DEVELOPING AN EVIDENCE-BASED METHOD TO ASSESS THE IMPACT OF LOCAL DEVELOPMENT ON MOORLAND FRINGE BIRD POPULATIONS

CHRISTOPHER JAMES HARRISON

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Manchester Metropolitan University

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ABSTRACT

The local unitary authorities of Calderdale, Kirklees and Bradford in West Yorkshire have joint jurisdiction over the South Pennine Moors Special Protection Area (SPMSPA). This is an upland protected area in the North of England. The SPMSPA provides feeding and breeding habitat for an assemblage of bird species of international conservation concern. Knowledge of the habitat associations of these species within the fringe of the SPA is lacking. Thirteen species form the bird assemblage that has been identified in collaboration with the project partners as in most need of ecological evidence within the moorland fringe landscape. Within this PhD, the ecology of these species was investigated in the context of the immediate 1 km fringe outside of the SPMSPA. The habitat composition of this fringe was found to be a heterogenous mosaic, predominantly characterised by smaller fields dominated by species-poor agricultural habitats. Curlew *Numenius arquata*, Lapwing *Vanellus vanellus*, Snipe *Gallinago gallinago*, Wheatear *Oeanthe oeanthe* and Golden Plover *Pluvialis apricaria* were found to be associated with fields comprising tussocks, wet flush and evidence of intensive grazing. Species richness was found to be greatest in habitats not typical of moorland or farmland. Bird diversity and species richness were lowest within 100 m of Small Wind Turbines (SWTs), with Magpie *Pica pica* and Starling *Sturnus vulgaris* negatively associated with proximity to SWTs. Landsat 8 imagery were found to be a good predictor of the distribution of habitat suitability for five moorland fringe bird species, especially when used to supplement empirical data. Building density was an important predictor for the majority of these species. The lack of unimproved grassland and particularly high land cover of improved and semi-improved agricultural land indicate that the SPMSPA fringe landscape is suboptimal for the conservation of moorland fringe bird diversity. The results of this research can be used as ecological evidence to assist future planning decisions and the conservation of habitats within the SPA fringe for birds of conservation concern.

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CHAPTER 1: BIRDS IN A DEVELOPING MOORLAND FRINGE LANDSCAPE

1.1. Global drivers of biodiversity decline

The world is undergoing a period of dramatic decline in biodiversity as a consequence of human activity (Johnson et al., 2017). The anthropogenic causes of this biodiversity loss are diverse in nature and geographically widespread, with implications for ecosystems throughout the world (Steffen et al., 2015). The most publicised and widely recognised driver of biodiversity loss is climate change as the result of fossil fuel combustion (González-Orozco et al., 2016; Titeux et al., 2016). This focus on climate change is constructive in raising awareness of the negative ecological pressures of human activity, however other anthropogenic causes of biodiversity loss need to be considered. Climate change is certainly a driver of biodiversity loss at multiple biological scales including the genetic, species, community and ecosystem levels (Mantyka-Pringle et al., 2015) and across a broad range of taxa (Bellard et al., 2012). Recent climate change however is a result of human activity and as such it is not the root anthropogenic cause of these biodiversity declines. Biodiversity loss has been attributed to factors such as meat consumption in developed countries (Stoll-Kleemann and Schmidt, 2017), overfishing (Boudouresque et al., 2017), the spread of infectious diseases from domestic animals to wildlife (Daszak, 2007), introduced species (Doherty et al., 2016), habitat loss and habitat fragmentation (Bartlett et al., 2016). Many of these factors are directly or indirectly related to human activities that have a deleterious effect on natural ecological systems such as deforestation (Barlow et al., 2016), agricultural expansion (Moraes et al., 2017) and urban sprawl (Dupras et al., 2016). The resultant habitat modification from these activities fall under the umbrella concept of land-use change. The diversity and pervasiveness of global land-use change means that the associated impacts on ecological systems are numerous and complex, presenting a major threat to global biodiversity (Newbold et al., 2015).

The maintenance of biodiversity is essential for the health of ecosystem functionality (Oliver et al., 2015), which is in turn essential for the wellbeing of the human race through the provision of ecosystem services (Sandifer et al., 2015). Conservation efforts are critical in the preservation of biodiversity (Johnson et al., 2017), however uncertainty over the likely success of such efforts can be problematic to securing funding and resources to combat biodiversity loss (Waldron et al., 2017). As conservation resources are distributed at multiple spatial scales including at international, national and local levels, resolving these uncertainties requires the gathering of coherent ecological and conservation impact evidence at multiple spatial scales, often in the context of one another (Baylis et al., 2016).

1.2. Impacts of development on bird populations and communities

Modern society is facing a multitude of developmental pressures that have the potential to be extremely detrimental to biodiversity, including bird populations and communities. These pressures include the expansion of urban areas, large scale agricultural expansion, land-use change, climate change as a result of fossil fuel burning, and paradoxically, renewable energy developments (Maggini et al., 2014; Thaxter et al., 2015; Batáry et al., 2017; Quinn et al., 2017) . Studies have revealed that these forms of development can have pronounced negative impacts on residential and migratory bird populations and communities through habitat loss, degradation and fragmentation (Marzluff, 2001; Filippi-Codaccioni et al., 2008; Marzluff and Ewing, 2008). Negative impacts include bird disturbance (Drewitt and Langston, 2006), adverse changes to bird behaviour (Larsen and Guillemette, 2007), changes in community composition (Blair and Johnson, 2008), direct mortality (Grecian et al., 2010), loss/avoidance of nesting and breeding sites (Morrison et al., 2011), reduction in food resources (Mennechez and Clergeau, 2006) and increased competition from colonial and successional/invasive species that are more adapted to urban habitats (Bonier et al., 2007).

The expansion of urban development and associated changes in land-use poses a number of threats to biodiversity and ecosystem function (Seto et al., 2012). The allocation of Special Protection Area (SPA) or Special Area for Conservation (SAC) status to key areas important for biodiversity provides a means of directly avoiding the physical effect of urban development by prohibiting or heavily restricting development in these areas (Morris, 2011). Nevertheless, it is important to understand how development and anthropogenic activity outside the boundary of a protected area might affect the ecological processes within, especially with regard to bird populations (Mas, 2005; Martínez et al., 2007; Kharouba and Kerr, 2010; Guix and Arroyo, 2011; Pérez-García et al., 2011). External pressures with the potential to affect wildlife within protected areas include factors such as increased recreational pressure, increased pollution (light, noise and chemical), increased predation from domestic pets and the alteration or loss of habitat (Yalden, 1992; Pearce-Higgins et al., 2007; Reed and Merenlender, 2008; Mcdonald et al., 2009; Aubrecht et al., 2010; Hölker et al., 2010; Radeloff et al., 2010; Wierzbowska et al., 2012). A literature review by Mcdonald et al. (2009) examined 163 studies and found 22 potential negative effects of urbanization on protected areas including some of the ecological impacts outlined above.

Bird species that are highly mobile may require habitat that extends beyond the boundary of a protected area as well as the habitat contained within. For example, in the UK, Golden Plover *Pluvialis apricaria* often fly greater than four kilometres between nesting and feeding sites (Whittingham et al., 2000). This species utilizes enclosed fields with abundant tipulid larvae as feeding habitat (Whittingham et al., 2000; Pearce-Higgins and Yalden, 2003) and breeds on heather moorland and blanket bog (Pearce-Higgins and Yalden, 2004). Thus, if breeding habitats are within the boundary of a protected area, but feeding habitats are situated mainly outside the boundary, there is the potential for urban development outside of the protected area to affect the survival of breeding populations within. One proposed solution to this problem involves 'biodiversity offsetting', where equivalent habitats are created elsewhere in place of another lost to development (Regnery et al., 2013). This is a strategy that has been implemented for a variety of habitats around the world, including forest and shrubland in New Zealand (Norton, 2009), wetlands in the United States of America (Zedler, 1996), sub-montane forest in the Republic of Guinea (Kormos et al., 2014), freshwater and marine habitats in Canada (Quigley and Harper, 2006) and habitats protected under the EU Habitats and Birds directives in France (Regnery et al., 2013). In the UK, DEFRA have published guidelines for the practice of biodiversity offsetting (DEFRA, 2011), however scientific case studies in the UK are elusive. For biodiversity offsetting to be successful it is necessary to understand the ecological requirements of the species assemblage at both landscape and temporal scales within the target habitat (Maron et al., 2012). Quantifying these requirements is difficult and as such the effectiveness of biodiversity offsetting remains largely untested, disputed or subject to suggestions for improvement (Hayes and Morrison-Saunders, 2007; Gordon et al., 2009; Quétier and Lavorel, 2011; Bull et al., 2013). One of the core concepts of biodiversity offsetting is the that of 'no net loss' to biodiversity, or more optimistically, a 'net gain' where possible (Schoukens and Cliquet, 2016; Bull and Brownlie, 2017). The uncertainty in quantifying baseline biodiversity at development sites for offsets to be measured against has resulted in controversy over the implementation of biodiversity offsetting (Gordon et al., 2015). Part of this controversy comes from the fact that biodiversity offsets are often only gauged by the losses likely to be incurred as a direct result of a particular development, meaning that any losses that are predicted to occur under scenarios despite development can be transferred to the offset site, maintaining an overall biodiversity decline (Maron et al., 2015). Another controversial aspect of biodiversity offsetting is that ecological damage will undoubtedly be caused in the area where development takes place, with no guarantee that the offset site will provide equivalent opportunities for biodiversity (Evans et al., 2015), partially through uncertainty over the success of mitigation activities at the offset site, but also through the inherent difficulty in quantifying geographically separate ecosystems as comparable (Apostolopoulou and Adams, 2017). In addition, there are concerns over the ability of authorities to monitor adherence to biodiversity offsetting programs and to avoid the introduction of counter incentives that undermine biodiversity offsetting efforts (Maron et al., 2016).

The ecological benefits of considering the wider landscape and its associated land use in the role of preserving biodiversity is important, but can be difficult to achieve due to the complexity of ecological systems where community diversity, functional ecology, structural ecology and genetics all contribute to the health of biodiversity (Waldhardt, 2003). In places where urban development encroaches on protected areas, it may be necessary to consider biodiversity offsetting in areas outside of the protected area, in an attempt to compensate for any negative effect on biodiversity, and to maintain habitat heterogeneity in the context of the wider landscape (Santos et al., 2008). The interface between a protected area and the unprotected surrounding habitats should be taken into consideration if we are to fully understand how biodiversity and ecosystems may be affected by urban encroachment in the vicinity of protected areas (Palomino and Carrascal, 2006; Filippi-Codaccioni et al., 2008; Knapp et al., 2008; McDonnell et al., 2008), particularly if unprotected habitats are known to support species of conservation importance from nearby protected areas (Santos et al., 2008).

In the United Kingdom, local governments are under increasing pressure to grant planning applications for new developments as a result of policies including the Local Development Framework (LDF) (DCLG, 2008) and the Strategic Housing Land Availability Assessment (SHLAA) (DCLG, 2007). These require the identification of suitable development sites, including areas that are potentially high in biodiversity such as rural settlements, brownfield sites outside of settlement boundaries and greenfield sites (Adams, 2011). The National Planning Policy Framework was established in order to provide a framework for sustainable development in the UK, facilitating economic and social development with some emphasis on environmental protection (DCLG, 2012). This includes avoiding any adverse effects of local development by adhering to obligations set out by the European Commission (EC) Birds and Habitats Directives (EEC, 1979, 1992; EC, 2009). This has resulted in a shift towards the empowerment of local governments and rural communities with regards to planning decisions and is reinforced by the 2010 Localism Bill (DCLG 2011). As such, local authorities are under obligation to prioritise areas for development, whilst maximising the integrity of local characteristics and biodiversity.

1.3. Impact of agricultural land-use practices on birds

In England, Agri-Environment Schemes (AES) are represented in the form of Environmental Stewardship programmes (ES). These were introduced in 2005 to take into account reform of the Common Agricultural Policy (CAP) and to place emphasis on financially reimbursing farmers for biodiversity enhancing activities (Baker et al., 2012). Historically, CAP rewarded farmers for intensification of farming practices, which led to severe declines in farmland bird populations (Donald et al., 2001; Gregory et al., 2005; Donald et al., 2006; Sanderson et al., 2006). The intensification of various agricultural practices e.g. increased livestock numbers and grazing activity (Fuller and Gough, 1999) rotational cropping, agro-chemical input and multiple silage cuttings have had a negative effect on the breeding success, diversity and population densities of bird species (Chamberlain et al., 2000; Guerrero et al., 2011) at local and landscape levels by impacting habitat quality (Wilson et al., 1997; Donald et al., 2001), changes in food availability, and predation pressure (Fuller and Gough, 1999).

The purpose of AES is to mitigate for any negative effects of farming intensification on bird populations and other aspects of biodiversity in Europe, and to enhance natural biodiversity in rural habitats. The uptake of AES on farmland close to areas of anthropogenic development has the potential to off-set some of the potential negative effects of local development on birds (Whittingham, 2011) by increasing the land coverage of food abundant habitats for birds. AES that adhere to the objective of improving bird populations have proven to be a success for a range of species including Cirl Bunting *Emberiza cilus* (Peach et al., 2001), Stone Curlew *Burhinus oedicnemus* and Corncrake *Crex crex* (Wilson et al. 2010). In addition to increasing food abundance, the promotion of a diverse crop and habitat structure has the potential to boost populations of birds that utilise moorland habitats such as Lapwing *Vanellus vanellus* , Redshank *Tringa totanus*, Skylark *Alauda arvensis*, Starling *Sturnus vulgaris*, Linnet *Carduelis cannabina* and Reed Bunting *Emberiza schoeniclus* by providing cover for feeding and for the avoidance of predation (Berg and Part, 1994; Wilson et al., 2005). AES are generally targeted at the local scale i.e. at individual field or farm level (Guerrero et al., 2012) and not necessarily at the landscape scale. Kleijn et al. (2006) and Vickery et al. (2009) argue that for AES to be successful in the recovery of farmland bird populations, habitat configuration at the landscape scale and the requirement of specific species need to be considered, especially where the conservation of endangered species is a priority.

Organic farming practices, leaving ‘set-aside’ (unfarmed land) and the maintenance of a mosaic habitat structure at the landscape level have all been suggested as important factors for the recovery of farmland birds (Wretenberg et al., 2010). Edge habitat and field margins have the potential to maintain habitat heterogeneity and should be considered as important components of the habitat landscape, however species specific requirements should be taken into consideration to optimise their efficacy (Sanderson et al., 2009; Kuiper et al., 2013). The combination of field margin maintenance and leaving set-aside are deemed to be extremely important for the recovery of many species, including Skylark (*Alauda arvensis*), Linnet (*Carduelis cannabina*), Common Whitethroat (*Sylvia communis*) and Whinchat (*Saxicola rubetra*) (Berg and Part, 1994). A mosaic of habitats with varied structural characteristics and plant species composition has been suggested as optimal for sustaining moorland bird population (Buchanan et al., 2006), suggesting that AES should be managed at a landscape level, where individual participants are not treated in isolation, but in the context of surrounding AES.

1.4. Wind turbines and bird populations

Research into the ecological effects of wind turbines has generally focussed on wind farms with multiple large turbines. With financial incentives available within the UK for small-scale electricity generation, there is an increasing trend towards the construction of small wind turbines (SWTs) in areas of high wind resource availability. The ecological effects of SWTs on UK biodiversity are not well understood, making it difficult for local authorities to make informed planning decisions. To date, only one experimental scientific paper has empirically quantified SWT-bird interaction. Minderman et al. (2012) examined bird flight behaviour within 20m of 20 individual SWTs but did not find any negative effect on the flight behaviour of birds within this distance of SWTs. Other research has addressed the issue of integrating ecological evidence into planning policy, using the lack of empirical ecological evidence regarding SWTs as an example for advocating better communication between scientists and policy makers and planning departments (Park et al. 2013). Consistent terminology in the scientific literature is regarded by many as key to the mutual understanding of concepts between scientists. ‘Small Wind Turbine’ (Minderman et al., 2012) and ‘Micro-Turbine’ (Park et al. 2013) are often used to mean an electricity generating wind turbine of generating capability <50kW.

There is a widespread misconception amongst some sections of society that the threat of wind turbines on birds is limited solely to the potential for bird strike (Leung and Yang, 2012). This is not aided by the fact that the majority of research attempting to reconcile bird ecology and wind turbines appears biased towards collision risk and direct mortality (e.g. De Lucas et al. 2008; Ferrer et al. 2012; Péron et al. 2013). There is a considerable and growing body of research that has focussed on the collision mortality of birds with onshore wind turbines, especially with regards to raptors (e.g. Barrios & Rodríguez 2004; De Lucas et al. 2008; Schaub 2012; Dahl et al. 2013; Hull & Muir 2013). Similarly, there is much research into the bird collision risk of offshore turbines for numerous migratory and marine birds (e.g. Plonczkier & Simms 2012; Johnston et al. 2014). Determining if the rate or risk of collision is of ecological significance to bird populations is extremely complex, as it is deemed to be species specific, location specific, and size specific (in terms of the size of a wind farm and the turbines), associated with topography, weather, season and land (Herrera-Alsina et al., 2013). Collision risk however is only one of many factors that could present a potential threat to the viability of bird populations around wind turbines. Other threats include displacement as a result of disturbance, habitat loss and degradation, and the creation of barriers to movement, altering the migration routes or daily movement patterns of birds (Drewitt and Langston, 2006; Masden et al., 2009, 2010; Plonczkier and Simms, 2012; Winiarski et al., 2014). This multitude of variables make it difficult to determine in advance whether a wind turbine development may adversely affect a bird population (Powlesland, 2009).

Using standardized pre-construction surveys, informed placement of turbines can theoretically minimise negative impacts (Madders and Whitfield, 2006). The current consensus in the ecological community appears to be that prior monitoring of a proposed wind turbine site for bird activity and placement based on a ‘least impact’ basis is the best way to minimise risk, i.e. by conducting an Environmental Impact Assessment (EIA) (Desholm et al., 2006). Adopting EIAs seems logical and relatively simple, but different guilds of birds require different survey methodologies, different seasonal emphasis, and in some cases long term monitoring covering several years in order to make sound estimates of abundance and distribution (Niemuth et al., 2013). Furthermore, there is some evidence to suggest that the spatial arrangement of turbines within the landscape can negatively affect bird species such as Red Kite (*Milvus milvus*) (Schaub, 2012). An approach has been proposed that involves pre-empting conflict between birds and wind turbines at the landscape level (Bright et al., 2008). This involves avoiding the overlap of turbine locations with areas of importance to birds that present a high turbine risk based on foraging range, collision risk and sensitivity to disturbance (Bright et al., 2008).

1.5. Predicting the responses of birds to forms of development

The terrain of UK uplands and the logistical constraints of conducting ecological surveys in these difficult to access areas makes estimating the abundance and distribution of upland birds problematic. Habitat quality and extent are known to be important determinants of bird densities on moorland (e.g. Haworth & Thompson 1990; Brown & Stillman 1993; Stillman & Brown 1994). As such, appropriate habitat data used in conjunction with reliable bird abundance–habitat association models could allow more accurate predictions of bird abundance across upland areas in relation to the potential for adverse effects from forms of development.

Predictive modelling is an increasingly important analytical tool which enables ecologists to assess the influence of environmental variables and anthropogenic development on bird populations without conducting exhaustive surveys over large areas and over extensive periods of time. Typically, previous studies have involved constructing forms of regression models such as General Linear Models, (Martínez-Abraín et al., 2012), Generalised Linear Mixed Models (GLMMs) (Devereux et al., 2008) and Generalized Linear Models (GLMs) (Pearce-Higgins et al., 2009). One increasingly common and useful predictive approach is to use Species Distribution Modelling (SDM) to produce expected occupancy values over large areas where complete field surveys are not feasible. MaxEnt software (Phillips et al., 2006) has increasingly been used in recent years to model species distributions across disturbed landscapes to assess the impacts of habitat loss, fragmentation and degradation (e.g. Lu et al., 2012). This maximum entropy modelling framework identifies the environmental factors that are most related to the distribution of a species and the probability of occurrence in a given area using presence-only occurrence data (Phillips et al. 2006). MaxEnt is capable of dealing with both continuous and categorical environmental variables simultaneously (Phillips et al. 2006), and is particularly well suited for small sample sizes that are typical of many species occurrence data sets (e.g. Pearson et al. 2006; Wisz et al. 2008). Fitzpatrick et al. (2013) argue that MaxEnt does not provide a direct estimate of occurrence probability, rather an index of habitat suitability. However, for planning decision making purposes this is likely to be sufficient.

There are other SDM models available and one approach is to adopt a consensus or ensemble approach, employing a suite of commonly utilized SDM techniques to project and compare predicted current species distributions and potential future species distributions (Araújo and New, 2007; Marmion et al., 2009). This approach can be implemented using the BIOMOD framework (Thuiller et al., 2009) within the R programming environment (R Core Team, 2013). The BIOMOD computational framework aims to maximize the predictive accuracy of current species distributions and the reliability of potential future distributions using several different statistical modelling techniques (Thuiller, 2003; Thuiller et al., 2009) including machine learning techniques and regression techniques. Stevens et al. (2013) used a combination of MaxEnt SDM analysis and Binary Logistic Regression models to determine the influence of wind turbines on the likelihood of habitat occupancy of several grassland-dependent bird species. No significant effects of the turbines were found for most of the target species except for Le Conte's Sparrow *Ammodramus leconteii*, where evidence for displacement by turbines was shown. The study did not take into account other environmental variables such as wind speed, rain or cloud cover.

Models have also been developed for the site selection of wind turbine development based on factors other than (but incorporating) ecological impact. One example is the use of Spatial Multi-Criteria Analysis (SMCA) in a Geographic Information System (GIS) to select suitable sites for turbine construction based on ecology, economics, wind resource and geology (van Haaren and Fthenakis, 2011). This approach does not replace EIA as a measure for site selection, but allows the filtering of sites prior to EIA, thus reducing cost and resource use whilst improving efficiency and efficacy.

1.6. Moorland habitats in the United Kingdom

The creation of SPAs and SACs have brought international recognition to the threatened biodiversity of semi-natural moorland habitats throughout the United Kingdom (Littlewood et al., 2006). The UK uplands are of international conservation importance for their range of moorland and blanket bog plant communities and associated breeding bird assemblage (Thompson et al., 1995). The upland moorland habitats of blanket bog and dwarf heath (including heather moorland) cover around 23.6% of Scotland, 3% of England and 6.2% of Wales, and are considered to be biodiversity action plan priority habitats by the Centre for Ecology and Hydrology (CEH) in the UK (Carey et al., 2008). There are 19 constituent plant communities of upland habitats, 13 of which are listed under the EC 'Habitats Directive' 92/43/EEC and five of which are almost entirely confined to Britain (Evans et al., 2006). Modern upland moorland in the UK is distinctive for the low, dense vegetation associated with these habitats, however these are the product of almost 4000 years of human land management (Birks, 1988 in Littlewood et al., 2014). Before this, the UK uplands were dominated by woodland and scrub, which was gradually deforested to make way for agriculture and in the 19th century the moors began to be manipulated for commercial scale grouse shooting (Simmons, 2002). The management practices of vegetation burning and livestock grazing over these years have led to complex successional changes in vegetation, and are vital in the maintenance of the semi-natural habitats of UK uplands (Simmons, 2002; Yallop et al., 2006). Moorland habitats are of high ecosystem service value, with provisions including water supply, climate regulation, carbon sequestration, recreation and aesthetic value (Bonn et al., 2009; Ostle et al., 2009). Moorlands also provide genuine economic income through employment and tourism associated with the perceived natural beauty and cultural heritage of the areas (Orr et al., 2008).

Between 1947 and 1980, around 20% of upland heather moorland present in England and Wales was transformed due to afforestation, agricultural reclamation, high grazing pressures and bracken *Pteridium aquilinum* invasion (Thompson et al., 1995). Of the remaining, 70% was estimated to be at risk of further change, with more recent research citing atmospheric deposition, climate change, and peat erosion due to the legacy of overgrazing as risks to moorland habitats (Holden et al., 2007). The international and national importance of biodiversity supported by the UK uplands is reflected by the fact that much of the area it covers is under legislative protection such as Special Protection Area (SPA), Special Area of Conservation (SAC) or Site of Special Scientific Interest (Orr et al., 2008). However, protected status isn't necessarily indicative of the health of upland habitats. For example 16% of UK uplands are designated as Sites of Special Scientific Interest (SSSI) (Reed et al., 2009) but a large proportion of these are in unfavourable condition (similarly for upland SACs) (Williams, 2006).

Ownership of land in the UK uplands is complex, with property rights distributed amongst stakeholders with different agendas and priorities (Quinn et al., 2010). The majority of moorland is privately owned and managed for red grouse and sheep production (Reed et al., 2013), but other landowners and stakeholders include water companies, the Forestry Commission and conservation NGOs (Quinn et al., 2010). This multitude of bodies alongside the largely unrestricted recreational access under the CROW (Countryside Rights of Way) act 2000 creates a difficult set of challenges when trying to gain consensus on the perceived threats to moorland habitats and their potential solutions.

1.7. Moorland bird communities: ecology and threats

Upland moorland habitats support internationally important breeding populations of migratory and resident bird species, including eight species listed under annex 1 of the EC birds directive (Thompson et al., 1995). These species are Peregrine *Falco peregrinus,* Golden Plover *Pluvialis apricaria*, Short-eared Owl *Asio flammeus* Merlin *Falco columbarius.* Hen Harrier *circus cyaneus,* Greenland White-fronted Goose *Anser albifrons*, Golden Eagle *Aquila chrysaetos* and Red Kite *Milvus milvus.* The first four of these are represented within the SPMSPA. Other species of international importance supported by moorland habitat include Greenshank *Tringa nebularia*, Curlew *Numenius arquata*, Meadow pipit *Anthus pratensis*, Dunlin *Calidris alpine*, Skylark *Alauda arvensis*, Great Skua *Stercorarius skua*, Whimbrel *Numenius phaeopus*, Twite *Acanthis flavirostris*, Raven *Corvus corax* and Red Grouse *Lagopus lagopus* (Thompson et al., 1995). Upland moorland also supports several additional species of UK conservation concern including Lapwing *Vannellus vannellus*, *S*nipe *Gallinago gallinago*, Redshank *Tringa totanus*, Common Sandpiper *Actitus hypoleucos*, Whinchat *Saxicola rubetra*, Wheatear *Oenanthe oenanthe* and Ring Ouzel *Turdus torquatus* (Eaton et al., 2009).

Forty percent of forty moorland bird species ranges and populations contracted or declined between the early 1970s and early 1990s as a result of afforestation, persecution, heavy grazing pressure and land drainage (Thompson et al., 1995). More recent research assessing the impacts of some of these pressures on moorland bird diversity as well as population trends for individual species are varied in their results. Pearce-Higgins and Grant (2006) show that declines in moorland heather cover over two decades have reduced habitat availability for only two of nine species, and that heterogeneity of vegetation composition and structure are more important for moorland bird diversity. This landscape mosaic approach is championed for upland bird diversity by others too, both within the core moorland area and in the surrounding landscape (Dallimer, Marini, et al., 2010). The management practices of vegetation burning and predator control undertaken on grouse moors can affect populations in both positive and negative ways for bird species (Grant et al., 2012).

1.8. Moorland fringe land-use and bird communities

Farmland often dominates the moorland fringe habitats that span protected area boundaries, meaning that there may be a conflict of interest between these programmes and the conservation objectives of the protected area. This problem is compounded when land owners manage land both within and outside a protected area. For example, there may be occasions when a farm implements AES within the protected area, but not in surrounding moorland fringe farmland (Dallimer, Marini, et al., 2010). For some moorland species such as Snipe and Curlew, there is a clear association between moorland habitat management and the management of the surrounding farmland in terms of the success of these species (Dallimer et al., 2012). The relationship between usage of these habitats differs between species, with some species such as Lapwing and Skylark being more typical of farmland but also utilizing moorland areas. Species such as Meadow Pipit, Snipe and Curlew favour both habitats whereas others such as Golden Plover favour moorland but also utilize farmland and moorland fringe to supplement their feeding ecology (Pearce-Higgins and Yalden, 2003; Dallimer et al., 2012). Within core moorland habitats, loss of nests at the start of the breeding season (April) due to heather burning may result in a decrease in breeding productivity for species such as Oystercatcher, Peregrine Falcon, and Wheatear (Moss et al., 2005). As such, it is important to consider how moorland bird species utilize the moorland fringe habitat surrounding the protected area boundary (at the species level and at the population level) and manage the landscape in a way that balances the survival and productivity of these with the livelihoods and practices undertaken by stakeholders and land managers. This is particularly relevant where the management practices undertaken on core moorland do not appear to explain moorland bird population trends (Calladine et al., 2014).

1.9. Moorland fringe landscape and local development

Policies which influence land use within SPAs and the surrounding moorland fringe are traditionally based on conservation practices that restrict planning or development (Bright et al., 2008). However, recent years have seen a significant increase in proposals for residential, commercial, recreational and renewable energy development on SPA moorlands and within the fringe bordering these protected areas (Pearce-Higgins et al., 2012) that pose a significant threat to the moorland birds (Douglas et al., 2012; Pearce-Higgins et al., 2012). Moorland fringe habitats are now viewed as transitional areas between proposals for local development and core areas for conservation (Dallimer, Gaston, et al., 2010). The benefits of wind-turbine technologies are well established in terms of the reduction of greenhouse gases (Pearce-Higgins et al., 2012) whilst further residential, commercial and recreational development are necessary to deal with the increasing urban pressures (Bright et al., 2008)(Bright et al., 2008). However, local governments lack the evidence base which would enable them to understand the impacts on threatened bird species of proposals within Local Development Frameworks that allocate areas for new development as well as setting out policies for renewable energy. Therefore, to ensure the international status of SPAs and their purpose in maintaining viable populations of threatened birds requires a better understanding of the distribution and abundance of habitats and birds within the moorland fringe landscape surrounding moorland SPAs and identifying causal links between forms of local development and the effects on bird populations across these landscape mosaics.

1.10. Overall Aim of the PhD and objectives of chapter

The overall aim of the PhD is to characterise the habitats and bird communities of the moorland fringe landscape of the South Pennine Moors Special Protection Area (SPMSPA), in northern England. These data will enable cross boundary local governmental co-operation to assess how these bird populations may be impacted by different development scenarios and permit sustainable local planning decisions to be made for residential and recreational development, and small scale (micro) wind turbine construction for the moorland fringe buffering the SPA. The SPMSPA consists of three spatially distinct areas of core moorland habitat embedded in a landscape mosaic of urban and fringe moorland habitats and as such, presents an important ecological site at UK national and European level.

The PhD has the following objectives:

(1) To map the distribution and estimate the extent of the different habitats within the moorland fringe landscape.

(2) To identify patterns of bird community composition and abundance across the moorland fringe landscape and determine how these are influenced by the habitat characteristics of developed and undeveloped areas within the SPA.

(3) To determine whether abundance and distribution of birds are influenced by the presence of small-scale (micro) wind turbines.

(4) To develop species distribution models for the most threatened moorland bird species that enables sustainable development planning decisions to be made by authorities with joint custody of the SPMSPA.

1.11. Scope of Chapters

Each chapter is designed to investigate an aspect of the moorland fringe bird community in relationship to either built development or agricultural activity. The findings are intended to provide evidence that can be used by the local authorities of Calderdale, Kirklees and Bradford in assisting planning decisions required within 1km of the SPMSPA.

Chapter 2: Characterization of the moorland fringe landscape around the South Pennine Moors Special Protection Area
This chapter describes and quantifies the habitats found across the SPA moorland fringe landscape. Data from field surveys, Centre for Ecology and Hydrology (CEH) datasets and Landsat images are used to quantify the extent of different habitats and examine temporal changes across the fringe within 1km buffer from the SPMSPA boundary from 1990, 2000, 2007, 2012 and 2013.

Chapter 3: Patterns of bird community composition and habitat associations of moorland fringe bird populations
In this chapter, data from line transect surveys are presented to reveal the patterns of species richness, diversity and evenness across the different habitat categories of the moorland fringe landscape. Non-metric multidimensional scaling (NMDS) and generalized additive models (GAMs) are used to identify the key features of the habitat that influence the abundance of the conservation priority bird species.

Chapter 4: Influence of small wind turbines on the abundance and distribution of moorland fringe bird species
In this chapter data from additional line transect and habitat surveys are used to examine the richness, diversity, abundance and habitat associations of birds in designated small wind turbines (SWTs) and to determine whether SWTs cause displacement of these species within the fringe landscape. Associations between the presence of individual bird species and distance to SWTs are investigated.

Chapter 5: Predicting species distributions across moorland fringe landscapes
This chapter will determine the distribution of conservation priority moorland fringe bird populations across the whole SPMSPA fringe landscape. Bird-habitat distributions are modelled using *biomod2* in R Development software. These models can then help frame an

appropriate conservation agenda for these species and facilitate sustainable development planning decisions to be made by local authorities and stakeholders with joint custody of the SPMSPA.

Chapter 6: Summary

In this final chapter I outline the main findings of the PhD and make recommendations for future bird research on moorland fringe landscapes.

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CHAPTER 2: HABITAT CHARACTERIZATION OF THE MOORLAND FRINGE LANDSCAPE

2.1. Abstract

Buffer zones of reduced anthropogenic activity around protected areas have the potential to help maintain the conservation integrity of protected areas. The South Pennine Moors Special Protection Area (SPMSPA) is an upland protected area in the North of England that is protected specifically for its breeding bird assemblage that is surrounded by historic industrial conurbations and farmland. Intensification of farming practices over the latter half of the twentieth century have resulted in the declines of many bird species across the United Kingdom and residential areas are under pressure to grow from governmental housing policy. This chapter aims to; (1) describe the landscape and quantify the habitats associated with the SPMSPA moorland fringe within the unitary authority areas of Calderdale, Kirklees and Bradford; (2) examine temporal habitat change within the fringe landscape; (3) Determine gradients in building density, elevation and moorland habitats as a function of increasing distance from the SPA and (4) Classify the moorland fringe landscape using Landsat 8 remotely sensed spectral bands.

Habitat surveys were undertaken in 2012 and 2013 for 1,284 fields within 1km of the SPMSPA boundary. Temporal change in habitat coverage in relation to the habitats within the SPMSPA was assessed using Centre for Ecology and Hydrology (CEH) data. Landsat 8 data were used to undertake classification of the moorland fringe landscape. Gradients in building density, elevation and habitat were assessed in the fringe landscape.

Fourteen habitat categories were recorded. Most fields comprised agricultural habitats. Upland habitats were in the minority. Upland habitats were found to increase in coverage from 1990 to 2000 and decrease again between 2000 and 2007. Building density was linear as a function of the SPMSPA boundary. As much of the fringe is improved, it is recommended that monitoring agricultural intensification and upland habitat loss would provide insight into SPMSPA bird conservation.

2.2. Introduction

For many protected areas, maintaining site integrity - a term used to describe an authority's responsibility for assessing potential adverse impact factors on protected areas - should involve the inclusion of buffer zones and connecting areas that extend beyond a protected area's boundary (Rees et al., 2013). These buffer zones have the potential to mitigate edge effects inflicted by encroaching anthropogenic development and activity (Gurrutxaga et al., 2010). The expansion of urban development and associated changes in land-use poses a number of threats to biodiversity and ecosystem function in these areas (Seto et al., 2012). The allocation of Special Protection Area (SPA) or Special Area of Conservation (SAC) status to key areas important for biodiversity provides a means of directly avoiding the physical effect of urban development by prohibiting or heavily restricting development in these areas (Morris, 2011). However, it is also important to understand how development and anthropogenic activity outside the boundary of a protected area might affect the ecological processes and species populations within, especially with regard to threatened bird populations in the case of SPAs (Mas, 2005; Martínez et al., 2007; Kharouba and Kerr, 2010; Guix and Arroyo, 2011; Pérez-García et al., 2011). External anthropogenic pressures outside protected areas with the potential to affect wildlife within the protected area include increased recreational pressure, increased pollution (light, noise and chemical), increased predation from domestic pets and the alteration or loss of habitat (Yalden, 1992; Pearce-Higgins et al., 2007; Reed and Merenlender, 2008; Mcdonald et al., 2009; Aubrecht et al., 2010; Hölker et al., 2010; Radeloff et al., 2010; Wierzbowska et al., 2012).

The moorland fringe habitats of the UK uplands have been subject to intense agricultural improvement over the latter half of the twentieth century through practices such as drainage, the application of inorganic fertiliser, reseeding for pasture, and increased sheep grazing (Dallimer et al., 2010). In contrast, farmland within the moorland fringe landscape is generally considered to be less intensively managed than land at lower altitudes due to remoteness, inaccessibility for heavy machinery and lower expected returns on agricultural intensification (Murray et al., 2016). In the UK bird species such as the declining Whinchat *Saxicola rubetra* rely on habitat that is not intensively managed, but is not at too high an elevation (Calladine and Bray, 2012). As such, the moorland fringe presents an ideal habitat for bird species that can tolerate or thrive in moderately high elevational areas, but require low intensity farmland. Upland bird species that utilise the moorland fringe such as Lapwing *Vanellus vanellus*, Snipe *Gallinago gallina*go, Skylark *Alauda arvensis*, Twite *Carduelis flavirostris* and Reed Bunting *Emberiza* *schoeniclus* have experienced declines in recent years in the UK (Fuller et al., 2002). As such, it is imperative to understand the agricultural landscape of moorland fringe areas in the context of the habitat requirements of birds such as these. Natural England describe the agricultural landscape in the South Pennines moorland fringe as a mosaic of small to medium fields dominated by relatively intense sheep farming, but with the presence of less improved habitats such as wet grassland, rush pasture and species rich meadows (Natural England, 2012). These fringe habitats have the potential to provide breeding and/or feeding grounds for threatened moorland bird species such as Snipe, Lapwing, Skylark and Twite (Fuller et al., 2002; Hoodless et al., 2007), including individuals that use the moorland SPA. In order to make inferences about how these birds might be distributed within the moorland fringe, it is important to understand the spatial configuration of the habitat types within the fringe, both in relation to one another and also in relation to the SPA boundary. Although there are some efforts to describe moorland SPA fringe landscapes in qualitative terms, e.g. as part of the National Character Profile for the South Pennines area in northern England (Natural England, 2012), few quantitative data are readily available. The collation and analysis of such data would improve the understanding of the relationship between SPAs and their surrounding landscape, providing a framework to plan and manage developmental pressures alongside agricultural trends whilst minimising impacts on SPA biodiversity.

Qualitatively, agricultural land utilised by some moorland birds within moorland fringe is often referred to as ‘in-bye’ but this lacks a tangible definition in terms of the habitat and landscape characteristics. French and Dolmans (2002) define in-bye based on the aggregation of broad habitat categories described in the 1980 UK Countryside Survey (Barr et al., 1993). The French & Dolmans (2002) definition of *in-bye* describes a landscape composed of all managed grasslands, including intensive and sown swards, less intensively managed grasslands in lowland or enclosed situations, dune grasslands and unmanaged grass/tall-herbs, usually in a lowland or in-bye situation but that does not include recreational grass or upland/moorland grass. This is extremely broad, and has the ambiguity of including the term ‘in-bye’ within the definition. French & Picozzi (2002) provide a similar definition, however provide some management context by including grazed pastures and hay/silage fields. Other studies highlight the practice of sheep farming as an important component of in-bye habitat (Mackay, 1975; Mitchell and Renton, 1983; Conington et al., 1995; Stott et al., 2012). Other definitions of in-bye are not habitat or land use specific, but rather relate to the spatial characteristics of the land. Examples include “the fenced in land nearest the homestead” (Royal Commission on Common Land, 1958), “enclosed ground” (Phillips, 2012) and “a field close to the centre of a farm, or a field on a farm” (OED, 2015). It would appear from these definitions that although in-bye may not be synonymous with moorland fringe, or indeed not even associated with moorland habitat, it is likely to make up a significant component of the SPMSPA fringe based on the qualitative landscape characteristics (Natural England, 2012). The term ‘moor edge’ was previously used by French & Picozzi (2002) in reference to the landscape immediately neighbouring moorland and upland habitats (except where this is in-bye) such as arable land and forest.

The agricultural and ecological landscape of SPA fringe habitats are interwoven with the ever-expanding landscape of urban development. There is currently extreme pressure on local government in the UK to identify areas for residential development (DCLG, 2012) and areas for other forms of development, potentially in close proximity to SPAs. To better understand the ecological pressures of development within the SPA fringe habitat, it is first necessary to understand the current spatial patterns of development and habitat types in relation to the location of the SPA boundary.

This chapter aims to describe and quantify the habitat configuration of a moorland fringe landscape surrounding the SPMSPA, in Northern England. An empirical spatial analysis of the moorland fringe landscape surrounding the SPMSPA will be provided and the configuration of habitats within 1 km of the SPMSPA boundary using data collected in habitat surveys conducted in 2012 and 2013 will be quantified. Temporal changes in habitat coverage will be examined using Centre for Ecology and Hydrology (CEH) Land Cover Map (LCM) datasets from 1990 (Fuller, 1995), 2000 (RM Fuller et al., 2002), and 2007 (Morton et al., 2011). Patterns in habitat coverage with increasing distance from the SPMSPA boundary and hotspots of moorland habitat in the SPA 1km fringe will be analysed also using LCM. Trends in topography and the quantity and density of buildings with increasing distance from the SPMSPA boundary will be investigated using Ordnance Survey data. Landsat 8 data will be used to perform a supervised classification of the SPMSPA fringe (within 1km of the SPMSPA) using the habitat data collected during habitat surveys. Using the results of this classification, the habitats of the SPMSPA fringe will be inferred and investigated in the context of Natural England’s National Character Profile (Natural England, 2012). If the hypothesis that farmland improvement and agricultural intensification is low within the SPMSPA moorland fringe (as described by Natural England), then this is likely to be beneficial for bird species utilising the moorland fringe, as agricultural intensification is known to cause declines in bird populations (Butler et al., 2010). If agricultural intensification is high, then this presents a potential mechanism for bird species declines within the moorland fringe landscape.

2.3. Methods

2.3.1. Study Site

The South Pennine Moors Special Protection Area (SPMSPA) phase 2 is a 2,800km^2 area of upland habitat located immediately north of the Peak District National Park (SPMSPA Phase 1) and south of the Yorkshire Dales National Park (Fig. 2.1). The industrial conurbations of Bradford and Huddersfield are in close proximity to the east of the SPA, with the large industrial areas of Greater Manchester and Lancashire situated to the west. In addition, a number of medium to large towns are in close proximity including Halifax, Keighley and Burnley. The SPMSPA phase 2 falls within the jurisdictional boundaries of eight local authorities, of which three encompass the target area for this project - the unitary authorities of Bradford, Calderdale and Kirklees. The joint boundaries of the SPMSPA and the Peak District Moors SPA align with the boundary of the South Pennine Moors SAC.

The SPMSPA phase 2 is one of 269 SPAs in the UK. It contains a landscape mosaic of remote but expansive upland moorland habitats including blanket bog, wet heath, dry heath, grassland and oak woodland (JNCC, 2011). The landscape of the SPA hosts a diverse range of historical and contemporary land-uses associated with northern English upland moorland habitats, including rough grazing of livestock, management for grouse shoots (vegetation burning and predator control), water reservoirs, recreational routes, transport routes and renewable energy production (Pearce-Higgins et al., 2007; van der Horst and Toke, 2010; Douglas et al., 2014). SPMSPA habitats provide breeding and foraging grounds for an internationally important assemblage of upland bird species including Curlew *Numenius arquata*, Short-eared Owl *Asio flammeus*, Merlin *Falco columbarius*, Golden Plover *Pluvialis apricaria*, Wheatear *Oenanthe oenanthe*, Ring Ouzel *Turdus torquatus*, Whinchat *Saxicola rubetra*, Lapwing *Vanellus vanellus* and Twite *Carduelis flavirostris* (JNCC, 2006). The immediate 1km fringe of the SPMSPA phase 2, constrained by the boundaries of Kirklees, Bradford and Calderdale unitary authorities are of primary interest to this project due to the development pressures faced by the three unitary authorities. Due to the close proximity and shared SAC status of the SPMSPA phase 1 and the SPMSPA phase 2, a portion of the SPMSPA phase 1 fringe within Kirklees unitary authority is included as part of the study site.

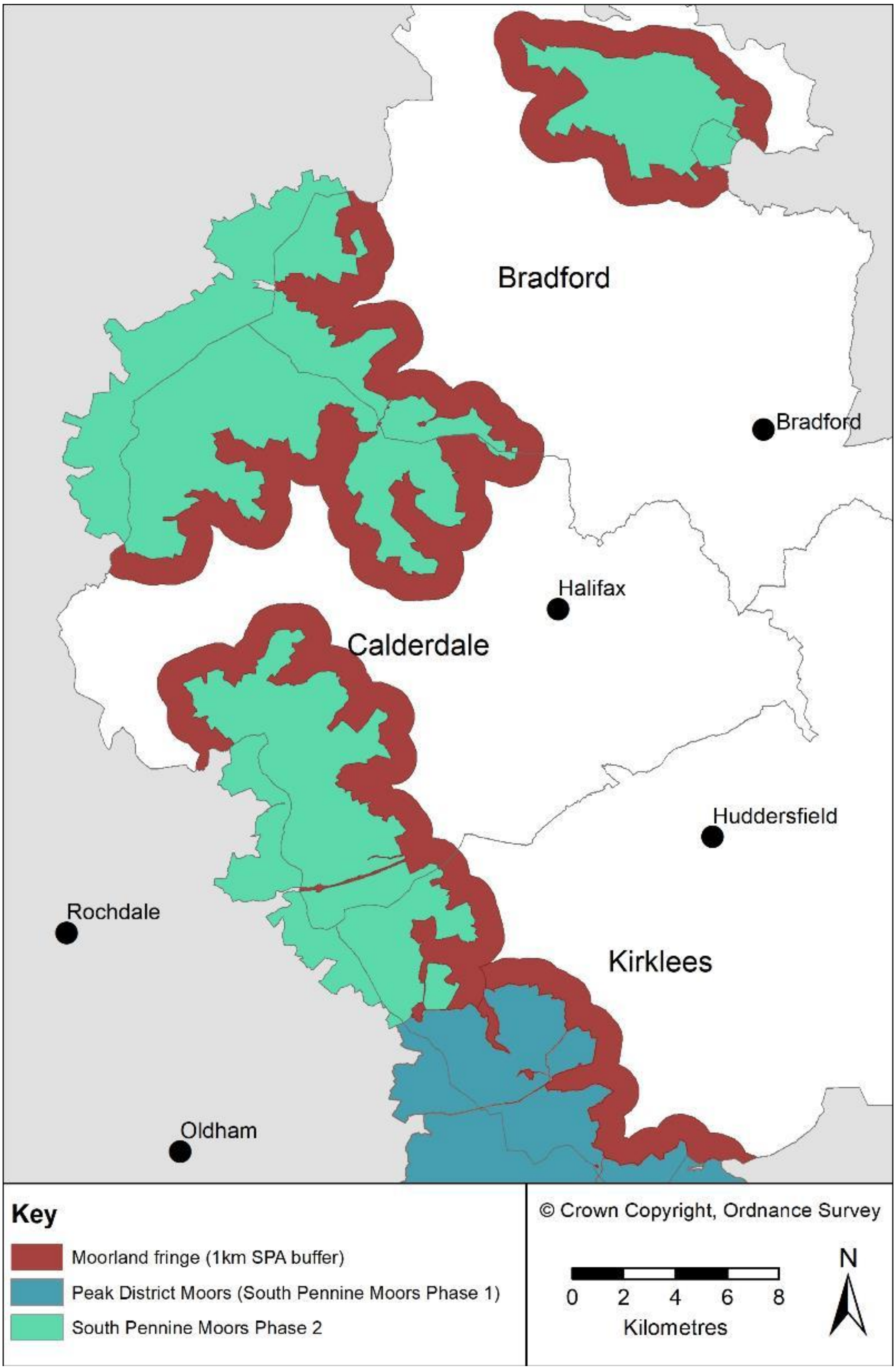

Figure 2.1 The SPMSPA and the extent of the 1km fringe study site constrained by the boundaries of Calderdale, Kirklees and Bradford unitary authorities.

2.3.2. Habitat surveys

Habitat surveys were undertaken in the SPMSPA fringe between 2012 and 2013. Kirklees and Calderdale authority moorland fringe areas (hereafter referred to as 'Kirklees' and

‘Calderdale’) were surveyed by staff from the ecological consultancy West Yorkshire Ecology, in July-September 2012. Bradford authority moorland fringe areas (hereafter referred to as ‘Bradford’) were surveyed by staff from the Urban Edge Environmental Consulting company and also West Yorkshire Ecology in June-July 2013. All surveys were conducted by recorders who were familiar with the habitats and the survey method. The survey undertaken in Bradford 2013 followed a different methodology (see below) and formed part of a separate project by a third party. All habitat surveys were undertaken at the field level, utilising landscape boundaries such as walls, fences, roads and paths as habitat unit divisions. In 2012, habitats were surveyed from line transects that were used for bird surveys in the same year. A total of 88 km of line transect was surveyed for habitat in Calderdale 2012 and 44 km of line transect was surveyed for habitat in Kirklees 2012. A similar line transect methodology was employed in 2013, however the data was supplemented with habitats inferred from aerial imagery and visits to fields that did not lie along line transects. Data are not available on the length of the line transects used in 2013.

The 2012 habitat surveys were conducted only in the Calderdale and Kirklees authority areas. Surveys were conducted during the late breeding bird season or immediately after the breeding bird season in 2012. Surveys were conducted only during conditions of good visibility whilst walking along line transects that were used for bird surveys previously the same year. Line transects sites were established by West Yorkshire Ecology in consultation with the three unitary authorities. The dominant habitat in each field (>75% cover) was classified according to a system developed by West Yorkshire Ecology (Appendix 1), based on the British Trust for Ornithology (BTO) Breeding Bird Survey (BBS) and Defra Environmental Stewardship guidance for improved and semi-improved grassland identification (Defra, 2005). Surveys were conducted primarily within the 1km fringe of the SPMSPA phase 2, however in the case of Kirklees, surveys extended to the 1km fringe of the Peak District Moors SPA (SPMSPA phase 1). Fields that lay completely outside of the 1km SPA fringe and fields that intersected the SPMSPA boundary were excluded from analyses.

Bradford habitat surveys in 2013 were undertaken by Urban Edge Consulting in association with West Yorkshire Ecology and under the guidance of Bradford unitary authority (Urban Edge Consulting, 2014). Surveys were undertaken at the individual field level, within 2.5km of the SPMSPA and within 1km of settlements. Survey sites were selected after consultation with the unitary authority, and were located in areas identified for potential development and in areas that were associated with perceived established patterns of key SPMSPA bird species occurrence. Habitat categories were designed to complement the 2012 Calderdale and Kirklees habitat surveys and in some cases, were directly comparable to the surveys of 2012. However, some of the habitat category definitions in 2013 differed to those of 2012 (Appendix 2).

2.3.3. LCM, Mastermap and elevation data

Land Cover Maps (LCM) covering the study site were downloaded from EDINA Digimap (University of Edinburgh, 2015). These datasets were produced by the Centre for Ecology and Hydrology (CEH) and represent land cover data for three years, 1990 (Fuller, 1995), 2000 (RM Fuller et al., 2002), and 2007 (Morton et al., 2011). Each dataset consisted of a raster at 25m x 25m resolution representing discrete habitat categories, created by the classification of satellite imagery. Habitat categories differed between years, both in number of categories (25 for 1990, 26 for 2000 and 23 for 2007) and in description of classes. The latest Ordnance Survey Mastermap dataset, from December 2014, and covering the study site was obtained from EDINA Digimap (University of Edinburgh, 2015). These data were vector datasets available as 10km x 10km parcels referenced to the GB national grid. Fourteen parcels were selected to cover the study area: SD82, SD83, SD90, SD91, SD92, SD93, SD94, SE00, SE01, SE02, SE03, SE04, SE10 and SE14. The parcels were merged and duplicate features removed using ArcGIS (ESRI, 2014). Buildings were extracted and cropped to the study area. Elevation data were obtained in the form of the Ordnance survey Terrain 5 DTM dataset for the study area was obtained from EDINA digimap (Ordnance Survey, 2015). This dataset is a 5m x 5m resolution raster containing an elevation value in metres.

2.3.4. Landsat 8 data collection

Landsat 8 satellite imagery was freely available through Google Earth Engine (GEE) from the US Geological Survey (USGS), with images regularly taken over the SPMSPA fringe. A set of four Landsat 8 composite images were created using the tools available in GEE, representing the four British seasonal periods, spring (March-May), summer (June-August), autumn (September - November) and winter (December-February. Initial inspection of images revealed that the study area was subject to high year-round cloud cover, resulting in large areas of unusable imagery in seasonal composites from a single year. To resolve this issue, seasonal composites were built using Landsat 8 imagery taken over a three-year period (2013, 2014 and 2015). A total of 148 Landsat 8 images were used to make seasonal composites with 31 images representing winter, 34 representing spring, 47 representing summer and 36 representing autumn. Prior to compositing, a cloud mask was applied to each of the 148 images and the images were cropped to the 1km SPMSPA buffer. In addition, permanent waterbodies were masked using the waterbody mask described by Hansen et al. (2013). The Landsat 8 images each consisted of 11 surface reflectance spectral bands representing wavelength ranges of electromagnetic reflection from the surface of the earth at a resolution of 30m x 30m (Table 2.1). The median value (per seasonal period) of individual Landsat 8 spectral bands was taken at each pixel location using the portions remaining from each image after cloud masking. The values for each spectral band from each seasonal composite were subsequently normalised. The Landsat 8 images used belonged to the Standard Terrain Collection (L1T) and had been pre-processed by USGS into topographically and radio-metrically corrected surface reflectance. These images represent the highest quality post-processed Landsat 8 imagery available from USGS and did not require any further post processing (USGS, 2017).

