitats, economically and ecologically, per unit area in the UK (Hansson et al., 2005). Their economic value comes through ecosystem services such as water quality improvement, biodiversity support, flood alleviation and carbon management (Zedler and Kercher, 2005). Those ecosystem services provide many jobs, as do conservation management and wetland visitor attractions (Dickie, 2001). Wetland ecosystems not only provide direct economic benefits, but they also provide indirect benefits in terms of nutrient retention and preventing eutrophication and redundancy of many waterways. Hence, wetlands help to maintain waterways as a resource for humans and a habitat for organisms (Hansson et al., 2005).

The recent intense scientific interest in biodiversity can be traced back to the Convention on Biodiversity in 1992, which highlighted international concern over species loss and habitat degradation (Junk et al., 2006). There has been an increased interest in wetland biodiversity since 1996 after the International Wetlands Conference of INTECOL. This conference highlighted the dearth of biodiversity research in wetland habitats, the increased concern over the loss and degradation of wetlands, and the rapid rate of wetland species decline (Junk et al., 2006). Though the knowledge of total wetland diversity is limited, wetlands often support a complex mixture of wetland specialist species, along with

terrestrial species from the surrounding habitats for which wetlands can offer a refuge (Junk et al., 2006). Wetland habitat loss is thought to be responsible for the extinction or threatened status of many British invertebrates, one third of which are wetland specialists (Rispoli and Hambler, 1999).

1.1.2 Reedbeds

Reedbeds are areas of wetland dominated by common reed (*Phragmites australis*) (BRIG, 2008). Reedbeds were once extensive in the UK, especially in the low-lying coastal plains of East Anglia (Hawke and Jose, 1996). However, reedbed habitat was reduced over a long period as a result of land drainage and vegetation succession, with estimated losses of between 10 – 40 % between 1945 and 1990 (Hawke and Jose, 1996). Reedbeds in the UK are of high conservation value in a global context because they are some of the farthest west in Europe. Although reedbeds in the UK tend to be much smaller than many found in mainland Europe they host high species diversity per hectare (Hawke and Jose, 1996). The remaining habitat is intensively managed and that in England is highly fragmented; there are approximately 5,000 ha of reedbed in England with only 50 of 900 sites over 20 ha (BRIG, 2008).

This fragmentation and vulnerability has led to reedbeds being given priority habitat status under the UK biodiversity action plan (BAP[1]) (BRIG, 2008). In addition many are designated Sites of Special Scientific Interest (SSSI) and the larger reedbeds and wetlands in the UK are also protected under International law, such as the Ramsar Convention with sites designated as Wetlands of International Importance (Paul et al., 2011). The Ramsar Convention "recognises that the loss of wetlands would be irreparable, and combines far-sighted national policies with coordinated international actions to secure healthy wetland ecosystems" (Lee, 2015). The Ramsar Covention targets especially large expanses of wetland, which include reedbeds. However specific protection of reedbed habitat is often underpinned by SSSI assignment; this provides protection under the Wildlife and Countryside Act 1981 (JNCC, 2015). Protection is also implemented through the UK BAP system which encourages

[1] The U.K. BAP system was incorporated into the U.K. post-2010 biodiversity framework in 2012, any further mention of the U.K. BAP system also refers to this.

the creation and enhancement of designated habitats and habitats for those species with BAP protection, of which wetlands and their associated species are a part (Birol and Cox, 2007).

1.1.3 Species associated with reedbeds

The loss and fragmentation of reedbed habitat in the UK has had a profound impact on the species that are associated with it. The bittern (*Botaurus stellaris*) is one of the most notable reedbed specialists, and it has suffered a severe decline due to habitat loss and fragmentation. The number of booming bitterns was reduced to 11 in 1997, from approximately 70 in the 1950s (Gilbert et al., 2005). Other reedbed and wetland specialists that have suffered noticeable declines are the harvest mouse (*Micromus minutus*) and water vole (*Arvicola amphibius*). A national harvest mouse survey carried out in 1999 only recorded harvest mouse nests from 29 % of those sites in which the species was found in the 1970s. Twenty-four percent of this loss was due to lack of suitable habitat (Flowerdew, 2004). The most recent water vole survey indicates that they have been lost from two thirds of the sites compared to 1989 - 1990, with the cause of decline attributed to loss and fragmentation of habitat (Carter, 2003). In addition it is thought that reedbeds provide a crucial refuge for water vole against the predation of the invasive species of American mink (*Mustela vison*) (Carter, 2003).

The species that are associated with reedbeds are often flagship species for conservation. Flagship species are attractive and charismatic, and can be used to focus public and policy maker attention on particular habitats. For reedbeds, flagship species include the bittern and the marsh harrier (*Circus aeruginosus*) (Feest et al., 2012). But reedbeds are an important habitat for many other species, including several passerine birds such as bearded tit (*Panurus biarmicus*), reed warbler (*Acrocephalus scirpaceus*) and sedge warbler (*Acrocephalus schoenobaenus*) (Valkama et al., 2008), and over 700 species of invertebrates (Hawke and Jose, 1996). Sixty-four insect species are dependent on common reed to some degree and 40 species of those are entirely dependent on it (Hawke and Jose, 1996).

Many of the reedbed-associated species in the UK have special conservation value, including six nationally rare Red Data birds and five GB Red Data Book invertebrates (BRIG, 2008). A recent study in three reedbeds in the UK found nine invertebrates on the UK National Red List and 56 rare or nationally scarce species (Hardman et al., 2012). In addition it also found five Lepidoptera (moth) species that are on the UK National Red List and 16 rare or nationally scarce species.

The reedbed-associated species of special conservation value are the ones that have driven the revitalisation of natural reedbeds, with up to 1,000 ha of reedbed for conservation being restored or created in the last two decades (Hardman et al., 2012). The motivation for much of this expansion, and especially the attempt to re-establish populations of breeding bitterns in the United Kingdom, is in the UK BAP system. The bittern has greatly benefited from this system which was established between 1995 and 1999. With the conservation and the creation of suitable habitat through the BAP system, the population has recovered from its all-time low in 1997, and there were 156 booming males recorded in the UK in 2015 (Hayhow et al., 2016). Reedbed birds are relatively well studied (Hardman et al., 2012) and so the effects of the habitat restoration and creation on their populations are quite well understood. However, the effects of these conservation efforts on other species remain unclear, although it is thought that they have been successful not just for bitterns but for other UK BAP species too (Wotton et al., 2009).

The management of reedbeds often centres around bitterns and other bird species (Fisher et al., 2011). For example, habitat association studies revealed that bitterns benefit from "wet" reedbeds with ditches and pools (Gilbert et al., 2005). The creation of wet reedbeds has also proven successful for several important invertebrate species, notably some threatened moths (Wotton et al., 2009). Often management aims to slow the successional transition of wet reedbeds to dry reedbeds, which can develop into willow carr or scrub. However there is evidence that these transitional stages are of benefit to species of conservation note too. Hardman *et al.* (2012) found that drier, older reedbed supported more

invertebrate species of conservation value and that sites with lower water levels had more reedbed specialist species.

1.1.4 Small mammals and invertebrates in reedbeds

Reedbeds are important for small mammal communities; they are the most diverse wetland habitat for small mammals (Scott et al., 2008, Slabova et al., 2008). They are especially important for the harvest mouse, which is a UK BAP priority species. Although the common name of this species reflects the fact that it is traditionally thought to be associated with farming, its habitat preferences are poorly known. There is evidence that harvest mice thrive in reedbeds: for example, nearly 90 % of individuals were caught in tall reed stands when studied in a habitat mosaic in Austria (Haberl and Krystufek, 2003) and recent studies from the UK show an association with wet reedbeds (Kettel et al., 2016). This suggests that they should benefit from the increased area of habitat created through the UK BAP system, especially considering that harvest mice, unlike other flagship species, require less than one hectare of reed to sustain a population (Hawke and Jose, 1996). The current conservation status of harvest mice is unknown, but they are thought to have been in decline up until the late 1990s (Flowerdew, 2004). More research is required to understand fully their distribution and conservation status.

The distribution and conservation status of most invertebrates associated with reedbeds are even less well understood than those of the harvest mouse. Aquatic macroinvertebrates are well studied because they are often used as an indication of water quality (Spieles and Mitsch, 2000), but terrestrial reedbed invertebrates of conservation concern are not as well documented. However, evidence suggests that newly created or restored reedbed habitat - the type encouraged through the UK BAP system - in the vicinity of established reedbed can provide suitable habitat for invertebrate species of conservation value (Hardman et al., 2012). Of the 40 species that are entirely dependent on common reed, there is a group of endophagous herbivores, several of which are UK BAP and Red Data species, such as the gall-forming fly *Lipara similis*. The ecological relationships between species and the effects of the herbivory on *Phragmites* are well understood (Tscharntke, 1999) but their distribution in the UK is not.

Another group of invertebrates that includes several species of conservation importance in reedbeds is moths (Lepidoptera). Many species that are reedbed or wetland specialists have been classified as common and widespread but rapidly declining (a 75 % decrease in abundance over 40 years) (Fox et al., 2013). The reasons for their decline are not fully understood and there is an urgent need for more research to understand fully the reasons for their decline (Fox et al., 2013). More specifically, there are nine species of moth that feed solely on *P. australis*; five of these are stem-boring wainscots, of which two are Red Data species because of their limited geographical distribution on the Norfolk and Suffolk coasts (Clements, 2000).

It has been suggested that bitterns require up to 20 ha of reedbed to breed, which is part of the reason why conservation efforts have often emphasised the importance of large areas of habitat dominated by common reed (Hawke and Jose, 1996). However, species like the reed warbler, harvest mouse and the reed invertebrates can survive in areas of reed less than a hectare (Hawke and Jose, 1996). This means that smaller reedbeds are still of conservation value, especially in the UK with harvest mice and several reedbed invertebrates being UK BAP species.

Even with the current expansion of reedbeds for bittern conservation there is still a demand to learn more about reedbeds and what influences species presence and populations. This is particularly true of mammal and invertebrate populations in small and marginal habitats. This demand is not just for those reedbeds managed for conservation purposes. Many reedbeds have been and continue to be created across the UK with a variety of uses. The need to understand the conservation value of smaller reedbeds is made all the greater by the creation of these small artificial reedbeds.

1.2 Constructed reedbeds

1.2.1 Constructed reedbed history and design in the UK

Reedbeds have the ability to improve the quality of surface and ground water, and as a result reedbeds are increasingly used in water treatment (Kobbing et al., 2013, Wang et al., 2010). Reedbeds that have been planted with specialist drainage networks and with water treatment in mind are typically referred to as "constructed" reedbeds. These constructed reedbeds are the focus of this thesis.

The first experiments to consider using wetland plants to treat wastewater were conducted in Germany in the 1950s (Vymazal, 2010). The first constructed reedbed was built in the UK in 1985 but their proliferation in the UK started 20 years ago, with only a few hundred built before this (Cooper and Green, 1995, Hawke and Jose, 1996). Constructed reedbeds are designed to treat a range of wastewater types, such as sewage, mine-water, sludge, industrial effluents and surface and road runoff (Cooper, 2007). In 2007 there was a record of 1,012 constructed reedbeds in the UK, but the estimated total was 1,100 to 1,200 (Cooper, 2007). My personal observations suggest that the number is now much higher, but no systematic database of sites exists. The average size of a constructed reedbed in the UK is thought to be less than 0.25 ha (Hawke and Jose, 1996). Over 85 % of the constructed reedbeds in the UK treat sewage, with the next most common wastewater being mine water (Cooper, 2007).

Constructed wetlands in the UK can be broadly split into horizontal and vertical flow reedbeds, and those with subsurface or surface flow. Horizontal flow systems have wastewater fed into them so that the water runs in a horizontal path, whereas in vertical flow systems the wastewater is stored and pumped either down through the gravel or filled from the bottom in batches (Vymazal, 2010).

Generally, both horizontal and vertical flow systems are lined with an impenetrable layer and filled with gravel. The *Phragmites australis* that is planted in these systems helps suspended solids to be retained through sedimentation. The roots and rhizomes provide a transport system for oxygen

and a substrate on which bacteria can grow, producing the right conditions for aerobic degradation of the wastewater (Vymazal, 2010). The rhizosphere helps with the reduction of the biological oxygen demand (BOD) and chemical oxygen demand (COD), which are important indicators of pollution (Bhatia and Goyal, 2014).

The main difference between horizontal flow and vertical flow systems is that the latter are drained periodically, enabling the diffusion of oxygen into the sediments, providing the conditions for aerobic degradation. Vertical flow systems are often more efficient and for this reason are used for primary and secondary treatment of wastewater, whereas horizontal flow systems are commonly used in tertiary treatment ("polishing") of already-treated and therefore relatively clean water. Both types of system are very efficient at removing suspended solids, but they differ in their ability to remove nitrogen (Vymazal, 2010). Vertical systems provide the aerobic conditions for high nitrification. On the other hand horizontal flow systems provide the conditions for denitrification, but neither system does both efficiently. Phosphorous removal in all types of constructed wetland is low as the primary sink is in soil. Heavy metals are removed in a similar way to phosphorous in that they precipitate out of the water and sediment out, as well as being absorbed and accumulated in the plants (Bhatia and Goyal, 2014).

Due to the different abilities of horizontal flow and vertical flow systems to treat wastewater they are often used in conjunction with each other (Vymazal, 2010). A constructed wetland treatment system will often have more than one component. Often they start with settling pools which flow into several treatment 'beds', which in Europe are typically either dominated by *Phragmites australis* or a mixture of *Typha latifolia* and *Iris pseudacorus* (Vymazal, 2010). Occasionally at the end of a treatment system there will also be a more "natural" pond that has vegetated edges and which is designed to benefit wildlife (personal observation).

1.2.2 Expected differences between constructed and natural reedbeds

The communities of animals and plants present in constructed and natural reedbeds are expected to differ for a number of reasons. Constructed reedbed communities are expected to be depauperate because of their young age, their small size, the fact that they are often surrounded by industrial or urban land, their contamination and perhaps a lack of management.

Island biogeography theory suggests that isolated islands of habitat will have lower equilibrium species richness because smaller habitat patches will have smaller population sizes and hence higher extinction risk and isolated habitat patches have reduced colonisation (Simberloff, 1976). Biogeography theory suggests that habitat fragmentation can lead to a reduction in diversity due to: loss of habitat, reduced habitat patch size and the isolation of habitat patches (Wilcox and Murphy, 1985).

Constructed reedbeds are regularly surrounded by a matrix of unsuitable habitat in urban and industrial settings and they are often built far from naturally occurring wetlands. As such constructed reedbeds could be considered isolated islands. Although the same could be said for natural reedbeds as they are highly fragmented in the UK. Additionally by their nature as a wetland habitat the plants and animals often associated with reedbeds require waterlogged soils and any surrounding habitat is hence inhospitable.

This highlights the importance of the density of wetlands in a habitat mosaic, which constructed wetlands may significantly influence. There is some evidence to suggest that constructed reedbeds can provide habitat connectivity for macroinvertebrates (Becerra-Jurado et al., 2011, Thiere et al., 2009). There is evidence from Europe that suggests the size and proximity of the closest wetland can have implications for diversity. One study found that the distance to the nearest reedbed over a hectare correlates positively with the abundance of species and diversity in a reedbed patch (Orlowski and Gorka, 2013).

These theories can be applied to the matrix of constructed reedbeds across the UK. The majority of constructed reedbeds in the UK are less than 0.25 ha (Hawke and Jose, 1996) and although it could be expected that the smaller and more isolated constructed reedbeds in the UK may be depauperate compared to their larger natural equivalents, they may still contribute significantly to wetland conservation.

The smaller constructed wetlands in the UK potentially provide suitable habitat connectivity to enable the persistence of metapopulations. The original concept of metapopulations was present in the literature from the 1970s, although similar theories were present beforehand they remained largely ignored before Levins' series of papers describing a population of populations (Hanski and Gilpin, 1991, Levins, 1969, Levins, 1970, Levins and Culver, 1971). Hanski and Gilpin (1991) review describes three spatial scales; local scale, metapopulation scale and geographical scale. Metapopulation scale being defined as "The scale at which individuals infrequently move from one place (population) to another, typically across habitat types which are not suitable for their feeding and breeding activities, and often with substantial risk of failing to locate another suitable habitat patch in which to settle". This concept becomes increasingly important with the consideration of habitat loss and the distance between these populations. Consequently, in a fragmented network of extinction-prone sub-populations, habitat patch connectivity plays a key role in facilitating the dispersal required to allow the persistence of the whole metapopulation (Gibbs, 1993). Hence, there is evidence to suggest that individual small wetlands may provide the connectivity to maintain whole wetland metapopulations (Gibbs, 1993). This is especially true for habitat specialists. It is thought that monophagous herbivores are especially vulnerable to habitat fragmentation due to their inability to exploit other habitats across a landscape (Tscharntke et al., 2002). Research supports the idea that connectivity reduces the risk of local extinctions in natural reedbeds, especially of small mammals and birds (Gibbs, 1993). There is also evidence to suggest that small areas of habitat across a landscape are just as important, if not more important, than large areas of habitat, which are often the focus of current conservation strategy (Tscharntke et al., 2002). Thus, even if individually they are small, if they contribute at all to wetland biodiversity, the overall impact of

constructed reedbeds in the UK could be very substantial, because there are more than 1,000 constructed sites, compared to ~ 900 natural sites (Cooper, 2007, BRIG, 2008).

Constructed reedbeds in the UK are often found in a matrix of industrial, urban and agricultural habitats (personal observation) which may leave them relatively isolated from natural wetlands which might serve as a source of colonising species. This may have contrasting effects on the conservation significance of constructed reedbeds, as compared with small natural reedbeds. On the one hand, they may have particularly low species richness. As expected from island biogeographic theory, there is evidence to suggest that the extent to which a reedbed is isolated (e.g. with no other reedbeds in close proximity) will impact on colonization and the initial establishment of reedbed specialists (Biggs et al., 2005). On the other hand, constructed reedbeds may contribute disproportionately to the total area of reedbed or wetland habitat in areas of the country where natural habitat has either been mostly drained or was not present in the first place.

The proliferation of constructed reedbeds only started around 20 years ago in the UK (Hawke and Jose, 1996, Cooper and Green, 1995). Although natural reedbeds have suffered severe loss, deterioration and fragmentation in the UK over the last century or more, and much "natural" habitat has been extensively managed or rehabilitated, they are still typically considerably older than constructed reedbeds (Hawke and Jose, 1996). The amount of time for colonisation could be expected to impact species diversity with younger habitats being less diverse than older ones (Valente et al., 2014). There is evidence from constructed reedbeds to suggest that species-rich invertebrate assemblages similar to those found in natural reedbeds may take up to 10 years to develop, although some of the first reedbed invertebrate specialists are expected to be present after 3 years (Bogusch et al., 2016, Athen and Tscharntke, 1999). Some species may take much longer than this to colonise, and this may explain why in the UK many rare reedbed specialists (e.g. *Lipara similis*) are only found in the remnants of ancient wetland habitat found in East Anglia.

Constructed reedbeds may also differ in their species composition compared to natural equivalents because they are young *and* artificial. Often diversity is different in artificial man-made habitats because they are young, and because initial vegetation composition is directly determined by humans. The rate and extent to which species colonize a barren/artificial habitat determines the community composition that develops, and the end result may differ from that found in a natural habitat (Pratt, 1994). Even if the plant species which would naturally dominate are present, and species colonization takes place, ecosystem function will only develop over time. Constructed reedbeds are typically planted with all *P. australis* and are managed (although see below) to maintain this dominance.

Despite the artificial nature of constructed reedbeds, the deterioration of natural reedbeds throughout the UK and mainland Europe may mean that constructed reedbed communities are not actually much less diverse. Evidence suggests that constructed wetlands created for habitat enhancement can have better macroinvertebrate species diversity and functionality (e.g. nutrient cycling, biomass decomposition, primary productivity) than the surrounding natural degraded habitat (Espanol et al., 2015). Ironically, this may be because natural reedbeds have become eutrophic, with high phosphorous levels, as a result of inadvertent pollution, leaving them in a state which is not dissimilar to that of constructed reedbeds used for treating sewage.

The build-up of sediment in the constructed reedbed systems means that often they are managed by periodic harvesting of the reed and dredging of the sediment. Both these processes can be destructive to the ecosystem that is developing, especially dredging (Nahlik and Mitsch, 2006). Similar management is sometimes used in natural reedbeds. Reed cutting is used to maintain reedbeds in their current state and prevent natural succession to wet woodland (Valkama et al., 2008). Dredging is used to increase the water capacity of reedbeds that have filled up with sediment, although in both natural and treatment wetlands this is typically a last resort (Nahlik and Mitsch, 2006). Although many constructed sites in the UK have management plans, often the reality is that these do not get implemented, and more drastic large scale and

therefore destructive management techniques such as dredging are more common than those less destructive ones such as reed cutting (personal observation).

1.2.3 Contamination and its implications for biodiversity

Despite the potential for constructed reedbeds to contribute to biodiversity, concern has been raised about attracting wildlife to a potentially toxic environment (Knight, 1997). Their use in wastewater treatment could influence the number and/or type of species found in constructed reedbeds. Evidence from studies in the UK suggest that the size, pH and concentrations of toxins can all affect the diversity of aquatic macroinvertebrates found in a constructed reedbed (Batty et al., 2005). Batty et al. (2005) suggest that although these aspects that reduce diversity are part of the treatment system, minor alterations to constructed reedbed design could result in the ability to support more species towards the system outlet. Even without these alterations, it is thought that the macroinvertebrate community in the constructed wetlands studied is sufficient to support organisms at higher trophic levels.

Constructed wetlands are often designed with a series of ponds and beds, and there is evidence that waterfowl are more inclined to use the areas of constructed wetlands closest to the outlets. This could reflect a gradient in the abundance of food and the *naturalness* of the habitat from the more contaminated to the less contaminated end of a site (Hamilton et al., 2005, Orłowski, 2013). The abundance of food may be affected by the contaminants in the system. Where in the reedbed these contaminants accumulate may indicate which species they impact and might result in differences between constructed and natural sites.

P. australis bioaccumulates heavy metals, although not to phytotoxic levels (Rzymski et al., 2014). Nevertheless those elements accumulating in the leaves are then accessible to other organisms with a potential to contaminate the food chain (Rzymski et al., 2014). Evidence suggests that *P. australis* accumulates most of the elements in its roots (Alhashemi et al., 2011, Rzymski et al., 2014). Additionally it seems that *P. australis* has the ability to move metals from roots to leaves at the end of the growing season into the senescent tissues (Bragato et

al., 2009). This may be a potential mechanism to remove toxic elements from the plant, since the leaves are shed in autumn.

Herbivore species may be most at risk from the accumulation of toxins in reeds, as they will take up an excess of the essential metals, potentially affecting their growth (Cheruiyot et al., 2013). There is a tendency for those metals that are essential to bioaccumulate through the food chain whereas those that are not will decrease at higher trophic levels (Zhuang et al., 2009). The requirement for an essential element will affect how much of that element invertebrates will take up in the first place (Wilson and Pyatt, 2007). Any bioaccumulation by invertebrates also then makes the metals available to predators. There is thus an expectation that metals will biotransfer through the food chain and the increased availability may lead to bioaccumulation and growth and development deficiencies (Cheruiyot et al., 2013). The foraging behaviours of various species in the same guild can result in different accumulation rates (Yigit and Altindag, 2006). The feeding mode may also affect the amount of contaminant received by the predator; those that chew or swallow their prey are more likely to receive a higher dose of the contaminant than other mechanisms (Cheruiyot et al., 2013).

Despite the concerns about treatment wetlands accumulation of heavy metals and other chemical contaminants through the food chain, a review suggests that with the correct protocols in place these areas can be maintained as important habitat for the species that use them without causing health risks (Murray and Hamilton, 2010). Additionally, this review states that understanding these constructed systems is essential if we are to protect species and maintain the use of constructed wetlands as both functioning water treatment systems and habitats (Murray and Hamilton, 2010).

There is also evidence that bioaccumulation of metals in artificial wetland ecosystems does not necessarily affect biodiversity. A study of stormwater wet detention ponds (SWDPs) showed that invertebrates found in these SWDPs accumulated more metals than invertebrates in the surrounding natural shallow lakes, but there was no difference in biodiversity (Stephansen et al., 2016). This

study showed that most SWDPs had similar community composition to natural lakes despite their contaminants.

1.2.4 Biodiversity in constructed wetlands

Constructed wetlands are primarily created for the treatment of water. Biodiversity and conservation benefits are often a coincidence or at most a secondary aim of the system. Plant species diversity is often a target of constructed wetland biodiversity studies because plants are crucial to the function of the wetland treatment system (Vymazal, 2010). There is evidence from some constructed wetlands that plant diversity is not necessarily beneficial in terms of ecosystem functioning and nutrient removal (Weisner and Thiere, 2010). However a long term study found that plant diversity increased rapidly over time from 12 species at the start of the survey to 63 species over 9 years later (Yang et al., 2006). With this increase in species diversity and the maturation of the ecosystem there was an increased capacity of the wetland to treat wastewater and the water quality improved.

Another focus of biodiversity studies in constructed wetlands is often aquatic macroinvertebrates. Aquatic macroinvertebrates are regularly used as an indicator of water quality (Spieles and Mitsch, 2000). They also provide an indication of wetland function and so may denote wetland health (Balcombe et al., 2005). The result of using macroinvertebrates as indicators in constructed wetlands is obviously influenced by the type of wastewater treatment involved. For example, high nutrient loads often reduce invertebrate diversity, although other factors such as age also play a part in determining diversity (Hansson et al., 2005). Higher densities and diversity of planktonic invertebrates occur in the final stages of a wetland treatment system (Hamilton et al., 2005). This correlates with water health indicators improving through the stages of the treatment system. In large water treatment systems it is possible that the later stages are able to support a longer food chain with the number of bird species increasing towards the later stages also (Hamilton et al., 2005).

The use of constructed wetlands by birds has also been fairly well documented, probably because natural reedbeds are widely known to be an important habitat for birds. The importance of constructed wetlands as a genuine alternative to the

depleted natural habitat for water birds has been observed (Murray and Hamilton, 2010). There is even evidence that constructed wetlands can contribute to the improved conservation status of some birds, with nutrient removal wetlands specifically having high species richness (Strand and Weisner, 2013).

Studies that cover multiple taxa in constructed reedbeds are limited, and most studies focus on only plants, aquatic macroinvertebrates or birds. A recent study found that wetland area was the key influential factor affecting biodiversity of several groups including birds, fish, aquatic macroinvertebrates and macrophytes (Hsu et al., 2011). High cover of aquatic macrophytes was also beneficial to all taxa. Despite this, various additional influences on species richness and abundance were detected for each taxon, which suggests that generalising from studies of just one taxon may be problematic.

1.3 Project aims

Large scale studies that assess the impact of constructed wetlands on conservation at a landscape level are rare. The majority of studies on constructed wetlands are small-scale (typically considering only one site) and rarely are *Phragmites* reedbeds targeted specifically. With the threatened status of reedbeds nationally, and the increasing popularity of constructed wetlands for water treatment, the potential contribution that constructed reedbeds could make to UK biodiversity is substantial. This project aims to assess the role of constructed reedbeds in conserving wetland species and their habitat. The primary role of these systems will always be to clean water but, with a greater understanding of what influences the use of these systems by wildlife, there is potential to manage and design them to maximise their conservation value. This project focusses on three groups of species which are likely to be diverse and of conservation significance in wetlands, but which have been almost entirely overlooked in studies of constructed reedbeds: small mammals, moths and reed stem-dwelling invertebrates.

The literature indicates that there could be an expectation that the typical attributes of constructed reedbeds discussed throughout this chapter may have impoverished fauna compared with natural equivalents. First, they are young, so the time available for colonisation has been limited. Second, they are small, so they are expected to support small population sizes, and have a lower equilibrium species richness than larger reedbeds. Third, they are potentially polluted, which may be detrimental to some parts of the animal community. Finally, they are often located in a matrix of industrial, urban and agricultural habitats which may leave them relatively isolated from natural wetlands which might serve as a source of colonising species. The expectation therefore is to find reduced diversity and species richness across all taxa in constructed reedbeds.

1.4 Thesis outline

In the remainder of this thesis, I aim to answer the following questions:

- How do constructed sites differ from natural sites in their size, age and other habitat characteristics? (Chapter 2)
- How do constructed reedbeds contribute to small mammal diversity? (Chapter 3)
- How do constructed reedbeds contribute to moth diversity? (Chapter 4)
- How do constructed reedbeds contribute to stem-dwelling invertebrate diversity? (Chapter 5)

Finally I compare diversity across all taxa and determine the overall conservation value of constructed reedbeds (Chapter 6).

The chapters in this thesis have been prepared for publication and as such there is small amounts of repetition in the introductions of the chapters.

Chapter 2: Methods

2.1 Introduction

With drainage for agriculture and development, truly natural reedbeds have long been a rare feature of the British landscape. Most large reedbeds now in existence are heavily managed, for conservation or commercial purposes, and many have been planted from scratch in recent years (Hawke and Jose, 1996). A clear distinction can be made between those reedbeds planted and/or managed for conservation purposes, and those that have been planted with specialist drainage networks with water treatment in mind. This thesis will refer to the latter category as "constructed" reedbeds. These constructed habitats have been designed to treat sewage wastewater, mine-water, sludge, industrial effluents and surface and road runoff (Cooper, 2007). "Natural" reedbeds in this study are those managed for conservation purposes. All of those sampled were actually relatively young, and had been planted at some point in the last 45 years.

The first constructed reedbed was built in 1985 (Hawke and Jose, 1996) but their proliferation in the UK started 20 years ago, with only a few hundred built before this (Cooper and Green, 1995). This was at least a decade after a significant expansion of natural reedbeds was seen in the UK, after it was recognised that natural reedbed and wetland habitat had been reduced over a long period as a result of land drainage and vegetation succession, and that action was required to protect and manage existing habitat, and create new habitat by reversing the hydrosere (Hawke and Jose, 1996). In general, this means that constructed reedbeds are younger than their natural equivalents, even if the latter have been planted and/or managed for conservation purposes.

The size of a constructed reedbed is dictated by the type of wastewater and the expected level of contamination. Constructed reedbeds are typically an order of magnitude smaller than natural reedbeds, with most being under 0.25 ha (Hawke and Jose, 1996). The average area of a constructed reedbed in the UK is 0.1 ha, whereas the average area of a natural reedbed is 5.5 ha (BRIG, 2008), although most sites are thought to be about 1 ha (Hawke and Jose, 1996).

In constructed reedbeds, habitat characteristics such as initial reed density and water depth are controlled by the engineers that design them. Constructed reedbeds are typically planted with a consistent water depth and reed density, so that the treatment process is even across the whole reedbed. Habitat characteristics across natural reedbeds vary depending on the stage of succession, with early succession reedbeds typically having standing water for much of the year and dense, monospecific stands of thick reed, whilst late succession reedbeds dry out, and have thinner, sparser reed, greater plant diversity, and areas of scrub (Hawke and Jose, 1996).

The design of a constructed reedbed is also dependent on the type of wastewater being treated. Typically, there are several compartments to a constructed site: they start with a settling pool (without any vegetation) and then have several treatment reedbeds or "beds". The treatment beds often contain different species: in the UK, a bed with a mix of *Typha latifola* and *Iris pseudacorus* is often used in conjunction with a bed containing a monoculture of *Phragmites australis* (see Chapter 1 for details). The species or combination of species used and the design (horizontal or vertical flow) will depend on the wastewater type. *Phragmites* constructed reedbeds which are planted as a monoculture are expected to have similar vegetation structure and composition to early succession (wet) natural reedbeds.

The typical attributes of constructed reedbeds suggest that they may have impoverished fauna compared with natural equivalents (see Chapter 1). The main objective of this thesis is to determine whether it is indeed the case that constructed reedbed communities are depauperate, or if in fact they support diverse communities of conservation value. In order to do this, my approach was to sample a range of constructed reedbeds which are typical of those found throughout the UK, and compare them to equivalent natural sites. I paired constructed sites with nearby natural sites with similar habitat characteristics (water level, reed density, vegetation composition) in order to remove as much as possible the effects of geographic location (latitude, longitude, altitude) and differences in successional stage. This chapter details how site selection was achieved, and tests for differences between paired natural and constructed in

habitat characteristics. The objective was to establish the factors which are most relevant in subsequent analyses of the animal communities in the two types of sites.

2.2 Methods

2.2.1 Site selection

A selection of 12 constructed reedbeds across the UK (Figure 2.1) was chosen after discussion with the Constructed Wetland Association (CWA) and several industry users. Sites were deemed suitable if they were: a) known to be dominated by *Phragmites australis*; b) site access could be arranged with the owner; c) sampling was possible without need of special health and safety measures. The 12 constructed sites selected treated three wastewater types: sewage (seven sites), mine-water (four sites) and run-off from a fire-training centre (one site). In all but one of these constructed sites there was more than one treatment bed. A treatment bed was chosen on the basis of: a) being dominated by *Phragmites australis,* b) having no known maintenance planned within survey timeframe, and c) ease of access. At five sites, only one bed was surveyed. In two cases, multiple treatment beds treating different types of effluent were surveyed in separate locations on site, and at the remaining five sites multiple treatment beds that were adjacent to each other and treating the same effluent were surveyed (indicated in Table 2.1).

