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Conservation Grazing with Native Irish Cattle in High Nature Value Environments

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Conservation grazing with native Irish cattle in High Nature Value Environments

This thesis is presented to the Institute of Technology, Tralee, in candidature for the degree of Doctor of Philosophy

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Abstract

Marrying conservation objectives with farming practices is a challenging necessity in the current era of anthropogenically-driven biodiversity loss. Upland habitats created by traditional farming are havens for wildlife but face multiple threats, including overgrazing and abandonment. Extensive cattle farming in the uplands has been mooted as a useful conservation measure, but Irish evidence supporting this approach is lacking.

This thesis examines home range, habitat preferences and activity budgets for Dexter cattle in an extensive upland setting in southwest Ireland. It investigates the effects of the grazing regime on EU-protected habitats and ground beetles.

Free-ranging cattle grazed the site over three seasons from 2013-2015. Cattle were tracked by GPS to establish home range and habitat preferences. Direct observation compared activity budgets of upland and lowland herds. Kernel Density Estimation was used to develop a utilisation score. Vegetation and ground beetle sampling examined the impact of the grazing in relation to utilisation.

Mean home range size was 122.7 ha. Wet heath constituted 46% of the home range; blanket bog, 23%; dry heath, 22%; wet grassland, 9%. Cattle showed significant preference for wet grassland and dry heath, and avoidance of blanket bog and wet heath. Activity budgets showed the upland herd spent significantly more time grazing than the lowland herd.

Stocking rates were 0.17 $LU.ha^{-1}$ for the whole study site, 0.12 $LU.ha^{-1}$ in wet heath, 0.20 $LU.ha^{-1}$ in dry heath, 0.14 $LU.ha^{-1}$ in blanket bog, and 0.42 $LU.ha^{-1}$ in wet grassland. At these densities the conservation status of the Annex I habitats was maintained or enhanced. Utilisation had a positive influence on plant species richness and community evenness, however variation between grazed and ungrazed plots was not significant. Beetle species richness did not vary between treatments. The abundance of large wingless ground beetles was consistently depressed across all grazed areas.

Conservation grazing prescriptions should account for availability and spatial distribution of habiat patches, preference of animals, and length of grazing season.

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In memory of Marlene,

who danced me to the end of love.

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Declaration

I declare that this thesis is my own work and I have not obtained a degree in this Institute, or anywhere else, based on this work.

1 Chapter 1: Introduction

Dexter cattle, Mt Brandon Nature Reserve county Kerry. Photo K.Kelly.

1.1 Background

Mountains, People and Biodiversity

Mountains are perhaps Earth's most arresting features and are associated with high levels of biodiversity (Antonelli et al., 2018; Perrigo, Hoorn and Antonelli, 2020). They cover about 25% of the world's terrestrial surface, and approximately 1.48 billion people live in or adjacent to mountainous regions (Edwards, 2005). More than half of the world's population is dependent on mountains for the resources and services they provide (Edwards, 2005). Both directly or indirectly, mountain ecosystems provide water, timber, food, medicines, pasture for livestock grazing and minerals for extraction. They also provide many non-material benefits, such as 'cultural services' e.g. diversity of culture and knowledge systems, 'regulating services' e.g. air quality and climate regulation, and 'supporting services' e.g. pollination, soil retention and nutrient cycling (Edwards, 2005).

Mountains are core areas for biodiversity and often contain organisms that are not found elsewhere. Factors leading to high levels of biodiversity include: habitat isolation and fragmentation, contrasting environmental conditions at different altitudes, compression of climatic zones over short distances, exposure, and variation in slope and aspect (Spehn, Rudmann-Maurer and Korner, 2010). Disturbances further increase habitat diversity in mountains and they occur on multiple scales, from avalanches and landslides to grazing and digging by animals. Mountainous or 'upland' regions often act as refugia for organisms that are either impacted by changes in climate or disturbances in adjacent lowlands

(Spehn and Körner, 2005). Furthermore, community-level diversity is often high on small spatial scales at altitude, due to the relatively small size of species that occur there (Spehn, Rudmann-Maurer and Korner, 2010).

The importance of mountains for biodiversity is well recognised, but defining 'upland' and 'mountain' is often challenging and has long been a source of controversy (Körner et al., 2017). Internationally, the Global Mountain Biodiversity Assessment (GMBA) uses 'ruggedness' to delineate mountain areas, where ruggedness is defined as the "*maximal difference of at least 200m in elevation among nine neighbouring grid points on the 30" grid of the WORLDCLIM database*" (Körner, Paulsen and Spehn, 2010). Using this definition, 12.3% of the terrestrial surface of the planet can be considered mountainous terrain.

No statutory definition of the uplands exists in Britain and Ireland. In Britain, they are described as being above limits of enclosed farmland and above 200 m by Ratcliffe (1977). In Ireland, many use the 150 m contour line to delineate the uplands e.g. Perrin et al., (2014) and Fossitt (2000). Considering those demarcations, the uplands cover almost a third of the land surface of both Britain and Ireland (Fielding and Haworth, 1999; Perrin et al., 2011a).

The Irish Uplands: Landform and Geology

Ireland is approximately 84,000 km^2 in size (Nolan, 2009). The majority can be considered lowland, with three-quarters under 150 m and only five per cent over 300 m (Aalen, 1997). The uplands are mainly found around the coastal fringes of the island, with the flat Central Lowlands in the middle, underlain with the largest continuous piece of carboniferous limestone in Europe (Aalen, 1997). Karst limestone covers over 40% of Ireland, including the main agricultural and populated areas (Drew, 2018).

The three major upland regions of Ireland are: ‘the north and west’ which take in much of Connaught and northwest Ulster; the south-eastern uplands that stretch from south Dublin to county Wexford; and the uplands of Munster spanning Waterford, Cork and Kerry. A fourth upland area occurs in the north-east of Ireland which is split into two areas: the igneous mountains of Mourne and the Carlingford peninsula, and the basaltic plateaus of county Antrim (Aalen, 1997).

The uplands of the north and west of Ireland, along with those of Leinster, were influenced by the Caledonian folding event and have a general north-east to south-west orientation. The uplands of Munster were formed during the Armorican building phases and the ridges and valleys of the region have a distinctive east-west orientation. The rock types of Ireland are generally very old, with the contemporary variations in relief being largely due to recurring periods of glaciation and differential resistance of rock types, rather than tectonic forces. The dominant glacial action in the lowlands was the deposition of drift sheets, whilst erosion has stripped the uplands of soil and left dramatic landscape features, such as corries, sharp ridges and deep valleys (Aalen, 1997).

The Caledonian folding events are responsible for the uplands of the north and west of Ireland, which are underlain with pre-Cambrian rocks (Sleeman, McConnell and Gatley, 2004). The upland areas are dominated by granites and metamorphic rock, and quartzite peaks form distinctive mountaintops in the region such as Croagh Patrick ('the Reek') in Mayo and Mt Errigal in Donegal. This folding structured much of north-central and north-western Europe, Scandinavia and Greenland, and is also responsible for the uplands of the east and south-east of Ireland. The mountains are formed from a granite mass which was exposed from its covering of Ordovician strata during recurring periods of glaciation. This, the 'Leinster Chain', forms the most extensive tract of mountainous land in the country, stretching from Dublin Bay to Barrow harbour in county Wexford (Aalen, 1997).

The uplands of Munster in the south and south west of Ireland were originally formed during the Armorican mountain building events around 250 m years ago. The ridges and valleys display a distinctive east-west trend, resulting from compression from the south. The ridges are Old Red Sandstone and the valleys are carboniferous limestone. At the eastern end of the range in county Waterford and southern Cork, extensive level surfaces are found, although peaks of over 600 m occur in the Comeraghs in Waterford. In west Cork and Kerry, the mountains become broader and higher and it is here that Ireland's highest peak is found: Corrán Tuathail (Carrauntoohil) at 1,038 m (Aalen, 1997).

The Irish Uplands: Climate, soil and vegetation

Ireland has an oceanic climate that is dominated by moist westerly airflows, frequent rainfall and small temperature ranges. The uplands are characterised by high rainfall and humidity, high windspeeds and near-continual ground wetness with low evaporation rates (Hodd, 2012). The oceanic climate of Ireland means that it does not have extremes in temperature, but it does have high altitudinal temperature lapse rates and high cloudiness, with frequent rainfall and humidity (Ratcliffe, 1968; Grace and Unsworth, 1988).

Ireland's mountainous western margin is subject to almost constant high humidity as it is the first landfall for Atlantic weather systems (Sweeney, 1997). Temperatures in Ireland are high for its latitude due to the influence of the Gulf Stream and temperatures do not rise as high as continental regions of similar latitude in summer, due to the cooling effects of the ocean. The topography of Ireland is such that the uplands are mainly coastal and there is high climatological contrast between the maritime margins and the interior of the country which is relatively continental (Sweeney, 1997). The eastern lowlands are the driest part of Ireland. Along with its proximity to Britain, this meant it became the effective centre of the country for cultural and economic development (Aalen, 1997).

Ireland's soils are a function of parent material, climatic conditions and local variations in relief. Precipitation-evaporation rates favour precipitation and leaching is the dominant soil process. This has led to widespread podzolisation, hard-pan development and acidic soils. Podzols are prevalent in the west and in the uplands, and peaty soils are characteristic of Ireland's mountains (Aalen, 1997). Soil and climatic influences on vegetation patterns in the Irish uplands are discussed further in Chapter 3.

The Irish Uplands: Cultural influences on the land

Ireland's uplands combine a varied ecology with a long history of human settlement and they share these parallels with Atlantic Europe. Pastoralism, dispersed settlements and infield-outfield cultivation have dominated historically. These features are shared with the western edge of Europe from the Iberian Peninsula, north to Britany, Cornwall, Wales, the Scottish Highlands and Norway (Aalen, 1997). Mesolithic hunter-gatherers settled in Ireland around 9,000 years ago and they largely confined themselves to the coasts, rivers and lakes, with no significant impact on the woodland ecosystems (Hall, 2011). Farming developed in Ireland around 6,000 years ago, influenced by immigrant Neolithic farmers. These early farmers spread throughout the country but only had limited tools, so favoured upland fringes and lighter soils where woodlands were thinner and soils free-draining (Aalen, 1997).

The Bronze and Iron Ages brought about technological advancements in agriculture and the heavier fertile soils of the central Irish lowlands were well-settled by the 6th century. The uplands were abandoned in favour of more productive ground in the lowlands and the woodlands that had been previously cleared for agriculture did not fully recover. This, in combination with climatic shifts, resulted in the development of extensive areas of heath and bog that were used by transhumance herders (Aalen, 1997). Thus, the regeneration of woodland was continually checked by grazing and burning, a feature that persists in modern Ireland. Throughout the Bronze and Iron age sequential waves of deforestation and re-vegetation occurred, driven by both climatic factors and human-mediated interference. However, by the Early Christian period (500-1000 AD) woodland clearance was extensive and permanent (Aalen, 1997; Hall, 2011).

The Irish human population grew rapidly after 1600, building from 1.5 million to 8.5 million by 1840, bringing settlement expansion into the uplands and intensification of land use. However, the Great Famine (1845 to 1849) and further population declines over the last 150 years, released population pressure on the land and led to abandonment of farms, particularly in the uplands. Legislation introduced in the 1800s began the process of dismantling the large estates of the earlier plantation period (seventeenth century) and tenants became owners of their holdings. By 1921, sixty per cent of people who were previously tenants, now owned their own land, and his converted Ireland into a nation of small farms (Aalen, 1997).

The Uplands: Contemporary patterns and threats

Three basic vegetation types have dominated Ireland since historic times: improved grassland on fertile lowland soils, moorland and bog on the uplands, and bog which can form on both the lowlands and uplands (Aalen, 1997). Woodlands have only survived where land was unsuitable for agriculture or was protected in an estate. Irish woodlands are typically small and isolated remnants of oak woodland, the potential climax community in Ireland (Cross, 2006). The uplands of Ireland have remained open, treeless expanses of peatland and heath until recent times, which contrasts with comparable upland regions in Europe that have remained wooded over the similar timescales e.g. Pyrenees and

Carpathians (Aalen, 1997). State supported forestry programmes through the latter half of the twentieth century transformed much of the Irish uplands, with rectilinear conifer plantations now dominant in some areas.

Transhumance or 'booleying' was a feature of the Irish rural economy throughout history, with grazing animals being moved to upland pastures during the summer months. Booleying ensured that the open character of the Irish uplands remained because it required extensive tracts of rough grazing. The practice was an important feature of agriculture up to the seventeenth century but thereafter declined, only remaining in remote areas through to the nineteenth and twentieth centuries (Aalen, 1997).

The twentieth century brought about fundamental changes in how the uplands are farmed and managed. The formation of the European Union and Ireland's enthusiastic involvement in its agricultural programmes has facilitated a move away from cattle grazing in favour of sheep in mountainous areas (O'Rourke et al., 2012). Subsidies paid to farmers under the Common Agricultural Policy (CAP) during the 1970s and 80s resulted in sheep becoming the dominant grazing animal in the Irish uplands. Concurrently, commercial forestry expanded because of State supported programmes and much of the Irish uplands, if not grazed by sheep, are now covered by non-native coniferous trees. In recent decades, wind-farm developments have become a feature of the Irish uplands. There are currently over 346 windfarms in operation on the island of Ireland, many of which are in upland areas (Irish Wind Energy Association, 2018). Counties Cork, Kerry and Donegal are the biggest producers, reflecting the western and upland trend in windfarm distribution.

Ireland's uplands are areas of exceptional natural beauty and remain openly accessible to recreational users despite being predominately in private ownership. Hill walking is the dominant recreational activity in the Irish uplands, bringing pressures of erosion, trampling and damage to vegetation. In 2017 over 2.1 million overseas visitors came to Ireland specifically for hiking (Fáilte Ireland, 2017), potentially generating €1.25 billion for the Irish economy (O'Dwyer, 2018). The Wicklow uplands probably experience the highest visitor numbers, being near the population centre of Dublin, followed closely by the MacGillycuddy's Reeks of county Kerry. The immediate area surrounding Carrantuohill in county Kerry (Ireland's highest peak) receives 125,000 visitors annually (P. Deane 2018, *pers.comm.* August 29th). The recent success of the Wild Atlantic Way marketing strategy for the west of Ireland has seen visitor numbers reach 10 million annually, further boosted by the success of recent TV and film productions (e.g. Game of Thrones and Star Wars) (Pollak, 2017). Continued growth in the sector has the potential to exert pressure on Ireland's uplands.

Farming in Ireland today

The Teagasc National Farm Survey (NFS) provides a snapshot of farming in Ireland each year and the results of the 2017 survey (Dillon et al., 2017) are summarised here. Approximately 4.4 m hectares of Ireland is taken up with agriculture. There are 139,600 farms in Ireland, with an average farm size of 45 ha. Grass, hay and silage take up 81% of agricultural land, with rough grazing taking up 11% and crops and fruit/horticulture making up the rest (approximately 8% each). The average family farm income in 2017 was €31,412, an increase of 32% on 2016. This increase can be attributed (and largely confined) to the considerable income increases across the dairy sector due to growths in milk prices (Dillon et al., 2017).

Farm incomes vary considerably between and among farming sectors in Ireland, with the dairy sector having the largest incomes (€86,069) and cattle rearing the smallest (€12,529). The cattle and sheep rearing sectors are characterised by lower profitability and smaller holdings. The average farm size for all sectors is 45 ha, with sheep farm sizes of 51 ha and cattle farms sizes of 35 ha (Dillon et al., 2017). Farm incomes are heavily reliant on direct payments in all sectors, but proportionately contribute the most to farm incomes in the cattle rearing sectors (114%). Geographically, farm incomes decline from south-east (~€40,000) to northwest (~€15,000), a trend that reflects both the type of farming and farm size. Tillage and dairy dominate in the east and south, whilst drystock farming is typical of the west and northwest (Dillon et al., 2017).