Table 2.1 Bands associated with Landsat 8 imagery. Each band in sensitive to a different wavelength of the electromagnetic spectrum (Roy et al., 2014).

Band identifier and colour sensitivity	Wavelength (μm)
Band 1, Ultra blue (coastal/aerosol)	0.43-0.45
Band 2, Blue	0.45-0.51
Band 3, Green	0.53-0.59
Band 4, Red	0.64-0.67
Band 5, Near Infrared (NIR)	0.85-0.88
Band 6, Shortwave Infrared 1 (SWIR1)	1.57-1.65
Band 7, Shortwave Infrared 2 (SWIR2)	2.11-2.29
Band 8, Panchromatic	0.50-0.68
Band 9, Cirrus	1.36-1.38
Band 10, Thermal Infrared 1 (TIRS1)	10.6-11.19
Band 11, Thermal Infrared 2 (TIRS2)	11.50-12.51

2.3.5. Habitat survey data analysis

Descriptive statistics for dominant habitats within the 1km SPMSPA fringe were calculated for each unitary authority and for all unitary authorities combined. Fields that were located partially within the SPA and fields that were completely outside the 1km fringe were excluded.

2.3.6. LCM data analyses

Land Cover Map datasets for the years 1990, 2000 and 2007 were each produced using a different remote sensing methodology, representing technological and software improvements over time. As a result, the spatial accuracy of habitat boundaries was different between years. This makes the datasets useful for habitat comparisons between different areas of the UK in the same year, but less suitable for the analysis of temporal change. This is compounded by the fact that that the habitat classification categories differ between years, creating further problems in making temporal comparisons. In order to standardise the datasets and allow temporal habitat patterns within the SPA fringe to be explored, a two-stage process was employed.

The first stage of the process was designed to overcome mismatches in habitat classification by reclassifying the habitats in each year into two broad categories that were consistent from year to year. The mean of the number of 25m x 25m pixels for each habitat type that fell within the SPMSPA phase 2 were calculated for each year of LCM data. Habitat types that were greater than the mean for a given year were deemed to be habitat characteristic of the SPMSPA phase 2. Pixels representing these habitats occurring within the SPA, within the SPA 1km fringe and within surrounding areas, were reclassified as 'typical of SPMSPA phase 2 habitat'. All other habitats were reclassified as 'not typical of SPMSPA phase 2 habitat'. The second stage of the process was designed to overcome differences in the spatial accuracy of habitat boundaries between years arising from differences in the remote sensing methodologies used by CEH. As CEHs methods were consistent within each year of Landcover data production, the ratio of pixels typical of SPMSPA phase 2 habitat within the SPMSPA phase 2 boundary to the pixels typical of SPMSPA phase 2 habitat within the SPA 1km fringe should be consistent between years. This allowed the coverage of reclassified habitat within the SPA 1km fringe as a proportion of reclassified habitat within the SPMSPA phase 2 to be compared between the three years.

The three reclassified LCM datasets were drawn as maps in ArcGIS and compared for temporal change. The ratios of habitat typical of SPMSPA phase 2 within the 1km fringe to the same habitat within the SPMSPA phase 2 were plotted and compared for each individual unitary authority and for all unitary authorities combined. As the 2007 landcover dataset was the most recent of the three datasets and represented the most advanced dataset in terms of the spatial accuracy of habitat boundaries, it was used to detect clustering of habitat typical of SPMSPA phase 2 within the 1km SPA fringe. This was achieved using zonal statistics within ArcGIS (see Fig. 2.2). Each pixel classed as 'habitat typical of the SPMPSA phase 2' was assigned a value of 1, and every other pixel was assigned a value of 0. The sum of pixels within a 100m distance of each individual pixel was calculated and colour coded on a scale of green to amber to red based on the magnitude of the summed value. Although pixels outside of the 1km SPA fringe were incorporated into this analysis to account for neighbour habitat, the results were cropped to the 1km SPA fringe.

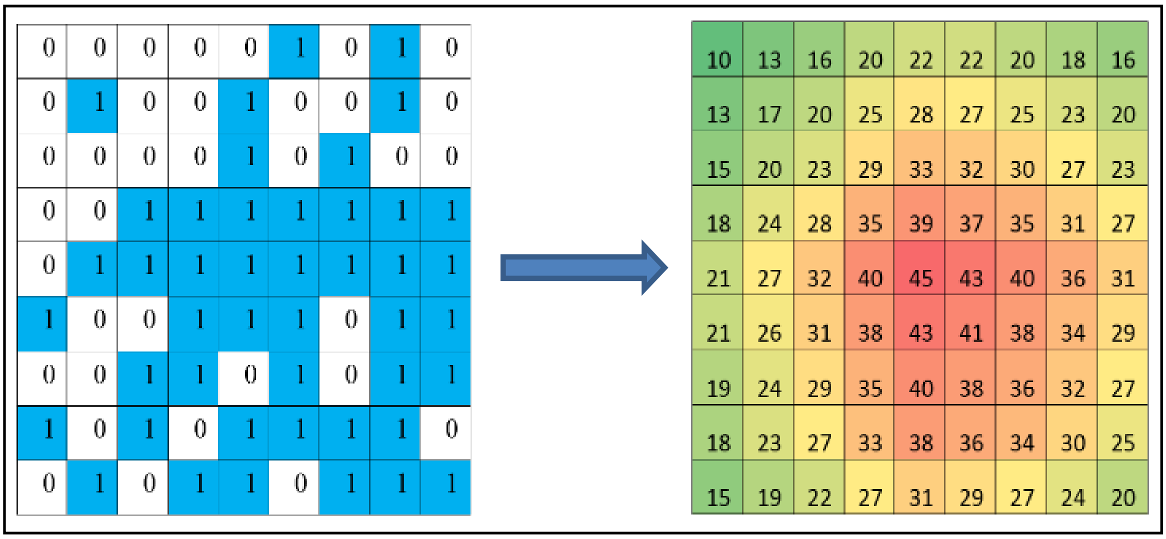

Figure 2.2 Illustration of the concept of focal statistics used to assess upland habitat clustering.

2.3.7. Gradient analysis

Ten buffers of 100m width were calculated outside of the SPMSPA boundary between 0m to 1,000m, and cropped to the boundaries of the unitary authorities in ArcGIS. The resultant ten distance bands were used to separate a number of developmental and landscape features, allowing patterns to be studied as a function of distance from the SPA boundary, here referred to as gradient analysis. The developmental features chosen were quantity of buildings and building density. Gradient analysis using these distance bands was also undertaken to observe patterns of 'habitat typical of the SPMSPA phase 2' within the 1km SPA fringe using the reclassed 2007 CEH landcover data and elevation patterns using the OS Terrain 5 DTM dataset.

Number of buildings and building density were calculated in ArcGIS using the OS Mastermap dataset. Building centroids were calculated and used to determine absolute number of buildings in each SPA distance band. Thiessen polygons were calculated from the building centroids with the area of the thiessen polygon associated with each building assigned to its respective building centroid. These values were inversely proportional to the proximity of a building to all other immediate surrounding buildings, creating a measure of density per building. This approach was favoured over simply calculating the density of building centroids within a distance band, because it allowed the mean density of individual buildings within a distance band to be calculated whilst taking into account the proximity of surrounding buildings that may fall into neighbouring distance bands.

2.3.8. Landsat 8 image Classification

The four seasonal Landsat 8 composite images were exported from GEE for analysis in R (R Core Team, 2013). Cirrus and aerosol/coastal bands were excluded from each composite image and remaining spectral bands (see Table 2.1) from each seasonal composite were compiled into a multiband raster comprising a total of 36 bands. Ordnance Survey Terrain 5 DTM elevation data was added to the multiband raster to improve classification performance. The DTM was resampled from 5m x 5m to 30m x 30m using cubic convolution in ArcGIS. The resultant 37 band raster represented 99.7% of the terrestrial coverage within the 1km SPMSPA fringe, (Fig. 2.3). Permanent waterbodies were excluded from classification.

Fields surveyed that did not have >75% coverage of a single habitat type, were ‘other’ habitats or were were excluded from supervised classification. The habitat type of the remaining fields will be referred to as ‘dominant habitats’. Dominant habitats were converted to point data, using the centroids of individual Landsat pixels that coincided with surveyed fields as extraction points. Habitats represented by less than 350 pixels in the Landsat data were excluded from further analysis due to underrepresentation. The excluded habitats were enclosed acid grassland, amenity grassland, dry dwarf shrub heath and wet heathland/mire leaving six habitat categories for classification (Table 2.2). Mastermap building data were aggregated so that any buildings within 60m of one another were merged into a single polygon and any resultant building polygons that were less than $900m^2$ in area were excluded. This was because $900m^2$ represents a single pixel of Landsat 8 data and therefore anything less than this area cannot be represented reliably at this resolution. These data were not included in classification, but were added to the final classification as a polygon mask. Habitat data points were split into a 50% training set and a 50% validation set data balanced by habitat class, using the caret package in R (Kuhn, 2008; R Core Team, 2013).

Table 2.2 Dominant habitat categories of the fields surveyed in 2012 and 2013 for use in supervised classification. Habitat categories that were underrepresented were excluded from analysis. The number of points represents the total number of pixels represented in the Landsat 8 data used in classification. These points were split into 50% training and 50% testing data.

Habitat category used in classification	Original habitat survey categories		Number of data points
	2012	2013	
Improved Grassland	Amenity Grassland Improved grassland	Amenity Grassland Improved grassland	10,451
Species poor semi improved grassland	Semi-improved, species poor grassland	Semi-improved grassland (species poor)	14,321
Species rich semi improved grassland	Semi-improved grassland	Semi-improved grassland	1,306
Dry heath/ acid grassland mosaic	Dry heath/ acid grassland mosaic	Dry heath/ acid grassland mosaic	1,717
Rush pasture	Rush pasture	Rush pasture	4,986
Wet heathland/ acid grassland mosaic	Wet heathland/ acid grassland mosaic	NA	2,808
NA	Dry dwarf shrub heath	Dry dwarf shrub heath	NA
NA	NA	Upland acidic grassland (enclosed)	NA

Supervised classification of the SPMSPA 1km fringe was undertaken using a Random Forest algorithm within the RStoolbox R package (Leutner and Horning, 2017). All 37 bands of Landsat 8 and elevation data were included as predictor variables. The model was trained using the habitat data training set and the testing set was used to validate the model using a confusion matrix to determine classification accuracy. To balance the habitat classes within the model, 600 data points were randomly selected from each habitat type within the training set and used for model construction. The value of 600 points was chosen as it approximately reflects the number of points available for training in the habitat category with the least data available (Species rich semi improved grassland. The Random Forest was constrained to 500 trees to avoid excessive computational time. A confusion matrix was constructed using the testing set and overall accuracy was used as a measure of model performance. Maps were subsequently produced using the random forest model to predict habitat categories in the entire SPMSPA 1km fringe.

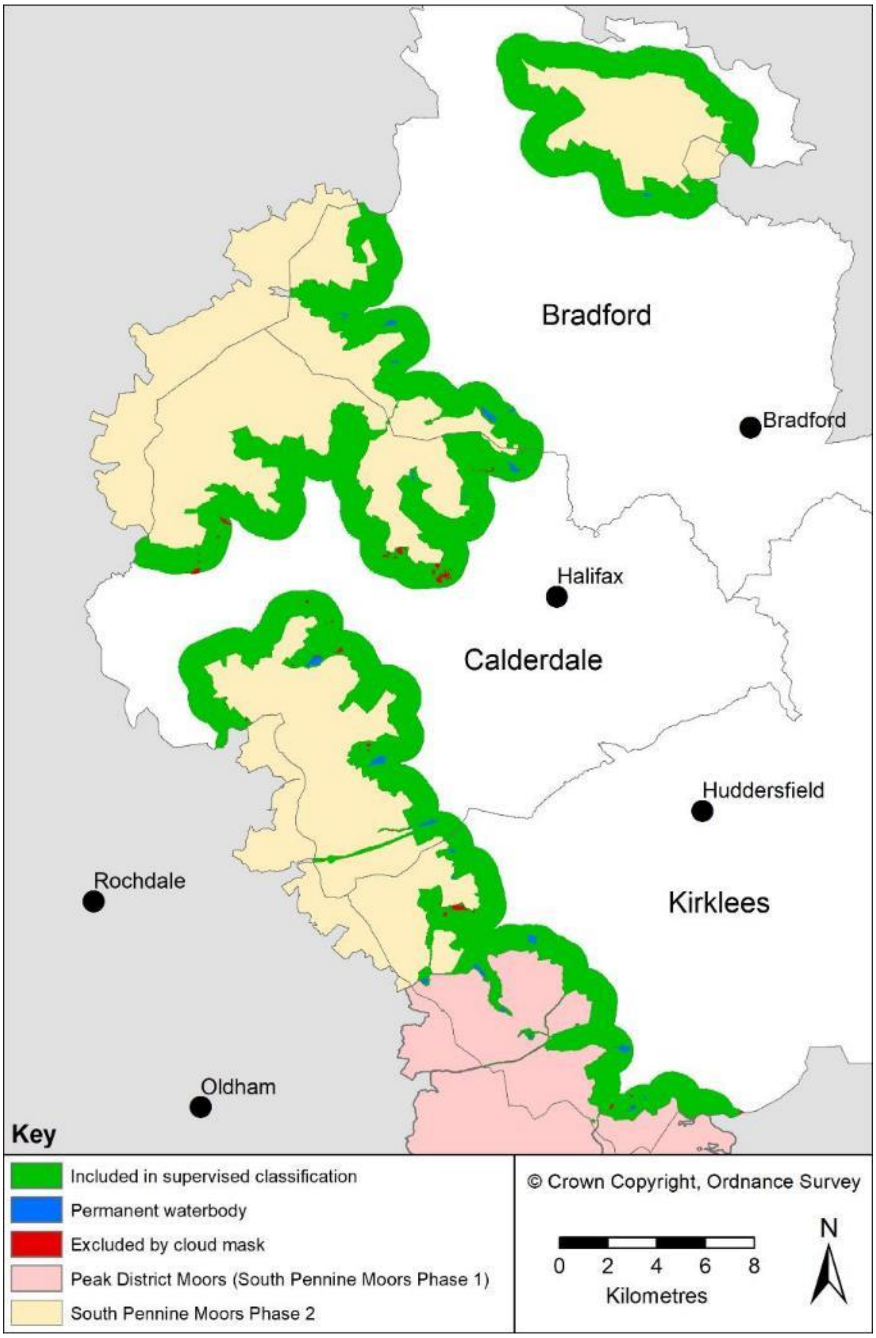

Figure 2.3 The area within the South Pennine Moors Special Protection Area 1km fringe where Landsat 8 imagery was available for supervised classification. Permanent waterbodies were excluded from analysis.

2.4. Results

2.4.1. Composition and extent of habitats within the SPMSPA fringe

A total of 548 fields were surveyed in Kirklees 2012 (Fig. 2.4), 889 in Calderdale 2012 (Fig. 2.5) and 1,504 in Bradford 2013 (Fig. 2.6). Following data cleaning, fields available for analysis (i.e. those with a dominant habitat of >75% coverage per field in 2012 and 2013 within the 1km fringe but not extending into the SPMSPA) totalled 360 in Kirklees 2012, 613 in Calderdale 2012, and 311 in Bradford 2012. In order to describe the habitats exclusively within the SPA fringe, fields that extended into the SPMSPA and fields that lay completely outside of the SPMSPA 1km fringe were removed, reducing the dataset to 360 fields in Kirklees 2012, 613 fields in Calderdale 2012 and 326 in Bradford 2013.

The fields surveyed within the SPMSPA 1km fringe consisted of a highly heterogeneous landscape mosaic, with fields of similar habitat within the 1km fringe generally appearing to be locally clustered (Figs 2.7-2.9). Fourteen dominant habitat types were recorded which varied in relative proportions between the three unitary authority areas. Of these 14 habitat types, nine were present across all three authority areas. Three of the habitats surveyed in Bradford 2013 had no directly comparable habitat category surveyed in Calderdale and Kirklees 2012, with only one habitat surveyed in Calderdale and Kirklees 2012 had no directly comparable habitat category in Bradford 2013 (Table 2.4). Nine dominant habitats types were recorded in Kirklees, 10 in Calderdale and 11 in Bradford (Table 2.5). Semi-improved (species poor) grassland made up the largest proportion of surveyed fringe habitat in Calderdale (35.4%), in Bradford (53.6%), across all three authorities combined (37.6%) and was the second most commonly encountered habitat in Kirklees (25.4%). Improved grassland was the most commonly encountered habitat in Kirklees (35.4%) and was the second most commonly encountered habitat in Calderdale (23.6%) and Bradford (20.6%). Between all three authority areas, improved grassland and semi-improved species poor grassland combined accounted for 63.5% of all habitat surveyed. Dry dwarf shrub heath had the lowest coverage of any habitat encountered in Kirklees whereas semi-improved grassland had the lowest coverage in Calderdale and dry heath/ acid grassland mosaic had the lowest coverage in Bradford. All habitats encountered in Kirklees were found in at least one other authority area.

Species rich semi-improved grassland/ unimproved grassland was surveyed in all three authority areas but was absent from Calderdale and Kirklees and only recorded as a dominant habitat in Bradford in two fields (Table 2.5). Wet upland habitats were rarely encountered in Kirklees where only a single field of wet heathland/ acid grassland was recorded. This habitat was relatively common in Calderdale, making up 17.47% of the total area surveyed in the Calderdale fringe. Wet heathland/ mire was absent in Kirklees and had low coverage in Calderdale. Blanket bog was not found in any authority area. Wet upland habitats were not recorded in Bradford; however it is unclear from the survey methodology whether these habitats were not encountered or simply not surveyed (Urban Edge Consulting, 2014). There was noticeable variation in the mean field areas and coverage of some habitats between authorities (Table 2.5). Generally, mean field area within the fringe was greatest in Bradford and lowest in Kirklees. The mean area of improved grassland fields in Bradford was almost twice that of those in Calderdale or Kirklees, however amenity grassland mean field size was much larger in Kirklees than in Calderdale or Bradford. Semi-improved grassland coverage was low in Kirklees and Calderdale but relatively high in Bradford. Dry heath/ acid grassland mosaic was far more commonly encountered in Kirklees than in Calderdale or Bradford and in larger fields. Rush pasture made up a large proportion of the total habitat surveyed in Kirklees and Bradford but was rarely encountered in Bradford.

Table 2.3 Comparison of the habitat categories recorded in 2012 and 2013. Equivalent habitats between the two years are shown side by side.

Kirklees and Calderdale 2012	Bradford 2013
Amenity grassland	Amenity grassland
Improved grassland	Improved grassland
Semi-improved, species poor grassland	Semi- improved grassland (species poor)
Semi-improved grassland	Semi-improved grassland
Species rich/ Unimproved grassland	Species rich semi-improved grassland/ Unimproved grassland
No directly comparable habitat	Rough grassland
No directly comparable habitat: no distinction in pH was made	Upland acidic grassland (enclosed)
Dry dwarf shrub heath	Dry dwarf shrub heath
Dry heath/acid grassland mosaic	Dry heath/ acid grassland mosaic
Blanket bog/ mire	No directly comparable habitat
Wet heathland/ mire	No directly comparable habitat
Wet heathland/ acid grassland mosaic	No directly comparable habitat
Rush pasture	Rush pasture
Other	Other

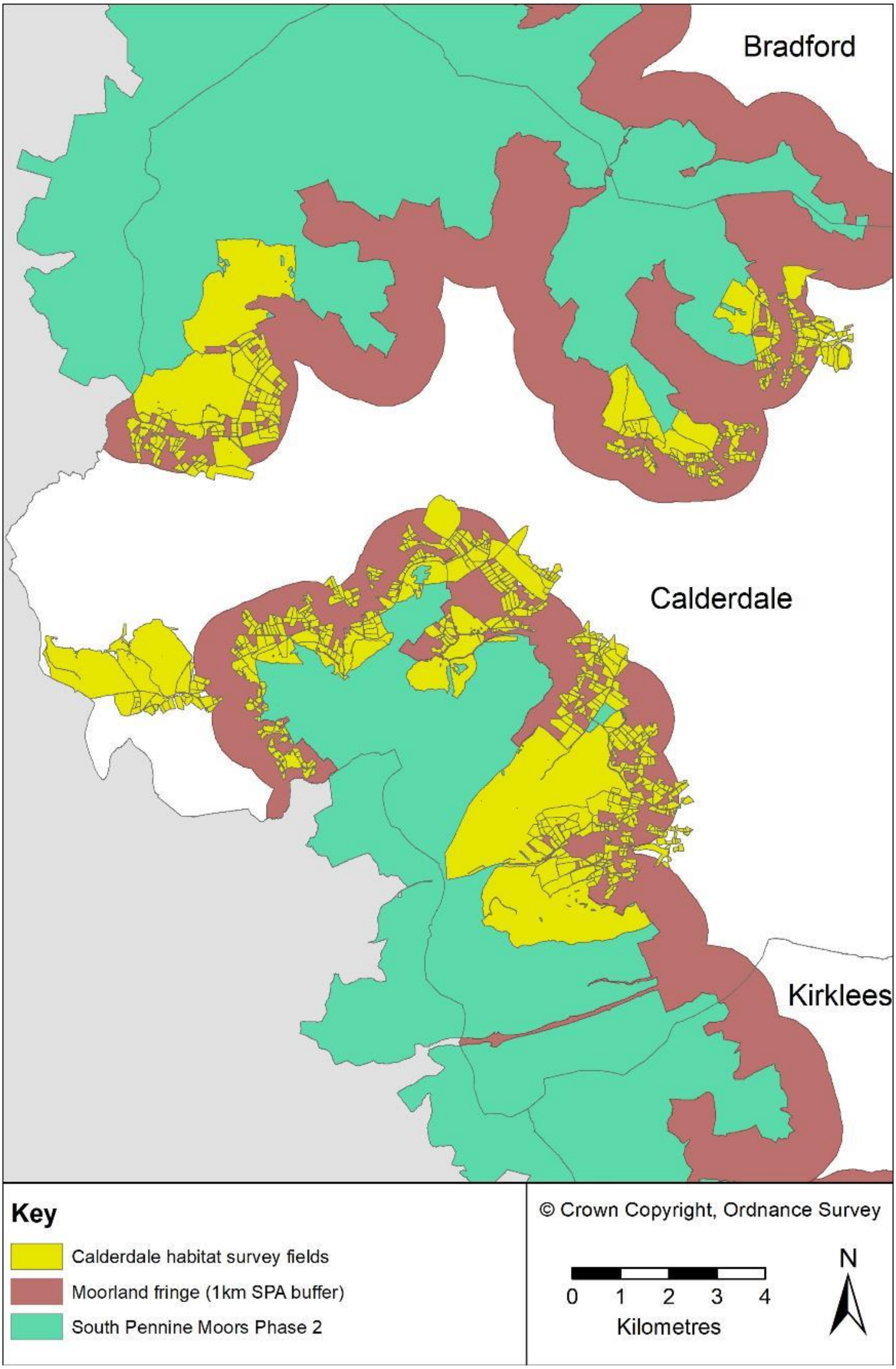

Figure 2.4 Locations of fields included in habitat surveys conducted Calderdale 2012, concentrating on the South Pennine Moors Special Protection Area 1km fringe.

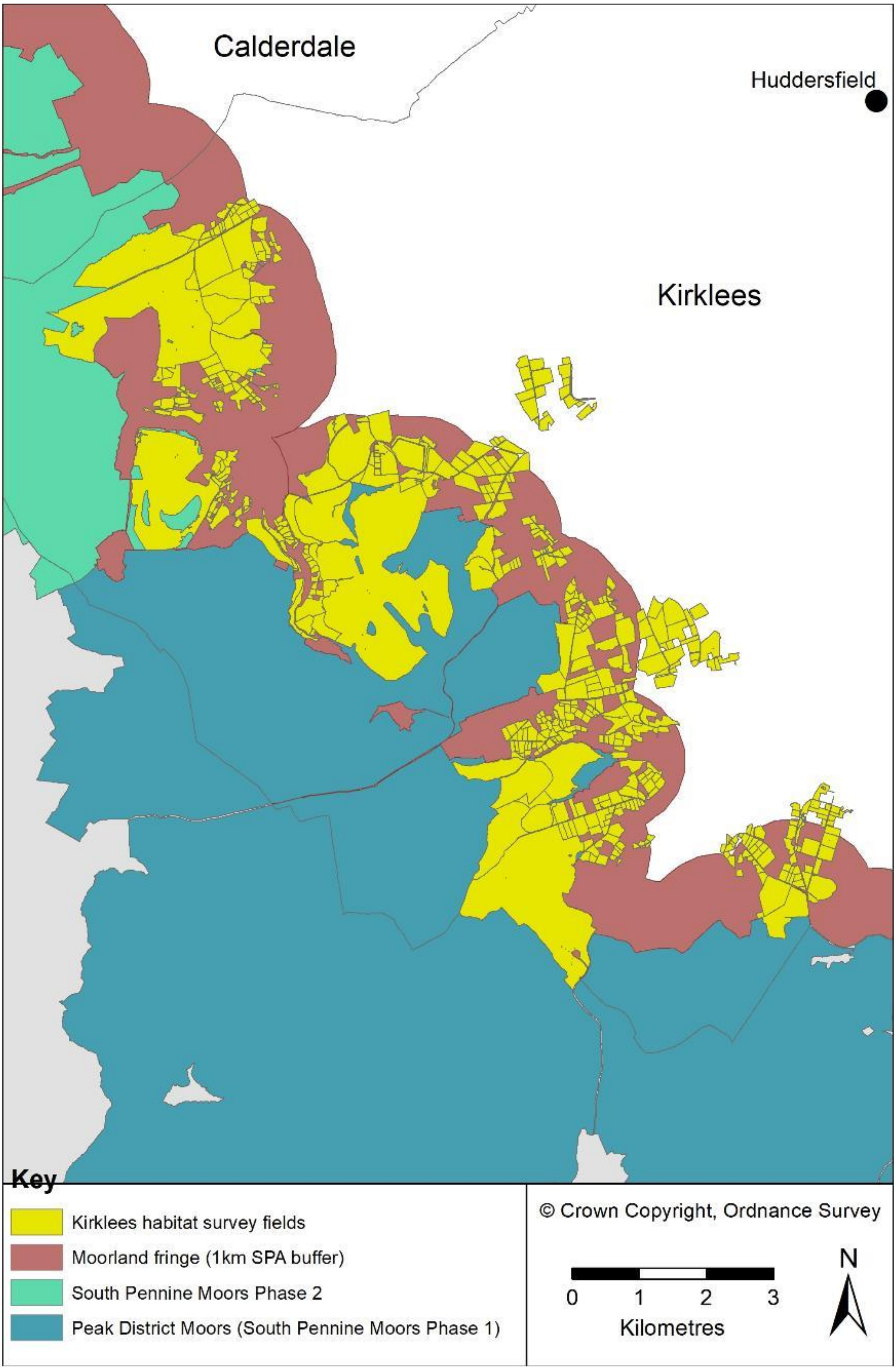

Figure 2.5 Locations of fields included in habitat surveys conducted in Kirklees 2012, concentrating on the South Pennine Moors Special Protection Area 1km fringe.

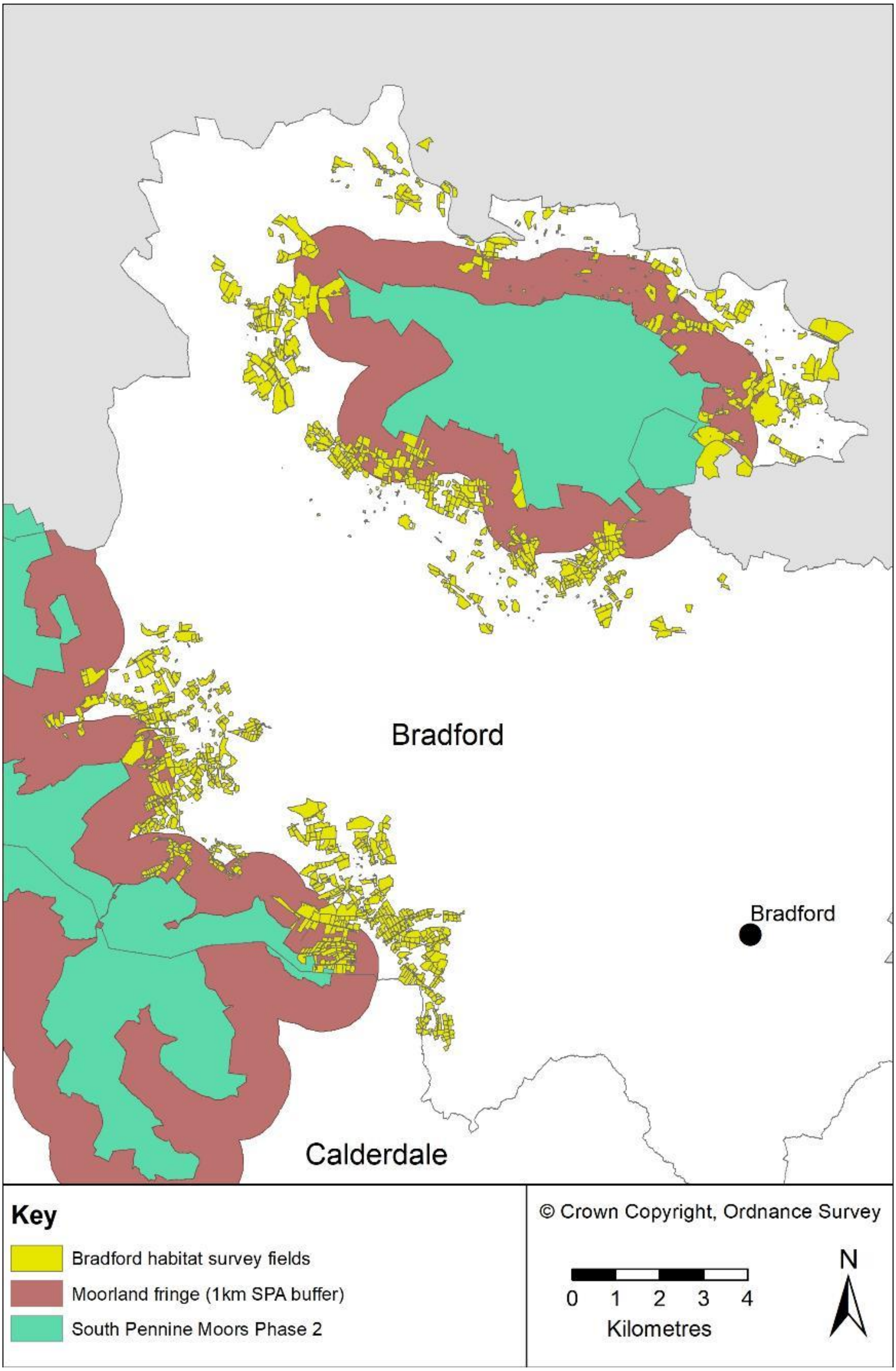

Figure 2.6 Location of fields included in habitat surveys conducted in Bradford 2013, extending to 2.5km from the South Pennine Moors Special Protection Area boundary.

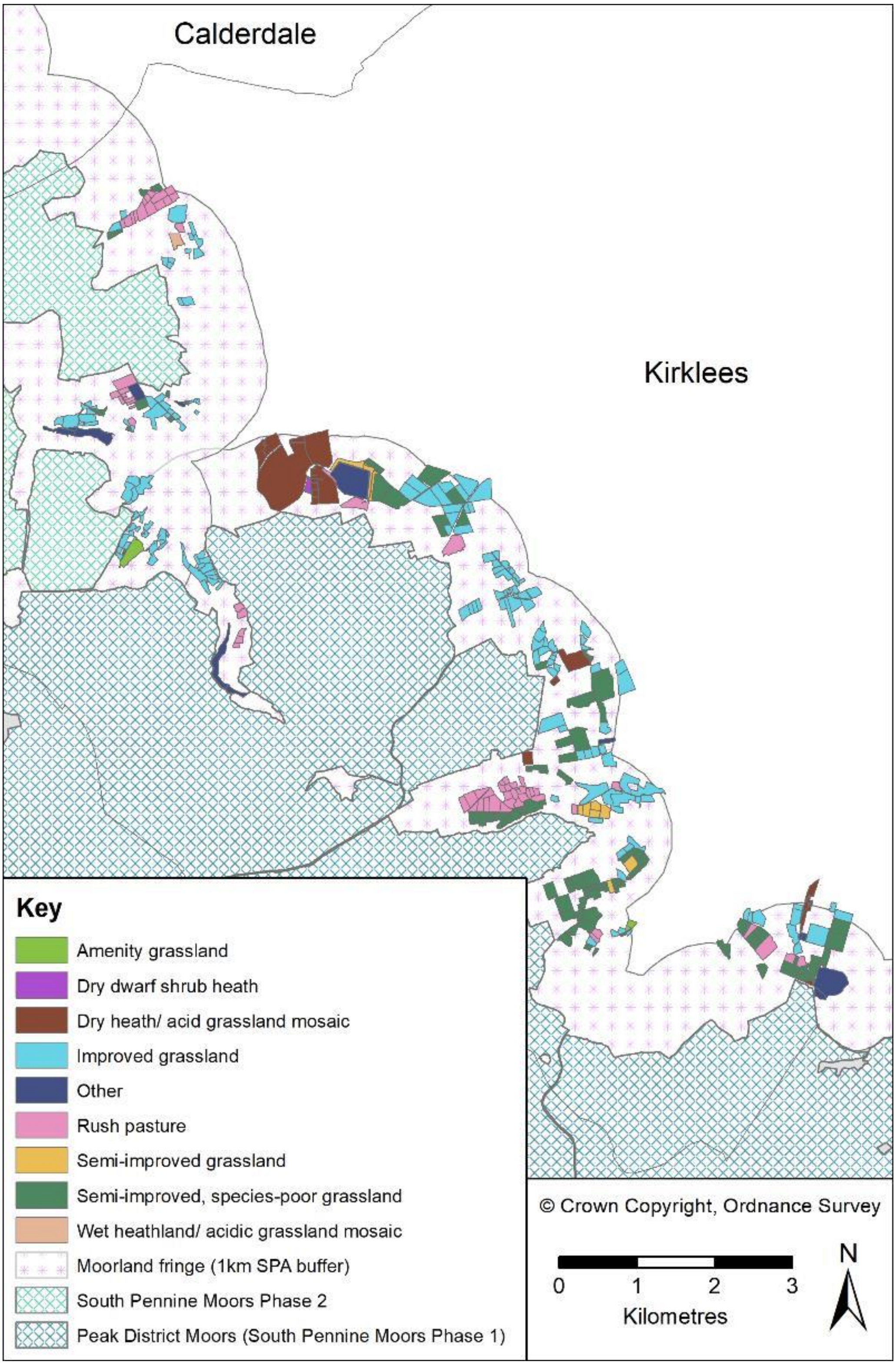

Figure 2.7 Distribution of fields with dominant habitats (>75% in a single field) surveyed in the Kirklees South Pennine Moors Special Protection Area 1km fringe in 2012.

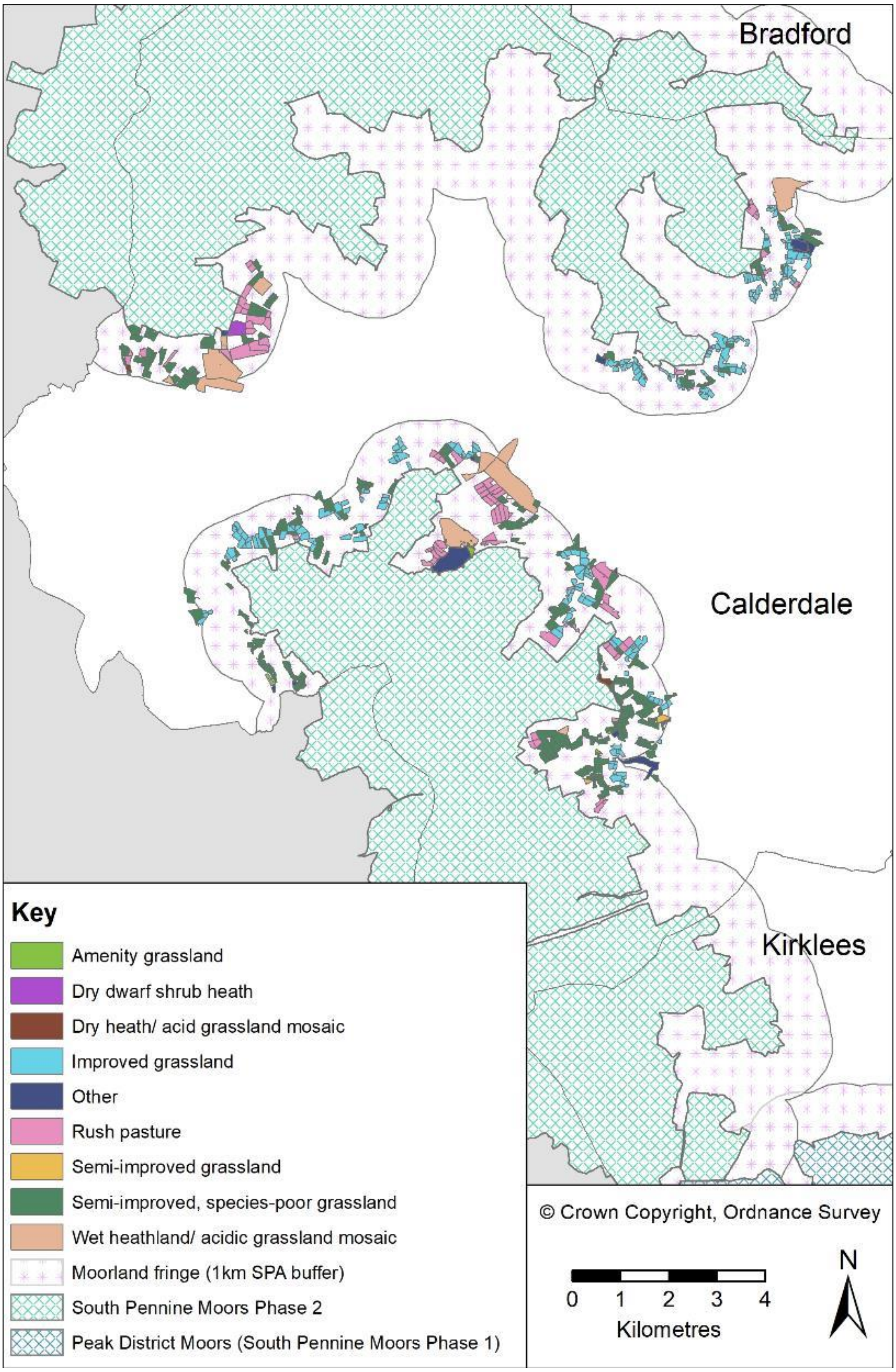

Figure 2.8 Distribution of fields with dominant habitats (>75% in a single field) surveyed in the Calderdale South Pennine Moors Special Protection Area 1km fringe in 2012.

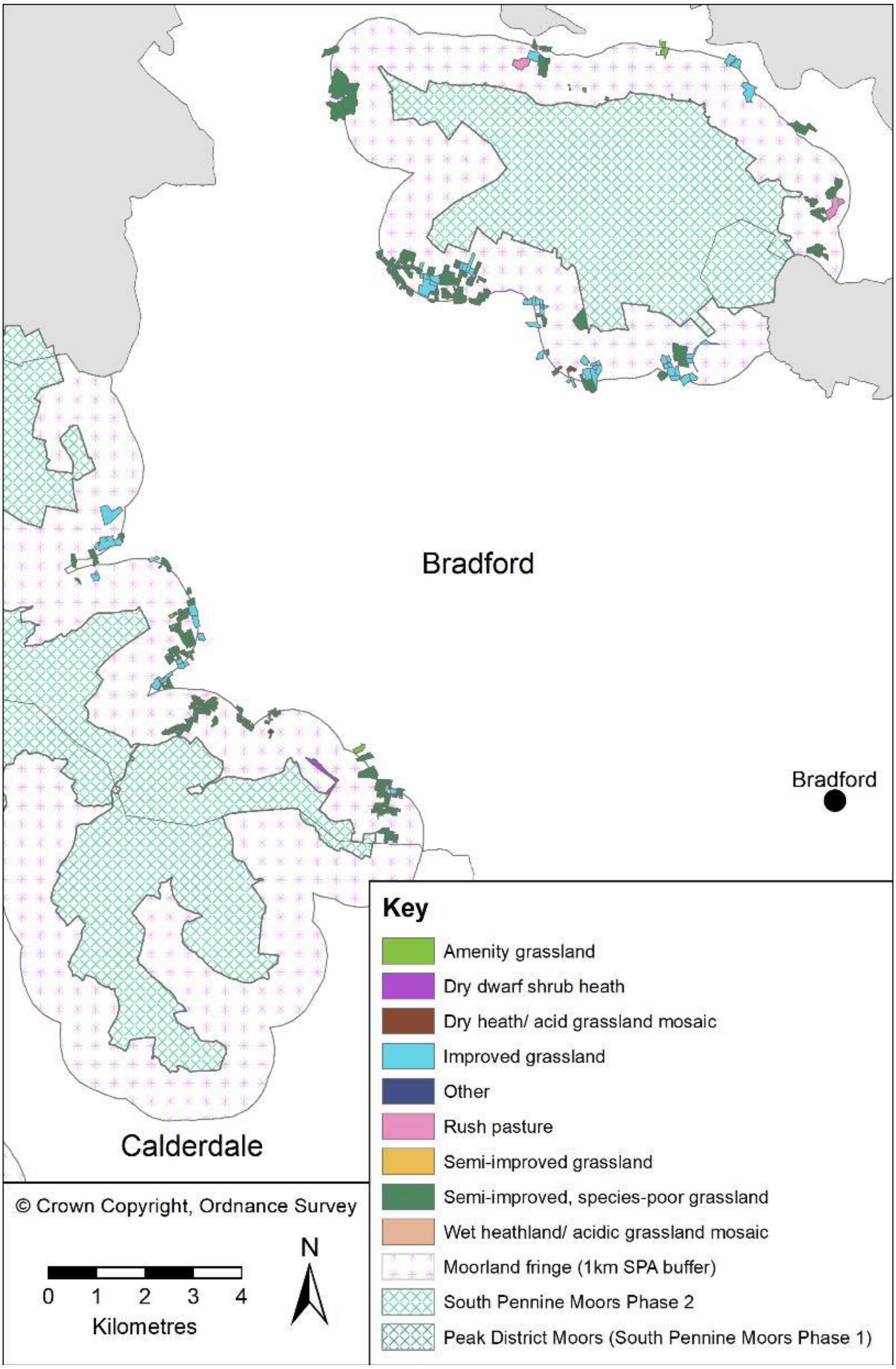

Figure 2.9 Distribution of fields with dominant habitats (>75% in a single field) surveyed in the Bradford South Pennine Moors Special Protection Area 1km fringe in 2013.

Table 2.4 Summary of the Special Protection Area 1km fringe habitats surveyed in Calderdale and Kirklees 2012 and Bradford 2013. Only fields that had >75% coverage of a single habitat type were included. Fields that extended into the South Pennine Moors Special Protection Area were not included. Habitats surveyed across all three authority areas are shaded in green. Habitat category follows terminology of the Bradford 2013 survey (see Methods). NP = not present, ND = no direct comparable habitat surveyed.

	Kirklees				Calderdale				Bradford				All three authorities			
Habitat	Number of fields	Mean field size (ha)	Total area (ha)	% of total area	Number of fields	Mean field size (ha)	Total area (ha)	% of total area	Number of fields	Mean field size (ha)	Total area (ha)	% of total area	Number of fields	Mean field size (ha)	Total area (ha)	% of total area
Amenity grassland	3	2.1	6.4	1.0	7	0.8	5.7	0.5	12	0.7	8.0	1.3	22	0.9	20.0	0.9
Improved grassland	163	1.4	219.5	35.4	228	1.2	267.9	23.6	58	2.2	125.9	21.1	449	1.4	613.3	26.1
Semi- improved grassland (species poor)	97	1.6	157.2	25.4	265	1.5	401.3	35.4	184	1.7	311.6	52.2	546	1.6	870.1	37.1
Semi-improved grassland	9	1.8	16.5	2.7	2	2.0	4.0	0.4	18	4.5	80.4	13.5	29	3.5	101.0	4.3
Species rich semi-improved grassland/ Unimproved grassland	NP	NP	NP	NP	NP	NP	NP	NP	2	2.9	5.7	0.9	2	2.9	5.7	0.2
Rough grassland	ND	ND	ND	ND	ND	ND	ND	ND	4	1.8	7.2	1.2	4	1.8	7.2	0.3
Upland acidic grassland (enclosed)	ND	ND	ND	ND	ND	ND	ND	ND	9	1.5	13.1	2.2	9	1.5	13.1	0.6
Dry dwarf shrub heath	1	1.3	1.3	0.2	2	4.8	9.7	0.9	1	7.1	7.1	1.2	4	4.5	18.1	0.8
Dry heath/ acid grassland mosaic	8	11.1	88.7	14.3	2	2.3	4.5	0.4	2	1.2	2.4	0.4	12	8.0	95.6	4.1
Blanket bog/ mire	NP	NP	NP	NP	NP	NP	NP	NP	ND	ND	ND	ND	NP	NP	NP	NP
Wet heathland/ mire	NP	NP	NP	NP	1	5.5	5.5	0.5	ND	ND	ND	ND	1	5.5	5.5	0.2
Wet heathland/ acid grassland mosaic	1	2.4	2.4	0.4	13	15.3	198.3	17.5	ND	ND	ND	ND	14	14.3	200.7	8.5
Rush pasture	66	1.2	78.3	12.6	80	2.3	182.8	16.1	3	4.6	13.7	2.3	149	1.8	274.8	11.7
Other	12	4.1	49.0	7.9	13	4.2	54.9	4.8	18	1.2	22.2	3.7	43	2.9	126.2	5.4
Total	360	1.7	619.4	-	613	1.9	1134.5	-	311	1.9	597.3	-	1284	1.8	2351.2	-

2.4.2. *Extent of SPMSPA Phase 2 habitat within the moorland fringe landscape*

Analysis of the 1990, 2000 and 2007 CEH LCM datasets showed that across all three authority areas in all three years, the SPA fringe contained a proportion of habitat that was representative of SPMSPA Phase 2 habitat and that the spatial distribution of this habitat appeared to differ between years (Figures 2.10a-2.10c). Moorland fringe SPA phase 2 habitat appeared to be locally distributed in 1990 when compared to 2000 and 2007 and more fragmented in 2000 than in the other years. The extent of this habitat increased from 1990 to 2000 within the 1km fringe but subsequently decreased between 2000 and 2007 for each unitary authority area and across all three authorities combine (Fig. 2.11). There were fundamental differences in the spatial distribution of data in 1990 compared to 2000 and 2007 which are likely due to changes in differences in data processing by CEH between years. Although this limits comparability, the standardisation of data into relative proportions of SPSMPA Phase 2 habitat within years allows the comparison of relative areas of SPMSPA Phase 2 habitat between years.

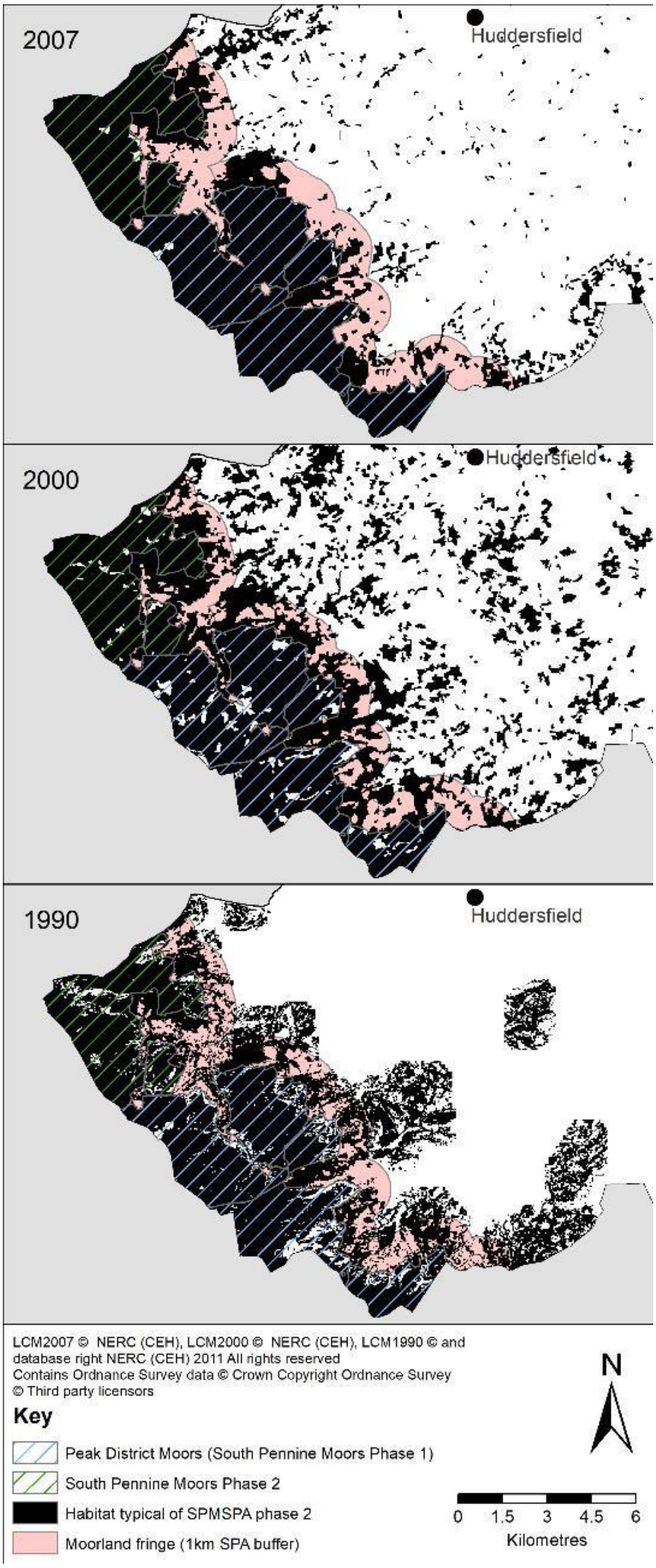

Figure 2.10a The distribution of habitat representative of the South Pennine Moors Special Protection Area phase 2 at 25m x 25m resolution in Calderdale derived from Centre for Ecology and Hydrology (CEH) Landcover Maps (LCM) datasets for the years 1990, 2000 and 2007.

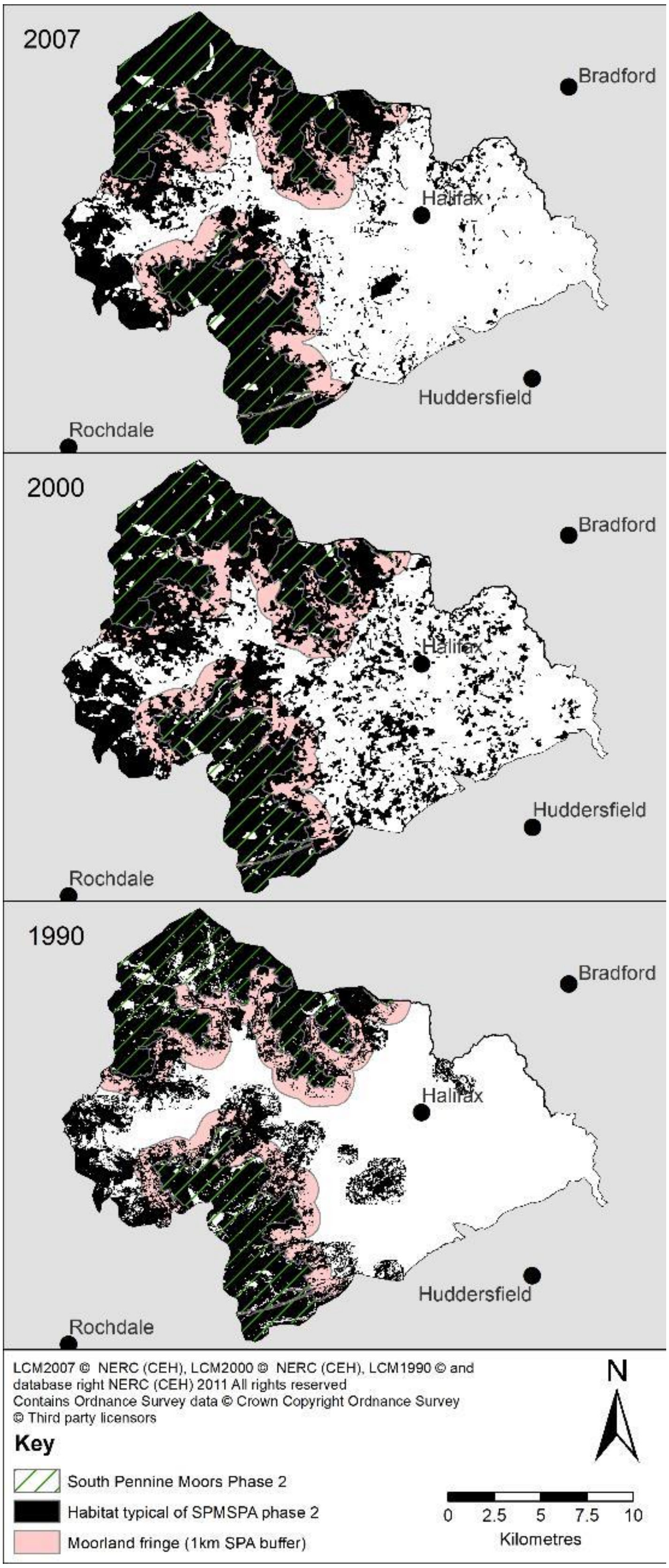

Figure 2.10b The distribution of habitat representative of the South Pennine Moors Special Protection Area phase 2 at 25m x 25m resolution in Calderdale derived from Centre for Ecology and Hydrology (CEH) Landcover Maps (LCM) datasets for the years 1990, 2000 and 2007.