Natural sites were chosen based on their proximity to the constructed sites but, as with constructed sites, access (at the right time of year; often restrictions applied due to breeding birds on natural sites) and health and safety considerations were taken into account. The nearest possible natural site to each constructed site was found through contact with local authorities and the Wildlife Trusts, and using MagicMap software (habitat mapping available online at http://www.magic.gov.uk/MagicMap.aspx). In only one case was access not granted to the nearest known natural site, and in this case the next nearest site was used. In all cases, matched pairs were within 16.2 miles of each other.

Surveys were undertaken at these 24 *Phragmites australis* reedbed sites in England and Wales (see Figure 2.1 and Table 2.1). For constructed sites, at least one treatment bed was surveyed to the fullest extent that access would allow. Access was often limited by health and safety, with some constructed reedbeds having areas of deep water or more often deep sludge. In these cases, surveys were conducted in transects at least 2 m into the reedbed, parallel to the perimeter. Effort was matched at natural sites. Within natural sites, which were typically larger, an area of equivalent size to the constructed site, matched as closely as possible in terms of water depth and habitat characteristics (see below), was surveyed. For those constructed sites where there were multiple small reedbeds or very large reedbeds, effort was maximised at 30 sampling points at 5 m intervals spanning one to four treatment beds (see Table 2.1). Where multiple small treatment beds were sampled, data from these were pooled for the analysis, unless it is indicated otherwise. Sites were sampled in either a 5 m grid or 5 m transect. A grid was used where possible, but for narrow or linear shaped reedbeds, transects were used. Wherever possible the same grid or transect was used throughout all surveys, but occasionally (nine out of 92 surveys) it was necessary to move from the original survey area, or reduce the area surveyed, owing to flooding or site maintenance. Any changes in sampling effort are reflected in the relevant analysis.

The total areas of reedbeds were estimated from the construction or management plans, measurements taken on the ground, or satellite images – whichever was deemed the most reliable given conditions at each site. The age of constructed sites was taken from the first installation of a reedbed at the site, not from any refurbishments. The age of natural sites was estimated from the first plantation of reed for the nature reserve, unless it was known that reed was present on site prior to this, in which case (once – Willow Tree Fen) it was estimated by previous land owners.

The percentage habitat cover of a strip of land 100 m wide surrounding the survey area was estimated by eye from satellite images and what was known from site visits (Table 2.2). Habitats were classified into seven categories: natural wetland, artificial wetland, open water, woodland, open habitat

(including grazed pasture, managed grassland), arable and hard standing. For those sites (two) where two reedbeds were surveyed in different locations on the same site, both locations were assessed separately for the habitat cover surrounding each reedbed. Although assessing habitat cover by eye has some limitations, time constraints restricted the use of more sophisticated methods to calculate the type of habitat surrounding the reedbed sites in this study. Further work in this area could utilise GIS software to calculate habitat maps in surrounding areas, in combination with a map of constructed reedbeds in the UK providing habitat connectivity information to inform analyses.

2.2.2 Survey methods

At each site, four types of survey (caterpillars, mammals, moths and stem-dwelling invertebrates) were conducted (for dates of each survey see Table 2.2). Detailed methods for each survey are described in Chapters 3-5. In addition, a survey of habitat characteristics and vegetation was conducted at each site. For vegetation surveys, each site was visited for one day in early summer, when most flowering plants were identifiable. For habitat characteristics (e.g. reed density, reed diameter, reed height and water depth), each site was visited for one day in early autumn. A 0.5 m by 0.5 m quadrat was placed at each point in the sampling grid or transect. In each quadrat, the total number of reed stems, and the height (measured from the base of the stem with a tape measure) and basal diameter (measured with callipers) of 20 randomly-selected reed stems, were recorded. Where the water level was above ground, the water depth in the centre of the quadrat was measured to the top sediment layer using a tape measure. The same quadrats were used to conduct plant surveys, where all flowering plant species were identified and their percentage cover estimated by eye.

Two sites were not accessible due to flooding for plant surveys. At both sites winter inspections of the vegetation suggested that *Phragmites australis* was the only flowering plant present, so in all analyses they are considered to have a plant species richness of one.

2.2.3 Comparison of habitats in constructed and natural sites

Habitat characteristics, many of which were likely to be inter-correlated, were subjected to Principal Component Analysis (PCA) in order to produce a small number of uncorrelated variables which describe the variation in habitat for use in subsequent analysis. This was done twice, once at the quadrat level and once at the site level, to allow for site level (moths and total diversity) and quadrat level (small mammals and stem-dwelling invertebrates) analyses. The following variables for each quadrat were included in both PCAs: the average reed stem height and diameter (within a quadrat or across a whole site), the density of reed stems per quadrat, the plant species richness, and the water depth.

With the exception of site age and area, every possible attempt was made to match habitat characteristics at the paired constructed and natural sites. This was done in order to allow a focus on site type as they key possible predictor of animal diversity (see later chapters). To determine whether or not this approach was justified, habitat data (including PC1 at the site level) from the two types of site were compared with Wilcoxon signed rank tests. All analyses were conducted in R version 3.3.2.

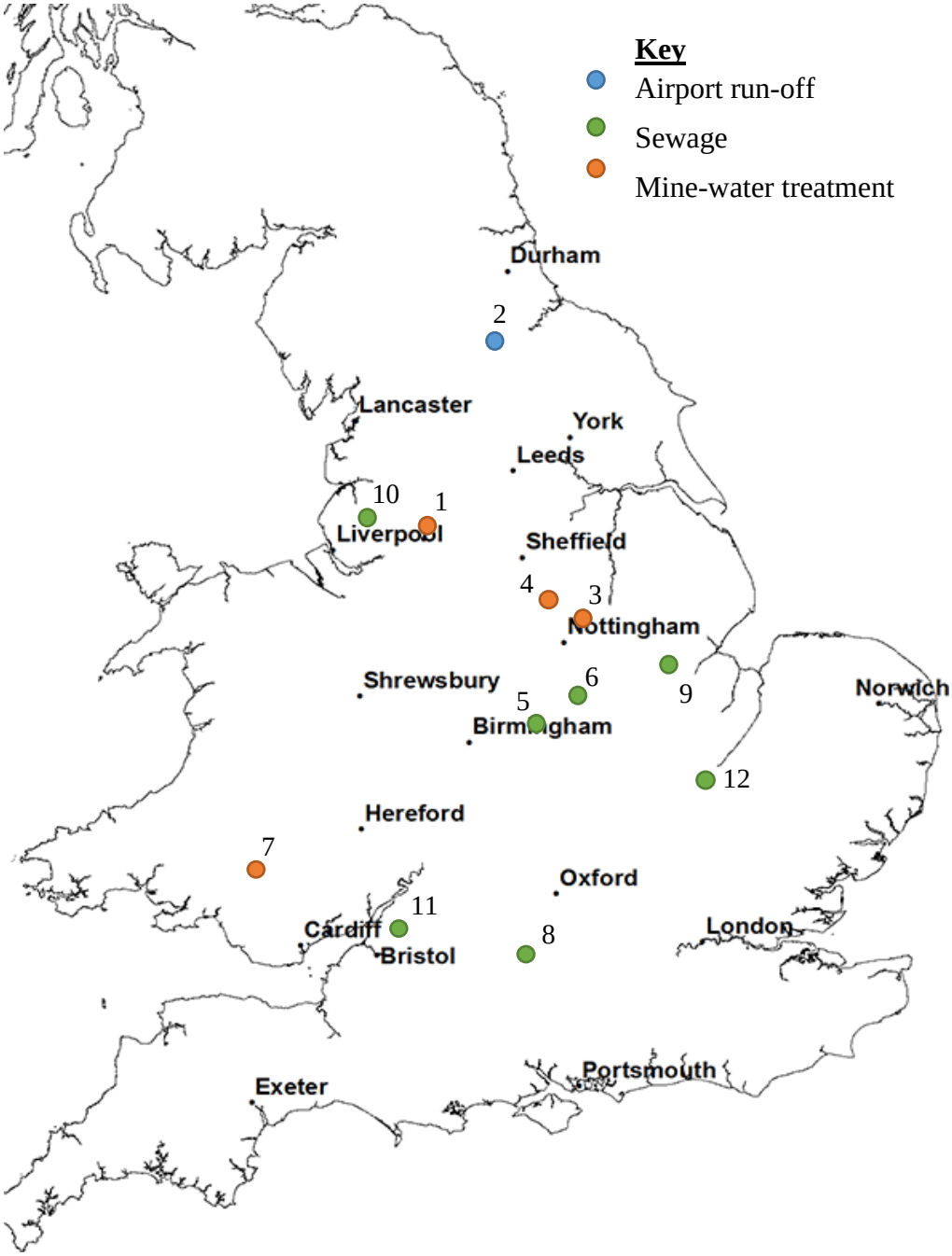


Figure 2.1: Map showing location of all paired sites and type of wastewater the constructed sites treated (see key). Table 2.1 contains corresponding site names and details.

Table 2.1: Site names and attributes. Matched pairs of constructed and natural sites are separated by a horizontal line. Each pair is identified in the text by the name of the constructed site. The location of the pair is indicated by a corresponding number in Figure 2.1.
*Two reedbeds in different locations on the same site

Pair (map number)	Name	Site type	Owner/manager	Number of reedbeds surveyed (no. of grid points)	Area (ha)	Age (years)	Distance between sites (miles)
Hockery Brook (1)	Amberswood	Natural	Wigan Council	1 (27)	0.214	25	0.9
	Hockery Brook	Mine Water treatment	The Coal Authority	1 (27)	0.44	12	
IFTC (2)	Bowesfield and Preston Farm (B+P Farm)	Natural	Teesside Wildlife Trust	1 (20)	3.2	11	4.2
	International Fire Training Centre (IFTC)	Run off from fire training centre	International Fire Training Centre	1(20)	0.1	13	
Shirebrook (3)	Carr Vale	Natural	Derbyshire Wildlife Trust	1 (28)	0.76	18	5.4
	Shirebrook	Mine Water Treatment	Nottinghamshire County Council and The Forestry Commission	3 (28)	0.36	17	
Pleasley (4)	Doe Lea NR	Natural	Derbyshire County Council	1 (26)	0.897	28	2.5
	Pleasley Pit	Mine Water Treatment		2* (26)	0.3	15	
Rugeley (5)	Drakelow	Natural	Derbyshire Wildlife Trust	1 (18)	2.0	40	10
	Rugeley	Small sewage treatment	ARM	3 (18)	0.038	31	

Table 2.1 continued:

Pair (map number)	Name	Site type	Company	Number of reedbeds surveyed (no. of grid points)	Area (ha)	Age (years)	Distance between sites (miles)
Benefield (6)	Titchmarsh	Natural	BCN Wildlife Trust	1 (12)	2.0	26	5.3
	Lower Benefield	Sewage Treatment	Anglian Water	1 (12)	0.028	15	
Taff Merthyr (7)	Parc Penallta	Natural	Caerphilly Council	1 (20)	0.6	16	2.6
	Taff Merthyr	Mine Water Treatment	The Coal Authority	1 (20)	2.215	15	
Sheepdrove (8)	Thatcham reedbeds	Natural	BBO Wildlife Trust	1 (24)	1.27	43	12.9
	Sheepdrove Organic Farm	Small sewage treatment	Sheepdrove Organic Farm	2 (24)	0.1	15	
Sleaford (9)	Willow Tree Fen	Natural	Lincolnshire Wildlife Trust	1 (30)	5.3	20	16.2
	Sleaford	Sewage Treatment	Anglian Water	1 (30)	0.73	13	
Martin Mere (10)	WWT Martin Mere	Natural	Wildfowl and Wetland Trust	1 (30)	20.0	13	0.5
	WWT Martin Mere	Nature Centre Sewage Treatment		3 (30)	1.35	19	

Table 2.1 continued:

Pair (map number)	Name	Site type	Company	Number of reedbeds surveyed (no. of grid points)	Area (ha)	Age (years)	Distance between sites (miles)
Slimbridge (11)	WWT Slimbridge	Natural	Wildfowl and Wetland Trust	1 (21)	7.8	27	1.0
	WWT Slimbridge	Nature Centre Sewage Treatment		2* (21)	1.5	21	
Welney (12)	WWT Welney	Natural	Wildfowl and Wetland Trust	1 (16)	6.7	37	0.2
	WWT Welney	Nature Centre Sewage Treatment		3 (16)	0.04	10	

Table 2.2: Percentage cover of habitat surrounding each reedbed
*Open habitat includes pasture and managed grassland

Site name	Habitat type						
	Natural wetland	**Artificial wetland**	**Open water**	**Wood land**	**Open habitat***	**Arable**	**Hard standing**
Amberswood	-	-	10%	80%	10%	-	-
Hockery Brook	-	15%	20%	10%	55%	-	-
Carr Vale	25%	-	25%	10%	10%	5%	-
Shirebrook	-	-	5%	40%	40%	5%	-
Doe Lea	5%	-	5%	65%	5%	15%	5%
Pleasley ROP	-	5%	5%	30%	40%	20%	-
Pleasley DP	-	-	10%	10%	80%	-	-
Drakelow	25%	-	25%	50%	-	-	-
Rugeley	-	-	-	5%	5%	40%	50%
Slimbridge N	40%	-	10%	10%	40%	-	-
Slimbridge T M	-	10%	30%	10%	10%	30%	10%
Slimbridge T SF	50%	30%	5%	5%	10%	-	-
Willow Tree Fen	30%	-	10%	10%	35%	10%	5%
Sleaford	-	20%	5%	10%	10%	5%	50%
Martin Mere N	80%	-	20%	-	-	-	-
Martin Mere T	30%	20%	50%	-	-	-	-
Welney N	95%	-	-	-	-	-	5%
Welney T	70%	-	10%	-	-	-	20%
Titchmarsh	20%	-	50%	10%	20%	-	-
Benefield	-	15%	-	10%	60%	10%	5%
Thatcham	80%	-	5%	15%	-	-	-
Sheepdrove	-	10%	20%	40%	25%	-	5%
Parc Penallta	10%	-	10%	50%	30%	-	-
Taff Merthyr	-	70%	10%	10%	10%	-	-
B&P Farm	25%	-	50%	-	20%	-	5%
IFTC	-	-	10%	-	-	60%	30%

Table 2.3: Dates of surveys conducted at reedbed sites. Sites are listed in their pairs, natural first then constructed. Surveys took one or two days as indicated, except for mammal surveys, which took five days starting on the date indicated. *Shirebrook was only accessible once for moth trapping due to anti-social behaviour

Site	Caterpillars & plants	Invertebrates & habitat characteristics	Mammals	Moths
Amberswood	26/06/15	23/10/14	09/02/15	10/07/15
		24/10/14		11/07/15
Hockery Brook	25/06/15	14/10/14	02/02/15	13/10/14
		15/10/14		14/10/14
Carr Vale	16/06/15	11/09/14	30/11/15	29/07/15
		12/09/14		30/07/15
Shirebrook	15/06/15	08/09/14	24/11/14	28/07/15*
		09/09/14		
Doe Lea	18/06/15	18/09/14	15/12/14	22/07/15
		19/09/14		23/07/15
Pleasley	17/06/15	15/09/14	08/12/14	17/07/15
		16/09/14		18/07/15
				20/07/15
				21/07/15
Drakelow	No access	24/09/14	17/11/14	23/08/14
				25/08/14
Rugeley	08/06/15	22/09/14	10/11/14	26/08/14
				03/09/14
Willow Tree Fen	10/06/15	05/10/15	12/01/15	04/08/15
				05/08/15
Sleaford	10/06/15	28/09/15	05/01/15	12/08/15
				13/08/15
Parc Penallta	04/07/16	29/10/15	09/11/15	05/07/16
				06/07/16
Taff Merthyr	02/07/16	31/10/15	02/11/15	03/07/16
				04/07/16
Bowesfield and Preston Farm	30/06/16	21/10/15	23/11/15	28/07/16
				29/07/16
IFTC	29/06/16	19/10/15	16/11/15	26/07/16
				27/07/16
Thatcham	No access	14/10/15	07/12/15	04/08/16
				05/08/16
Sheepdrove	23/06/16	13/10/15	30/11/15	02/08/16
				03/08/16

Table 2.3 continued:

Site	Caterpillars and plants	Invertebrates & habitat characteristics	Mammals	Moths
Titchmarsh	20/06/16	09/10/15	11/01/16	11/08/16 12/08/16
Benefield	07/07/16	30/09/15	04/01/16	09/08/16 10/08/16
WWT Slimbridge natural	07/07/15	09/10/14 10/10/14	27/10/14	06/07/15 07/07/15
WWT Slimbridge Treatment	30/06/15 02/07/15	29/09/14 30/09/14	01/11/14	29/06/15 30/06/15 01/07/15 02/07/15
WWT Martin Mere natural	28/06/16	22/09/15	21/09/15	20/07/16 21/07/16
WWT Martin Mere Treatment	27/06/16	15/09/15	14/09/15	19/07/16 22/07/16
WWT Welney natural	21/06/16	08/09/15	07/09/15	14/07/16 15/07/16
WWT Welney Treatment	21/06/16	01/09/15	31/08/15	12/07/16 13/07/16

2.3 Results

For the PCA at the quadrat scale, the first two components together explained over 58 % of the variation in the original five habitat variables (Table 2.4, Figure 2.2). PC1 is essentially a measure of reedbed "quality": quadrats that scored highly on this component were wet, had low plant species richness, and tall and thick reeds. PC2 captured variation in reed stem density, with high scores indicating low stem density. For the PCA analysis at the site scale, the first two components together explained over 61 % of the variation in the original five variables (Table 2.4, Figure 2.3). PC1 is again a measure of reedbed "quality": quadrats that scored low on this component were wet, had low plant species richness, and tall and thick reeds. PC2 again captured variation in reed stem density, with high scores indicating low stem density.

Constructed and natural sites did not differ in any of their habitat characteristics (Table 2.5, Figure 2.4, Figure 2.5), except for age and area. Natural sites were on average 10 years older (Figure 2.6) and an order of magnitude larger than constructed sites (Figure 2.7).

Table 2.4: Results of Principal Components Analysis (PCA) of five habitat variables at two scales, describing a total of 516 quadrats or 24 sites. Details are shown for the two most important components for both analyses which, together, explained over 58% (per quadrat) and 61% (per site) of the variance.

Per quadrat	**PC1**	**PC2**
Standard deviation	1.3589	1.0403
Proportion of variance	0.3693	0.2164
Loadings		
Diameter	0.594	-0.046
Height	0.427	-0.488
Density	-0.052	-0.813
Water depth	0.475	0.294
Plant species richness	-0.486	-0.110
Per site	**PC1**	**PC2**
Standard deviation	1.4125	1.0289
Proportion of variance	0.3991	0.2117
Loadings		
Diameter	-0.5663	-0.1490
Height	-0.3798	0.2865
Density	0.1219	-0.9035
Water depth	-0.4514	0.0474
Plant species richness	0.5624	0.2774

Table 2.5: Results of comparisons of habitat characteristics between paired natural and constructed reedbed sites using Wilcoxon signed rank test

Variable	**V**	**n**	**p**
Height	36	12	0.85
Water depth	28	12	> 0.99
Diameter	24	12	0.23
Density	26	12	0.34
Plant species richness	38.5	12	> 0.99
PC1 (Site level)	27	12	0.38
Area	67	12	**0.03**
Age	70.5	12	**0.02**

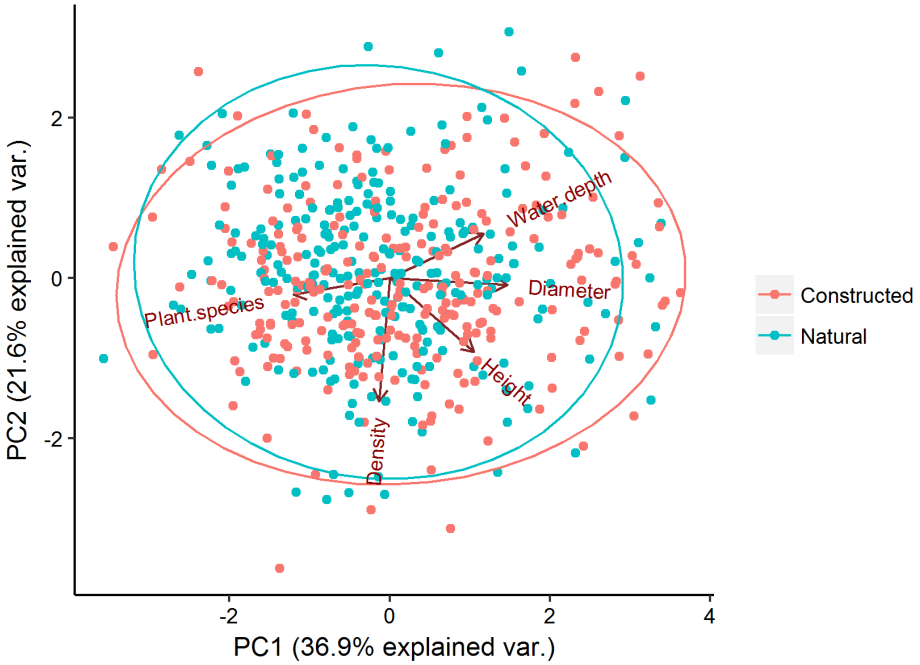


Figure 2.2: The first and second principle component scores for all 516 quadrats, separated by type (constructed or natural) of reedbed. See main text for details. Ellipse represent 95 % confidence intervals for the distributions of the two types of site.

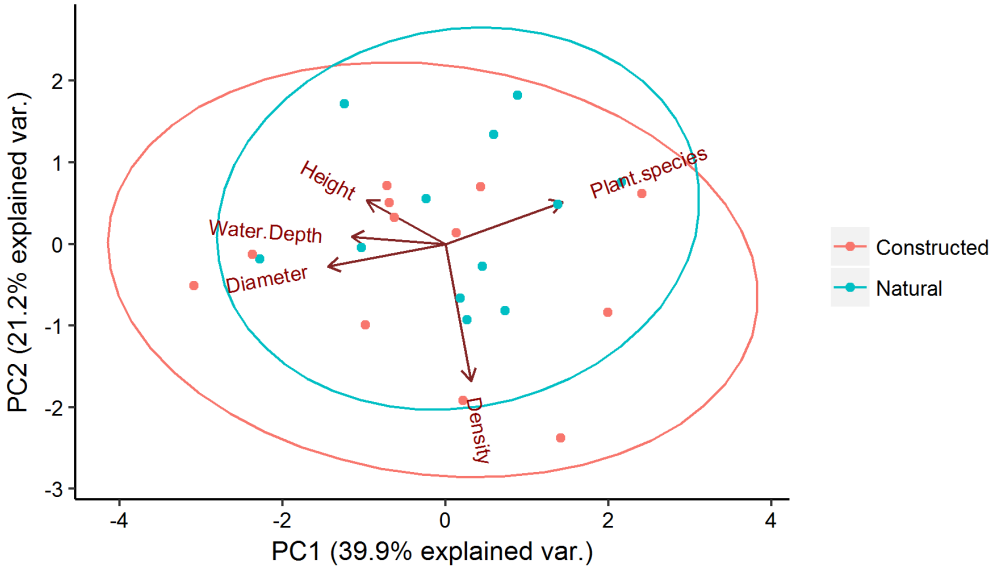


Figure 2.3: The first and second principle component scores for all 24 sites, separated by type (constructed or natural) of reedbed. See main text for details. Ellipse represent 95 % confidence intervals for the distributions of the two types of site.

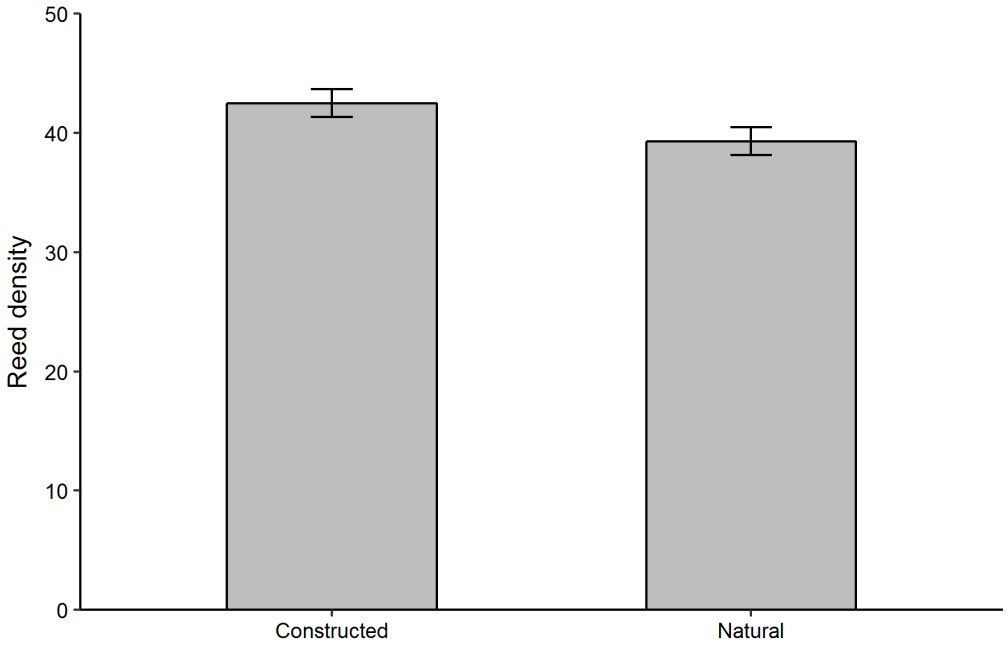


Figure 2.4: Mean density (+/- SEM) of stems in a 0.5 m by 0.5 m quadrat at constructed and natural sites.

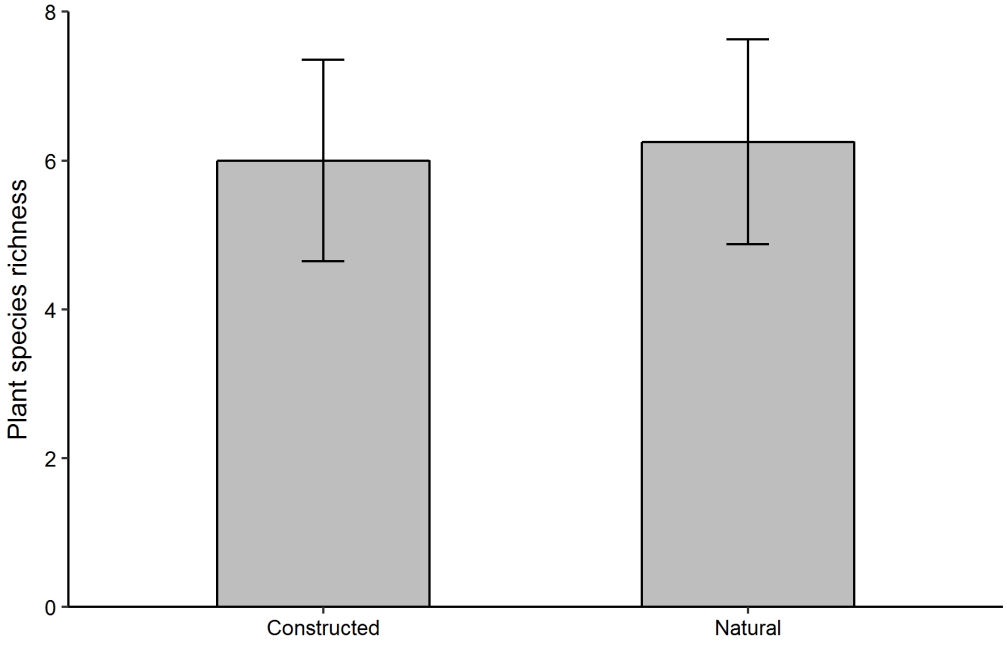


Figure 2.5: Mean plant species richness (+/- SEM) across each site in this study, shown for constructed and natural sites.

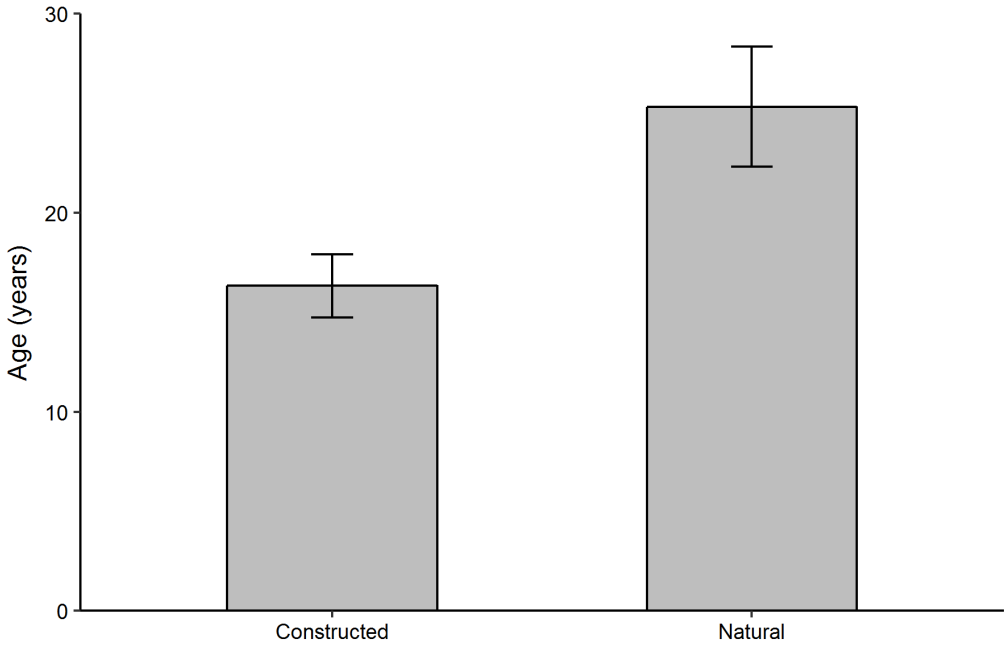


Figure 2.6: Mean age (+/- SEM) of 12 constructed and 12 natural reedbeds in this study

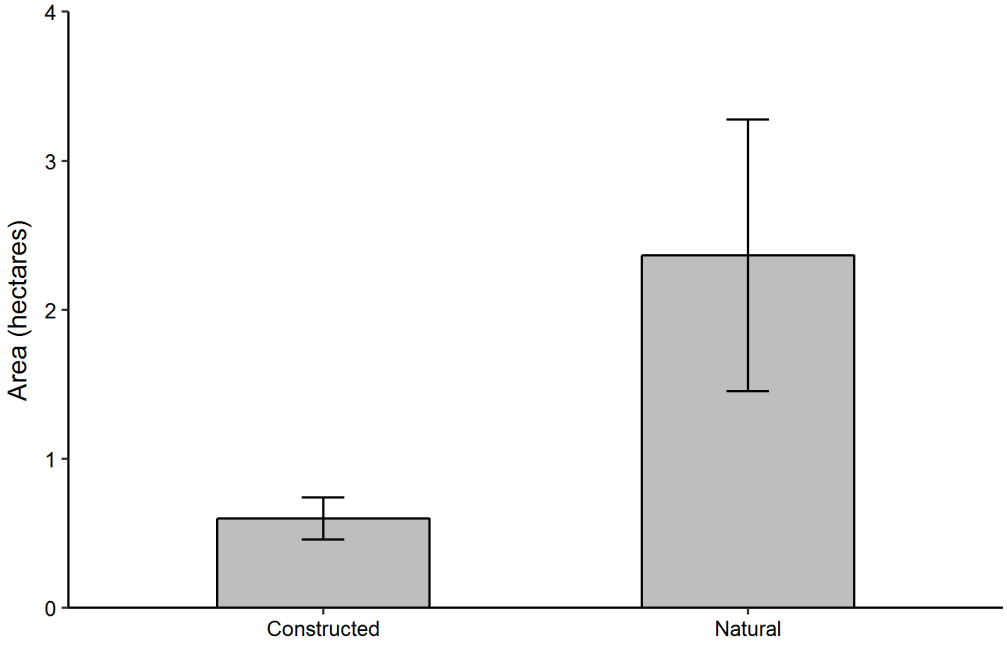


Figure 2.7: Mean area (+/- SEM) of 12 constructed and 12 natural sites used in this study.

2.4 Discussion

As expected constructed and natural sites differed in their age and area. The average area of a constructed site in this study (0.6 ha) was slightly bigger than that of the reported UK average (< 0.25 ha; (Hawke and Jose, 1996)), while the natural reedbeds surveyed were larger (3.3 ha) than the typical 1 ha for UK sites (Hawke and Jose, 1996).

The ages of constructed and natural sites in this study are in line with those reported in the literature (Hawke and Jose, 1996, Cooper and Green, 1995). Although natural sites were significantly older than constructed sites in this study, very few natural reedbeds in the UK are ancient. The only remaining old sites are in areas where reedbeds were once extensive, such as the low-lying areas of East Anglia and in some coastal regions (Hawke and Jose, 1996). Nevertheless, as discussed in Chapter 1, the observed differences in age and size, alongside other factors, are expected to limit the diversity and abundance of animals in constructed reedbeds.

The principal component analyses showed the major axis of variation among quadrats and sites surveyed was one which reflects variation along the hydrosere: wet sites are dominated by large, healthy reeds, whilst dry sites have more diverse vegetation and thinner, smaller reed. This represents the natural succession process; without management reedbeds gradually dry out and form more mixed stands with greater plant species richness and thinner, sparser reed (Hawke and Jose, 1996).