Farming in the Irish uplands

Approximately 1.1 million hectares of the land mass of Ireland can be considered 'upland' (Bleasdale and O'Donoghue, 2015), and extensive grazing is the dominant land use. The uplands are physically and economically remote and present considerable climatic challenges. Farm size diminishes towards the west and northwest of Ireland, corresponding to declining agricultural land quality (Emerson and Gillmor, 1999). A study on farming on the Iveragh peninsula in south west Ireland was conducted by O'Rourke et al., (2012). It found that the average farm size was 138 ha and the average age of farmers was 49 years. Seventy per cent of farms had shares in a commonage and those shares make up about 32% of the area farmed. Farms consisted of 59% upland, 20% improved grassland and 21% of rough land. Eighty six per cent of the farms were participating in the Rural Environmental Protection Scheme (REPS) and 52% of all holdings were designated as Special Areas of Conservation (SACs), *sensu* EU Habitats Directive of 1992 (O'Rourke et al., 2012).

Population structures in the uplands typically have high youth and elderly populations and a lower demographic vitality (Crowley, O'Keeffe and O'Sullivan, 2016). Consequently, communities experience outmigration of young adults, and thereby a loss of economic and reproductive potential. The remaining populations tend to contain a high proportion of early school leavers and fewer people with a third level education, especially among males. The female labour force has lower participation and employment rates compared with the rest of the Irish State, and males have higher unemployment rates (Crowley, O'Keeffe and O'Sullivan, 2016). Upland areas seem to remain attractive areas to raise a family, despite the challenges of health and childcare access, with 75% of upland households comprising of families with children which is 4% higher than the State. The physical constraints and natural resources in upland areas can restrict farming activities and thereby farm incomes and competitiveness, issues that threaten economic viability into the future (Crowley, O'Keeffe and O'Sullivan, 2016).

The threat to economic viability of upland areas has serious consequences for the conservation of protected upland habitats because they have developed with and because of farming practices. The drain of labour from hill farms resulting in under and over-grazing is an issue faced by HNV farmland across Europe (Costello, 2020).

The Uplands, Biodiversity and Conservation

The uplands form Ireland's greatest expanses of semi-natural habitats (Perrin et al., 2011a) and over 40% of the Natura2000 Network occurs in the uplands. These uplands are important areas for conservation and contain up to 14 habitat types listed in Annex I of the Habitats Directive (Perrin et al., 2011a). Annex I habitats are of EU community importance and Ireland is obliged to maintain them in 'favourable conservation status'. Strict criteria for achieving this status are outlined in the EU Habitats Directive (European Commission, 1992), detailed under the headings of: Area, Range, Structure and Function, and Future Prospects (Perrin et al., 2011a). Conservation status monitoring is discussed in the context of this thesis in Chapter 3 (impact of cattle grazing on upland vegetation).

National and European legislation requires certain habitats to be managed so that their conservation status is maintained or enhanced (i.e. Wildlife Acts of 1976 & 2000, EU Habitats Directive (1992), National Biodiversity Action Plans 2002 & 2011). However, even within sites where the primary objective is nature conservation, the appropriate management required to achieve conservation objectives is often lacking.

The latter half of the twentieth century has brought about increasingly widespread degradation of Irish uplands (Heritage Council, 1999; Perrin et al., 2011b). Reforms have attempted to redress the grazing imbalances in the uplands, much of which are deemed to be High Nature Value (HNV) farmland i.e. farming systems where traditional practices have maintained high levels of biodiversity (Beaufoy, 2008). Attempts have included agri-environment schemes such as the Rural Environmental Protection Scheme (REPS) and the Commonage Framework Plan (CFP), however, results have been ambiguous to date (Kleijn and Sutherland, 2003b; Whittingham, 2007; Finn and Ó hUallacháin, 2012a) and the status of the majority of EU protected upland habitats in Ireland is currently assessed to be in 'poor' or 'bad' condition (NPWS, 2019c).

Despite national and international legislation in place for conservation, it is now apparent that minimising adverse impacts on biodiversity is not enough. The EU Biodiversity Strategy for 2020 (European Commission, 2011) recognises that active conservation management through HNV farming plays a vital role in achieving Europe's biodiversity objectives. Therefore, there is an urgent need for a sustainable land management policy for HNV farming in the uplands, particularly regarding appropriate cattle management.

This Project

This work is part of an ongoing long-term research collaboration between the Institute of Technology, Tralee (ITT), the National Parks and Wildlife Service (NPWS) and Mr Paddy Fenton, a farmer in west Kerry and co-owner of The Dingle Dexter Beef Company.

Work by Williams et al. (2012) examined grazing and habitat selection in the context of sheep. However, to date there have been no reports on extensive upland HNV studies of cattle in an Irish context outside of the Burren Programme; a pioneering agri-environmental programme in Co Clare (detailed in section 1.1 – 'Policy and current agri-environment schemes'). An in-depth study of an extensive cattle-based upland grazing regime is therefore necessary and timely, since current literature is deficient on reports of such grazing regimes in the Irish uplands. Available prescriptions for peatland grazing are built on assumptions made from research done in neighbouring countries but with differing environmental conditions and traditions (e.g Bokdam and Gleichman, 2000; Holland et al., 2010; English Nature, 2005). This project is ideally placed to link Irish research to emerging Irish policy on best-practice approaches to managing HNV farmland in the uplands, and to Irish obligations under the Habitats Directive.

In 2011 Institute of Technology, Tralee (ITT), in collaboration with the NPWS, initiated a baseline study of the vegetation and macroinvertebrates associated with upland habitats in Mount Brandon Nature Reserve (MBNR) (Figure 1) in advance of the commencement of experimental grazing trials with Dexter cattle. The site has a history of heavy sheep grazing, which had been eliminated in recent years and land managers are now seeking to establish an appropriate grazing regime which will maintain and enhance the status of habitats on the site.

The research seeks to investigate the use of an extensive cattle grazing regime in a site of high conservation value. Identifying how free-ranging cattle behave in a heterogeneous upland setting, and assessing the impact they have on habitats and invertebrates, is the focus of this research. From this, identifying appropriate grazing regimes that facilitate the achievement of favourable conservation status for priority upland habitats is a key outcome. By providing evidence-based specific management recommendations for upland areas, the research intentions are to assist policy-makers in meeting requirements under national and international legalisation.

The research investigated the behaviour of Dexter cattle in an extensive upland setting using radio-telemetry. Floral community structure and composition were assessed, and macro invertebrate response were examined to consider the impact of grazing on biodiversity in upland habitats.

Figure 1 Map of Co Kerry, Mt Brandon cSAC and Mt Brandon Nature Reserve

Thesis Outline

- **Chapter 1**: The remainder of Chapter 1 consists of: (**1**) a review of Irish agri-environment policy and schemes and the role of cattle in them; (**2**) an overview of biodiversity conservation, conservation policy in Ireland and conservation grazing *per se;* and (**3**) and lastly, an examination of the historical and contemporary aspects of cattle in the Irish uplands is presented.
- **Chapter 2** examines the home range, habitat selection and behaviour of cattle in Mount Brandon Nature Reserve in Co Kerry.
- **Chapter 3** explores the impact of the grazing regime on Annex I upland habitats.
- **Chapter 4** presents a preliminary examination of carabid beetle communities in the Reserve and explores factors influencing them with focus on the grazing treatment.
- **Chapter 5** offers final conclusions and recommendations from the research.

1.1 Policy and current agri-environmental schemes

The Common Agricultural Policy

The Common Agricultural Policy (CAP), launched in 1962, is the agricultural policy of the European Union. CAP undertakes actions for agriculture in three areas: income supports *via* direct payments (under 'pillar one'); market measures for intervening in times of instability; and rural development measures ('pillar two') which address specific challenges in rural areas under Rural Development Programmes (RDPs). The 2018 CAP budget was €58.82 billion, financing income support (€41.74 bn), rural development (€14.37 bn) and market measures (€2.7bn) (European Commission, 2018).

The EU Rural Development Policy ('pillar two' of CAP), has been in effect since 2000 and aims to promote the economic, social and environmental development of the countryside. It is funded by the European Agricultural Fund for Rural Development (EAFRD), by national public budgets in member states, and in some cases by additional private funding from beneficiaries (European Commission, 2018).

CAP has undergone substantial reform since its inception, progressively moving away from being the wholly production and food security-oriented policy it was since the Treaty of Rome. Since 1992, CAP reforms have aimed to reduce the pressure of agriculture on the environment (European Commission, 2017). Fundamental CAP reform took place in 2003, when subsidies were decoupled from production and Single Farm Payments (SFP) were introduced under the

Single Payment Scheme (SPS). These payments are subject to cross-compliance conditions relating to environmental, food safety and animal welfare standards.

CAP beyond 2020

The Common Agricultural Policy (CAP) has evolved over many years, and in 2018 the European Commission presented legislative proposals for CAP beyond 2020. The budget for CAP comes from the EU's Multi-annual Financial Framework 2021-2027. In May 2020 the MFF was supplemented by the European Recovery Instrument (ERI), called 'Next Generation EU', due to the coronavirus pandemic. Lengthy negotiations led to a final CAP budget of 343.9 bn (a reduction of approximately 5% on the previous MFF).

Agri-environment schemes

Agri-environment schemes (AES) have been one of the key policy tools introduced to assist farmers in mitigating against the environmental impact of agriculture on the environment. AES are developed under each member state's Rural Development Programme (RDP) and are mandatory for administrations but voluntary for farmers (European Commission, 2017). Farmers in receipt of SFP must meet cross-compliance standards and may then opt for AES measures above and mandatory compliance elements. Agri-environment measures are wide ranging and include among others: options under organic farming, reduction of fertiliser inputs, improving habitats for wildlife, buffer strips to protect water courses, management of grazing pressure to prevent soil erosion, and conservation of genetic resources. Approximately 25% of agricultural land in the

EU is under AES measures and up to €23 billion was spent on AES from 2007-2013 (European Commission, 2017).

In Ireland, efforts to formulate agri-environment policy prior to the 1990s were limited and largely concerned with measures to control water pollution from silage and organic wastes. The Farm Modernisation Scheme (1970s), the Farm Improvement Programme (1980s) and later the Control of Farm Pollution Scheme (1994), focused on the provision of grant aid for buildings and storage facilities (Emerson and Gillmor, 1999). The application of the principle of Environmentally Sensitive Areas (ESAs), the first EU agri-environment instrument to provide a mechanism for compensating farmers for prioritising environmental considerations, was slow in Ireland. In 1991 the ESA scheme was piloted in the Slieve Blooms (midlands) and Slyne Head (west coast), but strict conditions for compliance and small budgets resulted in low uptake (Emerson and Gillmor, 1999). The introduction of the Rural Environmental Protection Scheme (REPS) in 1994 represented the first major transformation in Irish agri-environmental policy.

Rural Environmental Protection Scheme (REPS)

The first REPS programme (REPS1) included 11 basic measures that could be undertaken for a five-year period, with farmers receiving an annual payment of €151 per hectare per annum to a maximum of 40 ha. Additional funding was provided for supplementary measures, two of which were obligatory for land falling under designations of Natural Heritage Area (NHA) and Rejuvenation of

Degraded Area. Seventy percent of participants were mostly less intensive cattle and sheep farmers of the west of Ireland (Image, 2016).

REPS2, introduced in 2000 under the Rural Development Programme (2000-2006), succeeded REPS1 and included the basic same measures with the condition that plans would go beyond 'good farming practice'. Measures targeting protected areas (actual or proposed Natural Heritage Areas (NHAs) and Natura2000 sites) were also consolidated under 'Measure A', and smaller farms were given a 10% increase in payment. REPS3 (2004) again retained the same basic 11 measures but farmers were obliged to take on two additional measures: Measure A became mandatory, and farmers were offered a suite of additional measures to choose from as a second e.g. Traditional Irish Orchards and Specific Action for Corncrake. REPS4 (2007) was introduced under the next RDP period (2007 – 2013) and the 11 basic measures were retained (Emerson and Gillmor, 1999; Image, 2016). REPS closed to applicants in 2009 and by then had paid out €3.1 billion since 1994 and had included 45% of Irish farms (Finn and Ó hUallacháin, 2012a).

Agri-Environment Options Scheme (AEOS)

REPS was followed by the less well known Agri-Environment Options Scheme (AEOS) in 2010, which had a much smaller budget of €146.3 billion, reflecting the economic conditions during the financial crisis. AEOS was designed with a more targeted approach than REPS, with actions being carried out on parts of a farm rather than the whole-farm approach. Farmers could choose from a range of measures, which included among others: development of field margins, tree planting, species rich grassland, wild bird cover and stone wall repair (Image, 2016).

From Action-based to Results-based and Blended AE Models

The high levels of uptake into REPS (25% of all farmers within 5 years of commencement) demonstrated its success in engaging farmers with environmental objectives and in steering the industry away from being solely "*productivist*" (Emerson and Gillmor, 1999). However, research on the effectiveness of agri-environment schemes during its lifetime suggests that such broad-based schemes results are not effective in conserving biodiversity (Kleijn and Sutherland, 2003a; Kleijn et al., 2006; Finn and Ó hUallacháin, 2012a). This, coupled with the implementation of additional actions and designations resulting from European Court of Justice (ECJ) rulings against Ireland, led to the emergence of new forms of agri-environment schemes in Ireland which accessed funds outside the scope of the RDP (Image, 2016).

The Burren Farming for Conservation Programme (BFCP) in Co Clare began as a €2.5 m research programme funded under the EU LIFE Nature fund. Farmers in the Burren region, frustrated with restrictions imposed by Natura2000 designations and challenging REPS measures, initiated research with Teagasc and the NPWS (Dunford, 2015). The initial 'BurrenLIFE' project was a collaboration between the Burren Irish Farmers Association, the NPWS and Teagasc and it involved designing, testing (on 20 farms) and developing a sustainable farming programme for the Burren area between 2005-2010 (Dunford, 2015). The resultant design was rolled out to 160 farms from 2010-2015, jointly funded by the Department of Agriculture, Food and the Marine (DAFM) and the NPWS. It adopted a hybrid approach, whereby farmers were paid for actions as well as their environmental performance (Dunford, 2015). The farmer-driven, results-based, locally-based BFCP received widespread support and became the blueprint on which subsequent results-based agri-environment schemes have been designed. The BFCP, now 'the Burren Programme', is funded under the RDP 2014-2020 as a standalone locally led agri-environmental scheme and will receive €15m in funding over the duration of the current RDP (Bleasdale, 2018).

Early agri-environmental schemes in Ireland have taken a "*one-size-fits all approach*" (McGurn and Moran, 2013) and evaluations have shown that these programmes often fail to achieve desired results (Kleijn and Sutherland, 2003a; Kleijn et al., 2006; Hodge and Reader, 2010; Finn and Ó hUallacháin, 2012b). 'Outcome/results-based payment' schemes are based on delivery of environmental/ecological goods and services, whereas 'Prescription/action-based payments' are based on conducting land management practices that are expected to deliver. The latter are the most dominant in EU member states (McGurn and Moran, 2013) and constitute the provision of non-targeted horizontal payments to farmers (Burton and Schwarz, 2013).