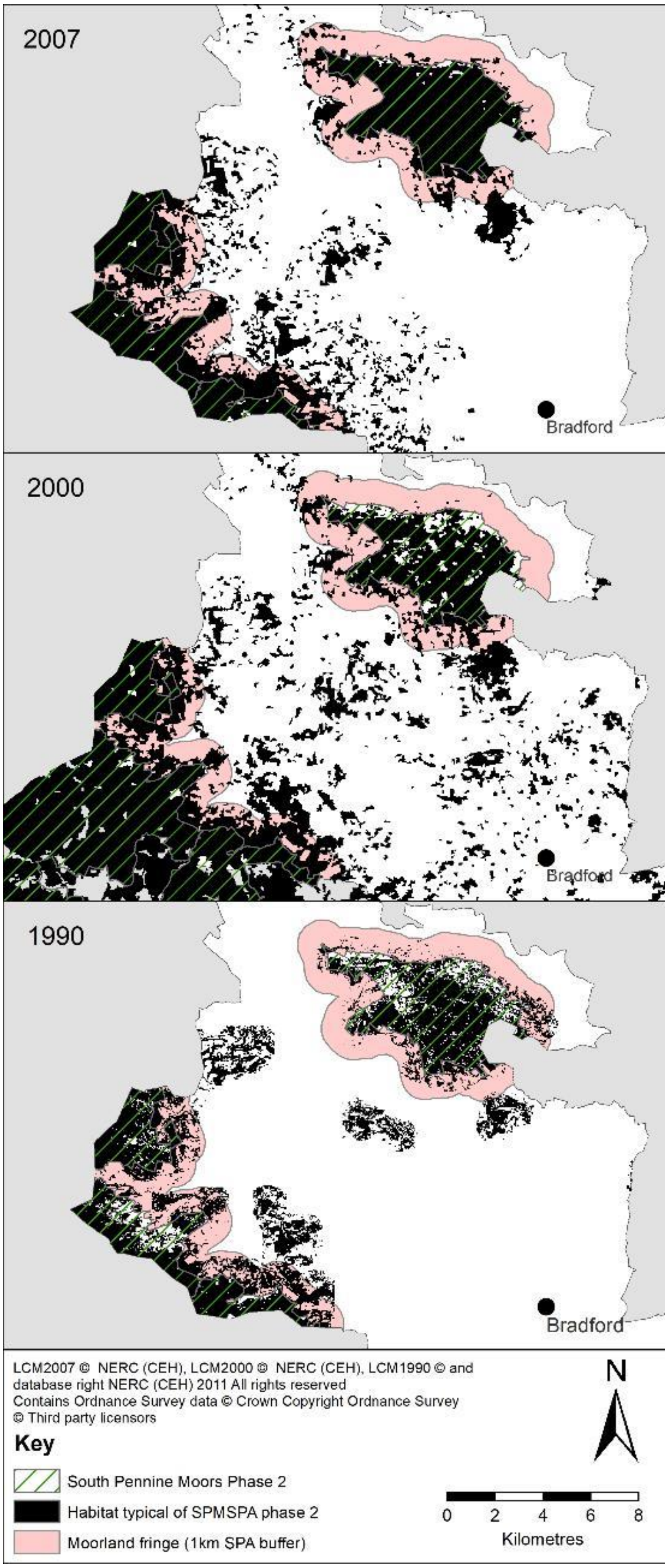

Figure 2.10c The distribution of habitat representative of the South Pennine Moors Special Protection Area phase 2 at 25m x 25m resolution in Calderdale derived from Centre for

Ecology and Hydrology (CEH) Landcover Maps (LCM) datasets for the years 1990, 2000 and 2007.

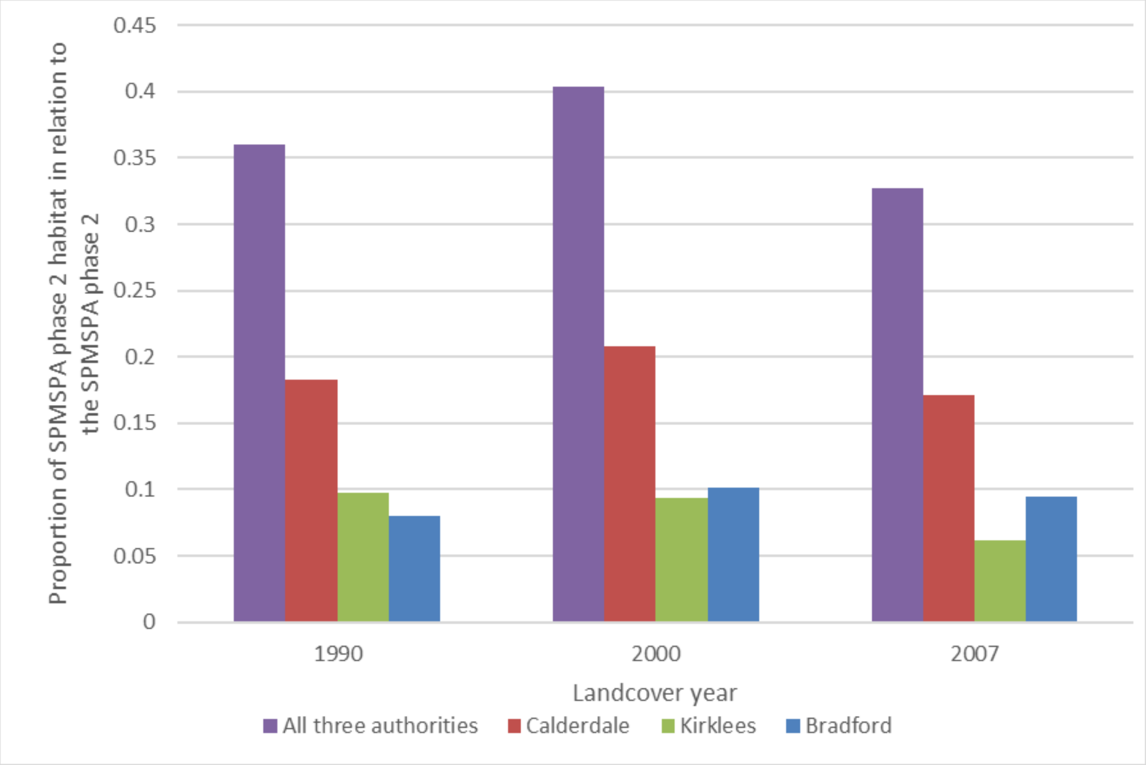

Figure 2.11 The proportion of habitat within the South Pennine Moors Special Protection Areafringe representative of SPMSPA phase 2 habitat for the years 1990, 2000 and 2007.

After applying focal statistics on the 2007 CEH Landcover data, Calderdale appears to have the greatest area of contiguous SPMSPA phase 2 habitat, with several clusters extending to the edge of the 1km fringe (Fig. 2.12). In Bradford, more clusters were apparent in the south than in the north part of the authority area. In Kirklees, clusters were more interspersed with areas not representative of the SPMSPA phase 2 habitat. In the whole SPMSPA 1km fringe, areas that were particularly dense in SPMSPA phase 2 habitat include the northern edge of the Peak District SPA, the south east of Oxenhope, the west of Queensbury, the south of Hebden Bridge, the north of Heptonstall, areas around Marsden and areas to the north of Lydgate (Fig. 2.12).

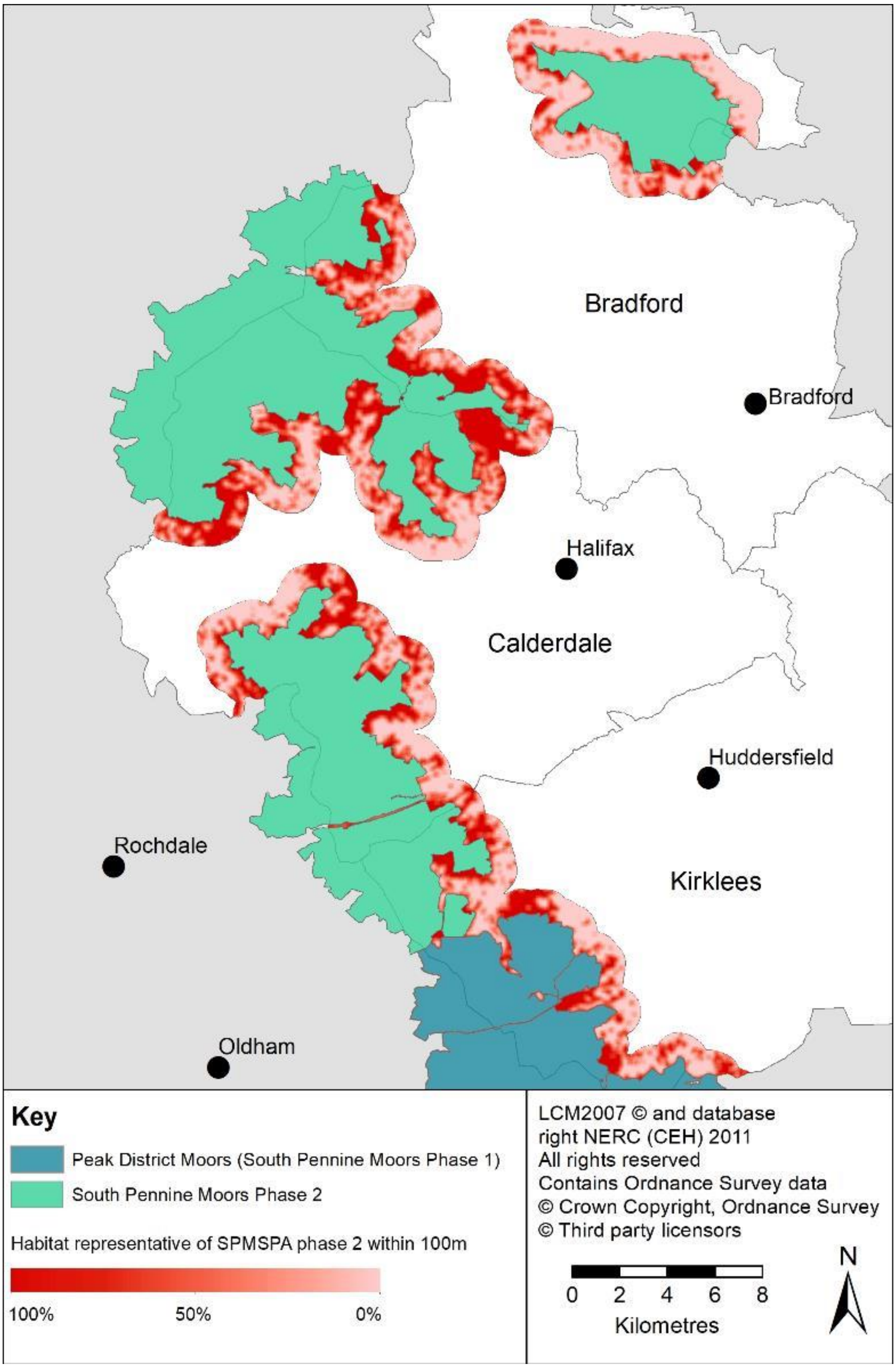

Figure 2.12. Heat map displaying the results of focal statistics applied to the 2007 Landcover data within the South Pennine Moors Special Protection Area 1km fringe.

The proportional coverage of habitat representative of SPMSPA phase 2 habitat with increasing 100m distance bands from the SPA boundary decreased as a function of distance from the SPA boundary (Fig. 2.13). Across all three authority areas, SPA phase 2 habitat was most common within the first 100m from the SPMSPA boundary. Beyond 100m the proportion of phase 2 habitat decreased sharply with increasing distance from the SPA until reaching a threshold between 500m and 600m from the SPA boundary (Fig. 2.13). The proportion of SPA Phase 2 habitat declined further beyond 700m in the Kirklees and Bradford fringe, but increased slightly in Calderdale. These patterns appeared to correspond to patterns of change in mean elevation as a function of distance from the SPA boundary (Fig. 2.14). Sharp declines in the proportion of SPA Phase 2 habitat were evident within the first 200m of the SPA boundary in the Kirklees SPA fringe, and within 300m of the SPMSPA boundary in the Bradford fringe (Fig. 2.13), however there were no apparent sharp declines in mean elevation within these areas at these distances from the SPA (Fig. 2.32).

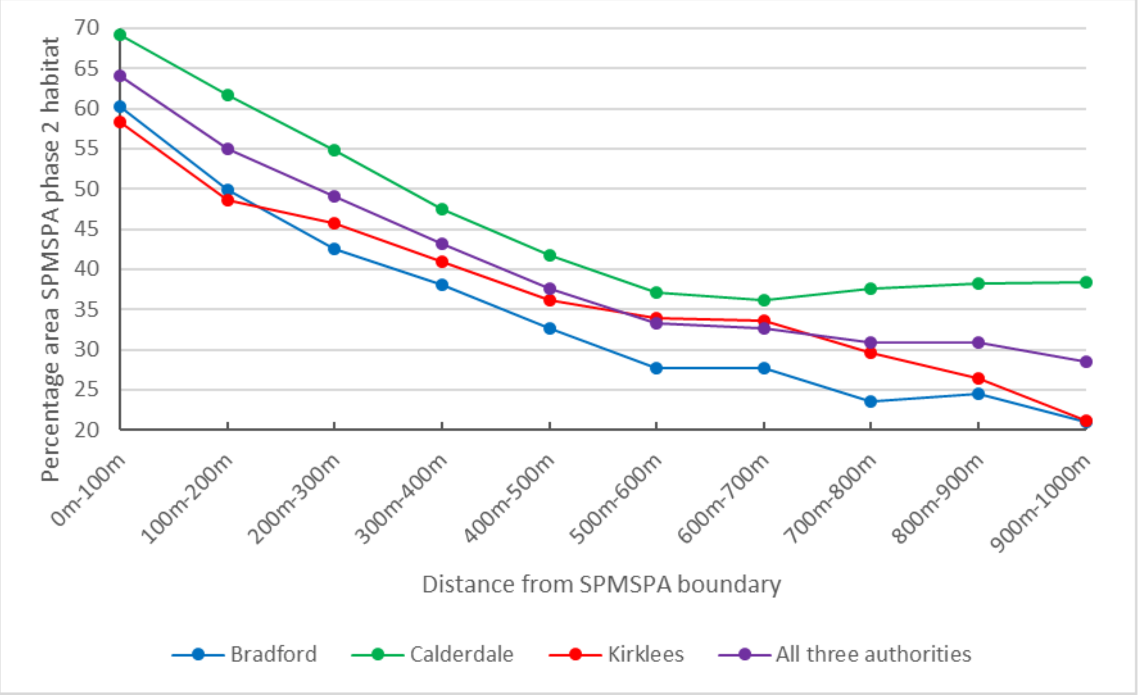

Figure 2.13 The proportion of habitat similar to that of the SPMSPA phase 2 in 100m distance bands extending from the South Pennine Moors Special Protection Area boundary to 1,000m for each authority area and all three areas combined.

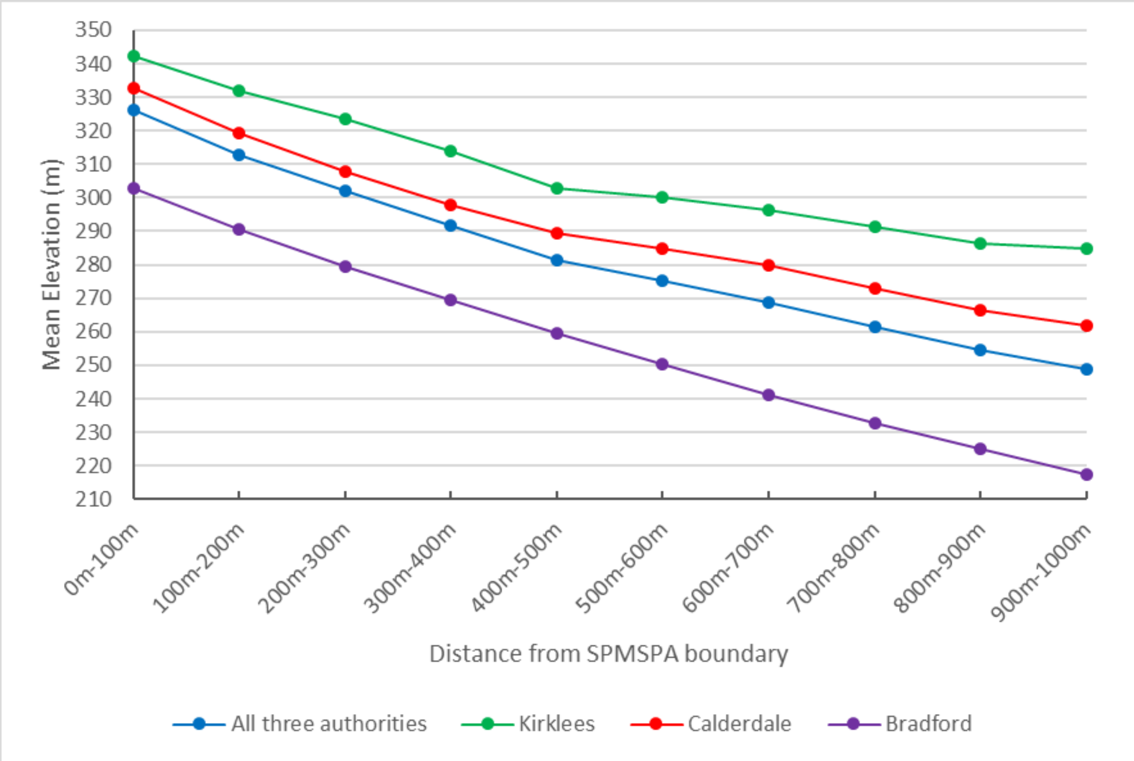

Figure 2.14 Mean elevation in the South Pennine Moors Special Protection Area 1km fringe in 100m distance bands extending from the SPMSPA boundary to 1000m for each authority area and all three areas combined.

2.4.3. Built development within the SPMSPA moorland fringe landscape

Buildings were a common landscape feature of the SPMSPA moorland fringe across all three authority areas. Both the number of buildings (Fig. 2.14) and density of buildings (Fig. 2.15) increased as a function of distance from the SPMSPA boundary. Buildings density was broadly linear across all councils, with increasing density towards the outside of the 1km SPMSPA fringe. This linear relationship indicates sparsely distributed buildings towards the edge of the SPMSPA probably largely represented by agricultural buildings and small villages. The more densely distributed buildings towards the outside edge of the fringe indicate an increase in larger residential and commercial components of the landscape. Sharp increases in the number of buildings within the 400-600m in the Calderdale area, between 500m and 800m in Kirklees and between 500m and 700m in Bradford revealed that the more intermediate distances of the SPA fringe were more heavily developed than areas close to the SPMSPA boundary, and in some cases than the edge of the 1km fringe. This is reflected when buildings in all three authorities are combined, where the number of buildings more than doubled between the SPA boundary and 400m before peaking at around 3,000 buildings between 700m and 800m with a subsequent decrease to the edge of the 1km fringe.

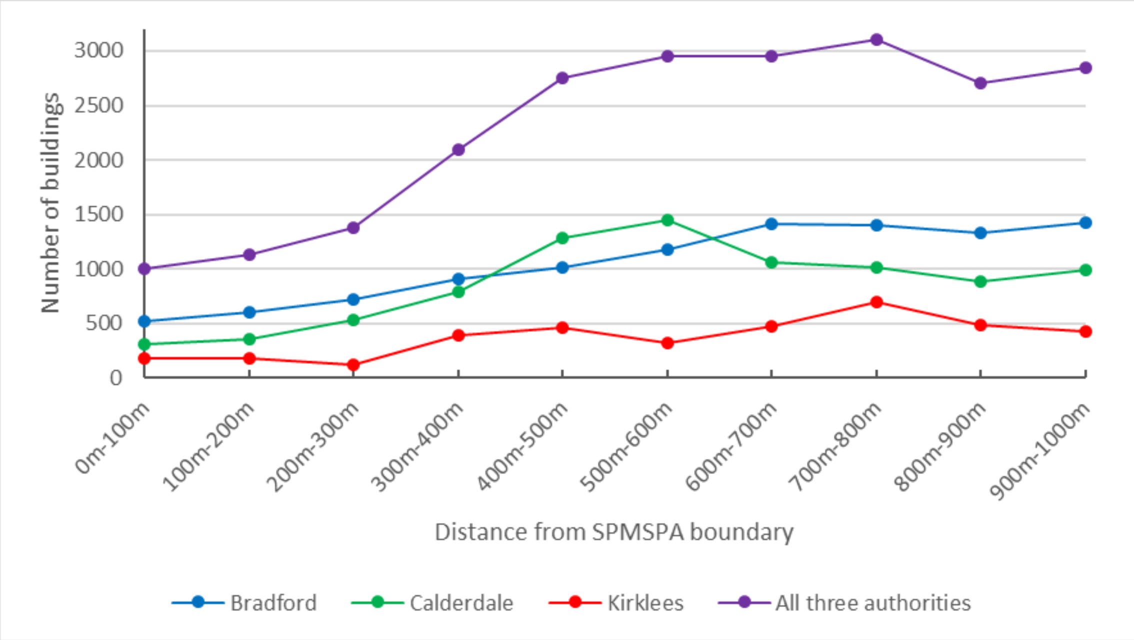

Figure 2.15 Total number of buildings within 100m distance bands extending from the South Pennine Moors Special Protection Area boundary to 1,000m for each authority area and in total.

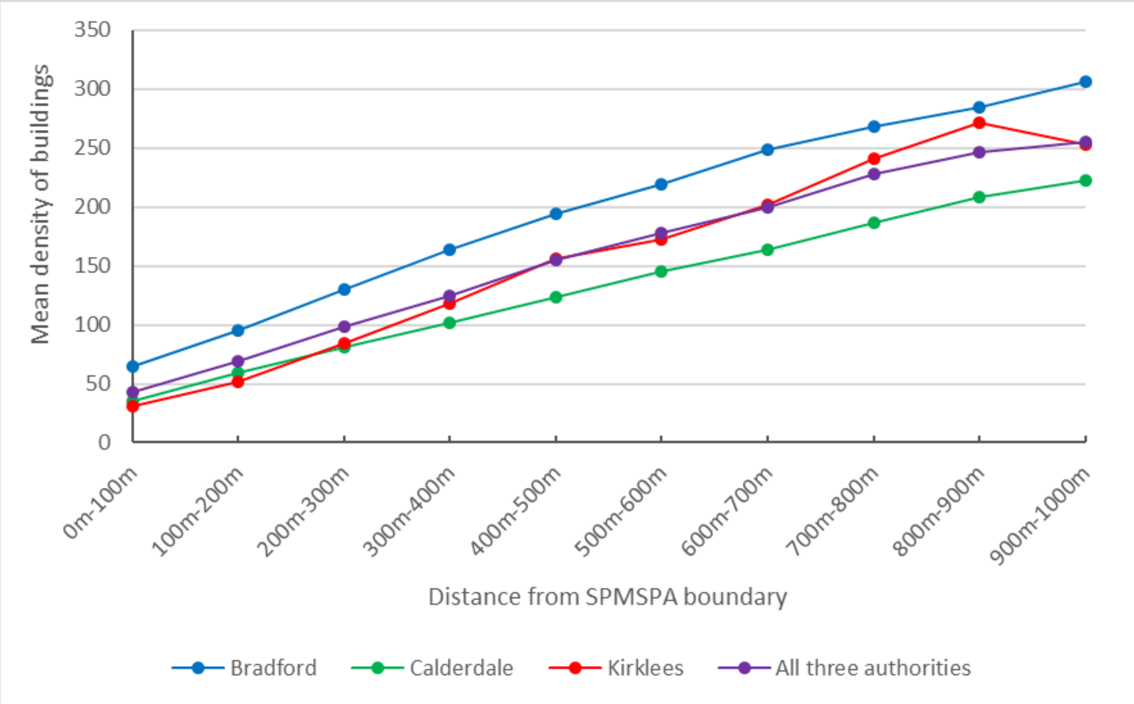

Figure 2.16 Mean density of buildings within 100m distance bands extending from the South Pennine Moors Special Protection Area boundary to 1,000m for each authority area and in total. Building density was represented by a 30m x 30m raster where each pixel the

number of buildings within a 500m radius of that pixel. Units on the y-axis are mean number of buildings within a 500m radius.

2.4.4. Landsat 8 classification of habitats within the SPMSPA fringe

Supervised classification of the SPMSPA 1km fringe was achieved with an overall accuracy of 93.11% (based on the testing dataset). A confusion matrix of classification results shows that user's accuracy and producers accuracy (per habitat class) are high, with every class achieving over 90% for both (Table 2.6). The greatest confusion was found between the habitat categories of improved grassland and semi-improved species poor grassland. There was also moderate confusion between semi-improved species poor grassland and semi-improved species rich grassland, and between rush pasture and semi improved species poor grassland.

Table 2.5 Confusion matrix displaying the results of supervised classification a = Dry heath/ acid grassland mosaic, b = Improved grassland, c = Rush pasture, d = Semi-improved, species-poor grassland, e = Semi-improved, species-rich grassland, f = Wet heath/ acid grassland mosaic.

		Reference							
		a	b	c	d	e	f	*Sum*	*User's Accuracy*
Prediction	a	**823**	9	8	5	4	6	855	*96.26%*
	b	8	**4850**	61	265	25	4	5,213	*93.06%*
	c	3	22	**2273**	105	4	21	2,428	*93.62%*
	d	21	339	131	**6732**	104	16	7,343	*91.68%*
	e	1	2	0	7	**516**	0	526	*96.26%*
	f	2	3	19	27	0	**1311**	1,362	*98.10%*
	Sum	858	5,225	2,492	7,141	653	1,358	17,727	
	Producer's accuracy	*95.92%*	*92.82%*	*91.21%*	*94.27%*	*79.02%*	*96.54%*	***Overall accuracy = 93.11%***	

Using the final random forest model, habitat categories were predicted for the entire SPMPSA 1 km fringe. Based on these predictions, coverage areas for each habitat were calculated using the number of pixels classified multiplied by the pixel area (900 m^2). the habitats with the greatest predicted areas were semi-improved species-poor grassland (145.0 km^2) followed by improved grassland (65.6 km^2). The habitat with the least predicted area was Semi-improved species rich grassland (5.0 km^2). Both dry and wet heath/ acid grassland mosaics were predicted to have very low areas (8.8 km^2 and 14.3 km^2) respectively (Figure 2.17).

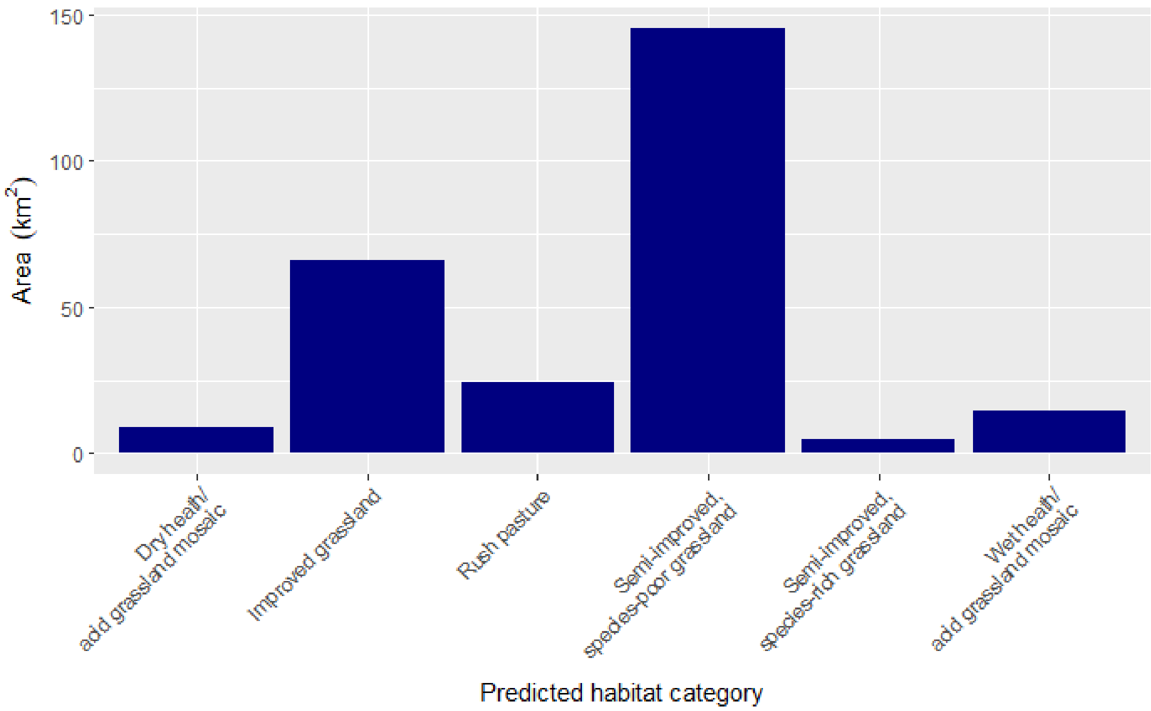

Figure 2.17 Areas of six habitat types within the South Pennine Moors Special Protection Area 1km fringe, as predicted by random forest classification.

Maps were subsequently created for the SPMSPA fringe (Figs 2.18-2.20) using the predictions for each habitat class. In Kirklees, wet heath/acid grassland mosaic occurred in largely contiguous areas, mostly close to the SPMSPA boundary whereas dry heath/ acid grassland mosaic extended to the edge of the SPMSPA 1km fringe with one particularly large patch in the north (Fig 2.18). In Calderdale, dry heath/ acid grassland mosaic was fragmented and occurred in small patches. In contrast, wet heath/ acid grassland mosaic occurred in large contiguous patches in Calderdale that often extended to the edge of the SPMSPA 1 km fringe (Fig. 2.19). Coverage of these two habitats was low in Bradford with small, fragmented patches occurring close to the SMPSMA boundary (Fig 2.20). Large, contiguous patches of species-poor semi improved grassland were present in the SPMSPA 1 km fringe of all unitary authories, becoming more fragmented towards the edge of the fringe and being replaced by improved grassland (Figs 2.18-2.20). Kirklees was dominated by improved grassland, especially in the north (Fig 2.18). Rush pasture was predicted to occur in large contiguous patches close to the SPMSPA boundary in both Calderdale and Kirklees (Figs 2.18-2.19). In Bradford, the SPMSPA 1 km fringe was dominated by species-poor semi improved grassland with rush pasture predicted to occur in small patches close to the SPMSPA (Fig 2.20).

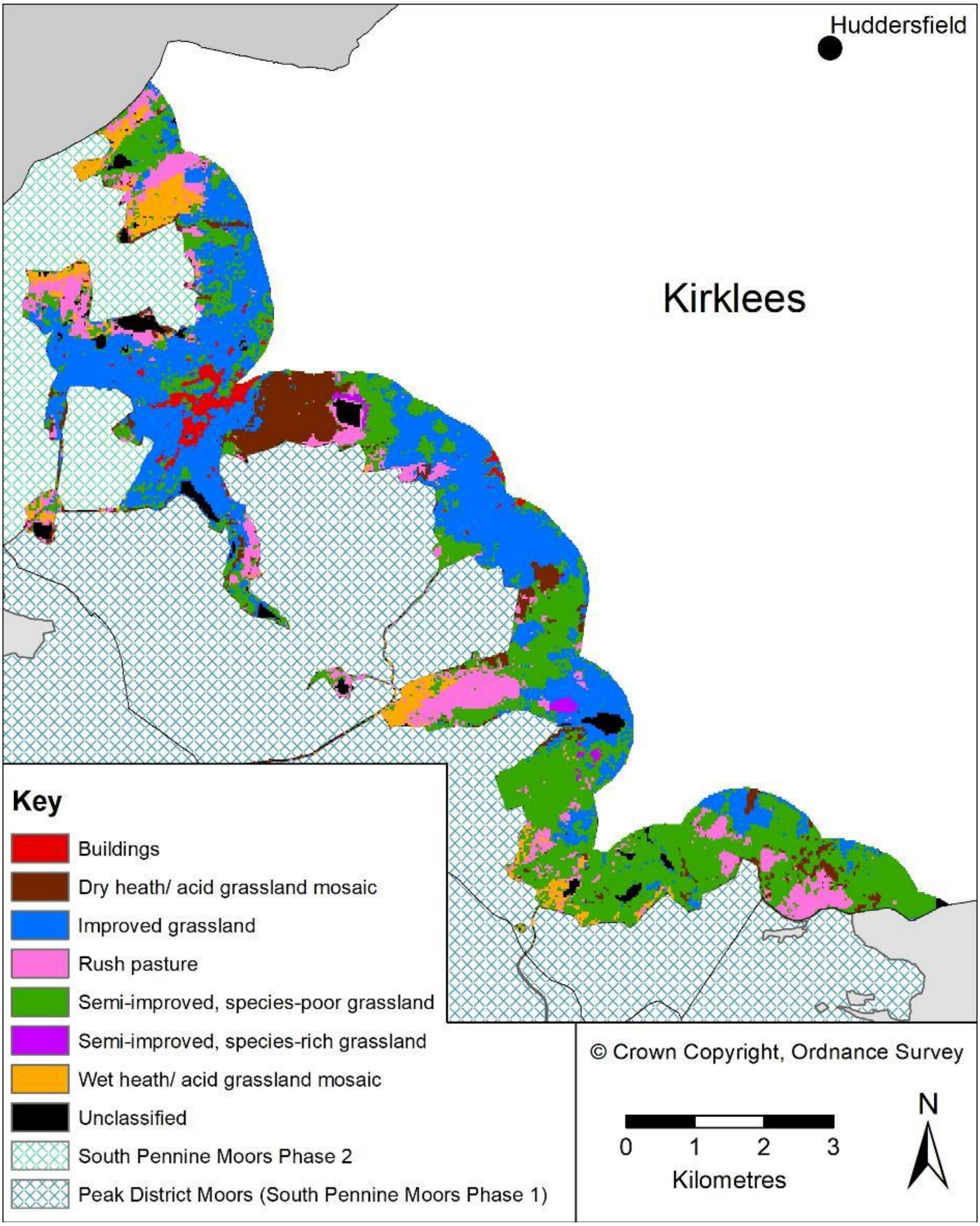

Figure 2.18 Habitat coverage map for Kirklees predicted by random forest supervised classification using Landsat 8 imagery.

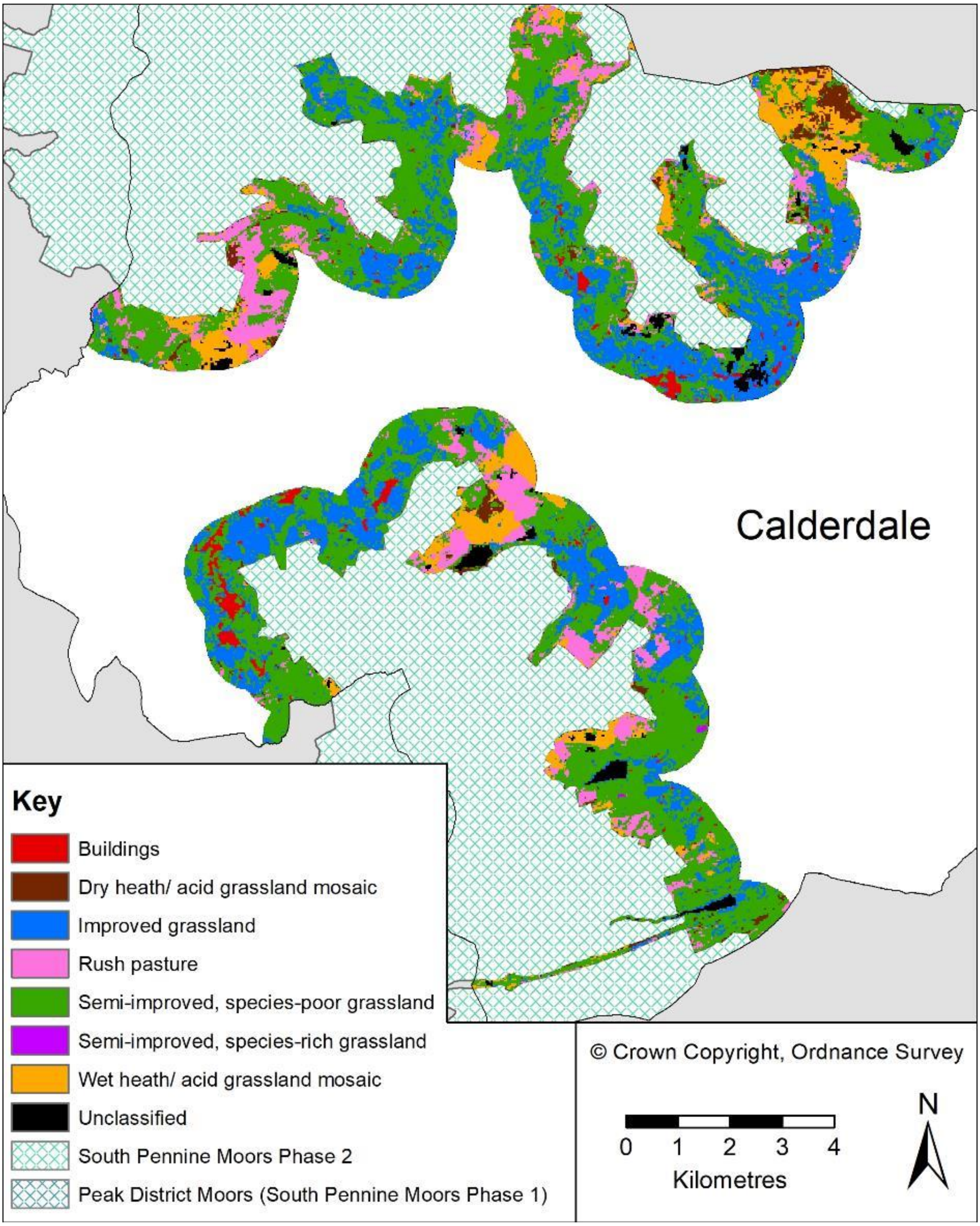

Figure 2.19 Habitat coverage map for Calderdale predicted by random forest supervised classification using Landsat 8 imagery.

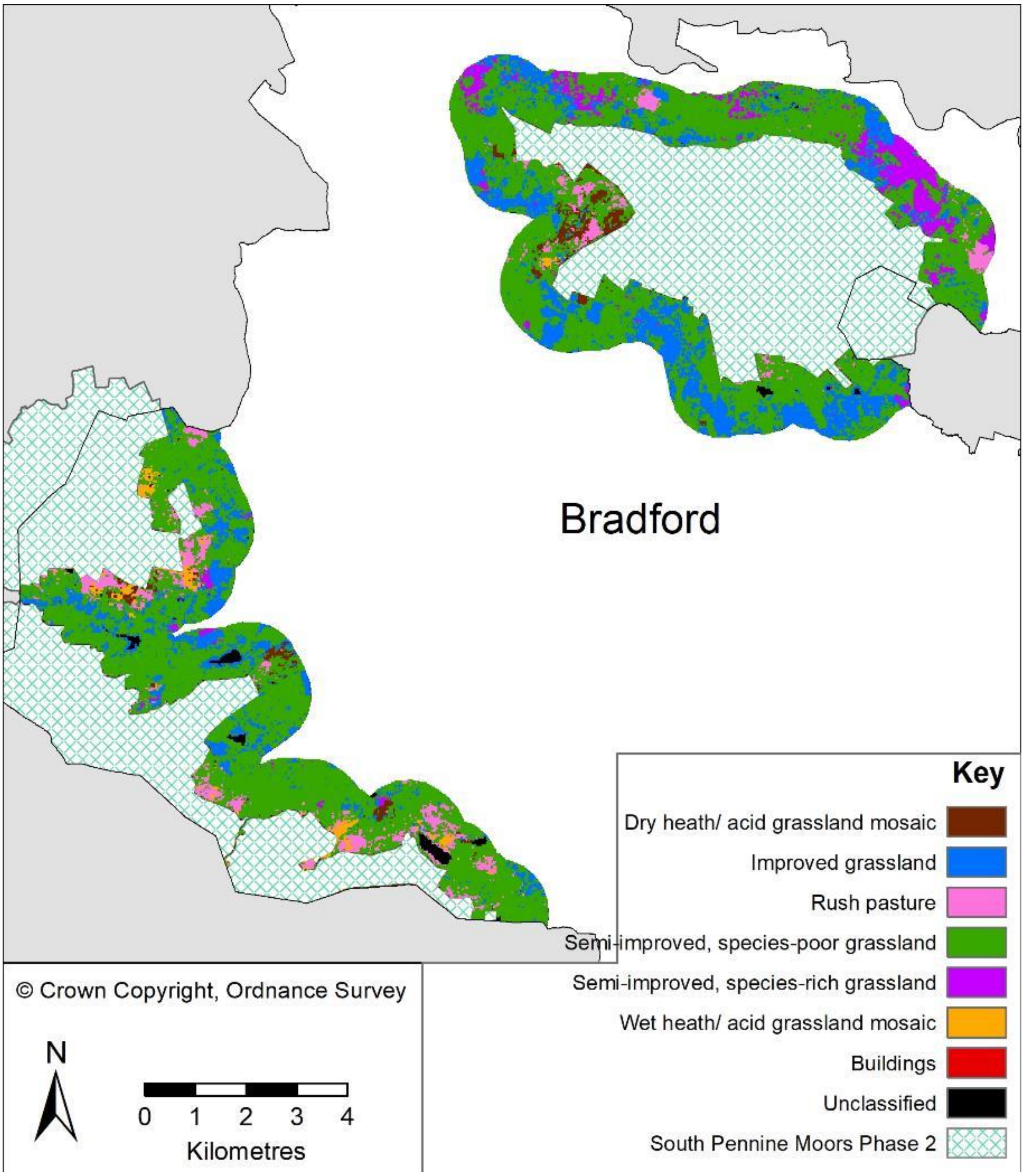

Figure 2.20 Habitat coverage map for Bradford predicted by random forest supervised classification using Landsat 8 imagery.

2.5. Discussion

The SPMSPA moorland fringe is a heterogeneous landscape mosaic containing agricultural habitats, upland habitats and building development with an elevational gradient that gradually decreases with increasing linear distance from the SPA boundary. The agricultural component of the fringe landscape is dominated by species poor semi-improved grassland, improved grassland and rush pasture whereas the more botanically diverse habitats of semi-improved grassland and unimproved grassland are rare. This is in keeping with the findings of Haines-Young et al. (2003) that documented the loss of acid grassland and its replacement with improved grassland in the British uplands over a decade ago, and may reflect the legacy of the Common Agricultural Policy (CAP), a long standing European initiative that has historically encouraged agricultural intensification (van Zanten et al., 2014). Tzanopoulos et al. (2012) suggest that reform of the CAP in 2003 may have resulted in livestock on farms being concentrated in fields that have better pasture-resulting in an increased cover of high intensity grassland, whilst rough grazing becomes neglected. For the SPMSPA this could be interpreted as the improvement of semi-improved fields to improved fields, resulting in a high proportion of improved fields and perhaps under-management or abandonment of rush pasture allowing rush species to dominate, resulting in a high proportion of fields with >75% rush cover. The heterogeneous nature of the study area is not completely atypical of what may be expected of moorland fringe landscapes (French and Dolmans, 2002; French and Picozzi, 2002), however it would appear than there is a strong bias towards intensively modified habitats within the SPMSPA fringe.

The scarcity of upland habitats (most notably blanket bog and mire) is in stark contrast to the habitat composition of the SPMSPA and SPMSAC where dry heaths and blanket bogs dominate (JNCC, 2011), suggesting that core habitat for SPMSPA moorland birds does not often extend into the SPA fringe study site. Interestingly, wet heathland/ acid grassland had relatively high cover in Calderdale and although this habitat was not a primary reason for the selection of the SPMSAC, northern Atlantic wet heath makes up 6.7% of the SAC (JNCC, 2011) and is on annex 1 of the habitats directive (EEC, 1992). Wet heath is an important breeding and/or feeding habitat for Merlin *Falco columbarius*, Golden Plover *Pluvialis apricaria*, Curlew *Numenius arquata*, Red Grouse *Lagopus lagopus*, Black Grouse *Tetrao tetrix* (Hampton, 2008) and Skylark *Alauda arvensis* (Chamberlain and Gregory, 1999), and as such represents an important component of the SPMSPA fringe for supporting upland birds.

Within the SPA fringe, habitats that are similar to the dominant habitats within the SPMSPA (i.e. upland habitats) are locally clustered and not evenly distributed over the study area. Part of this is likely to be due variation in topography, as these habitats are representative of the uplands and may be restricted to higher elevation areas. However, the concept of ‘upland’ in the UK is informally recognised as land above the line of enclosure, usually occurring at between 200m and 300m above sea level (Orr et al., 2008). Mean elevation for the SPMSPA 1 km fringe and cover of habitat representative of the SPMSPA were found to decrease gradually as a function of distance from the SPA, reinforcing the expectation that upland habitats are more likely to occur close the SPMSPA. Another explanation for the clustering of core moorland habitats in the SPA fringe may be the intensification of farming practices resulting in the expansion of improved farmland and potentially resulting in the fragmentation and replacement of upland habitats. This is evidenced by the temporal analysis of CEH Landcover datasets, which showed that habitats typical of those found in the SPMSPA were much less extensive in the fringe in 2007 than in 1990 or in 2000, suggesting that the range of upland habitat extending beyond the SPA boundary has been converted to non-upland habitats. As the moorland fringe upland habitats are likely to support a similar bird assemblage to that found within the SPMSPA, this pattern may have important conservation implications for upland specialist birds in the SPMSPA fringe landscape. An increase between 1990 and 2000 in SPA habitat in the fringe suggests that the decline is not constant and may be recoverable. It is likely that these observed patterns (especially in 1990) are not quantitatively accurate due methodological differences employed by CEH between years, a problem encountered by other research using CEH landcover to make temporal inferences using these data (e.g. Pearce-Higgins et al. 2006). Efforts were made to standardise differences between years during data analysis and as such the general temporal pattern of these results should not be disregarded, especially as the CEH LCM datasets represent the best available full landcover data in the UK (to the authors knowledge). These datasets are produced periodically, and the results presented here allow a benchmark for future patterns to be compared to. It would be prudent to apply the method used here to future datasets in order to assess further temporal patterns in the coverage of upland habitats within the SPMSPA fringe.

The number of buildings peaks within the central distance bands of the 1km fringe, suggesting that development is relatively high within the central portions of the SPMSPA 1 km fringe, but not in close proximity to the SPMSPA. Further information on this pattern can be gained from the fact that building density increasing linearly with increasing distance from the SPMSPA, suggesting that the buildings found in the central portions of the SMPSPA fringe are numerous, but spatially separate. This is indicative of less densely developed areas than might be expected towards the edge of the SPMSPA fringe where numerous medium sized towns lie. The effect of housing density on moorland fringe bird habitat suitability will be investigated in chapter 5. Urban encroachment on protected areas is a globally recognised phenomenon, with 25% of the worlds protected areas projected to be within 15km of a city with a population of at least 50,000 by 2030 (up from 17% in 1995) while in western Europe this figure is predicted to rise to 3% by 2030 (up from 4% in 1995) (Mcdonald et al., 2008). Previous studies from the USA have found that housing growth rates since the 1960s have been consistently greater within 1km of protected areas than the national average (e.g. Radeloff et al., 2010). In the UK, European protected sites including SPAs have been allocated Impact Risk Zones (IRZs) which serve to highlight situations where potential developments within certain distances of a protected area may impact on the ecology of the protected area (Natural England, 2014). Although the use of IRZs is designed to take the potential negative effects of development around protected areas into consideration, there is no scientific literature on these impact zones and the evidence used to create them is not readily available. Where the objective is to avoid the concentration of ecologically poor anthropogenic land cover (e.g. buildings and hardstanding), it may be appropriate to avoid further development in distance bands that are already highly developed and close to the protected area, and distribute development more evenly throughout the fringe landscape. The linear relationship between building density and distance from the protected area is encouraging, however a peak density in Kirklees at 800-900m suggests that there may be some encroachment of intense development within the SPMSPA 1km fringe. This should be monitored and the linear relationship of building density maintained. Historically mapped building data exists in the UK, and the use of these data to further analyse patterns of developmental encroachment on protected areas should be encouraged, in conjunction with research into the potential effects of these developments on the ecological functionality of protected areas.

Remote sensing is useful in classifying vegetative cover over large areas where field surveys are not possible (Xie et al., 2008), and for monitoring temporal changes in land-use and land cover. As such, there is a high potential application for remote sensing in the SPMSPA fringe because an understanding of land use change and habitat cover over time may be critical to the ecological functionality of the SPA. Remote sensing has been successfully used for applications similar to those required for the SPMSPA fringe, including classifying habitat cover in areas where natural and semi-natural habitats interface with agricultural land (Lu et al., 2012), for characterising moorland vegetation in the UK (Buchanan et al., 2005) and to model urban sprawl (Taubenböck et al., 2012). The use of satellite imagery for categorising agricultural grasslands on their management intensity is not a well-studied field (Franke et al., 2012). Nevertheless it has been proven possible with very high resolution (6.5m) satellite imagery (RapidEye) and with frequent images taken of the same area, reducing the potential of cloud cover and allowing many images representing a short time period to be incorporated into the classification algorithm (Franke et al., 2012).

The final objective of this chapter was to classify the SPA fringe landscape using remote sensing techniques. An accuracy estimation of 85% is widely adopted as the threshold for an acceptable degree of accuracy (Congalton and Green, 2009) so the SPMSPA moorland fringe classification accuracy of 93.11% can be viewed as extremely reliable, especially as intra-class accuracies were also all above 90%. The previously mentioned Natural England description of the SPMSPA fringe agricultural landscape as a mosaic of small to medium fields dominated by relatively intense sheep farming, but with the presence of less improved habitats such as wet grassland, rush pasture and species rich meadows (Natural England, 2012) is consistent with the habitat patterns found here. However, the coverage of less improved habitats such as wet and dry heath/ acid grassland mosaic and semi-improved species rich grassland were extremely low. Low intensity farmland provides structural heterogeneity which is beneficial to birds utilising farmland through the provision of foodplant diversity, cover from predators and refuge from extreme weather (Wilson et al., 2005). As such, the domination of improved grassland and semi-improved species-poor grassland throughout the SPMSPA fringe may be detrimental to bird populations that feed and breed within the fringe. The pattern of semi-improved species poor grassland towards the edge of the SPMSPA merging into improved grassland over much of the SPMSPA fringe suggests that areas close to the SPMSPA are less intensively managed. For birds commuting between the SPMSPA and the surrounding fringe, this is likely to be a benefit as these birds do not need to travel great distances to relatively low intensity farmland. It would be prudent to extrapolate the classification model constructed here on historic Landsat data to identify whether this ecotone is stable or represents encroachment of improved grassland from the outside edges of the SPMSPA fringe towards the boundary of the SPMSPA.

The ability of Landsat 8 imagery to provide accurate classification of farmland habitat is encouraging, especially considering that the division between habitat categories is not always a definitive boundary, but is more of a transitional continuum (e.g. semi-improved species poor grassland and semi-improved species rich grassland). As Landsat imagery is freely available and backlogged to the 1970s, it provides a great opportunity as a remote sensing tool to assess trends in agricultural intensification.

A limitation of the classification undertaken is the omission of habitat categories that were not surveyed such as woodland, hedgerows, rivers and bare rock and the omission of minority habitats. Although this was necessary for the improvement of classification accuracy within the six habitat categories used, it undoubtedly means that some habitats that have been classified here will in fact be habitats not included in the classification process. As such, the classification used here is useful for the study of landscape scale patterns, but should not be used as an exhaustive classification map. The next chapter will explore the relationship between the habitats described within this chapter and moorland fringe birds.