The results of the analysis presented here show that the attempts that were made to match pairs of constructed and natural sites for other habitat characteristics, such as reed stem density, height and diameter, were broadly successful: there were no differences in any of these variables between the two types of sites. This justifies the approach taken to the analysis of patterns in animal abundance and diversity in the following chapters, with the focus primarily being on the effects of site type, age and area.

Chapter 3: Small mammals in constructed and natural reedbeds in the UK

3.1 Introduction

Global habitat loss and fragmentation are key drivers in biodiversity loss, and with the additional and interacting threat of climate change, current conservation efforts may be inadequate to prevent a major extinction event (Mantyka-Pringle et al., 2012). The implications of this crisis for mammal conservation is evident, with 25 % of all mammal species that have been assessed being threatened with extinction (Visconti et al., 2011). Despite the fact that mammals are often charismatic and used as flagship species for conservation initiatives, their overall conservation status is precarious, with declines even in protected areas (Hoffmann et al., 2011).

Reedbeds are areas of wetland dominated by common reed (*Phragmites australis*) (BRIG, 2008). Reedbeds are threatened throughout the UK and Europe because of agricultural drainage, pollution, succession and predicted sea level rises (Paul et al., 2011). The remaining habitat in Europe is intensively managed, and that in England is highly fragmented (BRIG, 2008).

The loss and fragmentation of reedbed habitat in the UK has had a profound impact on the species that are associated with it. Reedbeds are especially important for small mammal communities; they are the most diverse wetland habitat for small mammals (Scott et al., 2008, Slabova et al., 2008, Marques et al., 2015), and are amongst the most diverse habitats of any kind for small mammals in the UK. Not only are reedbed small mammals of conservation significance in their own right, but they also support a large number of birds of prey and other predators (Buesching et al., 2008), which are often of threatened status and vulnerable to declines in small mammal abundance or diversity.

For conservation and management strategies to be effective, adequate monitoring of species abundance and distribution, as well as an understanding of habitat use and preference, are vital. Many mammals in the UK have suffered noticeable and serious declines (Battersby, 2005). However, despite being ecologically significant and important indicators of the quality of the

environment, there is a lack of robust information on UK mammal population trends and their drivers (Battersby, 2005).

Reedbeds are especially important for harvest mice (*Micromus minutus*), which is a UK Biodiversity Action Plan (BAP) priority species. In one study in Austria looking at harvest mouse distribution across a habitat mosaic, nearly 90% of individuals were caught in tall reed stands (Haberl and Krystufek, 2003). They are mainly associated with wet, early succession reedbeds in the UK (Kettel et al., 2016). Although the harvest mouse is in the "Least Concern" category on the IUCN red list (Aplin et al., 2016), it is unclear whether populations in the UK are healthy, and they were thought to have been in decline up until the late 1990s (Flowerdew, 2004). A national harvest mouse survey carried out in 1999 only recorded harvest mice nests from 29% of those sites in which the species was found in the 1970s. Twenty-four percent of this loss was due to lack of suitable habitat (Flowerdew, 2004). More research is required to understand fully their distribution and conservation status.

The recent revitalisation of natural reedbeds in the UK has been stimulated by attempts to conserve the flagship species of reedbeds, the bittern (*Botaurus stellaris*). Nearly 1,000 ha of reedbed have been created or restored in the last two decades (Hardman et al., 2012). The current expansion of reedbeds for bittern conservation has led to an increased understanding of factors affecting the bird's presence and population size (Wotton et al., 2009). However, our understanding of the impact of habitat creation and management on other reedbed species, including small mammals, remains limited. This is particularly true in the case of small and marginal habitats, and reedbeds which are not managed for conservation purposes, such as those used to treat polluted wastewater.

Reedbeds have the ability to improve the quality of surface and ground water (Wang et al., 2010). As a result reedbeds are increasingly used in water treatment (Kobbing et al., 2013). Reedbeds that have been planted with specialist drainage networks and with water treatment in mind are often referred to as "constructed"

reedbeds. They are designed to treat sewage wastewater, mine-water, sludge, industrial effluents and surface and road runoff (Cooper, 2007).

As with any artificial habitat, constructed reedbeds have the potential to host similar biodiversity to their natural equivalents, but this potential has seldom been explored in the literature. One study in the UK demonstrated the ability of constructed reedbeds to support a diverse aquatic macroinvertebrate community, which in turn may support other organisms, including small mammals (Batty et al., 2005). This study suggested that communities in constructed reedbeds were impoverish compared to natural systems, and typically comprised of pollutant-tolerant species. However, a study in Denmark found that toxicity and bioaccumulation had no effect on biodiversity and community composition in reedbed macroinvertebrates (Stephansen et al., 2016).

Very few constructed wetlands have been surveyed for small mammals. Studies from naturally heavy metal contaminated wetlands illustrate the effects that bioaccumulation can have on small mammal populations. Bioaccumulation levels are dependent on species, their feeding habits and dispersal ability, but the build-up of toxins in small mammals has the potential to lead to further bioaccumulation in predators such as birds of prey (Wijnhoven et al., 2007). However those species that have the highest bioaccumulation rates are carnivorous species (e.g. shrews), and these are generally less numerous and not as important prey items for typical reedbed predators (Wijnhoven et al., 2007). There is evidence from one study in Illinois to suggests that despite contamination, the abundance, demographics and reproductive activity of small mammals in constructed wetlands do not differ significantly from those seen in less contaminated habitats (Levengood and Heske, 2008).

Regardless of any difference in species composition between constructed and natural reedbeds, with the reduction in wetlands globally, constructed reedbeds could provide valuable additional areas of habitat, with significant impacts on the total area of habitat available to species of interest. Constructed wetlands may not only contribute in terms of area, but there is also evidence to suggest that small wetlands may provide the connectivity to maintain metapopulations

(Gibbs, 1993). This connectivity could reduce the risk of local extinctions, especially of small mammals (Gibbs, 1993).

This chapter assesses the conservation value of small mammal communities in constructed reedbeds. Mammals were live-trapped in 12 matched pairs of natural and constructed *Phragmites australis* reedbeds across England and Wales. The general objective was to discover whether the species which are typically found in large natural reedbeds are also present in these small and often polluted artificial habitats. The literature reviewed above indicates that there could be an expectation that generalists may be present in the smaller, polluted constructed reedbeds but that reedbed specialists like the harvest mouse may be absent from those smaller, polluted constructed reedbeds. Specifically, I tested for differences in small mammal abundance and diversity between natural and constructed reedbeds, and examined the effect of a range of other environmental variables on mammal abundance. I consider the implications of the findings in the context of small mammal conservation and constructed reedbed management.

3.2 Materials and methods

3.2.1 Field surveys

Twelve matched pairs of study sites were selected, as described in Chapter 2. Sampling points at 5 m intervals on a grid encompassing the survey area were used repeatedly for each survey method. The grid size varied depending on the area of habitat accessible at each artificial site (see Chapter 2, Table 2.1). At each point reedbed habitat characteristics were measured; details are given in Chapter 2.

Mammals were sampled by placing Longworth traps at each point in the 5 m grid. Traps were placed 1 m off the ground, attached to a bamboo cane with electrical tape. Longworth trapping in the stalk zone is the most efficient way to survey harvest mice in reedbeds, and it also captures a number of other reedbed species (Kettel et al., 2016). Some species, however, are much more likely to be caught on the ground. The nature of reedbeds is that they have fluctuating water

levels, meaning that ground trapping is sometimes impossible or unsafe (with a risk of drowning captured animals). Therefore, Longworth traps were also placed on the ground at each grid point, but only when this was judged to be safe. The traps were baited with bird seed (with a high concentration of sunflower seeds) and blowfly pupae and cotton wool for bedding. All the chambers of the traps were wrapped in bubble wrap for additional insulation.

Sites were surveyed between November 2014 and February 2015, during September 2015 and between November 2015 and January 2016 (see Chapter 2 Table 2.3 for details). At each site, traps were set for pre-bait on the first day and night. For the following three evenings traps were set for capture at 2000 h and checked at 0600 h. Traps were kept closed during the day. Individuals caught were identified to species, sexed and weighed using a spring balance. If an animal had not been captured before, its fur was clipped with a sharp pair of scissors to allow recaptures to be distinguished. Animals were then released at the site of capture.

3.2.2. Statistical Analyses

All analyses were conducted in R version 3.3.2. A series of three Hill numbers (N_0 – species richness, N_1 – the exponential of the Shannon index, and N_2 – the inverse of Simpson's index) were calculated for all sites, and paired natural and constructed site were compared using Wilcoxon signed rank tests. The catch per unit effort was calculated as the number of captures divided by the number of traps set for the three most common species: wood mouse (*Apodemus sylvaticus*), bank vole (*Myodes glareolus*) and harvest mouse. A correction was made for the fact that a trap could only capture one individual per night: traps that captured an individual of one species were assumed to be unavailable to the other species for half the night (Nelson and Clark, 1973).

The effects of various explanatory variables on the catch per unit effort at the scale of the site (for the three most common species), and the probability of capture at the scale of the trap location within the sampling grid (for each species separately, excluding sites at which the species was not found), were tested for the three commonest mammal species with generalised linear mixed models. For

catch per unit effort, each species at each site provided a single data point, and errors were assumed to be Gaussian. The fixed effects tested were species, total size of the reedbed treatment system or area of reedbed on the nature reserve, site type (natural or constructed), the age of the reedbed and the habitat surrounding the reedbed. For this last effect, percentage cover for three different habitat types were considered (calculated by eye - see Chapter 2): wetland, open habitat (pasture and managed grassland) and woodland. Random effects were site pair (because pairs of sites were deliberately matched – see Chapter 2) and site (because all three mammal species were analysed simultaneously). Preliminary analysis showed that other possible explanatory variables (habitat characteristics such as reed density and water level) did not vary significantly between constructed and natural reedbeds (see Chapter 2), and these variables were excluded from the analysis of catch per unit effort at a site scale to maximise our ability to test other effects of interest.

For analysis of the probability of capture at the scale of the trap location, only sites at which the species was found were used. Each trap location in the sampling grid on each night provided one data point, and errors were assumed to be binomial. Fixed effects were season (autumn (September to November) or winter (December to February), elevation (traps placed on the ground or elevated), site type (natural or constructed), water depth (wet, where the water level was above ground, or dry), reed density and habitat surrounding the reedbed. As before, for this last effect, percentage cover for three different habitat types were considered (calculated by eye - see Chapter 2): wetland, open habitat (pasture and managed grassland) and woodland. The only random effect was site; pair was not included as a random effect because in many cases individuals were only captured from one of the two sites in a particular matched pair.

In each model, all two-way interactions were considered, but high-order interactions were ignored. The most parsimonious models were selected using a partial forward step-wise consideration of terms; limited sample sizes restricted the fitting of a saturated model, and hence backwards step-wise model selection was not possible. First, a model with all main effects was fitted, and then each

two-way interaction was added separately and the effect of the addition on the deviance explained by the model was tested using an F test (Guassian model) or Chi-squared test (binomial models). Significant interactions were retained, and non-significant interactions were removed. Main effects were then tested by backwards deletion. It was not possible to fit all three surrounding habitat variables (percentage wetland, open habitat and woodland) in the same model, so three models, each with one of the three habitat variables as a main effect, were compared in the initial step of model selection. The best of these models (according to AIC) was used for the remainder of the model selection procedure. To find the optimal structure of the random component, the AIC scores of a model with both random effects was compared to models with one or no random effects. In both cases, the AIC was only slightly improved by removing the random effects, so to be conservative the random effects were retained.

Data on the body mass of the three most common mammal species were also analysed to test the effect of sex and date caught using a Gaussian linear mixed model, with site as a random effect. To find the optimal structure of the random component, the AIC scores of a model with the random component site was compared to one without. For bank vole the AIC was improved with the random component. For harvest mice and wood mice the AIC was slightly improved by removing the random effect, but to be conservative the random effect was retained.

3.3 Results

In total traps were set for capture on 2,523 occasions, with 559 individuals caught (Table 3.1). This is a capture rate of 22 % across all sites. The highest capture rate at any site was 49 % at Drakelow, a natural site. The lowest capture rate was 3 % at Hockery Brook, a constructed site. The average capture rate for ground traps was 45 %, with the highest of 83 % at Benefield, a constructed site and the lowest of 8 % at Amberswood, a natural site. The average capture rate for elevated traps was 10 %, with the highest at 50 % in one section of the Slimbridge constructed reedbed. However, as a single site, Sleaford constructed reedbed had the highest elevated trap capture rate at 25 %. Three sites had no

elevated traps used by small mammals: Amberswood, a natural site, and two constructed sites, Taff Merthyr and IFTC.

Seven species of small mammal were recorded: wood mouse, bank vole, harvest mouse, common shrew (*Sorex araneus*) and water shrew (*Neomys fodiens*) (Table 3.1), plus single individuals of pygmy shrew (*Sorex minutus*) and house mouse (*Mus domesticus*). Wood mice were caught at all bar two sites (Sheepdrove Organic Farm, a constructed site and Amberswood, a natural site) with 293 individuals being caught in total. Bank voles were caught at the majority of sites, with 154 individuals caught in total. Harvest mice were caught at six constructed and six natural sites with a total of 79 individuals. Common shrews were caught in low numbers at eight sites, totalling 21 individuals. A total of ten water shrews were caught at one natural site, Martin Mere. All shrew species; common, water and pygmy were caught only on the ground.

There was no difference in species richness of small mammals, or other measures of diversity, between constructed and natural sites (Wilcoxon paired tests for each of three Hill's numbers: N_0: V = 31.5, n = 12, p = 0.30, N_1: V = 31.5, n = 12, p = 0.30, N_2: V = 31.5, n = 12, p = 0.30). Seven species were observed across all sites, and the maximum species richness per site was four, observed at five natural sites and one constructed site (Table 3.2, Figure 3.1). Diversity ranged from 1.0 at several natural and constructed sites to 3.66 (N_1) and 3.34 (N_2) at one natural site (Table 3.2). The species composition at constructed and natural sites did not differ substantially. The most common species - wood mice, bank vole and harvest mice - were recorded at both constructed and natural sites. Common shrews were recorded at eight sites, five natural and three constructed. Three of the seven species recorded were only recorded at one site, all of which were natural.

Table 3.1: The number of individuals of each species caught at each site and the catch per unit effort (the corrected percentage capture – see text). Site type: N = natural, C = constructed. [1]One trap opened over one night equals one trap night.

		Number of individuals									
Site	**Site type**	**Wood mouse**	**Bank vole**	**Harvest mouse**	**Common shrew**	**Water Shrew**	**Pygmy Shrew**	**House Mouse**	**Total**	**Number of trap nights[1]**	**Catch per unit effort**
Amberswood	N	0	6	0	0	0	0	0	6	150	0.04
Hockery Brook	C	2	0	0	0	0	0	0	2	75	0.03
Carr Vale	N	15	6	11	2	0	0	0	34	93	0.45
Shirebrook	C	23	5	12	1	0	0	0	41	93	0.57
Doe Lea	N	13	25	1	0	0	0	0	39	141	0.32
Pleasley	C	11	0	0	0	0	0	0	11	84	0.14
Drakelow	N	7	28	15	3	0	0	0	53	108	0.65
Rugeley	C	11	0	3	2	0	0	0	16	96	0.18
Willow Tree Fen	N	4	0	0	0	0	0	0	4	90	0.05
Sleaford	C	13	2	8	0	0	0	0	23	90	0.29
Slimbridge Natural	N	8	8	1	0	0	0	0	17	126	0.14
Slimbridge Treatment	C	13	2	5	0	0	0	0	20	102	0.22
Parc Penallta	N	4	0	0	0	0	0	0	4	60	0.07
Taff Merthyr	C	21	2	0	0	0	0	0	23	120	0.21
B & P Farm	N	7	6	9	0	0	0	0	22	78	0.33
IFTC	C	5	2	0	0	0	0	0	7	75	0.10
Thatcham reedbeds	N	38	7	0	1	0	1	0	47	144	0.39
Sheepdrove Organic Farm	C	0	9	11	0	0	0	0	20	144	0.15
Titchmarsh	N	1	0	2	0	0	0	0	3	36	0.09
Benefield	C	26	1	1	0	0	0	0	28	66	0.54
Martin Mere Natural	N	22	10	0	8	10	0	0	50	180	0.32
Martin Mere Treatment	C	10	27	0	0	0	0	0	37	180	0.23
Welney Natural	N	23	3	0	2	0	0	1	29	96	0.36
Welney Treatment	C	16	5	0	2	0	0	0	23	96	0.27
Grand total		**293**	**154**	**79**	**21**	**10**	**1**	**1**	**559**	**2523**	**0.25**

Table 3.2: Species richness (N_0) and diversity (N_1 and N_2) for all sites, sites are arranged in their pairs. Site type: N = natural and C = constructed.

Site	Site type	Species richness (N_0)	Diversity	
			N_1	N_2
Amberswood	N	1	1.00	1.00
Hockery Brook	C	1	1.00	1.00
Carr Vale	N	4	3.31	2.99
Shirebrook	C	4	2.80	2.40
Doe Lea	N	3	2.10	1.91
Pleasley	C	1	1.00	1.00
Drakelow	N	4	3.07	2.63
Rugeley	C	3	2.29	1.91
Willow Tree Fen	N	1	1.00	1.00
Sleaford	C	3	2.46	2.23
Slimbridge Natural	N	3	2.40	2.24
Slimbridge Treatment	C	3	2.35	2.02
Parc Penallta	N	1	1.00	1.00
Taff Merthyr	C	2	1.34	1.18
B & P Farm	N	3	2.95	2.91
IFTC	C	2	1.81	1.68
Thatcham reedbeds	N	4	1.85	1.47
Sheepdrove Organic Farm	C	2	1.99	1.98
Titchmarsh	N	2	1.88	1.80
Benefield	C	3	1.35	1.15
Martin Mere Natural	N	4	3.66	3.34
Martin Mere Treatment	C	2	1.79	1.65
Welney Natural	N	4	2.05	1.54
Welney Treatment	C	3	2.21	1.85

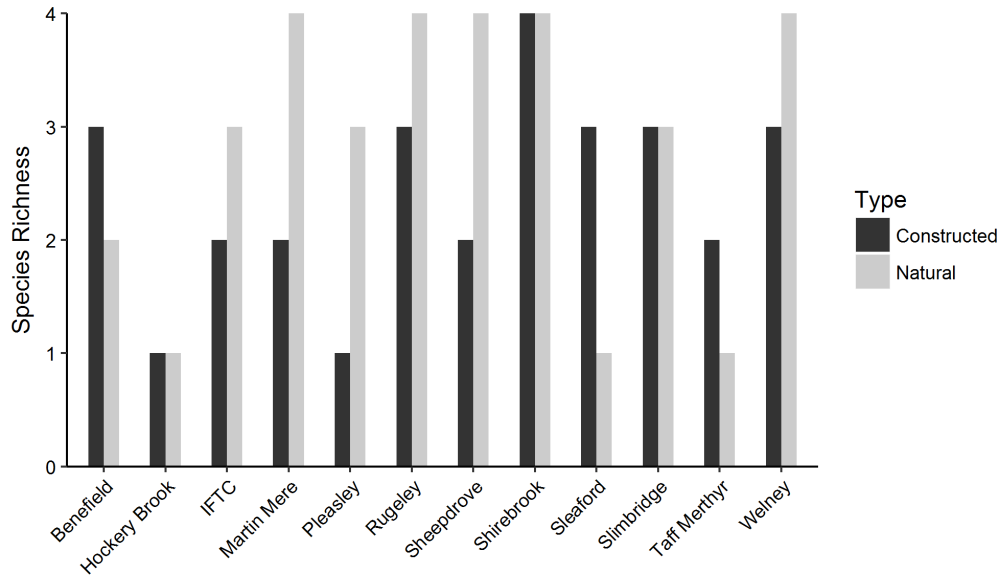


Figure 3.1: Small mammal species richness of paired constructed and natural sites. Name given is of constructed site in the pair (see Chapter 2 Table 2.1 for site information).

There were significant differences in the catch per unit effort among species; wood mice were the most abundant species, followed by bank voles and harvest mice (Table 3.3, Figure 3.2). Reedbed size and age had no significant effects on catch per unit effort (Table 3.3). Importantly, there was no difference in mean catch per unit effort between constructed and natural sites (Table 3.3, Figure 3.3), although there was an interaction of site type with the percentage of open habitat surrounding the reedbed. Catch per unit effort decreased at natural sites as the percentage of open habitat surrounding the reedbed increased whereas at constructed sites there was no effect (Figure 3.4).

Table 3.3: Results of generalized linear mixed model testing effects of various variables describing site and habitat characteristics on catch per unit effort and the probability of capture of the three most common small mammal species. All two-way interactions were considered; those not shown were not significant. Of three habitat variables (percentage wetland, open habitat and woodland surrounding the reedbed), results are reported for models which included only one (the one which explained the most variance when initially fitted as a main effect).

Analysis	**Random effect**	**Dependent variable**	**Independent variable**	χ^2	**df**	**P value**
Gaussian GLMM	Site Pair	Catch per unit effort	Species	17.023	2	**0.0002**
			Size	0.0402	1	0.8411
			Site type	0.0048	1	0.9447
			Age	1.7011	1	0.1921
			Open habitat	0.3374	1	0.5613
			Open habitat : site type	6.4663	1	**0.0109**
Binomial GLMM	Site	Harvest mouse capture	Season	0.0936	1	0.7596
			Site type	3.9157	1	**0.0478**
			Water	3.3935	1	0.0654
			Density	0.1753	1	0.6772
			Elevation	0.4808	1	0.4881
			Wetland habitat	0.5849	1	0.4444
			Density : Season	10.391	1	**0.0012**
			Season : wetland habitat	19.622	1	**< 0.0001**
Binomial GLMM	Site	Wood mouse capture	Season	1.2630	1	0.2611
			Site type	0.9586	1	0.3275
			Water	0.9435	1	0.3314
			Density	0.9948	1	0.3186
			Elevation	136.20	1	**< 0.0001**
			Wetland habitat	0.0614	1	0.8042
			Elevation : wetland habitat	6.7413	1	**0.0094**
Binomial GLMM	Site	Bank vole capture	Season	0.0800	1	0.7773
			Site type	0.6810	1	0.4093
			Water	3.2781	1	0.0702
			Density	0.0807	1	0.9938
			Elevation	184.31	1	**< 0.0001**
			Wetland habitat	2.4113	1	0.1205

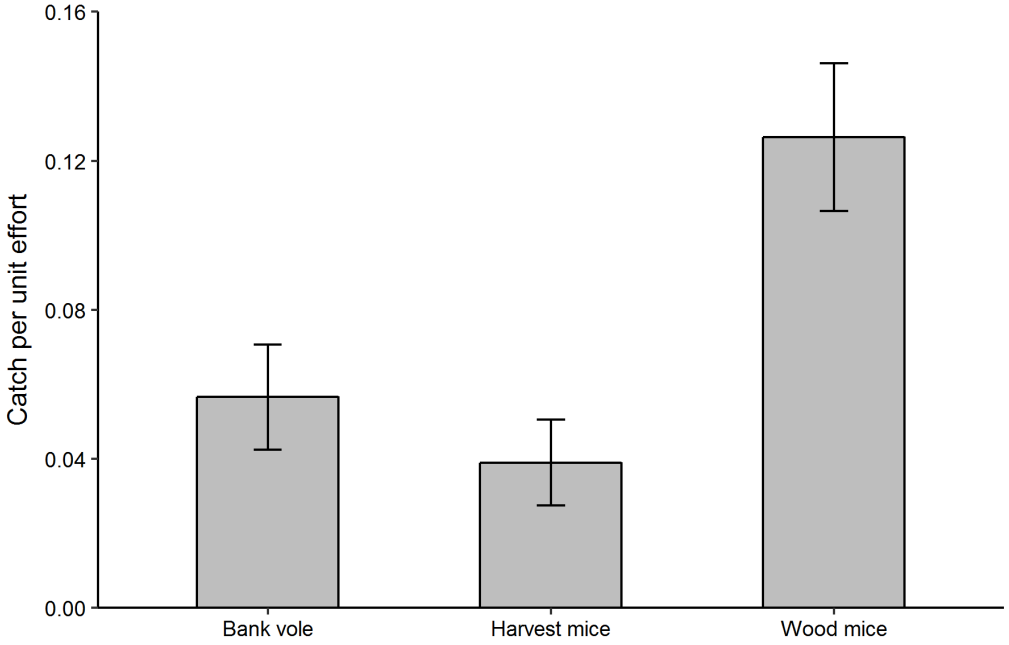


Figure 3.2: Mean catch per unit effort (+/- SEM) of the three most common species: bank vole, harvest mice and wood mice. GLMM (Table 3.3) indicates significant difference between catch rate of species (χ^2 = 17.023, df = 2, p = 0.0002).

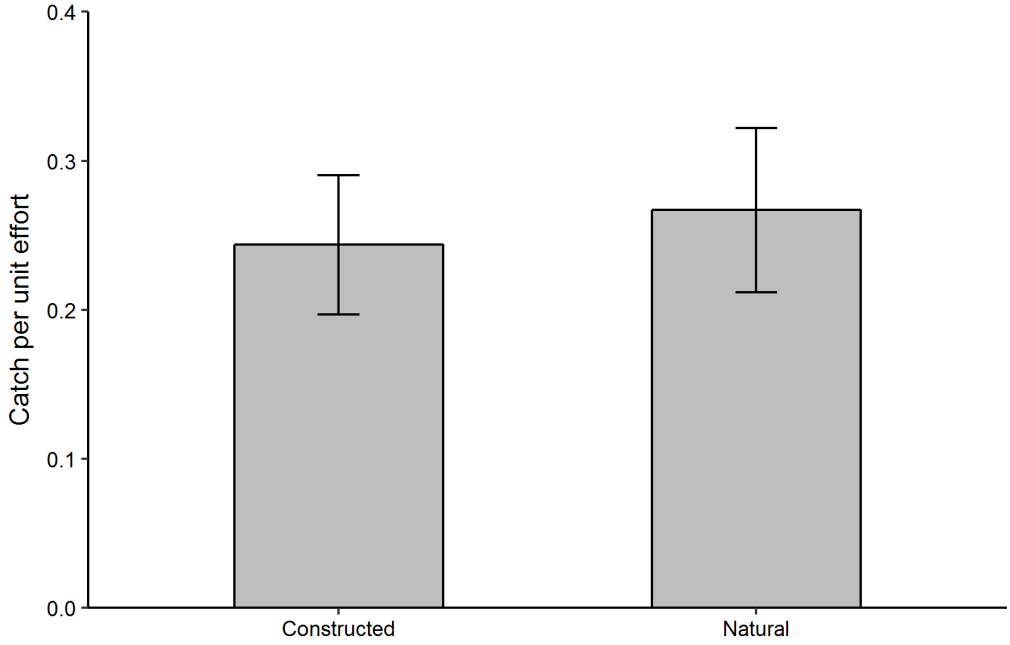


Figure 3.3: Mean catch per unit effort (+/- SEM) of all small mammal species combined in 12 constructed and 12 natural reedbeds. GLMM (Table 3.3) indicates no significant difference between site type (χ^2 = 0.0048, df = 1, p = 0.9447).

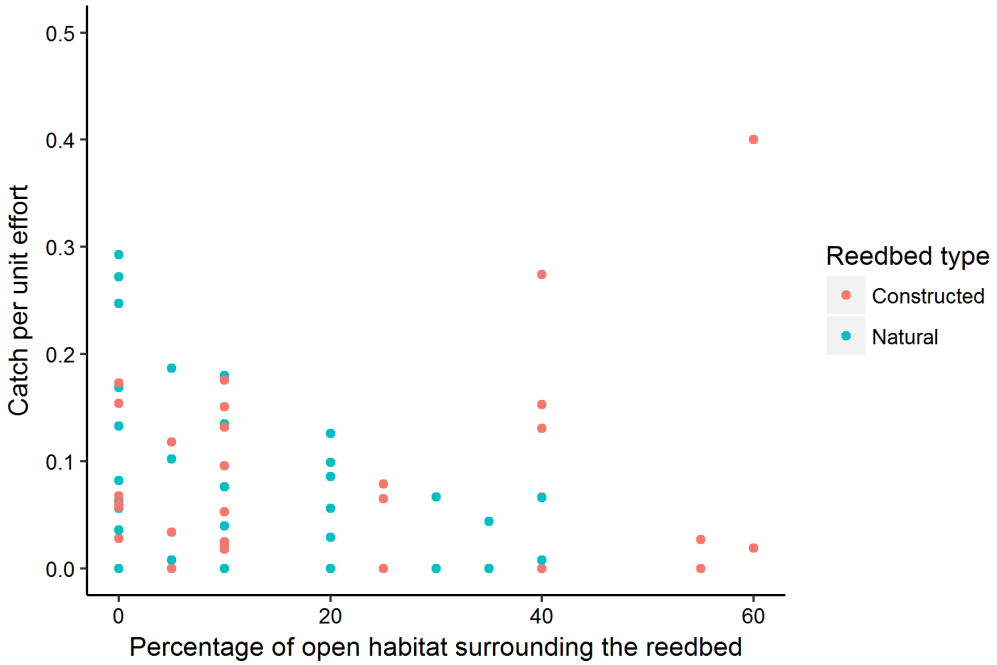


Figure 3.4: The effect of the percentage of open habitat (pasture and managed grassland) surrounding the reedbed on catch per unit effort of each species (woodmouse, bank vole and harvest mouse) for constructed and natural reedbeds. GLMM (Table 3.3) indicates the percentage open habitat surrounding the site affected the catch per unit effort differently at constructed and natural sites (χ^2 = 6.4663, df = 1, p = 0.0109).

There were two significant interactions on the probability of capturing a harvest mouse at a particular trap site: that between season and percentage wetland habitat surrounding the reedbed, and that between season and density (Table 3.3). In the autumn the probability of catching a harvest mouse was significantly greater in denser reed, whereas in winter the probability of catching a harvest mouse was the same regardless of the density of reed (Figure 3.5), although it should be noted that in autumn there were only four trap sites that captured of harvest mice compared to 51 in winter. In winter there was a greater chance of catching harvest mice at sites that had more wetland surrounding them, whereas in autumn there was only a minimal negative effect of the percentage of surrounding wetland (Figure 3.6). Although reedbed type as a main effect was also marginally significant, with harvest mice slightly more likely to be caught in constructed reedbeds, this result was only significant in the presence of the interactions of season and density, and season and percentage wetland habitat. There were 30 out of 197 trap sites that captured harvest mice at constructed

sites compared to 25 out of 194 at natural sites. Additionally, harvest mice were more likely to be caught in elevated traps than on the ground (Figure 3.7), but this effect was not significant.

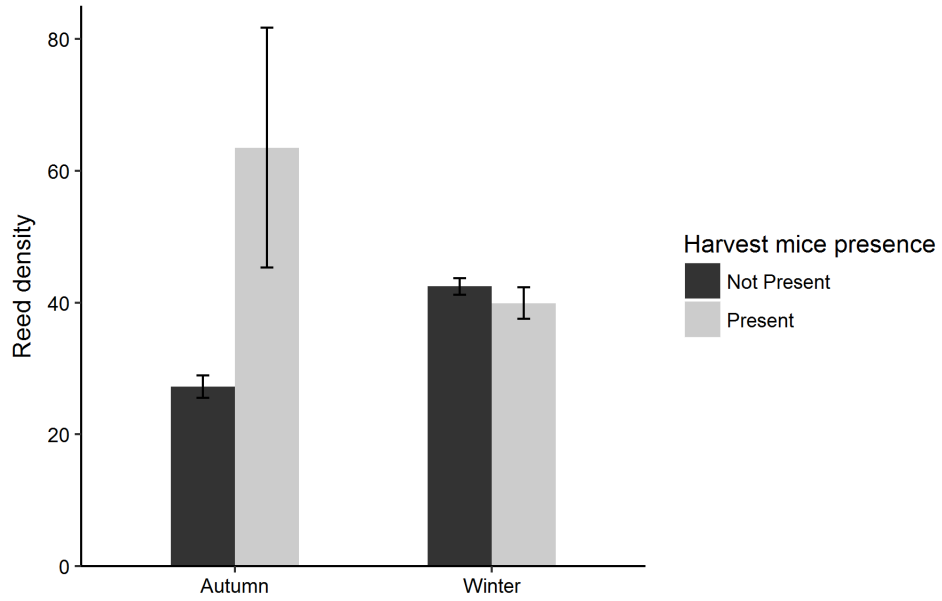


Figure 3.5: Mean reed density (+/- SEM) at trapping sites when harvest mice were present and absent during autumn and winter. GLMM (Table 3.3) indicates the reed density affected harvest mouse presence differently in autumn and winter (χ^2 = 10.391, df = 1, p = 0.0012).