A national outcomes-based approach for Ireland was first developed and proposed for the RDP 2014-2020 by McGurn and Moran (2013). The 'RBAPS Project' (Developing Results Based Agri-Environment Payment Schemes in Ireland and Spain, available at (https://rbaps.eu/) further developed and tested the approach in Ireland and Spain. The RBAPS approach is simple in concept:

habitat condition is scored (e.g. on a scale of 1-10), with the highest payments being awarded to the best quality habitat. Assessments are habitat-specific and based on indicators (e.g. indicator species, habitat mosaics etc.) developed from specific biodiversity targets (e.g. species rich grasslands, hedgerow networks, breeding waders). Model development and design of habitat-specific scoring cards is further described by Byrne (2018).

The RBAPS approach is not without risks e.g. risks to farmers in relation to factors outside their control (climate, behaviour of neighbouring farmers/partners etc.), risks to farmers that 'over supply' at their own cost where thresholds are in place for payments, and also the potential risk of opening up farmers to greater public scrutiny because they are now 'selling public goods' (Burton and Schwarz, 2013). Further barriers and risks to the RBAPS approach include a reduced budget for Pillar 2 after 2021, resistance and lack of ambition at political level, and no current consequences for not delivering results (Bleasdale, 2018). However, the RBAPS system, with its origins in the Burren Programme, is now being implemented into Locally Led schemes and European Innovation Partnership (EIP) farming for conservation programmes in Ireland (e.g. the Hen Harrier Project and the BRIDE Farming for Nature Project). Most agri-environmental schemes in Ireland now adapt a results-based or a blended approach.

Green Low-Carbon Agri-environment Scheme (GLAS)

The RDP 2014-2020 introduced the Green Low-Carbon Agri-environment Scheme and it has interlinked goals under biodiversity, climate change and water quality. GLAS has core requirements for entry, which include preparation of an application by an approved GLAS planner, nutrient management plans and knowledge transfer actions. It focuses on being outcomes-based and has a tiered structure for farmers with 'Priority Environmental Assets' who conduct 'Priority Environmental Actions' (Image, 2016).

GLAS is a three-tier scheme as follows (adapted from (DAFM, 2015b)):

Tier 1 (a): is the highest priority access to the scheme for landowners with 'Priority Environmental Assets', such as Natura sites, High Status Water Areas, Farmland birds (breeding waders, corncrake, twite, chough, geese/swans, grey partridge and hen harrier), Commonages and Rare Breeds.

Tier 1 (b): is for farmers that engage with Priority Actions, which include minimum tillage, catch crops establishment from a sown crop, low emission slurry spreading and wild bird cover.

Tier 2: is the second level of entry and is for farmers that do not have Priority Assets but whose lands include Vulnerable Water Area, or in the absence of a Vulnerable Water Area applicants may still qualify if one of the actions of Tier 1(b) is chosen. Unlike Tier 1, access to the scheme is not guaranteed.

Tier 3: is for general actions and can be chosen in addition to Tier 1 and 2 options or on their own but choosing only Tier 3 actions does not ensure access to the scheme. There are 20 actions under tier 3 and they include: wild bird cover, bird boxes, catch crops, laying hedgerows and protection of archaeological sites.

An annual payment of €5,000 is available under GLAS (or €7,000 for GLAS+, where farmers undertake particularly challenging actions) and the scheme expected to attract up to 50,000 farmers.

Rare breeds are considered under Tier 1 of GLAS with payments of €200/LU/Year available, to a maximum of 10 LUs. The objective of the action is to "*retain and where possible increase populations of specific rare breeds to ensure long term survival of the breeds.*" (DAFM, 2016) and entry is only available to farmers that have registered with the relevant breed society. Additionally, the list of further actions that can be undertaken on the same LIPIS parcel (Land Parcel Identification System) is restricted.

Table 1 Eligible livestock breeds

Cattle	Horses and Ponies	Sheep
Kerry	Connemara Pony	Galway
Dexter	Irish Draught	
Irish Maol	Kerry Bog Pony	

Livestock Units are calculated as follows and apply if an animal is kept for one year: bovines of 6 months to 2 years = 0.6 LU, bovines over 2 years = 1 LU, equines over 6 months = 1 LU, ewe (+/- lamb at foot) = 0.15 LU, ewe lamb 6 months – 1 year = 0.10 LU and Ram = 0.15 LU.

NPWS Farm Plan Scheme

The NPWS Farm Plan Scheme (FPS) was open to applicants from 2006 to 2010 and applied a targeted approach to assisting farmers with designated areas to manage land for conservation. Farm plans under the scheme were bespoke, with payments made for costs incurred and income forgone. The programme has paid out €25 million across 685 plans (Bleasedale and O'Donoghue, 2015) and many of the actions utilised under the scheme have been incorporated into Pillar II funding mechanisms (GLAS/Locally Led schemes). The FPS continues on a small scale and focuses on trialling novel methods (Image, 2016).

Locally Led agri-environment schemes (LLAES)

Locally led AES are designed to develop solutions to address environmental and biodiversity challenges at local level. The Burren Programme created the blueprint for contemporary locally led, results-based agri-environment schemes and has become the flagship farming for conservation programme in Europe. It has now been subsumed into the RDP and is funded as a standalone LLAES under "*non-productive investments linked to the achievement of agri-environment-climate objectives*" (DAFM, 2018b).

Ireland's RDP 2014-2020 further reflects the move away from action-based AES to results-based and hybrid schemes e.g. the Hen Harrier Programme (results-based) and e.g. GLAS (hybrid). Under the RDP legal mechanisms (specified in DAFM, 2018, p.311) and funding measures have been provided to support LLAESs under the following measures:

1. European Innovation Partnerships (EIP) Operational Groups – General EIPs (budget = €4 m)
2. EIP Operational Groups – Locally Led Hen Harrier and Freshwater Pearl Mussel Projects (Budget = €34 m)
3. EIP – Locally Led Environmental and Climate Projects (Budget = €20 m)
4. Collaborative Farming.

The 'General EIPs' (no.1 above) aim to support projects generating innovative solutions to target issues of farm viability, modernisation, economic performance, food chain organisation and creation of local supply circuits. The EIP Operational Group Projects (Hen Harrier and Freshwater Pearl Mussel projects) cover two national priorities and were awarded to two separate operational groups following a competitive selection process (DAFM, 2018b).

The 'EIP Locally Led Environment and Climate Project' measures are expected to support projects that contribute towards (among others) biodiversity, HNV farming and landscapes, water management, soil erosion, increasing energy efficiency and reducing GHG emissions from agriculture (DAFM, 2018b). In 2018 the DAFM awarded 12 projects with funding under this EIP call, three of which specifically target biodiversity and HNV farming in the uplands, these being: The Sustainable Uplands Agri-environment Scheme ('SUAS') project in County Wicklow, the Blackstairs Farming Futures project in (Carlow/Wexford), and the MacGillycuddy's Reeks project (county Kerry). These projects are led locally by operational groups in each region. Operational groups are made up of farmers, plus representatives from local authorities, academic institutes, NPWS and recreational groups. The 'collaborative farming' sub-measure will focus on (1) economic performance and modernisation and (2) entry into the agricultural sector and generational renewal (DAFM, 2018b).

Agri-environment schemes and the uplands

REPS/AEOS did not target upland areas directly but there is evidence that these schemes were favoured in the uplands, particularly in the west and northwest of Ireland. The uplands constitute an area of over 1 million hectares, 35% of which is designated land, 76% is under commonage (Bleasdale and O'Donoghue, 2015) and much of which are found in the west and northwest of Ireland. The level of adoption of REPS was greatest in the west and north west of Ireland, where farm sizes are small (40 ha) and drystock farming of cattle and sheep dominates (Emerson and Gillmor, 1999), with approximately 75% of participants being either cattle or sheep rearing systems (Finn and Ó hUallacháin, 2012a). REPS payment structures were such that the larger the farm, the less financial incentive there was: area-based REPS payments were capped at 40 ha. Furthermore, farms in excess of 40 ha were more likely to be full-time enterprises and run on an intensive basis, thus less likely to qualify for AES (Emerson and Gillmor, 1999). Thus for farmers in the west and northwest, meeting the requirements of the scheme was less problematic and costly compared with dairying, arable or pig production which dominated in the east and south of Ireland (Emerson and Gillmor, 1999).

AES in Ireland have generally had good uptake from farmers in the uplands. O'Rourke, Kramm and Chisholm (2012) found that 86% of farmers in the Iveragh uplands of county Kerry participated in REPS. Most of those farmers felt that the scheme had "*only a slight or no impact on how they farmed their land*". Nineteen percent of these farmers kept cattle in the uplands at low stocking densities of 0.02 $LU.ha^{-1}$ and had switched from store cattle (usually Kerry cattle) to spring calving suckler cows in line with subsidies and market demands (O'Rourke et al., 2012). This switch in system has implications for the uplands, as the tradition of outwintering the older animals on the hills is lost and the suckler cow do not graze on the higher ground. Farmers in the study cited REPS as a reason for the decline in upland cattle grazing, as under the scheme cattle had to be housed overwinter and mountain streams had to be fenced if cattle were to be grazed in the hills (O'Rourke et al., 2012).

AEOS was small and short lived compared to REPS and comparatively less is known about it (Image, 2016). There was considerable overlap between the options and measures in REPS and AEOS (Finn and Ó hUallacháin, 2012a). Indecon (2017) conducted an evaluation of the RDP 2007-2013 and applied GIS spatial analysis techniques to examine the extent to which AES were applied to areas of environmental/ecological interest. The results indicate that although REPS/AEOS funding was distributed across Ireland, it was focused on areas that contained the largest expanses of Natura 2000 land. Their results also show that the western counties have the highest area of REPS/AEOS funding by county (Indecon, 2017, p.104).

It has been recognised that prescription-based AES have not delivered adequately (Bleasdale, 2018). GLAS takes a hybrid approach, whereby farmers are paid for actions as well as their environmental performance. Although it does not include actions for upland farming *per se*, famers with Priority Assets (e.g. Natura 2000 sites, Hen Harrier and Commonages) are prioritised for entry to the scheme. With 40% of the Natura network occurring in the uplands, many upland farmers will have priority access to the scheme. However, GLAS does not target upland areas specifically, and locally led solutions may be preferable, considering the complex management issues that exist in the uplands (Bleasdale and O'Donoghue, 2015). In 2018 the DAFM funded three upland farming for conservation projects in Ireland which specifically target biodiversity and HNV farming in the uplands, these being: The Sustainable Uplands Agri-environment Scheme ('SUAS') project in County Wicklow, the Blackstairs Farming Futures project in (Carlow/Wexford), and the MacGillycuddy's Reeks project (county Kerry). Although currently in design phase, the general approach of these projects will be outcomes-based and locally led, using the RBAPS approach. These programmes run from 2018 to 2021.

Cattle and AES:

The majority of farmers engaged in REPS1-4 and the follow-up AEOS were cattle or sheep farmers in the west and northwest (Emerson and Gillmor, 1999; Finn and Ó hUallacháin, 2012a), and as O'Rourke, Kramm and Chisholm (2012) noted, farmers held the scheme at least partly responsible for the decline in cattle numbers in the uplands. REPS brought about a move towards spring calving suckler cows in place of traditional store cattle that would have been outwintered in small numbers and grazed higher ground. Requirements relating to fencing e.g. of upland water courses, were also implicated by farmers in the reduction of cattle numbers in the uplands. This decline in upland cattle was further linked by O'Rourke, Kramm and Chisholm (2012) to the spread of bracken (*Pteridium aquilinum*), gorse (*Ulex europaeus*) and hard rush (*Juncus inflexus*).

Commonage in Ireland

There are around 4,500 commonages in Ireland and they are important for conservation of the uplands and in sustaining rural livelihoods (Van Rensburg, Murphy and Rocks, 2009). Commonages cover approximately 478,930 ha of land in Ireland and 270,516 ha (60%) is upland and 261,130 ha (58%) of it is designated land (Bleasdale, 2018). They are typically areas of high conservation value as indicated by the proportions of designated land: the 58% of commonage land that is designated accounts for 90% of Ireland's SAC network, 10% of SPAs and 60% of NHAs (Van Rensburg, Murphy and Rocks, cited in Finn and Ó hUallacháin, 2012).

The introduction of Direct Payments under CAP had a dramatic impact on how commonages were managed in Ireland, particularly the ewe premium or 'headage payments', which resulted in a rapid increase in stock numbers on the Irish uplands, without reference to the size of holdings (Monaghan, 2012). This led to overgrazing in many commonages and subsequent damage to vegetation, in some cases leading to severe erosion soil loss. In the 1990s an initial cut of 30% in sheep numbers was imposed on farmers in 6 western counties. By 1998 Commonage Framework Planning, a join imitative between the NPWS and DAFM, was introduced to develop sustainable management plans for commonages in Ireland. Commonage Framework Plans (CFP) led to rapid destocking of commonages (and thus the uplands) from 2003 onwards. The introduction of SFP further reduced stock numbers as the link between direct payments and ewe numbers was broken i.e. decoupling of direct payments. Commonage Framework Planning was reviewed by the DAFM and NPWS in 2011, which led to the prescription of both minimum and maximum stocking densities. GLAS offers priority entry to commonage farmers, provided that a grazing agreement is in place that most shareholders in a commonage have signed up to.

The DAFM have identified the following breeds as being suitable for commonages (DAFM, 2012):

- Sheep: Blackface Mountain, Cheviot or a cross between these breeds.
- Cattle: Angus, Hereford, Kerry, Irish Maol, Dexter, Shorthorn, Galloway and Highlander or a cross between these breeds

Biodiversity Conservation

Biodiversity, the variability among living organisms from all sources (United Nations, 1992), underpins all ecosystem processes (Millenium Ecosystem Assessment, 2010). The diversity of genes, species, communities, populations and ecosystems, is essential in the provisioning of all ecosystem services that ultimately affect human beings. However, humans are significantly and perhaps irreversibly changing the diversity of life on Earth, with these changes mostly involving the loss of biodiversity (Millenium Ecosystem Assessment, 2010). Biodiversity loss leads to reduced ecosystem function such as nutrient cycling, pollination, pest control, climate regulation and soil retention (Maestre et al., 2012; Oliver et al., 2015), and further to reduced ecological resilience (Tilman and Downing, 1994). Recognition of the role of biodiversity in ecosystem functioning and resilience has led to the development of international policy aimed at halting biodiversity loss e.g. the United Nations Convention on Biological Diversity (United Nations, 1992) and the EU Habitats Directive (European Commission, 1992). The 'Natura 2000' is a network of sites protected for the conservation of biodiversity in the EU and is the largest network of protected areas in the world, covering 18% of the EU's land area and 6% of the marine region (European Commission, 2018).

In Ireland the main biodiversity conservation tool is the designation of protected areas. The Wildlife Act (1976 & 2000) is the primary piece of national legislation concerned with wildlife protection. Under the Wildlife Act, statutory Nature Reserves (NR) and Natural Heritage Areas (NHA) are designated for site protection, and the Flora Protection Order (FPO) for protection of plant species wherever they occur. Special Protection Areas (SPAs) are designated under the EU Birds Directive and Special Areas of Conservation (SACs) are designated under the EU Habitats Directive and together they make up the Natura 2000 network. The Irish Natura 2000 network covers approximately 14% of Ireland and contains 430 SACs covering 583,500 ha, and 153 SPAs covering 13,500 ha.