2.6. References

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CHAPTER 3: MOORLAND FRINGE BIRD COMMUNITY COMPOSITION AND HABITAT ASSOCIATIONS

3.1. Abstract

The moorland fringe landscape is composed of a heterogenous mosaic of habitats including upland semi-natural habitat, farmland habitat and residential areas (Chapter 1). Bird species that use moorland protected areas also use farmland habitat within the fringe landscape. The intensification of agriculture in recent years has negatively impacted many farmland bird species, and has the potential to negatively impact moorland bird species that utilise agricultural habitats. The expansion of residential areas may further compound this. This chapter investigates the associations of field level habitat characteristics such as dominant habitat type, management regime, wildflower richness and wet flush presence and gradient with five moorland fringe bird species that are known to depend on an inland Special Protect Area (SPA). Field level habitat characteristics are presented as three moorland fringe habitat gradients that were determined using Non-metric Multidimensional Scaling (NMDS). The patterns of association with these gradients and Curlew *Numenius arquata*, Lapwing *Vanellus vanellus*, Snipe *Gallinago gallinago*, Wheatear *Oenanthe oenanthe* and Golden Plover *Pluvialis apricaria* were investigated. The relative bird diversity of moorland fringe habitats was also explored.

Evenness was relatively low across within the moorland fringe with 71% of all bird records represented by only five species. Bird species richness was greatest in habitats not typical of moorland or farmland indicating the importance of broad habitat diversity in maintaining bird diversity within the moorland fringe. All five moorland fringe bird species that were investigation for habitat gradient associations showed a significant preference for fields that have tussocks, wet flush and are intensively grazed and a preference against fields where the vegetation is mechanically cut. In addition, Snipe and Wheatear have preference against semi-improved grassland and a strong preference for fields with a dominant cover of rush.

3.2. Introduction

Anthropogenic land use around protected areas is a source of potential stress to ecological processes within protected areas (Hansen and DeFries, 2007), especially where habitat outside of the protected area plays a role in supporting species that the designated area is designed to support (Berger, 2004; Hamilton et al., 2013; Xun et al., 2014). Many species are ecologically dependent not only on the habitats within protected areas, but also on the landscape within the immediate vicinity through the provision of additional feeding and breeding habitat (Fahrig, 2007; Gaston et al., 2008). At the edges of protected uplands in the UK, where moorland habitats interface with farmland and developed land (i.e. villages and small towns), bird species may use multiple habitats. The preferences of upland birds for these habitats vary between species. Lapwing *Vanellus vanellus* and Skylark *Alauda arvensis* have preferences for farmland but also use moorland. Some species such as Meadow Pipit *Anthus pratensis*, Snipe *Gallinago gallinago* and Curlew *Numenius arquata* favour both habitats, and others such as Golden Plover *Pluvialis apricaria* favour moorland but also use farmland to supplement feeding (Pearce-Higgins and Yalden, 2003; Dallimer et al., 2012). In cases like this, where bird species are protected within an area but are likely to use habitat outside of the protected area, it is important to understand the bird-habitat associations within the fringe landscape of the protected area. As an extension of this, the management of habitats within the fringe areas may play an important role in the conservation of a species within a protected area. This is true for moorland bird species such as Snipe and Curlew for which there is a clear association between moorland habitat management and the management of the surrounding farmland in terms of the success of these species (Dallimer et al., 2012).

The landscapes surrounding protected areas are often heterogeneous matrixes (Maestas et al., 2003; Hamilton et al., 2013), containing a variety of habitats that incorporate remnant or similar habitats found within the protected area, as well as more anthropogenic disturbed habitats that are used primarily as agricultural land use or for housing development (Maestas et al., 2003; Joppa et al., 2009). The land-use gradient within these landscapes often follows a general trajectory from more undisturbed, natural areas in close proximity to the boundary of the protected area, to areas of more intensive land use, dominated by land converted for agricultural and housing development, at increasing distances from the boundary (Łowicki and Walz, 2015). In addition, where protected areas lie in close proximity to urban areas, management practices and local planning decisions can have a profound effect on the extent to which the protected area fringe is able to support biodiversity and ecosystem services (Łowicki and Walz, 2015).

The intensification of agricultural land-use practices and associated degradation and fragmentation of habitat suitable for birds is a problem that is global in scale (Kehoe et al., 2015). It has been advocated that a landscape scale approach to the management of agricultural intensification may be more beneficial to preserving the biodiversity of a protected area than local level management (Tscharntke et al., 2005). However, in the case of ground nesting farmland birds in Europe, a combination of landscape level and field level approach to management should be considered in order to maximise conservation benefits to breeding birds (Guerrero et al., 2012). There are many circumstances in which birds use different habitat types on a localised or landscape basis (Dunning et al., 1992; Dallimer, Gaston, et al., 2010) because they are mobile and require different resources from these habitats (Whittingham et al., 2000; Söderström et al., 2001; Mckenzie et al., 2013; Galitsky and Lawler, 2015). The conservation of bird populations across these heterogeneous landscapes therefore relies not only on the extent and quality of multiple habitat types (Dallimer, Marini, et al., 2010), but also on knowledge of the bird community composition and the attributes of the habitat influencing their abundance. This represents a significant conservation challenge for local government authorities, especially where protected area designation focus on a single or few habitats, excluding the adjacent matrix and potentially over simplifying ecological processes by focussing on the spatial constraints of protected areas (Fischer and Lindenmayer, 2007; Santini et al., 2016). In addition, the composition of habitat external to protected areas should be managed to avoid losing connectivity to habitat that may be used by species within the protected area (Goetz et al., 2009). In the United Kingdom, this is especially true of Special Protection Areas (SPAs) created to protect moorland habitats and their bird populations. Across Europe, upland SPAs are typically designated based on bird breeding sites and consequently can exclude important matrix habitats such as the surrounding agricultural areas. The creation of SPAs has brought benefits in terms of the international recognition to the biodiversity of semi-natural UK moorland habitats. These habitats are particularly important for populations of numerous threatened bird species of European significance, as they are highly dependent on core moorland areas and the surrounding fringe landscape for breeding and foraging (Pearce-Higgins and Grant, 2006; Dallimer, Marini, et al., 2010; Dallimer et al., 2012). Recent years have seen a significant increase in anthropogenic activity in and around moorland SPAs including planning proposals for residential development, renewable energy developments and increased recreational access (Pearce-

Higgins et al., 2007, 2008). In addition, recent changes in government farming subsidies have resulted in a transitional agricultural landscape that has the potential to affect populations of moorland birds using agricultural land (Acs et al., 2010).

In the previous chapter, the types and extent of habitats that make-up the moorland fringe landscape surrounding the South Pennine Moors Special Protection Area (SPMSPA) were characterised. The spatial data analyses revealed that the SPMSPA moorland fringe is a heterogeneous landscape mosaic containing various agricultural habitats, a smaller proportion of upland habitats and areas of building development, with an elevational gradient that gradually decreases with increasing linear distance from the SPA boundary. In this chapter, the associations between field level habitat variables and the bird community composition, across the SPMSPA moorland fringe landscape were explored to help guide strategies for bird conservation and development proposals for the three local government authorities (councils) responsible for managing the SPA and fringe landscape. This involved comparing bird species richness, diversity and abundance across all moorland fringe landscape habitats, and examining whether these patterns are similar across the three different regions managed by the three different local councils, compared to the whole SPA moorland fringe landscape. Additionally, gradients in habitat and management characteristics of moorland fringe fields across the landscape were investigated using ordination techniques. The associations between these gradients and the abundance of some of the conservation-priority bird species were investigated and the implications for the conservation management of these habitats and species was discussed.

3.3. Methods

3.3.1. Study site

Bird and habitat surveys were conducted at sites located within the SPMSPA moorland fringe landscape. A detailed description of the SPMSPA and the composition and configuration of the habitats of the surrounding fringe landscape are provided in Chapter Two.

3.3.2. Bird surveys

Bird surveys were conducted within the British breeding bird season during April-July of 2012, 2013 and 2015, with two visits undertaken in each of these years (early season and late season Survey methodology was based on the British Trust for Ornithology (BTO) Common Bird Census (CBC) and Breeding Bird Survey (BBS) methods (Marchant 1983; Risely et al 2013). All surveys were conducted only during hours and days of suitable weather conditions. Surveys were undertaken by employees of West Yorkshire Ecology (WYE), the ecological records service for West Yorkshire working on behalf of the local authorities of Calderdale, Kirklees and Bradford. As a professional ecological service, WYES used surveyors competent in upland bird survey techniques and bird identification.

A series of 241 1km^2 quadrats were selected within 1 km of the SPMSPA boundary in 2012 (n = 53) and 2013 (n = 107), and within 2.5 km in 2015 (n = 64). Within each quadrat, two line transects along public rights of way were established, each approximately 1 km in length. Transects within 1 km survey squares were positioned as near to parallel as possible and separated by 500m where possible. Transects were walked at a steady pace (1km per hour) between the hours of 0800hrs and 1800hrs. Where transects were repeated during the same breeding season, the direction of travel was reversed to maximise observer visual coverage of the area without bias caused by topographic gradients and visual blockages. Vantage point (VP) surveys were undertaken by WYE in addition to the line transect surveys and due to inseparability of the VP data from the line transect data, were included as supplementary data in analysis for this chapter. All bird encounters (sight and sound) within 250m of the transect were recorded on a 1:5000 map, with notes on activity (flight, breeding and feeding). Although distance sampling would have been beneficial for this study, distances were not recorded and transect routes were not available for all transects undertaken prohibiting post-hoc inferences of distances.

Thirteen bird species were identified by project partners as of special survey interest due to their conservation significance and their possible utilization of the moorland fringe landscape (Table 3.1). in relation to the SPMSPA (table 1), however the survey method implemented allowed all species encountered to be surveyed.

3.3.3. Habitat surveys

Habitat surveys were undertaken in the SPMSPA fringe between 2012 and 2013. Kirklees and Calderdale were surveyed by staff from the ecological consultancy WYE in July-September 2012. Bradford moorland fringe areas were surveyed by staff from Urban Edge Environmental Consulting company and WYE in June-July 2013. All surveys were conducted by professional ecologists who were familiar with the habitats and the survey method. Methodology was agreed between surveyors to provide standardisation of results. Further details are provided in Chapter Two.

Table 3.1 List of the Conservation priority bird species and their UK conservation status (as of 2014).

Target species	RSPB conservation status	Rationale for inclusion as a target species
Dunlin (*Calidris alpina*)	Red	Protected under the jurisdiction of the SPMSPA. Red status in the UK.
Twite (*Carduelis flavirostris*)	Red	Historic stronghold around the SPMSPA. Uses SPMSPA as a breeding ground. Red status in the UK.
Ring Ouzel (*Turdus torquatus)*	Red	Uses SPMSPA as breeding ground. Red status in the UK.
Lapwing (*Vanellus vanellus*)	Red	Uses SPMSPA as breeding ground. Red status in the UK.
Common Sandpiper (*Actitis hypoleuco*s)	Amber	Uses SPMSPA as breeding ground.
Short-eared Owl (*Asio flammeus*)	Amber	Protected under the jurisdiction of the SPMSPA.
Merlin *(Falco columbarius*)	Amber	Protected under the jurisdiction of the SPMSPA.
Snipe (*Gallinago gallinago*)	Amber	Uses SPMSPA as a breeding ground
Curlew (*Numenius arquata*)	Amber	Uses SPMSPA as breeding ground. Near threatened status on IUCN red list
Wheatear (*Oenanthe oenanthe*)	Amber	Uses SPMSPA as breeding ground.
Golden Plover (*Pluvialis apricaria*)	Amber	Protected under jurisdiction of SPMSPA.
Whinchat (*Saxicola rubetra*)	Amber	Uses SPMSPA as breeding ground.
Redshank (*Tringa tetanus*)	Amber	Uses SPMSPA as breeding ground.

3.3.4. Measures of bird community composition

Flying bird records and duplicated individuals (i.e. where the same individual was recorded in multiple localities) were removed from subsequent analyses, leaving only perched individuals. Where duplicated individuals existed in the dataset, only the first recorded detection was used. As the primary objective of this chapter was to explore field level bird-habitat associations, birds recorded in fields that were not included in the habitat survey (see Chapter Two) were removed from analysis.

Measures of community composition were calculated for all habitats within the three unitary authority areas, and for all three authorities combined using the *BiodiversityR* package (Kindt and Coe, 2005) using the R Development Software (R Core Team, 2013). Bird species diversity was represented by three different indices: the Shannon–Wiener index (H'), Simpson's diversity index (1/D) and the Berger-Parker diversity index. Simpson's index is weighted by the commoner species in a sample, whereas the Shannon-Wiener index is weighted by the rarer species and by species richness (Magurran, 2004). As both emphasise different aspects of biodiversity, and to facilitate direct comparison with other studies (future and past) it was decided that both Simpson's index (D) and the Shannon-Wiener index (H') would be calculated and discussed separately in light of the biases of each. The value of D decreases as diversity increases, therefore $1/D$ was calculated as a more intuitive index (increase in value represents increase in diversity). In an effort to describe evenness separately from species richness, Shannon's measure of evenness (J') and Simpson's measure of evenness ($E_{1/D}$) were calculated. The Berger-Parker (d) index is a simple, intuitive and biologically meaningful measure of diversity that describes the relative importance of the most dominant species in an assemblage (Magurran, 2004). This was calculated to allow conclusions to be drawn on consensus between multiple biodiversity indexes (see Appendix 3 for diversity index equations).

To allow comparison of species richness between habitats where sampling effort differed, individual based rarefaction curves were calculated for bird species richness by habitat. Individual based rarefaction is calculated by repeatedly randomly subsampling *n* individuals from a sample of birds (in this case, the sample was habitat) and averaging the number of species present in gradually increasing values of *n* (Gotelli and Colwell, 2010). Further comparison of estimated species richness between habitats was achieved by extrapolation for all unitary authorities combined, a technique that extends the rarefaction curve to the number of individuals present in the largest sample (Gotelli and Colwell, 2010). In addition, 95% confidence intervals were calculated for each rarefaction curve. In a pairwise comparison of rarefaction curves, where the 95% confidence interval of the smaller sample does not overlap the rarefaction curve of the larger sample, a difference in species richness with $P<0.05$ can be assumed (Gotelli and Colwell, 2010) Rarefaction and extrapolation were calculated using *EstimateS* software (Colwell, 2013).

3.3.5. Habitat gradient associations of moorland fringe birds

Ordination of field level habitat characteristics within the moorland fringe landscape was undertaken using Non-metric multidimensional scaling (NMDS), a well-established technique that has been used extensively to identify ecological communities and gradients in bird communities (e.g. Clough et al., 2009; Borges et al., 2016; Fazaa et al., 2017), but can also be used to determine habitat gradients (e.g. Laurance, 1994). In this study, NMDS was used to cluster relationships between measured environmental variables and to provide a quantitative measure of gradients along these habitat variables. The variables measured are summarised in table 3.2. In order to emphasise habitat gradients and remove ambiguity, categorical variables were converted to dummy variables (i.e. binary variables for each category). Dummy variables and binary variables where presence represented <5% of the total number of fields or did not fall into a definitive category (i.e 'other') were removed from analysis. As multiple variable types were used in NMDS analysis (binary, ordinal and continuous) a Gower dissimilarity matrix was deemed to be most appropriate (Gower, 1971) and due to the relative sparsity of presences for some categories after conversion to dummy variables, a step-across transformation was applied (Williamson, 1978; Bradfield and Kenkel, 1987). Two, three and four NMDS dimensional axes were tested in analysis with the final number of dimensions selected using the lowest convergent stress value produced after 100 iterations (three axes). The NMDS procedure was undertaken using the vegan package in R (Oksanen, 2008).

Table 3.2 Descriptions of the variables used to describe habitat gradients and determine bird-habitat gradient associations.

Habitat variable	Description
Dominant habitat	Categorical variable. Habitat with greater >75% coverage within a single field. After removal of categories with low representation, habitat types were; Improved grassland; Rush pasture; Semi-improved species poor grassland. See chapter two For full descriptions.
Management	Categorical variable. Categories are; none (no obvious management); Cut/Mown; Grazed.
Flush	Binary variable. Presence or absence of a flush (waterlogged land fed by ground water) within a field.
Molehills	Binary variable. Presence or absence of molehills within a field.
Tussocks	Binary variable. Presence or absence of tussocks within a field.
Dry stone wall	Binary variable. Presence or absence of dry stone wall as a field boundary.

Gradient	Ordinal variable. Steepness of slope within a field. 0 = flat, 1 = <5º, 2 = 5 º -10 º, 3 = >10 º.
Grazing intensity	Ordinal variable. How heavily is the vegetation grazed within a field. 0 = ungrazed, 1 = lightly grazed, 2 = moderately grazed, 3 = heavily grazed.
Wildflower	Ordinal variable. Of four wildflower types (dandelion, sorrel, thistle and hawkbit), how many are present in a field.
Building density	Continuous variable. The inverse density of buildings as calculated using thiessen polygon. See chapter two for details.

Using the resultant NMDS axis scores as a proxy for habitat gradient, habitat gradient associations of conservation-priority bird species (table 3.1) were assessed using Generalized Additive Models (GAMs). Only species with greater than 20 presence records were included in analysis. The five species remaining for analysis were Curlew, Snipe, Golden Plover, Lapwing and Wheatear. All GAMs were fitted using the package *mgcv* in R (Wood, 2006), using an automated algorithm for optimising splines. Each species was fitted to the habitat gradients using seven GAMs, representing all additive combinations of the three habitat gradient NMDS axes used in analysis. Akaike's Information Criterion (AIC) was used to determine the best fitting model for each species and the significance of each axis on the presence of these species calculated using a chi-squired test.

3.4. Results

3.4.1. *Measures of moorland fringe bird community composition*

During the surveys conducted across 2012, 2013, and 2015 a total of 6,142 records, numbering 9,303 individual perched adult birds, and corresponding to 70 species across 15 habitat types, distributed over 2,903 fields across Kirklees, Bradford and Calderdale unitary authorities were recorded (Table 3.3). The most commonly encountered species were Starling *Sturnus vulgaris* (n=3,303), Lapwing *Vanellus vanellus* (n=1,050), Carrion Crow *Corvus corone* (n=842), Curlew (n=741) and Meadow Pipit *Anthus pratensis* (n=706) where n is the number of individuals. Of the conservation-priority species, Lapwing was the most commonly encountered species, followed by Curlew, Golden Plover *Pluvialis apricaria* (n=167), Wheatear *Oenanthe oenanthe* (n=122), Twite *Carduelis flavirostris* (n=64), Snipe *Gallinago gallinago* (n=60), and Short-eared Owl *Asio flammeus* (n=28). There were very few records for the remaining conservation-important species: Redshank *Tringa totanus* (n=14), Common Sandpiper *Actitis hypoleucos* (n=12), Whinchat *Saxicola rubetra* (n=6), Merlin *Falco columbarius* (n=5), Ring Ouzel *Turdus torquatus* (n=4) and Dunlin *Calidris alpine* (n=1). The moorland fringe bird community for each unitary authority area followed a similar pattern in that they were dominated by a few abundant species with only a few rare species. This was also true for all unitary authorities combined. This is illustrated by the fact that the five most abundant species represented 82.2% of all birds recorded in Bradford, 69.7% in Calderdale, 73% in Kirklees and 71.4% for all unitary authorities combined. A rank abundance plot (Fig. 3.1) reveals similarities in evenness between Calderdale, Kirklees and all unitary authories combines, whereas Bradford was less even with a relatively low number of species dominating the bird community. Conversely, Bradford had fewer species of low abundance that other unitary authorities, but also a much lower species richness. Species richness was greater when all three unitary authorities were treated collectively as opposed to any one authorities being treated individually (Fig. 3.1).

Measures of moorland fringe bird community composition are shown in Table 3.3 for all unitary authority areas combined. The habitats which supported the greatest number of individual birds were semi-improved species-rich grassland and improved grassland, whereas habitats with the least number of individual birds were dry dwarf shrub heath, blanket bog/mire, enclosed upland acidic grassland and wet heathland/mire (Table 3.3). Bird species richness (S_{obs}) was highest in habitats not typical of moorland or farmland, such as woodland or gardens. Other habitats with high observed species richness included improved grassland, semi-improved species poor grassland and fields with no single habitat with >75% coverage (Table 3.3). Species poor-habitats (those with very low S_{obs}) included dry dwarf shrub heath, blanket bog/mire, rough grassland, unimproved grassland and enclosed upland acidic grassland, all of which had fewer fields (Table 3.3). Patterns of estimated bird species richness (S_{est}), achieved through extrapolation broadly matched observed species richness, with semi-improved species poor grassland, improved grassland, fields with no single dominant habitat and those not typical of moorland or farmland having the greatest species richness. Species poor habitats were rough grassland, dry dwarf shrub heath, blanket bog/ mire, unimproved grassland, enclosed upland acidic grassland, amenity grassland, wet heathland/mire, and rush pasture (Fig. 3.3).

Shannon-Wiener (*H'*), inverse Simpsons (1/D) and Berger-Parker indices all estimated that bird diversity was highest in habitats not typical of moorland or farmland and rush pasture, with broad agreement between these indices that diversity was high in fields with no single habitat with >75% coverage and heathland/ acid grassland mosaics (wet and dry). In addition, *H'*, 1/D and Berger-Parker estimated that rough grassland, amenity grassland, dry dwarf shrub heath were the least diverse (Table 3.3). Bird species diversity was most evenly distributed in upland acidic grassland, dry dwarf shrub health and blanket bog/ mire, with amenity grassland and improved grassland having the least even

diversity.

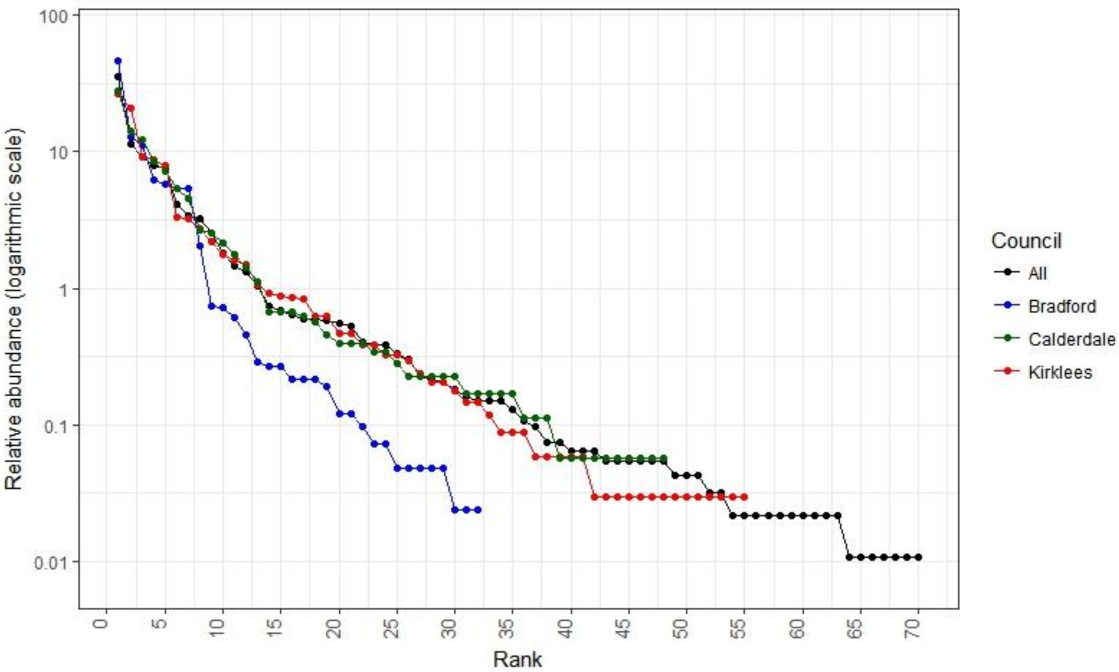

Figure 3.1 Rank abundance of moorland fringe bird species from moorland fringe habitats across all through unitary authorities.

Rarefaction analysis was difficult to interpret due to the large number of habitat categories used (Fig. 3.2), however after grouping number of individuals (n) by the sample size of each habitat type to aid comparability, some significant patterns became apparent (Fig.3.2). Significance here was determined by assessing whether confidence intervals overlap. This method is endorsed by the authors of *EstimateS* in the assessment of significance at $P \leq 0.05$ when comparing rarefaction curves (Colwell et al., 2004, 2012). The method is conservative, meaning that any significant differences found are likely to exceed the level of $P \leq 0.05$, however some true differences between species richness may be identified as not different (Colwell et al., 2012). The habitats of dry dwarf shrub heath, blanket bog and enclosed upland acidic grassland were grouped into the category of $n<14$ and did not show any difference in rarefied species richness. In the category of $n > 18$ but <69, wet heathland/ mire and unimproved grassland were of comparable rarefied species richness, however at equivalent number of individuals, the species richness of both of these habitats were significantly greater than that of rough grassland. Where $n > 111$ but <189 the three habitats of amenity grassland, dry heath/ acid grassland mosaic and wet heath/ acid grassland mosaic were all significantly different in species richness from one another. Of these, amenity grassland had the lowest species richness, followed by wet heathland/ acid grassland mosaic and then dry heath/ acid grassland mosaic. Where n is >253 but <670, fields with no dominant habitat were significantly greater in species richness than semi-improved species rich grassland and rush pasture. Semi-improved species rich grassland had significantly higher species richness than rush pasture. In the category of $n>$ 779 but < 3468, representing the most commonly encountered habitats, habitats not typical of moorland (‘other’) had a significantly greater species richness than improved grassland and semi-improved species poor grassland.

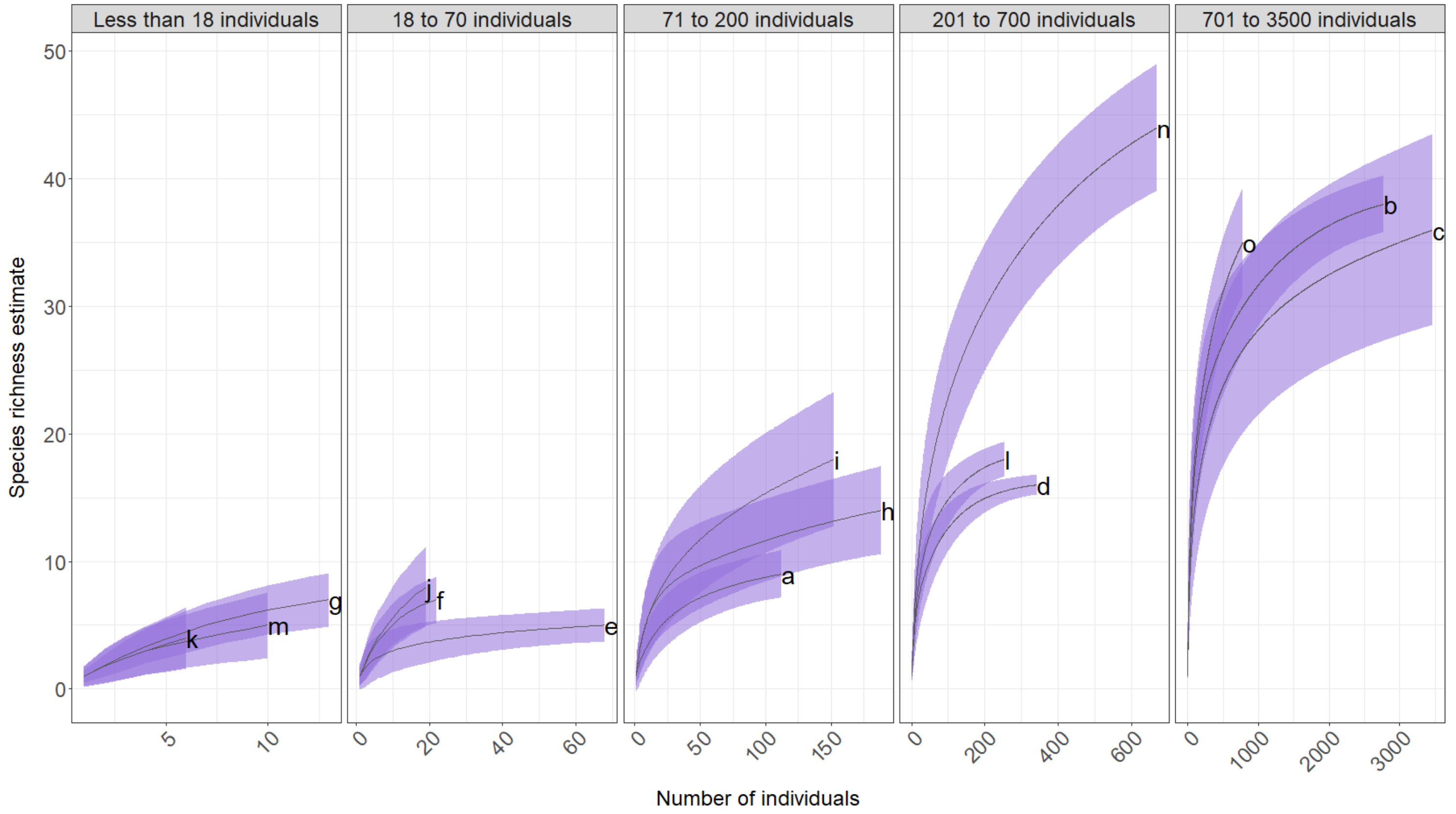

Figure 3.2 Rarefaction curves of estimated species richness by habitat across all councils, separated into groups by sample size (number of individual birds detected). Curves are presented side by side to facilitate comparison of species richness estimates between habitats. Habitats are indicated by letters; a = Amenity grassland; b = Improved grassland; c = Semi-improved species poor grassland; d = Semi-improved species rich grassland; e = Rough grassland; f = Unimproved grassland; g = Upland acidic grassland (enclosed); h = Wet heath/ acid grassland mosaic; i = Dry heath/ acid grassland mosaic; j = Wet heathland/ mire; k = Dry dwarf shrub heath; l = Rush pasture; m = Blanket bog/ mire; n = Other; o = No dominant habitat.

Extrapolation has the advantage of allowing comparability for all habitat type simultaneously. However, it has the disadvantage of larger confidence intervals for categories with low encounter rates of birds. Despite this, after extrapolation habitats that were not typical of moorland habitats were significantly greater in species richness than all other habitat types other than where fields did not contain a single dominant habitat, however this difference was close to significant based on the overlapping of confidence intervals (Figure 3.3). There appeared to be two distinct groupings of species richness with semi-improved species poor grassland, improved grassland, no dominant habitat and habitats not typical of moorland habitats all having significantly greater species richness than all other habitats except for dry heath/ acid grassland mosaic. The confidence intervals of dry heath/ acid grassland mosaic were too large for any meaningful differences between this and other habitats to be investigated through extrapolation (Figure 3.3).

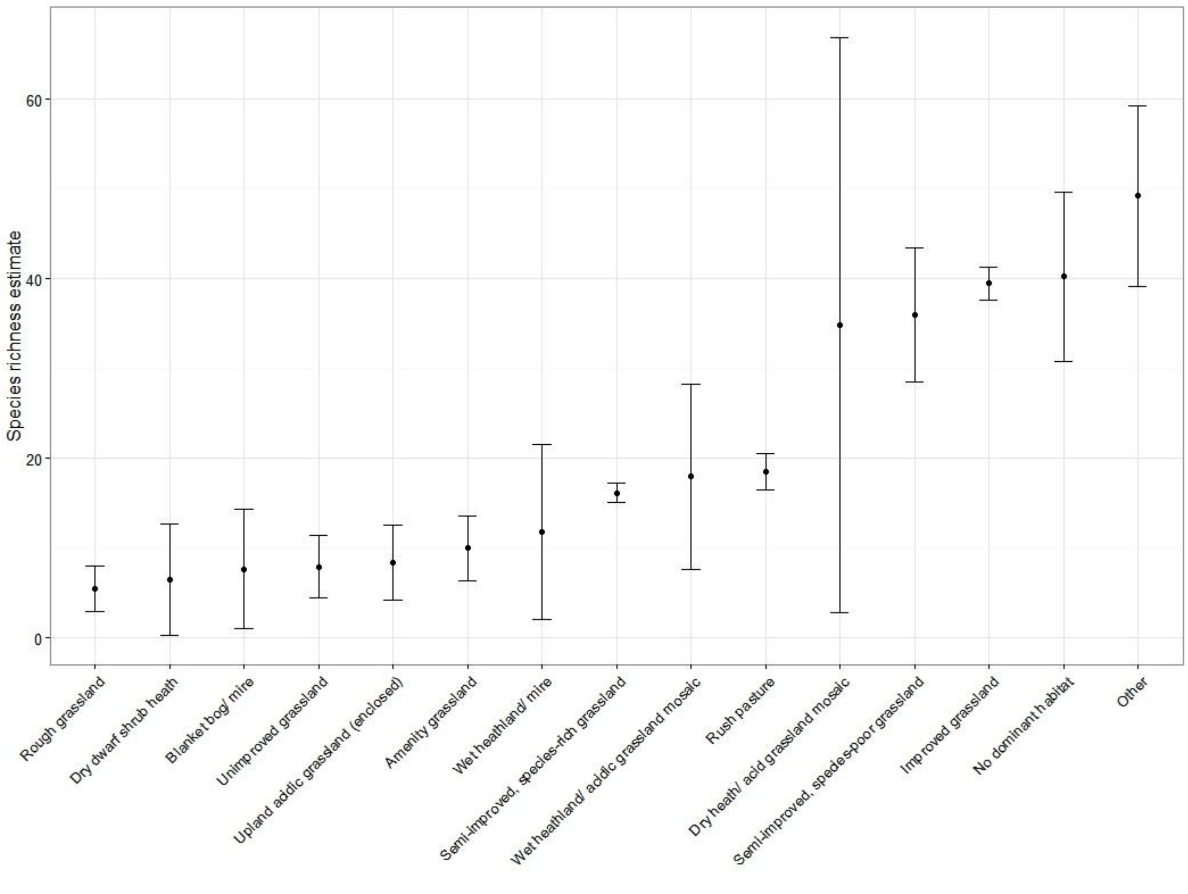

Figure 3.3 Extrapolated species richness with 95% confidence intervals of bird species recorded in each habitat across all three unitary authority regions.

There was considerable variation in the number of fields surveyed representing the different habitat types between the three council moorland fringe areas (Tables 3.4 to 3.6), with the most commonly encountered habitats corresponding to improved grassland and semi-improved species rich grassland across all three regions. Number of individual birds and observed species richness (S_{obs}) were highest in improved grassland at Kirklees (Table 3.3), whereas more individuals were recorded from semi-improved species-rich grassland habitat from the other two council areas (Tables 3.4 and 3.5). Observed species richness was highest in fields with no single habitat with >75% coverage at Calderdale and semi-improved species-rich grassland in Bradford. There was pronounced variation in the different measures of community composition between habitats within each of the three council areas (Tables 3.4, 3.5 and 3.6). Berger-Parker, Simpsons and Shannon-Wiener diversity indexes were in agreement that fields with no dominant habitat and habitats not typical of moorland/farmland were the most diverse habitats within Kirklees (Table 3.3). Similarly, fields with no dominant habitat were the most diverse according to these three diversity indexes in Bradford. However, in Calderdale neither of these habitat were the most diverse according to any diversity index, instead improved grassland and rush pasture were the most diverse habitats (Table 3.5).

Least diverse bird communities were found in semi-improved species poor grassland and amenity grassland in Kirklees according to the Berger-Parker index, with these habitats scoring low using Simpsons index and Shannon-Wiener also. Interestingly, species-rich semi improved grassland had lower biodiversity than species-poor semi improved grassland according to the Shannon-Wiener index in Kirklees, however sample size for species-rich semi improved grassland was low at 11 fields. In contrast to the relatively high diversity associated with fields not typical of moorland or farmland when the three councils are combined, this habitat was the lease diverse according to the Berger-Parker index and Simpsons index in Calderdale, suggesting that although diversity in habitat types is important at the landscape scale in maintaining bird species diversity, at the regional scale (i.e. between council areas), this is variable. Amenity grassland was lower in species diversity according to all three biodiversity indexes in Bradford by a considerable margin.

The habitat type with the greatest evenness in Kirklees was the category of habitats not typical of farmland or moorland. At face value this was not surprising, as multiple habitats were included in this category. However, this hypothesis does hold when applied to Calderdale where habitats not typical of moorland or farmland scored intermediate evenness. Rush pasture, wet heathland and dry heath/ acid grassland mosaics had the most even communities in Calderdale, whereas habitats not typical of moorland or farmland, Rush pasture and upland acid grassland supported the most even bird communities in Bradford. When habitats are compared between council moorland fringe areas, evenness was consistently higher in Kirklees than in Calderdale or Bradford. A graphical representation of all diversity indexes and evenness scores can be found in Figure 3.4.

Table 3.3 Measures of bird community composition in habitats within the SPMSPA fringe habitat across all three council areas. S(obs) = observed species richness; Sest = estimated species richness (extrapolated to 3,467 individuals) ± 95% confidence intervals; Berger-Parker = Berger-Parker diversity index; *H'* = Shannon-Weiner index; 1/D = Inverse Simpson's Diversity, *J'* = Shannon's measure of evenness; $E_{1/D}$ = Simpson's measure of evenness.

	Amenity grassland	Blanket bog/ mire	Dry dwarf shrub heath	Dry heath/ acid grassland mosaic	Improved grassland	Fields no single habitat >75% coverage	Habitats not typical moorland/farmland	Rough grassland	Rush pasture	Semi-improved, species-poor grassland	Semi-improved, species-rich grassland	Unimproved grassland	Upland acidic grassland (enclosed)	Wet heathland/ acidic grassland mosaic	Wet heathland/ mire
Number of fields	62	2	8	43	790	431	166	27	128	1112	68	9	17	36	4
Number of individuals	112	10	6	156	2834	863	669	68	254	3745	344	22	13	188	19
S (obs)	9	5	4	19	39	35	43	5	18	36	16	7	7	14	8
S (est) ± 95% CI	9.9 ± 3.6	7.7 ± 6.6	6.5 ± 6.2	34.9 ± 32	39.5 ± 1.8	40.3 ± 9.4	49.2 ± 10	5.5 ± 2.6	18.5 ± 2	36 ± 7.5	16.2 ± 1.1	7.9 ± 3.5	8.4 ± 4.2	17.9 ± 10.3	11.8 ± 9.7
Berger-Parker	0.68	0.4	0.5	0.28	0.47	0.24	0.2	0.63	0.3	0.39	0.3	0.45	0.31	0.24	0.42
H'	1.19	1.42	1.24	2.14	2.15	2.43	2.79	1.01	2.22	2.16	2.02	1.62	1.82	2.06	1.75
1/D	2.07	3.57	3	5.81	4.02	7.7	10.49	2.15	6.33	5.06	5.6	3.78	5.45	6.27	4.25
J'	0.54	0.88	0.9	0.73	0.59	0.68	0.74	0.63	0.77	0.6	0.73	0.83	0.93	0.78	0.84
$E_{1/D}$	0.36	0.83	0.87	0.45	0.22	0.33	0.38	0.55	0.51	0.24	0.47	0.72	0.88	0.56	0.72

Table 3.4 Measures of bird community composition in habitats within the SPMSPA fringe habitat in Kirklees council region. S(obs) = observed species richness; Berger-Parker = Berger-Parker diversity index; *H'* = Shannon-Weiner index; 1/D = Inverse Simpson's Diversity, *J'* = Shannon's measure of evenness; $E_{1/D}$ = Simpson's measure of evenness. NA indicates no community measure due to low sample size.

	Amenity grassland	Blanket bog/ mire	Dry dwarf shrub heath	Dry heath/ acid grassland mosaic	Improved grassland	Fields no single habitat >75% coverage	Habitats not typical moorland/farmland	Rush pasture	Semi-improved, species-poor grassland	Semi-improved, species-rich grassland	Unimproved grassland	Wet heathland/ acidic grassland mosaic	Wet heathland/ mire
Number of fields	3	1	2	26	215	65	27	43	118	11	0	4	0
Number of individuals	22	7	3	83	1411	183	413	144	594	92	0	4	0
S(obs)	4	4	1	13	26	18	30	13	13	6	0	3	0
Berger	0.64	0.57	NA	0.39	0.59	0.26	0.26	0.52	0.66	0.5	NA	0.5	NA
H'	1.03	1.15	NA	1.9	1.67	2.12	2.45	1.59	1.2	1.12	NA	1.04	NA
1/D	0.55	0.61	NA	0.78	0.63	0.83	0.87	0.68	0.53	0.6	NA	0.62	NA
J'	2.2	2.58	NA	4.61	2.69	5.92	7.79	3.08	2.14	2.49	NA	2.67	NA
$E_{1/D}$	1.43	3.88	0.53	4.33	4.53	4.95	7.43	3.47	2.35	1.44	NA	5.45	NA

Table 3.5 Measures of bird community composition in habitats within the SPMSPA fringe habitat in Calderdale council region. S(obs) = observed species richness; Berger-Parker = Berger-Parker diversity index; *H'* = Shannon-Weiner index; 1/D = Inverse Simpson's Diversity, *J'* = Shannon's measure of evenness; $E_{1/D}$ = Simpson's measure of evenness. NA indicates no community measure due to low sample size.

	Amenity grassland	Blanket bog/ mire	Dry dwarf shrub heath	Dry heath/ acid grassland mosaic	Improved grassland	Fields no single habitat >75% coverage	Habitats not typical moorland/farmland	Rush pasture	Semi-improved, species-poor grassland	Semi-improved, species-rich grassland	Unimproved grassland	Wet heathland/ acidic grassland mosaic	Wet heathland/ mire
Number of fields	10	1	4	5	274	119	31	78	353	3	0	32	4
Number of individuals	1	3	3	9	379	275	139	102	656	2	NA	184	19
S(obs)	1	1	3	6	22	24	20	12	23	1	NA	13	8
Berger	NA	NA	NA	0.22	0.26	0.32	0.55	0.23	0.53	NA	NA	0.25	0.42
H'	NA	NA	NA	1.74	2.34	2.24	1.82	2.06	1.74	NA	NA	2.01	1.75
1/D	NA	NA	NA	5.4	7.3	6.1	3.09	6.44	3.16	NA	NA	6.07	4.25
J'	NA	NA	NA	0.97	0.76	0.71	0.61	0.83	0.56	NA	NA	0.78	0.84
$E_{1/D}$	NA	NA	NA	0.94	0.47	0.39	0.31	0.66	0.25	NA	NA	0.57	0.72

Table 3.6 Measures of bird community composition in habitats within the SPMSPA fringe habitat in Bradford council region. S(obs) = observed species richness; Berger-Parker = Berger-Parker diversity index; *H'* = Shannon-Weiner index; 1/D = Inverse Simpson's Diversity, *J'* = Shannon's measure of evenness; $E_{1/D}$ = Simpson's measure of evenness. NA indicates no community measure due to low sample size.

	Amenity grassland	Dry dwarf shrub heath	Dry heath/ acid grassland mosaic	Improved grassland	Fields no single habitat >75% coverage	Habitats not typical moorland/farmland	Rough grassland	Rush pasture	Semi-improved, species-poor grassland	Semi-improved, species-rich grassland	Unimproved grassland	Upland acidic grassland (enclosed)
Number of fields	48	2	11	274	229	100	27	3	623	53	9	17
Number of individuals	89	0	60	979	322	117	68	8	2217	249	22	13
S(obs)	6	0	5	20	16	9	5	4	26	12	7	7
Berger	0.85	NA	0.73	0.41	0.29	0.24	0.63	0.5	0.51	0.42	0.45	0.31
H'	0.62	NA	0.91	1.88	2.1	1.84	1.01	1.21	1.76	1.8	1.62	1.82
1/D	1.36	NA	1.78	4.27	6.31	5.62	2.15	2.91	3.41	4.19	3.78	5.45
J'	0.35	NA	0.56	0.63	0.76	0.84	0.63	0.88	0.54	0.73	0.83	0.93
$E_{1/D}$	0.31	NA	0.5	0.33	0.51	0.7	0.55	0.84	0.22	0.51	0.72	0.88

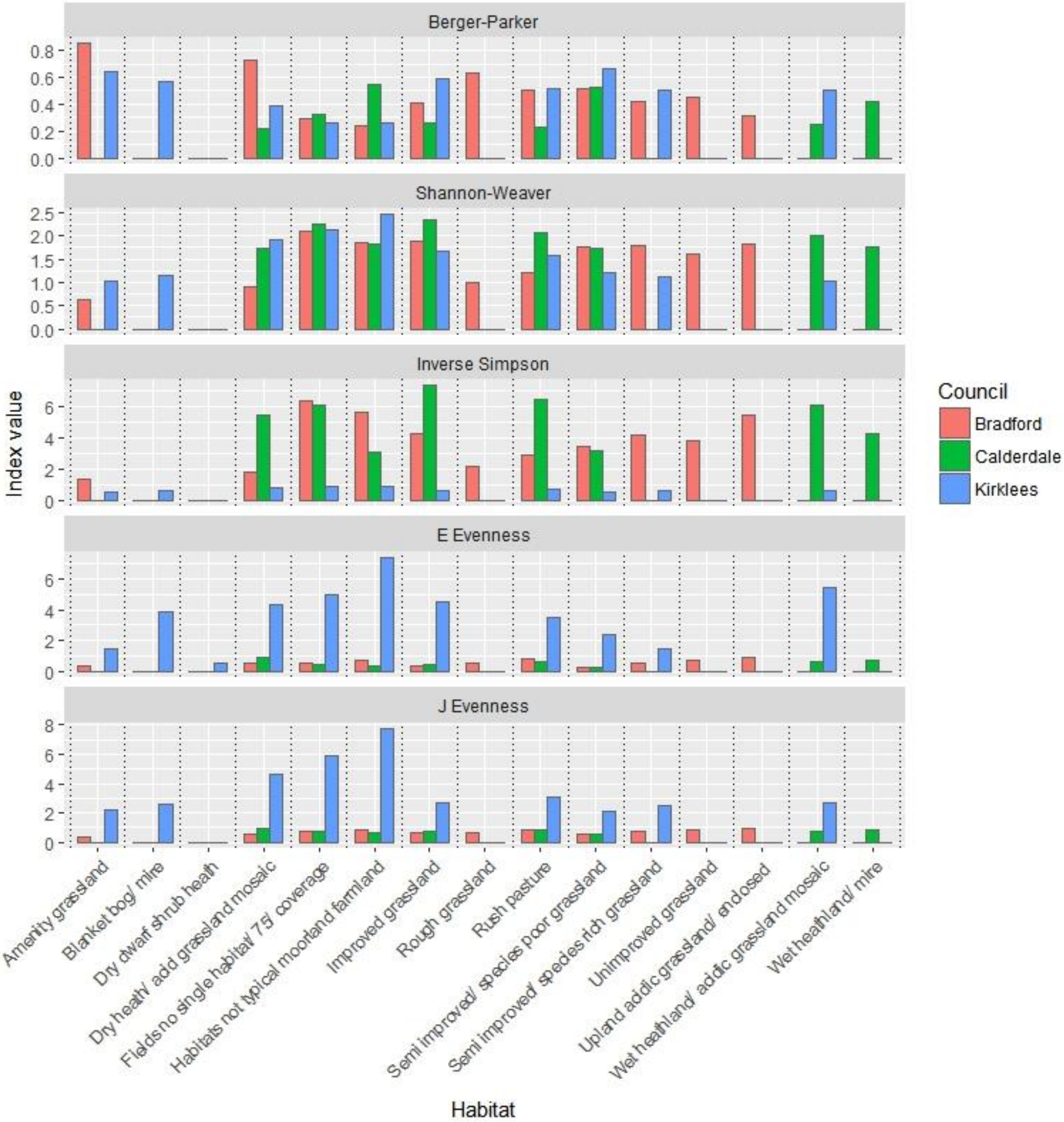

Figure 3.4 Relative diversity index scores between council moorland fringe areas and between habitat types.

3.4.2. Moorland fringe habitat gradients

Non-metric multidimensional scaling (NMDS) analysis resulted in three habitat gradients across the moorland fringe landscape. The loadings for NMDS1 (Fig 3.5) suggest that this axis corresponded to a habitat gradient spanning fields that are at least partially wet (i.e. contain a flush), which contain tussocks and are grazed intensively (negative end of the axis) to fields that have been managed through grass cutting (positive end of the axis). NMDS1 therefore corresponded to a gradient from fields that are typically upland in their characteristics, but still actively used for grazing, reflecting traditional methods of land management close to or encroaching onto moorland where sheep grazing predominates.

The other end of the gradient reflects fields that may be cut for silage or hay (fields that are used for the growth of crops for feedstock) but with no single habitat dominating NMDS1.

Fields described by NMDS2 (Fig. 3.5) correspond to a habitat gradient spanning predominantly improved grassland that are grazed (negative end of the axis) to fields that are at least partially wet (i.e. contain a flush), contain tussocks and are dominated by semi-improved grassland (positive end). Thus, NMDS2 describes a gradient spanning fields that are composed of lush grass that are used for intensive grazing, to fields that are less intensively managed and ungrazed. Fields at the positive end of NMDS2 fall broadly in line with field that are typical of 'set-aside'- areas that may be under Environmental Stewardship (ES) subsidy which have been left unfarmed to promote biodiversity, whereas the fields at the negative end of the axis are typical of fields that are not in receipt of ES subsidies and are intensively farmed purely for agricultural output.

Interpretation of NMDS3 characteristics (Fig. 3.4) were less clear at the positive end of the axis, due to relatively low loading values for all variables, but the presence of tussocks, flush and rush vegetation were common. The other end of the gradient was composed of fields dominated by semi-improved grassland, suggesting that the NMDS3 gradient incorporates fields that are less intensively managed (wildflower richness and molehills also featured at the negative end of NMDS3), however these fields appear to grazed intensively. The habitat gradient moves toward fields that featured fields dominated by rushes along with flushes, tussocks and dry stone walls. This may reflect fields that are on the very edge of the SPA that are wet and more typical of upland farmland, however the fact that improved grassland is included at this end of the gradient makes interpretation less clear.

3.4.3. Bird-habitat associations

NMDS stress is a measure of the goodness of fit of the resultant ordination feature space as compared to the original data feature space, where zero is equal to perfect representation (Boyra et al., 2004). Three-dimensional NMDS ordination resulted in a stress value of 0.16, with convergence in stress achieved within 50 permutations. Pairwise biplots of the first two NMDS dimensions revealed significant overlap between the habitat associations of Curlew, Lapwing, Snipe, Golden Plover and Wheatear, however the associations of Snipe and Golden Plover appeared more constrained and directional than other species (Fig. 3.5). Snipe were associated with the presence of tussocks, flush, rush pasture, wildflower diversity and gradient, whereas Golden Plover were associated with similar variables (Fig. 3.4). The presence of these five conservation-priority species ordinated along the negative end of the NMDS1 gradient, whereas less presence/absence association was evident with either NMDS2 or NMDS3, except for Snipe.

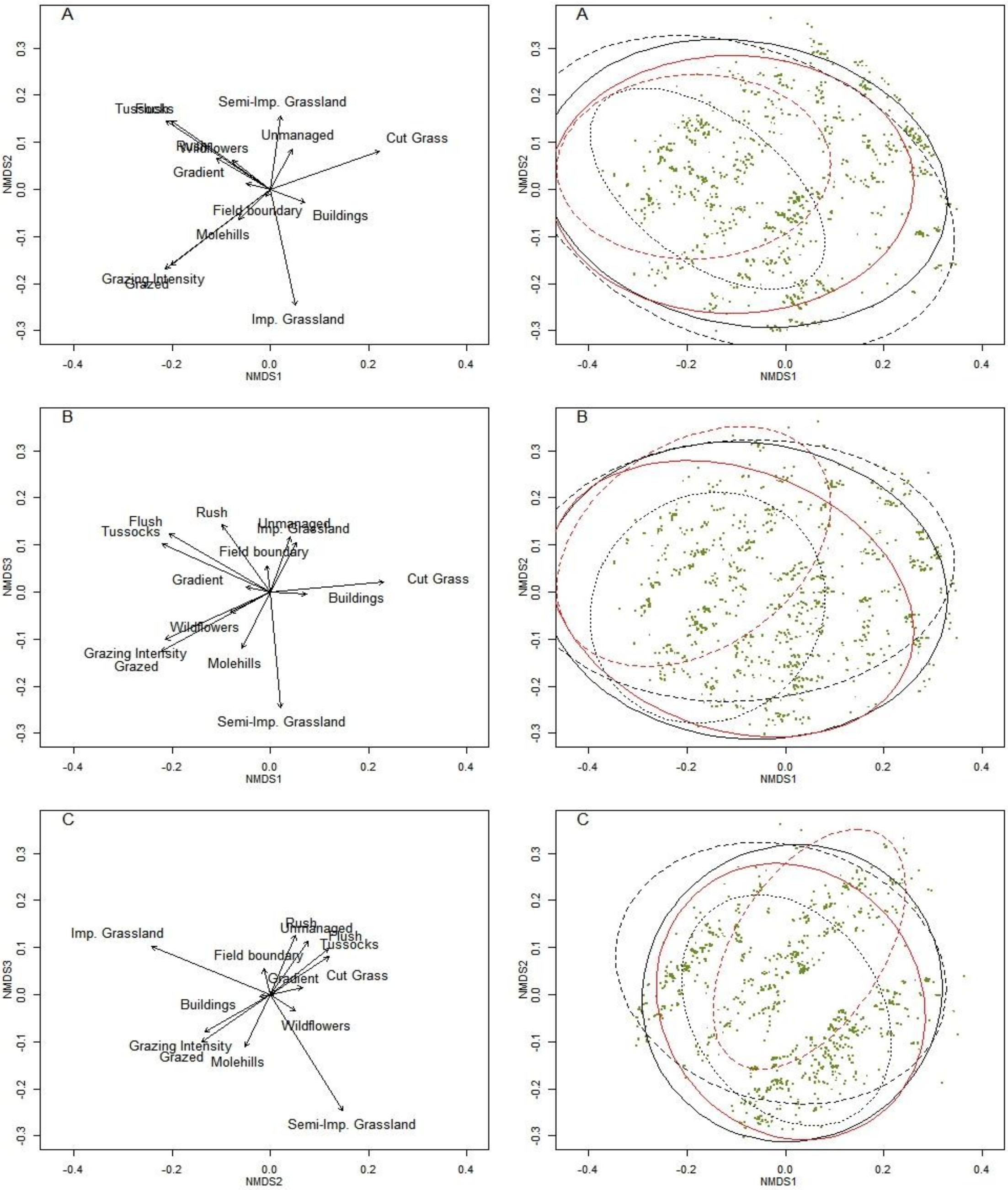

Figure 3.5 NMDS ordination showing pairwise biplots for three axes of habitat variability (3.5A shows NMDS1 vs NMDS2; 3.5B shows NMDS1 vs NMDS3; 3.5C shows NMDS2 vs NMDS3). Loadings for habitat characteristics are displayed in the left column of figures, with the length of arrows indicating association with NMDS axes. The column on the right shows individual fields as green points. 95% confidence ellipses are shown for Curlew (black solid line), Lapwing (red solid line), Golden Plover (black dotted line), Wheatear (black dashed line) and Snipe (red dashed line), indicating fields associated with these species.