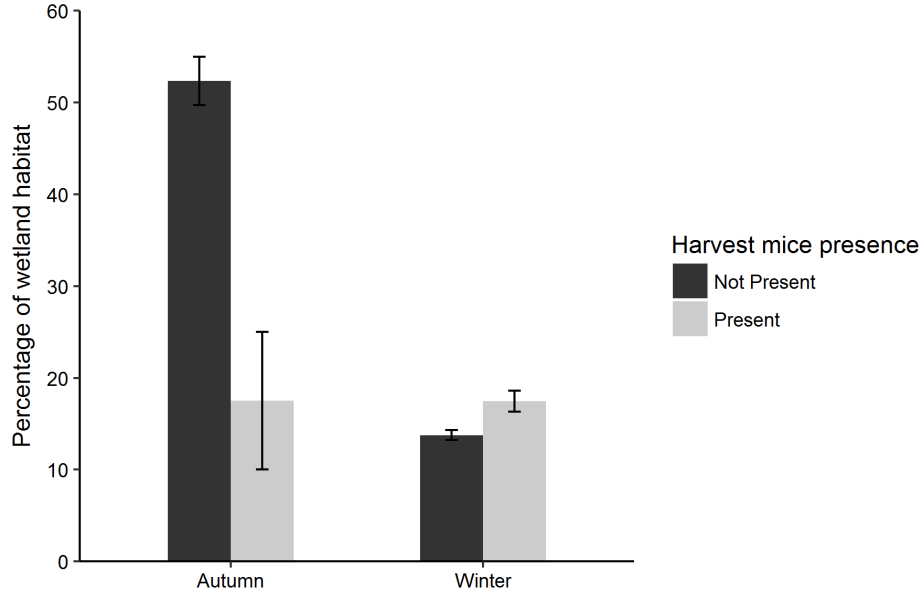


Figure 3.6: Mean percentage wetland habitat surrounding the reedbed (+/- SEM) at trapping sites when harvest mice were present and absent during autumn and winter. GLMM (Table 3.3) indicates the percentage of reedbed habitat affected harvest mouse presence differently in autumn and winter (χ^2 = 19.622, df = 1, p < 0.0001).

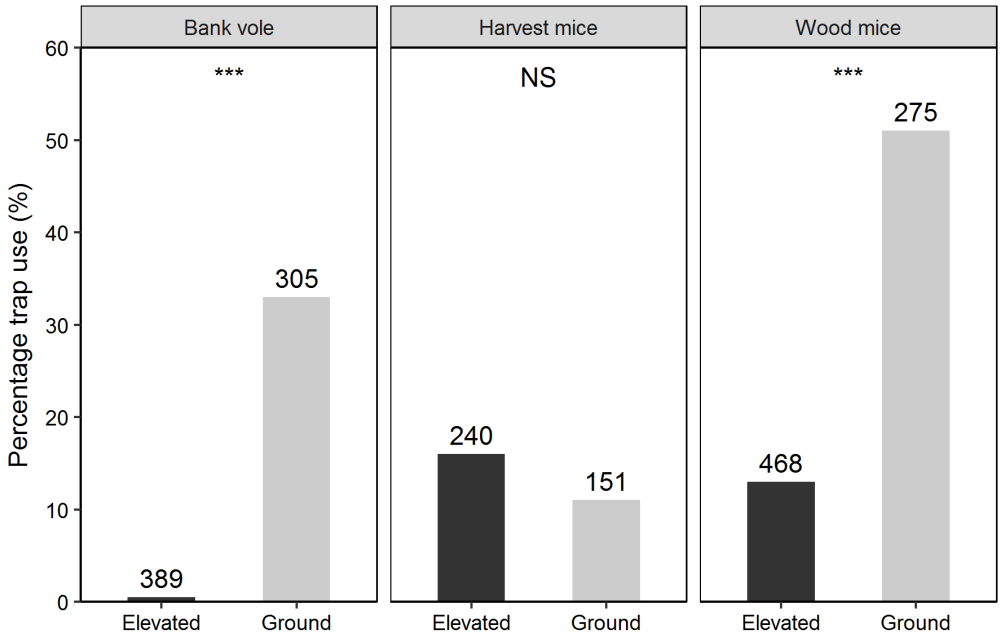


Figure 3.7: The percentage of traps used at either elevated or ground level for the three most common species. Significant differences (*** $p < 0.001$) are shown, as indicated by GLMMs for each species (Table 3.3). Sample sizes (number of trap sites) are indicated above each bar.

Wood mice were significantly more likely to be caught on the ground than in elevated traps (Table 3.3, Figure 3.7). There was also a significant interaction between elevation and the percentage of wetland habitat surrounding the reedbed. A ground trap was more likely to catch a wood mouse if there was more wetland surrounding the reedbed whereas a cane trap was more likely to catch a wood mouse if there was less wetland surrounding the reedbed (Figure 3.8). Bank voles were caught on the ground all but twice (at Sleaford, a constructed site) and the effect of elevation on capture probability for this species was not surprisingly significant (Table 3.3 and Figure 3.7). There were no other significant predictors of the probability of capturing a wood mouse or bank vole.

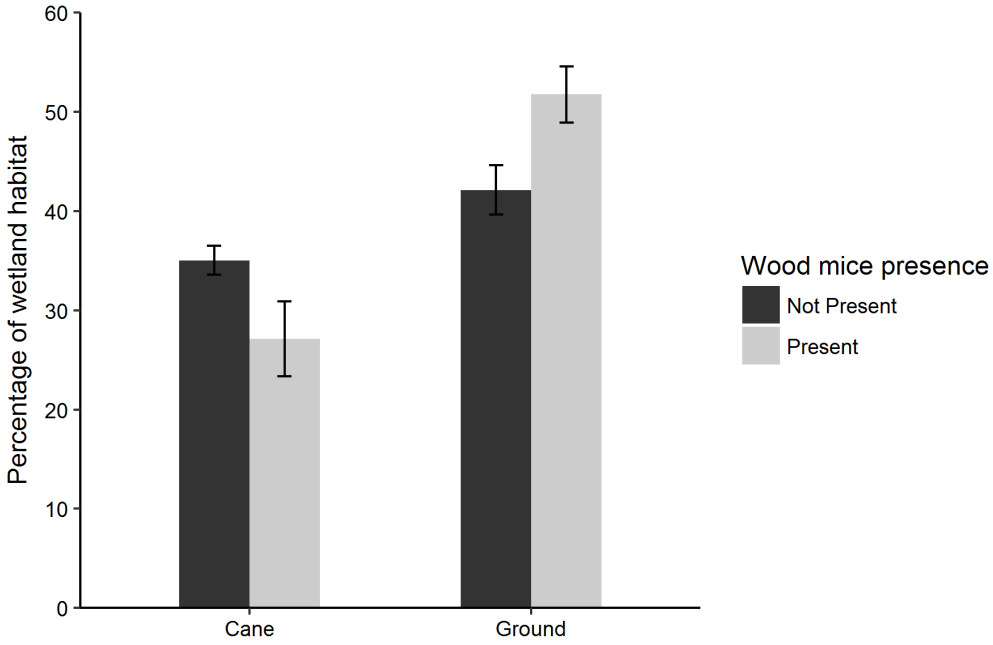


Figure 3.8: Mean percentage wetland habitat surrounding the reedbed (+/- SEM) at trapping sites on a cane and on the ground when harvest mice were present and absent. GLMM (Table 3.3) indicates the percentage of wetland habitat affected wood mouse presence differently dependent on trap placement (χ^2 = 6.7413, df = 1, p = 0.0094).

There was no effect of sex or capture date on the body mass of harvest mice or bank voles, but there was a significant effect of sex on body mass in wood mice, with males being slightly heavier than females (Table 3.4, Figure 3.9).

Table 3.4: Results of generalised linear mixed model testing the effects of sex and date on the body mass of the three most common small mammal species. All two-way interactions were considered; those not shown were not significant.

Analysis	**Random effect**	**Dependent variable**	**Independent variable**	**χ^2**	**df**	**P value**
Gaussian GLMM	Site	Body mass wood mouse	Sex	4.377	1	**0.0400**
			Date	0.0618	1	0.8037
Gaussian GLMM	Site	Body mass harvest mouse	Sex	0.633	1	0.4262
			Date	0.0095	1	0.9224
Gaussian GLMM	Site	Body mass bank vole	Sex	0.9944	1	0.3187
			Date	1.2469	1	0.2642

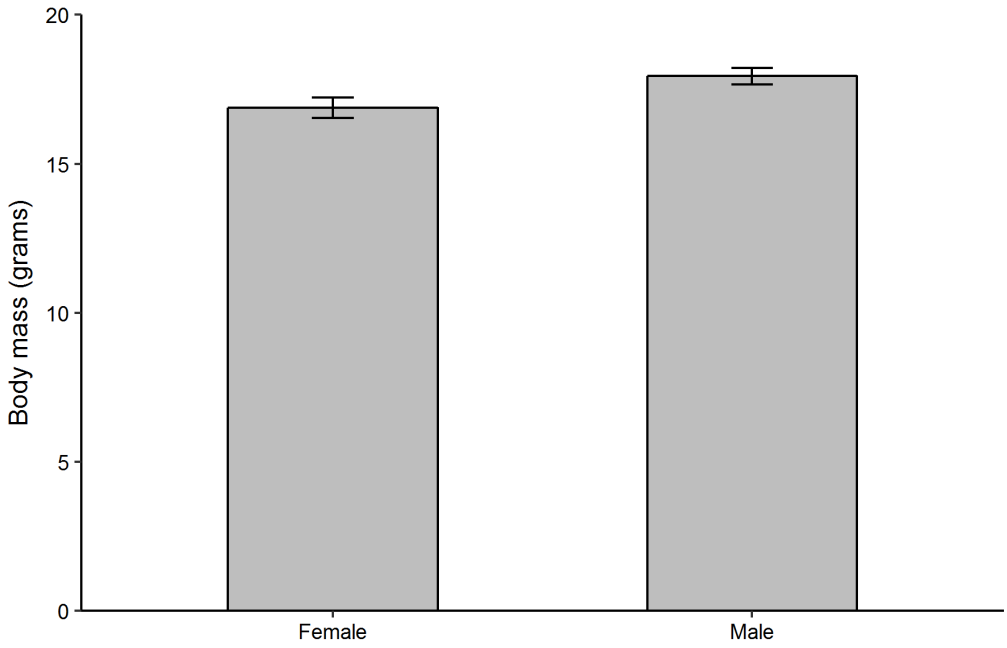


Figure 3.9: Mean body mass (+/- SEM) of male and female wood mice. GLMM (Table 3.4) indicates a significant difference between body mass in male and female wood mice (χ^2 = 4.377, df = 1, p = 0.0400).

3.4 Discussion

This study compared the small mammal communities in constructed and natural reedbeds, and there was no clear evidence that site type had an effect on species richness, diversity or catch per unit effort. This suggests that, despite their small size, young age and potential contamination with pollutants, constructed reedbeds, which are designed and managed for wastewater treatment, can have additional benefits by providing habitat for small mammal communities similar to those seen in equivalent natural habitat. This community contains most of the UK's small mammal species, and is therefore of conservation interest in its own right. Furthermore, the expectation is that this community can provide prey for birds of prey and other predators, some of which are of potential conservation concern, such as barn owl (*Tyto alba*), marsh harrier (*Circus aeruginosus*) and pine martin (*Martes martes*) (Flowerdew, 2004, Buesching et al., 2008).

3.4.1 Abundance and diversity of small mammals in UK reedbeds

The results of this study support previous research which has shown that reedbeds are some of the most diverse habitats for small mammals (Scott et al., 2008, Slabova et al., 2008, Marques et al., 2015). Seven species of nine small mammal species present in the UK were captured in this survey, although two species (house mouse and pygmy shrew) were only caught once, and neither of these species are necessarily associated with wetland habitats. This is more than the five species caught in a recent survey of natural reedbeds in the UK (Kettel et al., 2016), and similar to that seen in other types of habitat such as woodland (Moro and Gadal, 2007).

There was no main effect of reedbed type (constructed versus natural) on the abundance of small mammals. This finding contrasts with those of previous studies of other habitat types, which have indicated that various types of restored habitats may have lower small mammal abundance than natural equivalents (Moro and Gadal, 2007). However contaminated areas of habitat restored from coal mining, similar to the constructed sites treating mine water sampled in this study, suggested that restored wetland habitat is more diverse in comparison to other habitat types (Slabova et al., 2008).

The only main effect on catch per unit effort was species. The results showed that wood mice, which are regarded as habitat generalists (Sozio et al., 2013), were the most abundant species in the reedbeds sampled, whilst other more specialist species such as harvest mice were recorded at lower abundances. This is consistent with findings for a range of habitats in the UK (Moro and Gadal, 2007), where wood mice are often the commonest small mammal. Wood mice may be best placed to exploit heterogeneous habitats within and surrounding reedbeds, especially small reedbeds. Harvest mice, by contrast, may be restricted by the total amount of reed or wetland in the surrounding habitat, although this effect may be seasonal (see below).

Constructed reedbeds are often used for the treatment of water contaminated with organic waste, suspended solids and heavy-metals (Vymazal, 2010). One might expect such contamination to impact negatively on wildlife, including

small mammals. Most studies in the literature consider the effect of bioaccumulation on individuals of small mammal species and not on populations. A few studies have considered the effect of metal contamination and bioaccumulation on small mammal fecundity, the results are mixed (Miska-Schramm et al., 2014, Levengood and Heske, 2008) and the limited literature also suggests that not only is bioaccumulation species-dependent but so is tolerance of contaminants (Nunes et al., 2001). I found no difference in small mammal diversity, richness or abundance between constructed and natural sites, which suggests that the contamination in constructed reedbeds has little or no effect on small mammal populations, or at least that the other factors affecting those populations are much more important. Previous surveys have shown that the abundance, demographics and reproductive activity of small mammals are not negatively affected by contaminants, even when they exceed critical concentrations observed in experimental studies (Levengood and Heske, 2008). This is supported by other studies that show contamination and bioaccumulation do not affect the invertebrate community which are often the prey items for small mammals (Stephansen et al., 2016). Even when the community structure of invertebrates in contaminated wetlands is representative of pollutant tolerant species, the community still has the capacity to support small mammal species (Batty et al., 2005). However most reedbeds in the UK treat tertiary sewage wastewater (Cooper, 2007), and are thus unlikely to be heavily contaminated with heavy metals and other pollutants of concern for bioaccumulation. Seven of the 12 sites in this study treated sewage which may be why we saw no overall effect on abundance, although CPU and species richness did not differ between sewage and mine-water sites (average CPU at sewage sites was 0.268 and at mine-water sites was 0.237, average species richness at sewage sites was 2.71 and at mine-water sites was 2.0).

3.4.2 Species composition

The species present in natural and constructed reedbeds did not differ markedly, although harvest mice were slightly more likely to be caught at constructed sites in this study. As discussed previously there is conflicting evidence about how contaminants in constructed reedbeds may or may not affect the species using them. Some evidence suggest shrews, due to their carnivorous diet, are more

susceptible to effects of contamination (Wijnhoven et al., 2007), although this was not obvious in the present study, with shrews being recorded at both natural and constructed sites.

3.4.2.1 Harvest mice

The harvest mouse, a UK BAP species, was recorded at half the sites surveyed, six natural and six constructed sites. This, in addition to discussions with nature reserve and constructed site managers, indicates harvest mice are more widely distributed in the UK, at least in wetlands, than is generally realised. At five out of the six constructed sites, the species was recorded for the first time. The one site with a record was WWT Slimbridge, where harvest mice had been recorded on site but not in the constructed reedbeds. Some natural sites only had previous records from nest searching – live animals had never been seen. Of the twelve remaining sites, at which harvest mice were not recorded, two were outside the known range of the species and four on the periphery of their range (NBN, 2011c). Of the remaining six sites within the species' range, Thatcham, Pleasley, Willow Tree Fen and IFTC were paired with sites at which harvest mice were recorded, confirming that harvest mice were present in the same general area of the country. Site-specific factors may have reduced the chances of harvest mice establishing at these particular sites. For example, Willow Tree Fen is a new nature reserve and has only been managed for seven years, despite the presence of reed for 20 years. IFTC is a very small isolated site alongside an airport with only part of the site bordered by agricultural land. Both these aspects of the site may indicate why harvest mice were not present at these sites despite being present at their matched paired site. It is also possible that the survey effort in this study was sometimes not sufficient to detect harvest mice when they were present. Norfolk is known for its expanse of wetland habitats and contains 30% of UK reedbed habitat (Hawke and Jose, 1996). With the harvest mouse being strongly associated with reedbeds, it might therefore be expected that harvest mice would be recorded at Welney natural and treatment reedbeds in Norfolk, despite the failure to record it in the present study. Further, lengthier surveys may be needed to confirm that the species is really absent from these sites.

Previous studies of harvest mice have found that they prefer wetter, denser areas of tall reed that have low plant diversity (Kettel et al., 2016). However my study found that habitat preferences were linked to season, with a preference for denser reed only being evident in autumn, when in general they were scarce. This apparent change in habitat preference may indicate that in autumn, when food is more readily available, the species tends to remain in core reedbed habitat which provides suitable cover for nesting. In contrast, in winter when resources are scarce, they may use more diverse habitats in search of food. The capture of only four harvest mice in autumn limits confidence in such speculation, and could equally indicate that harvest mice preferentially use non-reedbed habitats in autumn, and move to reedbeds in winter. This is also supported by results from this study that suggest harvest mice preferentially use reedbeds with more wetland surrounding them in the winter. Studies of other small mammal species suggest that there are seasonal patterns to habitat use, often linked to predation risk and food availability (Todd et al., 2000), and recent evidence suggests that harvest mice may leave reedbeds during the summer and return only in autumn (Reader et al. unpublished data).

Harvest mice preference for elevated traps (16 % use compared to 11 % use of ground traps), although not statistically significant in this study, is in line with other research, which suggests that aerial trapping is the best method to study harvest mice (Kettel et al., 2016). As suggested by Kettel *et al.* (2016) and others (Buesching et al., 2008), conventional ground trapping could miss harvest mice. In this study, ground traps failed to record harvest mice at five of the twelve sites where they were caught in elevated traps. If this is the case across the UK, where only conventional trapping is used, it may be that harvest mice are under-recorded and hence their conservation value would need to be reassessed with this in mind. Although the effect of elevation was not significant, recording harvest mice mainly in the canopy of the reedbed may indicate a habitat preference. Their ability to utilise this part of the reedbed is be due to their agility and ability to climb compared to other species of small mammals and hence reduce inter-specific competition (Ishiwaka and Mori, 1999, Buesching et al., 2008).

3.4.2.2 Other species

Wood mice were the most abundant species caught during this survey. Wood mice are a common and widespread species but little is understood about their use of reedbeds. They are known to use arable land in large numbers in the summer (Todd et al., 2000). However in winter, when my survey was conducted, wood mice are known to seek habitats that provide better cover from predation (Todd et al., 2000). When arable fields are bare they may move to reedbeds which would provide adequate cover from predators.

Adequate cover from predators is one of the theories to explain arboreality in small mammals. The more agile and fast moving species are capable of avoiding predation even in more exposed, less dense vegetation associated with arboreality (Buesching et al., 2008). In a previous study most arboreal activity took place at 1 m in woodlands, and this resulted in wood mice being arboreal for 20 % of the time and bank voles 10 % of the time (Buesching et al., 2008). In parallel with this my study found that, trapping at 1 m, wood mice were caught off the ground 26 % of the time, although they were still significantly more likely to be caught on the ground. Interestingly, although they are capable of climbing amongst standing reed stems when released from the hand, they lack a prehensile tail and are noticeably less agile and less able to remain off the ground than harvest mice (personal observation).

Bank voles were only recorded twice in elevated traps at Sleaford, and just prior to trapping at this site heavy snowfall had flattened the reedbed and made the elevated traps very accessible without the need to climb reed stems. Buesching *et al.* (2008) noted that those places where bank voles were arboreal were the areas with densest vegetation in woodlands. Areas of reedbed do not provide the same structural support as dense areas of woodland and it may be this factor that reduced the activity of bank voles in the canopy of the reedbed in my study.

Time of year (season) did not affect catch per unit effort of wood mice and bank voles. This suggests that there is little immigration into reedbeds, as was apparently seen with harvest mice in this study. In the absence of migration, wood mice and bank vole numbers would typically be expected to fall from late

autumn to late winter, as a result of the cessation of breeding, and the increased mortality seen in winter in small mammals (Jackson et al., 2001). Winter mortality is strongly affected by temperature (Jackson et al., 2001) and it may be that sampling from diverse locations around the UK meant that any overall effect of winter mortality was masked by local variation in conditions and hence mortality.

The body mass of two of the species commonly encountered in this study, bank vole and harvest mice, was not significantly affected by sex or date. This is probably because the sample size was relatively modest and all the animals captured were adults. By comparison, nearly 300 wood mice were captured and this revealed that males were on average over 1 gram heavier than females. There is contrasting evidence on sexual dimorphism in small mammals, with some evidence suggesting an expectation of larger males, but much of the literature indicating there is no significant difference (Lu et al., 2014). There is an expectation for a decline in body mass over the winter as resources become scarce, and metabolic costs increase, but this was not observed in my study. However, there is some evidence to suggest that both large and small individuals would have increased winter mortality due to not meeting their physiological optimum over winter (Klemme et al., 2008).

Little is understood about the water shrew's distribution and abundance in the UK. There has never been a nationwide survey and they are especially hard to survey because they are elusive and have relatively low population densities (Greenwood et al., 2002). Although water shrews are known to use reedbeds, more research is needed to understand their habitat preferences (Greenwood et al., 2002). Their range encompasses most of the sites surveyed, but they were only recorded at one site. To establish whether or not water shrew may use constructed reedbeds, further surveys will be needed. A less labour-intensive way to survey for water shrews than live-trapping is using bait tubes (Churchfield et al., 2000) and identifying the aquatic invertebrates in their scat. Many constructed reedbeds have been surveyed for aquatic macro-invertebrates; this offers the potential of utilising those records to establish whether the prey items of water shrew are present, and hence whether the contamination

associated with constructed reedbeds could affect their presence or not. There is some evidence to suggest that within local populations of water shrews, there is avoidance of areas with low water quality (Greenwood et al., 2002). This suggests that constructed reedbeds may not be as suitable for water shrews as they are for other species of small mammals.

3.4.3 Size and age of constructed reedbeds

In spite of considerable variation in age among the reedbeds sampled in this study, including an average difference of 10 years between constructed and natural sites, there was no effect of age of the reedbed on catch per unit effort of small mammals. There is evidence that some species of small mammal can colonise new habitat and reach maximum population densities within three years (Fox and Fox, 1984). All of the sites in this study (both constructed and natural) were older than this and this may be why no effect of age was detected. In addition, a previous study found that small mammal abundance and diversity did not increase with age but that changes in seral stage and the structural diversity of the habitat with age were most important (Moro and Gadal, 2007). There is little change in structural diversity within managed reedbeds, where the objective is typically to stall the hydrosere, to maintain a monospecific stand of reed. This is especially true in constructed reedbeds that are designed and managed specifically to maintain water depth and reed density (but see Chapter 1 and 6). Although natural reedbeds are also managed with the aim of keeping a monospecific stand of reed, varying water levels are often a key target (Hawke and Jose, 1996). Successional change in a reedbed leads to the development of woodland, and these areas were avoided in my study, so any effects of age may have been eliminated here.

There was no effect of size of the reedbed on catch per unit effort, despite the general expectation that large habitat patches should maintain higher biodiversity. Specifically for small mammals, it has been shown that the smaller the habitat fragment size, the lower the diversity and abundance (Pardini et al., 2005). The average size of a natural reedbed (~ 3.3 ha) was significantly larger than the average size of a constructed reedbed (~ 0.6 ha) in this study, and this was predicted to be a possible driver of differences in mammal abundance and

diversity between the two types of site, but again there was no such effect evident in the analysis.

It may be that the reedbeds I sampled do not function as isolated islands of suitable habitat, but instead they are part of a complex habitat mosaic supporting small mammal populations over a larger area. This is especially likely to be true for habitat generalists such as wood mice and bank voles, which are likely to be able to exploit the woodland and pasture which surrounded most of my sites. For harvest mice, however, which appear to be heavily reliant on habitat dominated by tall grasses, and perhaps more specifically reed, the results of this study strongly suggest that small constructed reedbeds can support viable populations, even when they are surrounded by a matrix of largely unsuitable habitat. It has previously been shown that harvest mice require less than 1 ha of reedbed to sustain a population (Hawke and Jose, 1996), and my study suggests that much smaller reedbeds may be sufficient. Having said this, the results do hint that the surrounding habitat matrix may matter for harvest mice: they were caught more often in areas surrounded by more wetland in the winter (at which time they were most abundant generally). However, the populations found at two of the smallest constructed sites (Rugeley and Benefield), which are 0.038 ha and 0.0285 ha respectively, were mostly surrounded by hard standing and arable in or pasture, underlining the potential for very small fragments of habitat to support the species.

The persistence of small mammal populations in small constructed reedbeds may have significance for the persistence of the same species at a larger spatial scale. Wetland species are more susceptible to local extinction in habitat patches due to the unpredictable nature of their habitat, associated with water level changes (Gibbs, 1993). If patches of artificial habitat help to reduce the isolation of natural reedbeds, by providing "stepping stones" for dispersing animals for example, these sites may contribute to the stability of larger meta-populations (Gibbs, 1993). There are over 1,000 constructed reedbed sites in the UK, nearly all of which have been built in the last 30 years (Cooper, 2007). This is more than the estimated 900 natural sites (BRIG, 2008), and hence may be having a

substantial impact on the persistence of wetland species by providing connectivity across the landscape.

3.4.4 Other factors affecting small mammals in UK reedbeds

Habitat characteristics such as reed density, water depth and vegetation composition were not significant in determining the catch per unit effort for any of the species. These habitat characteristics were deliberately matched at natural and constructed paired sites (see Chapter 2), to maximise my ability to detect differences in the animal community which are driven by factors specifically related to the artificial nature and industrial function of constructed reedbeds. This may have limited my ability to detect other effects of habitat characteristics which may actually be important in determining the nature of the animal community present. Other wetland studies of small mammals have shown preferences for areas with increased plant cover and plant diversity (Canepuccia et al., 2015). The structure of the vegetation has also been shown to be important, providing shelter and climbing opportunities for small mammal species (Wijnhoven et al., 2005). Constructed reedbeds are designed as monocultures, with uniform water depth and reed density, and little opportunity for vegetation succession, providing limited habitat heterogeneity. By contrast, natural reedbeds often provide a succession of habitat types, from wet reedbed to dry reedbed to scrub. Many of these habitat types were ignored in the present study, and so my finding that constructed reedbeds have very similar mammal communities to natural sites may only tell part of the story.

3.5 Conclusions

Conservation efforts do not and cannot rely solely on natural pristine habitats and hence the importance of artificial habitats is increasing. The conservation of species in natural habitat is reliant upon the management of adjacent artificial habitats (Toral et al., 2012). Despite their industrial function, and some concerns over contamination with pollutants, my study shows that constructed reedbeds provide a suitable habitat for many of the UK's small mammal species, and conservation efforts could utilise this fact. Natural reedbeds are managed for the benefit of the species that use them. Constructed reedbeds will always be used

first and foremost for wastewater treatment, but if they are designed and managed to maximise the diversity and abundance of small mammals and other wildlife, this could provide a significant boost for wetland biodiversity in UK, Europe and beyond. The main species of conservation concern in this study of small mammals is the harvest mouse. The utilisation of constructed reedbeds by the harvest mice, potentially more so than natural reedbeds, indicates that they already provide a conservation benefit.

Chapter 4: Moths in constructed and natural reedbeds in the UK

4.1 Introduction

It is projected that 50 – 90 % of the existing insect species are yet to be discovered and known insect species already provide 50 % of known biodiversity (Thomas, 2005). Despite this, knowledge of trends in insect diversity and population size, and factors affecting them, is lacking, especially compared to vertebrate and plant groups (Thomas et al., 2004). Those insect taxa that are well-studied are often utilised as indicator taxa. The use of indicator taxa to track changes in environments and or in biodiversity is a concept that has been around for decades and utilises an indicator species or taxa as an index of attributes too difficult, inconvenient or expensive to measure (Landres et al., 1988). Although controversial, the appeal of indicator taxa to conservation organisations and governments as a time and cost effective monitoring tool is overwhelming (Carignan and Villard, 2002). Well-studied insect taxa can act as indicators of broader patterns such as changes in biodiversity, and responses to threats such as climate change, which is adding to the already detrimental effects of global habitat loss and fragmentation (Mantyka-Pringle et al., 2012).

Although, it should be considered that global and even continental extrapolations of trends identified from a small number of taxa are problematic because insect species often have localised populations and specialised niches (Thomas, 2005). And despite the large body of literature identifying the limitations associated with the use of indicator species, they may still be useful if caution is applied during their selection and in the interpretation of their responses to environmental change (Carignan and Villard, 2002). The monitoring of exhaustive groups of taxa and species to monitor trends and responses to threats such as climate change is unrealistic. However, the monitoring of a wide range of indicator taxa, across broad geographic areas, enabling the understanding of the interacting effects of climate change, habitat loss and fragmentation to inform policy is crucial for current conservation efforts to be effective (Mantyka-Pringle et al., 2012).

Reliable indicator taxa are crucial to monitoring changes in insect distributions, abundance and diversity (Fox et al., 2011). Considerable advances in the

measurement of patterns of insect decline and how indicator species represent other taxa have been made in the last few decades (Thomas, 2005). It has been suggested that one of the only insect taxa which can realistically be monitored with precision across many parts of the world is the butterflies (Thomas, 2005). However, despite being monitored effectively in the UK, butterflies are relatively species-poor, and are not therefore ideal indicator taxa. The closely-related macro-moths, by contrast, are a relatively well-studied and diverse group of taxa in the UK which are ideally suited to studies seeking to detect geographic and temporal patterns in insect biodiversity.

The macro-moths comprise an ecologically diverse group of insects in the UK, with some 900 readily-identifiable species, including representatives which exploit almost all terrestrial biomes (Fox et al., 2011). They are well recorded in the UK, with only birds, plants, butterflies and spiders having more records (Thomas, 2005). The diversity of the group, and the wealth of historic records collected by moth recorders, provide an ideal opportunity for moths to be used as indicator taxa for the study of insect biodiversity (Fox et al., 2011). Macro-moths are also reported to provide an excellent indication of ecosystem health (Kitching et al., 2000).

Many of the common macro-moth species in Britain are undergoing a major population decline (Conrad et al., 2006). Their decline is more severe than those recorded for British birds and plants, and similar to that recorded for butterflies. This suggests that British insect biodiversity is declining fast, with potentially serious implications for the provision of ecosystem services (Conrad et al., 2006). Macro-moth species in all life stages provide a range of ecosystem services. For example, they are the dominate pollinators of flowering plants after dark in temperate zones (Devoto et al., 2011). During their larval stage moths are significant nutrient recyclers and primary consumers (Merckx et al., 2012). They are also important prey items for many other taxa, including bats and birds. For example, blue tit (*Cyanistes caeruleus*) chicks eat an estimated 35 billion caterpillars in Britain every year (Merckx et al., 2012). A review of the diets of British bats showed that all 15 species studied included Lepidoptera as prey items, and four of them (included two endangered species) have moths as their

main prey item (Vaughan, 1997). Hence, moths are not only a significant component of biodiversity in many habitats – they also support a range of other species of conservation value.

As of 2007, 81 species of moth are UK Biodiversity Action Plan (BAP) species, with a further 71 others added to 'stimulate research' as they are widespread but rapidly declining (Fox, 2013). Those listed for research purposes are included to encourage study to understand the effects on species with which they interact, including predators, and also to understand how wider changes in UK habitats might be affecting moth abundance.

As well as taxa of conservation concern, the UK BAP also lists a number of threatened habitats, one of which is reedbed. Reedbeds are areas of wetland dominated by often monospecific stands of common reed (*Phragmites australis*) (BRIG, 2008). Reedbeds are important for invertebrates: over 700 species of invertebrate associated with reedbeds, 64 species are dependent on common reed to some degree and 40 of those are entirely dependent on it (Hawke and Jose, 1996). Eleven of the species that are totally dependent on common reed are moths (Waring and Townsend, 2009).

The majority of the moth species that are dependent on *Phragmites* are stem-boring moths and cause characteristic damage. The stem-boring larvae kill the growing shoot causing side-shoots to grow from the internodes below the damage (Tscharntke, 1999). The larvae will typically eat out several internodes before moving on to another stem, they will use at least three stems before pupating and overwintering inside the stems (Tscharntke, 1992). In optimal reedbed habitat - those areas with thick stems - some species can reach high population densities which can result in up to 96 % of stems being damaged. These high population densities result in flush-crash cycles and although not optimum habitat, small reedbed patches, that are likely to have thin, stressed reed still experience flush-crash cycles and cannot sustain a population without a stable population nearby (Tscharntke, 1992).

Reedbed habitat in the UK is highly fragmented: there are approximately 5,000 ha in England, with only 50 of 900 sites being over 20 ha. There has been a recent proliferation of large reedbeds (totalling up to 1,000 ha) in the UK in recent years, primarily as a result of efforts to provide habitat for large charismatic species of conservation interest, such as the bittern (*Botaurus stellaris*) (Hardman et al., 2012). Recent studies have shown that this proliferation has been beneficial for other taxa. For example, recently restored or created reedbeds adjacent to older reedbeds support similar numbers of reedbed-specialist invertebrates (Hardman et al., 2012).