Conservation in the Irish Uplands

The Irish uplands are a stronghold for biodiversity rich blanket bog and heath habitats (O'Rourke et al., 2012). Thirty five per cent of Ireland's uplands are designated for biodiversity conservation (Bleasdale and O'Donoghue, 2015) and they contain up to 40% of our terrestrial SAC network (Perrin et al., 2011a). Up to 27 EU Habitats Directive Annex I habitats occur in the Irish uplands and twelve of these were the focus of the National Survey of Uplands Habitats (NSUH) (Perrin et al., 2011a). The dominant habitat types in the Irish uplands are blanket bog, heaths (wet and dry) and grasslands (semi-natural dry/humid acid grassland and wet grassland). The classification and vegetation composition of the key upland habitats is discussed in detail in sections 3.1.4 generally, and site-specifically in section 3.7.6.

The conservation status of upland habitats is monitored by the NSUH and reported under Article 17 of the Habitats Directive every 6 years (this process is described in detail in chapter3, section 3.1.4). In the 2007, 2013 and 2019 Article 17 reports, the overall conservation status for Northern Atlantic Wet Heath with *Erica tetralix* (4130) and Active Blanket Bog (7130) was 'Bad'. For European Dry

Heaths (4030), it was 'Inadequate' in 2007 and 'Bad' in 2013 and 2019. Conservation status reports for these habitats (and majority of EU protected upland habitats) are showing continuing negative trends, indicating that designation alone is insufficient to halt biodiversity loss and protect habitats (Bleasdale, 2018).

Ground beetles

Arthropods typically contribute over half of the metazoan species in any habitat, occupy several trophic levels and fulfil vital functions (Dennis, 2003). They affect a vast array of ecosystem services including pollination, decomposition and nutrient cycling (Prather et al., 2013). Arthropods are impacted by grazing management (Van Klink et al. 2013) and it affects them in a variety of ways, with most studies showing that diversity and abundance increase at low stocking densities (Dennis, Young and Gordon, 1998; Dennis et al., 2004, 2008; Pöyry et al., 2006). However, some studies have shown that a lack of grazing can be unfavourable for arthropods (González-Megías, Gómez and Sánchez-PiÑero, 2004; Debano, 2006). There have been very few studies on invertebrates in the Irish uplands (McCormack et al., 2009; Anderson, 2013), and none specifically relating to the response of cattle grazing in Annex I habitats.

Carabidae (Coleoptera) is a taxonomically and functionally diverse family of surface active arthropods that commonly occur in agricultural systems (Cole et al., 2002). In the uplands carabids ('ground beetles') constitute a significant part of faunal biodiversity and are of ecological importance as part of the food-web (Dennis, 2003; Pearce-Higgins, 2010). Carabids are sensitive to habitat heterogeneity and land use (Ribera et al., 2001; Cole et al., 2002) and are emerging as useful bioindicators (Rainio and Niemelä, 2003; Avgın and Luff, 2010). Extensive practices can promote structurally diversity swards and habitat heterogeneity (Dennis, Young and Gordon, 1998). Many studies have examined the spatial and functional displacement of beetles in relation to grazer impact (McFerran et al., 1994; Cole et al., 2002; Dennis, 2003). In general, these have found that large, poorly-dispersing specialist species decrease with increased grazing pressure, while smaller generalist species increase.

Birds

Ireland's upland habitat supports a range of rare and threatened bird species. Birdwatch Ireland identified twenty two species in its Action Plan for Upland Birds 2011-2020 (BirdWatch Ireland, 2010). Ten of these are Red listed on the Birds of Conservation Concern 2014-2019 (Colhoun and Cummins, 2013). Several of the species are rare breeders e.g. nightjar (*Caprimulgus europaeus*), ring ouzel (*Turdus torquatus*), short-eared owl (*Asio flammeus*), red-throated diver (*Gavia stellate*), merlin (*Falco columbarius*), golden plover (*Pluvialis apricaria*) and dunlin (*Calidris alpina*). Two species of concern are the focus of national conservation efforts that relate to the uplands: the hen harrier (*Circus cyaneus*) and curlew (*Numenius arquata*).

1.2 Grazing and Conservation

The use of grazing for conservation is becoming increasingly popular and examples exist across Europe of where grazing has led to high levels of biodiversity. In many cases, grazing regimes follow traditional practices for the maintenance or enhancement of semi-natural habitats. In others, grazing is adjusted and targeted to meet specific goals or objectives such as restoration of degraded ecosystems (Wallis de Vries, 1998). Grazing is a process that affects the structure and composition of plant communities and is widely regarded as an essential tool in conservation (Tallowin, Rook and Rutter, 2005). 'Conservation grazing' is used to describe the use of grazing animals to maintain or enhance biodiversity in semi-natural habitats (Small, 2003). Many habitats valued for their biodiversity are as a result of, or have been, maintained by grazing management. Grasslands and heaths are examples of where, in the absence of wild grazers, domestic animals may halt successional processes and thus conserve habitats.

Grazing is a process that directly affects the structure and composition of plant communities (Wallis de vries, 1998b) through processes of selective defoliation, trampling and excretion of urine and faeces. Ecosystem processes of nutrient cycling, turnover and productivity may be modified in the long term by grazing. It can further induce indirect cascading effects on the structure of entire ecosystems. For the purposes of wildlife conservation in semi-natural systems, grazing can be manipulated to induce desired impacts on target habitats and ecosystems. By influencing ecosystem processes, large herbivores can play a key role in conservation management (Wallis De Vries, 1998b).

Different species of grazer will induce differing effects. Grazing animals select certain plant species in preference over others and this has an impact on vegetation structure and composition. Large herbivores graze in a variety of different ways (Wright et al., 2006). Physiological differences contribute to differences in grazing and browsing behaviours, which affect plant selectivity. Cattle are unique in that they are the only large herbivores that use their tongues in a tearing action when grazing (Wright et al., 2006). They pull tufts of vegetation into their mouths and leave tussocks. This method of pulling and tearing means that they have a low ability to be selective and could be described as “Bulk Roughage Feeders” in that their inability to be selective makes them obligate grazers of grasses, sedges, rushes and herbs in a mixed sward setting (Wright et al., 2006).

Breed and rearing experience have effects on behaviour of cattle in semi-natural habitats. Orr et al., (2014) found that choice of breed has consequences for foraging behaviour. A traditional breed (North Devon) with extensive rearing experience (reared on semi-natural grassland) had higher rates of ‘Total Jaw Movement’ compared to cattle of a commercial breed (Hereford x Friesian). This indicated that they were better foragers in semi-natural swards of heterogenous structure. The cattle with experience of intensive settings grew less well in during trials in extensive settings compared to those that were reared in semi-natural grasslands (Orr et al., 2014).

Hessle, Rutter and Wallin, (2008) also showed that breed affects behaviour (in traditional Vaneko cattle versus Charolais), with the traditional breed roaming further and grazing over a greater proportion of their surroundings. Heifers of the traditional breed had higher grazing activity rates than the commercial breed in semi-natural grasslands. Season also had an impact on behaviour in this study, with cattle spending a greater proportion of available daylight hours grazing in autumn compared to summer or spring.

Impact on vegetation: excluding animals

Large herbivores graze in different ways and have varying impacts on sward characteristics and plant communities (Milne et al., 1998; Wright et al., 2006). Effects may be induced by the exclusion or introduction of grazing animals. The impacts of excluding grazing animals are reviewed by Bakker (1998) and the following generalisations are drawn. Grazed areas harbour pioneer species such as annuals and biennials, low stature species and rosette species. If grazing is excluded, tall grasses and herbs take over and leaf litter accumulates, followed by subsequent colonisation by shrub and trees (Bakker, 1998) i.e. the process of succession.

Various models have been developed to describe the relationship between species diversity, above ground standing crop, and disturbance gradients (Bakker, 1998). Disturbance is the sum of mechanisms that limit plant biomass (burning, grazing, cutting) i.e. the intensity of management. A low standing crop and low species diversity can be correlated with high levels of disturbance (Grime, 1979). However, a high standing crop is likely to mean high productivity and some species may be limited due to competitive exclusion (Bakker, 1998). Al-Mufti et al (1977) and Grime (1973, 1979) found that species diversity can be plotted as a bell-shaped curve along gradients of maximum standing crop (cited in Bakker (1998)). Huston (1979) found that moderate levels of disturbance and environmental stress are required for high species diversity. In mesotrophic conditions the removal of above ground biomass has been shown to result in high species diversity, with the same effect in oligotrophic environments if the biomass is removed only occasionally (Bakker, 1998).

Succession to scrub and woodland is often recorded after grazing is ceased but studies have shown that unidirectional change is not always a given and site history is a key factor. For example, Schreiber (1997) found that after cessation of grazing, fields previously used for arable showed greater scrub encroachment rates than fields that had been continuously grazed. Heathlands have been shown to show differential responses to grazing cessation in relation to site history (Wahren, Papst and Williams, 1994).

The type of diversity measure (species richness, evenness) and the scale (spatial and temporal) of the measure are important when capturing the response to grazing cessation. Equitability (evenness) has been shown to decrease prior to, and quicker than, the number of species, so both timescale and type of measure are important factors (Bakker, 1985; Bobbink and Willems, 1987). Excluding cattle from a wet grassland in Sweden showed a decline from 35 to 18 species per meter squared, but at a plot level (810 m^2) the decline was relatively much lower, going from 113 to 89 species. This revealed a progression towards a more heterogeneous or patchy sward (Persson, 1984) and showed the necessity for measurement at appropriate spatial scales (Bakker, 1985). The cessation of grazing may cause the number of species within a community, and the number of communities, to decrease in open vegetation (Bakker, 1985). Although the first number of years after the cessation in grazing may show an increase in flowering plants, this is a short-term effect (Bakker et al., 1997). Furthermore, studies have also shown that changes at a small scale are relatively bigger than changes at a large scale e.g. (Persson, 1984).

Impact on vegetation: introducing animals

The effects of introducing grazing animals are varied and summarised in (Bakker, 1998. p.114). Species richness has been shown to have a positive relationship with grazing intensity in grasslands of Westerholt over a 10-year period. It also transformed a large uniform area with few plant communities to one with a pattern of small patches (Bakker, 1989; Bekker and Bakker, 1989). On large sites that vary abiotically, as is often the case with study sites, effects on plant communities vary because patches are not uniformly attractive to herbivores (Bakker, 1998). In the Netherlands the impact of cattle grazing varied within a 25-ha site due to differences in ground wetness. Cattle avoided wetter ground resulting in transformation of *Calthion palustris* communities with *Carex acutiformis,* transforming into species-poor *Carex acutiformis* communities. Cattle rested on the driest part of the site but did not graze there, resulting in dung accumulation and the development of tall herbs which displaced low-growing species. Where cattle grazing was intensive, the soil became impoverished which resulted in a reduction in species of eutrophic soils and the expansion of *Agrostis capillaris*, a species of mesotrophic soils. High grazing intensity was indicated by the emergence of rosette plants (Van Den Bos and Bakker, 1990).

Species and stocking levels

Different species and stocking rates result in varying impacts on plant communities which is related to selectivity. It has been suggested that cattle are less selective than sheep (Grant et al., 1985; Fraser et al., 2009). Cattle ingested more *Nardus stricta* than sheep in grasslands in southern Scotland (Grant et al., 1985; Hodgson et al., 1991a). Cattle have been shown to decrease the cover of *Nardus* from 55% to 30% over five years, whereas under sheep grazing it increased to 80% in the same period. The utilisation *Nardus* increased as the height of between-tussock grasses (*Agrostis* spp., *Deschampsia flexuosa* and *Festuca ovina*) decreased. Scottish research indicates that heavy grazing favours grasses and herbs and light grazing favours Ericoid species and lichens (Welch, 1984). Over 20 years heavy grazing reduced Ericoid species, lichens and *Deschampsia,* whereas *Agrostis capillaris, Anthoxanthum odoratum, Festuca ovina, Gallium saxatile, Luzula multiflora, Nardus and Rhytidiadelphus loreus* increased significantly. *Calluna* increased under light grazing conditions, with corresponding declines in graminoids and some bryophyte species (Welch and Scott, 1995).

In summary, conservation grazing may be animal species orientated e.g. for wintering geese (Eerden, Van, 1996), or focused on plant species diversity or community diversity, or more generally on wilderness (Van Wieren and Bakker, 1998). Effects induced vary depending on whether animals are excluded or introduced, with grazer species and with site (and within sites). Effects also vary at spatial and temporal scales and occur on macro and micro levels (Bakker, 1998). If the goal is 'object' orientated (species-focused) it is possible to control and target grazing. However, if the goal is 'system' focused (habitat conservation) then outcomes are less well defined (Van Wieren and Bakker, 1998). Conservation goals should be defined explicitly so as to decide on appropriate management and also to enable evaluation of the actions (Van Wieren and Bakker, 1998).

Grazing for conservation in Ireland

The issue of grazing has been a sensitive one in Ireland, particularly in the last two decades, and opinions of political, agricultural and environmental groups are often conflicting. In general, conservation efforts have focused on adjusting stocking densities in reactive response to negative impacts on habitats, particularly so in the uplands in respect of agri-environment schemes. The Burren Programme was the first project to employ grazing from a goal-orientated perspective i.e. conservation grazing for the purposes of 'producing' species rich grasslands. No research has been conducted to date in the Irish uplands regarding conservation grazing on Annex I habitats.

This project was a 'system-orientated' conservation grazing venture, the primary goal of which was to develop prescriptions for the maintenance or enhancement of upland habitats. Chapter 2 examines the home range, habitat selection and behaviour of Dexter cattle in Mt Brandon Nature Reserve, while Chapter 3 explores the impact on habitats and species within the Reserve. Chapter 4 studies ground beetle communities of the Reserve and looks at the response of them to the grazing. Chapter 5 provides final conclusions and recommendations. Chapters 2, 3 and 4 were written as stand-alone chapters, so note that there is some repetition regarding site descriptions and experimental design.

Cattle in the Irish uplands

Opposing and controversial views on the nature of the Neolithisation process in Britain and Ireland have emerged in recent decades (Tresset, 2003). In Britain, fully domesticated animals seem to have appeared suddenly in a Neolithic context, with cattle and sheep having been imported from the Paris and Rhine basins (Tresset, 2003). In Ireland the situation is different, with domestic cattle appearing in a late Mesolithic context and perhaps from a southern route *via* western France or Iberia rather than from Britain (Tresset, 2003). Radiocarbon-dated domestic cattle bones from Ferriter's Cove in Co Kerry and Kilgreany Cave in Co Wexford date to the end of the 5th millennium BC. One bone, although species identification is uncertain, may date to the mid-6th millennium BC (Woodman, Mccarthy and Monaghan, 1997). Little is known about exactly how or when culture and lifestyle associated with the Neolithic spread to Ireland but it seems to have been developing from around 4000 BC and these developments involved the introduction of arable farming and domesticated animals, most notably of cattle and sheep (Woodman, 2016).

The introduction of cattle to Ireland and to a lesser extent sheep in the Neolithic period had substantial consequences for both arriving Neolithic farmers and existing Mesolithic hunter-gatherers (Woodman, 2016). Large mammal species present in other parts of Europe during the mid-Holocene were absent from Ireland. Although bear (*Ursus arctos*) and wild pig (*Sus scrofa ferus*) were present, aurochs (*Bos primigenius*), European elk (*Alces alces*), roe deer (*Capreolus capreolus*) and probably red deer (*Cervus elaphus*) were absent (Woodman, 2016). More large mammal species present in continental Europe provided not only more food sources but also resulted in the development of numerous different tool types made from antlers and bones which were substituted in Ireland with stone tools (Woodman, 2016). For Neolithic farmers that arrived in Ireland the presence of cattle would have been crucial, and for the Mesolithic hunter-gatherer communities already present in the country, they would have been a remarkable addition (Woodman, 2016)

Although cattle have been present in Ireland since as early as *c* 4350 BC (~6300 BP) (Tresset, 2003), the construction of fences and controlled herding may have been important in the development of distinct breeds. The earliest evidence for controlled cattle grazing is found at the Céide Fields in Co Mayo, northwest Ireland. The fields which are between 5 and 50 ha, are bounded by walls made from soil and rubble and represent one of the oldest enclosed farmland systems in the world (Hall, 2011), dating to the early Neolithic around 5750 – 5150 cal BP (Verrill and Tipping, 2010). Until such large-scale controlled grazing came about, the mating of selected animals necessary for breed formation would have been difficult or impossible (Curran, 1990).