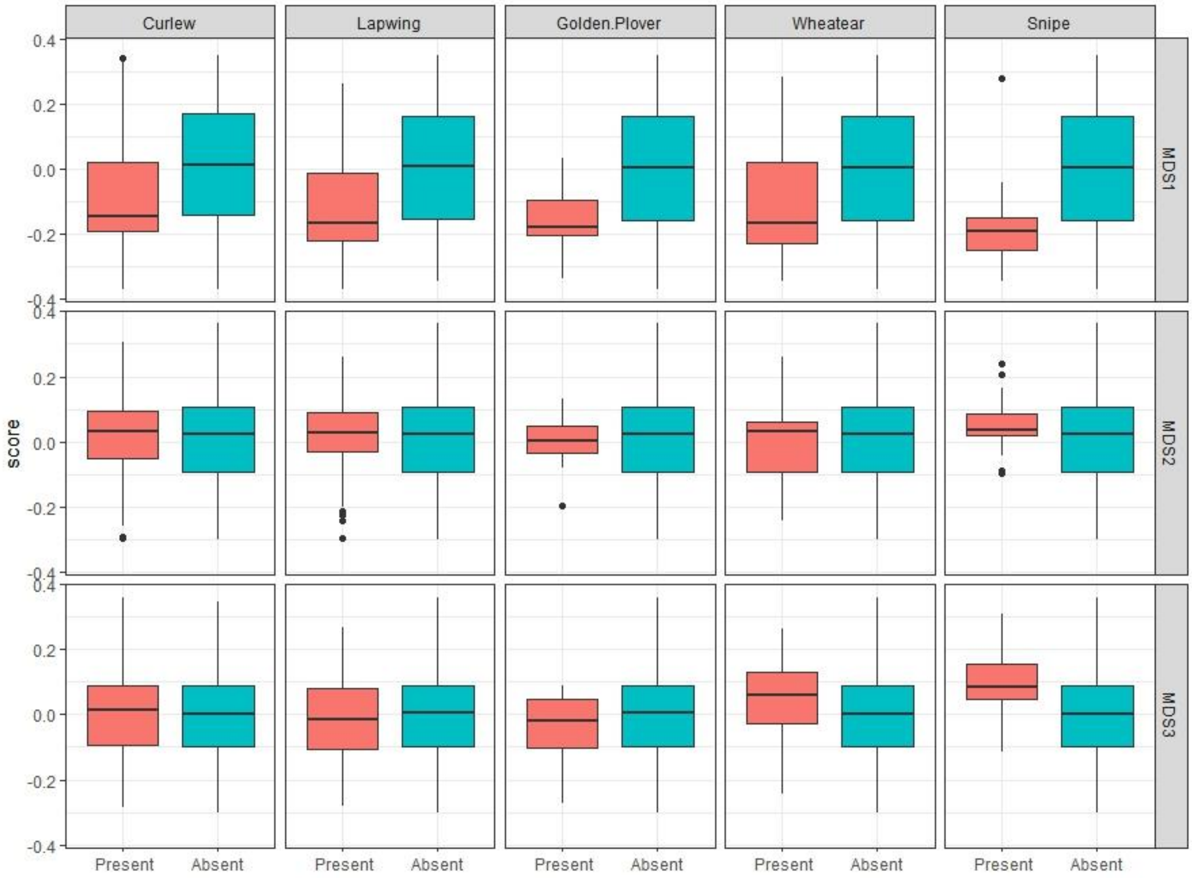

Figure 3.6 Distribution of non-metric multidimensional scaling ordination scores for fields with presence or absence of Curlew, Lapwing, Golden Plover, Wheatear and Snipe. Statistical associations are shown in Table 3.7.

Examination of the distribution of NMDS ordination scores (Fig 3.6) reveals a clear differentiation in the axis scores for presence versus absence for all five species across NMDS1. All species showed clear association for presence with negative scores whilst absences were distributed over the entire range of NMDS1 scores (Fig 3.6). Differences between the distribution of presence and absence ordination scores across NMDS2 were less clear, however Golden Plover and Snipe appeared to be associated with central scores, indicating that these species do not prefer the extremes at either end of this habitat gradient. (Fig 3.6). The distribution of NMDS3 scores were similar for Curlew, Lapwing and Golden Plover, in contrast to Snipe and Wheatear where presences appeared to favour the positive end of the habitat gradient, especially for Snipe (Fig 3.6). The best explanatory combinations of habitat gradients NMDS1, NMDS2 and NMDS3 were determined using Generalised Additive Models (GAMs). The Statistical relationships between habitat gradients used in the best performing GAMs as determined by Akaike Information

Criterion (AIC) and the five bird species were tested for significance using a chi-squared test.

The results of Generalised Additive Models (GAMs) exploring the relationship between the five analysed bird species and three habitat gradients represented by the NMDS axes are presented in Table 3.7. The habitat gradient described by NMDS1 was chosen in all best performing models, with significant associations found in all species (table 3.7). Curlew was the only species that was best described solely by NMDS1 with a complex relationship relative to other species showing a general trend towards presence of this species at the negative end of the gradient (Figure 3.7a). This pattern of association with NMDS1 is more pronounced in Lapwing and Snipe (Figure 3.7b and 3.7g), where automated spline choice was simplified to a linear relationship. Although the relationships between NMDS1 and Golden Plover and NMDS1 and Wheatear appear less pronounced (Figures 3.7d and 3.7j), there was a significant decrease in presences towards the positive end of the gradient (Table 3.7). These results show that all five species have a preference for fields that are less intensively managed for agriculture and are more similar to the core SPA habitats in their characteristics than intensively managed fields that are cut for silage or hay. Although NMDS2 contributed to the best models for Golden Plover and Snipe, these associations were not significant and AIC was not greatly higher than that of the next best model (Table 3.7). Taking this into account along with the fact that NMDS2 played no role in the best performing models for Curlew, Lapwing or Wheatear, it is fair to say that that these species showed no preference for any fields along a gradient from high agricultural output improved fields to land that has been removed from active agriculture, perhaps intentionally to promote biodiversity under ES. The habitat gradient represented by NMDS3 contributed to the best performing models for all species except Curlew. For Lapwing and Golden Plover however this contribution was not significant. For Wheatear, the association with NMDS3 was near significant (P=0.06), and for Snipe NMDS3 contributed significantly in explaining presence of this species (Table 3.7). The presence of Snipe and Wheatear were linearly positively associated with the values of the NMDS3 axis (Figure 3.7i and 3.7j). This suggests that these species have a preference for wet fields with Tussocks and Rush and are significantly unassociated with semi-improved grassland.

Table 3.7 Generalised Additive Models (GAMs) showing influence of the three NMDS ordination habitat gradients (described in the text) on the relative abundance of five conservation-priority bird species. Significance of association was testing using χ^2 where na = not applicable as the habitat gradient was not included in the best model. AIC = Akaike's Information Criteria; ΔAIC = change in AIC values between models. Lower AIC values indicate better models.

Species	**Model (* indicates best performing model)**	**AIC**	**ΔAIC**	**Contribution of habitat gradients to best performing model**
Curlew	NMDS1*	1639.6	0.0	NMDS1: χ^2=99.43, P<0.001 NMDS2 = na NMDS3 = na
	NMDS2	1718.2	78.6	
	NMDS3	1738.0	98.4	
	NMDS1+NMDS2	1641.3	1.7	
	NMDS1+NMDS3	1641.3	1.7	
	NMDS2+MDS3	1719.8	80.2	
	NMDS1+NMDS2+NMDS3	1643.0	3.4	
Lapwing	NMDS1	720.8	0.4	NMDS1: χ^2= 34.62, P<0.001 NMDS2 = na NMDS3: z= 3.22, P=0.24
	NMDS2	751.6	31.2	
	NMDS3	760.4	40.1	
	NMDS1 + NMDS2	721.3	1.0	
	NMDS1 + NMDS3*	720.4	0.0	
	NMDS2 + NMDS3	750.9	30.6	
	NMDS1+NMDS2+MDS3	720.8	0.4	
Golden Plover	NMDS1	234.5	6.3	NMDS1 χ^2=- 8.36, P= 0.02 NMDS2: χ^2= 4.03, P=0.24 NMDS3: χ^2= 5.93, P=0.17
	NMDS2	249.2	21.0	
	NMDS3	251.1	23.0	
	NMDS1 + NMDS2	233.7	5.5	
	NMDS1 + NMDS3	229.5	1.3	
	NMDS2 + NMDS3	241.6	13.4	
	NMDS1+NMDS2+NMDS3*	228.2	0.0	
Snipe	NMDS1	289.0	15.5	NMDS1: χ^2= 16.96, P<0.001 NMDS2: χ^2=3.46, p=0.46 NMDS3: χ^2=13.13, P<0.001
	NMDS2	321.1	47.6	
	NMDS3	315.1	41.6	
	NMDS1 + NMDS2	287.4	13.9	
	NMDS1 + NMDS3	274.4	0.8	
	NMDS2 + NMDS3	300.5	27.0	
	NMDS1+NMDS2+NMDS3*	273.5	0.0	
Wheatear	NMDS1	331.9	1.4	NMDS1: χ^2= 7.54, P<0.05 NMDS2 = na NMDS3: χ^2=3.48, P=0.06
	NMDS2	342.1	11.6	
	NMDS3	338.1	7.5	
	NMDS1 + NMDS2	333.4	2.8	
	NMDS1 + NMDS3*	330.5	0.0	
	NMDS2 + NMDS3	339.8	9.3	
	NMDS1+NMDS2+NMDS3	331.9	1.4	

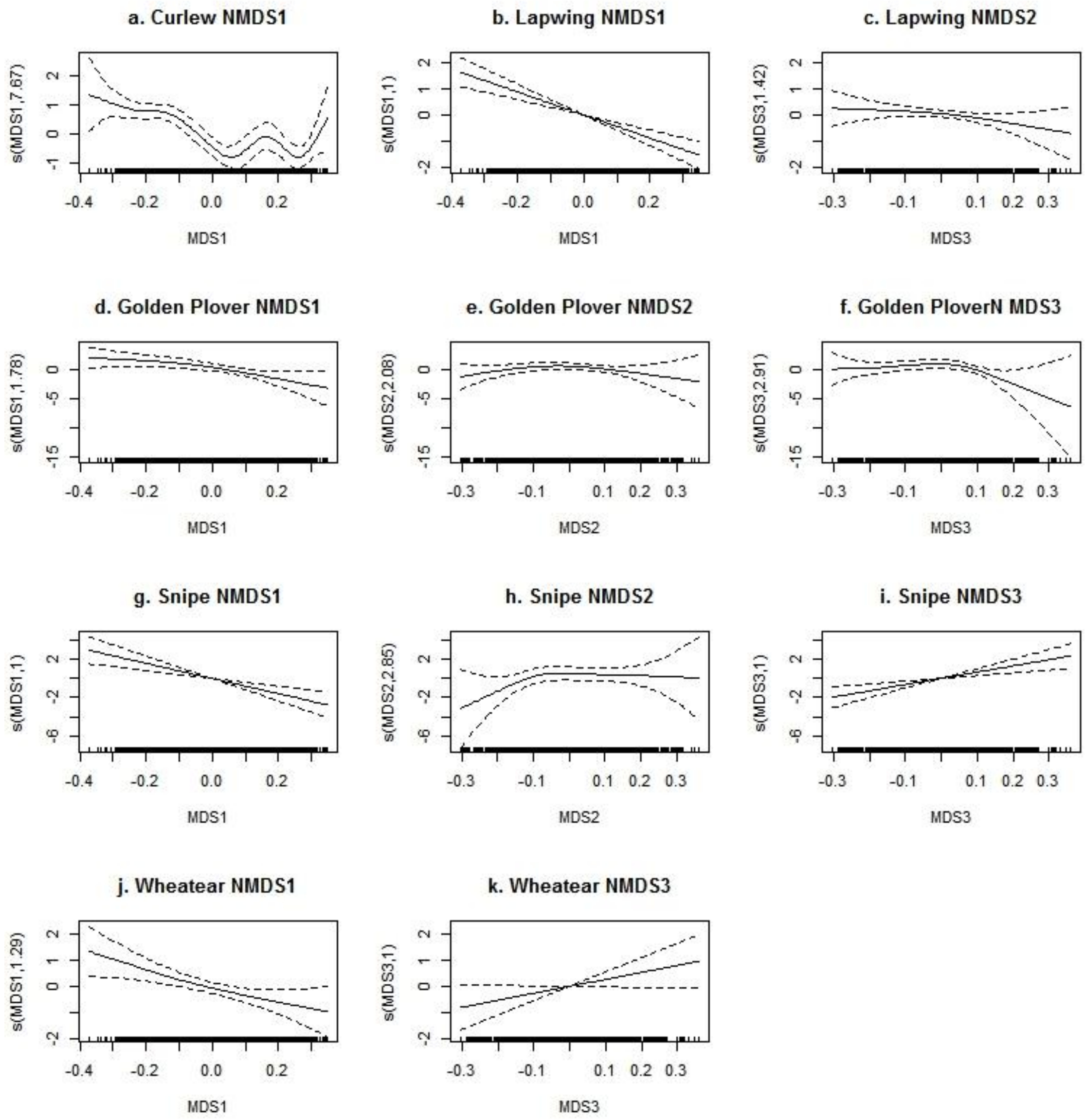

Figure 3.7a-k The relationship between the abundance of the five conservation-priority bird species, and the NMDS habitat gradients that exhibited significant influences on bird abundances identified from Generalized Additive Models (GAMs).

3.5. Discussion

Fields with a heterogeneous habitat arrangement were estimated to have amongst the highest bird diversity by all calculated diversity indices. Heterogeneous fields were represented by heathland and acid grassland mosaics, fields with no single dominant habitat and fields with habitats not typical of moorland or moorland fringe farmland. Maintaining habitat heterogeneity within upland vegetative communities has long been understood to be of conservation importance for upland bird species diversity in the UK (Usher and Thompson, 1993) and similarly for farmland bird species, where bird species with different life history strategies require structural variation to accommodate for variety in predator avoidance responses, feeding requirements and breeding behaviour (Benton et al., 2003). The results of this study show that habitat heterogeneity at the landscape level is important for the conservation of bird diversity in moorland fringe. From a management perspective, this means incorporating habitat types into the SPMSPA fringe that are not typical of moorland or farmland such as woodland, waterbodies and gardens. In addition to this, bird diversity was high in fields with no single dominant habitat, showing that heterogeneity of habitats at the field level is also important for maintaining high bird diversity. Previous studies have shown that in-field heterogeneity is also important for breeding waders including Lapwing and Redshank (Verhulst et al., 2011), highlighting that maintaining a variety of habitats within individual field may benefit birds associated with the SPMSPA as well as bird diversity as a whole. This highlights the importance of taking a multiscale approach to the conservation of birds in moorland SPA fringe areas where bird diversity is a priority. Many other studies have advocated a multiscale approach to studying and conserving bird diversity in a broad range of habitats including urban environments (Jokimäki and Kaisanlahti-Jokimäki, 2003), rural-urban interfaces (Taylor et al., 2016), woodland (Grand et al., 2004) , grassland (Thompson et al., 2014), farmland (Rudolphi et al., 2014) and upland habitats (Mahon et al., 2016). Multiscale approaches to bird conservation have been used extensively to study the effects of anthropogenic stresses including wildfire (Herrando and Brotons, 2002), urbanisation (Gagné et al., 2016) and agricultural intensification (Jeliazkov et al., 2016). This is often achieved through spatially oriented predictive modelling within a species distribution modelling or habitat suitability modelling framework. This will be discussed further in Chapter Five.

Species richness and evenness were lower within the SPA fringe of the unitary authority of Bradford than in Calderdale or Kirklees. Additionally, species richness was much higher when all three authorities were combined than any one individually. These results suggest that the combined efforts of multiple decision making authorities with joint jurisdiction over a protected area are highly important in preserving bird diversity. The importance of cross boundary co-operation in the management of protected areas is recognised at the Country level (i.e. where protected areas span two or more nations) to promote and enhance biodiversity corridors without anthropogenic barriers to movement (Zimmerer et al., 2004). This concept can be extended to boundaries at other spatial and political scales including unitary authorities, where distinct decision making units are responsible for managing the same landscape. One could argue that it is indeed more important at this scale, where in addition to cross boundary ecological similarity, the political motivations and overarching policy goals are set by a central government (Westminster and Brussels in the case of English SPAs) and therefore conservation goals should align between local authorities.

The presence of Curlew, Golden Plover, Snipe, Lapwing and Wheatear were all significantly negatively associated with fields described by the habitat gradient presented by NMDS1, i.e. a gradient from fields with wet flush and tussocks that are heavily grazed (negative end of the habitat gradient) to fields where the vegetation is mechanically cut (positive end of the habitat gradient). This could be advantageous from a landscape management perspective as encouraging the implementation of practices that broadly match the negative end of this habitat gradient is likely to benefit all five of these species. Using habitat gradients to inform management practices has the advantage of maintaining generalisation and is less prescriptive than using individual habitat components. This allows an approach to be undertaken that is less concerned with individual field composition (which may result in field heterogeneity, but increase homogeneity at the landscape level) and instead allows a more casual approach to management. In general terms for the five species, grazing is more beneficial than cutting, fields with a waterlogged portion and a high proportion of tussocks are more important than wildflower diversity or molehill cover. The importance of grazing and ground wetness has previously been shown for the presence of breeding waders such as Lapwing, Redshank and Black-tailed Godwit *Limosa limosa* (Tichit et al., 2005; Smart et al., 2006; Verhulst et al., 2011) at wetland and coastal sites. This study shows that the same is true for upland waders and passerines using the moorland fringe. The habitat gradient described by NMDS3 had an influence in addition to NMDS1 over the presence or absence of Snipe and Wheatear. Whereas NMDS1 shows field improvement/ semi-improvement and the presence of dry stone walls to have relatively little influence over the presence of Curlew, Lapwing and Golden-Plover in comparison to other habitat characteristics, this is not true for Snipe or Wheatear. For these species, (especially Snipe) there is a strong preference against semi-improved grassland and a strong preference for fields with a dominant cover of rush. Molehill cover does not appear to have a positive effect on the presence of Snipe or Wheatear. Molehill abundance has been shown to be positively related to the presence of some bird species, with the hypothesis that molehills indicate a proxy for earthworm abundance (Atkinson et al., 2005). The results of this study suggest that earthworms are not an important food source for Snipe or Wheatear, or that molehills are not a good indicator of earthworm abundance.

Of the 13 species of conservation importance identified as part of this study, less than half were encountered frequently enough within the SPMSPA fringe for habitat association analysis. These species were Curlew, Lapwing, Golden Plover, Wheatear and Snipe. An explanation for the fact that eight species were only found in low numbers could lie in that most the SPMSPA fringe landscape was composed of habitats mainly associated with farmland as opposed to habitats typical of moorland (see Chapter Two). Breeding Curlew, Lapwing and Snipe have all been shown to be associated with farmland habitats, especially less intensively managed (i.e. less improved) farmland (Henderson et al., 2002). In contrast to this, Curlew and Snipe have also been shown to be associated with improved grassland cover by Whittingham et al. (2010). Golden Plover have been shown to feed in enclosed farmland, with males and females commuting to these habitats at night and day respectively with greater commuting distances in the day (Pearce-Higgins and Yalden, 2003). As bird surveys undertaken as part of this study were only undertaken during the day, it is possible that an incomplete picture of the habitat associations of this species have been explored. Golden Plover are associated with cotton grass and other sedges at the field level (Dallimer, Marini, et al., 2010), suggesting that this species has strong preference against improved fields. Molehills have been shown to be a good predictor of Golden Plover foraging in enclosed fields, possibly representing a surrogate for earthworm abundance (i.e. prey availability) hence their inclusion in this study (Whittingham et al., 2000). Species with very low encounter rates included Short-eared Owl, Merlin, Dunlin, Ring Ouzel and Whinchat. These species are generally associated with typical upland habitats such as heather, heather/ grassland mosaics, blanket bog (Stillman and Brown, 1994; Buchanan et al., 2003). Twite are known to use habitat within moorland fringe (Wilkinson and Wilson, 2010), preferring sites that are close to water bodies and low in rush cover (Brown et al., 1995) and with high coverage of flower meadows (Langston et al., 2006). Unfortunately, this species has declined markedly in its breeding population and distribution across the whole of the UK, and indeed within the SPMSPA (Raine et al., 2009). The last available data in the literature are from 2004/05 showing only 10 known historical Twite breeding sites remaining in Lancashire and West Yorkshire out of 43 in 1967/68 (Raine et al., 2009). Taking this into account, the low encounter rate of Twite in this study is not surprising. Redshank are known to use moorland fringe habitats including enclosed meadows (Jefferson, 2005; Moss et al., 2005), however have shown population declines in recent years in upland habitats (Jefferson, 2005). Breeding Common Sandpiper are associated with the banks of waterbodies such as reservoirs and rivers in the uplands (Holland and Yalden, 1991; Yalden, 1992), a component of the SPMPSA that may require further investigation to determine the assosications of Common Sandpiper.

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CHAPTER 4: INFLUENCE OF SMALL WIND TURBINES ON MOORLAND FRINGE BIRDS

4.1. Abstract

There is a considerable body of literature on the ecological impacts of large wind turbines and wind farms, however there is almost no scientific literature exploring the ecological effects of Small Wind Turbines (SWTs). Here we adopt the definition of an SWT to mean a wind turbine of energy generating capability <50kW. These are usually free-standing turbines that are used to supplement electricity supply in a domestic setting or on a farm. This chapter aims to investigate the ecological effect of SWTs on bird communities and the presence of moorland fringe bird species. The aims of the chapters are; (1) To determine habitat composition around SWTs; (2) To determine bird community composition around SWTs; (3) To investigate the effect of distance from SWTs on the presence of bird species.

Bird surveys were at undertaken at 16 SWT sites. Habitat surveys were undertaken along the same transects used for bird surveys. Habitat heterogeneity was assessed between turbine sites and between distance bands from as were bird-habitat associations. Logistic regression was used to determine the effect of distance from SWTs on birds.

A total of 16 different habitat types were recorded around SWTs. No differences in habitat between distance bands from turbines were detected, however habitat composition differed significantly between turbine sites. The surveyed bird community around SWTs comprised 54 species. Species diversity was lowest within 100m of SWTs, suggesting that that there may be a displacement effect within 100m of SWTs. Magpie *Pica pica* and Starling *Sternus vulgaris* were found to be significantly associated with distance from SWT when controlling for habitat type. The effect of distance from SWTs is discussed in the light of these results and results from previous studies on larger turbines.

4.2. Introduction

Research into the ecological effects of wind turbines has largely focussed on wind farms with multiple large turbines. With financial incentives available within the UK (as of 2013) for small-scale electricity generation, there is an increasing trend towards the construction of small wind turbines (SWTs) in areas of high wind resource availability. Consistent terminology in the scientific literature is regarded by many as key to the mutual understanding of concepts between scientists. 'Small Wind Turbine' (Minderman et al., 2012) and 'Micro-Turbine' (Park et al. 2013) are often used to mean an electricity generating wind turbine of generating capability <50kW. The international safety standard for SWTs states that Micro-Turbines have a generating output of <500W, whereas SWTs have <50kW output. As the latter definition best fits the turbines investigated by (Minderman et al. (2012), (Park et al. (2013) and by the research conducted in this chapter, the term SWT will be used here. In addition to power output, SWTs are defined by a swept rotor area of <200m^2 which translates to a rotor length of approximately 8m. The ecological effects of SWTs on UK biodiversity are not well understood, making it difficult for local authorities to make informed planning decisions (Minderman et al., 2012). Some studies have addressed the issue of integrating ecological evidence into planning policy, using the lack of empirical ecological evidence regarding SWTs as an example for advocating better communication between scientists and policy makers and planning departments (Park et al. 2013). There is a widespread misconception that the threat of wind turbines on birds is limited solely to the potential for bird strike (Leung and Yang, 2012). This is not aided by the fact that the majority of research attempting to reconcile bird ecology and wind turbines appears biased towards collision risk and direct mortality (e.g. De Lucas et al. 2008; Ferrer et al. 2012; Péron et al. 2013). A considerable body of research has focussed on the collision mortality of birds with onshore wind turbines, especially with regards to raptors (e.g. Barrios & Rodríguez 2004; De Lucas et al. 2008; Schaub 2012; Dahl et al. 2013; Hull & Muir 2013). Similarly, there is much research into the bird collision risk of offshore turbines for numerous migratory and marine birds (e.g. Plonczkier & Simms 2012; Johnston et al. 2014). Determining the rate or risk of collision is of ecological significance to bird populations is extremely complex, as it is deemed to be species specific, location specific, and size specific (in terms of the size of a wind farm and the turbines), associated with topography, weather, season and land (Herrera-Alsina et al., 2013). This multitude of variables make it difficult to determine in advance whether a wind turbine development may affect a bird population (Powlesland, 2009). Collision risk however is only one of many factors that could present a potential threat to the viability of bird populations around wind turbines. Other threats include displacement as a result of disturbance, habitat loss or degradation, and the creation of 'barriers' (i.e. the ' barrier effect') altering migration or daily movement patterns (Drewitt and Langston, 2006; Masden et al., 2009, 2010; Plonczkier and Simms, 2012; Winiarski et al., 2014).

Using standardized pre-construction surveys, informed placement of turbines can theoretically minimise these negative impacts (Madders and Whitfield, 2006). The current consensus appears to be that prior monitoring of a proposed wind turbine site for bird activity and placement based on a 'least impact' basis is the best way to minimise risk, i.e. by conducting an Environmental Impact Assessment (EIA) (Desholm et al., 2006). Adopting EIAs seems logical and relatively simple, but different guilds of birds require different survey methodologies, different seasonal emphasis, and in some cases long term monitoring, covering several years in order to make sound estimates of abundance and distribution (Niemuth et al., 2013). Furthermore, there is some evidence to suggest that the spatial arrangement of turbines within the landscape can affect bird species such as Red Kite (*Milvus milvus*) (Schaub, 2012). An approach has been proposed that involves pre-empting conflict at the landscape level (Bright et al., 2008) which involves avoiding the overlap of turbine location with areas of importance to birds that present a high turbine risk, based on foraging range, collision risk and sensitivity to disturbance (Bright et al., 2008). In this chapter the effect of Small Wind Turbines (SWTs) on moorland fringe bird species is investigated. Specifically, the following research questions apply: (1) does the habitat composition around SWTs differ as a function of distance from SWTs? (2) Does habitat composition differ between SWT sites? (3) do measures of bird community composition differ with distance from SWTs; (4) is the presence of individual bird species affected by proximity to SWTs?

4.3. Methods

4.3.1. Study site

Bird and habitat surveys were conducted at wind turbine sites located within the South Pennine Moors Special Protection Area moorland fringe landscape. Details of the SPMSPA and the surrounding fringe are provided in Chapter Two. In order to select suitable sites, the UK planning portal database was searched using various synonyms and variations of the keyword 'turbine' in the districts of Calderdale, Bradford and Kirklees. Turbines fitting the defined criteria of SWT and having gained planning permission were identified, and these sites were subsequently reduced to turbines within 3 km of the SPMSPA boundary. All sites that met these criteria were visited and inspected for construction status prior to ecological survey (n= 95). Turbines that were positively identified as constructed were split into two categories: (1) *Individual turbines* that were more than 1 km in distance from any other SWT (n=16); (2) *Cluster of turbines* where two or more turbines were less than 1 km apart (n=42). Buffers were calculated and drawn around the turbine locations (as defined in the planning permission documentation) in ArcGIS, at 100m radial intervals up to 500m, hereon referred to as 'distance bands'.

4.3.2. Bird surveys

The bird survey method was based on the British Trust for Ornithology (BTO) Common Bird Census (CBC) and Breeding Bird Survey (BBS) methods (Marchant 1983; Risely et al 2013). Survey transects following public rights of way that crossed as many distance bands as possible, with minimal intersection as possible were identified and selected from ordnance survey maps. A total of 23 turbines were selected for surveys, comprising of both individual turbines (n=15), and one cluster of turbines (n=8) (Appendix 4). Between 16th May 2013 and 18th July 2013, a total of 65.4km of line transect were surveyed, a period representing the breeding bird season in the UK. Transect surveys were categorised into early and late breeding season period, with the early period ending on 14th June 2013 and the late period beginning on 30th June 2013. Most turbine sites were surveyed twice (once in the early period and once in the late period), however time and resources available restricted some turbine sites only having one survey. All bird surveys were undertaken between the hours of 0800 and 1800 to avoid peak activity and any associated bias in bird detectability and the direction of travel along transects was rotated for transects that were surveyed twice. Two surveyors undertook the surveys, each working individually with walking rate standardised at 1 km/h. All bird encounters within a 100m perpendicular distance of the line transect were recorded, and all individuals up to 50m in front of the observer. No bird records behind the perpendicular of the line transect and the surveyor were recorded. Birds observed only in flight were recorded if the bird was observed to cross the perpendicular of the transect at the point of the surveyors location. Bird encounters were digitally projected in space using a handheld Global Positioning System (GPS) unit and a laser range finder to provide more accurate bird locations and to facilitate entry into a GIS. Where multiple individuals were encountered in the same group, the location of the group was recorded at the closest bird to the surveyor and the number of individuals was noted. Turbine locations were verified using a projected GPS waypoint where access to the turbine was not available, or by an unprojected waypoint where access was possible. A 'burn in' distance was applied to the wind turbine line transects which involved walking into the survey area at the standardised walking pace from a minimum radial distance of 1,000m from the turbine location. This was to minimise any disturbance created by observers preparing to begin surveys. Environmental variables recorded at the start of the burn in location including wind speed, temperature, cloud cover, visibility and rainfall. Where possible, the rate of spin of the turbine blades was recorded on a four-point scale.

4.3.3. Habitat surveys

Habitats were surveyed in 50m^2 quadrats at each intersection where a length of line transect crossed the midpoint of a turbine distance band. Habitat variables were selected based on their potential significance in influencing bird distributions around turbines, with the intention to use these data as covariates in later statistical analyses. Both qualitative and quantitative habitat variables were recorded at every intersection between the midpoint of a turbine distance band based on the location of the turbine as described in the planning permission documentation and a bird survey line transect (Table 4.1). The habitat survey area was defined by three points along the line transect; a point at the intersection between the line transect and the midpoint of the distance band; a point 25m in one direction along the line transect from the intersection; a point 25m in the opposite direction along the line transect from the intersection. A perpendicular distance of 50m on either side of the transect at the location of these points dictated the two dimensional area of the habitat survey area (hereon referred to as 'habitat quadrat'). The areas on either side of the line transect were treated and surveyed as two distinct habitat quadrats. Primary and secondary habitat describe the two habitats that compose the largest and second largest proportion of a habitat quadrat by surface area coverage (Table 4.2).

4.3.4. Spatial data processing

Bird records were entered into a GIS attribute table in ArcGIS and associated with their respective GPS coordinates. All line transects were digitised with 100m buffers on either side of the transect to simulate the area surveyed. Bird records were cropped by these buffers to remove all bird records outside of the survey area and misplaced GPS points. Sampling effort was determined by intersecting the 100m line transect buffers with the turbine distance bands using Quantum GIS and calculating the subsequent area. Survey effort was then calculated as m^2 per turbine, m^2 per distance band and m^2 per distance band per turbine. Turbine locations were verified and corrected (where necessary) in ArcGIS, post survey using the GPS waypoints collected during surveys and aerial imagery (Table 4.3). Radial distance bands of 100m increments were calculated around the corrected turbine locations, and bird records associated with the distance band they geographically fell within. For the analyses the distance bands were increased to 600m to account for potential shifts in line transect location relative to the turbine positions. Bird records were then allocated habitat data based on the closest habitat quadrat. Where a bird record had equal proximity to two or more habitat quadrats, one habitat quadrat was randomly assigned. Habitat quadrats were split between turbine distance bands for analysis independent of bird records in two ways. Where area of habitat was of interest, the quadrats were intersected by the turbine distance bands in ArcGIS and the resultant total of each habitat per distance band calculated. For analysis of habitat quadrat counts, habitat area split by distance band biased the data and forfeited statistical integrity (because an artificially high number of habitat units was generated through intersection), thus the centroid of the habitat quadrat was calculated and allocated to a distance band.

Table 4.1 Quantitative and qualitative habitat measurements taken at each habitat sampling site

Measurement variable	Data type	Description
Primary habitat	Categorical (see table 4.2)	The habitat category that makes up the largest proportion of the sampling area. If two habitats are equal, both habitats will be recorded as a primary habitat in the format x/y where x is one habitat type and y is another.
Secondary habitat	Categorical (see table 4.2)	The habitat category that makes up the second largest proportion of the sampling area. If there is only one habitat present, secondary habitat will be recorded as 0. If two habitats are equal in abundance by area, secondary habitat will be recorded as 0.
Livestock	Binary (presence/absence), Categorical (livestock species)	An indicator that livestock were present in the habitat quadrat during habitat survey. If livestock were present in the same field as a field occurring within the quadrat, livestock was recorded as present. Species of livestock were recorded. Present =1, absent = 0.
Trees	Binary (presence/absence)	One or more trees occur within the habitat sampling quadrat. Present =1, absent = 0.
Hedgerows	Binary (presence/absence)	One or more hedgerows occur within the habitat sampling quadrat. Present =1, absent = 0.
Buildings	Binary (presence/absence)	One or more buildings occur within the habitat sampling quadrat. Present =1, absent = 0.
Visibility	Continuous	A measurement in metres of perpendicular ground visibility from the transect line. Measurements >100m were recorded as >100m.

Table 4.2 primary and secondary habitat categories. Habitats were categorised prior to habitat survey based on informal observation during earlier bird survey. These categories relate to primary habitat (the habitat with the highest proportion of land cover within the habitat quadrat) and secondary habitat (the habitat with the second highest proportion of land cover within the habitat quadrat).

Habitat type	Habitat description
Woodland	An area dominated by trees
Scrubland	An area dominated by low lying, dense vegetation
Farmland (unimproved)	An area of land clearly used for agricultural purposes with high vegetative species diversity and is not dominated by bright green/lush grasses.
Farmland (semi-improved)	An area of land clearly used for agricultural purposes with medium vegetative species diversity and/or is not dominated by bright green, lush grasses.
Farmland (improved)	An area of land clearly used for agricultural purposes with low vegetative species diversity and/or dominated by bright green, lush grasses.
Grassland (unimproved)	An area of land that is not clearly used for agricultural purposes with high vegetative species diversity and is dominated by grasses.
Grassland (semi-improved)	An area of land that is not clearly used for agricultural purposes with medium vegetative species diversity and is dominated by grasses.
Grassland (improved)	An area of land that is not clearly used for agricultural purposes with low vegetative species diversity and is dominated by grasses.
Garden	An area of land that represents an outdoor section of a dwelling.
Bare rock	An area of land with no soil substrate and visible bedrock or boulders.
Running water	A visibly mobile body of water (e.g. rivers and streams).
Standing water	A body of water that appears to be non-mobile (e.g. lakes, ponds, canals).
Bog/waterlogged land	An area of land that has a soil substrate but is visibly saturated with water.
Moorland	An area of land that is dominated by moorland plant species (e.g. Calluna vulgaris, Erica tetralix, Vaccinium myrtilis, Juncus spp.).
Other	Any area of land that does not fall into the above categories.
None	Only relevant to the secondary habitat categories. Indicates that only a primary habitat is present.

4.3.5. Habitat and bird community data analysis

Primary and secondary habitat composition between distance bands was analysed using a Pearson's chi squared test of association in SPSS Statistics (IBM Corp, 2012). The centroid of each habitat quadrat was used to group habitat by turbine distance band. Bird community composition for each turbine distance band was examined using Shannon-Weaver index (*H'*), Simpson's index (*D*), Shannon's measure of evenness (*J'*), and Simpson's measure of evenness ($E_{1/D}$) calculated using the BiodiversityR package in R (Kindt and Coe, 2005; R Core Team, 2013) (See Appendix 3 for equations). Rank abundance curves were calculated for each turbine distance band in order to assess and compare species richness and evenness. Rarefaction curves and extrapolated species richness values were calculated for each distance band using the computer program EstimateS (Colwell, 2013).

One-way ANOVA with Tukeys Honestly Significant Difference (HSD) tests were used to examine overall and pairwise differences in habitat sampling area between distance bands with 'area sampled per distance band per turbine' as the dependent variable and 'turbine distance band' as the independent variable. For species with > 20 detections, habitat associations were determined initially using Pearson's chi-squared test of association in SPSS (IBM Corp, 2012) with primary habitat used as the categorical independent habitat variable and bird species presence or absence as the dependent variable. Presence data were used as they are more reliable than counts of individual birds for agile species and species that flock in large numbers (Stevens et al., 2013).

In order to test for associations between distance band and bird species presence, binary logistic regression was undertaken using the *glm* function in the *stats* R Package (R Core Team, 2013) with a logit link function. Species presence/ absence was used as the dependent variable. Turbine distance band was included as a numeric predictor variable and primary habitats were included as a categorical predictor variable. As categorical variables are converted to dummy variables by *stats*, habitats were reclassed in order to reduces the number of factor levels and by extension, reduce the total number of predictor variables used in modelling. All semi-improved and unimproved grassland and farmland were relabelled 'semi improved and unimproved'; improved grassland and improved farmland were grouped to become 'improved'; running water and standing water were grouped into 'water bodies'; the single record of bare rock was grouped with moorland (this record was known to occur in a moorland area) to become 'moorland'. Woodland, scrubland and garden remained the same. 'Other' habitats were excluded from analysis, as were any bird records associated with this habitat. This resulted in seven distinct habitat categories. Only distance bands with over 33ha of survey area were used for the analysis, excluding the 500m-600m band. Species with <20 records were excluded from binary logistic regression. The Z-statistic returned for all predictor variables (including individual habitat categories) by *glm* was used to assess significance and direction of association.

4.4. Results

4.4.1. Composition of moorland fringe habitats around Small Wind Turbines

A total of 16 turbine sites were surveyed, comprising of 15 individual turbines (density = 0.76 turbines/km^2 per turbine site within 500m radial distance) and a single cluster of eight turbines (density = 0.39 turbines/km^2 within 500m combined radial distance of all turbines). Correction of turbine geographic location from the planning permission documented location to confirmed turbine location resulted in a mean geographic shift of 68m ± 19.2m (SE). In total, 544 habitat quadrats were sampled (Fig. 4.1) corresponding to an area of 139.5 ha ($\bar{x}$ quadrat area= 0.257 ha ±0.002 ha). Mean habitat sampling areas per turbine distance bands were significantly different from one another (F = 2.496, P = 0.037). Pairwise post-hoc analysis showed that these differences were between 200m-300m and 500-600m distance bands (P = 0.036) and 300-400m and 500m-600m distance bands (P = 0.047). All other pairwise comparisons of habitat sampling areas were not significantly different, suggesting that only the 500m-600m distance band was underrepresented.

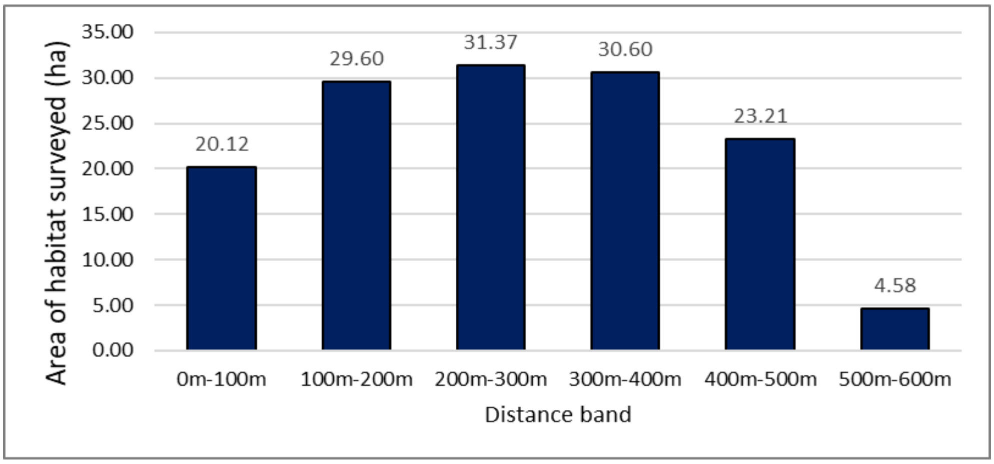


Figure 4.1 Total area of habitat sampled across all turbine distance bands for sixteen SWT sites within the SPMSPA fringe habitat. The 0-100m distance band was under-sampled as a result of a reduced total area due to a naturally reducing function of area by radial concentric bands around a point. The 500m-600m distance band was under-sampled due to a difference between expected turbine site locations versus actual turbine site locations and the associated shift in relative line transect position to the turbines.

Improved farmland accounted for over 40% of all primary habitat in all turbine distance bands except 500m-600m (Fig. 4.2). Semi-improved grassland made up a larger proportion of primary habitat in the 0m-100m distance band that in any other distance band (Fig. 4.2). Both primary and secondary habitat composition were heterogeneous within distance bands and appeared to be relatively evenly distributed between distance bands (except for 500m-600m). Few immediate differences in primary or secondary habitat composition by area between turbine distance bands were apparent, except for the furthest 500m-600m band. A large proportion of the habitat quadrats were composed only of one habitat type i.e. the primary habitat. Likelihood ratio chi squared test results for similarity in habitat between turbine distance bands and between turbine sites revealed that primary habitat was significantly different between distance bands when all distance bands were included in the analysis (Table 4.4). When the data for the 500m-600m distance band was removed from the analysis, there was no significant difference in primary habitat composition between distance bands, indicating that primary habitat was similar across all sites between all distance bands up to 500m. Secondary habitat was significantly different between turbine distance bands whether or not the furthest distance band data was included. All combinations of primary and secondary habitat were significantly different between distance bands when the 500m-600m distance band was excluded from analysis, but were similar across distance bands when 500m-600m was included. Both primary and secondary habitat were significantly different in composition between all turbine sites.

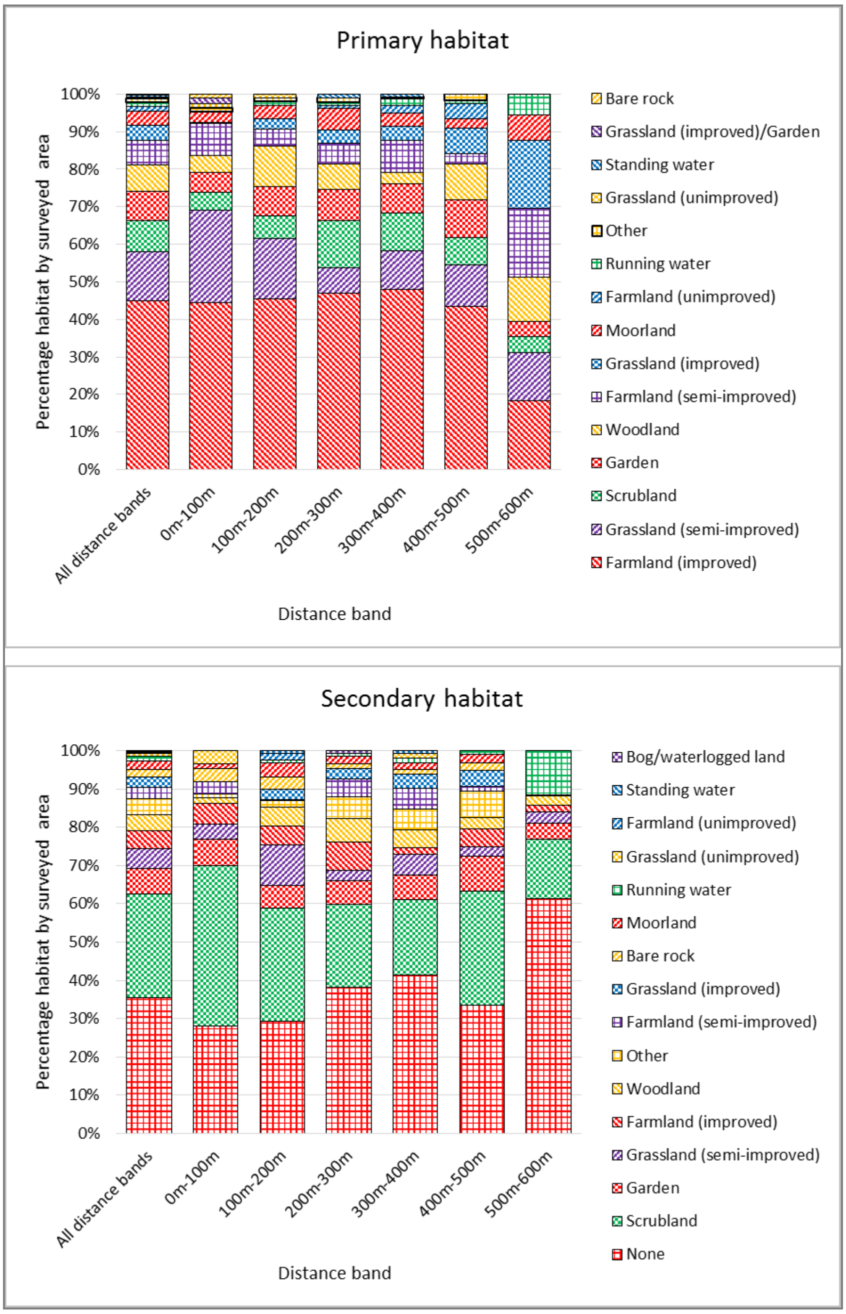


Figure 4.2 Composition of habitats by area across the different turbine distance bands, for 16 small wind turbine sites, in the SPMSPA fringe habitat. Primary habitat represents the most abundant habitat per quadrat, secondary habitat represents the second most abundant habitat.

4.4.2. Bird species abundance, richness and diversity around SWTs

A total of 54 bird species were recorded within 600m of turbines, comprising 1,360 detections and 2,687 individuals. Two measures of community composition, the Shannon-Wiener and Simpson's indices revealed that bird diversity was lowest within 100m of SWTs (Table 4.4) with diversity also being low between 200-300m of SWTs. Shannon-Weiner *H'* index of bird species diversity was highest within 300-400m of SWTs, whereas the Simpsons index revealed bird diversity highest at 400-500m (Table 4.4). Bird community evenness was greater at distances greater than 300m from the wind turbines (Table 4.4).

Figure 4.3 shows the relative abundance of species within each distance band. Jackdaw *Corvus monedula* was the most abundant species, dominating the 0-100m distance band with Starling *Sturnus vulgaris* being the most abundant at 100-200m from SWTs. The relative abundance of the most abundant species in closer distance bands to SWTs generally decreased with increasing distance from the SWTs. Unique species were encountered in all distance bands except 0-100m but most of these species were represented by <3 individuals e.g. Little Owl *Athene noctua*, Mallard *Anas platyrhyncos*, Nuthatch *Sitta europaea*, Treecreeper *Certhia familiaris* and Sparrowhawk *Accipiter nisus* were all unique to the 100m-200m turbine distance band, Wheatear was the only species unique to 200m-300m, Grasshopper Warbler *Locustella naevia*, Grey Wagtail *Motacilla cinerea* and Sand Martin *Riparia riperia* were all unique to 300-400m, Kingfisher *Alcedo atthis* was unique to 400m-500m and Grey Heron *Ardea cinerea* was unique to 500m-600m.

Table 4.3 Likelihood ratio chi squared test results for similarity in habitat between turbine distance bands and between the different turbine sites within the SPMSPA fringe habitat. Significant P values are highlighted in bold. Likelihood ratio statistic were chosen over Pearson's chi squared statistic due to the fact that many expected values were <1.

Variables used in analysis	**Filter parameters for analysis**	**Likelihood ratio**	***df***	***P***
Primary habitat, turbine distance band		97.0	70	0.040
Primary habitat, turbine distance band	No distance band 500m-600m	70.7	56	0.090
Secondary habitat, turbine distance band		104.6	75	0.014
Secondary habitat, turbine distance band	No secondary habitat 'none'	92.6	70	0.037
Secondary habitat, turbine distance band, no 500m-600m		85.0	60	0.018
Secondary habitat, turbine distance band,	No 500m-600m, distance band, no secondary habitat 'none'	77.6	56	0.030
Primary habitat, secondary habitat		367.0	210	<0.001
Combination of primary and secondary habitat, turbine distance band		443.5	420	0.206
Combination primary and secondary habitat, turbine distance band	No distance band 500m-600m	387.8	332	0.019
Primary habitat, turbine site		534.2	210	<0.001
Secondary habitat, turbine site		333.6	225	<0.001
Secondary habitat, turbine site	No secondary habitat 'none'	294.1	210	<0.001

Table 4.4 Measures of bird community composition across the different distance intervals from SWTs at 16 sites within the SPMSPA fringe. Measures represent the Shannon-Weiner index (H'), Simpson's index (as 1-D), Shannon's measure of evenness (J') and Simpson's measure of evenness ($E_{1/D}$). Ranking of each distance band according to the index value (where 1 = least diverse or even and 6 = most diverse or even) is presented in brackets.

	0m-100m	**100m-200m**	**200m-300m**	**300m-400m**	**400m-500m**	**500m-600m**	**Total**
Detections	147	332	298	338	196	49	1360
Abundance	269	662	620	708	344	84	2687
Species richness	26	44	38	40	33	22	54
H' (rank)	2.58 (1)	2.91 (4)	2.71 (2)	2.97 (6)	2.96 (5)	2.80 (3)	3.03
1-D (rank)	0.866 (1)	0.900 (3)	0.890 (2)	0.929 (5)	0.932 (6)	0.926 (4)	0.923
J' (rank)	0.793 (3)	0.770 (2)	0.744 (1)	0.805 (4)	0.848 (5)	0.906 (6)	0.760
$E_{1/D}$ (rank)	0.29 (3)	0.23 (1)	0.24 (2)	0.35 (4)	0.44 (5)	0.61 (6)	0.24

Rank abundance plots reveal that bird community composition across all distance bands consisted of a range of rare, uncommon, and abundant species (Fig. 4.3). Individual-based rarefaction (Figs. 4.4a-o) showed that estimated species richness (S_{est}) approached an asymptote towards observed species richness (S_{obs}) for all turbine distance bands except 500m-600m. Species richness at equivalent n was lower within the 0m-100m turbine distance band than 100-200m, 200-300m, 300-400m and 400-500m. Species richness at equivalent n was higher within the 100-200m turbine distance band than within 200-300m, 300m-400m and 400m-500m. Species richness at equivalent n was almost the same within the 300m-400m turbine distance band as within 400-500m and 500-600m. When extrapolated to the highest abundance of birds found in any turbine distance band (300-400m, n =708), S_{est} was highest in the 100m200m distance band, followed closely by 400m-500m, 300m-400m and 200m to 300m. 500m-600m and 0-100m both had a much lower S_{est} than other turbine distance bands.

Individual based species richness extrapolation was calculated to the size of the largest turbine distance band sample (n=708) using the methods provided within the framework of the computer program EstimateS (Colwell, 2013). Extrapolation to 708 individuals found that the 500m-600m turbine distance band had the lowest estimated species richness (S_{est} = 25.6), followed by 0-100m (S_{est} = 26.7), 200-300m (S_{est} = 39), 300m-400m (S_{est} = 40), 400-500m (S_{est} = 40.3) and the highest estimated species richness within the 100-200m turbine distance band (S_{est} = 44.6). Confidence intervals overlapped between turbine distance bands (Fig. 4.5).