There has also been a proliferation of smaller reedbeds designed to treat wastewater. Reedbeds that have been planted with specialist drainage networks and with water treatment in mind are often referred to as "constructed" reedbeds. They are designed to treat sewage wastewater, mine-water, sludge, industrial effluents and surface and road runoff (Cooper, 2007). But in contrast with the rarer, larger reedbeds, almost nothing is known about their contribution to UK biodiversity. There are also concerns about bioaccumulation of toxins from wastewater, but there is evidence to suggest that wildlife is able to thrive in constructed reedbeds despite raised levels of pollutants (Stephansen et al., 2016). There is also evidence that management of constructed reedbeds could be adjusted to benefit biodiversity without compromising water quality (Worrall et al., 1997). However, existing studies have typically focussed on one or two constructed sites at a time, and have never systematically compared the communities supported by natural and constructed reedbeds.

Understanding moth habitat preferences is crucial to maintaining their populations and hence the ecosystem services that they provide. Reports of moth reedbed habitat preferences in the literature are scarce. A recent survey of RSPB reedbeds identified a preference for lower water levels and deep litter by reedbed specialist moths, and a greater total abundance of moths at sites with partial flooding (Hardman et al., 2012). However, these data come exclusively from large reedbeds managed for conservation purposes, and it remains unclear the extent to which the patterns identified are applicable in the small and fragmented sites which make-up most of the UK's reedbed habitat.

Most of the small and fragmented reedbeds in the UK, including constructed reedbeds for wastewater treatment, have not been surveyed for moths, and it remains unclear whether they support a diverse community. It is thought that the rare reedbed specialists can persist in small habitat fragments (Tscharntke, 1992), but again this has not been extensively tested. Amongst smaller habitat patches, connectivity and isolation are thought to exert an important influence on patch colonisation, local population persistence and metapopulation stability, especially for invertebrates (Gibbs, 1993, Tscharntke, 1992). Thus, newly-created constructed reedbeds may support important moth diversity in their own right, and they may also support diversity more broadly by providing stepping stones between existing habitat patches, reducing isolation or increasing connectivity.

This chapter assesses the conservation value of moth communities in constructed reedbeds. Light traps were used to trap moths at 12 matched pairs of natural and constructed reedbeds across England and Wales. These surveys aimed to determine whether small constructed reedbeds being used for wastewater treatment support significant moth diversity, and whether these constructed reedbeds differ from natural sites in moth diversity and the abundance of protected reedbed specialists. The literature reviewed above indicates that there could be an expectation that generalists may be present in the smaller, polluted constructed reedbeds but that reedbed specialists, for example the wainscot species, may be absent from those smaller, polluted constructed reedbeds. I also examined the effect of a range of environmental variables on moth diversity and abundance. I consider the implications of these findings in the context of moth conservation and constructed reedbed management.

4.2 Materials and Methods

4.2.1 Field surveys

Twelve matched pairs of study sites were selected, as described in Chapter 2. A Heath moth trap with a 15 W actinic bulb attached to a lithium battery was placed in the middle of the survey area (survey area/grid described in Chapter 2). The

trap was light-activated, and the mechanism was checked each time the trap was left. Surveys took place during July and August 2015 and 2016. One constructed site, Rugeley had to be completed in 2014 due to planned management works. The matched natural site, Drakelow, was also surveyed at the same time (exact dates for each site can be found in Chapter 2, Table 2.3). The trap came on at dusk (approximately 22.00) and the trap was retrieved at 06.00. Sites were surveyed on two separate nights. At sites where two separate reedbeds were surveyed (see Chapter 2), each reedbed was surveyed for two nights. One site (Shirebrook) could only be surveyed once due to it being unsafe to return to site due to anti-social behaviour. Each individual moth was identified to species and photographed. Those individuals too faded or otherwise unidentifiable were excluded from the analysis of species richness and diversity but they were included in analyses the total number of individuals caught.

Variables thought to predict moth diversity were measured. Plant species richness and habitat characteristics were recorded at each point in the survey area (see Chapter 2). Weather data were taken from published records for nearby recording stations (UK Met Office historical site data). The minimum and average overnight temperature was calculated between 22.00 and 06.00. Average wind speed was also calculated from between 22.00 and 06.00, as was total rainfall.

Caterpillar searches were also conducted in the stems of the *Phragmites australis*. Each quadrat in a 5 m grid system (see Chapter 2) was searched for characteristic damage caused by stem-dwelling moth species (all other stem-dwelling species were surveyed at a different time – see Chapter 5). The species present and their abundance was recorded within each quadrat. Two sites were not accessible for these searches: water levels at Drakelow and Thatcham were too high at the time of sampling to allow safe access. Due to different sampling techniques the results from caterpillar searches are not included in GLMM analyses (described below) of total moth abundance, species richness and diversity, but they were included in the analysis of species richness for reedbed specialist moths.

4.2.2 Statistical analysis

All analyses were conducted using R (version 3.3.2). A series of three Hill numbers (N_0 – species richness, N_1 – the exponential of Shannon index, N_2 – the inverse of Simpson's index) were calculated for all moth species for each trap night at each site. Only individuals identified to species level were included in the analysis. Reedbed habitat characteristics which might predict moth abundance and diversity are known to be correlated, with wetter sites typically having higher reed densities and lower plant diversity. To remove this correlation prior to analysis of patterns in moth data, Principal Component Analysis (PCA) was conducted (see Chapter 2).

The effects of various explanatory variables on total moth abundance, species richness (N_0) and diversity (N_1 and N_2) were explored using Gaussian general linear mixed effects models (GLMMs). Reedbed type (constructed or natural), trap night (first or second), minimum overnight temperature, average wind speed, total overnight rainfall, size (area of reedbed over the whole site), trapping year (2014 / 2015 or 2016; only two sites were visited in 2014 so data were pooled from 2014 and 2015), date (days since start of trapping season), site age (approximate age of specific area surveyed, not the whole site) and habitat characteristics (PC1, see Chapter 2) were all fitted as fixed effects. Random effects were site pair (because pairs of sites were deliberately matched – see Chapter 2) and site (because both trapping nights were analysed simultaneously). In each model, all two-way interactions were considered, but high-order interactions were ignored. The most parsimonious models were selected using a partial forward step-wise consideration of terms; limited sample sizes restricted the fitting of a saturated model, and hence backwards step-wise model selection was not possible. First, a model with all main effects was fitted, and then each two-way interaction was added separately and the effect of the addition on the deviance explained by the model was tested using an F test. Significant interactions were retained, and non-significant interactions were removed. Main effects were then tested by backwards deletion. To find the optimal structure of the random component, the AIC scores of a model with both random effects was compared to models with one or no random effects. In both cases, the AIC was

only slightly improved by removing the random effects, so to be conservative the random effects were retained.

A separate analysis was conducted for reedbed specialist moths only, but small sample sizes did not fitting of a suitable GLMM. Instead, species richness and abundance of reedbed specialist moths were compared at paired natural and constructed sites using Wilcoxon signed rank tests. Reedbed specialist moths were defined as those that rely on *Phragmites australis* for at least one part of their life cycle according to Skinner and Wilson (2009).

4.3 Results

A total of 1,792 individual moths of 190 species were recorded on 51 trapping nights across 26 trapping locations on 24 sites. Fifty individuals could not be identified to species level due to being faded and or damaged. The highest abundance and richness was recorded at Martin Mere (a natural site) with 154 individuals of 34 species (Table 4.1). Over 100 individuals were recorded at five other sites. The lowest species richness was found at Carr Vale, a natural site, with nine individuals of five species, and at its matched constructed site Shirebrook, with seven individuals of seven species. Diversity ranged from 3.37 (N_1) and 1.78 (N_2) at Rugeley to 25.30 (N_1) and 21.51 (N_2) at Amberswood, a constructed site and a natural site respectively (Table 4.1).

Over 100 individuals of three species were recorded: the large yellow underwing (*Noctua pronuba*; 154 individuals) was recorded at all bar three sites, the southern wainscot (*Mythimna straminea*; 162 individuals) at all bar four sites, and the smoky wainscot (*Mythimna impura*; 118 individuals) at all bar seven sites. Four species of IUCN 'Endangered' status (Fox, 2013) were recorded across five sites: dusky thorn (*Ennomos fuscantaria*), double dart (*Graphiphora augur*), garden dart (*Euxoa nigricans*), dark barred twin-spot carpet (*Xanthorhoe ferrugata*). Eighteen species listed as 'Vulnerable' by the IUCN were recorded across 20 sites. 'Endangered' or 'Vulnerable' species were recorded at all bar three sites: Carr Vale, a natural site, IFTC, a constructed site and B+P farm, a natural site.

Table 4.1: Moth species richness and abundance per night of trapping and total abundance, species richness (N_0), and diversity (N_1 and N_2) per site. Abundance data are for every individual caught, whilst richness and diversity only include species identified to species level. *A second night of trapping was not possible at Shirebook due to anti-social behaviour. Site type: N = natural, C = constructed.

Pair	Site	Site type	Night 1		Night 2		Total richness	Total abundance	Diversity	
			Abundance	Richness	Abundance	Richness			N_1	N_2
Hockery Brook	Amberswood	N	34	21	20	15	29	54	25.30	21.51
	Hockery Brook	C	60	20	49	26	33	109	20.47	13.72
Shirebrook	Carr Vale	N	11	5	1	1	5	12	4.16	3.52
	Shirebrook	C	7	7	*	*	7	7	7.00	7.00
Pleasley	Doe Lea	N	40	13	40	18	22	80	13.80	9.84
	Pleasley DP	C	11	8	9	8	27	96	16.46	11.27
	Pleasley ROP	C	21	14	55	16				
Rugeley	Drakelow	N	4	2	28	11	11	32	6.84	4.61
	Rugeley	C	39	8	43	10	15	82	3.37	1.78
Sleaford	Willow Tree Fen	N	49	19	47	19	28	96	13.83	8.60
	Sleaford	C	28	12	50	16	22	78	13.96	9.71

Table 4.1 continued.

Pair	Site	Site type	Night 1		Night 2		Total richness[1]	Total abundance	Diversity	
			Abundance	Richness	Abundance	Richness			N_1	N_2
Slimbridge	Slimbridge N	N	60	17	59	20	27	119	14.20	9.30
	Slimbridge T SF	C	24	14	10	7	41	144	16.94	6.92
	Slimbridge T M	C	31	13	79	25				
Taff Merthyr	Parc Penallta	N	31	14	9	6	16	40	9.98	6.72
	Taff Merthyr	C	19	15	11	8	20	30	17.23	14.51
IFTC	B&P Farm	N	36	12	71	19	24	107	11.38	7.58
	IFTC	C	17	8	31	11	14	48	8.10	5.38
Sheepdrove	Thatcham	N	25	18	31	16	26	56	17.18	10.46
	Sheepdrove	C	16	11	39	20	25	55	22.10	19.84
Benefield	Titchmarsh	N	30	1	53	27	32	83	22.08	15.07
	Benefield	C	18	8	17	10	12	35	6.82	3.98
Martin Mere	Martin Mere N	N	96	24	58	21	34	154	15.45	8.60
	Martin Mere T	C	37	17	77	19	27	114	17.68	12.78
Welney	Welney N	N	12	5	39	11	12	51	6.55	4.27
	Welney T	C	57	22	53	21	31	110	17.07	9.76

Species that are identified as preferring wetland or marshy habitat, according to Waring and Townsend (2009), were recorded at all bar two sites (Rugeley and Taff Merthyr, both constructed sites), with 26 such species recorded in total. Although drinker (*Euthrix potatoria*) caterpillars, which are wetland specialists, were spotted at both sites outside the survey periods.

A total of 322 individuals of eight reedbed specialist species (dependent on *Phragmites australis*) were captured as adult moths across 22 of the sites. The only two sites at which reedbed specialists were not recorded were Rugeley and Taff Merthyr, both constructed sites (Table 4.2). The eight species were: *Chilo phragmitella*, *Donacula forficella*, fen wainscot (*Arenostola phragmitidis*), obscure wainscot (*Leucania obsoleta*), *Schoenobius gigantella*, silky wainscot (*Chilodes maritima*), southern wainscot, twin-spotted wainscot (*Lenisa geminipuncta*). Martin Mere (a natural site) had the highest abundance of these specialists, with 78 individuals of six different species.

Only two species were found as caterpillars in the stems of common reed; brown-veined wainscot (*Archanara dissoluta*) and fen wainscot (Table 4.2). The majority of individuals recorded were brown-veined wainscots (217 out of 235). At one constructed site, Hockery Brook, 91 brown-veined wainscots were recorded from 27 quadrats, yet the adult moth was not caught there, or at any other site. At two sites (Carr Vale and Rugeley) only fen wainscot caterpillars were recorded. Fen wainscots were recorded as adult moths at seven sites, but at only four of these were they recorded as caterpillars. At one of the reedbeds at WWT Slimbridge treatment there was abundant moth damage, but all the caterpillars had left the stems and hence no species identification was possible. This site was visited relatively late in the season. However, in the same week, brown-veined wainscot caterpillars were still present in the smaller treatment reedbed at this site.

Table 4.2: Reedbed specialist moth abundance and species richness (N_0). Species richness includes species found through caterpillar searches. *Caterpillar species: BVW = Brown-veined wainscot, FW = Fen wainscot ** Not accessible for caterpillar searches. Site type: N = natural, C = constructed.

Pair	Site	Site type	Reedbed specialist moth abundance	Reedbed specialist moth species richness	Average caterpillar abundance per quadrat	Caterpillar species*
Hockery Brook	Amberswood	N	8	3	0.22	BVW
	Hockery Brook	C	21	4	3.37	BVW
Shirebrook	Carr Vale	N	2	2	0.03	FW
	Shirebrook	C	1	1	0.21	BVW
Pleasley	Doe Lea	N	3	1	0.38	Both
	Pleasley DP	C	5	3	1.00	BVW
	Pleasley ROP	C	12	1	0.00	None
Rugeley	Drakelow	N	1	1	0.00	None**
	Rugeley	C	0	0	0.27	FW
Sleaford	Willow Tree Fen	N	5	3	0.00	None
	Sleaford	C	22	2	0.33	BVW
Slimbridge	Slimbridge N	N	23	2	0.00	None
	Slimbridge T SF	C	6	2	0.00	None
	Slimbridge T M	C	0	0	0.12	BVW
Taff Merthyr	Parc Penallta	N	2	2	0.20	BVW
	Taff Merthyr	C	0	0	0.00	None

Table 4.2 continued.

Pair	Site	Site type	Reedbed specialist moth abundance	Reedbed specialist moth species richness	Average caterpillar abundance per quadrat	Caterpillar species*
IFTC	B&P Farm	N	30	4	1.50	Both
	IFTC	C	6	2	0.95	BVW
Sheepdrove	Thatcham	N	16	3	0.00	None**
	Sheepdrove	C	6	2	0.54	Both
Benefield	Titchmarsh	N	14	4	0.08	BVW
	Benefield	C	3	2	0.25	Both
Martin Mere	Martin Mere N	N	78	6	0.00	None
	Martin Mere T	C	15	3	0.03	BVW
Welney	Welney N	N	34	3	1.50	BVW
	Welney T	C	9	4	0.31	Both

There was no effect of site type (constructed or natural) on total moth abundance, species richness (N_0), or diversity (N_1 and N_2) (Table 4.3). There was also no difference between site types in reedbed specialist moth species richness (Wilcoxon paired: V = 50.5, n = 12, p = 0.122) or abundance (V = 56.5, n = 12, p = 0.182).

Minimum overnight temperature significantly and strongly affected total moth abundance, species richness (N_0) and diversity (N_1 and N_2) (Table 2, Figure 4.1). The lower the minimum temperature, the lower the diversity, richness and abundance. Abundance was positively affected by habitat variable PC1 (see Chapter 2), with fewer moths being caught in the wet, dense reedbeds typical of the early part of the hydrosere (Figure 4.2). There was also a significant difference between trap nights, with both species richness and abundance being slightly higher on the second trap night (Figure 4.3).

Table 4.3: Results of generalised linear mixed models testing effects of various variables describing site and habitat characteristics on total moth diversity, abundance and species richness. All two-way interactions were considered; those not shown were not significant.

Dependent variable	Independent variable	χ^2	df	P value
Diversity N_1	Minimum temperature	20.792	1	**<0.001**
	Wind speed	0.0890	1	0.7654
	Year	0.0000	1	0.9833
	Size	2.4495	1	0.1176
	Habitat	0.1882	1	0.6645
	Night	0.5679	1	0.4511
	Rainfall	1.9196	1	0.1659
	Date	0.0101	1	0.9199
	Type	1.4067	1	0.2356
	Age	1.0072	1	0.3156
	Type : Age	9.4366	1	**0.0021**
Diversity N_2	Minimum temperature	10.329	1	**0.0013**
	Wind speed	0.5013	1	0.4789
	Year	0.0576	1	0.8104
	Size	2.8783	1	0.0897
	Habitat	0.4748	1	0.4908
	Night	0.7804	1	0.3770
	Rainfall	0.9692	1	0.3249
	Date	0.1856	1	0.6666
	Type	0.3541	1	0.5518
	Age	0.7641	1	0.3820
	Type : Age	6.3645	1	**0.0116**
Species richness	Minimum temperature	63.948	1	**<0.001**
	Wind speed	0.0105	1	0.9183
	Year	0.4486	1	0.5030
	Size	0.8602	1	0.3537
	Habitat	1.0458	1	0.3065
	Night	9.7198	1	**0.0018**
	Rainfall	8.6589	1	**0.0032**
	Date	0.3894	1	0.5326
	Type	1.3055	1	0.2532
	Age	1.9109	1	0.1669
	Rainfall : Date	4.5207	1	**0.0333**
	Type : Age	11.635	1	**<0.001**
Abundance	Minimum temperature	30.522	1	**< 0.001**
	Wind speed	5.1362	1	**0.0234**
	Year	2.8765	1	0.0898
	Size	0.0516	1	0.8202
	Habitat	5.9097	1	**0.0151**
	Night	4.7898	1	**0.0286**
	Rainfall	2.8693	1	0.0903
	Date	0.1230	1	0.7258
	Type	3.2967	1	0.0694
	Age	0.5985	1	0.4392
	Type : Size	4.3177	1	**0.0377**
	Type : Age	4.9623	1	**0.0259**

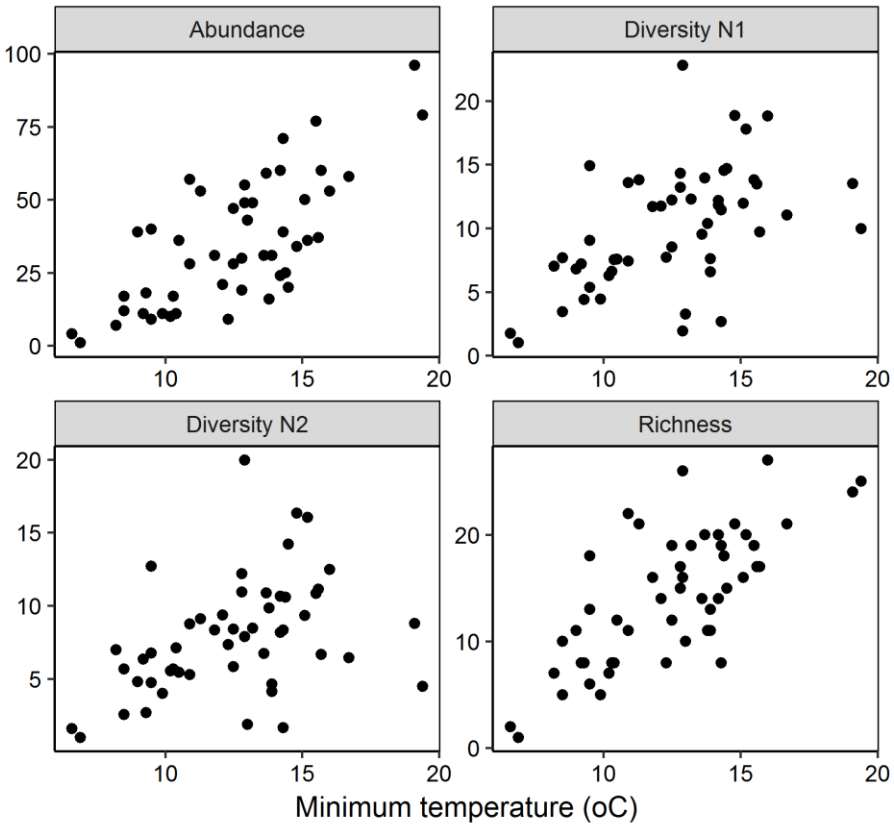


Figure 4.1: The effect of minimum temperature on moth abundance, diversity (N_1 and N_2) and species richness (N_0). GLMM (Table 4.3) indicates significant effect of minimum temperature on: abundance (χ^2 = 30.522, df = 1, p < 0.001), diversity N_1 (χ^2 = 20.792, df = 1, p < 0.001), diversity N_2 (χ^2 = 10.329, df = 1, p = 0.0013) and species richness (χ^2 = 63.948, df = 1, p < 0.001).

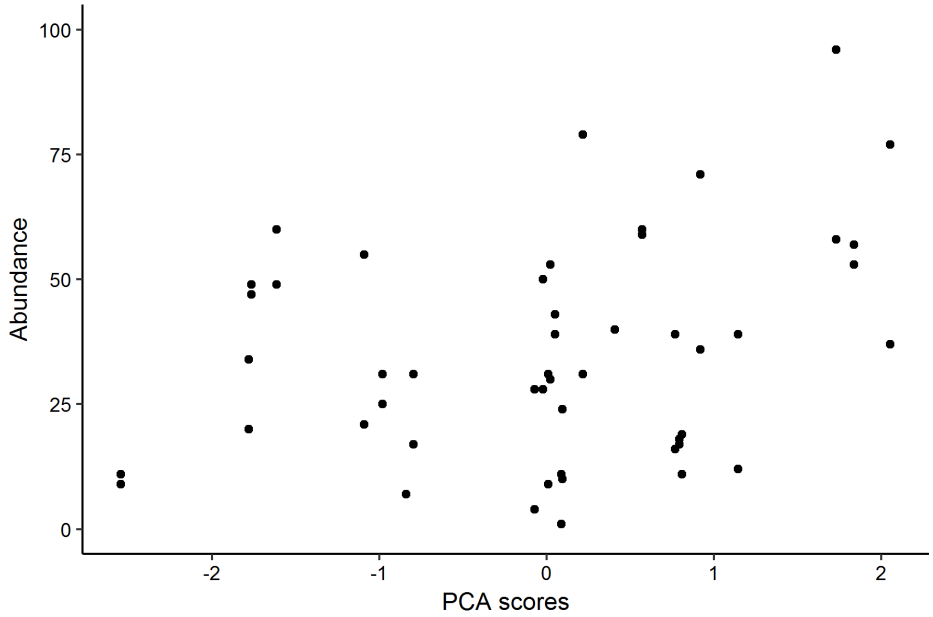


Figure 4.2: The effect of the habitat variable PC1 (see Chapter 2) on moth abundance per night of trapping. GLMM (Table 4.3) indicates significant effect of habitat PCA scores on abundance (χ^2 = 5.9097, df = 1, p = 0.0151)

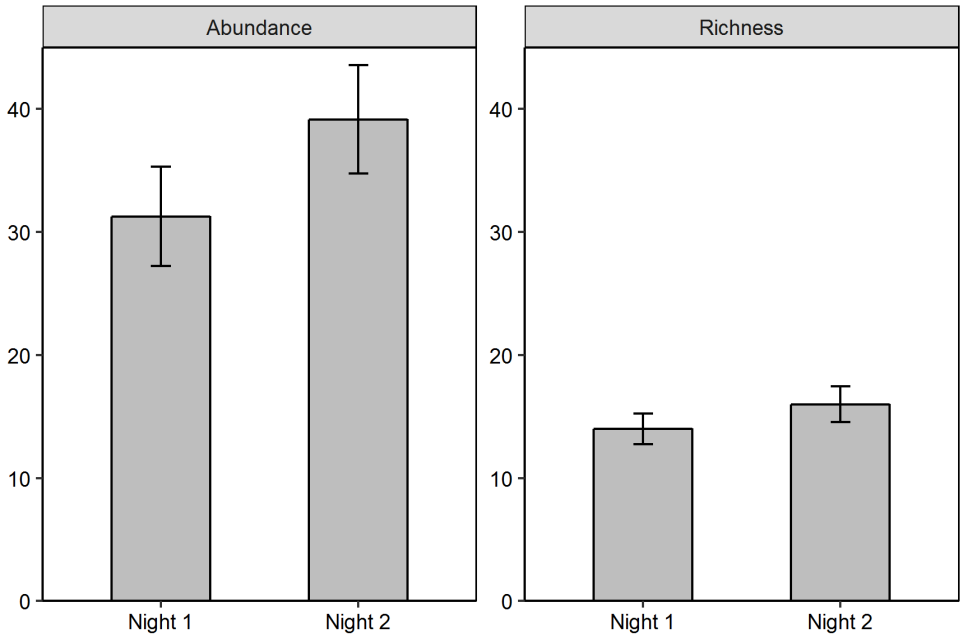


Figure 4.3: The effect of trap night on mean moth abundance and species richness (+/-SEM). GLMM (Table 4.3) indicates significant effect of night on: abundance (χ^2 = 4.7898, df = 1, p = 0.0286) and species richness (χ^2 = 9.7198, df = 1, p = 0.0018).

For all four moth community variables (abundance, species richness (N_0), and two measures of diversity (N_1 and N_2), there was an interaction between site type and age. For constructed sites, older sites tended to have slightly lower moth abundance, species richness and diversity, whereas at natural sites there was no such effect. This effect was strongest for the inverse of the Shannon index (N_1) (Figure 4.4). For abundance, there was also an interaction between reedbed type and size, with size positively affecting abundance at natural sites but having no effect at constructed sites (Figure 4.5). Abundance was also slightly negatively affected by wind. Species richness was also affected by rainfall. Rain only occurred on 15 of 51 trap nights, but it had a slightly negative effect on species richness. There was also a significant interaction of rainfall and date on species richness.

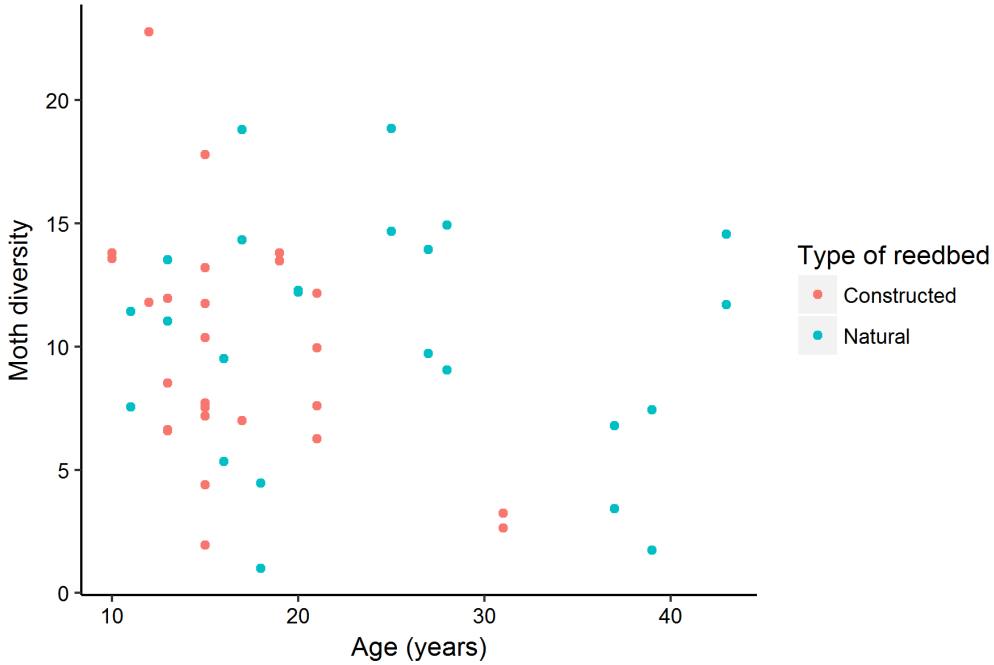


Figure 4.4: The effect of age (years) on moth diversity (N_1) for two types of reedbed, constructed and natural. GLMM (Table 4.3) indicates the age of the reedbed affected the moth diversity differently at constructed and natural sites (χ^2 = 9.4366, df = 1, p = 0.0021).

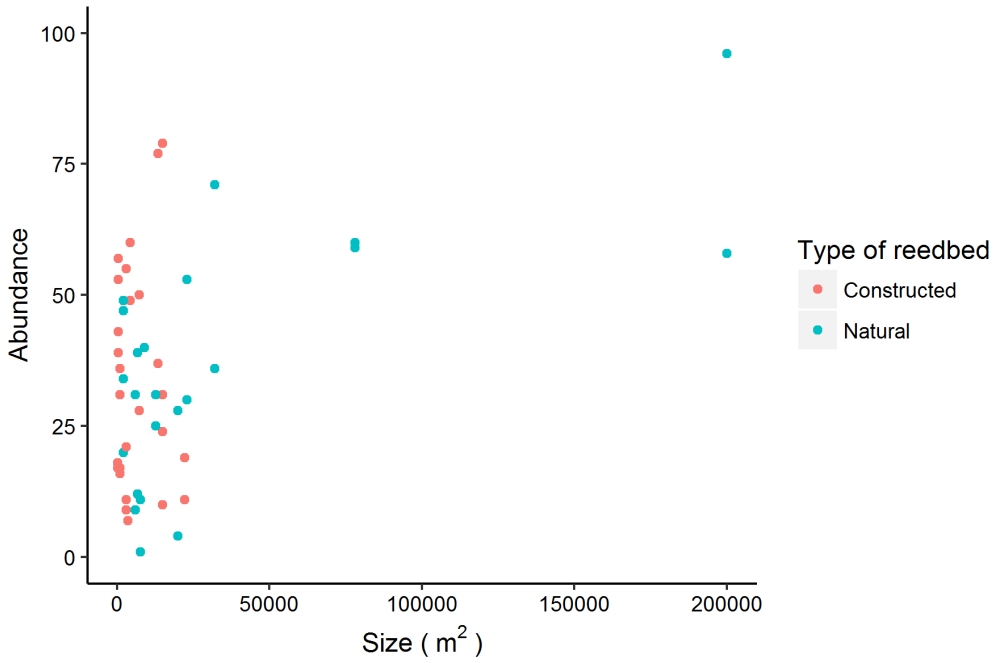


Figure 4.5: The effect of reedbed size (m^2) on moth abundance for 12 constructed (25 trap nights) and 12 natural sites (26 trap nights). GLMM (Table 4.3) indicates the size of the reedbed affected the abudance of moths differently at constructed and natural sites (χ^2 = 4.3177, df = 1, p = 0.0377).

4.4 Discussion

Despite considerable observed variance in the moth community among sites, there was no significant difference in moth abundance, species richness or diversity between constructed and natural reedbeds. Neither was there any difference in the abundance or diversity of reedbed specialist moths between the two types of site. This result suggests that, despite being contaminated, small and relatively young, constructed reedbeds are just as diverse for moths as natural ones.

The abundance and diversity of moths caught in a moth trap is predominately determined by weather and the light source (Yela and Holyoak, 1997, Truxa and Fiedler, 2012). This means that comparisons between studies and habitats for species richness, diversity and abundance can be extremely unreliable. However, a recent study by Hardman et al. (2012) in three UK reedbed nature reserves, using the same type of trap and light source as in the present study, recorded a total of 202 species over three nights. They also recorded 16 reedbed specialist species. These results are very similar to the results seen in this study (190 species and eight reedbed specialists) and suggests that constructed reedbeds provide valuable habitat for moth species in the UK.

4.4.1 Species composition

The three most abundant species were smoky wainscot, large yellow underwing and southern wainscot. The first two of these are generalist and their larvae mainly feed on grasses (Skinner and Wilson, 2009). This was the case for 164 of the species recorded: they were either generalists or fed on several species of common plant. The lack of specialist species may be related to the idea that weak artificial light sources, like that of the light trap used in this study, can only attract species from a 10 m radius (Truxa and Fiedler, 2012), and that reedbeds generally have a low plant species richness. The generalists recorded in this study mainly feed on grasses which means they are probably able to feed on *P. australis*, and may explain why most of the species recorded were generalists.

The southern wainscot, the third most common species in this study, is a reedbed specialist feeding on common reed and canary grass (*Phalaris canaruensis*). It was recorded at 22 of the 26 trapping locations, indicating that constructed reedbeds provide adequate habitat even for reedbed specialist species. Reedbed specialist species were caught in fairly high abundance, with 322 individuals of eight species, and they were caught at all bar two constructed sites. This suggests that despite their primary plant host *Phragmites australis* having potentially bio-accumulated many toxins (Alhashemi et al., 2011), constructed reedbeds are still able to support a range of specialist species.

Previous studies have raised concerns about bioaccumulation through the food chain and toxin build-up within the *Phragmites australis* itself (Alhashemi et al., 2011). However, there is no evidence from this study that, even if bioaccumulation occurs, there is any effect on the species feeding on them. There is no difference between constructed and natural sites in species richness and abundance of the reedbed specialist species. In line with this finding, other surveys in wetlands have shown that even when bioaccumulation occurs, invertebrate biodiversity is not affected and is the same in contaminated and natural sites (Stephansen et al., 2016).