Wilde (1862) described four forms of early Irish cattle. The 'Common Cow' was small, had moderate sized wide horns and was principally black. The second, perhaps similar to the modern Kerry cow, was "*exceedingly hardy*" and largely confined to the south west (Feehan, 2003; Wilde, 1862). The 'Irish Longhorn' was the third breed described of the time. It was a large animal and a result of crossing existing Irish breeds with imported cattle, possibly Lancashire or Craven cattle (Feehan, 2003). The fourth breed of the time was 'maol' or 'moyle' and like a breed that appeared later, the Angus. Larger than the Kerry and smaller than the Irish Longhorn, the 'moyleen' was usually dun, black or white in colour (Wilde, 1862; Feehan, 2003).

By the mid-eighteenth century, a breed of black cattle had become an established feature of Kerry and west Cork and was known as the 'poor man's cow' because of its ability to produce high quality milk from a diet which is mainly roughage (Alderson, 1981; Curran, 1990), and it was around this time that the first references to "Kerry Cows" arose. The popularity of Kerry cattle grew during the 19th century and pedigree breeding was prevalent.

The origins of the Dexter breed are difficult to establish. Separation of true Kerries from Kerry-Dexters by the Royal Dublin Society (RDS) officially occurred in 1863, but it seems clear that Dexter-type cattle were always an important component of the Kerry breed (Curran, 1990). Various authors have attempted to track down the origins of both the Dexter breed and of the Dexter name. Low

(1845) stated that a Mr Dexter of Co Tipperary had produced his breed from the mountain cattle of the area. However, Wilson (1909) concludes that no clear evidence exists that Mr Dexter started the breed. He used numerous sources, such as statements from agricultural writers of the period (Tighe, 1802; Wakefield, 1812; Youatt, 1834), and archaeological evidence, to show that Dexter-type cattle existed in Ireland long before the name was applied to them. Curran (1990) suspects that the name 'Dexter' as applied to the breed, represents an example of Irish humour, in that the 'Mr Dexter' referred to by Low and Wilson, was a small, stocky man who happened to deal with small, stocky cattle that existed at the time, out of which the pedigree Dexter and Kerry breeds were to later emerge.

The archaeological evidence suggests that Dexter-type cattle were in Ireland for many centuries and were an integral part of the Kerry breed, only coming to prominence during the eighteenth and nineteenth centuries (Curran, 1990). By the mid-eightieth century the Dexter began to emerge as being distinct from the Kerry breed and Walsh (2017) highlights cattle shows in 1850 and 1861 that award a '…*Kerry cow, Dwarf*' and a '..*Kerry Co, Dexter*'. The Royal Dublin Society (RDS) split the Kerry and Kerry-Dexter types in 1863 but merged them again the following year, with formal recognition of the Dexter breed not occurring until 1876 (Walsh, 2017).

The Dexter breed declined in Ireland during the 1900s due to breeding issues and achondroplasia which results in 'bulldog' calves (Walsh, 2017). Affected animals display extreme dwarfism, short vertebral columns, large abdominal hernia and a large head with a retruded muzzle and a cleft palate (Harper et al., 1998). This presented major difficulties for breeders during the early 1900s. The favoured phenotype tends to be heterozygous for the bulldog mutation and thus selecting for it maintains the lethal allele at high frequencies (Cavanagh et al., 2007). Bulldog dwarfism is cause linked to mutations in the aggrecan, a protein that is encoded by the ACAN gene (Cavanagh et al., 2007).

The numbers of Dexters declined in Britain and Ireland during the 1900s and by 1930 the breed had reached a bottleneck. Numbers were so low in Ireland by 1928 that the RDS stopped accepting Dexters into the herd book. In the 1970s and 80s Dexters were re-introduced to Ireland from the UK and in 2015 the Dexter Cattle Society (Rep. of Ireland Group) was established and has around 140 members (Walsh, 2017).

Dexter is a hardy and adaptable dual-purpose breed and can be red, black or dun. Average weights are 325 kg for cows and 475 kg for bulls. Average heights (to withers) are 104 cm for cows and 114 cm for bulls. There are around 2,000 registered cows in Ireland (data from the Dexter Cattle society, summarised in Walsh (2017, p. 71)). Two types are recognised; short legged (Dexter-type) or long legged (Kerry-Dexter types). They have a short, broad head with a wide jaw and prominent eyes (Dexter Cattle Society, 2014; Curran, 1990).

1.3 Aims and Objectives

Aim

To provide specific evidence-based management recommendations for upland areas with a view to optimising biodiversity in farming compatible systems.

Objectives

1. To explore the home range behaviour of Dexter cattle in a 462-hectare upland Reserve comprising of an assortment of habitat types in a traditional summer grazing regime.
2. Identify habitat preferences and environmental factors influencing them.
3. Examine the impact of low-density seasonal grazing regimes on upland biodiversity, using vegetation and ground-dwelling invertebrates to assess changes in the condition of three Annex I habitat types: Active Blanket Bog, Northern Atlantic Wet Heath with *Erica tetralix* and European Dry Heath.
4. To make recommendations in relation to appropriate management of upland High Nature Value (HNV) farmland systems for sustainable agriculture and biodiversity.

2 Chapter 2: Home Range, Habitat Selection and Activity Budgets of free-ranging Dexter cattle *Bos taurus* in south west Ireland

Dexter cow with GPS collar, Mt Brandon Nature Reserve. Photo K.Kelly.

Dexter cow on blanket bog/wet heath. Mt Brandon Nature Reserve. Photo K.Kelly.

2.1 Introduction

2.1.1 Home Range

Understanding how organisms use their habitats is one of the fundamental questions in ecology and many methods exist for studying animal behaviour (Mech and Barber, 2002). Telemetry allows remote sensing and reporting of information and is described by Loureiro and Rosalino (2009) as being one of the most important tools used by ecologists. Global Positioning Systems (GPS) telemetry has now become an important and frequently used tool for studying wildlife generally (Augustine, Crowley and Cox, 2011; Kenward, 2001), and it has also greatly enhanced the ability to study free-ranging livestock (Augustine and Derner, 2013). Animal movement is complex and can be driven by multiple factors, including internal states (e.g. memory and perception) or external factors related to the environment (Liu et al., 2015). GPS systems have facilitated great advances in studying distribution patterns of free-ranging livestock, but these advances are complicated by challenges of experimental design and statistical analyses (Stephenson and Bailey, 2017).

'Home Range' is a core concept in spatial ecology. It describes the space in which animals live (Kenward, 2001) and estimating home range using data collected *via* GPS is now widely used for management and conservation purposes (Aebischer, Robertson and Kenward, 1993; Kenward, 2001; Browne and Aebischer, 2003). Methods of estimating home range have been developed since Mohr (1947) first introduced the concept of 'minimum home ranges' and started using minimum convex polygons (MCPs) to delineate ranges (Laver and Kelly, 2008). Two main families of home range estimation exist: parametric methods based on location density distributions and non-parametric methods based on linkage distances between individual locations (Kenward, 2001).

MCPs were some of the first home range estimates and are still widely used, making them useful for comparative purposes (Laver and Kelly, 2008; Enright, 2012). MCP is a linkage method and is the simplest way of defining a home range. It involves drawing a boundary around the outermost GPS locations and creates a home range based on the extent of all locations recorded. They are easy to draw and often the first step in estimating the home range of an animal (Kauhala and Auttila, 2009). With MCPs there is no assessment of preferred areas and no consideration is given to underlying statistical distributions.

The need to statistically analyse home range estimations has led to more explicit definitions of home range and Kernel Density Estimation (KDE) is a now a widely used technique (Kie, Baldwin and Evans, 1996; Seaman and Powell, 1996). Kernel methods can reveal one or more core activity areas (Worton, 1989) and are better estimators of home range than MCP (Seaman and Powell, 1996; Kenward et al., 2001a, 2008; Mitchell and Powell, 2008). KDEs are parametric models of utilisation distributions and produce density estimates based on Gaussian or compact kernels (Lyons, Turner and Getz, 2013). KDEs allow for multi-nuclear distributions to be developed over a matrix of intersections and then contours are interpolated between those intersections. However, a smoothing factor is applied when creating the contours. This process may over-smooth the contours delineating used areas, especially if outliers extend the distribution (Kenward, 2001). Furthermore, the greater the smoothing applied, the less precise the fit to the actual pattern of locations (Kenward, 2001).

Recent home range methods combine the simplicity of MCP with the statistical strength of kernel estimation. Objective-Restricted-Edge Polygon (OREP) estimation, an outlier-exclusion linkage approach, developed by (Kenward, 2001), combine the simplicity of MCPs with the strength of kernel estimation. OREPs have been used on cattle data by Bevan (2008), by Williams et al. (2009, 2010a) on sheep, and by Enright (2012) on goats. This method is equivalent to cluster analysis with objective coring by Kenward et al. (2001b) and to local nearest-neighbour convex hulls, *sensu* Getz and Wilmers (2004). It objectively defines an outlier exclusion distance to exclude locations with neighbour distances beyond the normal distribution and produces an 'excursion-exclusive' home range (Kenward, 2001). OREPs unify analyses based on grid-cell, polygon and location density techniques (Kenward et al., 2008).

Challenges in animal movement studies include practical problems of cost and labour, but also issues of replication, auto-correlation and sample size (Liu et al., 2015). Home range methods often treat GPS locations as being independent, an assumption contravened by regularly sampled GPS locations (Kenward, 2001; Lyons, Turner and Getz, 2013) and one of the issues with wildlife telemetry data is serial correlation. These correlations may be simple or complex, in terms of repeating patterns in time (monotonic) or space (Euclidean distances) (Fieberg et al., 2010a). Causes of correlations in GPS studies can be linked to physical or physiological differences (animals cannot access certain places, speed of animal movement etc.), behavioural characteristics of the animal (ruminating herbivores, roosting birds etc.), or external factors such as prey selection, limitations in resource selection, seasonality or pressures from other species (Johnson et al., 2002; Fieberg et al., 2010b).

Repeated observations on an individual may give rise to within-group correlation structures i.e. a constant correlation may be assumed with observations on the same individual, to a lesser degree with animals from the same herd and independent among observations from different herds (Fieberg et al., 2010a), and the term 'autocorrelation' is reserved for within animal correlation patterns. Autocorrelation is where the position of an animal at *time t* + Δt is dependent on its position at *time t* (Rooney, Wolfe and Hayden, 1998) and where it is possible to predict an animal's position based on its last position. With data that are autocorrelated there is a risk that home range sizes could be overestimated (Swihart and Slade, 1985). Therefore, time-to-independence should be calculated (Rooney, Wolfe and Hayden, 1998), or repeated observations on an individual should be pooled i.e. the collared animal is the sample unit, not the GPS locations (Kenward, 2001). In studies of cattle behaviour, animals are not considered to be independent of each other because of effects of social behaviour, however studying behaviour in settings where animals roam freely is logistically challenging, so treating individuals as independent samples is often beneficial (Stephenson and Bailey, 2017).

The number of animals required to be representative of the whole is a challenge in GPS tracking studies. Monitoring subsets of a herd reduces costs and labor but may result in incomplete information on the whole. Liu et al. (2015) found that travel distances and speed were overestimated with smaller sample samples. However, analysis of kernel density estimataion showed that animals had high levels of spatial occupancy and that monitoring of an appropriate sub-set of preserved most information with accepatable levels of error (Liu et al., 2015).

Home range estimations can be combined with habitat data to examine habitat use i.e. comparing what is used by the animal with what is available. Johnson (1980) proposes four levels of selection. First-order selection is the selection of a broad geographical area and second-order is the uptake of a home range of an individual/group in that area. Third-order selection is the use of various habitat components within the home range and fourth-order is the actual selection of food items. Aebischer, Robertson and Kenward (1993) proposed a two-stage approach to examining resource selection: first by comparing the proportion of habitats available in the study area with those used within the home range and second by comparing those available in the home range with those used at location.

2.1.2 Activity budgets of cattle

Activity budgets are used to assess how an animal divides its time and Arnold & Dudzinski (1978) were the first to use the terms 'free-ranging' and 'ethology' in the study of grazing animals (Arnold & Dudzinski 1978, cited in Kondo (2011). How animals budget time and how they interact with their environment is a balance of meeting energy requirements, and investing energy in survival and reproduction (Fan et al., 2008). Many studies have demonstrated that budgets vary in relation to the abundance, quality and spatial distribution of food resources (Altmann and Muruthi, 1988; Hanya, 2004; Vasey, 2004, 2005).

Activity budgets have been studied in many wild and free-ranging domestic ruminant species, including deer (Parker et al., 1999; Zhang, 2000), goats (Shi et al., 2003; Enright, 2012), oryx (Ruckstuhl and Neuhaus, 2009) and sheep (Arnold, 1984; Liu et al., 2005; Pokorná et al., 2013) and the dominant activity in ruminant species is usually feeding. The grazing behaviour of grazing animals is therefore an important component on research into grazing systems (Hirata et al., 2002).

The performance of grazing animals depends on their ability to ingest an adequate diet to meet nutritional requirements (Higashiyama and Hirata, 1995; Hasegawa and Hidari, 2001; Braghieri et al., 2011). The amount of time spent by ruminants in grazing activities depends on climatic and pasture conditions (i.e. the quality and availability of forage) and physiological states of animals (Hirata et al., 2002). Therefore, activity patterns vary with, *inter alia,* forage availability and quality (Festa-Bianchet, 1988; Fryxell, 1991; Moncorps et al., 1997; Shi et al., 2003).

A study on Kerry cows in 'semi-wild' conditions showed that total grazing time in cattle tends to be stable over a season (Linnane, Brereton and Giller, 2001), yet the intensity of the grazing may affect the trade-offs cattle make between forage quality and quantity (Wallis de Vries and Schippers, 1994; Newman et al., 1995). Heavy stocking tends to reduce grazing time because animals have to spend more time searching (Hepworth et al., 1991).

Hejcmanová et al. (2009) found that heifers spent more time grazing in intensively grazed systems than in extensively grazed pasture, although not significantly so. Temperature and season had an impact on grazing times, with a reduction found at times of high temperatures. The amount of time grazing increased month to month, with consistently decreasing biomass growth rates (Hejcmanová et al., 2009). These trends were also reported by Linnane, Brereton and Giller, (2001) in Kerry cows.

The ecology of grazing has been studied extensively since the 1980s, with Senft et al. (1987) proposing the concept of a decision hierarchy at different spatial and temporal scales. This was later expanded by Bailey et al. (1996) to include bite, feeding station, patch, feeding site, camp and home range (Kondo, 2011). Forage intake is determined by bite, biting rate and grazing time (Allden and McDWhittaker, 1970; Hodgson et al., 1991b; Forbes, 1988; Hejcmanová et al., 2009). Where forage quality is poor, longer feeding bouts are required to fill the rumen (Bourgoin et al., 2008).