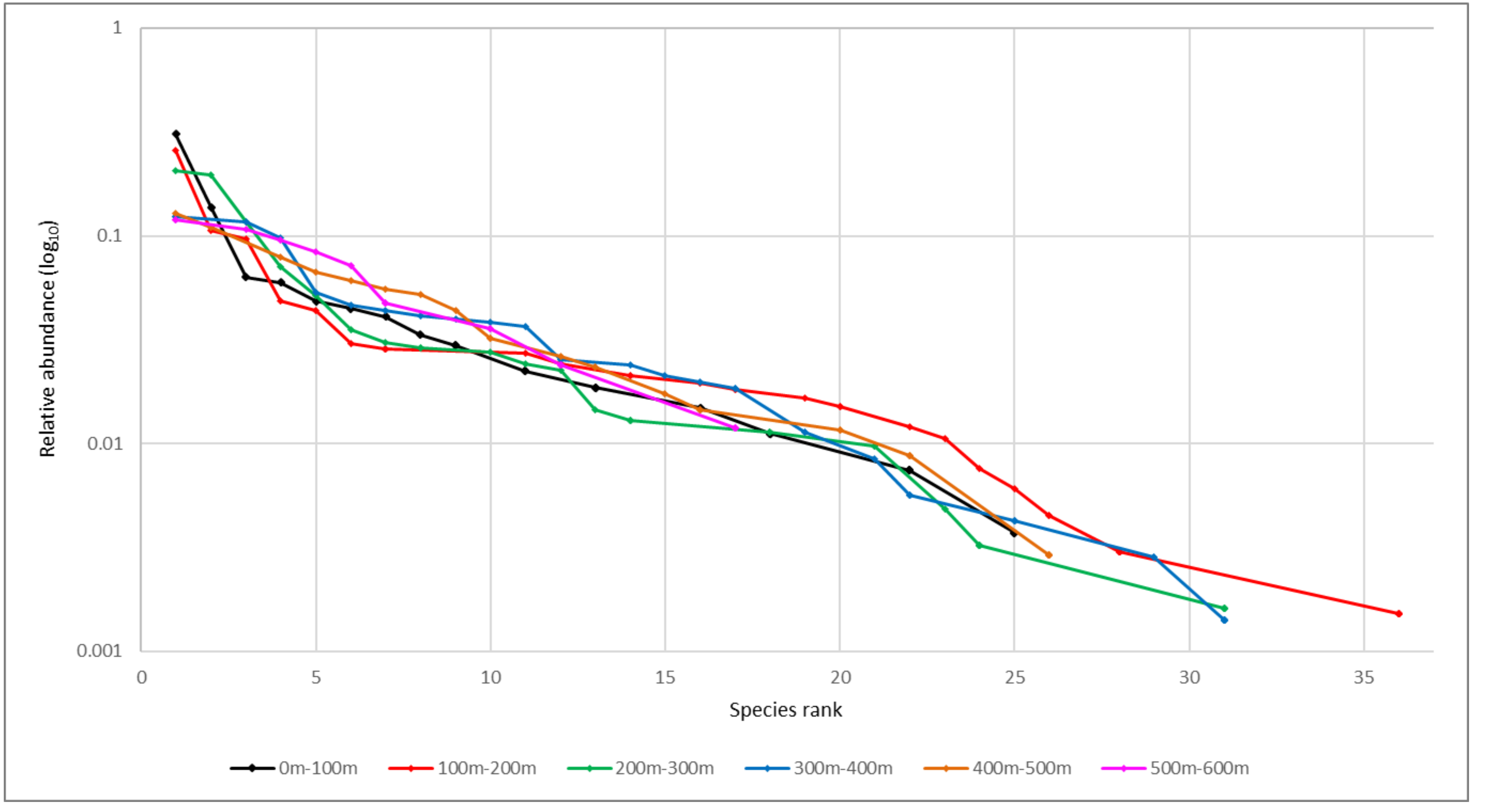


Figure 4.3 Rank abundance plots for each turbine distance band. Evenness is similar between all distance bands, however species richness is lower in the 0m-100m and 500m-600m distance bands than any other.

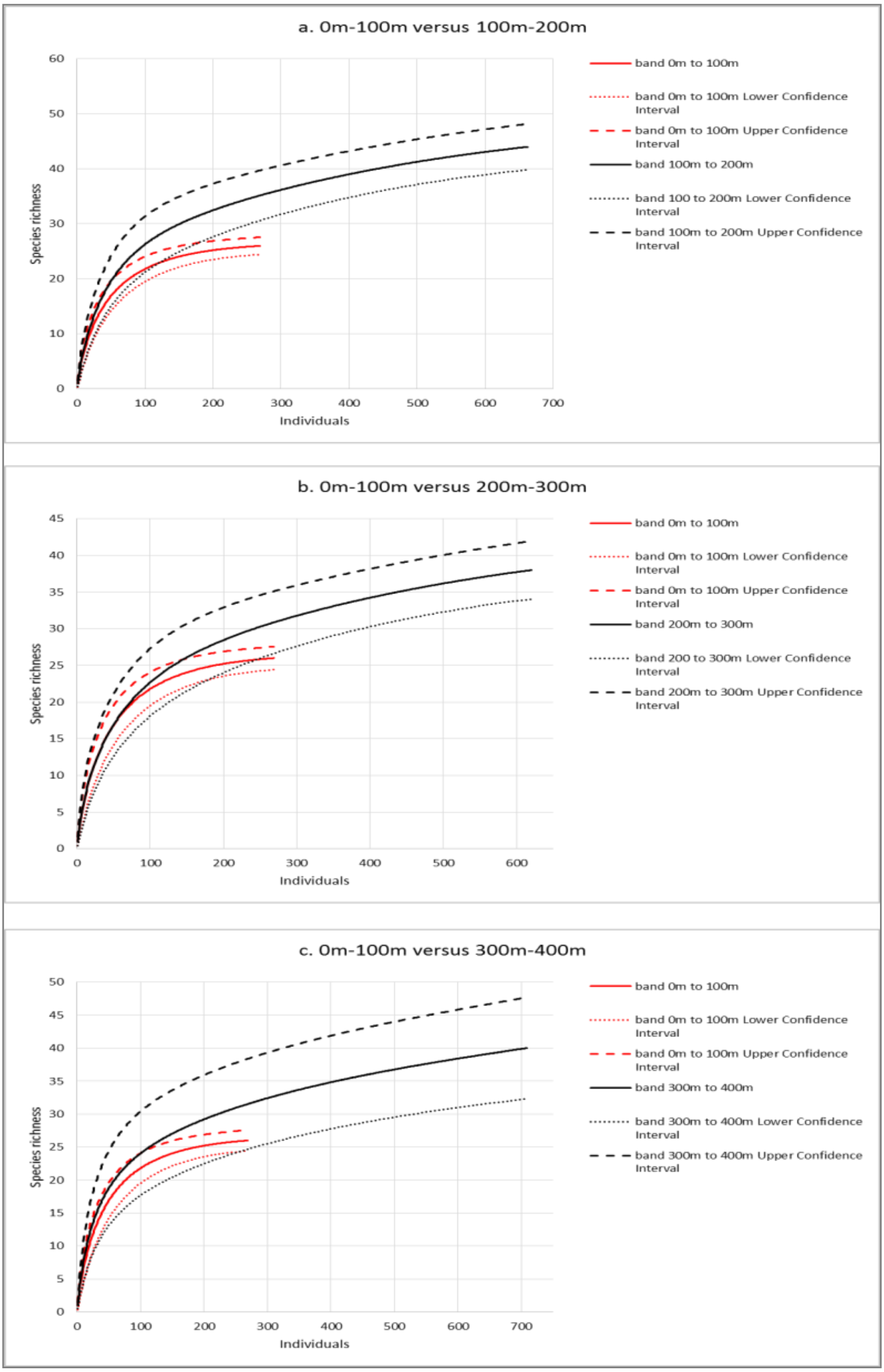
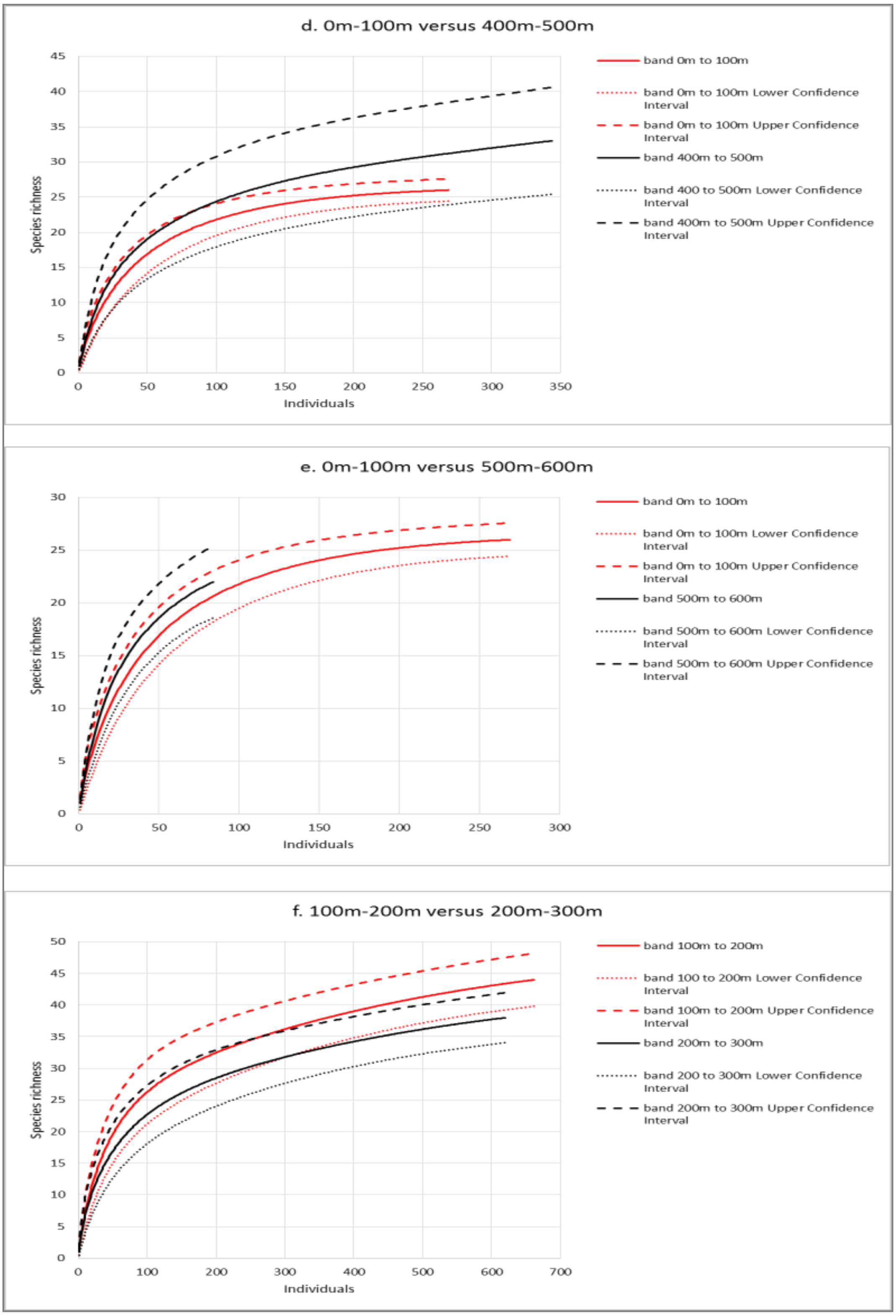
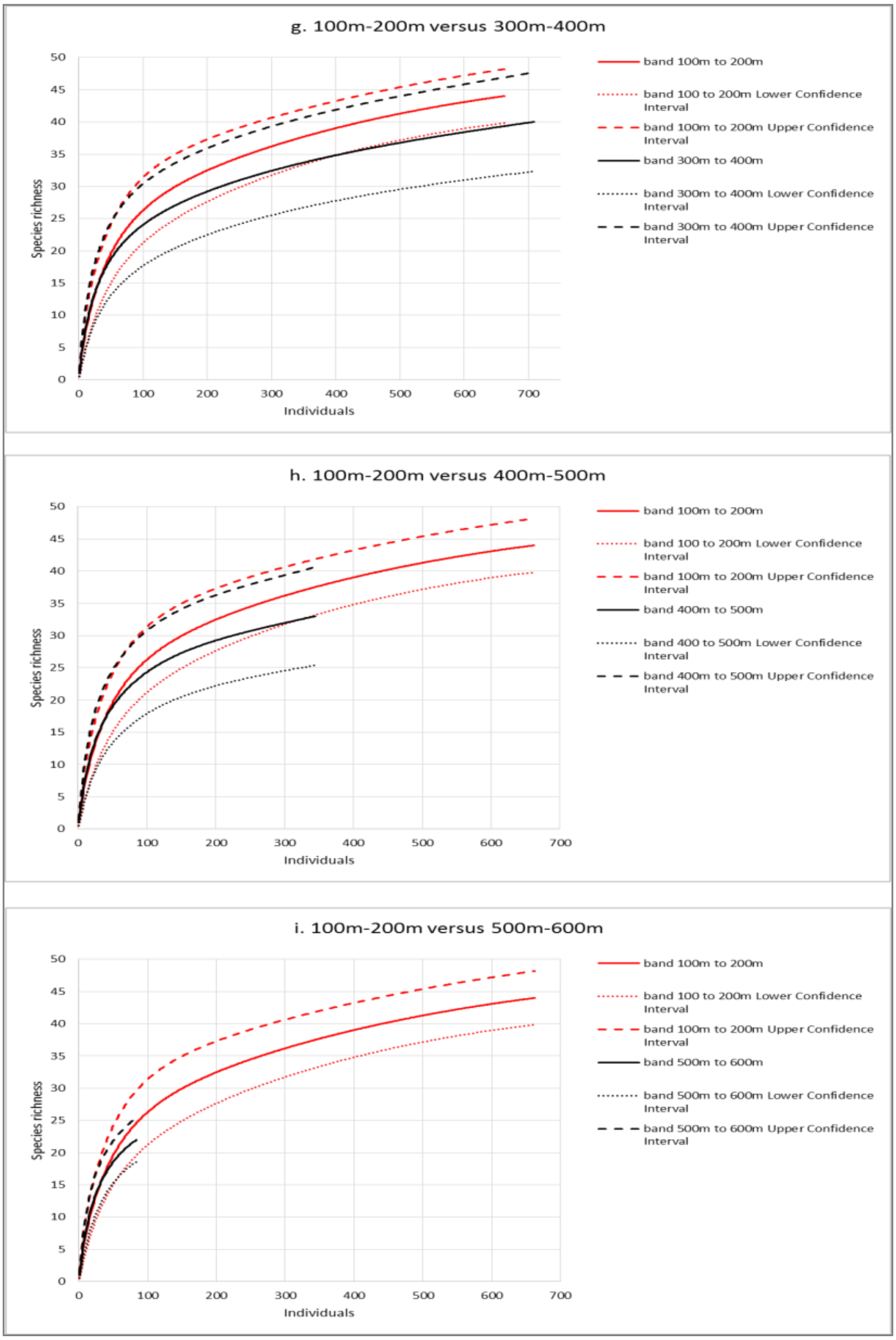


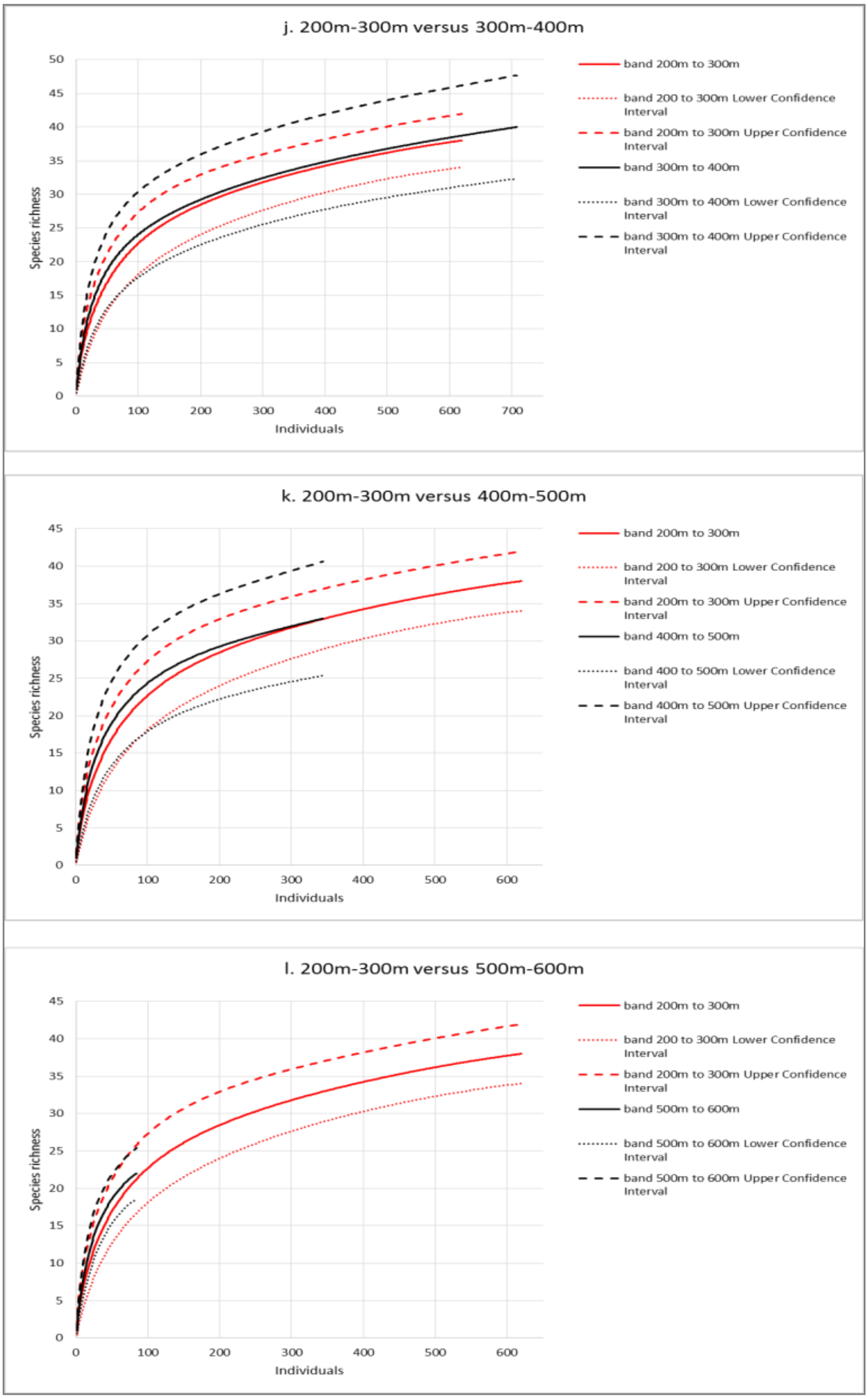
Figure 4.4a-c Pairwise rarefaction curves. Rarefied species richness is lower in the 0m-100m distance band than in all other distance band other than 500m-600m. No other distance bands have any difference in species richness to other distance bands.



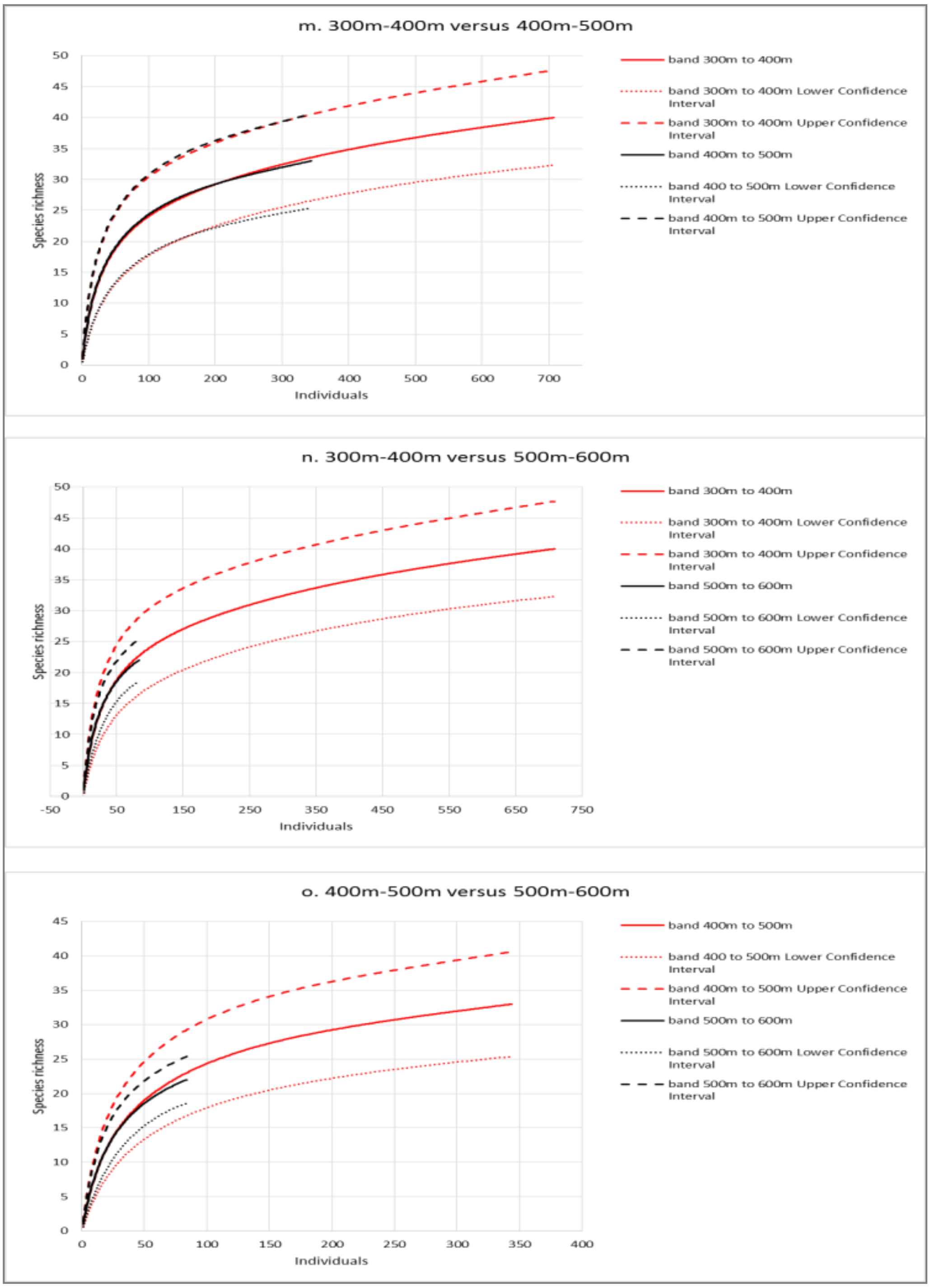
Figures 4.4d-f Pairwise rarefaction curves continued.



Figures 4.4g-i Pairwise rarefaction curves continued.



Figures 4.4j-l Pairwise rarefaction curves continued.



Figures 4.4m-o Pairwise rarefaction curves continued.

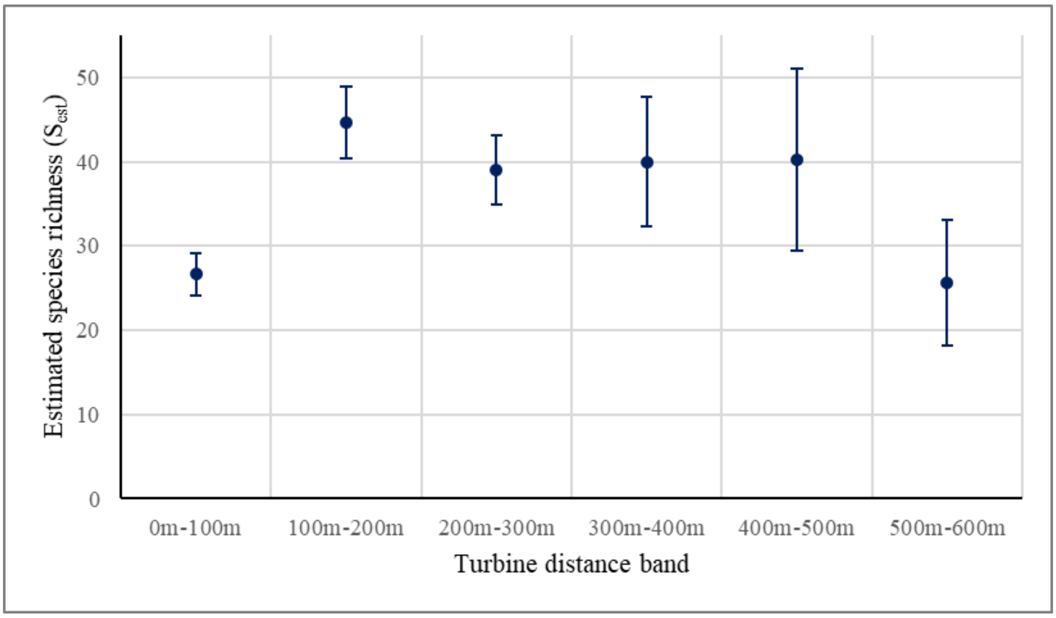


Figure 4.5 Estimated species richness by individual based extrapolation. The 0m-100m distance band had lower species richness than all other distance bands except 500m-600m. The 500m-600m distance band had a lower extrapolated species richness than 100m-200m and 200m-300m.

4.4.3. Bird-habitat associations and SWT effect

Survey effort differed significantly between at least two distance bands (F = 8.215, P <0.001). These differences were apparent between the 0-100m distance band and all other distance bands except 400-500m and 500-600m (100-200m, P = 0.043; 200-300m, P = 0.002; 300-400m, P = 0.005). Significant differences were also found between the 500-600m distance band and all other distance bands other that 0m-100m (100-200m, P = 0.02; 200-300m, P <0.001; 300-400m, P <0.001; 400-500m, P = 0.011). Consequently, survey area per turbine distance band per turbine site were used as a measure of survey effort and as a weighting variable in further analyses. Bird species with >20 total detections were analysed for species-habitat association using a Likelihood ratio Chi-squared test of association. Of the 21 species analysed, thirteen found to have significant habitat associations (P < 0.05). These included: Blackbird *Turdus merula*, Chaffinch *Fringilla coelebs*, Collared Dove *Streptopelia decaocto*, Great Tit *Parus major*, Greenfinch *Carduelis chloris*, House Sparrow *Passer domesticus*, Jackdaw *C. monedula*, Meadow Pipit *Anthus pratensis*, Robin *Erithacus rubecula*, Starling *S. vulgaris*, Swallow *Hirundo rustica*, Willow Warbler *Phylloscopus trochilus* and Wren *Troglodytes troglodytes*. Species that did not display significant habitat associations were Magpie *Pica pica*, Woodpigeon *Columba palumbus*, Blue Tit *Cyanistes caeruleus*, Carrion Crow *Corvus corone*, Pied Wagtail *Motacilla alba*, Goldfinch *Carduelis carduelis*, Linnet *Carduelis cannabina*, and Dunnock *Prunella modularis* (Table 4.6).

Table 4.5 Habitat associations of twenty-one bird species with >20 detections. A Likelihood ratio $\chi 2$ test of association was used. Significant habitat associations are highlighted in bold.

Primary habitat category	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**11**	**12**	**13**	**14**	**16**	**17**	**Total**	***Likelihood ratio χ^2***	***P***
Number of sampling 'sites'	56	77	5	47	348	4	88	32	81	1	14	1	20	6	780		
Corvids																	
Jackdaw	**6**	**6**	**0**	**3**	**44**	**0**	**15**	**5**	**15**	**0**	**7**	**0**	**7**	**2**	**110**	**29.2**	**0.006**
Magpie	5	3	0	1	27	0	2	1	6	0	1	0	0	0	46	13.1	0.443
Carrion Crow	3	7	0	1	36	0	6	4	5	0	1	0	0	1	64	13.1	0.439
Granivorous passerines																	
House Sparrow	**2**	**1**	**0**	**0**	**7**	**0**	**4**	**1**	**16**	**0**	**0**	**0**	**1**	**0**	**32**	**40.7**	**<0.001**
Greenfinch	**2**	**5**	**0**	**0**	**12**	**0**	**5**	**7**	**11**	**0**	**0**	**0**	**0**	**0**	**42**	**32.1**	**0.002**
Chaffinch	**11**	**11**	**0**	**3**	**18**	**0**	**7**	**4**	**11**	**0**	**2**	**0**	**0**	**0**	**67**	**25.5**	**0.020**
Linnet	0	2	0	1	9	0	0	0	0	0	0	0	0	0	12	12.2	0.510
Goldfinch	4	7	0	3	24	1	10	2	9	0	2	0	1	1	64	6.8	0.911
Insectivorous passerines																	
Robin	**11**	**7**	**0**	**1**	**9**	**0**	**8**	**2**	**3**	**1**	**1**	**0**	**2**	**0**	**45**	**34.0**	**0.001**
Willow Warbler	**8**	**6**	**0**	**0**	**3**	**1**	**7**	**0**	**1**	**0**	**1**	**0**	**0**	**0**	**27**	**40.8**	**<0.001**
Wren	**7**	**4**	**0**	**0**	**3**	**1**	**4**	**1**	**1**	**0**	**1**	**0**	**0**	**2**	**24**	**34.6**	**0.001**
Great Tit	**7**	**6**	**0**	**0**	**6**	**0**	**2**	**2**	**4**	**0**	**0**	**0**	**0**	**0**	**27**	**23.8**	**0.033**
Blue Tit	9	8	0	5	13	1	7	2	6	0	1	0	1	0	53	17.5	0.177
Dunnock	3	6	0	2	14	0	2	2	8	0	1	0	0	0	38	10.8	0.627
Pigeons																	
Collared Dove	**3**	**1**	**0**	**0**	**2**	**0**	**1**	**1**	**6**	**1**	**0**	**0**	**0**	**0**	**15**	**26.9**	**0.013**
Woodpigeon	3	5	0	6	11	0	2	1	1	0	0	0	0	0	29	15.1	0.300
Starlings																	
Starling	**2**	**1**	**0**	**1**	**36**	**0**	**9**	**3**	**14**	**0**	**1**	**0**	**1**	**0**	**68**	**24.9**	**0.024**
Swallows and swifts																	
Swallow	**1**	**11**	**2**	**5**	**77**	**1**	**20**	**7**	**7**	**0**	**2**	**0**	**3**	**0**	**136**	**33.8**	**0.001**
Thrushes																	
Blackbird	**15**	**15**	**0**	**3**	**28**	**0**	**8**	**2**	**16**	**0**	**2**	**0**	**0**	**1**	**90**	**33.8**	**0.001**
Wagtails and pipits																	
Meadow Pipit	**1**	**6**	**3**	**12**	**49**	**0**	**11**	**3**	**0**	**0**	**0**	**1**	**5**	**0**	**91**	**58.1**	**<0.001**
Pied Wagtail	0	1	0	2	26	0	2	2	2	0	0	0	0	0	35	20.8	0.078

Only two species were found to have a significant association with turbine distance band. These were Magpie and Starling (Table 4.7). To corroborate these results, similar binary logistic regression models were built for these two species, using only turbine distance as a predictor variable. Results were compared to a null model using a z-statistic, further reinforcing this significant relationship: Magpie (deviance from null model = 6.6, $p<0.01$); Starling (deviance from null model = 6.9, $p< 0.01$). The presence of both Magpie and Starling showed a positive relationship with distance from turbine, with median distance band for absence records at 200m-300m and 300-400m for presence records in both species (Figure 4.6).

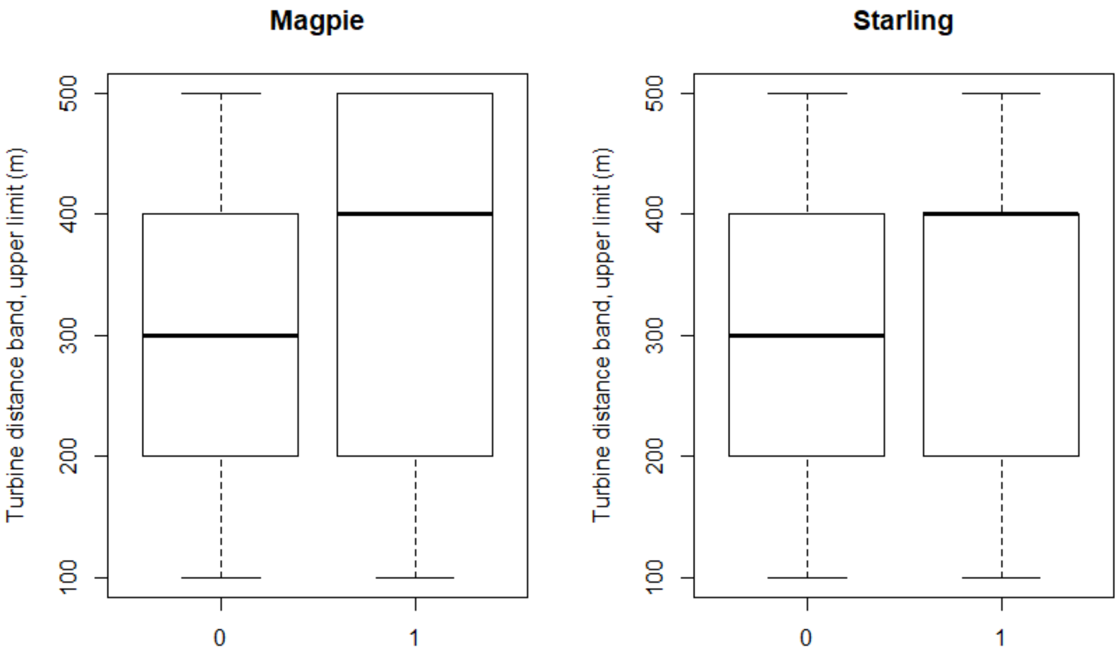


Figure 4.6. Differences in presence records (1) and absence records (0) for Magpie and Starling by distance from turbines in 100m bands. These species were the only two species found to have a significant association with turbine distance bands (table 4.7).

Table 4.6 The results of binary logistic regression examining species-habitat association and species-turbine distance band associations. Z-test scores are shown for all species all predictor variables. Significant associations are highlighted in bold.

Species	*Number of detections*	*Turbine Distance band*		*Improved*		*Semi improved and unimproved*		*Moorland*		*Scrubland*		*Woodland*		*Waterbody*	
		Z	*P*	Z	*P*	Z	*P*	Z	*P*	Z	*P*	Z	*P*	Z	*P*
Blackbird	85	-1.01	0.31	**-3.4**	**<0.01**	**-3.01**	**<0.01**	-0.03	0.98	-0.26	0.80	0.62	0.54	-0.63	0.53
Blue tit	51	1.56	0.12	-1.04	0.30	0.73	0.47	-0.31	0.76	0.80	0.43	1.69	0.09	-0.09	0.93
Carrion Crow	63	0.94	0.35	1.06	0.29	0.05	0.96	-0.02	0.99	0.54	0.96	-0.14	0.89	-0.05	0.96
Chaffinch	65	-0.78	0.43	**-2.43**	**0.02**	-1.37	0.17	-0.02	0.99	0.12	0.85	1.07	0.28	0.06	0.95
Dunnock	36	0.07	0.94	-1.80	0.07	-1.74	0.08	-0.02	0.99	-0.34	0.73	-1.14	0.25	-0.33	0.74
Goldfinch	63	-0.54	0.59	-1.41	0.16	-0.58	0.56	-0.93	0.35	-0.57	0.57	-0.82	0.41	0.18	0.86
Great Tit	23	-0.17	0.86	-1.17	0.24	-0.99	0.32	-0.01	0.99	0.98	0.33	1.217	0.22	-0.01	0.99
Greenfinch	41	-1.00	0.32	**-3.54**	**<0.01**	-1.86	0.06	-0.02	0.99	-1.58	0.12	**-2.10**	**0.04**	-0.02	0.99
House Sparrow	30	1.27	0.20	**-5.13**	**<0.01**	**-3.76**	**<0.01**	-1.51	0.13	**-2.83**	**<0.01**	**-2.34**	**0.02**	-0.02	0.99
Jackdaw	106	-1.55	0.12	-1.60	0.11	-1.43	0.15	1.26	0.21	-2.09	0.04	-1.66	0.10	**2.24**	**0.03**
Magpie	41	**2.53**	**0.01**	0.10	0.92	-1.41	0.16	-0.02	0.99	-0.77	0.44	0.85	0.40	-0.15	0.88
Meadow Pipit	90	1.19	0.24	0.02	0.98	0.02	0.98	0.02	0.98	0.02	0.98	0.02	0.98	0.02	0.98
Pied Wagtail	34	-0.71	0.48	1.48	0.14	0.12	0.91	-0.01	>0.99	-0.56	0.56	-0.01	0.99	-0.01	>0.99
Robin	43	-0.11	0.91	-0.97	0.33	0.8	0.42	1.62	0.11	1.23	0.22	**2.72**	**<0.01**	0.46	0.65
Starling	67	**2.51**	**0.01**	-1.84	0.07	-2.38	0.17	-1.43	0.15	**-2.75**	**<0.01**	**-2.09**	**0.04**	-1.21	0.23
Swallow	132	0.52	0.60	**2.43**	**0.02**	**1.98**	**<0.05**	0.65	0.52	0.95	0.34	-1.55	0.12	0.50	0.62
Willow Warbler	25	-1.56	0.12	-0.39	0.70	0.85	0.39	-0.02	0.99	1.72	0.09	**2.30**	**0.02**	1.27	0.20
Woodpigeon	25	-1.73	0.08	0.01	0.99	0.01	0.99	<0.01	>0.99	0.01	0.99	0.01	0.99	<0.01	>0.99
Wren	21	-0.92	0.36	-0.37	0.71	0.89	0.37	-0.02	0.99	1.26	0.21	**2.00**	**<0.05**	1.22	0.22

4.5. Discussion

4.5.1. Composition of fringe habitat around Small Wind Turbines

Fringe habitat composition within 600m of turbine sites was heterogeneous with sites differing in the proportion of available habitats. However, this heterogeneity was found to be non-uniformly distributed between turbine distance bands. Habitat composition also differed significantly between turbine sites. This confirms that SWTs are non-randomly positioned and that the selection criteria used to determine their location are not habitat specific. Site descriptions of onshore wind turbine studies in the UK include unenclosed upland habitats such as moorland, rough grassland and blanket bog (Pearce-Higgins et al., 2009), with others more generally categorising sites as coastal and upland (Bassi et al., 2012) or coastal and inland (Sinden, 2007). Minderman et al. (2012) collected environmental and linear feature data in their study on the effects of SWTs on birds e.g. proximity of turbines to hedgerows and treelines. However these authors did not further assess habitat composition in the vicinity of SWTs. Large scale wind turbine sites are not randomly distributed in a landscape, rather they are selected based on minimising negative impacts such as visual impact, noise impact, and ecological impacts (Saidur et al., 2011) and maximising potential energy yield through wind resource (Sturge et al., 2014). Similar considerations are given to Small Wind Turbines (Bahaj et al., 2007; Allen et al., 2008), however the potential ecological effects of SWTs are not as well studied as those of large wind turbines (Minderman et al., 2012).To the best of my knowledge, this research represents the first attempt to describe and quantify the habitat characteristics at a small number of selected SWT sites in the UK.

The most commonly encountered primary habitat in all distance bands was improved farmland, indicating that in this area, farmers are maximising the use of their land by allowing the constructing and use of turbines on worked agricultural farmland. It is not uncommon for large turbines to be built on farmland in the UK, however there is limited evidence of the effect of these turbines on birds, with one study suggesting that negative impacts may be limited (Devereux et al., 2008). Surprisingly, moorland habitat was not common within 600m of the selected SWTs, making up only 3.8% of primary habitat and 2.1% of recorded secondary habitat at the study sites. Of the 31 habitat quadrats that contained moorland as either a primary or a secondary habitat, 24 were within 1km of the SPMSPA boundary including eight quadrats that were within the SPMSPA. A further two quadrats were found within 1km-2km of the SPMSPA, and five moorland habitat quadrats were found within 2km-3km. Considering that the study area was immediately adjacent to the SPMSPA (an extensive area of moorland habitat), this may seem surprising. However, as the SPMSPA is also an SAC with the specific purpose of protecting the moorland habitat, this result may simply reflect the fact that SWTs are not permitted to be built within the boundary of the SPMSPA, and that the majority of moorland habitat is enclosed within the SPMSPA (see Chapter Two). Another explanation is that SWTs are selected at sites that are not in close proximity to moorland areas outside the SPMSPA. For this to be confirmed, a more comprehensive survey of the habitats in the SPMSPA fringe would have to be undertaken in relation to the location of SWTs.

Although a total of 54 bird species were recorded around the SWT sites, only three of these were target conservation priority species: Curlew, Lapwing and Wheatear, all of which were recorded in very low numbers. Considering the proximity of the survey sites to the SPMSPA and its associated moorland habitat, the low abundance of target species was unexpected, especially considering that moorland birds are anecdotally expected to utilise the surrounding farmland. Curlew *Numenius arquata* and Snipe *Gallinago gallinago* have both been shown to prefer habitats with a heterogeneous structure and composition (Pearce-Higgins and Grant, 2006). This may explain these species low abundance as the majority of habitat around SWTs consisted of improved farmland, where sward structure is likely to be uniform and vegetation diversity low. Pearce-Higgins and Yalden (2004) show that Golden Plover chicks show very low preference for grassland within their home range when there are moorland habitats nearby. If this is also apparent for the adult Golden Plovers, then the proximity of the SPMSPA and its associated habitats may deter individuals from the surrounding moorland fringe landscape. Elsewhere in UK uplands, Golden Plovers are known to utilise enclosed fringe habitat abundant in tipulids for feeding (Pearce-Higgins and Yalden, 2003). No data were collected on invertebrate abundance for this study, but it is possible that food scarcity in the study area and a lack of suitable tipulid populations may have been a factor, and this merits further research in the future.

Despite the low numbers of target species, 26 of the species recorded are of UK conservation concern, with seven listed as red and 19 listed as amber by the RSPB (Eaton et al., 2009). Of these, two red listed species (House Sparrow and Starling) and four amber listed species (Willow Warbler, Swallow, Meadow Pipit and Dunnock) were recorded in sufficient numbers for analysis. Of these, Starling had a significant negative association with proximal turbine distance bands. Magpie was the only other species to exhibit any significant relationship with turbine distance band. In a study of non-breeding bird communities, Devereux et al., (2008) also found no evidence of displacement in three functional groups of wintering farmland birds or in wintering Skylark in the United Kingdom. Other studies have found displacement and avoidance effects on birds within set distances of wind turbines, but the displacement effect and extent of displacement varies among sites, operational status of turbines and is also species-specific (Barrios and Rodríguez, 2004; Smallwood et al., 2009; Stevens et al., 2013). Larsen and Madsen (2000) report a displacement effect on the foraging behaviour and habitat utilization by Pink-footed Geese within 100m or 200m of wind turbines, depending on whether the configuration of turbines was linear or clustered respectively. Pearce-Higgins et al (2009) found displacement evidence in species densities extending 100–800 m from turbines for seven moorland species studied within the United Kingdom. However, this study did not have identical control sites for the surveyed turbine sites - a problem that is ubiquitous across heterogeneous landscapes which are subject to turbine development. Amongst wintering grassland-dependent birds in the USA, studies have shown that displacement by wind turbines on habitat occupancy tends to be species specific (e.g. Stevens et al., 2013) with turbines influencing the densities of grassland birds within 180m of wind turbines (Leddy et al., 1999) and that displacement may depend in part on the extent of habitat modification during wind turbine construction (Pearce-Higgins et al., 2012). Within this study, extrapolated bird species richness was found to be significantly lower within 100m of SWTs than all other distance bands out to 500m and greatest 100-200m from turbines. In addition, bird species diversity according to the Shannon-Wiener index and Simpson's index was found to be lowest within 100m of SWTs. These findings suggests that there may be a displacement effect on the bird community within 100m of SWTs. It is clear that SWT development proposals must consider the habitats proposed for citing and construction. Planning decisions for the citing of SWTs should consider the potential for small-scale displacement effects on fringe bird species within 100m of the SWT, but landscape level factors are critical in the decision making for granting planning applications for clusters of SWTs. One of the major concerns for unitary authorities is the pressure for further housing development within the moorland fringe landscape, which has the potential to reduce the amount of suitable land for bird populations and SWTs. Thus it is important to identify where are the most important sites for birds within the moorland fringe, particularly for the conservation-priority bird species. This will be the focus for the next chapter of this PhD.

4.6. References

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CHAPTER 5: HABITAT SUITABILITY MODELLING OF BIRDS IN THE MOORLAND FRINGE

5.1 Abstract

Encroaching urban development and agricultural land intensification have the potential to negatively affect the efficacy of protected areas in their conservation objectives by modifying supplementary habitat and disrupting corridors to movement. However, planning departments lack sufficient evidence to make ecologically sound planning decisions with regards housing developments. In the case of upland SPAs such as the South Pennine Moors Special Protection Area (SPMSPA), breeding bird species of conservation concern such as Golden Plover *Pluvialis apricaria*, Lapwing *Vanellus vanellus* and Curlew *Numenius arquata* are likely to use farmland outside of the SPA as well as moorland habitat within the SPA for feeding and breeding. This chapter aims to; (1) Develop Habitat Suitability Models for Lapwing *Vanellus vanellus*, Curlew *Numenius arquata*, Golden Plover *Pluvialis apricaria*, Snipe *Gallinago gallinago*, Reed Bunting *Emberiza schoeniclus*, and Wheatear *Oenanthe oenanthe*; (2) Assess the predictive capability of Landsat 8 data for these models (3) To assess the relative importance of empirical predictors in predicting habitat suitability (4) To assess a suite of algorithms for Habitat Suitability Modelling to determine the most appropriate algorithms for these species. Landsat 8 spectral bands performed well as predictor variables in habitat suitability modelling, especially when used to supplement empirical data. Building density was an important predictor variable for all species except Golden Plover. Indicators of agricultural activity did not contribute much to models. The best performing modelling algorithms were consistently Random Forest and Generalised Boosting Models.

5.2 Introduction

The expansion of urban development and the creation of new housing is necessary to accommodate a globally increasing human populous, but has the potential to negatively impact biodiversity (Seto et al., 2012; Güneralp and Seto, 2013). The negative ecological impacts of housing development are taxonomically broad and complex in nature (McKinney, 2002), but includes detrimental consequences to bird species populations (Sushinsky et al., 2013) and bird diversity (Pidgeon et al., 2014). Encroaching urban development and agricultural land intensification have the potential to disrupt the conservation value of protected areas through the modification of supplementary habitat required by species within the protected area, and by limiting connectivity to these habitats (Radeloff et al., 2010). In landscapes where urban-rural or urban-natural gradients exist and protected areas are nearby, the complexity of the landscape means that conservation oriented planning decisions are complicated by a multitude of interacting ecosystem processes (Radeloff et al., 2005; McDonnell et al., 2008; Lookingbill et al., 2014; Baró et al., 2017). Intuitively, one might expect larger residential developments to have the greatest impact on protected areas.However it has been proposed that lower density residential developments may have more of an impact on species within protected areas, due to the fact that they are more likely to occur within close vicinity of a protected area than large urban conurbations (Hansen et al., 2005).

Rural areas are generally higher in biodiversity than urban areas, even when both are protected for their ecological value (Knapp et al., 2008). In the case of Europe and the United Kingdom (UK), farmland birds attract conservation consideration as many species have shown dramatic declines in recent years due to agricultural intensification (Donald et al., 2006; Sanderson et al., 2013; Aebischer et al., 2016). In addition, in the UK uplands, moorland specialist breeding bird species such as Golden Plover *Pluvialis apricaria*, Short-eared Owl *Asio flammeus* and Dunlin *Calidris alpina* are often protected by Special Protection Areas (SPAs) (Hancock et al., 2009; Pendlebury et al., 2011; Hayhow et al., 2015). Development is heavily restricted within the boundaries of these SPAs, however some species such as Golden Plover use farmland that may fall outside of SPAs as supplementary feeding habitat (Whittingham et al., 2000). As such, any planning developments in these upland SPA/farmland/urban mosaics should consider the potential for a development site to harbour not only typical farmland bird species, but also moorland bird species. It is therefore important to understand the effects that urban development, farming practises and environmental characteristics have on upland birds within the moorland fringe. This chapter aims to investigate these factors on thirteen bird species that have been identified as priority species of conservation concern (see chapter three).

Investigating the habitat associations of birds often takes the approach of measuring environmental variables in the field, conducting bird surveys and statistically relating one to the other. This is robust and allows for detailed habitat associations to be determined, however without complete geographical coverage of the environmental variables, this approach is of limited use for the evidence based site selection of planning developments where least ecological impact is desired. As ecological surveys rarely have complete coverage due to resource limitations, other approaches to determining suitable areas for development (or conversely, suitable habitat for a species) are required. One family of methods used to determine relative habitat suitability over large areas is Species Distribution Modelling (SDM). Different terminology is often used for different applications of SDM, which include Habitat Suitability Modelling and Ecological Niche Modelling. The differences between these is subtle and relate mainly to the desired practical application. For the purpose of this chapter, the term Habitat Suitability Modelling will be used, as it reflects the application of determining the relative value of habitat throughout the moorland fringe study site.

Landcover data are often used to investigate bird habitat associations where complete coverage of a region is needed, for example Corine in Europe (e.g. Radović and Tepić, 2009), National Land Cover Data in the United States (e.g. Wood et al., 2014) and Land Cover Map in the UK (e.g. Fuller et al., 2004). Such datasets are usually reliant on the supervised classification of remotely sensed data from satellite imagery (Yu et al., 2014). A degree of uncertainty is inherent in the process of modelling remotely sensed data into discrete classes (Congalton et al., 2014), and similarly a degree of uncertainty is inherent in the process of producing Habitat Suitability Models (Lin et al., 2015). As an alternative to using classified spectral data, a potential method for reducing cumulative error is to use continuous variables in the form of unclassified remotely sensed spectral data, or derived indices such as Normalised Difference Vegetation Index (NDVI) to create Habitat Suitability Models (Bradley and Fleishman, 2008; Shirley et al., 2013). The Landsat satellite program has been in service since 1972, with the current Landsat 8 satellite producing freely available images in ten spectral bands at 30m x 30m resolution. Using these raw data to model habitat suitability provides an opportunity for low cost, standardised, fine scale Habitat Suitability Models to be built, with the potential opportunity for historic trends in suitability to be identified using time-series analysis (Shirley et al., 2013; Dutrieux et al., 2016).

The aims of this chapter are; (1) Assess the efficacy and accuracy of raw Landsat 8 spectral data in modelling habitat suitability for five moorland bird species of conservation concern, within a moorland fringe landscape and determine the most appropriate spectral bands in modelling each of these species; (2) Create Habitat Suitability Models for these five species using fine scale (30m x 30m) predictor variables encompassing measures of the built environment, farming practises and topographical factors and compare these models to models built using Landsat 8 data at the same resolution; (3) Assess whether combings Landsat 8 data improves Habitat suitability models (4) Assess which spectral bands and which environmental variables contribute to the best models for each species; (5) Use a suite of algorithms for Habitat Suitability Modelling to determine the most appropriate algorithms for these species.

5.3 Methods

5.3.1. Bird survey method and species selection

Bird surveys were conducted within the British breeding bird season during April-July of 2012, 2013 and 2015 by qualified ecologists from West Yorkshire Ecology (WYE), commissioned by council ecologists from Calderdale, Kirklees and Bradford. Full details on the bird survey method and survey sites are provided in Chapter Three. Thirteen conservation priority species identified by the three local authorities were initially selected for analysis: Lapwing *Vanellus vanellus,* Curlew *Numenius arquata*, Golden Plover *Pluvialis apricaria*, Common Sandpiper *Actitis hypoleucos,* Short-eared Owl *Asio flammeus*, Snipe *Gallinago gallinago*, Reed Bunting *Emberiza schoeniclus*, Twite *Linaria flavirostris* Ring Ouzel *Turdus torquatus*, Merlin *Falco tinnaeus,* Dunlin *Calidris alpine,* Whinchat *Saxicola rubetra* and Wheatear *Oenanthe oenanthe*. These species were considered conservation priority species because of their dependence in part, on moorland habitats, and their populations are in regional and national decline (see Chapter Three). Presence records for these species obtained from the bird surveys were cropped to the SPMSPA 1km fringe. This geographical constraint resulted in extremely low sample sizes for seven of the 13 species, resulting in model failure during preliminary investigation for Short-eared Owl (n = 17), Common Sandpiper (n= 15), Reed Bunting (n = 13), Twite (n = 11), Whinchat (n = 5), Ring Ouzel (n = 3), Merlin (n = 2) and Dunlin (n = 1). As modelling could not be completed for these species, they were omitted from analysis.

5.3.2. Selection of landscape predictor variables

A series of candidate predictor variables were selected for modelling habitat suitability of the five conservation priority bird species. All candidate predictor variables belonged to one of two categories; (1) Landsat 8 predictor variables and (2) Empirical predictor variables. These two sets of predictor variables will be described below.