The abundance of caterpillars identified at constructed sites also suggests that there is little negative effect of any toxins associated with constructed reedbeds. Internal plant feeders found are thought to be at risk of the negative effects of the toxic environment (Ye et al., 1997). However, 91 stem-dwelling brown-veined wainscot caterpillars were recorded at one constructed site, despite this being a site treating mine-water with high levels of iron and other heavy metals (The Coal Authority, pers comm). Although none of these caterpillars were recorded as adult moths, which might imply high mortality, neither were they recorded as adult moths at any other site, including natural sites, where pollution is not of concern.

The recording of rare species of conservation status at all bar three sites indicate reedbeds provide essential habitat for species in need of a helping hand. If constructed reedbeds can provide adequate habitat for those moth species of

conservation concern, they may significantly boost the number of sites and potential area of habitat available in the UK. As reported in previous chapters, it is estimated that there are over 1,000 constructed reedbeds in the UK, with an average size of 1,000 m^2 (Cooper, 2007). This is more than the 900 natural sites (BRIG, 2008), although the majority of these are an order of magnitude larger. Although this may limit the contribution of constructed sites to the total area of reedbed in the UK, connectivity plays a vital role in maintaining meta-populations (Gibbs, 1993), and constructed reedbeds may provide stepping stones between natural sites for species of conservation concern.

4.4.2 Age and size

The opposite effects of age on species richness, abundance and diversity, dependent on reedbed type, could be an indication of the impact of constructed reedbed management and design on moths. These results need to be interpreted with caution, as they were strongly influenced by data from the oldest single constructed site, but they may be explained if toxins accumulate with age in constructed reedbeds. The expectation if this was causing a detrimental effect would be to see this effect especially on reedbed specialists, and especially those species whose caterpillars are internal feeders on reed, but reedbed specialists and caterpillars were found at both constructed and natural sites. As already stated, bioaccumulation has been shown to have no effect on invertebrate biodiversity in wetlands (Stephansen et al., 2016). Bioaccumulation and toxicity data were not available for this project; these data alongside further biodiversity studies would give a clearer indication of whether any bioaccumulation was having a negative effect on moth diversity over time.

The maintenance of species richness, abundance and moth diversity as natural sites get older is as expected. These sites are designed and managed for biodiversity. However, in contrast, constructed sites are designed and managed with water quality in mind, and very few are managed at all. These results may be an indication that the lack of management in constructed reedbeds results in less favourable conditions for moths. This may be because of the build-up of leaf litter in constructed reedbeds (personal observation). Whilst in natural reedbeds it has been shown that deeper litter increases moth diversity (Hardman et al.,

2012), the deep litter that accumulates with lack of management at constructed sites is regularly inundated with waste water and for this reason is unlikely to benefit biodiversity. It may not be the waste water per se that is the problem; there is evidence to suggest even in natural reedbeds that annual alternation between submergence and exposure is highly disruptive for invertebrates and results in low species diversity (Bedford and Powell, 2005).

The increased moth abundance seen with the increase in size of natural sites has to be interpreted with caution, due to the fact that only a few large sites were included in this study. There is little evidence of a relationship with reedbed size in constructed sites. Previous research in the UK has shown that the overriding effect on moth abundance is weather (as seen in this study, see below) but habitat size is still thought to be important, especially for specialist species (Botham et al., 2015). The effect of site age was not tested separately for reedbed specialist species in this study because of small sample sizes, but their richness and abundance was not significantly different at constructed and natural sites, despite constructed sites being significantly smaller (see Chapter 2).

4.4.3 Environmental effects on moth diversity, species richness and abundance

It has been show previously that low temperatures and high wind speed reduce the abundance of moths that come to a light trap (Yela and Holyoak, 1997). This is expected, as moths struggle to fly in high winds, especially the smaller species. At low temperatures only the larger species can generate enough heat to activate their flight muscles. My study showed that moth diversity, species richness and abundance were all positively affected by minimum overnight temperature and moth abundance was negatively affected by wind speed. Yela and Holyoak (1997) suggest that temperature is more important than wind speed in determining the abundance of moths. My results are in line with this with a strong effect of minimum temperature in all models. The effect of wind on abundance may have been significant because all moth species, including micro-moths which are more vulnerable to high wind speeds, were analysed in this study, whereas the previous study by Yela and Holyoak (1997) only reported effects on macro-moth species.

Rainfall prior to and during moth trapping can affect species richness and abundance (Wilson et al., 2015). Again, as with other weather effects, this is as expected, because moths struggle to fly in heavy rain-fall. This especially affects smaller species. In the present study, rainfall had a slight negative effect on species richness, whereas most previous studies record a stronger relationship with abundance. Again this may be a result of including micro-moths in the analysis, compared to other studies that only considered macro-moths. Wilson et al. (2015) showed that during the flight season of many common British moths, rainfall has a negative effect on abundance. There is also evidence that rainfall of up to 1.5 cm 24 hours prior to and during trapping has a negative effect on moth abundance (Butler et al., 1999). The present study only recorded a few nights of rainfall, and this may be why no relationship with rainfall was detected for abundance.

Few habitat preference studies have been conducted for moths in reedbeds. However, one recent study by Hardman et al. (2012) showed higher moth species richness in areas with higher plant species richness, and more reedbed specialist moths in areas with lower levels of standing water and deeper litter layers. The present study found similar results, with higher moth abundance at sites with higher plant species richness and lower water levels. The study by Hardman et al. (2012) also found that reed diameter was important for reedbed specialist moths, with thicker stems producing more reedbed specialists.

4.4.4 Other factors affecting moth abundance, species richness and diversity

Trap night affected moth abundance and species richness, with both variables increasing on the second night. This may be because some individuals are attracted close to the trap on the first night, and are then more likely to be trapped on the second night (personal observation).

4.5 Conclusions

Despite concerns about toxicity affecting herbivores that feed on the *Phragmites australis* stems, and the fact that constructed reedbeds are typically smaller and younger than natural sites, there seems to be no reduction in specialist reedbed

species in constructed sites. More generally, the evidence from this study is that constructed reedbeds support the same moth diversity as do natural reedbeds. As expected the over-riding factor affecting moth abundance, species richness and diversity was weather. However, some results indicated that some aspect of a constructed reedbed is limiting the diversity as they get older. Constructed reedbeds will always be first and foremost for wastewater treatment, but if they could be designed and managed to benefit moth species and other wildlife this could provide the UK and Europe with additional areas of habitat and connectivity. Despite their contamination, young age and small size constructed reedbeds provide a promising artificial habitat which conservation efforts could utilise and benefit.

Chapter 5: Stem-dwelling invertebrates in constructed and natural reedbeds in the UK.

5.1 Introduction

Gall-forming insects contribute substantially to global biodiversity with the number of gall-forming insects on all plants estimated between 21,000 and 211,000 species (Espirito-Santo and Fernandes, 2007). The structure of a plant can help predict the number of insect species per plant; amongst grasses, shoot length predicts species richness (Tscharntke and Greiler, 1995). According to a review by Tscharntke and Greiler (1995), an average of 34 endophagous insect species feed on each perennial grass species. Two thirds of them were parasitoids and half were monophagous.

Phragmites australis, the longest grass, supports the most insect species, amongst grasses (Tscharntke and Greiler, 1995). It is estimated to support over 100 species with 93 % of them being monophagous. Those that form galls also support relatively closed communities of specialised occupants (Stone and Schonrogge, 2003). Areas of wetland dominated by *P. australis* are known as reedbeds (BRIG, 2008). In the UK this habitat is sparse and highly fragmented with only 5,000 ha remaining.

The most common gall-forming species on *P. australis* in the UK belong to the genus of flies called *Lipara*. *Lipara* species induce a single characteristic cigar-shaped gall at the top of the stem (Bogusch et al., 2016). Two species are common within the UK - *L. lucens* and *L. rufitarsis* – and two rare species are also present - *L. pullitarsis* and *L. similis* (UK BAP priority species). The galls of all four species provide a discrete microhabitat which support a large number of inquilines (Bogusch et al., 2016). The *Lipara* genus also supports a large range of parasitoids (Nartshuk, 2006).

Other common species such as *Giraudiella inclusa* and *Steneotarsonemus phragmitidis* are also known to cause characteristic damage on *P. australis* stems in the UK. *G. inclusa* forms ‘rice grain’ like galls (Tscharntke, 1988) and *S. phragmitidis* forms ‘corkscrew’ galls (Haflieger, 2009). Damage by several moth species, the caterpillars of which feed inside the stem without forming a gall, is also common (Haflieger, 2009) (see also Chapter 4).

Studies of stem-dwelling species of *P. australis* in natural reedbeds in the UK are rare. Little is understood about their populations and even less about their natural enemies, parasitoids and predators. Studies from Europe show *L. lucens* and *L. rufitarsis* attack those reed shoots that are under abiotic stress and are less than 4.5 mm whereas *L. pullitaris* and *L. similis* attack reed shoots of any diameter, although mainly 5 to 7 mm (Tscharntke, 1992). This leads to different habitat utilization by the different species: *L. lucens* and *L. rufitarsis* use small patches of reed and the edges of large reedbeds, due to the high abundance of stressed (and therefore thin) shoots, whilst the other two species are more common in large reedbeds with healthy thick reed stems.

G. inclusa show a preference for thinner 'stressed' shoots in dry reedbeds or for thicker shoots with side shoots induced by moth damage in wet reedbeds (Tscharntke, 1992). Because of the preference for thinner shoots *G. inclusa*, can utilise small habitats down to 120 m^2 even when the reed is very stressed.

There has been a proliferation of smaller reedbeds in the UK designed to treat wastewater (Cooper, 2007). Reedbeds that have been planted with specialist drainage networks and with water treatment in mind are often referred to as "constructed" reedbeds. They are designed to treat sewage wastewater, mine-water, sludge, industrial effluents and surface and road runoff (Cooper, 2007). Contamination by heavy metals can induce thin reeds, like those preferred by the two common *Lipara* species, similar to those induced by other forms of abiotic stress (Bogusch et al., 2016). There are some concerns about bioaccumulation of toxins from wastewater in phytophagous insects and other invertebrates in constructed reedbeds, but there is evidence to suggest that invertebrates are able to thrive in such habitats, despite raised levels of pollutants (Stephansen et al., 2016).

Stem-dwelling insects have seldom been studied in constructed reedbeds. One study in sewage purification reedbeds showed that the young age and reduced size of constructed reedbeds used to treat wastewater had a negative effect on the abundance of two gall-forming species (Athen and Tscharntke, 1999). The

presence of parasitoids was also affected by reedbed age. This is expected as young habitats cannot host complex food webs as the parasitoids require their host populations to be well established before colonisation (Athen and Tscharntke, 1999). This was also supported by a study of post-industrial reedbeds (e.g. gravel-sandpits, stone quarries, colliery dumps and reclaimed lignite open-cast mines) which suggested that in the first 10 years after reedbed formation it was mainly *Lipara* spp. that were present, with species-rich assemblages occurring later (Bogusch et al., 2016).

Notwithstanding possible limitations on constructed reedbed communities associated with their small size, they may be beneficial because they provide connectivity to species of conservation value in the UK. There are an estimated 900 natural sites in the UK, whilst there are over 1,000 constructed sites (Cooper, 2007, BRIG, 2008). Research has suggested that monophagous herbivores and their parasitoids, including the stem-dwelling invertebrate assemblages dependent on common reed, are vulnerable to the effects of fragmentation due to a lack of connectivity or continuous habitat (Tscharntke et al., 2002). The additional areas of habitat provided by constructed sites, although small, may create the connectivity essential to maintain biodiversity in the landscape, and could be even more important than creating large areas of habitat as is the current conservation strategy (Tscharntke et al., 2002).

This chapter assesses the conservation value of constructed reedbeds for stem-dwelling invertebrate communities. Reed stem searches were carried out at 12 matched pairs of natural and constructed reedbeds across England and Wales. These surveys aimed to determine whether the community of stem-dwelling invertebrates differs between small constructed reedbeds being used for wastewater treatment and nearby natural reedbeds. The literature reviewed above indicates that there could be an expectation that the smaller, younger constructed reedbeds may only host the initial gall-forming species and fewer parasitoids and predator stem-dwelling species. Specifically I tested for any effect of site type, age, size and other habitat variables on the species richness of stem-dwelling invertebrates in general, and the abundance of *Lipara* spp. and their parasitoids in particular. I consider the implications of these findings in the

context of stem-dwelling invertebrate conservation and constructed reedbed management.

5.2 Materials and methods

5.2.1 Field studies

Twelve matched pairs of study sites were selected, as described in Chapter 2. Each quadrat in a 5 m grid system was searched for characteristic damage caused by stem-dwelling invertebrates (excluding moth damage, which was surveyed separately - see Chapter 4). These surveys took place during September and October of 2014 and 2015 (exact dates for each site can be found in Chapter 2, Table 2.3). Those stems showing characteristic signs of damage of any stem-dwelling species were removed and taken back to the lab. Stems were opened and pupae were stored in appropriate conditions to encourage hatching. Once hatched, individuals were identified to species where possible. Only hatched individuals from *Lipara* galls were counted as being present, due to the potential of the presence of parasitoids and predators.

5.2.2 Statistical analysis

All analyses were conducted using R version 3.3.2. A series of three Hill numbers (N_0 – species richness, N_1 – the exponential of the Shannon index and N_2 – the inverse of the Simpson's index) were calculated for all sites. The effects of various explanatory variables on stem-dwelling invertebrate species richness (N_0) and diversity (N_1 and N_2) were explored using general linear mixed effect models (GLMMs) with Gaussian errors. Reedbed type (constructed or natural), size (area of reedbed over whole site), age (approximate age of specific area surveyed, not the whole site), survey year (2014 or 2015) and habitat characteristics (PC1, see Chapter 2) were all fitted as fixed effects. The random effect was pair (because pairs of sites were deliberately matched – see Chapter 2).

The effects of various explanatory variables on proportion of stems damaged by *Lipara* spp. within a quadrat were also explored using GLMMs with binomial errors. Reedbed type (constructed or natural), size (area of reedbed over whole

site), age (approximate age of specific area surveyed, not the whole site), survey year (2014 or 2015) and average diameter of stems within a quadrat were all fitted as fixed effects. The random effects were site (because all quadrats within a site were analysed simultaneously) and pair (because pairs of sites were deliberately matched – see Chapter 2).

The effects of various explanatory variables on the proportion of *Lipara* galls per site in which larvae had been parasitised were also explored using GLMMs with binomial errors. Reedbed type (constructed or natural), size (area of reedbed over whole site), age (approximate age of specific area surveyed, not the whole site) and survey year (2014 or 2015) were all fitted as fixed effects. The random effect was pair (because pairs of sites were deliberately matched – see Chapter 2).

In each of these models, all two-way interactions were considered, but high-order interactions were ignored. The most parsimonious models were selected using a partial forward step-wise consideration of terms (limited sample sizes restricted the fitting of a saturated model, and hence backwards step-wise model selection was not possible). First, a model with all main effects was fitted, and then each two-way interaction was added separately and the effect of the addition on the deviance explained by the model was tested using an F test (Gaussian) or Chi-squared (binomial). Significant interactions were retained, and non-significant interactions were removed. Main effects were then tested by backwards deletion. To find the optimal structure of the random components, the AIC scores of a model with the random effect was compared to models with no random effects. The AIC was only slightly improved by removing the random effects, so to be conservative the random effects were left in.

5.3 Results

A total of 873 *P. australis* stems were found to show signs of stem-dwelling invertebrate damage across the 24 reedbeds surveyed, ranging from 239 stems (19.3 %) at one natural site, Doe Lea, to only two stems (0.22 %) from both Welney Natural and Taff Merthyr, a constructed site (Table 5.1).

The highest stem-dwelling invertebrate species richness was at Doe Lea, where seven species were recorded, and Slimbridge natural, where six species were recorded, both of which were natural sites (Table 5.1). *Lipara rufitarsis* was recorded at all but one site (Welney Treatment), where characteristic damage of *Lipara* spp. was present but no individuals hatched. So it could be suggested that *Lipara* spp. were present at this site for the gall to form that year. *L. lucens* was recorded at seven sites. Parasitoid or predatory species of *Lipara* spp. were recorded at 16 sites, the most abundant parasitoid was *Stenomalina liparae*. The predatory species *Cnemopogon apicalis* was recorded at only six sites, all of which were natural (Carr Vale, Doe Lea, Drakelow, Martin Mere, Slimbridge and Thatcham). *Giraudiella inclusa* was recorded at 11 sites. *Steneotarsonemus phragmitidis* was recorded at six sites (Figure 5.1).

Table 5.1: Stem-dwelling invertebrate species richness (N_0), diversity (N_1 and N_2) and the average number of damaged stems per quadrat at each site. Parasitoid and predatory species include: *S. liparae, Endromopoda phragmitidis, Polemochartus liparae* and *Cnemopogon apicalis*.
*In brackets is the percentage parasitism of all hatched individuals from *Lipara* galls
**Damage caused by *Lipara* spp. present at this site, but no individuals were hatched

Pair	Site	Site type	Average no. of damaged stems per quadrat	Species richness	Diversity		Presence of parasitoids or predatory species*
					N_1	N_2	
Hockery Brook	Amberswood	N	2.6	4	3.37	3.17	Yes (44%)
	Hockery Brook	C	0.5	1	1.00	1.00	No
Shirebrook	Carr Vale	N	3.6	4	3.78	3.61	Yes (41%)
	Shirebrook	C	3.14	4	1.98	1.49	Yes (10%)
Pleasley	Doe Lea	N	9.19	7	3.35	2.59	Yes (47%)
	Pleasley	C	1.15	4	2.79	2.17	Yes (85%)
Rugeley	Drakelow	N	1.05	3	2.87	2.77	Yes (60%)
	Rugeley	C	2.22	4	2.82	2.45	Yes (7%)
Sleaford	Willow Tree Fen	N	0.1	1	1.00	1.00	No
	Sleaford	C	1.17	2	1.53	1.35	Yes (15%)
Slimbridge	Slimbridge N	N	1.66	6	3.31	2.32	Yes (11%)
	Slimbridge T	C	2.29	4	2.94	2.52	Yes (13%)

Table 5.1 continued.

Pair	Site	Site type	Average no. of damaged stems per quadrat	Species richness	Diversity N_1	Diversity N_2	Presence of parasitoids or predatory species*
Taff Merthyr	Parc Penallta	N	0.4	3	2.46	2.13	Yes (25%)
	Taff Merthyr	C	0.1	1	1.00	1.00	No
IFTC	B&P Farm	N	0.95	1	1.00	1.00	No
	IFTC	C	0.25	2	2.00	2.00	No
Sheepdrove	Thatcham	N	1.38	5	3.53	2.96	Yes (20%)
	Sheepdrove	C	0.42	1	1.00	1.00	No
Benefield	Titchmarsh	N	0.42	2	1.75	1.60	Yes (25%)
	Benefield	C	4	4	3.58	3.29	Yes (51%)
Martin Mere	Martin Mere N	N	0.23	1	1.00	1.00	Yes (100%)
	Martin Mere T	C	0.2	2	2.00	2.00	Yes (100%)
Welney	Welney N	N	0.125	1	1.00	1.00	No
	Welney T	C	0.25	0**	0.00	0.00	No

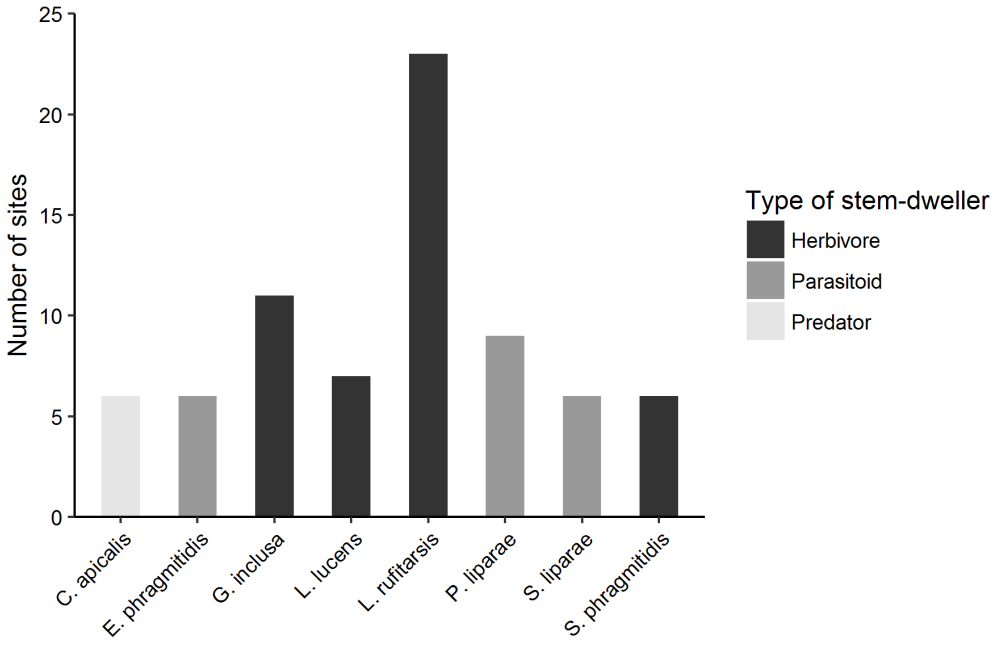


Figure 5.1: The number of sites at which reed stem-dwelling predators, parasitoids and predators were present.

There was no difference between site types in their stem-dwelling invertebrate richness or diversity (Table 5.2, Figure 5.2). Species richness and diversity were different between the two sampling years, with higher species richness and diversity being recorded in 2014 than 2015 (Table 5.2, Figure 5.3). There was a significant positive effect of age on species richness and one measure of diversity (N_1) (Table 5.2, Figure 5.4). There were no other significant predictors of stem-dwelling invertebrate species richness or diversity.

Table 5.2: Results of generalized linear mixed model testing various variables describing site and habitat characteristics on stem-dwelling invertebrate species richness (N_0), diversity (N_1 and N_2), percentage *Lipara* damage and percentage parasitism of *Lipara* galls. All two-way interactions were considered those not shown were not significant.

Dependent variable	**Independent variable**	χ^2	**df**	**p**
Species richness	Site type	0.1438	1	0.7045
	Size	0.1737	1	0.6768
	Age	4.7437	1	**0.0294**
	PCA	0.5102	1	0.4757
	Year	10.348	1	< **0.001**
Diversity N_1	Site type	0.4077	1	0.5231
	Size	0.4087	1	0.5226
	Age	4.3471	1	**0.0370**
	PCA	0.2232	1	0.6366
	Year	8.4243	1	**0.0037**
Diversity N_2	Site type	0.5317	1	0.4659
	Size	0.6947	1	0.4046
	Age	3.7897	1	0.0515
	PCA	0.1911	1	0.6620
	Year	6.3635	1	**0.0116**
Percentage *Lipara* damage	Site type	0.3278	1	0.5669
	Size	0.8380	1	0.3600
	Age	0.0059	1	0.8081
	Average diameter	29.395	1	< **0.001**
	Year	5.1071	1	**0.0238**
	Year : diameter	17.299	1	< **0.001**
Percentage parasitism of *Lipara* galls	Site type	4.1734	1	**0.0410**
	Size	0.1383	1	0.7100
	Age	2.3502	1	0.1253
	Year	0.5346	1	0.4647

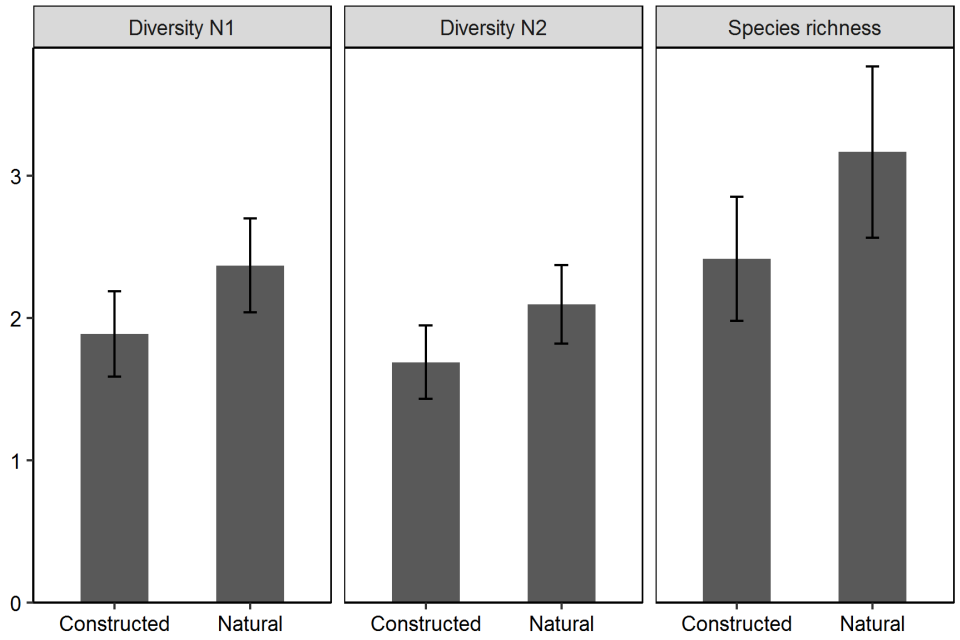


Figure 5.2: Mean species richness and diversity of stem-dwelling invertebrates (+/- SEM) in 12 constructed and 12 natural reedbeds. GLMM (Table 5.2) indicates no significant difference between site types for species richness ($\chi^2 = 0.1438$, df = 1, p = 0.7045) and diversity (N_1, $\chi^2 = 0.4077$, df = 1, p = 0.5231, N_2, $\chi^2 = 0.5317$, df = 1, p = 0.4659) of stem-dwelling invertebrates.

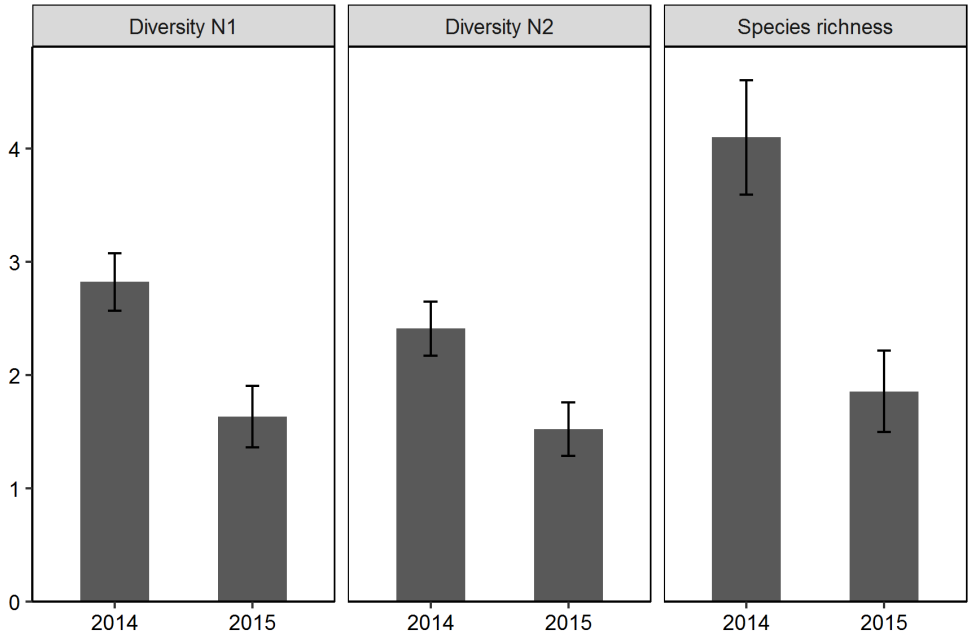


Figure 5.3: Mean species richness (N_0) and diversity (N_1 and N_2) of stem-dwelling invertebrates in two survey years (+/- SEM). GLMM (Table 5.2) indicates a significant difference between years for species richness ($\chi^2 = 10.348$, df = 1, p < 0.001) and diversity (N_1, $\chi^2 = 8.4243$, df = 1, p = 0.0037, N_2, $\chi^2 = 6.3635$, df = 1, p = 0.0116) of stem-dwelling invertebrates.

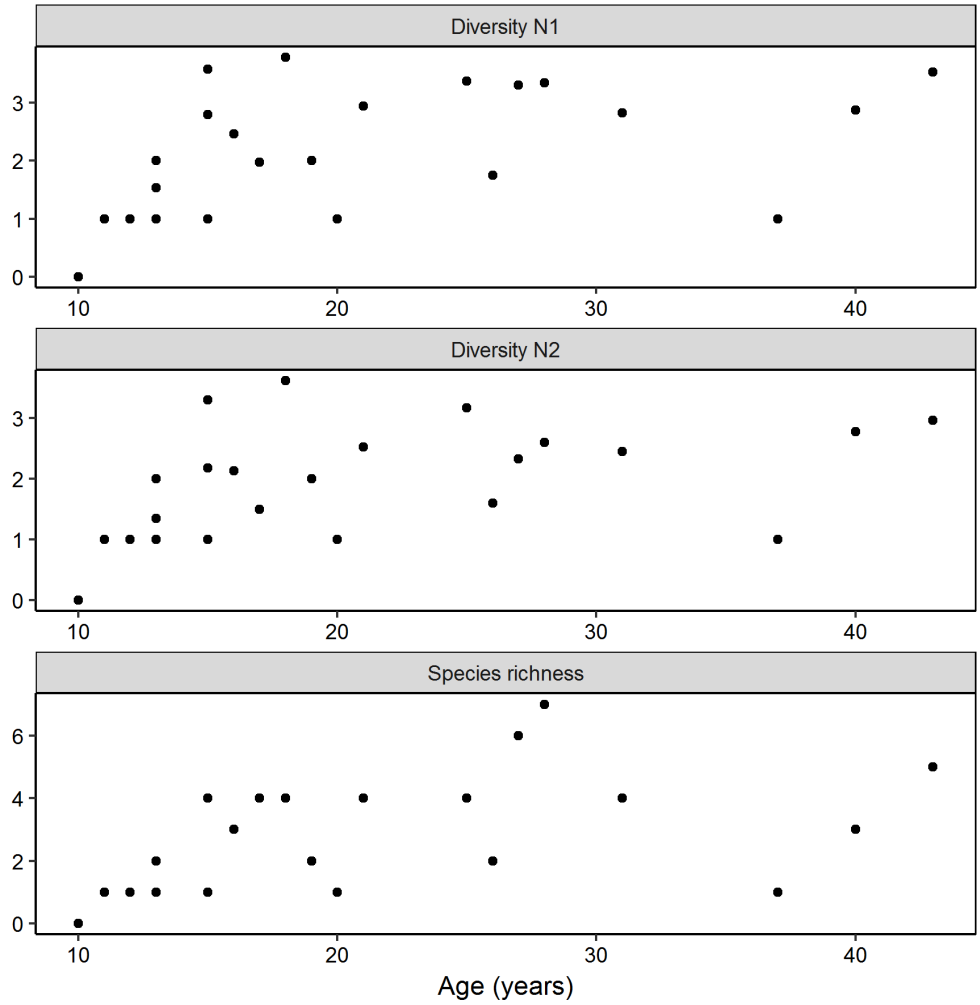


Figure 5.4: The effect of age on species richness (N_0) and diversity (N_1 and N_2). GLMM (Table 5.2) indicates a significant effect of age for species richness (χ^2 = 4.7437, df = 1, p = 0.0294) and diversity N_1 (χ^2 = 4.3471, df = 1, p = 0.0370) and no significant difference for diversity N_2 (χ^2 = 3.7897, df = 1, p = 0.0515) of stem-dwelling invertebrates.

The percentage *Lipara* damage was significantly higher in 2014 than 2015 (Table 5.2, Figure 5.5). Generally, more stems were damaged by *Lipara* spp. in quadrats with a lower stem diameter, but this relationship was stronger in 2014 than in 2015 (Table 5.2, Figure 5.5). The number of *Lipara* stems damaged in 2014 was 419 compared to 188 in 2015. The percentage damage did not differ between constructed and natural sites.

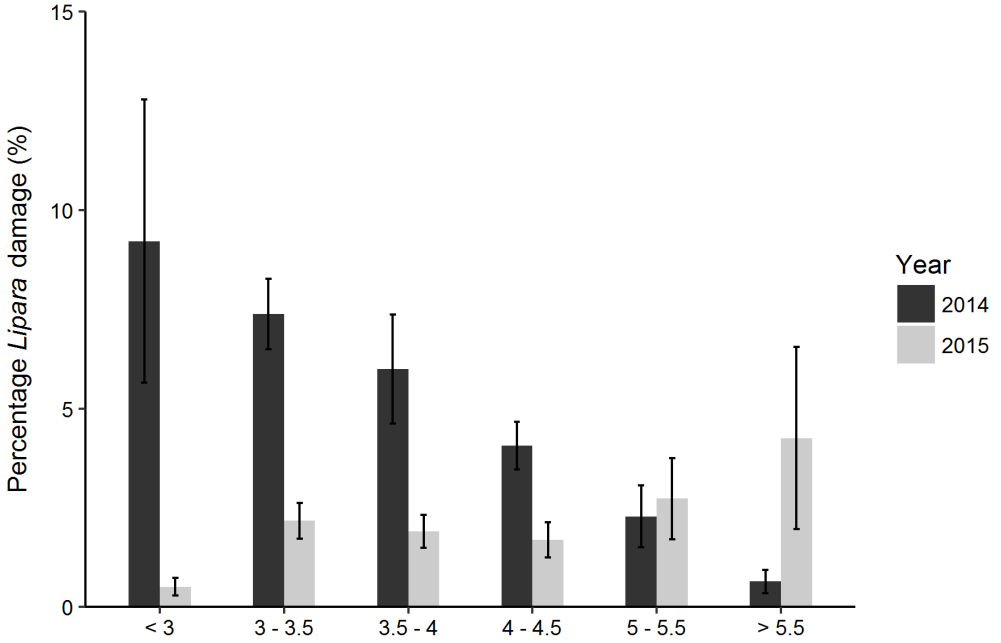


Figure 5.5: Mean percentage *Lipara* damage per quadrat (+/- SEM) against the average reed stem diameter per quadrat in two sampling years. GLMM (Table 5.2) indicate the effect of average diameter on percentage *Lipara* damage was dependent on year (χ^2 = 17.299, df= 1, p < 0.001).