The effects of breed, experience and season may also influence behaviour. Orr et al., (2014) found that rearing experience and breed had an impact on foraging behaviour. Traditional breeds with experience of unimproved grassland pastures were able to forage more effectively than commercial animals in tussocky semi-natural grasslands swards (as measured by 'Total Jaw Movement' i.e. bites). Hessle, Rutter and Wallin (2008) also showed that breed influenced behaviour, along with season and moisture content. They found that traditional breeds had higher activity rates than commercial breeds in semi-natural grasslands; heifers of the traditional breed roamed more broadly over the available area compared to commercial animals.

Kilgour et al. (2012) reviewed the literature on the behaviour of cattle and observed that early studies focused on the three primary activities of grazing, ruminating and resting. Later studies added additional behaviours such as travelling, licking and drinking e.g. Cory, (1927) and Reppert (1960). Compton & Brundage (1971) were the first to describe a time budget which included all behaviours (cited in Kilgour et al., 2012). Approximately 40 categories of cattle behaviours can be described, often grouped into the categories of maintenance, self-expression and social (Kilgour et al., 2012). Hall (1989) also added categories of 'social-actor' and 'social-acted' to describe where the animal is engaged in giving or receiving social interaction e.g. grooming, licking or displacing behaviour.

Grazing is the most common behaviour in cattle, and they allot 4.5 – 9.3 hours to it in daylight hours. Cattle graze less at night and reported times vary from 0.4 h to 4.5 h across studies (Hughes and Reid, 1951; Low et al., 1981; Orr et al., 2012). Next to grazing, rumination is the most important behaviour for cattle and they allocate 1.4 h to 6.9 h to it in daylight hours, and up to 10.2 h over a 24 hour period (Kilgour et al., 2012). Rest is the third most important behaviour for cattle (and most ruminants) and average times vary from 3.6 to 10.3 h over a 24 h period. Overall, cattle allocate 90-95% of their time to the three primary activities of grazing, ruminating (while standing or lying) and resting. Across 5 studies reviewed by Kilgour (2012) an average of 34% of time was allocate to grazing and 28% and 22% to ruminating and resting respectively.

2.1.3 Dexter cattle

By the mid-eighteenth century a breed of black cattle had become an established feature of Kerry and west Cork and was known as the 'poor man's cow' because of their ability to produce high quality milk from a diet of poor quality forage (Alderson, 1981; Curran, 1990). It was around this time that the first references to 'Kerry Cows' arose. The popularity of Kerrys grew during the 19th century and pedigree breeding was prevalent.

Separation of true Kerry cattle from Kerry-Dexters by the Royal Dublin Society (RDS) officially occurred in 1863 and it is likely Dexter-type cattle were always an important component of the Kerry breed (Curran, 1990). The Dexter-type cattle were in Ireland for many centuries but only became popular during the eighteenth and nineteenth centuries when selection for particular traits was made possible by land enclosures (Curran, 1990). The Dexter, on which this study is based, is considered a hardy and adaptable dual-purpose breed suited to a range of geographical locations. Two forms are recognised; short legged ('Dexter'-type) or long legged ('Kerry-Dexter' types) and they have a short, broad head with a wide jaw and prominent eyes (Curran, 1990; Wilson, 1909; Dexter Cattle Scoiety, 2014).

In Ireland the uplands (land over 150 m) form our greatest expanses of semi-natural habitats (Perrin et al., 2010). Almost 29% of the land mass of Ireland is considered upland and over 40% of the Natura 2000 Network (sites designated under the EU Habitats Directive) in Ireland occurs in the uplands. Uplands are important areas for conservation and they contain up to 14 habitat types listed in Annex I of the Habitats Directive (Perrin et al., 2009). Annex I habitats are of community importance and under the Directive Ireland is obliged to maintain priority habitats in favourable conservation status. Strict criteria for achieving this status are detailed in the Directive under the headings of Area, Range, Structure and Function, and Future Prospects (European Commission, 1992).

The latter half of the twentieth century has brought about increasingly widespread degradation of Irish uplands (Heritage Council, 1999). Reforms have tried to redress the grazing imbalances in the uplands, much of which are deemed to be HNV farmland (i.e. farming systems where traditional practices have maintained high levels of biodiversity) using a wide range of measures, including agri-environment schemes such as the Rural Environmental Protection Scheme (REPS) and the Commonage Framework Plan (CFP). However, results have been ambiguous to date (Kleijn and Sutherland, 2003b; Whittingham, 2007; Finn and Ó hUallacháin, 2012a) and the status of the majority of EU Protected upland habitats in Ireland is currently assessed to be in poor or bad condition (NPWS, 2008, 2013, 2019c).

Recent work examined the home range and habitat preferences of sheep (Williams et al., 2009, 2010, 2012a) and goats (Enright and Williams, 2010; Enright, 2012).

However, there have been no reports on extensive upland cattle grazing in an Irish context outside of the Burren (Burrenbeo Trust). Linnane, Brereton and Giller (2001) studied the behaviour of Kerry cows in a 'semi-wild setting' in Killarney National Park, but this study was conducted in one 4.7 ha enclosed field in a lowland setting. An in-depth study of an extensive cattle-based upland grazing regime is therefore necessary since current literature is deficient of reports on such grazing regimes in the Irish uplands. A key first step is an examination of home range, resource selection and activity budgets of cattle in a free-ranging setting. Analysis of spatio-temporal data derived from GPS tracking of cattle was used in this study to explore these relationships.

2.1.4 Developing stocking rates for upland habitats

A brief review of different approaches to livestock units (LUs) and stocking rates (SRs) follows here, leading into the development of an evidence-based approach for the Irish uplands.

Livestock Units and Stocking Rates

Livestock Units (LU) came about to estimate the overall grazing pressure on land and allow for the aggregation of stock from different species, breeds and age class (European Council, 2009) . They were originally developed for ruminants but are now also applied to equines, pigs and poultry.

Livestock Units, Stocking Rates and Stocking Densities measure the stock of grazing animals per area over time (e.g. LUs/ha/year). The European Commission sets a reference Livestock Unit and the coefficients (multipliers) can be established based on the nutritional requirement of different species and breed of animal. Establishing the correct stocking rate is critical for optimising both forage and animal performance, and the management of semi-natural habitats.

Many factors can affect stocking rates, including management goals, animal species and breeds, class of stock (dry cow, bull, steer etc.), available area of land, topography, soils and habitats, forage quality and palatability, forage species composition, and land management practices (Meehan et al., 2018). Various approaches are used for developing stocking rates depending on requirements or major drivers. For example, a weight-based approach is used in the US, an economic output approach in the EU, a habitat management approach in the Irish and British uplands (e.g. Commonage Management), and a nutrient input approach in the Irish lowlands.

The EU Commission reference unit is the Livestock Unit (LU) and 1 LU is equivalent to one adult dairy cow over 2 years old producing 3,000 kg of milk annually without additional concentrates (European Commission, 2013). Multipliers are then applied for other species and breeds and programmes across Europe adopt this approach (Table 2). In Environmental Stewardship Schemes in the UK, one LU is equal to one dairy cow (weight equivalent = 650 kg), a heifer or steer under 2 years is equal to 0.6 LU and one medium sheep is 0.08 LU (Chesterton, Condliffe and Peel, 2006).

Table 2 Animal species and ages with equivalent livestock units (LU) (European Commission, 2013).

Animal	LU value
Female or male cattle over 2 years old	1
Female or male cattle 2 years old or younger	0.6
Female or male sheep	0.15
Female or male horse	1
Female or male donkey	1
Female or male goat	0.15
Female or male deer	0.3

Animal Units (AU) and Animal Unit Months (AUM) are used in the United States. AUMs are based on age, class and size of stock, and on the amount of forage they consume in one month (Meehan et al., 2018). A standard 'Animal Unit' is defined as a mature 1000 lb cow (453.6 kg) with a calf of less than 6-months at foot, and adjustments are made depending on breed, class and size of livestock (Basarab and Gould, 2001). A cow suckling a calf is assumed to require 26 lbs of dry matter per day (11.8 kg). This amount of consumption with additions for waste and trampling, amounts to an estimated requirement of 1000 lbs (454 kg) of dry matter per month. As the relationship between animal weight and metabolic requirement is not linear, Animal Unit Equivalents (AUEs) are used to adjust the basic AUM. For example, a mature bull is assigned an AUE of 1.4 and is assumed to require 1,295 lbs (587 kg) of dry forage per month, whereas a mature sheep with a lamb is assumed to require 182 lbs per month and is assigned an AUE of 0.2. Stocking rates are then calculated by multiplying the number of animals to be grazed on pasture by the AUE and number of months planned to graze (Meehan et al., 2018).

Livestock units and Stocking rates in Ireland

In Ireland, the EU Livestock Unit is used, where a cow over 2 years old is the reference unit and coefficients are applied for different breeds and ages (previously in table 2). Stocking rates are then applied per unit area for area-based payments, such as under the Basic Payment Scheme ANC scheme, as well as agri-environment schemes such as GLAS (DAFM, 2015a, 2018a).

Stocking rates on Irish lowland farms are based on the amount of nitrogen produced per year on a holding. For example, a dairy cow produces 85 kg of nitrogen per year, a suckler cow produces 65 kg and a mountain hogget produces 4 kg (DAFM, 2017b). Articles 14 and 20 of S.I. No. 605 of 2017 ('Good Agricultural Practice for the protection of waters') set out the annual nutrient excretion rates for livestock. Limits are set to 170.ha.yr^{-1} and 90% of holdings in Ireland are less than this (DAFM, 2017b).

Stocking the Uplands

The Areas of Natural Constraint (ANC) scheme provides support for people farming in land designated as disadvantaged and aims to ensure continuation of farming practices in these areas. It is a scheme for land situated on the mainland of Ireland that is designated as 'Disadvantaged/Constrained', pending delineation under the provisions of Regulation (EU) No. 1305/2013 (DAFM, 2018a). It is coupled with the areas of Specific Constraint (ASC) scheme which deals with offshore islands. The ANC/ASC designations are based on biophysical criteria such as low temperature, excessive soil moisture, limiting soil drainage, shallow rooting depth, unfavourable texture, and most townlands of Irish uplands are eligible for the scheme (Figure 2).

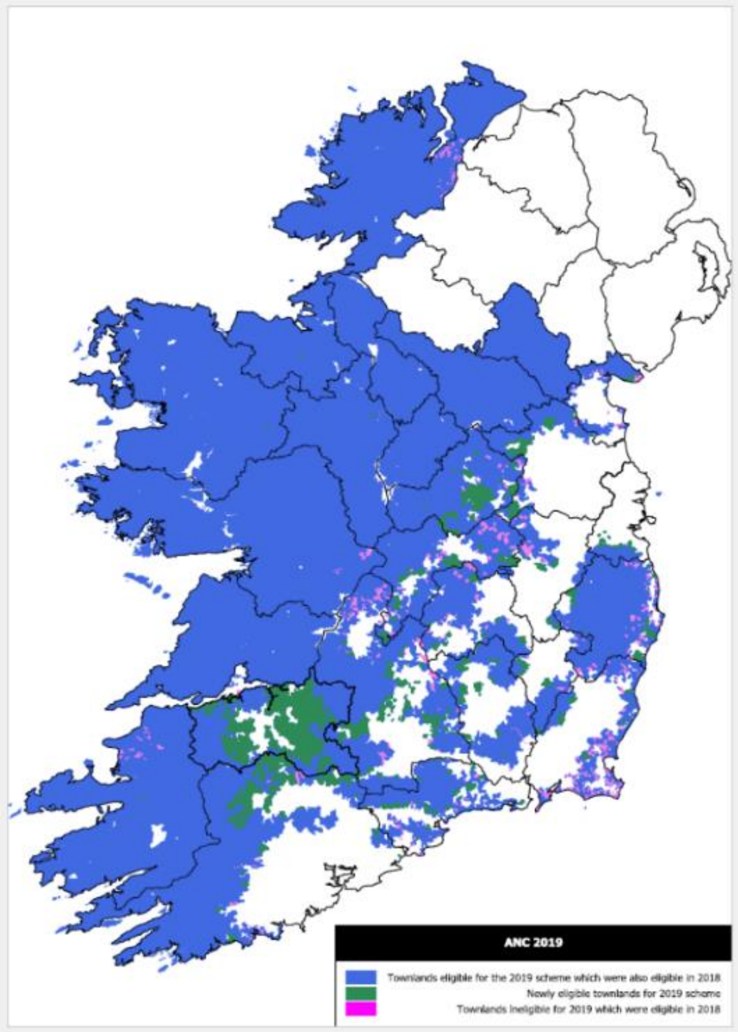


Figure 2 Map of ANC eligible townlands 2019 (source, (DAFM, 2019)).

Under the ANC/ASC scheme, the minimum retention period is seven consecutive months and the stocking density "*has to be equal to or greater than 0.15 livestock units per hectare*" (DAFM, 2019). In addition to the seven-month retention period participants must also maintain a stocking density of 0.15 livestock units per forage hectare calculated over 12 months of the calendar year (DAFM, 2019). The application of minimum stocking densities for upland areas under the ANC/ASC and under Commonage Management is in contrast with UK based schemes where no such minima are applied.

The ‘Guide to the completion of the GLAS Commonage Plan’ (DAFM, 2015a) sets out a formula for determining sustainable stocking rates on commonages based on habitat condition. Stocking rates are divided among active shareholders to meet the minimum. Once the minimum is achieved, any surplus above the recommended is distributed among the active farmers. The condition of a commonage is subsequently assessed and amendments to stocking rates can be proposed by an agricultural advisor. Advisors recommend adjustments to stocking levels depending on whether the commonage is being ‘adequately grazed’, ‘overgrazed’ or ‘undergrazed’ (referenced to Habitat Condition assessment cards). A ‘sustainable stocking rate’ can be calculated with reference to recommended ‘undamaged stocking rates’ for different habitats by applying a stocking co-efficient that is dependent on Habitat Condition (Table 3 & 4) (DAFM, 2015a). The formula for determining the sustainable stocking rate is as follows:

‘Sustainable Stocking Rate = Undamaged Stocking Rate X *Stocking Co-efficient’*

Table 3 DAFM recommended stocking rates for Undamaged Habitats under Commonage Management Plans (DAFM, 2015).

Habitat	**Undamaged Stocking Rate ($ee.ha^{-1}$)***
Blanket Bog	0-0.75
Wet Heath	0.75-1.0
Dry Heath	1.0-1.5
Upland Grassland	1.5-5.0
Other	Variable

* Ewe equivalents per hectare. A Ewe Equivalent for cattle over 2 years is 6.7, cattle under two years is 4.0, a hogget (and older) is 1.0, horses and donkeys are 6.7, goats are 1.0, and farmed deer are 2.0.

Table 4 Habitat condition codes and stocking co-efficient to be applied to stocking rates (DAFM, 2015).

Habitat Condition Code	Stocking Co-efficient
U (Rank/Undergrazed)	1.0-1.25
U (Undamaged)	1.0
MU (Moderate Undamaged)	0.80-0.98
MM (Moderate Damage)	0.40-0.99
MS (Moderate Severe)	0.2-0.39
S (Severe)	0-0.19
S* (Very severe)	0

In Ireland stocking rates do not consider the different sizes and demands of individual breeds. Breeds of the same species will have different live weights (Table 5 & 6), which affects the amount of forage they require (Martin et al., 2013). The Rural Development Service (now Natural England) developed a more systematic approach based on food intake and performance of breeds, an approach that is currently lacking in Ireland.

Table 5 Animal weight categories (RDS/Natural England), adapted from (Martin et al., 2013).

	Liveweight (kg)		
	Small	Medium	Large
Sheep	< 50	50-70	>70
Cattle	< 500	500-700	> 700
Horses	< 300	300-600	> 600

Table 6 Livestock grazing units for different weights, adapted from (Martin et al., 2013).