5.3.3. Landsat 8 predictor variables

These data comprised four composite raster images, each with nine bands representing surface reflectance electromagnetic spectral wavelengths corresponding to Blue, Green, Red, Near Infrared (NIR), Shortwave Infrared (SWIR) 1 and 2, Panchromatic and Thermal Infrared (TIRS) 1 and 2 (see table 2.1). Cirrus and coastal/aerosol spectral bands were omitted from each composite image. The rasters were constructed from Landsat 8 scenes taken in 2013, 2014 and 2015 and represented the British seasonal periods (spring, summer, winter and autumn) in the location of the SPMSPA 1km fringe. The composite images were created in Google Earth Engine (GEE) using cloud free, orthorectified and topographically corrected Landsat 8 pixels. These data were identical to the Landsat 8 data used to perform classification of the SPMSPA fringe landscape in chapter 2 (see chapter 2 for further details). The four seasonal composite images were stacked into a single multiband raster (36 bands in total), with each band representing a potential predictor variable for habitat suitability modelling. Pearson product moment correlation coefficient and VIF (Variance Inflation Factor) were used to reduce collinearity amongst predictor variables in models using the *usdm* package in R (R Core Team, 2013; Naimi, 2015). The *vifcor* function was used, which calculates the maximum pairwise Pearson product moment correlation coefficient between all variables and removes from the pair the variable with the greatest VIF (Naimi, 2015). This process was reiterated until all pairwise correlation coefficients are below a predefined threshold, in this case $r<0.7$. The resultant set of variables were used to investigate the efficacy of Landsat 8 data in habitat suitability modelling of the five moorland fringe bird species.

5.3.4. Empirical predictor variables

A total of eight variables were considered for use as empirical model predictors. These variables were either direct representations of third party data or calculated from third party data and were selected based on their potential importance in predicting the habitat suitability of the five species to be modelled. Collinearity between numerical variables was determined using the *vifcor* method described in section 5.3.3. The eight predictor variables are described below.

Elevation and slope were chosen as predictor variables due to known associations of Lapwing (Smart et al., 2013), Curlew (Douglas et al., 2014), Golden Plover (Whittingham et al., 2000), Snipe (Amar et al., 2011) and Wheatear (Henderson et al., 2004) with the UK upland landscape. Elevation and slope were derived from Ordnance Survey Terrain 5 DTM data. The raw DTM data were presented as a raster of elevation in metres at a spatial resolution of 5m x 5m. Cubic convolution was used to resample elevation to 30m x 30m resolution, and slope was calculated from the resampled DTM in angular degrees using the slope tool in ArcMap 10.2.2. Elevation is shown in Figure 5.1 and slope is shown is Figure 5.2.

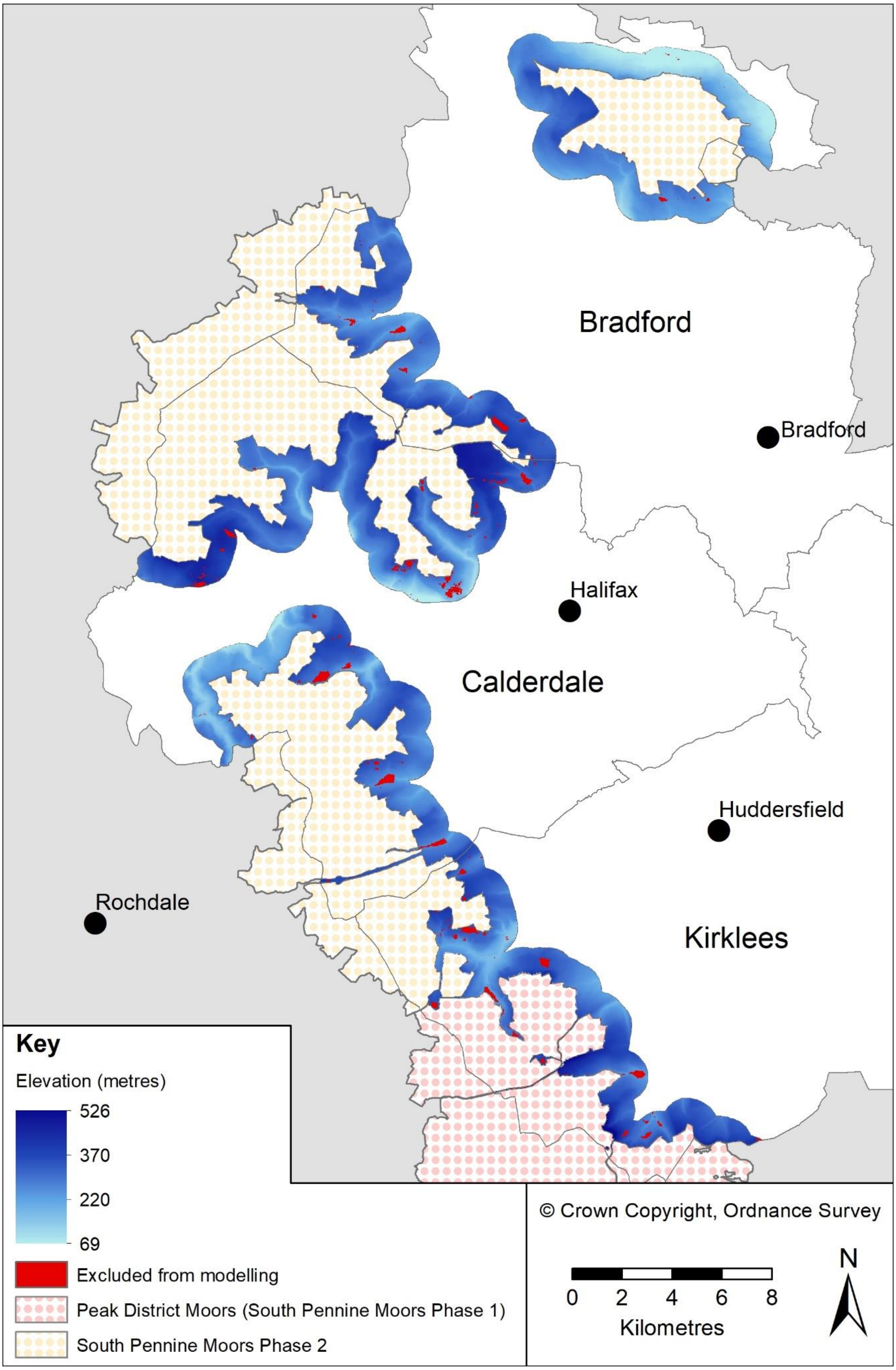

Figure 5.1 Elevation within the SMPSPA 1km fringe. Areas excluded from modelling due to incomplete coverage of at least one predictor variable are shown. Resolution is 30m x 30m.

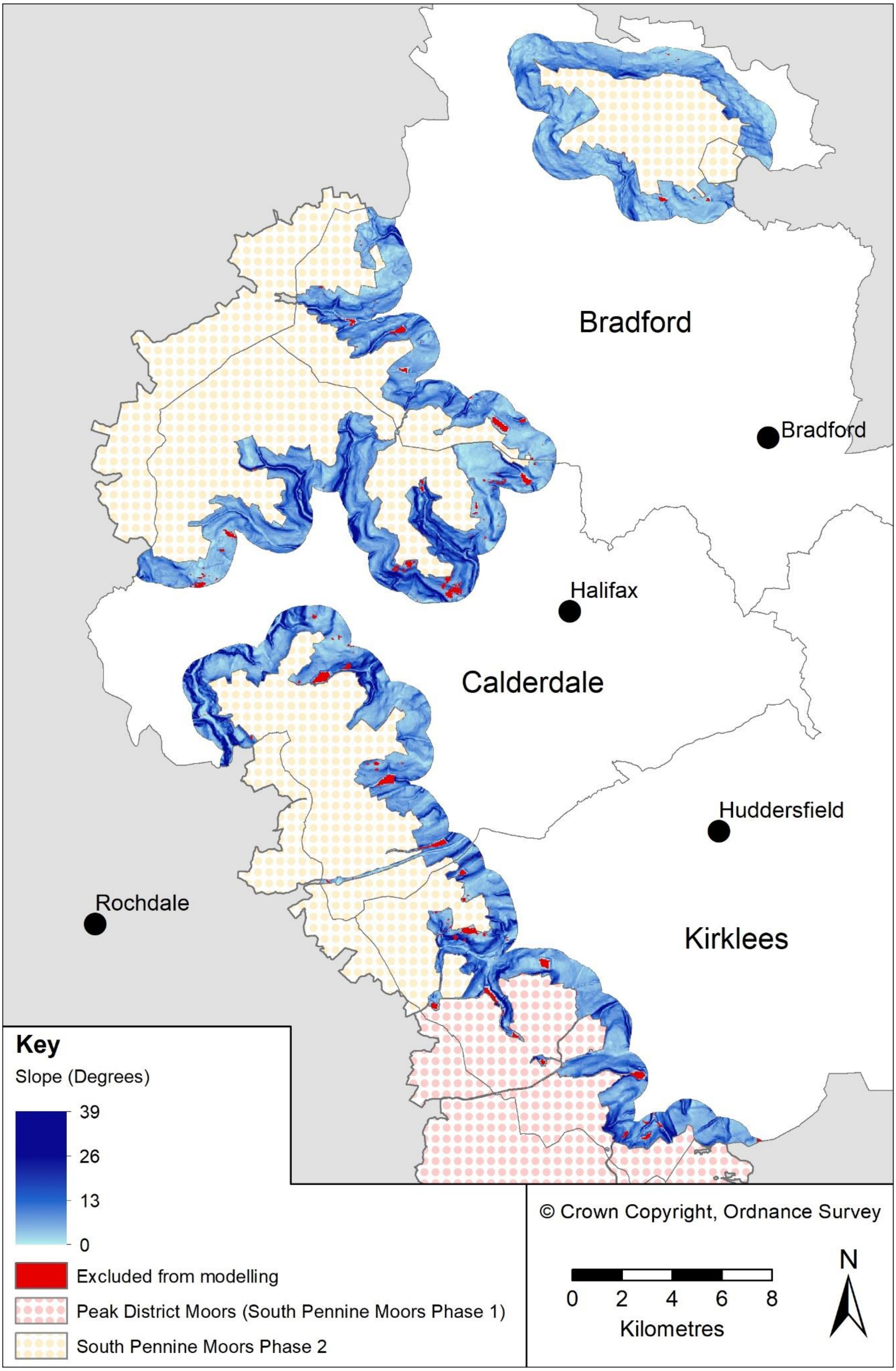

Figure 5.2 Slope within the SMPSPA 1km fringe. Areas excluded from modelling due to incomplete coverage of at least one predictor variable are shown. Resolution is 30m x 30m.

Roads have the potential to cause ecological disturbance to birds through a variety of mechanisms including habitat loss and fragmentation (Kociolek et al., 2011), traffic noise (Ware et al., 2015) and collision mortality (Summers et al., 2011). These factors can lead to the avoidance by birds of areas in close proximity to roads, reducing the area of available habitat (Thompson et al., 2015). As such, density of roads was included as a predictor variable in habitat suitability models. An OS Mastermap Integrated Transport Network (ITN) dataset was used to calculate road density. All roads were extracted from the dataset as line vectors and converted to a 30m x 30m resolution raster using the Line Density tool in ArcMap 10.2.2. Each raster cell represented the length of road within a 6om radius from the centre of that cell. Road density is shown in Figure 5.3. Building density was calculated using Ordnance Survey Mastermap data. All buildings in the study area were extracted as polygons and centroids for individual buildings were calculated. A building density raster was created at a resolution of 30m x 30m using the point density tool in ArcMap 10.2.4. The values of the raster represented the number of buildings within a 500m radial distance of any given pixel. Building density is shown in Figure 5.4. The impact of Small Wind Turbines (SWTs) on birds within the 1km SPMSPA fringe were investigated in Chapter Four, however the species under investigation in this chapter were not encountered in sufficient numbers for analysis. In this chapter, the impact of SWT density on the habitat suitability of Golden Plover, Snipe, Curlew, Lapwing and Wheatear will be investigated by including SWT density as a variable in habitat suitability modelling and determining the variable importance of this predictor relative to other predictor variables. Density of SWTs was calculated using point data for each SWT site that had been confirmed visually as built (n= 58) within 3km of the SPMSPA in Bradford, Calderdale and Kirklees (see Chapter Four). The Point Density tool in ArcMap 10.2.2 was used to create a raster at 30m x 30m resolution where each pixel was represented by an integer value indicating the number of SWT sites within a 1km distance of each pixel. SWT density is shown in Figure 5.5.

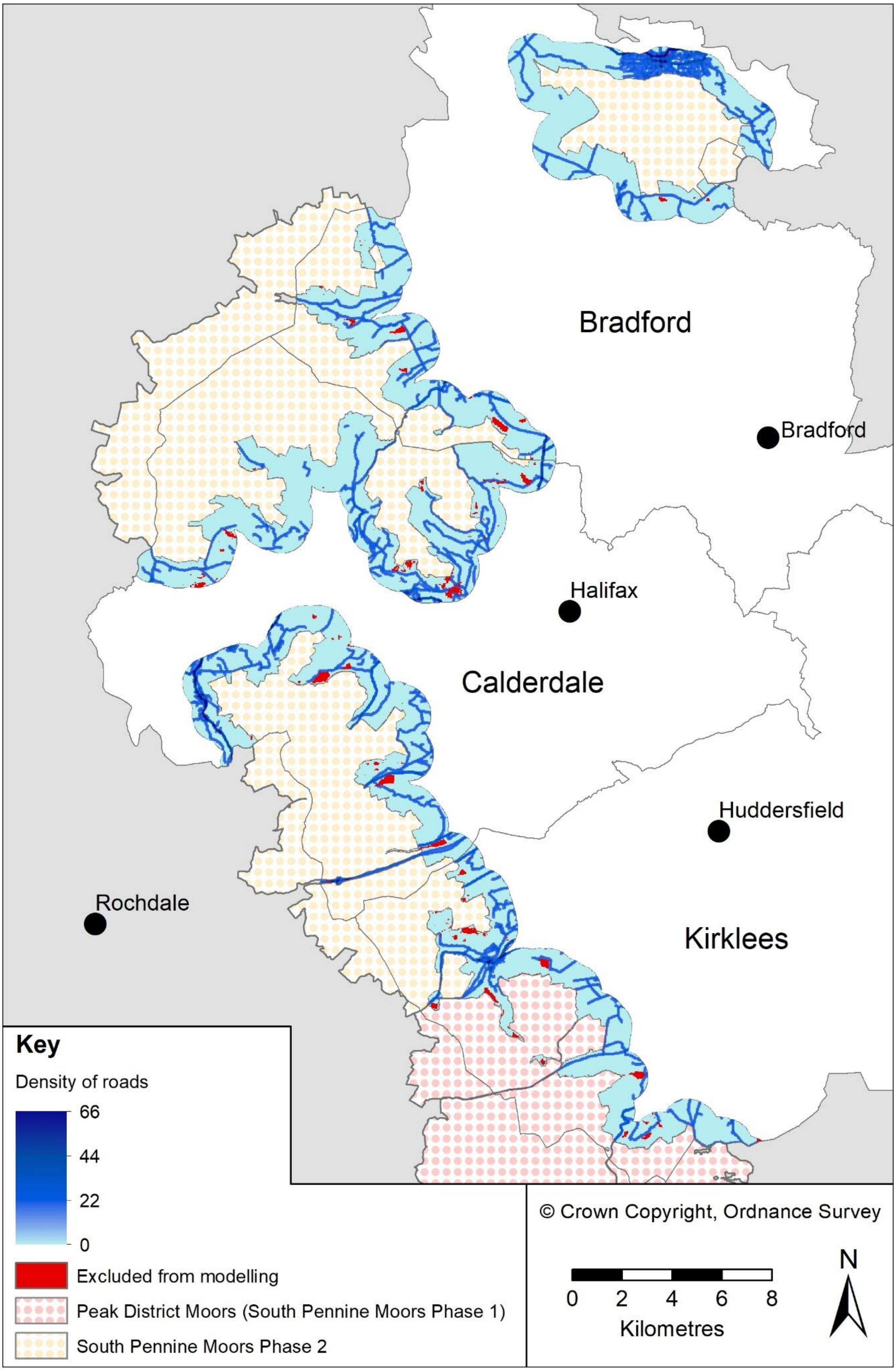

Figure 5.3 Density of roads within the SMPSPA 1km fringe. Areas excluded from modelling due to incomplete coverage of at least one predictor variable are shown. Spatial resolution is 30m x 30m.

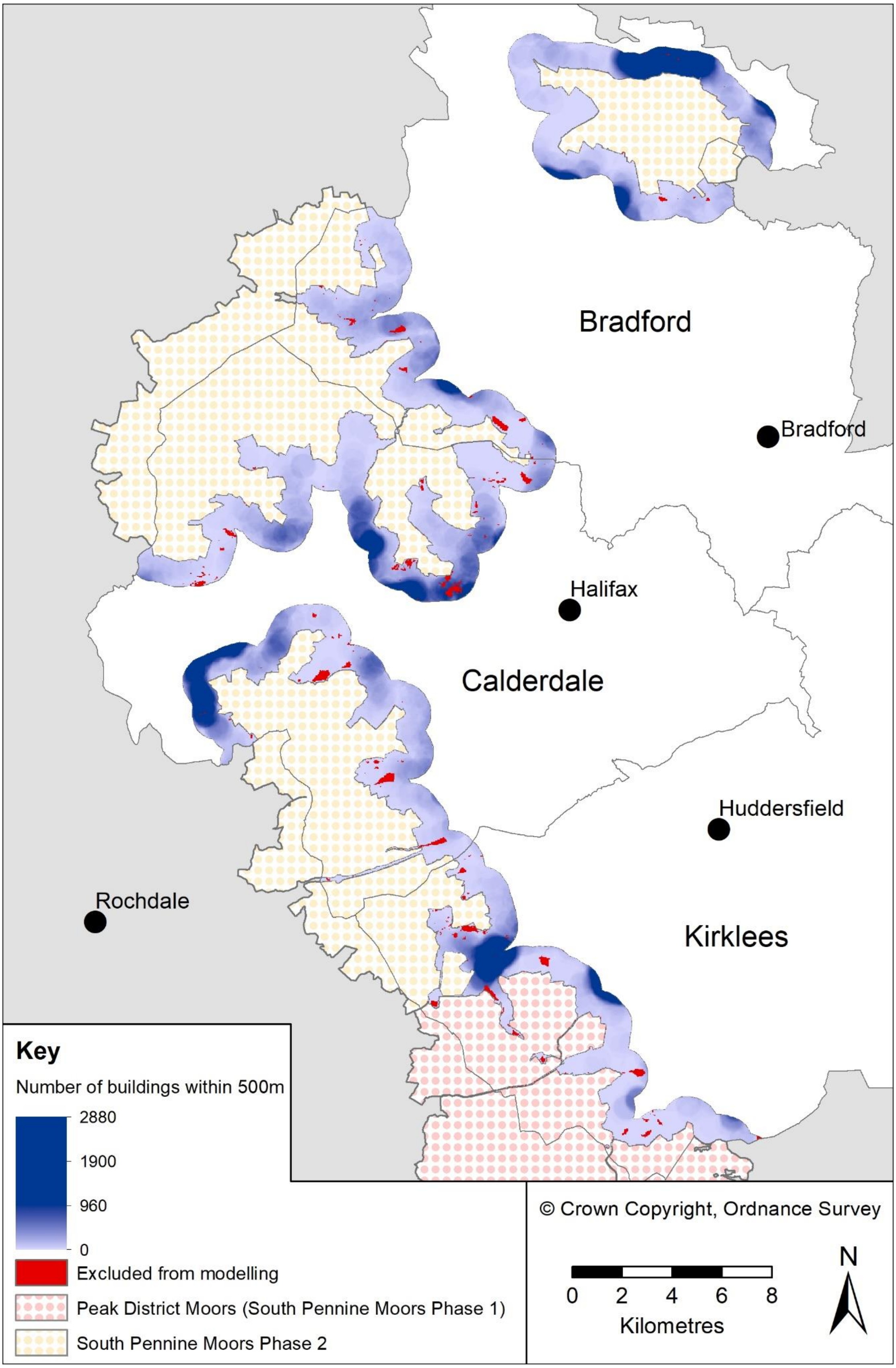

Figure 5.4 Density of buildings (number of buildings within 500m) within the SMPSPA 1km fringe. Areas excluded from modelling due to incomplete coverage of at least one predictor variable are shown. Spatial resolution is 30m x 30m.

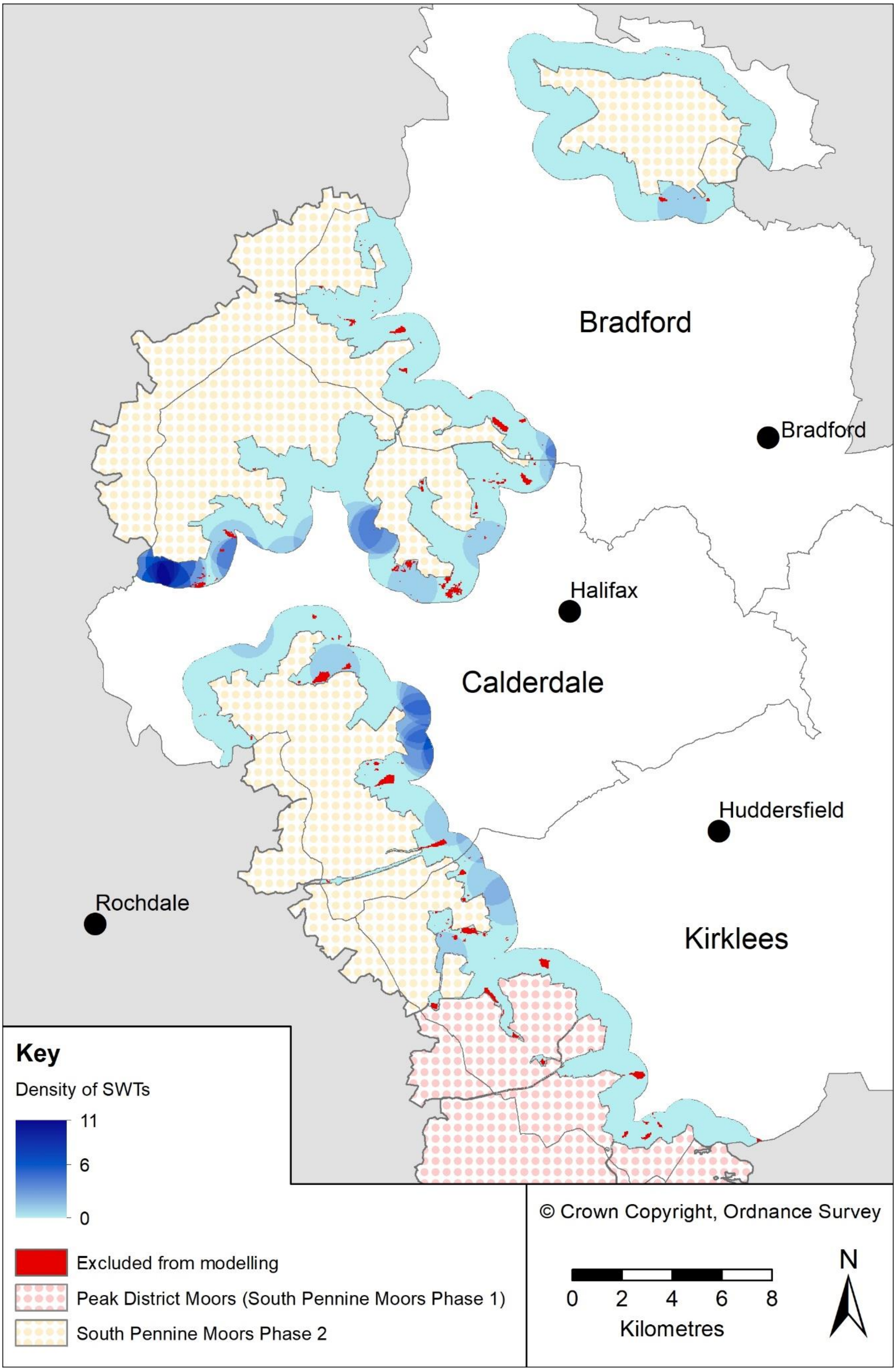

Figure 5.5 Density of Small Wind Turbines (SWTs) within the SMPSPA 1km fringe. Densities represent SWTs within 500m. Areas excluded from modelling due to incomplete coverage of at least one predictor variable are shown. Spatial resolution is 30m x 30m.

Agricultural intensification is a known driver of farmland bird declines in the UK, and as such the agricultural landscape was deemed to be integral in modelling the habitat suitability for the five species of moorland fringe bird species within this chapter. Three predictor variables encapsulating agricultural activity were included in habitat suitability models. The first was derived from the Provisional Agricultural Land Classification (ALC) and was obtained from Natural England (Natural England, 2012). This dataset represented agricultural land quality and are derived from factors including climate (temperature, rainfall, aspect, exposure, frost risk), site (gradient, micro-relief, flood risk) and soil (depth, structure, texture, chemicals, stoniness). and consisted of seven categories representing five qualitative grades of agricultural land (grade 1 = best, grade 5 = worst) and two non-agricultural categories (urban and ‘non-agricultural land’). Data were converted from spatial vector to a 30m x 30m resolution raster (Figure 5.6). The habitat classification map produced in Chapter Two was included as a predictor variable and represented dominant habitats within fields in the SPSMPA 1 km fringe at a resolution of 30m x 30m. Habitats represented were improved grassland, species rich semi-improved grassland, species poor semi-improved grassland, wet heath/ acid grassland matrix, dry heath/ acid grassland matrix and rush pasture (Figure 5.7). See Chapter Two for further details on the method used to create the habitat classification map.

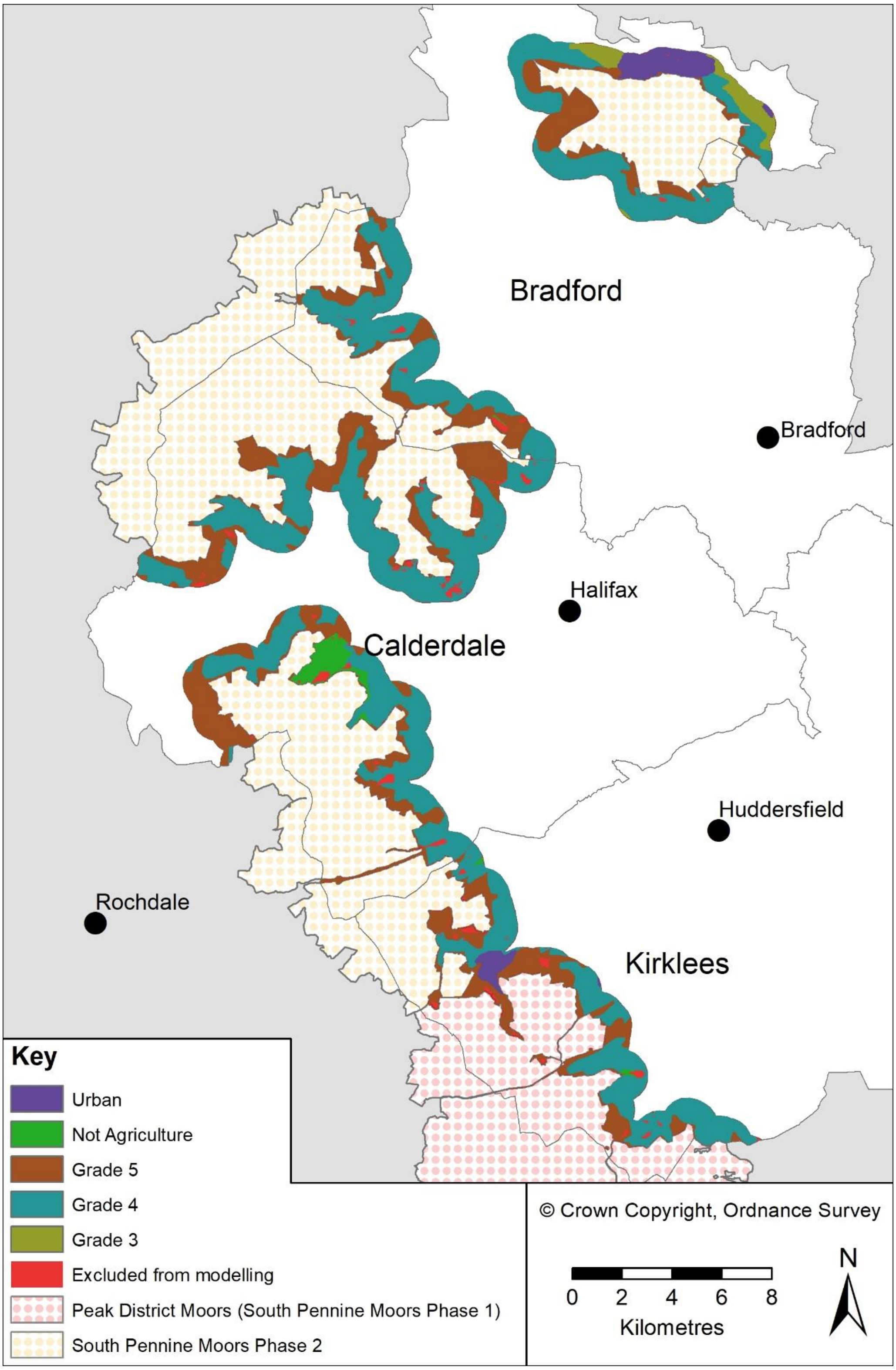

Figure 5.6 Agricultural grade of land according to the Natural England Agricultural Land Classification (ALC) within the SMPSPA 1km fringe. Nationally, Grade 1 represents the lowest quality agricultural land and Grade 5 the highest quality. Areas excluded from modelling due to incomplete coverage of at least one predictor variable are shown. Spatial resolution is 30m x 30m.

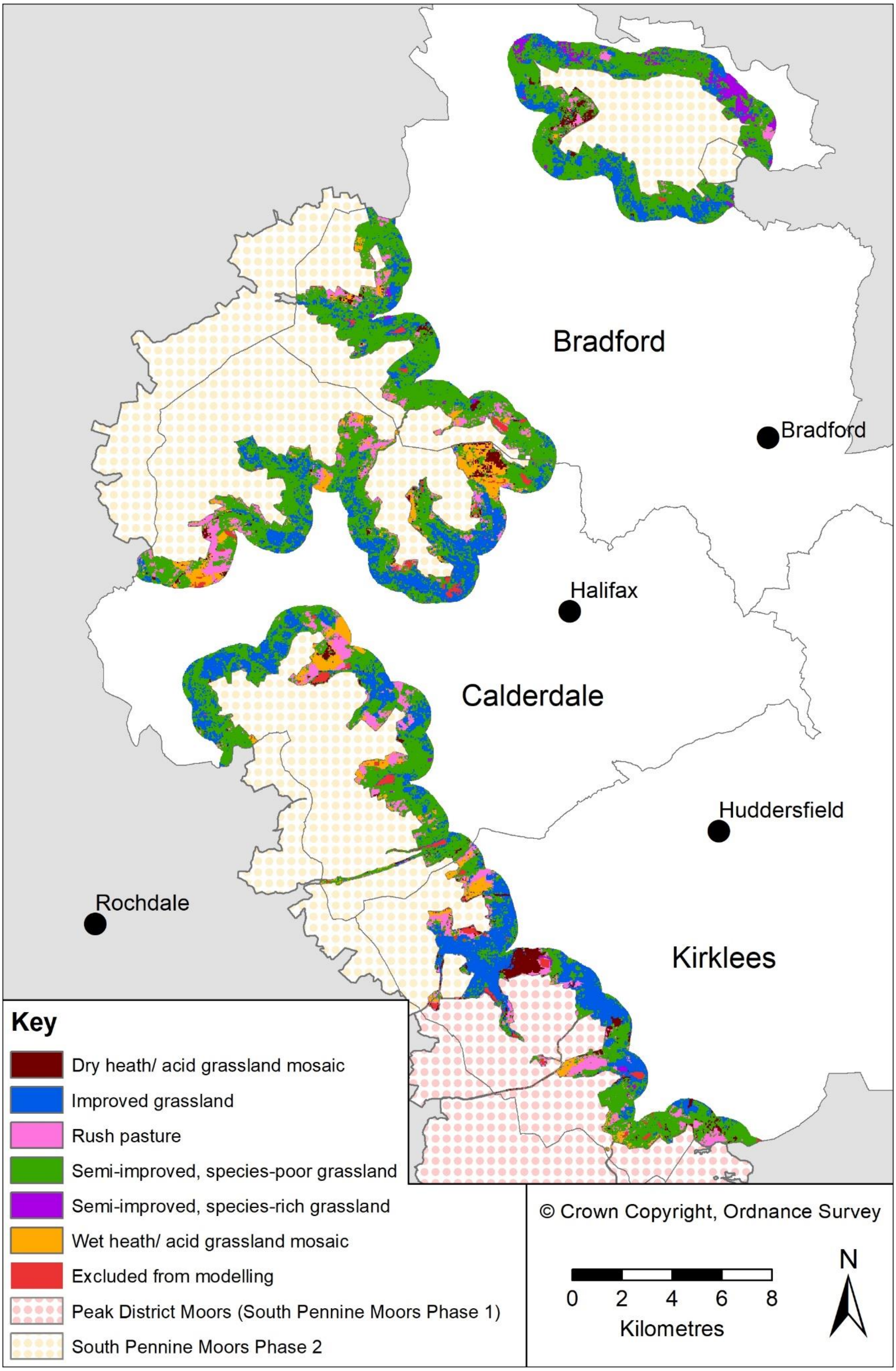

Figure 5.7 Distribution of habitats within the SMPSPA 1km fringe as modelled in Chapter Two. Areas excluded from modelling due to incomplete coverage of at least one predictor variable are shown. Spatial resolution is 30m x 30m.

The final empirical predictor variable represents Environmental Stewardship (ES) uptake within the SPMSPA 1km fringe landscape. Area of ES was calculated using a combination of ES data obtained from Natural England through a Freedom of Information Request and OS Mastermap data. The ES data consisted of point data for each land parcel within the study site that was engaged in an ES scheme from 2005 (the year ES was implemented) to 2015. This dataset was standardised (temporally) with the bird survey data by extracting ES sites that were live in Bradford in 2013, Calderdale in 2012 and Kirklees in 2012. As Calderdale was surveyed in 2015 as well as 2012, the best margin of error was achieved by including live ES agreements from 2015, but only in areas covered by the 2015 bird survey squares. Duplicated parcels were removed using parcel reference ID numbers (to avoid artificially inflating area of ES where multiple ES schemes were implemented on the same patch of land). In addition, ES schemes that were deemed irrelevant to moorland fringe bird species were removed from the dataset leaving 71 scheme types across Entry Level Stewardship (ELS), Higher Level Stewardship (HLS) and Organic Entry Level Stewardship (OELS). Each data point indicated area of ES was assigned to land parcels from Mastermap. Where more than one data point fell into a land parcel (e.g. two parts of a single field were used for spatially separate ES schemes), the area of the agreement was summed for that parcel. The resultant dataset was a vector layer representing the continuous variable of area of ES per land parcel (Figure 5.8).

5.3.5. Combined Landsat 8 and Empirical predictor variables

In order to assess whether introducing Landsat 8 data improves habitat suitability models based on empirical variables, a 44 band raster was created using all Landsat 8 predictor variables and empirical variables. In order to reduce collinearity between the numerical variables within this raster, the *vifcor* method was employed as described in section 5.3.3.

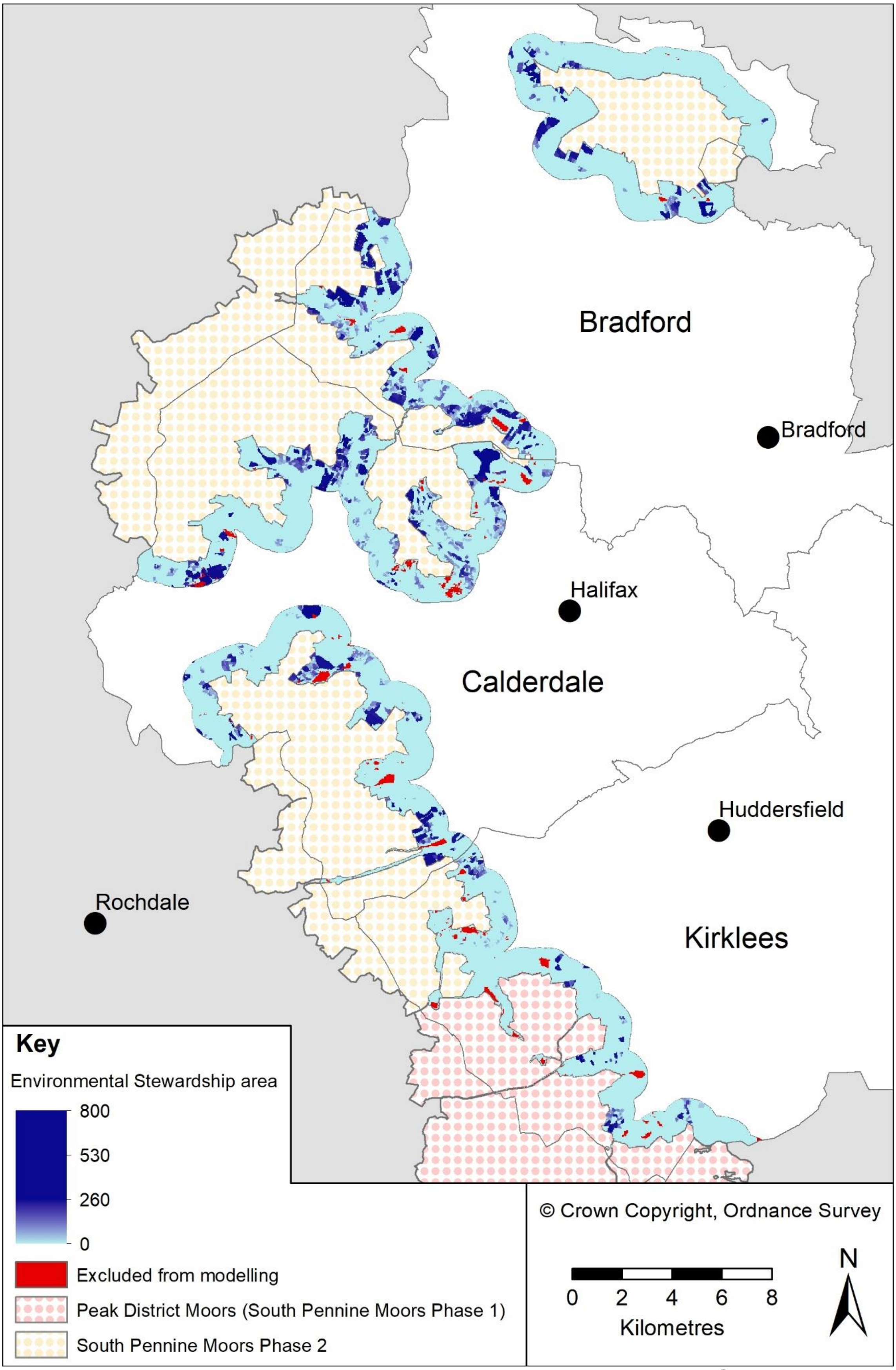

Figure 5.8 Area of land committed to Environmental Stewardship in m^2 (see text for details). Areas excluded from modelling due to incomplete coverage of at least one predictor variable are shown. Spatial resolution is 30m x 30m.

5.3.6. Habitat Suitability Modelling

After pre-processing predictor variables as described above, the R package *biomod2* (Thuiller et al., 2016) was used to perform habitat suitability modelling for Lapwing, Curlew, Snipe, Golden Plover and Wheatear within the 1km SPMSPA fringe.

Each species was modelled separately using (1) Landsat 8 predictor variables, (2) Empirical predictor variables, and (3) combined Landsat 8 and Empirical Predictor variables. For each model, a sample of pseudo-absences were randomly chosen from the background data using a 'disk' approach. This meant that only background data outside a radial distance of the square root of the area of the resolution of the predictor data of each presence point for each model could be selected as a pseudo-absence location. As the resolution of all predictor variables was 30m x 30m, the minimum distance a presence point could be from a pseudo-absence was 30m. The number of pseudo-absence was set at 10,000 which provided a reasonable compromise between computational time and pseudo-absence sample size.

Pseudo-absences were randomly chosen separately for each species, and predictor variable set, resulting in 15 modelling datasets. As an objective of this chapter was to investigate the relative performance of different modelling algorithms, each of the 15 datasets was used for habitat suitability modelling with nine different modelling algorithms. Pseudo-absences remained constant between models using the same dataset to facilitate comparability of results between algorithms within each modelling group. The modelling algorithms selected for habitat suitability modelling were a mixture of statistical and machine learning methods. The *biomod2* package calls upon a mixture of internal R functions and other packages to implement the model fitting process. The algorithms and associated packages used for this chapter were Generalised Linear Model (GLM) from the *glm* function in package p*stats* (R Core Team, 2013); Generalised Additive Model (GAM) from the *gam* package (Hastie, 2016); Generalised Boosting Model (GBM) from the *gbm* package (Ridgeway, 2015); Classification Tree Analysis (CTA) from the *rpart* package (Therneau et al., 2015); Artificial Neural Network (ANN) from the nnet package (Venables and Ripley, 2002); Surface Range Envelope (SRE), from the *biomod2* package; Multiple Adaptive Regression Splines (MARS) from the *earth* package (Hastie and Milborrow, 2016); Random Forest (RF) from the *randomForest* package (Liaw and Wiener, 2002) and Maximum Entropy (MaxEnt) which is linked to a standalone software package (Phillips et al., 2006). Bird presence data were split into 70% training data and 30% test data for model training and validation respectively. Due to the large number of models (n = 135), most hyperparameters were left at the default settings in *biomod2*. Exceptions were of the number of splines used in GAMs (k = 4) and the number of additional cross folds performed by ANN and GB (set to 2) in Landsat 8 and combined Landsat 8/Empirical models (due to model failure at default settings).

5.3.7. Model validation and thresholds

Area Under the Curve (AUC) of the Receiver Operating Characteristic (ROC) (Fielding and Bell, 1997) was calculated for each model undertaken using the 30% testing data. There are many validation metrics that can and have been applied to accuracy assessment in species distribution modelling (Liu et al., 2011), however AUC was chosen as it is a threshold-independent measure of model accuracy that illustrates a model's discrimination ability between two categories (Fielding & Bell, 1997). An AUC value of 0.5 indicates a model with no better discrimination than chance and an AUC value of 1 indicates perfect discrimination. The modelling algorithm and predictor dataset with the greatest mean AUC for a given species was deemed to be the best habitat suitability model.

Thresholding can be used to produce a set of binary prediction maps (i.e. suitable habitat versus unsuitable habitat. There are many available methods for the calculating dichotomous thresholds (Liu et al., 2005) and each thresholding method will produce a different dichotomous map of habitat suitability. In order to remove subjectivity, a thresholding method was not used here. Habitat suitability outputs were kept as raw probability outputs on a scale of 0 to 1. These outputs do not directly represent probability of occurrence, but represent a ranked scale of habitat suitability which allows the visual identification of the most suitable areas of habitat for a given species.

5.3.8. Variable importance and model comparisons

As part of the modelling process, biomod2 allows a relative variable importance score to be calculated for each independent term specified in the model. This algorithm shuffles each predictor in turn and compares the predictive output to the output of the model with unshuffled data, using Pearson's product moment correlation (Thuiller et al., 2016). This procedure was undertaken during the validation process with a relative importance score calculated for each predictor. In order to determine the most appropriate spatial resolution for each species. AUC was compared between empirical and Landsat models in order to determine which of the two predictor variable groups is the better predictor for each of the moorland fringe species.

5.4 Results

5.4.1. Variable selection

There was high collinearity between Landsat predictor variables. After VIF selection collinearity was removed and the Landsat predictor set was reduced to 13 variables (Table 5.1). None of the continuous Empirical predictor variables had collinearity issues as determined by the VIF protocol (max r = 0.35), and as such all seven predictor variables were used in modelling (Table 5.1). Presence data varied between species and across resolutions. After masking to the SPMSPA 1km fringe, five species remained with sufficient detections for modelling. Detection rates were: Curlew (n = 901), Lapwing (n = 604), Snipe (n = 85), Wheatear (n = 85) and Golden Plover (n = 68).

Table 5.1 Predictor variables remaining after removing collinearity by reducing correlation to less than 0.7 (r<0.7) and minimising VIF.

Landsat 8 Spring	Landsat 8 Summer	Landsat 8 Autumn	Landsat 8 Winter	Empirical
Blue	Blue	Blue	NIR	Slope
NIR	NIR	Panchromatic	SWIR2	Road density
TIRS2	SWIR2	TIRS2	TIRS1	Habitat classification
	TIRS1			Environmental Stewardship
				Elevation
				Building density
				SWT density
				Agricultural class

5.4.2. Model performance

Model performance as determined by AUC varied between modelling algorithms, between species and between predictor variable sets (Fig. 5.9). GBM and RF consistently performed better than other model algorithms, with GBM best predicting the habitat suitability of Golden Plover (AUC = 0.88) and Wheatear (AUC = 0.70) and RF best predicting habitat suitability for Curlew (AUC = 0.79), Lapwing (AUC = 0.90) and Snipe (AUC = 0.84) . Combining Landsat and Empirical data produced the best models for all species, except for Wheatear which was best modelled using Empirical data only (Fig. 5.2).

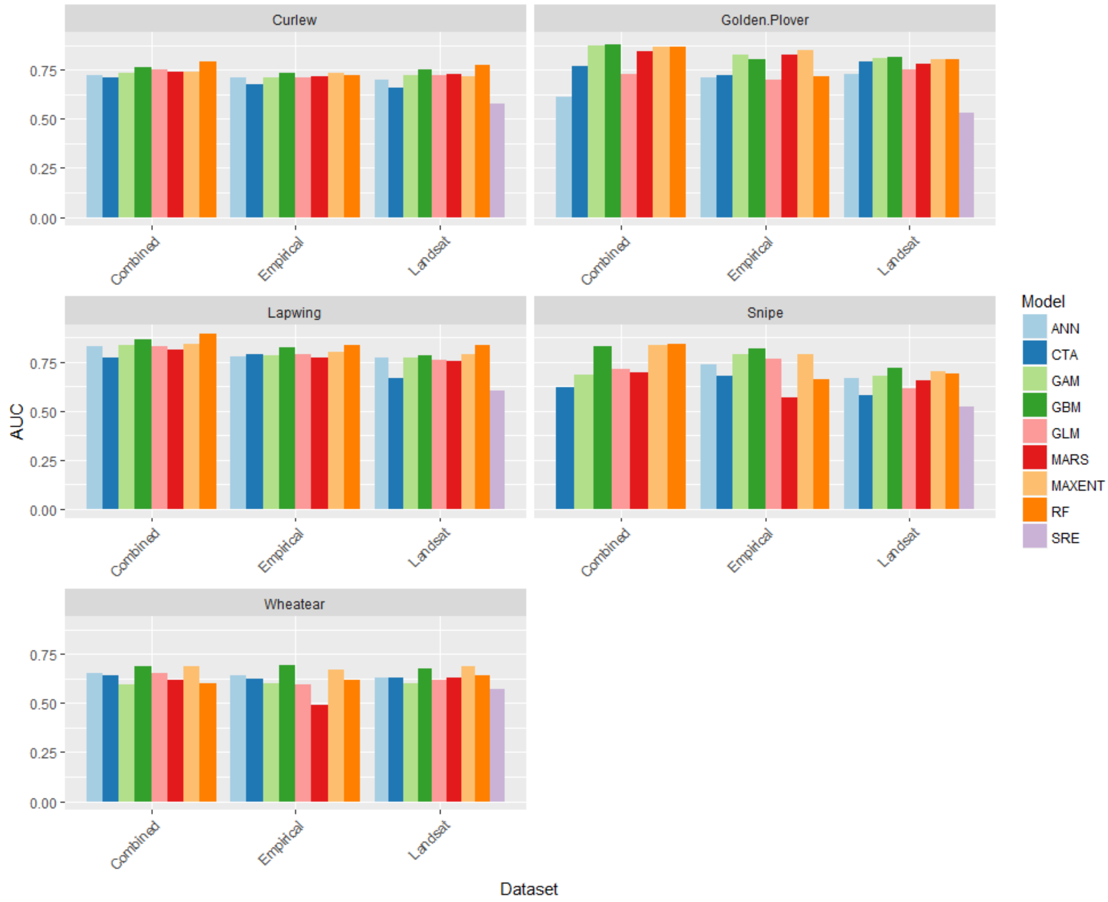

Figure 5.9 AUC values for all habitat suitability models constructed. AUC values were calculated using 30% testing data.

5.4.3. Variable importance

Predictor variable importance scores for the best performing models varied between species (Fig. 5.10). The most important variables for predicting the habitat suitability of Curlew were Landsat 8 Near Infrared (NIR) data from Spring and building density (Fig 5.3). Building density was also an important predictor for Wheatear, Snipe and Lapwing. The most important Landsat 8 predictor variables for Lapwing were NIR from Spring and Summer and Spring NIR for Snipe. Landsat 8 variables did not contribute to the habitat suitability model for Golden Plover, where Elevation appeared to be the most important predictor by far. Elevation was also important in the model for Snipe and Wheatear. Slope was the most important predictor for lapwing and contributed highly to the model for Wheatear. Of the empirical predictor variables, Environmental Stewardship area, SWT density, road density, agricultural class and habitat classification (as produced in chapter 2) did not contribute much to any of the habitat suitability models.

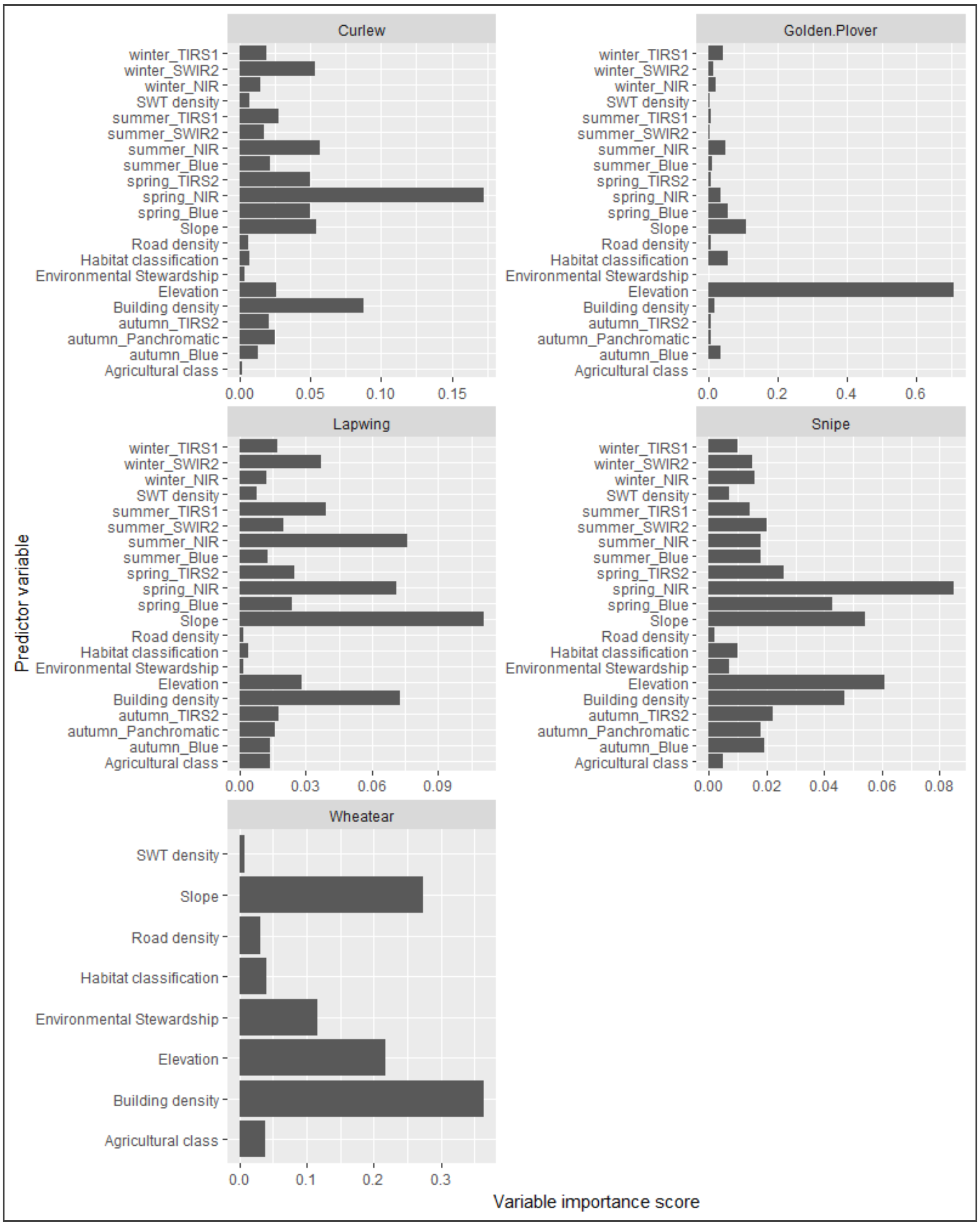

Figure 5.10 Variable importance scores values for the best performing models for each species (Curlew = Random Forest , Lapwing = Random Forest, Snipe = Random Forest, Golden Plover = Generalised Boosting Model, Wheatear = Generalised Boosting Model). Compined Landsat 8 and Empirical predictors produced the best models for all species except Wheatear which was best described by Empirical predictor variables. Scores are relative within models and are not directly comparable between models.