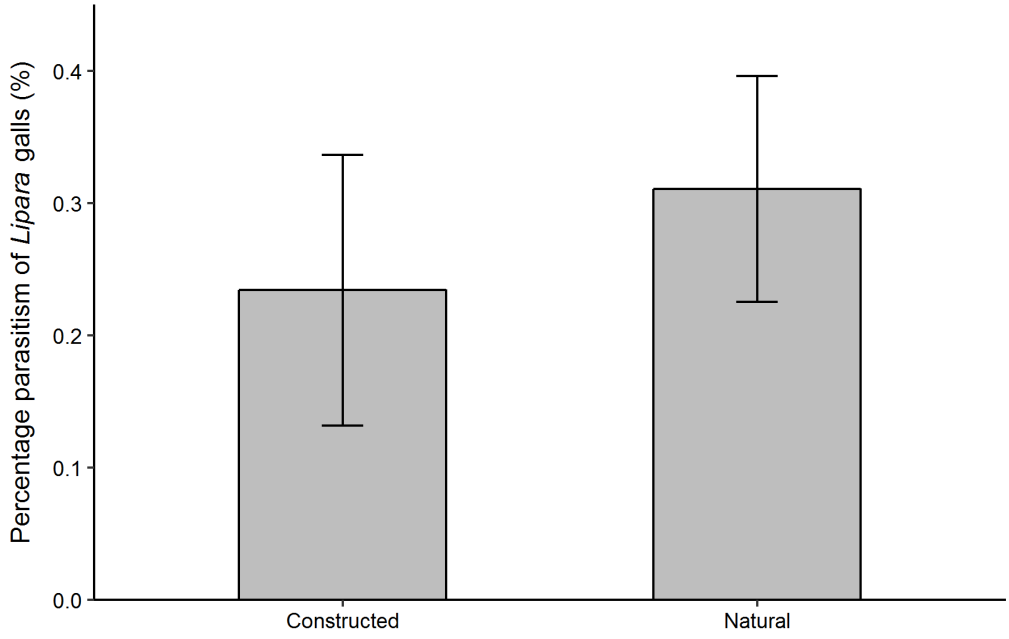


Figure 5.6: Mean percentage parasitism of *Lipara* galls (+/- SEM) in 12 constructed and 12 natural sites. GLMM (Table 5.2) indicates a significant difference between constructed and natural sites (χ^2 = 4.1734, df = 1, p = 0.0410).

There was a slight significant difference between rates of parasitism of *Lipara* spp. at constructed and natural sites (Table 5.2, Figure 5.6). There were no other significant predictors of rates of parasitism (Table 5.2).

5.4 Discussion

Despite concerns about toxicity levels for invertebrates living in *Phragmites australis* stems in constructed reedbeds, no effects were seen of reedbed type on the species richness or diversity of the stem-dwelling community. This suggests that, despite their small size, young age and potential contamination with pollutants, constructed reedbeds, which are designed and managed for wastewater treatment, can have additional benefits by providing habitat for stem-dwelling invertebrate communities similar to those seen in equivalent natural habitat, and are therefore of conservation interest in their own right. The species studied in this chapter are habitat specialists and as such may be especially vulnerable to habitat fragmentation. Their presence at constructed reedbed sites throughout the UK suggests these sites might provide valuable stepping stones between areas of natural habitat.

5.4.1 Species composition

Species composition did not differ markedly between constructed and natural sites. The majority of species recorded in this study were found at both natural and constructed sites. *Lipara rufitarsis* was recorded at all sites apart from one constructed site from which no stem-dwelling invertebrates were hatched. *Lipara lucens* was recorded at five natural sites and two constructed sites. The rare members of this genus, *L. similis* and *L. pullitarsis*, were not recorded at any of the sites in this study, although this is not surprising because they are mainly recorded from a small number of sites in East Anglia (NBN, 2011b).

The presence of the two dominant gall-forming invertebrates found in this study (*L. ruffitarsis* and *L. lucens*) at constructed sites (and natural sites) is of significance because, even though they are not of conservation concern in their own right, they are known to provide a discrete microhabitat for a range of other

species (Nartshuk, 2006). They are also known to be prey items for several bird (Kiviat, 2013) and mammal (Reader, unpublished data) species.

Most parasitoids and predators of *Lipara* spp. were recorded at both constructed and natural sites. One exception was *C. apicalis*, which was only recorded at natural sites. Very few studies have recorded *C. apicalis*. It is known to use *Lipara* galls in *Phragmites* stands, but its habitat preferences are unknown (Grochowska, 2006). The typical species-rich assemblages with which the presence of predatory species like *C. apicalis* would be associated are thought to take over 10 years to develop and colonise (Bogusch et al., 2016). Although all sites in this study were 10 years or older, it is notable that it was at some of the oldest sites in this study that *C. apicalis* was recorded.

Species richness of stem-dwelling invertebrates was different in the two years of sampling. Flush-crash cycles are known from other stem-dwelling invertebrates of *Phragmites* such as *Archanara geminipuncta*, where local population levels drop very low every 3 or 4 years (Tscharntke, 1992), but this has not been observed for the species that were recorded in this study. Alternatively, it may be that the difference in the two years is a geographic effect: more of the sites that were studied in the second year were on the periphery of the range of many of these species in the UK (NBN, 2011a), where densities are perhaps lower.

5.4.2 Abundance of stem-dwelling invertebrates in reedbeds in the UK

Abundance of *Lipara* spp. did not differ between constructed and natural sites. This is in-line with a recent study that showed that *Lipara* species were equally abundant in near natural and post-industrial habitats (e.g. gravel-sandpits, stone quarries, colliery dumps and reclaimed lignite open-cast mines) (Bogusch et al., 2016). This study along with others (Athen and Tscharntke, 1999) showed that the main effect on stem-dwelling invertebrates in *P. australis* was the size and age of the habitat (discussed below).

The abundance of parasitoids was slightly greater at natural sites, although this result should be interpreted with caution as levels of parasitism were often based on only a few individuals. Natural sites were significantly older than constructed

sites (see Chapter 2), and although age was not significant it may be that all natural sites were old enough to record parasitoids and the youngest constructed sites were not. This is in-line with other research that suggests age of a habitat can predict the abundance of parasitoids, as discussed below (Athen and Tscharntke, 1999, Bogusch et al., 2016). It may be that the sample sizes of galls taken from each site, or the range of site ages, in the present study were too limited to detect an effect of age on parasitism.

5.4.3 Age and size

Previous research has shown that the presence of stem-dwelling invertebrates and their parasitoids can be predicted by the age and size of the habitat (Athen and Tscharntke, 1999). Species richness and diversity were predicted by the age of a reedbed but not size. Athen and Tscharntke (1999) suggested that after three years (*Lipara spp.*) and six years (*G. inclusa*) there was a 50 % probability of finding these species but parasitoids would take longer to colonise. Other research has shown that after 10 years the species-rich assemblages associated with parasitoids develop after this time (Bogusch et al., 2016). The present study suggests that it might take longer than 10 years for species-rich assemblages to develop, as often only the main gall-forming species, and not their parasitoids, were recorded at the youngest sites in this study, and all were 10 years or older.

Research suggests that habitat areas as small as 120 m^2 are still able to support the minimum populations for local persistence of *G. inclusa* (Tscharntke, 1992). The smallest site in this study was 285 m^2, which suggests all sites were capable of hosting *G. inclusa*. Although it was only found at 11 of 24 sites in this study, its presence is harder to detect than other stem-dwelling invertebrates, because its "rice-grain" galls are a less conspicuous form of stem damage. Tscharntke (1992) also suggested that *L. lucens* and *L. ruffitarsis* species are able to maintain populations in small fragments of *Phragmites* due to their preference for thin reed. However the study by Tscharntke (1992) does not take account of dispersal ability. Research suggests that monophagous herbivore species and their parasitoids (like those studied in this chapter) are at a disadvantage when colonising fragmented habitats (Tscharntke et al., 2002), like reedbeds in the UK, due to a lack of connectivity across the landscape. So despite all the sites in

this study theoretically being large enough to support the species, it may be that isolation has limited colonisation of some of them. Although constructed sites contribute substantially to the total number of reedbeds in the UK, they are often located in a matrix of industrial, urban and agricultural landscapes, where natural reedbeds are not abundant (BRIG, 2008, Cooper, 2007).

5.4.4 Habitat associations

L. lucens and *L. rufitarsis* prefer stems with diameters less than 4.5 mm (Tscharntke, 1992). Similar results were observed in the present study, with higher average diameter stems in a quadrat being associated with lower levels of *Lipara* damage. This relationship was mainly observed in 2014, the year when most study sites were in the centre of the distribution of *Lipara* spp. whereas in the 2015 most sites were on the periphery of the current known range of *Lipara* spp. (NBN, 2011a).

There were no associations between habitat characteristics and species richness and diversity. This may be due to opposing habitat preferences of the different species identified. For example, although both *Lipara* species and *G. inclusa* prefer thin stems (< 4.5 mm), the latter species has a particular preference for the side shoots generated by moth damage, and moths typically prefer thicker stems (Tscharntke, 1992).

5.5 Conclusions

This chapter has highlighted some differences between constructed and natural reedbeds. Abundance of parasitoids of *Lipara* were lower at constructed sites. Regardless of these differences, species richness and diversity were the same at constructed and natural sites. This means that despite their contamination, young age and small size, constructed sites have the ability to host the same diversity as natural ones. They also have the potential to provide stepping stones between natural sites throughout the UK, which may reduce the detrimental effects of fragmentation on these vulnerable species.

Chapter 6: Conservation value of constructed reedbeds

6.1 Introduction

Direct observations of total biodiversity and abundance changes are difficult to obtain, and it is therefore inevitable that generalizations will be made from indicator taxa (Thomas, 2005). Insect taxa are often used as indicators of changes in biodiversity, and their responses are used to understand the ecological impact of climate change and habitat loss and fragmentation (Thomas, 2005). However, the fact that insect species often have localised populations and specialised niches is problematic when making global and even continental extrapolations of trends (Thomas, 2005). Additionally indicator taxa need to be surveyed regularly to establish trends over time (Gregory et al., 2005). Potential groups to serve as environmental indicators for wetlands in the UK include birds, small mammals, bats, dragonflies, butterflies, amphibians and aquatic macroinvertebrates (Everard, 2008). Many of these require skilled surveyors for reliable identification, and different groups are likely to respond differently to environmental changes. Even taxa with strong ecological links have been shown to respond differently to environmental changes. For example insectivorous mammals like shrews and bats did not respond in the same way as their prey to agricultural intensification, despite being closely linked (Pocock and Jennings, 2008). This means that without adequate monitoring programmes of a wide range of indicator taxa, across broad geographic areas, understanding the interacting effects of climate change, habitat loss and fragmentation to inform policy will not be possible and current conservation efforts may be ineffective (Mantyka-Pringle et al., 2012).

This thesis has considered the factors that lead to the prediction that communities in constructed reedbeds may be depauperate compared to their natural equivalents, including their small size, young age and contamination. For each of several taxonomic groups, no large effects of these factors were evident, since communities in constructed and natural sites were similar in species richness and diversity. However, although species richness and diversity indices for individual groups of species provide a useful and rapid univariate summary of biodiversity, they necessarily simplify community composition substantially. Multivariate analyses, which take into account species identity, and consider a

range of taxonomic groups, may reveal differences between natural and constructed sites which were previously undetected.

Although other taxa have largely been overlooked, there are some detailed studies of communities of aquatic macroinvertebrates in constructed reedbeds. These studies provide contradictory results about how community composition may be expected to differ between constructed and natural sites due to contamination (Batty et al., 2005, Stephansen et al., 2016). Batty et al. (2005) found mine-water treatment wetlands to contain aquatic macroinvertebrate communities of pollutant tolerant species whereas Stephansen et al. (2016) found that aquatic macroinvertebrate communities in storm water detention ponds did not differ from those in natural ponds. However, little is known about whether the findings of these studies will be reflected in other groups of species, such as those surveyed in the present study.

In this chapter, I summarise the results of the previous chapters, and contemplate the conservation value of constructed reedbeds. Specifically, I analyse the effect of site type, and those aspects of a constructed reedbed expected to restrict diversity (size and age), on community composition using data from all the taxa surveyed in this project. I attempt to evaluate which of these taxa might serve as good indicators for overall diversity, and in identify where future research efforts might be concentrated in order to understand better how to maximise the conservation value of constructed reedbeds.

6.2 Methods

6.2.1 Field studies

Twelve matched pairs of study sites were selected, as described in Chapter 2. All results from surveys of small mammals, caterpillars, adult moths and stem-dwelling invertebrates are combined; see Chapters 3 – 5 for descriptions of each survey, and Chapter 2 for site access dates.

6.2.2 Statistical analysis

All analyses were conducted in R version 3.3.2. A series of three Hill numbers (N_0 = species richness, N_1 = the exponential of the Shannon index, N_2 = the inverse of the Simpson's index) were calculated for all species collected in the four surveys of each site.

The effects of various explanatory variables on total species richness (N_0) and diversity (N_1 and N_2) were explored using general linear mixed effect models (GLMMs). Errors were assumed to be Gaussian. Reedbed type (constructed or natural), size (area of reedbed over whole site), age (approximate age of specific area surveyed, not the whole site) and habitat characteristics (PC1, see Chapter 2) were all fitted as fixed effects. The random effect was pair (because pairs of sites were deliberately matched – see Chapter 2).

In both models, all two-way interactions were considered, but high-order interactions were ignored. The most parsimonious models were selected using a partial forward step-wise consideration of terms (limited sample sizes restricted the fitting of a saturated model, and hence backwards step-wise model selection was not possible). First, a model with all main effects was fitted, and then each two-way interaction was added separately and the effect of the addition on the deviance explained by the model was tested using an F test. Significant interactions were retained, and non-significant interactions were removed. Main effects were then tested by backwards deletion. To find the optimal structure of the random components, the AIC scores of a model with the random effect was compared to models with no random effects. The AIC was only slightly improved by removing the random effects, so to be conservative the random effects were left in.

Spearman's rank correlation analyses were performed to test the correlation of species richness and total diversity (N_1 and N_2) against all three taxa: moth and caterpillar richness and diversity (N_1 and N_2), mammal richness and diversity (N_1 and N_2) and stem-dwelling invertebrate richness and diversity (N_1 and N_2). Spearman correlation analyses were also performed to test the correlation of the three taxa against each other.

All previous analyses of the communities surveyed in this thesis have considered species richness and diversity. Further analyses were conducted to look for any effect of site type (constructed or natural) on community composition. A hierarchical cluster analysis was performed on all species abundance data from each site, using Euclidean distances. Principal component analysis (PCA) was performed on all those species that occurred at three sites or more. PCA was based on a Euclidean distance matrix with chord transformation. The first analysis used variance and so only consider abundant species with high variances (e.g. common species). Although rare species in this study may not be truly rare due to low sampling effort (e.g. two nights sampling for moths), a second PCA was performed whereby all species were given equal weighting. A multi-variate analysis of variance (MANOVA) was performed to test the effect of reedbed type on all species that occurred at eight or more sites (scarce species were ignored due to limited degrees of freedom).

6.3 Results

A total of 205 species were observed across all 24 sites. The majority of these were Lepidoptera (190), with eight species of stem-dwelling invertebrates and seven of the nine British small mammal species also recorded. Total species richness varied from 13 at Carr Vale, a natural site, to 48 at Slimbridge Treatment (Figure 6.1). Diversity (N_1) varied from 5.91 at Shirebrook a constructed site to 25.21 at Titchmarsh, a natural site and diversity (N_2) varied from 3.64 at Rugeley, a constructed site to 18.18 at Amberswood, a natural site (Figure 6.1).

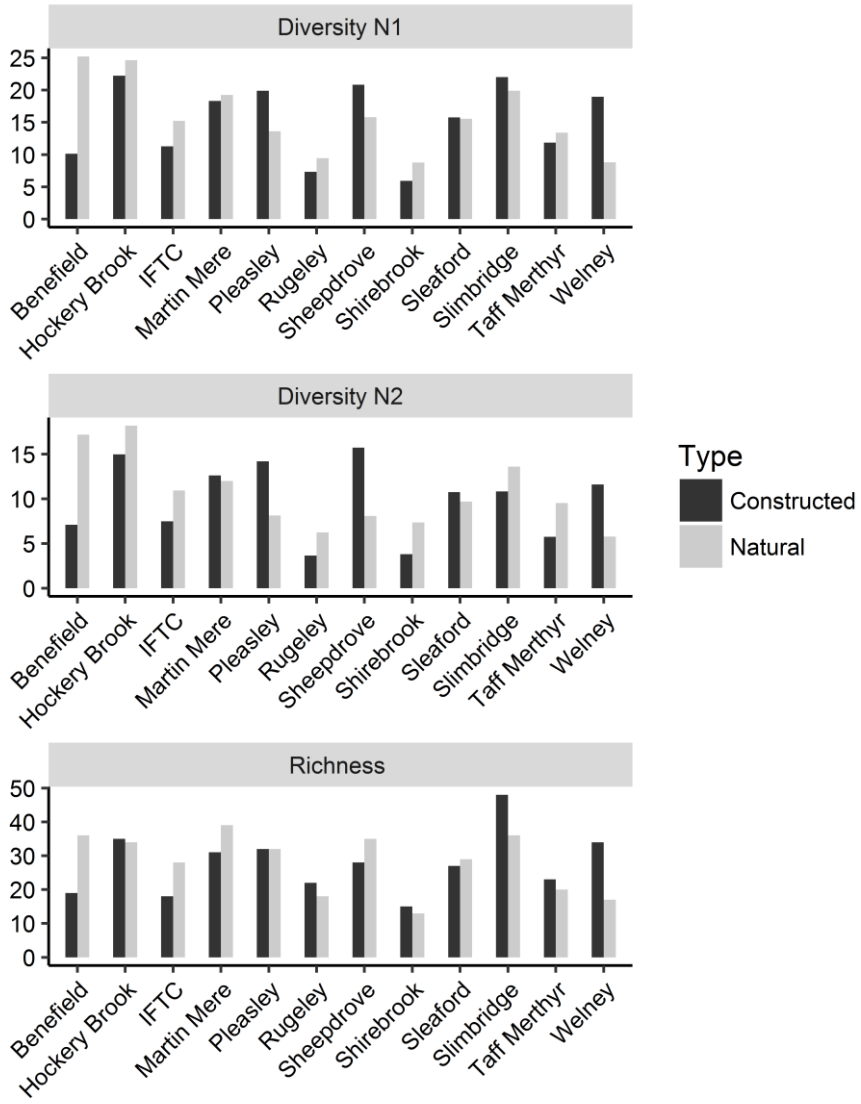


Figure 6.1: Species richness (N_0) and diversity (N_1 and N_2) at 12 constructed and 12 natural sites, construction site named for pairs (see Table 6.1 for pairs).

There were no significant predictors, including site type, of either species richness (N_0) or diversity (N_1 and N_2) (Table 6.1, Figure 6.2).

Table 6.1: Results of generalized linear mixed model testing various variables describing site and habitat characteristics on total species richness (N_0) and diversity (N_1 and N_2). All two-way interactions were considered those not shown were not significant.

Dependent variable	Independent variable	χ^2	df	p value
Diversity N_1	Type	1.4702	1	0.2253
	Size	0.0627	1	0.8022
	Age	1.3612	1	0.2433
	Habitat variable	0.9668	1	0.3255
Diversity N_2	Type	2.6380	1	0.1043
	Size	0.0212	1	0.8842
	Age	1.8486	1	1.8486
	Habitat variable	1.0352	1	0.3089
Richness N_0	Type	0.2314	1	0.6305
	Size	2.2549	1	0.1332
	Age	0.0215	1	0.8834
	Habitat variable	0.6934	1	0.4050

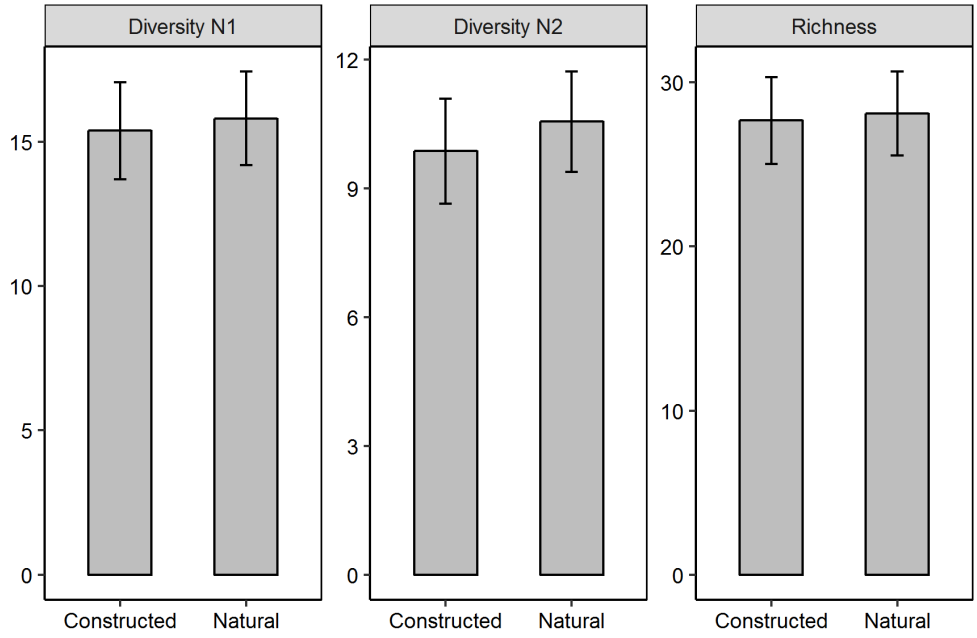


Figure 6.2: The mean diversity (N_1 and N_2) and species richness (N_0) of 12 constructed and 12 natural sites (+/- SEM). GLMM (Table 6.1) indicates no significant difference of species richness ($\chi^2 = 0.2314$, df = 1, p = 0.6305) and diversity (N_1, $\chi^2 = 1.4702$, df = 1, p = 0.2253, N_2, $\chi^2 = 2.6380$, df = 1, p = 0.1043) at constructed and natural sites.

In general, variation in the richness or diversity of one of the surveyed taxa did not predict variation in the richness or diversity of the others (Table 6.2, Table 6.3, Table 6.4). The only significant correlations in diversity across sites among the three taxa surveyed (moths, mammals and stem-dwelling invertebrate) was a weak negative correlation in the exponential of the Shannon diversity index (N_1) between moths and mammals (Table 6.3, Figure 6.3). Unsurprisingly, since 92.7 % of the species found were moths, there was a strong positive correlation between moth richness or diversity and total richness or diversity (Figure 6.4, Figure 6.5, Figure 6.6).

Table 6.2: Spearman's rank correlation analysis of the relationship among sites in species richness (N_0) among the different taxa surveyed.

	Moth species richness	**Mammal species richness**	**Stem-dwelling invertebrate species richness**
Total species richness	-0.23 **p <0.001**	0.94 0.2774	0.02 0.9083
Moth species richness		-0.39 0.0566	-0.22 0.3067
Mammal species richness			0.20 0.3474

Table 6.3: Spearman's rank correlation analysis of the relationship among sites in the exponential of the Shannon diversity index (N_1) among the different taxa surveyed.

	Moth diversity	**Mammal diversity**	**Stem-dwelling invertebrate diversity**
Total diversity	0.87 **p <0.001**	-0.30 p = 0.1488	-0.15 p = 0.4799
Moth diversity		-0.41 **p = 0.0489**	-0.25 p = 0.2310
Mammal diversity			0.03 p = 0.9037

Table 6.4: Spearman's rank correlation analysis of the relationship among sites in the inverse of the Simpson's diversity index (N_2) among the different taxa surveyed.

	Moth diversity	**Mammal diversity**	**Stem-dwelling invertebrate diversity**
Total diversity	0.71 **p <0.001**	-0.15 p = 0.4733	-0.18 p = 0.3893
Moth diversity		-0.32 p = 0.1323	-0.28 p = 0.1873
Mammal diversity			0.00 p = 0.9885

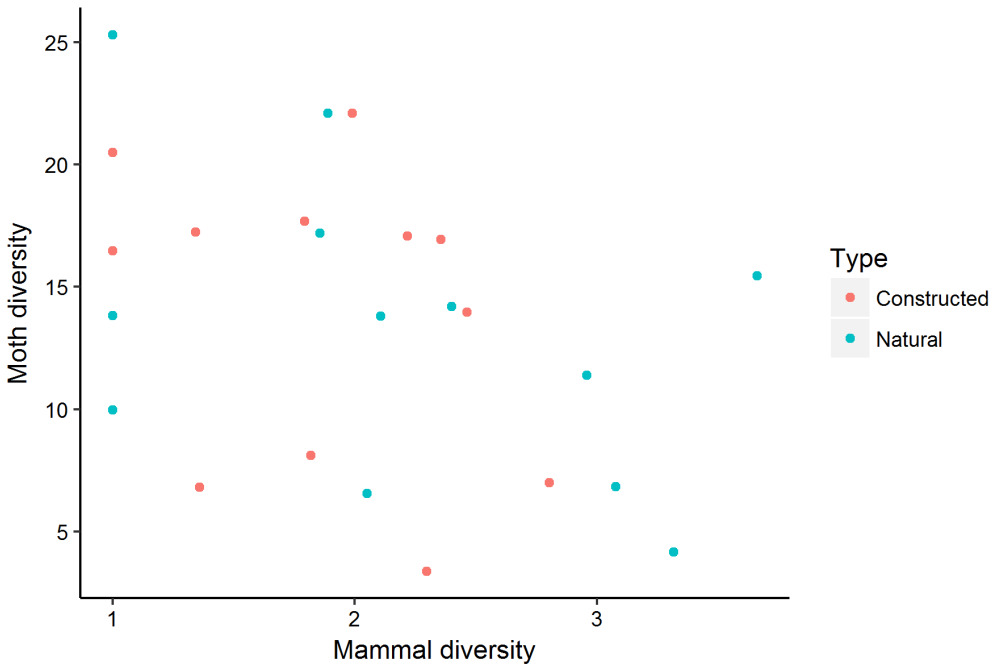


Figure 6.3: Correlation between moth diversity (N_1) and mammal diversity (N_1), separated by type of reedbed – constructed or natural.

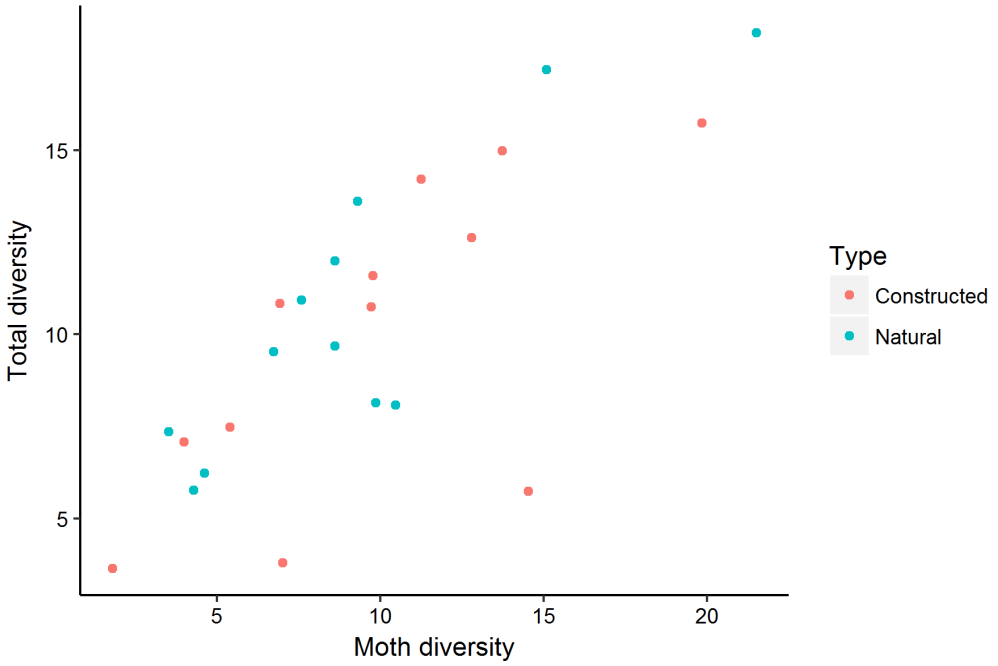


Figure 6.4: Correlation between moth diversity (N_2) and total diversity (N_2), separated by the type of reedbed – constructed or natural.

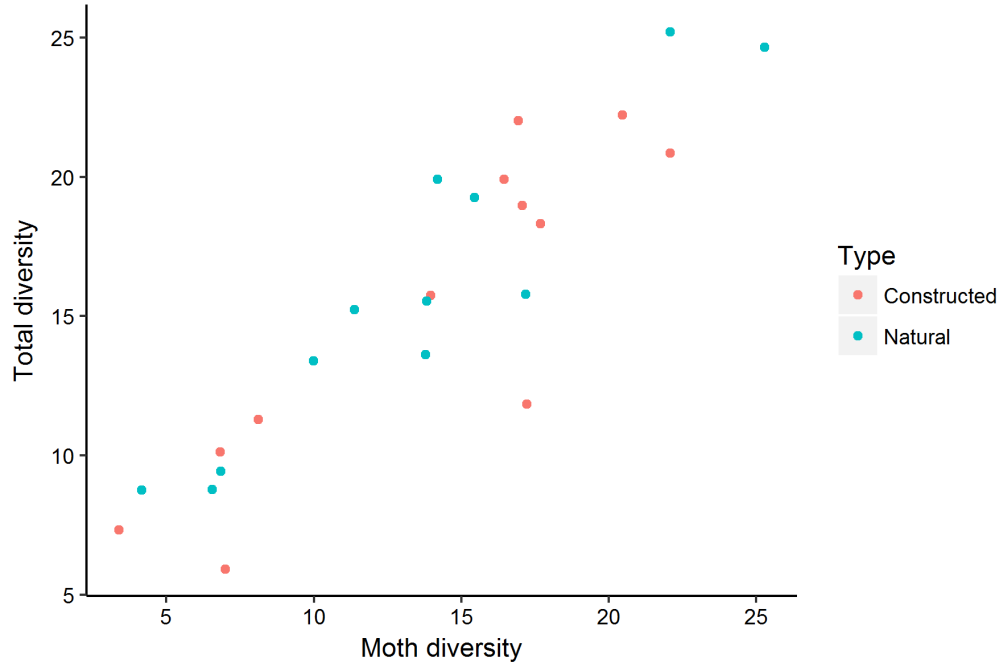


Figure 6.5: Correlation between moth diversity (N_1) and total diversity (N_1), separated by the type of reedbed – constructed or natural.

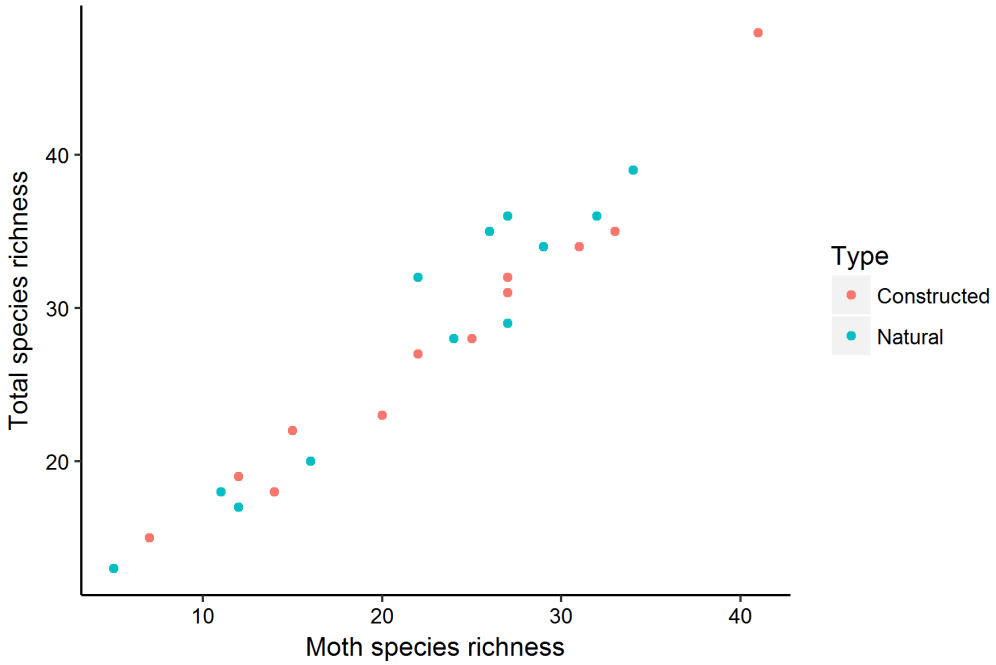


Figure 6.6: Correlation between moth species richness (N_0) and total species richness (N_0), separated by the type of reedbed – constructed or natural.