	Livestock Unit		
	Small	Medium	Large
Ewe	0.08	0.1	0.1
Dairy cow	0.58	1	1.1
Suckler cow (incl. calf at foot)	0.7	0.9	1.1
Other cattle > 24 months	0.6	0.7	1
Weaned cattle < 24 months	0.5	0.6	0.7

Furthermore, stocking rates for upland areas are reactionary, based on habitat condition at the end of an agreement cycle (e.g. four year under Commonage Management Plan or GLAS scheme agreement). Blanket stocking rates are often applied, irrespective of habitat type of grazing behaviours, which may reflect a lack of evidence supporting more bespoke prescriptions.

Table 7 provides a summary of agri-environment schemes in Britain and Ireland.

Table 7 Habitats, typical plants, habitat codes per Fossitt (2000), National Vegetation Classification (NVC) and EU Habitats Directive codes (if applicable), with indicative stocking rates (SR) in Scotland (SCOT) (Farm Advisory Service, 2017), England (ENG) (Martin et al., 2013), Wales (Tubridy, 2013) and Ireland (IRL) (DAFM, 2015a)

Habitat	**Typical plants**	**Habitat Type**(Fossitt (2000), NVC, Hab.Dir.)	**SCOT**** (LU/ha)	**ENG** (LU/ha)	**Wales***** (LU/ha)	**NI**	**IRL** (LU/ha)
Blanket bog	*Calluna vulgaris*, *Sphagnum* sp., *Eriophorum* sp.	PB2, M17-19, 7130 (if active)	0.02 (0.00-0.05)	0.07	-	0.075	**ANC:** 0.15 for seven consecutive months. Overall rate of 0.15 maintained over 12 months. **Commonage Management**: 0-0.75 Deviation of +/- 20% to calculate min/max (all habitats)
Wet Heath	*C.vulgaris, Erica tetralix, Molinia*, *Carex* sp.	HH3, M15-16, 4010	0.08 (0.05-0.10) (Guidance given regarding timing but flexibility offered)	0.09	<0.4LU Apr-June <0.2LU July-Sept <0.1LU/Ha Oct-Mar. Never below 0.2LU Apr-Sept 0.05LU July-Sept.	0.25	**ANC:** 0.15 for seven consecutive months. Overall rate of 0.15 maintained over 12 months. **Commonage Management**: 0.75-1.0
Dry Heath	*C. vulgaris, Erica cinerea, Ulex gallii, Agrostis* sp., *Nardus* sp.	HH1, H9/H10/H12/H15, 4030	0.12 (0.10-0.2)	0.2	-	0.3 Mar-Oct	**ANC:** 0.15 for seven consecutive months. Overall rate of 0.15 maintained over 12 months. **Commonage Management**: 1.0-1.5
Bracken	*Pteridium aquilinium*	HD1, U20	0.00-0.10	N/A	-	-	-
Grasslands							
Poor	*Molinia/Nardus*	GS3/GS4, U5 M25	0.25 (0.2-0.4)	-	-	-	**ANC** 0.15 for seven consecutive months. Overall rate of 0.15 maintained over 12 months. **Commonage Management** 1.5-5.0
Moderate	*Festuca ovina, Agrostis capillaris*	GS3, U4 CG10	0.50 (0.4-0.6)	-	-	-	
Good	*Festuca rubra, Cynosurus cristatus*	MG3, MG5	0.7 (0.6-0.8)	0.2*	-	-	
Semi-improved	*Lolium perenne, Cynosurus cristatus*	MG6	0.8-1.0	0.2*	-	-	

*based on cross-compliance recommendation for calcareous grassland in (Martin et al., 2013). **agri-environment scheme plans can propose manipulation of stocking rates and offer support for shepherding and introduction of cattle. *** Min of 30% of LU to be cattle

2.1.5 Aim and Objectives

This chapter aims to explore the movement and behaviour of free ranging cattle in a HNV upland setting in order investigate how they use available space and habitats.

The objectives of the research were to:

1. To explore and elucidate the home range behaviour of free-ranging cattle in a 462-ha nature reserve in the uplands of southwest Ireland
2. To examine the differential use of various habitats within the home range of the cattle
3. To investigate the activity budgets of free-ranging cattle in upland and lowland settings
4. To calculate stocking densities using home range results and habitat data in order to inform prescriptions for sensitive upland habitats.

2.2 Methodology

2.2.1 Overview

In 2011 a five-year grazing plan for Mount Brandon Nature Reserve was agreed between the National Parks and Wildlife Service (NPWS) and Mr Paddy Fenton, an organic beef farmer in Ventry County Kerry. The Institute of Technology, Tralee, entered the agreement and carried out a baseline survey of vegetation and macroinvertebrates in the Reserve prior to commencement of the grazing. The dominant habitats in the Reserve are Blanket Bog, European Dry Heath, Northern Atlantic Wet Heath and Wet Grassland, and these were the focus of the grazing trials. Control plots (grazing exclosures) were created in 2011 by the NPWS and one was built in each of the four dominant habitats.

Thirty Dexter cattle, which included a small number of Dexter x Angus, ranged freely on the 462 ha site between July and October from 2011 to 2015 as per the grazing agreement. In keeping with the husbandry requirements of the grazier, the herd size varied occasionally. Four heifers were sold mid-way through the 2014 season and 8 calves were removed in September of 2015 for weaning. The maximum number of cattle on site at any time was 37 in 2014 and the minimum number on site was 29 during 2015. One calf died in 2012 and a cow died in 2015, both of unknown causes. A cow wearing a GPS collar was lost in 2014 and not recovered despite intensive searching of the study area and extensive searching of the surrounding lands and coastlines.

The cattle were left undisturbed during the study and it was a 'free-choice' system. Fieldworkers and the grazier checked on the animals several times each week to monitor health and condition. There were occasions when moving the cattle was necessary, for example they escaped from the study area on at least three occasions in 2014 and once in 2015.

A study on the home range and habitat selection of the cattle was conducted between 2013 and 2015 using GPS collars and a behavioural study using Direct Observation methods was carried out in 2014. The data from the GPS Tracking and the Direct Observation sampling were analysed to investigate the spatiotemporal aspects of how the cattle used the site.

2.2.2 Location

Mount Brandon Nature Reserve is a 462-hectare statutory nature reserve located 35 km west of Tralee town on the northern edge of the Dingle Peninsula in county Kerry in south west Ireland. The reserve was established in 1986 under Statutory Instrument No.420: (Nature Reserve (Mount Brandon) Establishment Order, 1986). The reserve makes up 3% of the Mount Brandon Candidate Special Area of Conservation (cSAC). Mount Brandon cSAC (Habitats Directive (92/43/EEC; site code 000375) is 14,355 hectares and is designated due to the presence of seven Annex I habitats. These include blanket bog, a priority habitat under the Directive, with northern Atlantic wet heath, alpine and boreal heath, vegetated sea cliffs, chasmophytic vegetation and nutrient-poor lakes (NPWS, 2009). The site has further designation under Annex II due to the presence of freshwater pearl mussel *Margaritifera margaritifera* and

Killarney fern *Trichomanes speciosum*. Más an Tiompán (763 m) is the highest peak in the reserve.

The underlying geology of the reserve is sandstone, conglomerates and siltstones of the Upper Devonian and Lower Carboniferous periods (Jackson, 1994; NPWS, 2009). The soils of the reserve are comprised of poorly drained peaty podzols with associated lithosols and blanket peat (NPWS, 2009). Sheep grazing predominated in the area until the Reserve was established in 1986 and since then it has been grazed by small numbers of sheep and by Kerry x Highland cattle during the late 1990s (Tim O' Donoghue, *pers. comm.*) but details on the number of animals and exact timing of the grazing are unclear. A herd of approximately 60 feral goats *Capra hircus* has been free-ranging in the Reserve since its establishment.

2.2.3 Home Range and Habitat Selection

Tellus Domestic tracking collars from FollowIt Wildlife (Lindesberg, Sweden) programmed to record a GPS location every 2 hours, 00:00-2200 Greenwich Mean Time (GMT) daily, were used to monitor the movements of cattle during the grazing seasons (July - October) in 2013, 2014 and 2015.

Figure 3). The GPS units were accurate to approximately 7.5m radius (Followit support, *pers. comm.*).

Three animals of suitable size, age and experience were chosen at the beginning of each grazing season for GPS sampling. Collars were fitted in early July, prior to turn-out. In GPS tracking studies the individual animal can be considered as 'the sample unit' (Kenward, 2001). In order to achieve as many independent replicates as possible, different animals were selected for collaring each year.

Figure 3 Selected images from collar programming, fitting and deployment.

GPS coordinates are stored on-board the collars and can be subsequently downloaded directly to computer once removed from the animal. They can also be downloaded *via* UHF in the field to a laptop. Two new collars were purchased in 2014 and these allowed access *via* a web portal provided by Followit as data were uploaded daily to remote servers.

GPS coordinates obtained from the collars were converted from Latitude/Longitude (LL) to Irish National Grid (IG) using the Transverse Mercator Calculator v2.0a (Morton, 2012), available at http://www.dmap.co.uk. If locations took longer than 90 seconds to be obtained, they were deemed invalid ('Time-Outs') and were eliminated from analysis. Locations that had a Dilution of Precision (DOP) of greater than 10 were also eliminated. DOP is a measure of satellite geometry and when satellites are close together while a GPS fix is being attempted, the DOP value will be high i.e. lower in accuracy (Adrados et al., 2002).

Ranges8 (Anatrack Ltd, Dorset, UK) was used to perform home range analyses. Minimum Convex Polygon (MCP) and Outlier Restricted Edge Polygon (OREP) home range estimates were calculated. ArcGIS 10.0 (ESRI, 2016) was used to produce maps. Habitat maps, orthographic photographs and 1:50 000 Ordinance Survey Ireland (OS*i*) maps were provided by the NPWS (OS*i* licence No. EN 0059212). MCPs are presented in the results, however, great emphasis should be placed on the OREP (and later the Kernel Density Estimates for utilisation scores) when considering the findings. MCPs are commonly reported, yet the other methods provide more realistic fits to the true movements of the animals and their home ranges.

Dynamic interaction analysis was also performed using Ranges8 to explore the cohesiveness of the collared animals. This routine gives an index of cohesiveness, showing the tendency of animals to avoid or adhere to each other (Kenward, 2001; Enright, 2012). The programme provides a single statistic, where values close to -1 indicate avoidance and values close to +1 indicate cohesiveness (Kenward et al., 2008).

Habitat availability and selection was studied using both compositional analysis (Aebischer, Robertson and Kenward, 1993) and Jacobs Index (Jacobs, 1974) available in Ranges8. Compositional analysis of selection ratios (% habitat used / % habitat available) was conducted using *Compos Analysis* v6.3 (Smith Ecology Ltd), an Excel add-in available from http://www.smithecology.com (licence no. CompAn-01-165A). Compositional analysis uses Multivariate analysis of variance (MANOVA) to analyse data represented as proportions and it can be used to determine statistical significance and rank order of differences between variables (Aebischer, Robertson and Kenward, 1993; Kauhala and Auttila, 2009).

The *Compos* Add-in tool for Excel implements the method of Compositional Analysis as described by (Aebischer, Robertson and Kenward, 1993). The Wilk's lambda and *t*-values are determined by randomisation tests as recommended by (Aebischer, Robertson and Kenward, 1993) to overcome lack of multivariate normality (Smith, 2005) and the default of 1000 iterations recommended by (Manly, 1997; Smith, 2005) to test at a significance level of 0.05 was accepted.

Jacobs Index, 'index D' in Jacobs (1974) is calculated using $D = (r-p)/[(r+p)-2rp]$, where r is the proportion of habitat used and p is the proportion available. D varies from -1 (avoidance) to +1 (reference), with D = 0 indicating habitat use in proportion to availability (Browne and Aebischer, 2003; Enright, 2012). Jacobs index was calculated for each animal at both the broad and detailed selection level.

A two-step approach to habitat selection was followed in the manner of Johnson (1980) and Aebischer, Robertson and Kenward (1993);

1. Compare habitat composition of home ranges (used) against overall availability in the study area (available) i.e. broad selection level
2. Compare habitat composition at GPS locations (used) against the various estimates of home range (available) i.e. detailed selection level.

2.2.4 Kernel Density Estimates for Utilisation scores

Kernel Density Estimation (KDE) (Worton, 1989b) was used to create Utilisation Scores (US) for the site. These were developed to examine the space used by the cattle within home range, and to score locations (i.e. sample plots/pitfall traps/quadrats) based on their position in the distribution. KDE is a contour method and it generates high-density centres based on contour lines (kernel isopleths). These lines indicate the probability of occurrence from 95%, 90%, 85% etc., down to 5%. Thus the highest US values are at the centre of the kernel and decreases towards the edges (Ziesemer and Meyburg, 2015).

KDEs using fixed kernel methods were calculated in RANGES8 (Kenward et al., 2008). KDEs, first described by Silverman (1986), are non-parametric models of utilisation distributions and produce density estimates based on Gaussian or compact kernels (Lyons, Turner and Getz, 2013). KDEs allow for multi-nuclear distributions to be developed over a matrix of intersections and then contours are interpolated between the intersections. A smoothing factor (h) is applied when creating the contours to define the search radius around each point, and this determines which points contribute to the density value at any given point. Fixed kernel methods were chosen as they are preferable to adaptive kernel methods which biases area estimates upwards (Worton, 1995; Seaman and Powell, 1996).

GPS data from all samples (all collars and all years, n = 9) were pooled to form one data set and this was imported into RANGES8. Locations falling outside the study area were deleted. KDEs at 5% intervals were calculated from 5% to 95% using fixed kernels, a reference smoothing parameter (*hRef*) of 4.7 and a smoothing multiplier of 1. *hRef* is the standard deviation of x and y coordinates divided by the sixth root of the sample size (Kenward et al., 2008). The matrix cell size was set to 15 m because the GPS units are accurate to approximately 7.5m radius (Followit support, *pers. comm.*).

KDEs were exported to ArcMap 10.0 (ESRI, 2016) as polyline shapefiles. Shapefiles of the reserve habitats, the plot and quadrat locations were added for analysis. The KDE contour that quadrats fell within was then recorded. Thus, Utilisation values were calculated for quadrats in the grazed plots. Values were subtracted from 100 to ensure that a low-to-high scale indicated low-to-high levels of utilisation. For example, quadrats falling in the 95% contour were furthest from the centre of the distribution, which indicated lower levels of grazing; this was converted to a score of 5%. Scores were averaged to give a plot-level utilisation score. Figure 4 shows an example for one plot (BB2).