5.4.4. Habitat suitability estimates

Spatial predictions for habitat suitability from best performing models were relatively similar for all five species (Figures 5.11-5.14). Generally, the most suitable areas are towards the edge of the SPMPSA, however for all five species there are relatively suitable areas of habitat that extended across the entire SPMSPA 1km fringe. Where the fringe cuts into the SPMSPA, there are large areas of relatively suitable. As building density was an important predictor for most species, built development should be avoided in these areas. A notably consistent area that is predicted to have low habitat suitability is in the north of Bradford. This area is the location of Ilkley- a medium sized town with population of approximately 14,000. This pattern reinforces the importance of areas that are not developed for maintaining suitable habitat for moorland fringe bird species of conservation concern. As the values shown in Figures 5.11-5.14 are relative scores of habitat suitability, it is difficult to make inferences about areas of suitable habitat. However, pixels with relatively high scores appear to be grouped in several locations for all species. These areas should be the focus of habitat conservation efforts to benefit the bird species studied here.

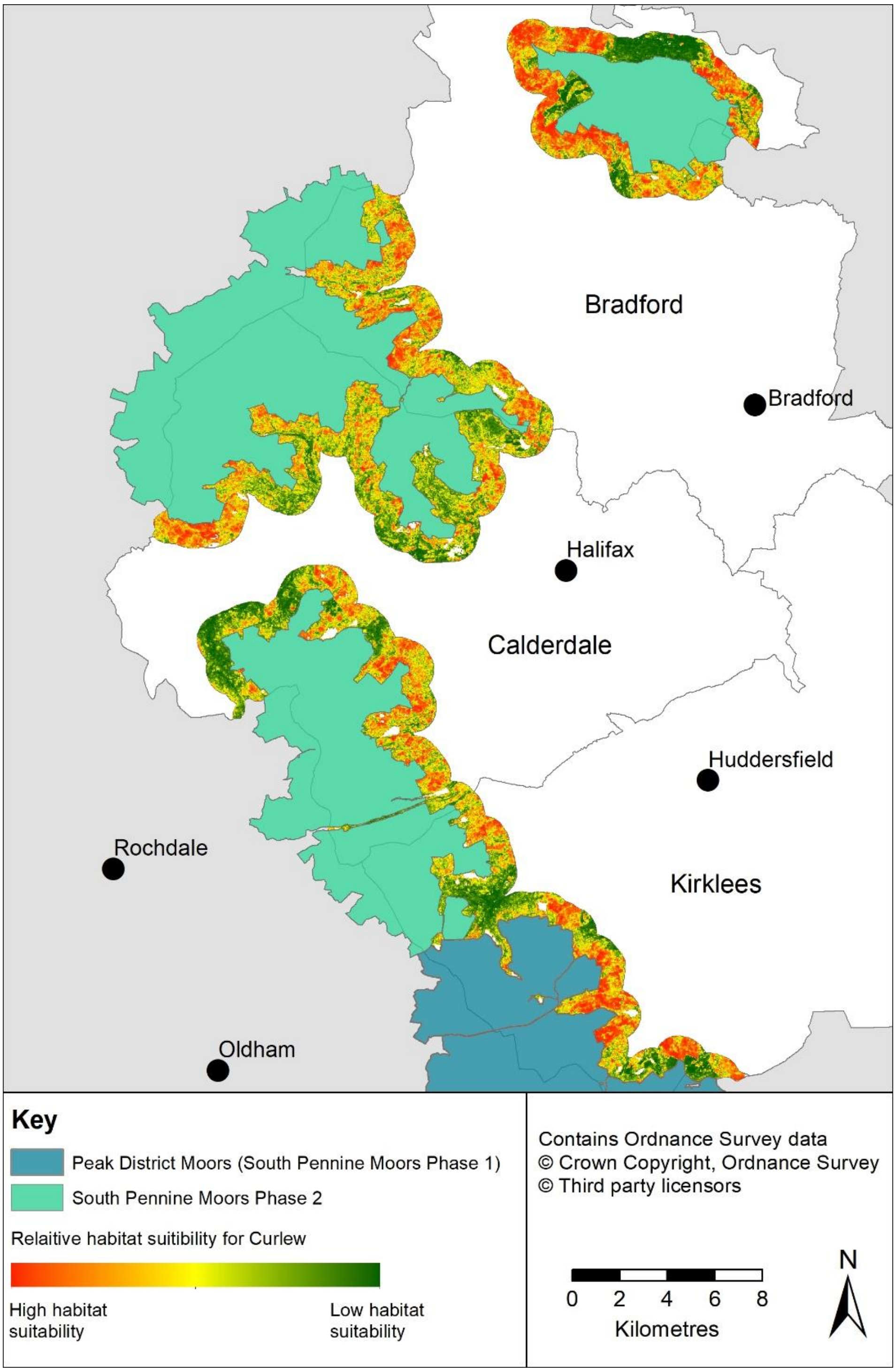

Figure 5.11 Map of habitat suitability for Curlew at 30m x 30m resolution, modelled using Random Forest with empirical predictor variables combined with Landsat 8 variables. Colours show relative suitability and are histogram equalised.

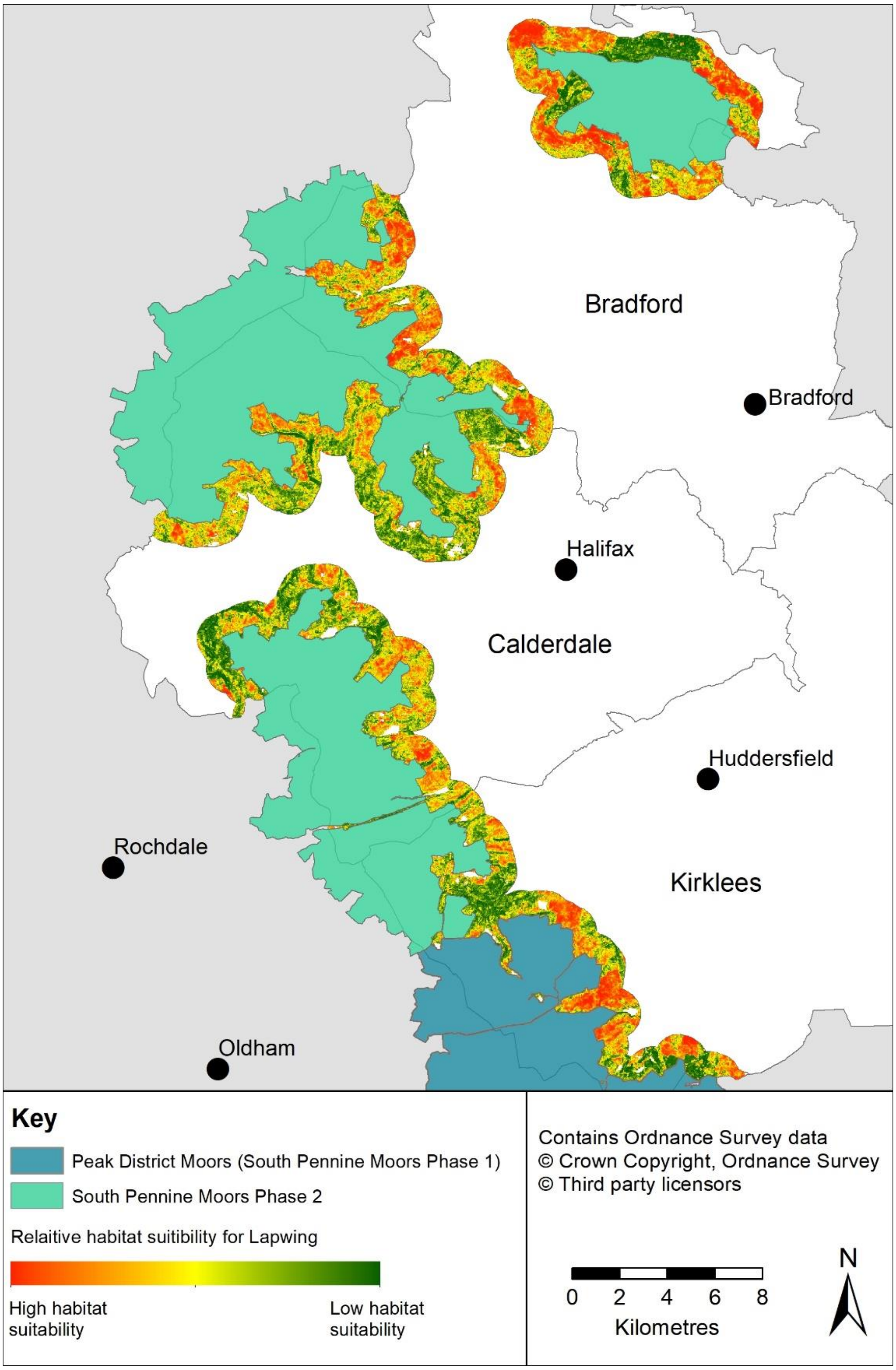

Figure 5.12 Map of habitat suitability for Lapwing at 30m x 30m resolution, modelled using Random Forest with empirical predictor variables combined with Landsat 8 variables. Colours show relative suitability and are histogram equalised.

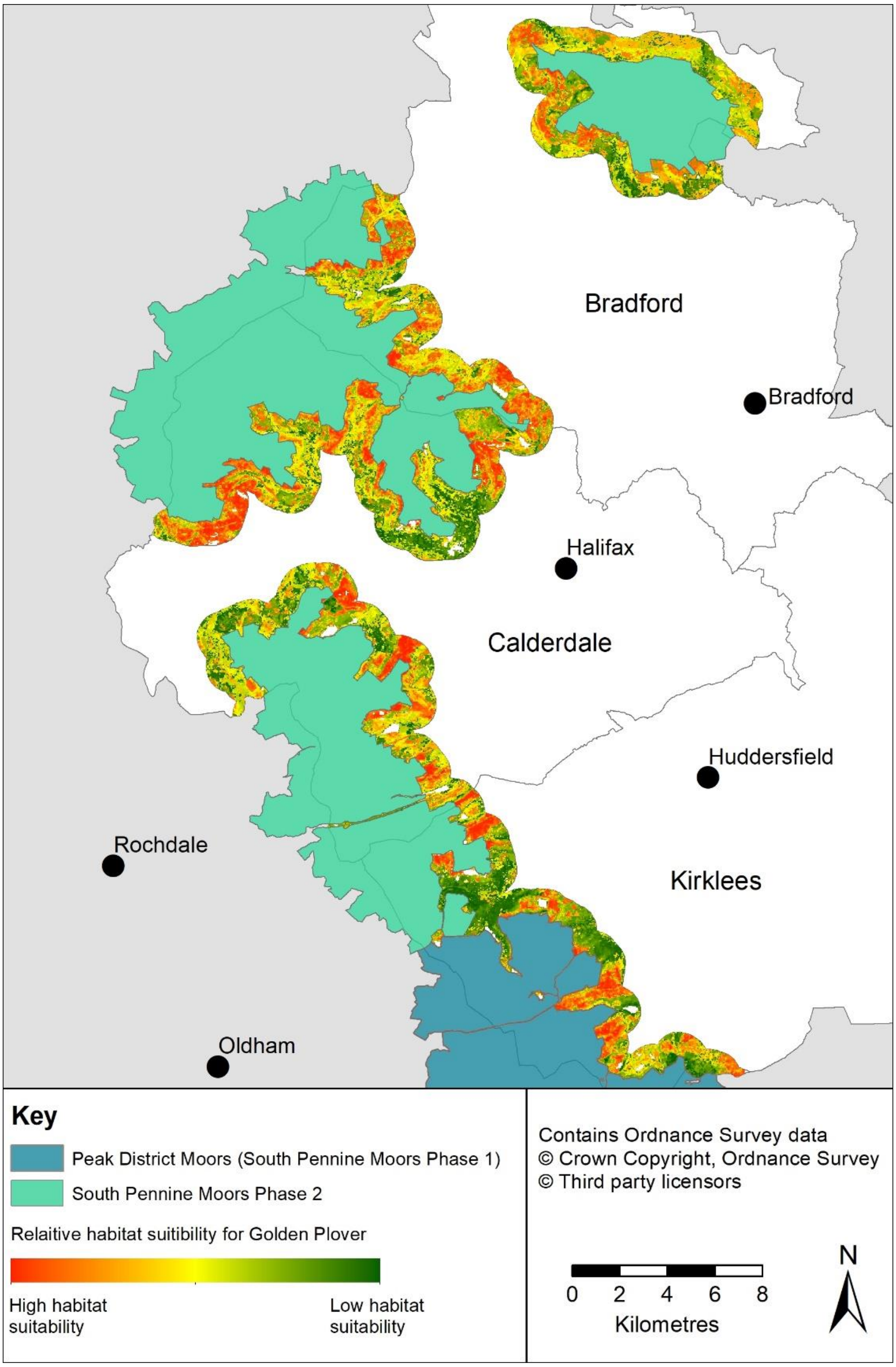

Figure 5.13 Map of habitat suitability for Golden Plover at 30m x 30m resolution, modelled using Random Forest with empirical predictor variables combined with Landsat 8 variables. Colours show relative suitability and are histogram equalised.

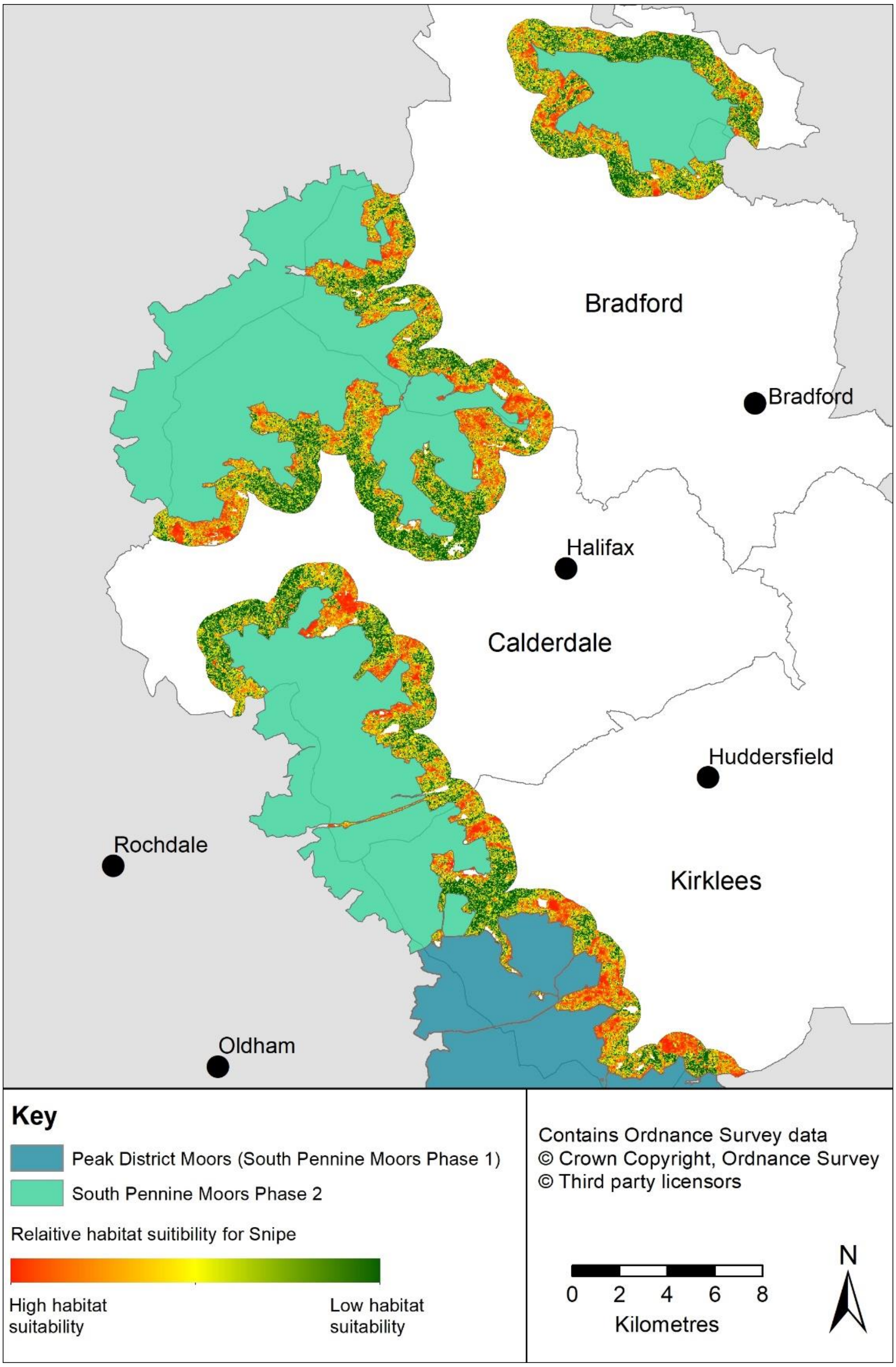

Figure 5.14 Map of habitat suitability for Snipe at 30m x 30m resolution, modelled using Random Forest with empirical predictor variables combined with Landsat 8 variables. Colours show relative suitability and are histogram equalised.

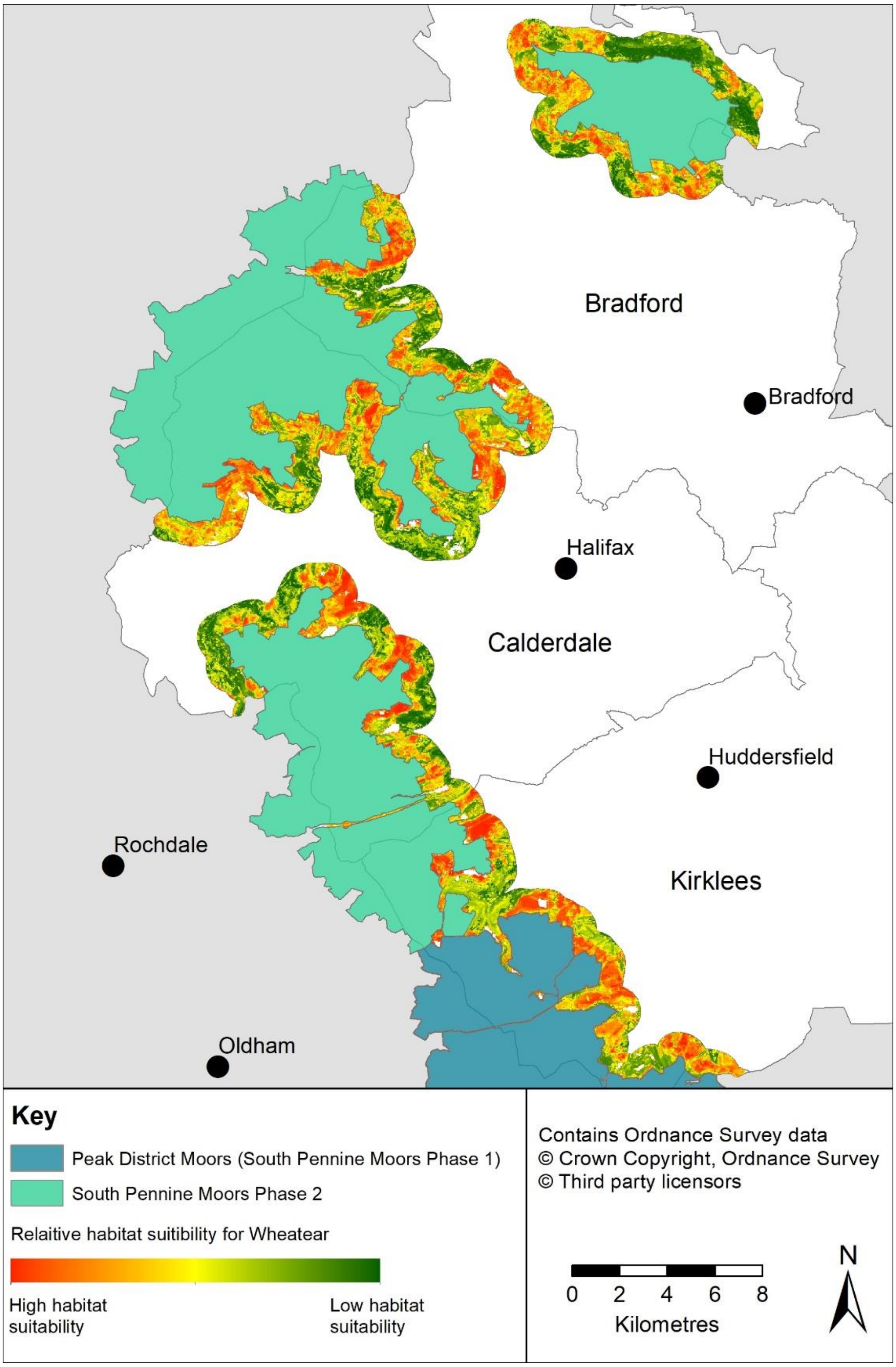

Figure 5.15 Map of habitat suitability for Wheatear at 30m x 30m resolution, modelled using Random Forest with empirical predictor variables combined with Landsat 8 variables. Colours show relative suitability and are histogram equalised.

5.5 Discussion

Raw Landsat8 data have been shown to have good predictive accuracy when used at the native resolution of 30m^2 to model species distributions (Shirley et al., 2013), however Landsat data is rarely used in its raw form for such applications. It has been advocated that when fine scale data is available and the resolution is ecologically relevant, then these data should be used for SDMs (Merow et al., 2014). The current study found that Landsat 8 data provides good predictive capability for moorland fringe bird habitat suitability modelling, but is nest when used in conjunction with other more ecologically interpretable data.

Maps of continuous values were chosen over logistic thresholding. This has the benefit of allowing relative site habitat suitability to be determined. In terms of local planning decisions, this may be beneficial if several sites are being considering for a development, as it allows sites to be ranked by probability of impact upon a given species. If a dichotomous distribution map is explicitly required, then careful assessment of the output should be undertaken, perhaps with expert opinion as a validation method (Gastón et al., 2014).

No one single set of spectral bands were found to be the best predictors of the species distributions within this study. This makes justifying the use of remotely sensed spectral data for use in Species Distribution Models difficult in terms of ecological relevance. As reducing collinearity between spectral bands was undertaken using a mathematical process and not ecological theory in this study, this problem is compounded. Vegetative indices have been developed to overcome this problem (Cohen and Goward, 2004) and the use of such indices to supplement empirical data can improve model performance. As raw Landsat 8 data was used here, vegetative indices were not employed. Further investigation into the complementarity of raw spectral data with non-correlated vegetative indices is recommended, as it may have the potential to improve habitat suitability performance. The results of this study suggest that the raw spectral data does not detract from good model performance, with some models attributing greater importance to raw spectral bands than empirical data.

The use of empirical data in this study represents a more traditional approach to Species Distribution Modelling, however studies are normally conducted at coarser resolutions using km^2 rather than m^2 (e.g. Araujo et al., 2005). Species Distribution Models should be used at a resolution that is ecologically relevant to the study species (Austin and

Van Niel, 2011). In the case of this thesis, the usage of fields by moorland fringe birds is of primary interest. As such, coarse resolutions would reduce the interpretability of distribution models at the field level. Good predictive accuracy was found for all species in this study using empirical data at $900m^2$ indicating that this is an appropriate resolution. Across the Landsat and Empirical models, no one algorithm performed better than any other, although RF and GBM were consistently best performing. This shows the importance of using multiple modelling algorithms to guide conservation practice where Species Distribution Models are used as a tool. Based on the results of this study, GBM or RF are recommended as first port of call modelling algorithms for habitat suitability modelling. Although no consensus approach was used here, model averaging techniques exist (Araújo and New, 2007; Zhang et al., 2015) and may be appropriate where there is disagreement between modelling algorithms. This is especially true as the link between ecological theory and different modelling methods is poorly understood (Elith and Graham, 2009).

In summary, Landsat reflectance data has the potential to be a very useful tool in species distribution mapping, but is best used in conjunction with empirical data. The empirical data used here also proved to be a good predictor of moorland fringe bird species habitat suitability. Empirical data collection can be costly in terms of time and resources, especially if primary data collection is involved. This study shows that where resources are not available for such data collection, the use of freely available Landsat 8 data may be a good proxy. The predictor variables contributing to the habitat suitability models for each species differed, indicating that the moorland fringe should be managed taking the requirements of each priority species into account individually. Lapwing, Curlew, Snipe and Wheatear appear to be the most sensitive species to increased housing density.

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CHAPTER 6: CONCLUSIONS AND FUTURE RESEARCH

6.1 Main research findings

6.1.1. Moorland fringe habitat characteristics

The UK uplands are of international conservation importance and for their range semi-natural moorland habitats (Littlewood et al., 2006) and many of these landscapes have been granted SPA (Special Protection Area) status due to their breeding bird assemblages (Thompson et al., 1995). The upland moorland habitats of blanket bog and dwarf heath (including heather moorland) cover around 23.6% of Scotland, 3% of England and 6.2% of Wales, and are considered to be biodiversity action plan priority habitats by the Centre for Ecology and Hydrology (CEH) in the UK (Carey et al., 2008). The South Pennine Moors Special Protection Area (SPMSPA) is a typical upland SPA in the North of England that is surrounded by historically industrial town and cities and a matrix of farmland and smaller residential areas. Between 1947 and 1980, around 20% of upland heather moorland in England and Wales was subject to adverse land-use change (Thompson et al., 1995). Of the remaining, 70% was estimated to be at risk of further change, with more recent research citing atmospheric deposition, climate change, and peat erosion due to the legacy of overgrazing as risks to moorland habitats (Holden et al., 2007).

Chapter two of this thesis aimed to investigate the composition of habitats within the moorland fringe landscape using the SPMSPA as a case study. The key findings were:

- Temporal change in the coverage of habitats typically associated with the SPMSPA but within the SPMSPA fringe were investigated. It was found that between the years 1990 and 2000 these habitats increased in proportional coverage by around 5%, with a subsequent decrease in proportional coverage of around 7% between 2000 and 2007. If generalised, this finding is concerning as this would represent significant degradation and loss of upland habitats is prevalent in the fringe landscapes of upland protected areas in the UK.

- The SPMSPA fringe landscape is composed of a heterogenous habitat mosaic, predominantly composed of smaller fields dominated by agricultural habitats with larger but less frequently encountered patches of upland habitat. From a conservation perspective, the most concerning finding was that improved grassland and semi-improved species poor grassland were the most abundant habitats, dominating 78% (63% by area) of all fields surveyed within 1km of the SPMSPA whereas semi-improved species rich grassland and unimproved grassland was only encountered in two fields in a single unitary authority (Bradford).

- A supervised classification of the SPMSPA fringe habitat was achieved with an overall accuracy of 93.11%. This high accuracy provides evidence that farmland habitat categories incorporating an element of management intensity can be inferred from Landsat 8 data. Predicting the entire SPMSPA fringe habitat showed that the pattern of high coverage semi-improved species poor grassland and improved grassland is generalisable.

6.1.2. Moorland fringe bird habitat associations

Upland moorland habitats in the United Kingdom (UK) support important breeding populations of migratory and resident breeding bird species. Some of these species are listed under annex 1 of the EU birds directive (Thompson et al., 1995) including Golden Plover *Pluvialis apricaria*, Short-eared Owl *Asio flammeus* and Merlin *Falco columbarius*. Other species of international conservation concern that are supported by the uplands within the UK include Dunlin *Calidris alpina*, Curlew *Numenius arquata* and Twite *Acanthis flavirostris* (Thompson et al., 1995) alongside species of UK conservation concern including Lapwing *Vannellus vannellus*, *S*nipe *Gallinago gallinago*, Redshank *Tringa totanus*, Common Sandpiper *Actitus hypoleucos*, Whinchat *Saxicola rubetra*, Wheatear *Oenanthe oenanthe* and Ring Ouzel *Turdus torquatus* (Eaton et al., 2009). These thirteen species were identified in collaboration with the local unitary authorities of Calderdale, Kirklees, and Bradford in West Yorkshire, UK to be of priority for the study as part of this project into their relationship with the moorland fringe landscape of the SPMSPA. (Dallimer et al., 2010). It is known that there is an association between moorland habitat management and the management of the surrounding farmland in terms of the success of some of these species (Dallimer et al., 2012). Lapwing, Snipe, Curlew and Golden Plover are amongst the species that are known to use both moorland and farmland habitat to varying degrees in their breeding and feeding ecology (Pearce-Higgins and Yalden, 2003; Dallimer et al., 2012). Chapter two described habitat gradients within the moorland fringe landscape using NMDS (Non-metric Multidimensional Scaling) with a variety of quantitative field-level variables. The associations between these habitat gradients and the presence of five moorland fringe bird species were investigated (Golden Plover, Lapwing Snipe, Curlew, Wheatear). Numbers of records of other species were two low for analysis. Overall bird diversity was also explored in relation to the primary moorland fringe habitat types surveyed as part of Chapter two. Key findings of Chapter three were:

- The presence and absence of Golden Plover, Lapwing, Snipe, Curlew and Wheatear can all be described by the same habitat gradient. The end of this gradient that is favoured by these species comprises fields with tussocks, wet flush and evidence of intensive grazing. All this species showed preference against fields where the vegetation is mechanically cut.

- The habitat preferences of Snipe and Wheatear included a second gradient where fields dominated with rush were preferred over semi-improved grassland. Building density did not appear to influence the presence of these species over field level habitat and management variables.

- In terms of bird diversity, extrapolated species richness was found to be greatest in habitats not typical of moorland or farmland (e.g. woodlands, gardens and waterbodies). This shows the importance of maintaining a broad range of habitats within the moorland fringe landscape to preserve overall bird diversity.

6.1.3. Small Wind Turbines and birds

The availability of financial incentives for small-scale electricity generation within the UK has led to an increasing trend towards the construction of SWTs (Small Wind Turbines) in areas of high wind resource availability. The impacts of large wind turbines on birds are well studied (e.g. Bright et al., 2008; Schaub, 2012; Péron et al., 2013), however the ecological effects of SWTs (energy generating capability <50kW) on UK biodiversity are not well understood, making it difficult for local authorities to make informed planning decisions. Minderman et al. (2012) is the only experimental scientific paper has empirically quantified SWT-bird interaction finding no negative effect on the flight behaviour of birds within 20m of SWTs. Chapter four investigated the habitat composition around SWTs, the effect of SWTs on bird diversity within 600m and the effect of SWTs on 21 bird species within 600m. The key findings were:

- No difference in habitat composition was found within 600m of individual SWTs, however there were significant differences in habitat composition between SWT sites. In addition, 15 habitat types were encountered within 600m of SWTs. This indicates that SWTs are sited in locally heterogenous areas, but that there is no consistency in the habitat configuration that SWTs are built in.

- Bird diversity and species richness was consistently found to be lower within 100m of SWTs than any other distance out to 600m. In addition, extrapolated species richness was greatest 100-200m from SWTs. This suggests that there may be a displacement effect on some species within 100m of SWTs.

- Distance from SWT (up to 500m) had a significant positive effect on the presence of Magpie *Pica pica,* Starling *Sternus vulgaris* relationship was found between distance from SWTs and other species.

6.1.4. Habitat Suitability Modelling of moorland fringe birds

The large extent of UK uplands and the logistical constraints of conducting ecological surveys in these remote areas, makes it difficult to estimate the abundance and distribution of upland birds. This also applies to the moorland fringe, where land ownership and lack of complete access can complicate surveys. Species Distribution Modelling (SDM) or Habitat Suitability Modelling is a well-established technique that is used to produce expected occupancy values/ habitat suitability index values over large areas where complete field surveys are not feasible. Many modelling algorithms have been developed or adapted to implement Habitat Suitability Modelling, however there is considerable debate and uncertainty over which of these are the most appropriate under different scenarios (Li and Wang, 2013). Habitat Suitability Models invariably rely on predictor variables that are easy to interpret such rainfall or vegetative habitat characteristics (e.g. Porzig et al., 2014). Remotely sensed satellite images such as those acquired by the Landsat missions have the potential to provide high resolution predictor variables for Habitat Suitability Models, but have only been used in a limited number of studies (e.g. Shirley et al., 2013). More frequently, classified satellite imagery is used, which results in loss of data (by converting continuous data to categorical) and the potential amplification of error through cumulative modelling (i.e. modelling habitats, then modelling habitat suitability). Sometime derived vegetative indices are used, which maintain resolution and data integrity, but only for a limited number of spectral bands. In Chapter five of this thesis, Habitat Suitability models were built for five moorland fringe bird species using readily interpretable environmental variables (referred to as Empirical variables) and raw Landsat 8 spectral data.

The key findings of Chapter five were:

- Random Forest (RF) and Generalised Boosting Model (GBM) were consistently the best performing modelling algorithms as determined by AUC. It Is recommended that these algorithms are the first choice for modelling the habitat suitability of moorland fringe bird species.

- Supplementing Empirical data with Landsat 8 data improved the predictive performance of habitat suitability models for Curlew, Lapwing, Golden Plover and Snipe.

- Of the empirical predictor variables, building density was the most important predictor of habitat suitability for Wheatear, Curlew and Snipe. Slope was the most important for Lapwing and elevation for Golden Plover. Road density, area of Environmental Stewardship land, SWT density, agricultural class and habitat classification (as produced in chapter 2) all had very little influence on the best performing models across all species.

6.2 Conservation and management implications

This thesis has attempted to assess the SPMSPA fringe landscape in its ability to support thirteen bird species that are inextricably associated with the SPMSPA alongside the fringe's importance for maintaining bird diversity. This was achieved by describing the habitats, topography and the built environment of the fringe landscape; determining habitat gradients within the landscape; investigating the associations of priority bird species and bird diversity with these landscape characteristics; assessing the importance of agricultural practices, building densities and Small Wind Turbines in influencing the distribution and habitat associations of bird species; and attempting to model suitable habitat for five of the conservation priority species. The aim was to provide the ecology and planning departments of three unitary authorities in West Yorkshire (Calderdale, Kirklees and Bradford) with an evidence base that can be used to inform ecologically sound planning decisions in an upland and agricultural landscape that is under pressure to be built on, and is home to an important upland SPA. The following conservation management recommendations apply:

- Upland habitats within the SPMSPA fringe were found to be regionally localised in Chapter two. As maintaining habitat connectivity around and between protected areas is important for providing corridors to movement for species, the conservation of these areas should be a priority task in the management of the SPMSPA fringe landscape in the context of planning development decisions. This is further emphasised by the fact that upland habitats were found to decrease in area in the SPMSPA fringe between the years 2000 and 2007. The loss of upland habitats in the SPMSPA fringe were determined using landcover data which is periodically produced by CEH (Centre for Ecology and Hydrology) in the UK. It is recommended that this trend be monitored over time by applying the methods used here on future datasets as they are released. This also allows geographical hotspots of change to be identified allowing intervention efforts to be focussed in areas of high risk.

- The lack of unimproved grassland and particularly high land cover of improved and semi-improved agricultural land should be addressed to maintain heterogeneity of the agricultural landscape. The improvement of agricultural land is associated with the loss of bird species richness and biodiversity as a whole (Billeter et al., 2008) indicating that the SPMSPA fringe landscape is sub optimum for the conservation of bird diversity. The pattern of loss of unimproved grasslands in favour of improved meadows has been documented elsewhere in upland landscapes, with Graf et al. (2014) reporting a 20% decrease in unimproved meadows between 1987 and 2010 in a region of the Swiss Alps. As the primary function of the SPMSPA is the conservation of breeding moorland birds, efforts should be made to promote less intensive agricultural practices in the moorland fringe.

- A single habitat gradient were found to be of importance to Snipe, Curlew, Lapwing, Golden Plover and Wheatear. These species show preference for grazed fields with tussocks and wet flushes and preference against mechanically cut grassland. In addition, Snipe and Wheatear show preference for fields dominated by rushes and preference against semi-improved grassland. As very little species rich-improved grassland was found within the SPMSPA fringe, this can be interpreted as species-poor semi improved grassland. Understanding the association of moorland fringe bird species with species-rich semi improved grassland and unimproved grassland would require further research at another location.

- The impact of SWTs (Small Wind Turbines) was found to be minimal within 200-600m, however with 100m there was a decrease in bird diversity. A radial distance of 100m should be interpreted as a distance between two SWTs of 200m. This should be taken into consideration when making planning decisions for SWTs where SWTs are already in situ to avoid cumulative effects. This is likely to be particularly important where many SWTs are located near to one another, however the effect of clustered turbines was not investigated within this project. There was an association between distance from turbine with the presence of Starling and Magpie. No other species appeared to show any association within 500m.

- The fact that habitat composition was significantly different between SWT sites but not between SWT distance bands indicates that SWTs are sited in locally heterogenous areas, but that there is no consistency in the habitat configuration where SWTs are built. This means that there is potential for interaction between SWTs and a broad range of bird species exhibiting different life history strategies. This is in contrast to large wind farms, where soaring and flocking migratory birds are often considered to be of primary concern (e.g. Barrios and Rodríguez, 2004; Plonczkier and Simms, 2012). This is highlighted by the fact that Magpie, Collared Dove, House Sparrow, Great Tit and Willow Warbler showed associations with SWT distance, suggesting that the assemblage of birds that have the potential to be negatively affected by SWT are not necessarily the same as those affected by larger wind turbines.

- Habitat Suitability Model maps produced in Chapter five can be used as a tool to assist planning permission decisions for development. Although these maps do not show the absolute probability of habitat suitability, they provide a relative measure. This gives a good indication of areas that are the most suitable for supporting Curlew, Lapwing, Golden Plver, Snipe and Wheatear.

The relative importance of empirical variables used in Habitat Suitability Modelling in Chapter five show that building density was the most important predictor of habitat suitability for Wheatear, Curlew and Snipe, indicating that building density within the SPMSPA fringe should be kept low wherever possible.

There are opportunities to expand on the research undertaken within this thesis. This would help to gain a better understanding of the relationship between the habitats within moorland fringe landscapes and the birds that use these habitats. Some suggestions are outlined below.

- Much of the SPMSPA fringe landscape is composed of agricultural grassland and the use of satellite imagery for categorising agricultural grasslands based on their management intensity is not a well-studied field (Franke et al., 2012). Exploring the classification of moorland fringe habitats using remotely sensed data in more detail would be useful. Alternative algorithms may yield better predictive performance than those achieved within chapter two of this study. Datasets such as Light Detection and Ranging (LIDAR) are increasing in their geographical coverage, which may add predictive capability to the landcover classification within the moorland fringe. Unmanned Aerial Vehicles (UAVs) are becoming widely recognised as useful tools to collect remotely sensed data from remote areas, allowing more data to be accrued to improve classification. If accuracy of classification is achieved, then it may be viable to use such datasets in Habitat Suitability Modelling. In addition, this would allow time-series analysis to be achieved which would provide valuable information on changes in land use configuration over time (Jia et al., 2014). As some remotely sensed products such as Landsat data are freely available, there is the potential for low cost tools to be developed that aid ecologically sound planning decisions for local authorities.

- As some or the priority bird species identified as part of this study were encountered too infrequently to be analysed, it would be useful to study the moorland fringe habitat requirements of these species in upland regions of the UK other the SPMSPA. These species include Short-eared Owl, Common Sandpiper, Twite, Dunlin, Whinchat, Ring Ouzel and Merlin.

- Bird-habitat associations and Habitat Suitability Modelling within this project relied on logistic presence-absence data. Distance sampling is a well-established method for obtaining densities of species, including birds (Buckland et al., 2005). Using distance sampling methods to survey birds within the moorland fringe would provide additional information in the form of species densities which could be used to better understand the relationship between these species and the moorland fringe landscape

- The study of SWTs within this project was limited to 21 turbines over the breeding season of a single year. As evidence is severely lacking for the ecological impact of SWTs, it is recommended that further research be undertaken with a larger sample size and over a longer time period. Ideally this would be undertaken using a Before and After, Control and Impact (BACI) approach.

- The potential for Landsat data to be used in Habitat Suitability Modelling was explored in Chapter five of this thesis. Further research to validate such models is required, as spectral data is difficult to interpret ecologically.

- Much of the conservation protection in the UK uplands is governed by EU (European Union) legislation, including the SPMSPA and other SPAs. Considering the UKs recent decision to withdraw from the EU it is more important than ever for ecologists to collaborate with UK national and local government in gathering and applying scientific ecological evidence to be used in guiding planning policy as well as conservation policy. This means undertaking more research and communicating results and recommendations effectively with policy makers and stake holders.

6.3 References

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APPENDIX

Appendix 1. Detailed descriptions of the habitat categories used for the habitat surveys of 2012 and 2014 in Calderdale and Kirklees (Taken from West Yorkshire Ecology, unpublished).

Habitat type	Habitat description
Amenity grassland	Intensively managed, regularly mown grasslands, typical of lawns, playing fields and golf course fairways (Phase I habitat J1.2).
Improved grassland	Meadows and pastures affected by heavy grazing, drainage, or the application of herbicides, inorganic fertilizers, slurry or high doses of manure, and have lost many sward species. Very limited range of grasses and a few common forbs, mainly those resistant to grazing (Phase I habitat B4). Cover of rye-grasses and white clover >30%. Sward is species-poor (up to 8 species/m2, including grasses). Low cover of wildflowers and sedges (< 10%), excluding white clover, creeping buttercup and injurious weeds (FEP Guidance 008).
Semi-improved, species-poor grassland	Cover of rye-grasses and white clover <30%. Moderately species-rich (more than 8 species/m2, including grasses). Cover of wildflowers and sedges greater than 10% or more, excluding white clover, creeping buttercup and injurious weeds. Fewer than 4 semi-improved wildflower indicators occasional in sward. (FEP Guidance 008). For neutral grassland see: Phase I habitats B6 and B2.2 (below). May also be acidic.
Semi-improved grassland	If neutral: Cover of rye-grasses and white clover <30%. Moderately species-rich (more than 8 species/m2, including grasses). Cover of wildflowers and sedges greater than 10% or more, excluding white clover, creeping buttercup and injurious weeds. Four or more semi-improved wildflower indicators occasional in sward. (FEP Guidance 008). For neutral grassland: Phase I habitat B2.2. Typically NVC communities: MG1, MG6, MG9-10, MG12-13. If acidic: Enclosed and often more species-rich than the acid grassland communities within the moorland line, it is often described as lowland acidic grassland. Phase I habitat B1.2. Typical NVC communities: U4 – Festuca ovina – Agrostis capillaris – Galium saxatile and others. More typical of lowland areas. Not apparently managed intensively for farming (i.e. probably no herbicides; probably no, or very low, fertilizer

	inputs). Typically occurring on upland, hill-grazing land, usually on peaty, acidic soils. Enclosed with fences, walls, hedgerows etc. Generally dominated by Sheep's fescue and Common bent, with a high proportion of herbs, such as Betony, Devil's-bit scabious, Bitter vetch, Harebell, Heath bedstraw, Lady's bedstraw and Mountain pansy.
Species-rich/ Unimproved grassland	If neutral: Cover of rye-grasses and white clover <10%. Species-rich (more than 15 species/m2, including grasses). Cover of wildflowers and sedges greater than 30% or more, excluding white clover, creeping buttercup and injurious weeds. (FEP Guidance 008). For neutral grassland: Phase I habitat B2.1. Typically NVC communities: MG1-5, MG8-10, MG12. If acidic: Upland (unimproved) acid grassland is widespread within the moorland line where it exists as extensive species-poor communities on the open fell or as enclosed rough grazing (rough grassland). Phase I habitat B1.1. Typical NVC communities: U2 – Deschampsia flexuosa; U5 – Nardus stricta – Galium saxatile; U4 – Festuca ovina – Agrostis capillaris – Galium saxatile (more typical of lowland areas). Unpalatable grasses such as mat grass may dominate. Not apparently managed intensively for farming (i.e. probably no herbicides; probably no, or very low, fertilizer inputs). Typically occurring on upland, hill-grazing land, usually on peaty, acidic soils, without fences, walls, hedgerows etc. Where it is more species-rich and enclosed it is often treated as lowland acidic grassland (see below).
Dry dwarf shrub heath	Occurs on well drained shallow peat <0.5m deep. Comprises heather, bell heather, bilberry, western gorse with tormentil and grasses (Phase I habitat D1).
Dry heath/ acid grassland mosaic	Occurs on well drained shallow peat <0.5m deep. Comprises heather, bell heather, bilberry and western gorse. Dominated by acidic grasses (Phase I habitat D5).
Blanket bog/ mire	Occurs on deep peat deposits >0.5m deep. Formed on areas normally over 200m. Numerous pools and raised hummocks, which are formed by Sphagnum mosses. Vegetation comprises heather, cross-leaved heath, cotton-grasses, deer-grass, crowberry, bog asphodel and sedges. Sphagnum mosses are frequent (Phase I habitat E1.6.1).
Wet heathland/ mire	Occurs on lower slopes too dry or steep for deep peat deposits normally under 200m. Peat depth is up to 0.5m. The vegetation comprises of heather, cross-leaved heath, bilberry, deer grass, and purple moor grass with Sphagnum mosses (Phase I habitat D2; Agri-environment Scheme Management Plan – Heather moorland).
Wet heathland/ acidic grassland mosaic	Occurs on lower slopes too dry or steep for deep peat deposits normally under 200m. Peat depth is up to 0.5m. The vegetation comprises of heather, cross-leaved heath, bilberry, deer grass, and purple moor grass with Sphagnum mosses. Molinia caerulea is abundant and domininant. Unpalatable grasses such as mat grass and purple moor-grass and rushes may be present – see Upland acidic grassland/ Grass moor (unenclosed) (Phase I habitat D6; Agri-environment Scheme Management Plan – Heather moorland). Full title used in survey was "wet heathland/ mire/ acidic grassland mosaic (*molinia* dominant)".

Rush pasture	Juncus-dominated (>75%) improved and semi-improved (including species-poor semi-improved) or acidic grassland.
Other	Abandoned (currently disused fields, often with some trees and shrubs), Garden (garden property , usually adjacent to a house), work sites, tilled land, bare ground, arable etc.

Appendix 2. Full descriptions of the habitat categories used for the habitat surveys of 2013 in Bradford, taken from Urban Edge Consulting (2014).

Habitat type	Habitat description
Amenity grassland	Intensively managed and regularly mown grasslands, typical of lawns, playing fields and golf course fairways. Often dominated by perennial rye-grass *Lolium perenne* and white clover *Trifolium repens and containing* forbs such as daisy *Bellis perennis* or broad-leaved plantain *Plantago major* (Phase I habitat J1.2). Generally treated with herbicide and fertiliser and may be disturbed by recreational use making them of little intrinsic botanical interest or value to feeding birds from the South Pennine Moors SAC and SPA.
Improved grassland	Includes meadows and pastures affected by heavy grazing, drainage, or the application of herbicides, inorganic fertilizers or high doses of manure. Contain very limited range of grasses, mainly those resistant to grazing. Perennial rye grass often formed > 90% of the sward with other grasses e.g. crested dogs tail *Cynosurus cristatus* and Yorkshire fog *Holcus lanatus* and ruderal vegetation being present along field edges. Conforms to Phase 1 habitat B4. Falls within MG7 *Lolium perenne* leys and related grasslands of the National Vegetation Classification (NVC; Rodwell ed., 1992). Agriculturally productive grasslands are generally species poor, heavily grazed or frequently mown.
Semi-improved grassland (species poor)	Meadows less affected by fertilizer or herbicide application but still contain a low numbers of species (more than 8 species/m^2, including grasses but less than four wildflower indicators (FEP Guidance 008). Common grasses included perennial rye grass, crested dogs tail, Yorkshire fog and cock's foot *Dactylis glomerata*. The majority of forbs usually comprised ruderals including thistle *Cirsium spp* and broad-leaved dock *Rumex obtusifolius* and creeping buttercup *Ranunculus repens.*
Semi-improved grassland	Meadows less affected by fertilizer or herbicide application and contained a reasonable diversity of species (more than 8 species/m^2, including grasses and more than four wildflower indicators (FEP Guidance 008). Commonly recorded forbs include common knapweed

	Centaurea nigra, pignut *Conopodium majus,* ox-eye daisy *Leucanthemum vulgare* and yellow-rattle *Rhinanthus minor.* Meadow buttercup *Ranunculus acris* also frequent. Corresponds to MG6 Cynosurus cristatus/Lolium perenne grassland) but not as heavily affected by applications of herbicide and fertiliser. Conforms to Phase 1 habitat type B2.2.
Species-rich/ Unimproved grassland	This habitat type was rarely recorded and often existed as remnant habitat within more improved grassland fields on steep inaccessible slopes. Contains low proportion of perennial rye-grass with Yorkshire fog, crested dog's tail, sweet vernal grass *Anthoxanthum odoratum,* red fescue *Festuca rubra* and tufted hair grass *Deschampsia caespitosa* being more abundant. Forbs abundant with >30% cover of wildflowers and sedges (FEP Guidance 008). Most common species include glaucous sedge *Carex flacca,* carnation sedge *Carex panacea,* lousewort *Pedicularis sylvatica,* yellow-rattle, common bird's-foot trefoil *Lotus corniculatus,* common knapweed *Centaurea nigra*, betony *Stachys officinalis* and devil's bit scabious *Succisa pratensis.* Conform to MG5 *Centaurea nigra – Cynosurus cristatus* grasslands.
Rough grassland (enclosed)	Unmanaged grassland often found at field edges, abandoned pastures or road verges, dominated by coarse grass tussocks e.g. cocksfoot, tufted hairgrass, false oatgrass *Arrhenatherum elatius* and Yorkshire fog (corresponds to NVC community MG1). Tall rank swards appear scruffy and unkempt but rich in weed seeds.
Upland acidic grassland (enclosed)	Typically recorded as part of an acid grassland and dwarf shrub heath mosaic. Grasses included wavy hair grass *Deschampsia flexuosa,* mat grass *Nardus stricta,* fescues such as *Festuca ovina* and bents such as *Agrostis capillaris or A. stolonifera.* Heather *Calluna vulgaris* occasionally present as are tormentil *Potentilla erecta,* heath bedstraw *Galium saxatile* and bilberry *Vaccinium myrtillus.* Cotton grasses *Eriophorum spp* also present. Typical NVC communities for this habitat type include U2 – *Deschampsia flexuosa* and U5 – *Nardus stricta – Galium saxatile*.
Dry dwarf shrub heath	Occurs on well drained shallow peat. Comprised of abundant heather, cross-leaved heath *Erica tetralix*, bilberry, and occasional western gorse *Ulex europaeus.* Tormentil, heath bedstraw and grasses. Characteristic acid grassland species sometimes present. Sometimes fragmented and restricted to steep slopes and moorland edges.
Dry dwarf shrub heath / Acid grassland mosaic	Mosaic of upland acidic grassland and dry dwarf shrub heath (see above).
Rush pasture	Typically grassland either semi-improved or improved with >75% coverage of rushes *Juncus spp.* Tend to be sheep grazed, although sometimes grazed by cattle. Occur more frequently towards the moorland fringe. Sward comprises of a combination of semi-improved acid and neutral grassland species interspersed with tussocks of soft rush *Juncus effusus* with frequent marsh thistle *Cirsium palustre.* Sometimes occurs in a mosaic with areas of acid grassland or semi-improved neutral grassland. Distinction between areas dominated by more or less than 75% rushes can be difficult. Occasionally recorded are marsh valerian *Valeriana dioica,* hemlock water dropwort *Oenanthe crocata,*

	square stalked St John's wort *Hypericum tetrapterum* and ragged robin *Lychnis flos-cuculi.* Rush-dominated areas sometimes subject to herbicide application and may be species poor.
Other habitats	Includes abandoned fields, often with some trees and shrubs, gardens, work sites, tilled land, bare ground, arable and woodlands.

Appendix 3. Diversity index equations.

$$D = \sum \left(\frac{n_i(n_i - 1)}{N(N - 1)} \right)$$

$$H' = -\sum p_i \ln p_i$$

$$d = N_{max}/N$$

$$J' = \frac{H'}{H_{\mathrm{max}}} = H'/\ln S$$

$$E_{1/D} = \frac{(1/D)}{S}$$

S = the number of species

n_i = the number of individuals in the *i*th species

N = the total number of individuals

p_i = the proportion of individuals found in the *i*th species (n_i/N)

Appendix 4. Descriptions of each Small Wind Turbine survey site. Of the sixteen turbine sites surveyed, fifteen were individual turbines separated by at least 1km from any other turbine and included all individual turbines within 3km of the SPA. The remaining site comprised of a cluster of eight turbines and was included for comparison to individual turbines.

Turbine site code	Number of turbines	Unitary authority	Corrected location (lat, long, WGS84)
1	1	Kirklees	53.642963, -1.904151
2	1	Calderdale	53.763759, -1.891442
3	1	Calderdale	53.752511, -2.032275
4	1	Calderdale	53.705692, -2.023518
5	1	Bradford	53.803501, -1.897331
6	1	Kirklees	53.631872, -1.893248
7	1	Kirklees	53.629814, -1.926745
8	1	Calderdale	53.718733, -2.074016
9	1	Calderdale	53.750541, -1.92775
10	1	Calderdale	53.745353, -2.05027
11	1	Calderdale	53.73565, -1.975745
12	1	Calderdale	53.650934, -1.95492
13	1	Kirklees	53.645026, -1.932935
14	1	Kirklees	53.602379, -1.956652
15	1	Calderdale	53.622555, -1.917172
16	8	Calderdale	53.741334, -2.125095 (mean centre)

Appendix References

Urban Edge Consulting (2014) *South Pennine Moorland Fringe Habitat Surveys: Supporting data to inform City of Bradford Metropolitan District Council Core Strategy Development Plan Document*.