PCA was used to analyse community composition including 205 species at 24 sites. For the analyses of the common species, PC1 explained 17.1 % of the variance and PC2 explained 14.5 % of the variance. When plotted in multivariate space, sites do not obviously group based on site type (Figure 6.7). When all species were weighted equally, PC1 explained 12.8 % of the variation and PC2 10.0 % of the variation. Again, sites were not obviously grouped by site type in multivariate space (Figure 6.8).

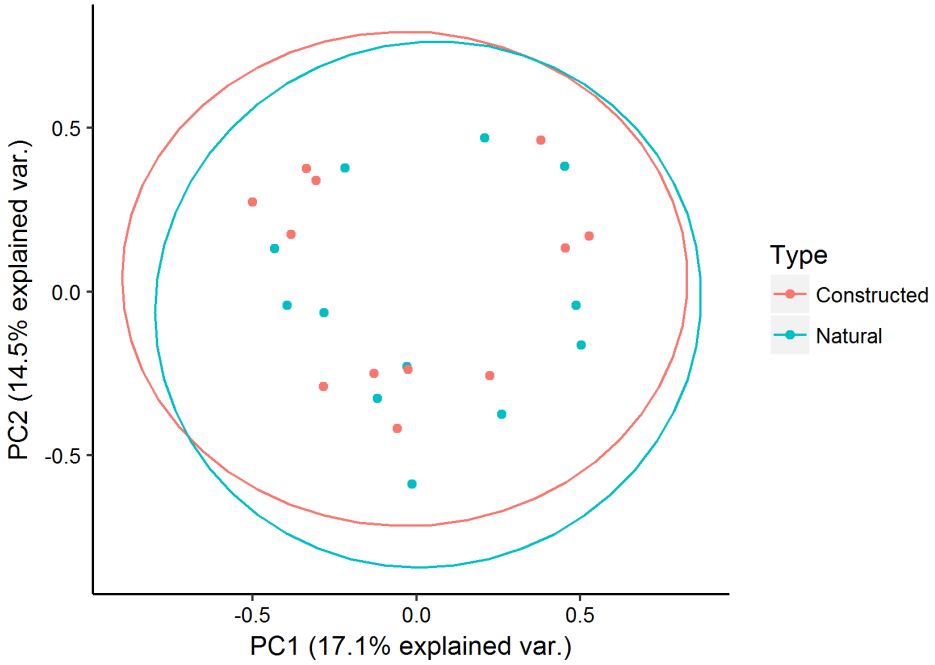


Figure 6.7: The first and second principle component scores for all 24 sites, separated by type (constructed or natural) of reedbed. Ellipse represent 95 % confidence intervals for the distributions of the two types of site.

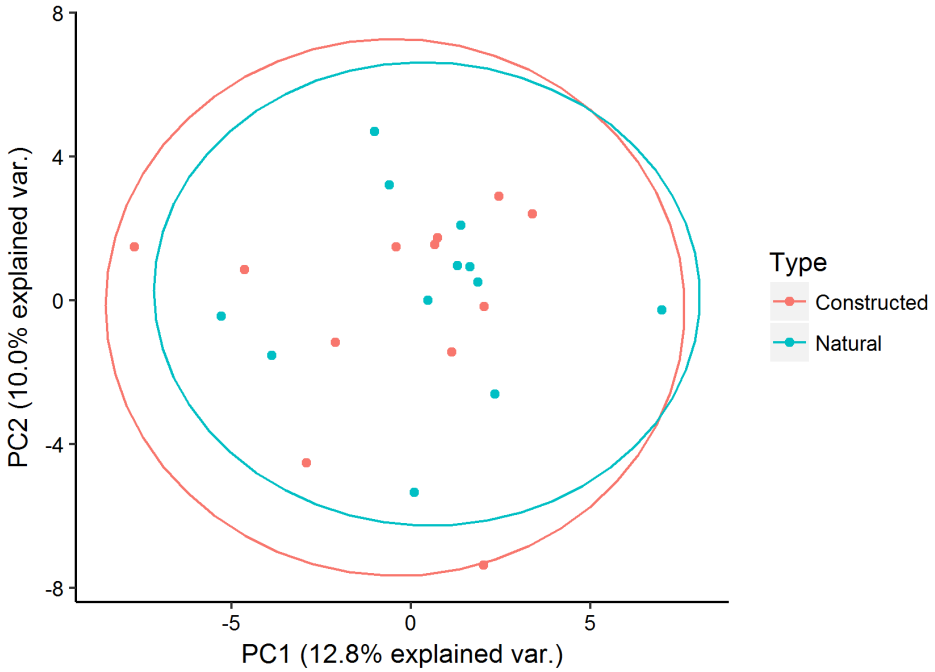


Figure 6.8: The first and second principle component scores for all 24 sites, separated by type (constructed or natural) of reedbed. Ellipse represent 95 % confidence intervals for the distributions of the two types of site.

Hierarchical cluster analysis showed no obvious groupings of sites (Figure 6.9). Natural and constructed sites were dispersed throughout the dendrogram, as were sites treating different wastewater types. There were no obvious patterns reflecting reedbed size or age or geographic regions (see Chapter 2, Figure 2.1 for locations of sites). Paired sites did not typically cluster together, with one exception of pair seven, Parc Penallta and Taff Merthyr.

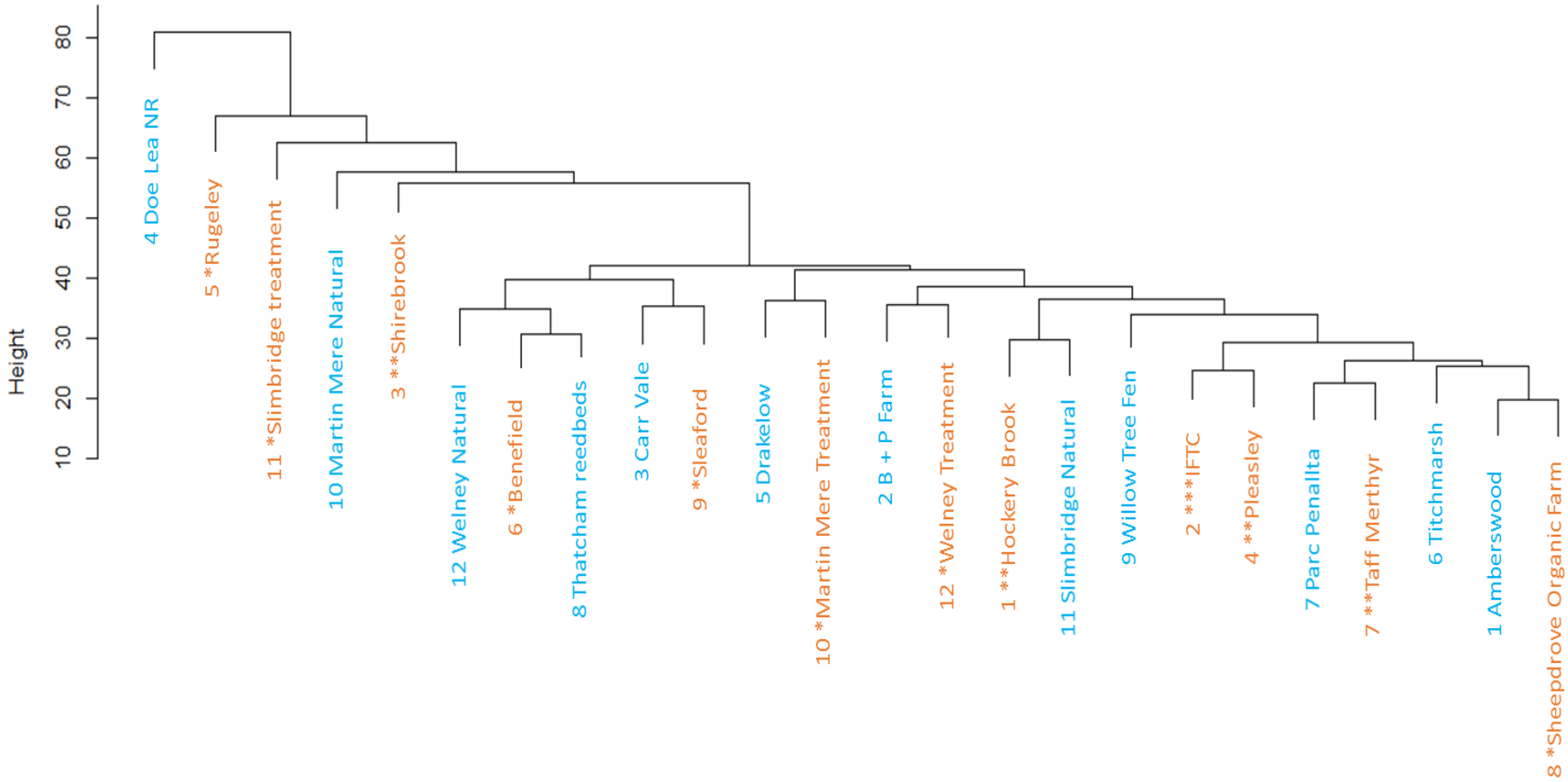


Figure 6.9: Hierarchical cluster dendrogram: natural sites are shown in blue, constructed sites are in orange, * = sewage treatment site, ** = mine water treatment site, ***airport run-off. Pairs are indicated by numbers and their geographical locations are shown in Chapter 2, Figure 2.1.

There was also no significant effect of site type (constructed or natural) on the community composition, considering species that were found at eight or more sites (MANOVA: Pillai = 0.96221, F = 1.1573, p = 0.6373).

6.4 Discussion

Constructed reedbeds do not differ in their diversity and species richness compared to the natural reedbeds surveyed in this study. Diversity and species richness were not predicted by the type of reedbed, constructed or natural, or the age and size of the reedbed nor the habitat type. This suggests that there is not a strong effect of any aspects of a constructed reedbed (that are different from that of a natural reedbed) on diversity and richness. This is highlighted when comparing the paired sites (Figure 6.1): although there was considerable variation among sites, most pairs have very similar biodiversity and so it may suggest local conditions (e.g. weather conditions for moths) are more important in determining biodiversity. Although community composition is not matched at paired sites (see below).

A previous study of one reedbed in the UK highlighted the ability of constructed reedbeds to have biodiversity benefits (Worrall et al., 1997). This research suggested that constructed reedbeds could be designed and managed to benefit wildlife without compromising water quality (Worrall et al., 1997). Especially if design and management is driven from a more ecological perspective rather than purely an engineering one. Although most constructed reedbeds are not actively managed (personal observation), the present study has highlighted the potential conservation benefit of constructed reedbeds and their ability to host similar biodiversity to that of their natural equivalents.

6.4.1 Age and size

In spite of significant differences in age and size between natural and constructed reedbeds, and concerns over pollution, there was no difference in richness and diversity. Furthermore, there was no identifiable effect of age or size themselves, despite the general expectation that large habitat patches should maintain higher biodiversity and that constructed reedbeds were significantly smaller their

natural equivalents. This is not a great surprise as age and size did not produce any strong effects on species richness or diversity of any of the groups of species that were sampled when they were considered individually (Chapters 3-5).

6.4.2 Community composition

The community composition analyses showed no differences between natural and constructed sites. Some previous research has indicated that constructed wetlands host communities that, although diverse, contain pollutant tolerant species (Batty et al., 2005) different from those that would be expected in a natural system. Batty et al. (2005) studied macroinvertebrates which may be more sensitive to the effects of water quality, and hence the contamination associated with constructed reedbeds, compared to the taxon surveyed in this study. Data from the present study suggest that small mammals, moths and stem-dwelling invertebrates are not strongly affected by contamination, including heavy metals. This is emphasised by the fact that the mine-water treatment sites sampled were not obviously separated from other constructed sites in the community composition analyses presented here. Although species richness and diversity were similar at the paired sites the community composition analysis suggests that community composition was not the same at paired sites, with one exception: Parc Penallta and Taff Merthyr.

6.4.3 Indicator taxa

Previous research has suggested that moths (and Lepidoptera generally) are good indicator species for overall diversity and trends associated with climate change and any effects of habitat loss and fragmentation (Thomas, 2005). This study found that moth diversity had no correlation with stem-dwelling invertebrate diversity and a weak *negative* correlation with small mammal diversity, suggesting that moths are not good indicators for other taxonomic groups in this kind of habitat. This is not a surprise as the analysis in previous chapters suggested that the diversity of each taxon is influenced by different variables. Moth diversity was mainly predicted by weather (Chapter 4) and not by habitat variables. Given how sensitive to temperature and wind light trapping is known to be (Yela and Holyoak, 1997), it may be that more long term studies, to

eliminate the effects of weather for moths, would be required to notice any trends related to other variables.

Identifying suitable indicator taxa is often difficult as different taxa respond differently to environmental changes (Everard, 2008). Insectivorous mammals such as shrews and bats have been shown to respond differently to agricultural intensification compared to their prey items (including moths) despite being closely linked ecologically (Pocock and Jennings, 2008). This suggests that it might be unlikely for mammal diversity to be predictive of moth and stem-dwelling invertebrate diversity, as seen in this study. Even birds, which have long been used as indicator taxa because they are conspicuous, mobile and supposedly respond rapidly to change, do not respond to all environmental changes equally in all habitats. For example improving water quality and invertebrate assemblages in the UK uplands is not mirrored by bird populations, which are still rapidly declining (Everard, 2008). This suggests that monitoring a single taxon is unlikely sufficient to understand a whole community's response to environmental variation, and that using one indicator taxon for comparisons of constructed and natural reedbeds may not be the best option. Interestingly, wherever water quality is concerned, there is a tendency to use aquatic macroinvertebrates as an indicator taxon, to the extent that they are used as indicators of ecological quality in the Water Framework Directive. Especially in wetlands with emergent vegetation, such as reedbeds, the present study suggests that such an approach may not capture all variation in community composition which may be of conservation significance.

6.5 Conclusions: the conservation value of constructed reedbeds

Very few studies have directly compared the communities in constructed and natural reedbeds, and to my knowledge this is the first study to do so in the UK on such a large scale. Additionally, very few studies have considered multiple taxa, or organisms other than plants, birds or aquatic macroinvertebrates. Thus, despite reedbeds being well known for supporting rich communities of (for example) mammals and terrestrial invertebrates, this study is the first to compare

the diversity and composition of multiple taxa in a large number of constructed and natural reedbeds across a large geographic area.

As discussed throughout the thesis, the expectation was that constructed reedbeds have impoverished fauna compared to natural reedbeds. However, this is not the case for the 12 constructed sites surveyed in this study: for all of the species groups surveyed, they are just as diverse as neighbouring natural sites. Although age affected the species richness and diversity of stem-dwelling invertebrates, this was not associated with constructed reedbeds per se, but with both reedbed types, and suggests that over time constructed reedbeds, although young now, can accumulate the same diversity as older natural reedbeds. All other significant effects on species richness and diversity could not be attributed to a reedbed being "constructed". Moth diversity was mainly determined by weather, as expected, and communities of both moths and mammals in constructed and natural reedbeds were diverse by UK standards.

Although a small number of the results present in this thesis suggest weak effects of site type on biodiversity (e.g. older constructed sites have reduced moth diversity, abundance and richness), these effects were never large in effect or consistent across taxa. The expectation if constructed reedbeds had a detrimental effect on wildlife or were unable to support any wildlife at all, would be for strong negative effects of constructed reedbeds to be seen across all taxa. Although it is very difficult to prove that any aspect of a constructed reedbed is not having any effect on the biodiversity it supports, this is the first study in the UK to suggest that constructed reedbeds are able to host similar diversity to that of their natural equivalents.

There are an estimated 1,000 constructed reedbeds in the UK (Cooper, 2007), although this number is expected to be considerably higher (personal observation). While most of these constructed reedbeds are small < 0.25 ha (Hawke and Jose, 1996) their ability to host similar diversity to that of their natural equivalents means the connectivity they provide could be substantial. There are only 50 of 900 natural sites over 20 ha (BRIG, 2008) and it is expected that small fragmented areas of reedbed provide critical connectivity to sustain

wetland biodiversity and maintain metapopulations, especially of wetland specialists (Gibbs, 1993).

All taxa considered in this thesis are understudied and underreported in the literature in the UK, even in natural reedbeds. The presence of many of the species normally found in natural reedbeds in these constructed reedbeds is promising and suggests that they already provide a conservation benefit. Other, anecdotal evidence from this study also suggests that constructed reedbeds provide a conservation benefit. Although birds were not specifically targeted, reedbed specialist birds such as reed bunting (*Emberiza schoeniclus*), reed warbler (*Acrocephalus scirpaceus*) and sedge warbler (*Acrocephalus schoenobaenus*) were sighted at most reedbeds, constructed and natural, large and small, during surveys of other taxa. The rarer wetland specialist water rail (*Rallus aquaticus*) was also heard regularly at both constructed and natural sites, and seen once at Taff Merthyr, a constructed site. Although never seen, multiple site managers also reported the presence of water vole (*Arvicola amphibius*) at both constructed and natural sites.

The presence of a species at a site indicates that a habitat is suitable, but does not provide any indication of habitat quality (Hazell et al., 2004). However, the scarcity and fragmentation of natural reedbeds in the UK makes species presence all the more important and means that constructed reedbeds, regardless of habitat quality, can provide a great conservation benefit to reedbeds and the species that utilise them nationally. This could be of great significance, as many of the reedbed species, and especially those that are considered reedbed specialists, are of special conservation concern. Although the primary purpose of constructed reedbeds is wastewater treatment, further research to understand their biodiversity benefits, especially of species of conservation concern, would enable designers and managers of constructed reedbeds to capitalise on this additional benefit.

It has been shown that management can have impacts on diversity in reedbeds (Valkama et al., 2008). My observations at both natural and constructed sites is that management is not done in a regimented way. This is especially the case in constructed sites, where regular management is often overlooked and results in

less frequent but more drastic management interventions such as dredging. Reliable data on management type and frequency are hard to gather, and were not considered in this study. If accurate details of past and current management practices for sites could be obtained, these data may provide explanations for variations in the communities among sites that were not explained by factors in this study.

Although not managed, constructed reedbeds are likely to persist across the UK because of their role as a water treatment process. The benefits they provide in terms of biodiversity may become crucial as pressures on small natural reedbeds, in ditches and small ponds are increasing. Those small natural reedbeds in agricultural and urban landscapes are being removed and hence constructed reedbeds in similar landscapes may provide the connectivity these natural reedbed fragments once provided.

This research may also provide support for conservation organisations to consider protecting small, young areas of habitat, and to consider the importance of habitats that are used for industrial purposes. Currently the main reason constructed reedbeds are used is because of their reduced costs and their *potential* for wildlife benefits. This study now provides evidence that constructed reedbeds do have conservation value and can support wildlife of conservation concern. Although it is difficult to assign a monetary value to biodiversity, this study suggests that constructed reedbeds could now be utilised for wastewater treatment even when conventional cost-benefit analysis is not necessarily strongly in favour of a reedbed, due to the additional biodiversity benefits they could provide. Conservation organisations such as WWT and Wildlife Trust have already installed constructed reedbeds because of their potential wildlife benefit. The additional evidence from this study supports this action, and should now encourage other organisations to consider constructed reedbeds as a wastewater treatment process.

Although the evidence from this study is promising, further work to understand fully the capacity of constructed reedbeds compared to larger, older natural reedbeds is needed. The natural reedbeds studied in this study were those closest

to the constructed reedbeds. However, the oldest natural reedbeds are mostly located in East Anglia or at the coast, far from the industrial/urban matrix within which constructed reedbeds are normally located. This study has shown that constructed reedbeds are similar to those natural reedbeds located in a similar geographical context, despite their young age and small size. Although this is important, an important next step for future work will be to consider how similar these constructed reedbeds are to the large expanses of reedbed that dominate low-lying coastal areas, and whether their key role might be in providing connectivity across otherwise hostile landscapes. Current work is ongoing (by the CWA and myself) to map all the constructed wetlands in the UK. Further research along with this information would enable a better understanding of the biodiversity benefits of constructed reedbeds in the UK on a landscape scale.

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Appendix 1

Species	Number of sites
Mammals	
Bank vole *Myodes glareolus*	18
Common shrew *Sorex araneus*	8
Harvest mouse *Micromus minutus*	12
House mouse *Mus domesticus*	1
Pygmy shrew *Sorex minutus*	1
Water shrew *Neomys fodiens*	1
Wood mouse *Apodemus sylvaticus*	22
Moths	
Acleris kochiella	3
Acrobasis advenella	1
Agapeta hamana	8
Agonopterix heracliana	1
Agonopterix kaekeritizana	1
Agriphila selasella	1
Agriphila straminella	4
Agriphila tristella	1
Barred straw *Gandaritis pyraliata*	5
Beautiful hook tip *Laspeyria flexula*	1
Bird cherry Ermine *Yponomeuta evonymella*	1
Black arches *Lymantria monacha*	2
Black neck *Lygephila pastinum*	2
Bordered white *Bupalus piniaria*	1
Bright-line Brown-eye *Lacanobia oleracea*	13
Broad bordered yellow underwing *Noctua fimbriata*	1
Broom moth *Chesias rufata*	1
Brown china mark *Elophila nymphaeata*	4
Brown line bright eye *Mythimna conigera*	2
Brown spot pinion *Agrochola litura*	1
Bryotrphha sp.	1
Buff Arches *Habrosyne pyritoides*	2
Buff Ermine *Spilosoma lutea*	6
Buff tip *Phalera bucephala*	5
Bulrush wainscot *Nonagria typhae*	4
Burnished brass *Diachrysia chrysitis*	3
Calamotropha paludella	3
Canary shoulder thorn *Ennomos alniaria*	1
Catoptria falsella	1
Celypha lacunana	1
Celypha striana	2
Chilo phragmitella	10

Species	Number of sites
Chrysoteucina culmella	14
Clay *Mythimna ferrago*	1
Cloaked minor *Mesoligia furuncula*	1
Clouded border *Lomaspilis marginata*	4
Clouded silver *Lomographa emerata*	1
Cnephasia sp	1
Cochylis atricapilana	1
Common carpet *Epirrhoe alternata*	1
Common footman *Eilema lurideola*	14
Common Rustic *Mespamea secalis*	25
Common wainscot *Mythimna pallens*	6
Common Wave *Cabera exanthemata*	4
Common white wave *Cabera pusaria*	3
Copper Underwing *Amphipyra pyramidea*	6
Coronet *Craniopora ligustri*	1
Coxcomb prominent *Ptilodon capucina*	1
Crambus ericella	1
Crambus peralla	1
Crambus perella	2
Crambus perlella	6
Crambus uligiosellus	1
Cream boarded green pea *Earias clorana*	1
Crescent *Helotropha leucostigma*	7
Cyclamen tortrix *Clepsis spectrana*	6
Dark Arches *Apamea monoglypha*	23
Dark barred twin spot carpet *Xanthorhoe ferrugata*	1
Dark-sword-grass *Agrotis ipsilon*	1
Dingy footman *Eilema griseola*	5
Dingy shears *Apterogenum ypsillon*	1
Donacaula forficella	6
Dot moth *Melanchra persicariae*	1
Dotted clay *Xestia baja*	5
Double dart *Graphiphora augur*	2
Double line *Mythimna turca*	2
Double lobed *Lateroligia ophigramma*	3
Double square spot *Xestia triangulum*	9
Drinker *Euthrix potatoria*	13
Dunbar *Cosmia trapezina*	1
Dusky Sallow *Eremobia ochroleuca*	4
Dusky thorn *Ennomos fuscantaria*	3
Ear moth *Amphipoea oculea*	1

Species	Number of sites
Early thorn *Selenia dentaria*	1
Elephant hawk moth *Deilephila elpenor*	7
Endotheria quadrimaculana	2
Eudonia	2
Eudonia lacustrata	5
Eudonia murana	1
Eudonia pallida	3
Euzophera pinguis	1
Evergestis palliclata	1
Eyed hawk moth *Smerinthus ocellata*	1
Fen wainscot *Arenostola phragmitidis*	9
Flame *Axylia putris*	2
Flame shoulder *Ochropleura plecta*	11
Flounced rustic *Luperina testacea*	10
Galium carpet *Epirrhoe galiata*	1
Garden dart *Euxoa nigricans*	1
Garden tiger *Arctia caja*	7
Gold triangle *Hypsopygia costalis*	1
Green carpet *Colostygia pectinataria*	1
Green pug *Pasiphila rectangulata*	1
Heart and Club *Agrotis clavis*	3
Heart and Dart *Agrotis exclamationis*	13
Hedya salicella	1
Helcystogramma rufescens	3
July highflyer *Hydriomena furcata*	2
Lackey *Malacosoma neustria*	2
Large Emerald *Geometra papilionaria*	1
Large fruit tree tortrix *Archips podana*	4
Large yellow underwing *Noctua pronuba*	35
Latticed heath *Chiasmia clathrata*	1
Lead belle *Scotopteryx mucronata*	1
Least yellow underwing *Noctua interjecta*	3
Lesser Broad bordered yellow underwing *Noctua janthe*	8
Lesser yellow underwing *Noctua comes*	3
Light arches *Apamea lithoxylaea*	3
Light Emerald *Campaea margaritaria*	6
Lime-speck pug *Eupithecia centaureata*	1
Limnaecia phragmitella	6
Lobesia abscisana	3
Lozotoeniodes formosana	1
Magpie *Abraxas grossulariata*	1
Map-winged swift *Korscheltellus fusconebulosa*	1
Marbled minor *Oligia strigilis*	3

Species	Number of sites
Marbled orchard tortrix *Hedya nubiferana*	2
Marbled white spot *Deltote pygarga*	1
Mere wainscot *Photedes fluxa*	1
Miller *Acronicta leporina*	1
Mother of Pearl *Pleuroptya ruralis*	2
Mottled beauty *Alcis repandata*	3
Mottled rustic *Caradrina morpheus*	4
Mouse Moth *Amphipyra ragopoginis*	8
Northern Drab *Orthosia opima*	1
Obscure wainscot *Leucania obsoleta*	11
Old lady *Mormo maura*	1
Orange Swift *Triodia sylvina*	1
Pale prominent *Pterostoma palpina*	1
Pebble prominent *Notodonta ziczac*	4
Peppered moth *Biston betularia*	2
Pinion streaked snout *Schrankia costaestrigalis*	1
Plain golden Y *Autographa jota*	1
Poplar grey *Subacronicta megacephala*	2
Poplar hawk moth *Laothoe populi*	15
Ptycholomoides aeriferanus	1
Red barred tortrix *Ditula angustiorana*	1
Red twin spot carpet *Xanthorhoe spadicearia*	1
Rhomboid tortrix *Acleris rhombana*	1
Riband wave *Idaea aversata*	8
Ringed china mark *Parapoynx stratiotata*	4
Ruby Tiger *Phragmatobia fuliginosa*	8
Rustic *Hoplodrina blanda*	2
Sallow *Cirrhia icteritia*	1
Sallow kitten *Furcula furcula*	3
Scalloped oak *Crocallis elinguaria*	1
Scarce footman *Eilema complana*	1
Schoenobius gigantella	3
Scoparia ambiguealis	1
Scoparia ancipitella	1
Setaceous hebrew character *Xestia c-nigrum*	3
Shaded broad bar *Scotopteryx chenopodiata*	7
Shark *Cucullia umbratica*	1
Shuttle shaped dart *Agrotis puta*	1
Silky wainscot *Chilodes maritima*	11
Silver Y *Autographa gamma*	2
Single dotted wave *Idaea dimidiata*	6
Six-striped rustic *Xestia sexstrigata*	1
Small blood vein *Scopula imitaria*	1

Species	Number of sites
Small china mark *Cataclysta lemnata*	6
Small clouded brindle *Apamea unanimis*	1
Small dotted buff *Photedes minima*	3
Small magpie *Anania hortulata*	2
Small phoenix *Ecliptopera silaceata*	1
Small rivulet *Perizoma alchemillata*	4
Small square spot *Diarsia rubi*	4
Small wainscot *Denticucullus pygmina*	1
Smoky wainscot *Mythimna impura*	34
Southern wainscot *Mythimna straminea*	33
Square spot rustic *Xestia xanthographa*	1
Straw dot *Rivula sericealis*	1
Straw underwing *Thalpophila matura*	3
Swallow prominent *Pheosia tremula*	3
Tawny shears *Hadena perplexa*	1
The Engrailed *Ectropis crepuscularia*	1
Turnip moth *Agrotis segetum*	1
Twin spotted wainscot *Lenisa geminipuncta*	1
Udea olivalis	1
Uncertain *Hoplodrina octogenaria*	15
Uncertain/rustic	5
Vines rustic *Hoplodrina ambigua*	1
Webbs wainscot *Globia sparganii*	2
White Ermine *Spilosoma lubricipeda*	1
White Satin *Leucoma salicis*	1
Yellow shell *Camptogramma bilineata*	1
Yellow tail *Euproctis similis*	1
Stem-dwelling invertebrates	
Cnemopogon apicalis	6
Endromopoda phragmitidis	6
Giraudiella inclusa	11
Lipara rufitarsis	23
Lipara lucens	7
Polemochartus liparae	9
Stenomalina liparae	6
Steneotarsonemus phragmitidis	6

Appendix 2



The University of Nottingham BBSRC Doctoral Training Partnership (DTP) is undertaken in partnership with Rothamsted Research. Its aim is to develop an innovative, integrated, student-led programme of individualised research training in the Biosciences through an effective cohort-based programme. Students are not recruited to an individual project but, through a series of lab rotations, identify a PhD programme that they wish to pursue in one of three BBSRC priority research areas (Food Security; Industrial Biotechnology and Bioenergy; World Class Biosciences)

Research Training Structure

The research training structure is framed by individual development needs analysis and has the following core components:

- Generic and research skills development
- High-level modular training
- Laboratory rotations
- Experiential learning through public engagement and outreach activity

Generic and research skills development comprises the following: induction, introductory training programme, annual 'Spring School' conference-style training event, participation in Sysmic (online systems and mathematical biology course), specialist training days at partner organisations, twilight (after-lab) training which is arranged on an *ad hoc* basis to meet identified needs, and seminar series.

In Year One there is an initial period of intensive training which takes place over the course of the first six months. This comprises a two-week introductory training programme, 24 weeks of laboratory rotations, followed by a week-long 'Spring School' (conference-style

training event for all DTP cohorts) and ending in a science outreach event. Each lab rotation is assessed focussing on a different form of scientific communication (Report, Poster, Presentation) and students receive feedback from supervisors. The PhD project begins in Month 7. Students are also encouraged to choose from a portfolio of relevant Master Courses which they can attend alongside their lab rotation projects.

In Years Two to Four training is mainly concentrated in the week long annual Spring School, although additional training and seminars are delivered where required throughout the year. A three month Professional Internship for PhD Students (PIPS) placement is usually taken in Year Three and, as BBSRC specifies, this is not in an area related to the students' PhD research programme.

The integrated programme of training is mapped to BBSRC priority areas, BBSRC Skills Statement: Scientific and Core Skills for UK Bioscience and the Researcher Development Framework (RDF) to ensure breadth and depth of provision, and to encompass vulnerable skills and strategic competencies. It builds on the best practice principles outlined in the Statement of Expectations for Doctoral Training, RCUK (2013) and the 'Good Practice Elements in Doctoral Training', LERU (2014).

It offers a wide range of skills development opportunities to prepare students for a successful bioscience-related career and supports the exchange of ideas and experiences within and across cohorts. It also generates a strong community of postgraduates and supervisors to stimulate the formation of cross-disciplinary links through cohort development, which prepares students for a career in an increasingly multidisciplinary environment.

Professional Internships for PhD Students
Reflection Form

Name of Organization

Middlemarch Environmental

Details of Placement

Please describe your main activities during the placement (150-200 words)

The placement involved working with senior ecologists to perform protected species surveys up and down the UK. The majority of time was spent performing great crested newt surveys and nocturnal and dawn bat surveys. In addition, I got the opportunity to monitor a large reintroduction program for water voles. I was also involved with a few of the larger, more complex projects at the company. One involved mitigation for a large, multi-species bat roost and another large scale wetland habitat creation in disused quarry lagoons.

Placement Achievements

Please detail all outcomes from the placement, including any publications, presentations given and reports written etc. (150-200 words)

The main output from the day to day surveys are recorded in reports to send to clients. This data is not publishable and is often confidential. Although as a result of my placement I was offered some freelance ecologist work and a job offer for when I complete my PhD.

Skill development

Has this Placement helped you developed any new skills or enhanced your previous skill set? (100-150 words)

I worked alongside knowledgeable ecologists who specialised in several protected species surveys and mitigation. I have developed my fieldwork skills generally but the placement also enabled me to develop new specific skills related to the protected species work associated with ecological consultancy. I had previous experience in dealing with site managers through my PhD but this placement enhanced my skills at dealing with clients who are often not interested in ecology or wildlife.

Future Work

Has this Placement influenced your future career aspirations? If so, in what way? (150-200 words)

I have been offered and accepted a job, with my placement company as a Senior Ecological Consultant with intentions to use the skills and contacts gained from my PhD in the specific area of wetland habitat creation.