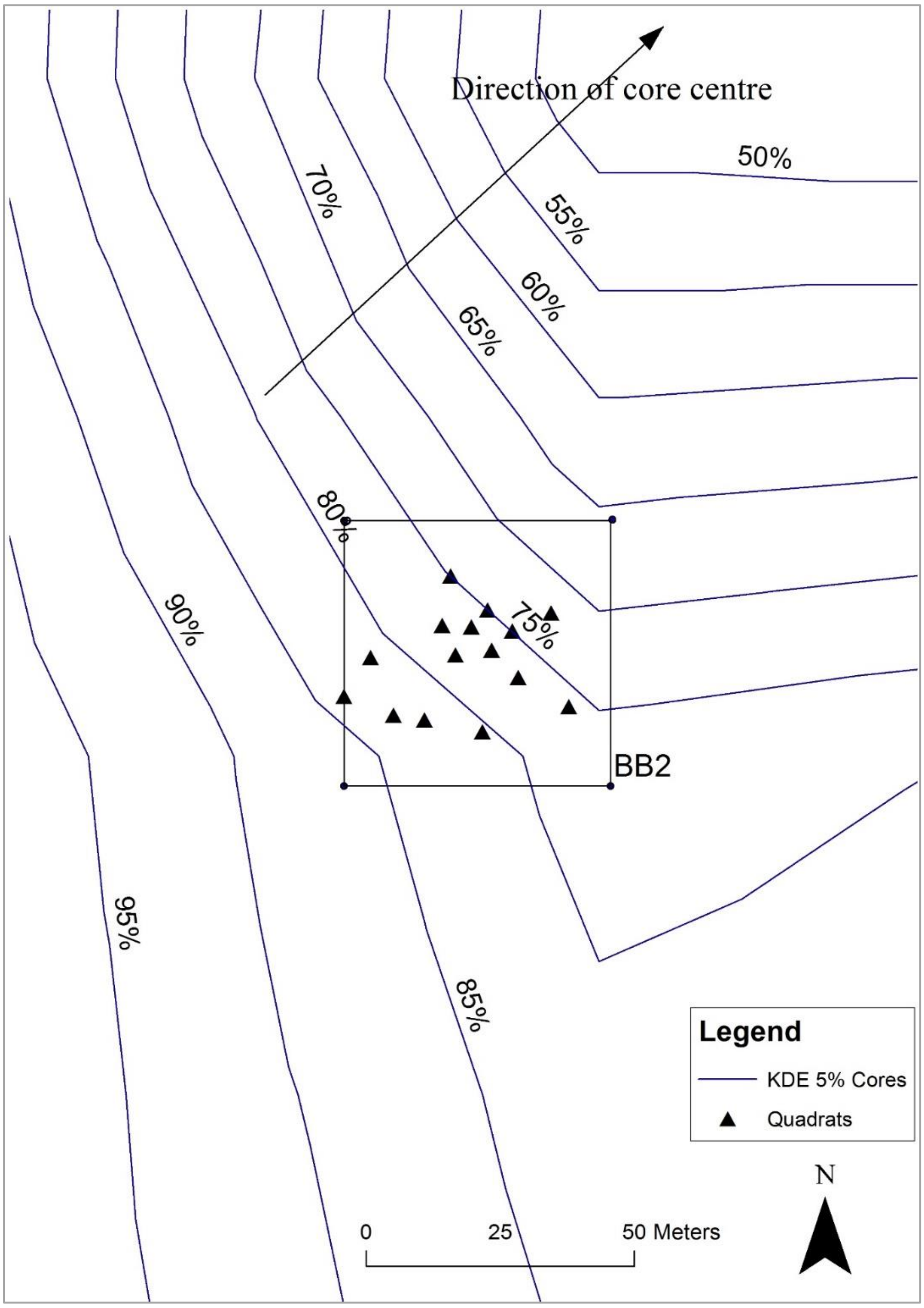


Figure 4 Example of the grazing scores for a blanket bog sample plot (BB2). 5-95% fixed kernels overlain on BB2 plot and quadrats in ArcMap. The quadrats in the middle of the plot fall into the 80% contour, resulting in a grazing score of 20 (low-moderate).

2.2.5 Direct Observation Study

Study areas

This study took place in two contrasting areas, Mt Brandon Nature Reserve as previously described and a lowland (10m ASL) 21 ha grass-fed organic cattle farm approximately 25 km south of the reserve near the village of Ventry (IG V 40969 98603). The farm is owned and managed by the same grazier (Mr Paddy Fenton) and holds a herd of 80 – 100 Dexter cattle, with additional small numbers of Aubrac, Angus and Highland cattle. The soils of the farm are loamy drift with siliceous stones, which sit on rhythmically bedded sandstone called the Ballymore Sandstone Formation (GSI, 2008). In 2014 the area received 1684 mm of rain and the mean monthly temperature was 10.8°C (Met Éireann, 2014).

Data Collection and Analysis

Prior to commencement of the sampling, 8 hours of preliminary observation was conducted at each site to assess the range of behaviours displayed by the animals. All activities were recorded, grouped into 6 categories and an ethogram was constructed (Table 8). Focal animal sampling in 15 min continuous recording blocks with intervening scan sampling *sensu* Altmann (1974) was used to collect behavioural data using the 6 behaviour categories identified. Sampling was carried out one day per week at each site during August-October in 2014.

For the focal animal sampling, one adult animal was randomly selected from the herd for observation on each of the sampling days. The focal animal was observed for 4 hours each day, with Opticron HR WP 8 x 42 binoculars and/or an Opticron HR 80 GA EED/45 field scope where necessary. Sampling intervals were 15 minutes in duration, with data recorded during the 1st and 3th quarter of every hour. The 15min sampling blocks were divided into 30 second periods and the dominant activity during each period was recorded. For the upland site the dominant habitat type was also noted.

For scan sampling, the whole herd was scanned twice per hour, immediately after the 15min focal sampling blocks. The number of animals engaged in each of the 6 behaviour categories was recorded.

Sampling days were divided into three 4-hour blocks (07:00-11:00, 11:00-15:00 and 15:00-19:00) with sampling conducted during one block each week. The sampling was then rotated through blocks from week to week i.e. on week no.1 sampling was carried out from 07:00-11:00 at each site, on week no.2 from 11:00-1500 and on week no.3 from 15:00-19:00. This continued until the end of October while day length allowed. Sampling was not conducted outside of daylight hours for health and safety reasons.

Table 8 Descriptive ethogram of observed cattle behaviours

Behaviour Category	Behaviour Type	Description of behaviour
Grazing	Maintenance	Head consistently down at feeding station. Biting or moving to take next bite.
Walking	Maintenance	Locomotion with head consistently raised for 3 seconds.
Standing	Maintenance	In a standing position, with head up for more than 3 seconds and not chewing the cud.
Lying	Maintenance	Lying down at rest, with head up or on the ground and not chewing the cud.
Interacting	Social/self-expression.	Engaged in social behaviour with another animal i.e. licking/grooming, vocalising, head-butting and pushing, suckling.
Ruminating	Maintenance	Chewing the cud, either lying or standing without any other activity.
Other		Animal scratching or licking itself, scratching against an object (post/gate/rock etc.), drinking, investigating new objects, comfort behaviour.

Focal animal sampling data were analysed using SPSS Statistics 22 (IBM Corp., 2013). Activity budgets were described for the upland and lowland cattle, expressing behaviours as a percentage of time observed and independent samples Mann-Whitney *(U)* tests were used to compare budgets.

For the scan sampling, data on the number of cattle engaged in each activity under examination was expressed as proportion number visible at time of observation. Independent samples Mann-Whitney *(U)* tests were used to test for differences between groups.

2.2.6 Stocking Rates

Outlier Restricted Edge Polygon (OREP) home range estimates, habitat composition of home range, and number of locations per home range were calculated using Ranges8 (Anatrack Ltd, Dorset, UK). Data were averaged from all samples across all years (n = 9). GPS tracking and home range analysis are described in section 2.2.3 and results are presented in section 2.3.1. Stocking rates were calculated by:

LU/(HR*%nLocs), where;

LU = Livestock Units,

HR = Home Range,

%nLocs = proportion of GPS locations that occurred in each habitat

The Livestock Units are based on 2014 cattle numbers; 15 Dexter cows with 15 followers and 10 steers (= 30 LU). Livestock Units were adjusted for the small size of Dexter cattle by applying coefficients for 'small cows' in Martin et al., (2013) i.e. small (< 500 kg) suckler cow (incl. calf at foot) = 0.7 LU (Table 9 & 10). Mean weight of cattle = 305.98 kg (weights based on July 2014 cattle data from Dineen (2016)).

Table 9 Animal weight categories (RDS/Natural England), adapted from (Martin et al., 2013).

	Liveweight (kg)		
	Small	Medium	Large
Sheep	< 50	50-70	>70
Cattle	< 500	500-700	> 700
Horses	< 300	300-600	> 600

Table 10 Livestock grazing units for different weights, adapted from (Martin et al., 2013).

	Livestock Unit		
	Small	Medium	Large
Ewe	0.08	0.1	0.1
Dairy cow	0.58	1	1.1
Suckler cow (incl. calf at foot)	0.7	0.9	1.1
Other cattle > 24 months	0.6	0.7	1
Weaned cattle < 24 months	0.5	0.6	0.7

2.3 Results

2.3.1 Home Range

MCPs and OREP estimates are presented below for completeness. However, a greater emphasis should be placed on the OREPs (and KDEs) than on MCPs when considering the findings. MCPs are commonly reported in the literature in animal movement studies, yet the other methods provide a better fit to the true movements of the animals and their home ranges.

Between 2013 and 2015 nine cattle were tracked in Mt Brandon Nature Reserve. In total, 12, 572 positions were obtained with high fix success rates of 85–98%. Cattle travelled a mean distance of 144 km (s = 20.2 km, N = 9) per season, with a mean daily distance of 1.4 km (s = 0.3 km, N = 9). Using OREP estimation, mean home range size was 122.7 ha (s = 21.0 ha, N = 9) and 177 ha (s = 38.9 ha, N = 9) using MCP. Table 11 and Figure 5 summarise the results and show that home range sizes varied from month to month, being largest in September and smallest in October. The cattle grazed at a mean altitude of 259 m ASL (s = 99.3 m, N = 9) and the mean temperature inside the collar units was 16° C (s = 3°C, N = 9). The temperature inside the unit is affected by the ambient temperature and the body temperature of the animal (Followit, 2012).

The Tellus GPS collars detect activity on two axes, with x-axis movement corresponding to the animal moving its head up and down ('nodding') and y-axis movement corresponding to the animal shaking its head (Followit, 2012). The units record any such activity for the length of time it takes to obtain a fix ('Time-to-Fix'), thus the proportion of 'active time' on each axis can be calculated. Activity was recorded on the x-axis (head nodding) 47% of the total time and on the y-axis 58% of the time (head shaking).

Data were split between day (0800 hrs to 22:00 hrs) and night (22:00 hrs to 08:00 hrs). Analysis revealed that the cattle were moderately less active at night, with x-axis movement detected 40% of the time and y-axis movement 51% of the time, compared with 53% and 65% respectively during the day, however this was not statistically tested.

Table 11 Descriptive statistics for MCP and OREP estimates. Full grazing season and monthly home range sizes for Dexter cattle in Mt Brandon Nature Reserve are given.

	Full season OREP | MCP (ha)	July OREP | MCP (ha)	Aug OREP | MCP (ha)	Sept OREP | MCP (ha)	Oct OREP | MCP (ha)
Mean	122.7 | 177.6	21.8 | 101.2	69.0 | 131.9	90.3 | 121.0	43.5 | 85.8
S.D.	21.0 | 38.9	13.9 | 37.1	16.7 | 28.3	41.8 | 57.5	42.9 | 60.8
Min.	89.6 | 113	0.5 | 46.9	42.6 | 82.6	24.2 | 43.4	3.9 | 7.2
Max.	148.2 | 211	35.9 | 138.0	84.9 | 164.0	143.0 | 186.0	128.5 | 150
N	9	8	8	9	8

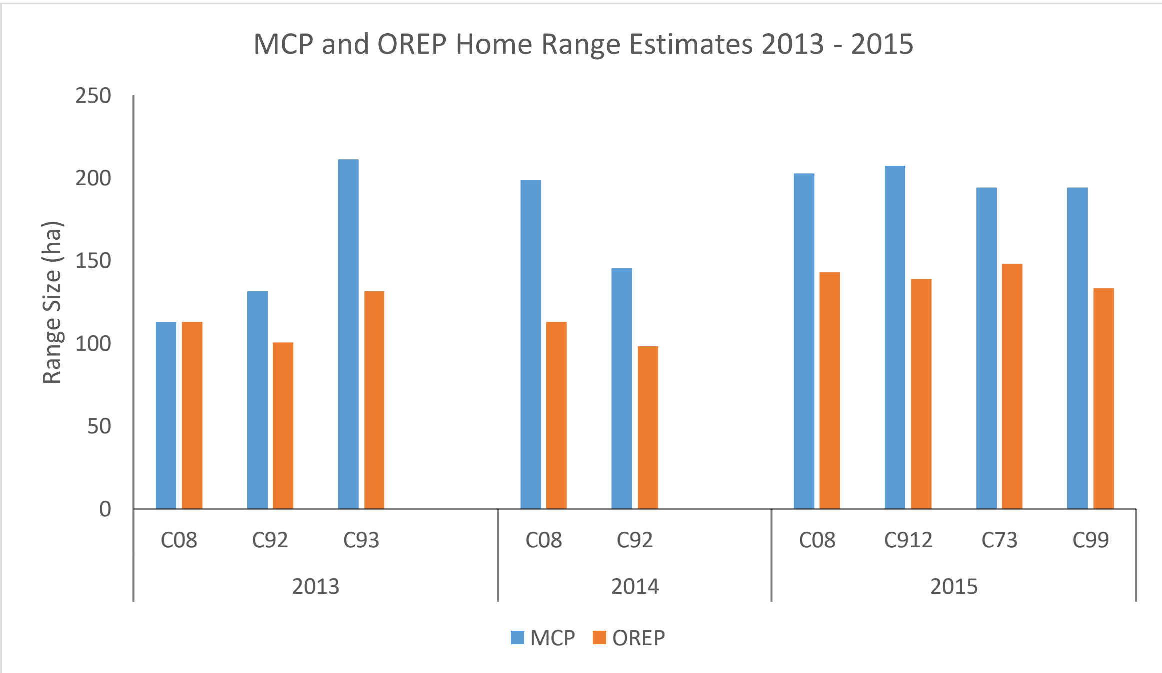


Figure 5 Mean home range of Dexter cattle in Mt Brandon Nature Reserve for full year and by month (N = 9).

In 2013 three Dexter cattle were fitted with GPS collars and released into the study area (SA) on July 14th. The collars recorded data over a total of 280 days, resulting in 3658 GPS positions. Two out of the three cattle were tracked for the full grazing season (July 14th to October 31st). One collar was out of commission from July 19th to August 29th (41 days), due to the animal getting a leg stuck in the collar. The collar was damaged, had to be repaired and was later fitted to another animal and redeployed. Mean home range size in 2013 was 152 ha using MCP (s = 52, N = 3) and 115 ha using OREP (s = 15, N = 3).

In 2014 three different cattle were fitted with the GPS collars and turned out into the SA on July 12th. One of these animals was lost during the month of October and was never found. The technology in use was such that the data from the collar were not recoverable, thus only two samples were available for analysis from 2014. These two collars recorded data over a total of 222 days and 3830 attempted fixes, 86% of which were successful. The average home range size was 172 ha using MCP (s = 37 ha, N = 2) and 105 ha (s = 10, N = 2) with OREP.

In 2015 the cattle were released into the SA on July 18th with four animals fitted with collars. They were on site until October 26th and the units recorded data over a total of 420 days which resulted in 5084 attempted fixes, 97% of which were successful. Home range size was 199 ha (s = 6.5 ha, N = 4) using MPC and 140 ha using OREP (s = 6.3 ha, N = 4).

Visual assessment of home ranges indicated that they overlapped. Dynamic interaction analysis available in Ranges8 compares the observed mean distance between dyads (pairs of coordinates) with the expected mean and results are combined with Jacobs index (1974) to give an unbiased interaction coefficient, where -1.0 indicates avoidance and +0.1 indicates cohesion (Kenward et al., 2008). The results of dynamic interaction and two-dimensional overlap analysis in Ranges8 showed that there was an 82% overlap across all ranges (s = 9%, N = 9). Within years the cohesion index was D = 0.98 (2013), D = 0.98 (2014) and D = 0.99 (2015), demonstrating that the collared animals were in the same place at the same time.

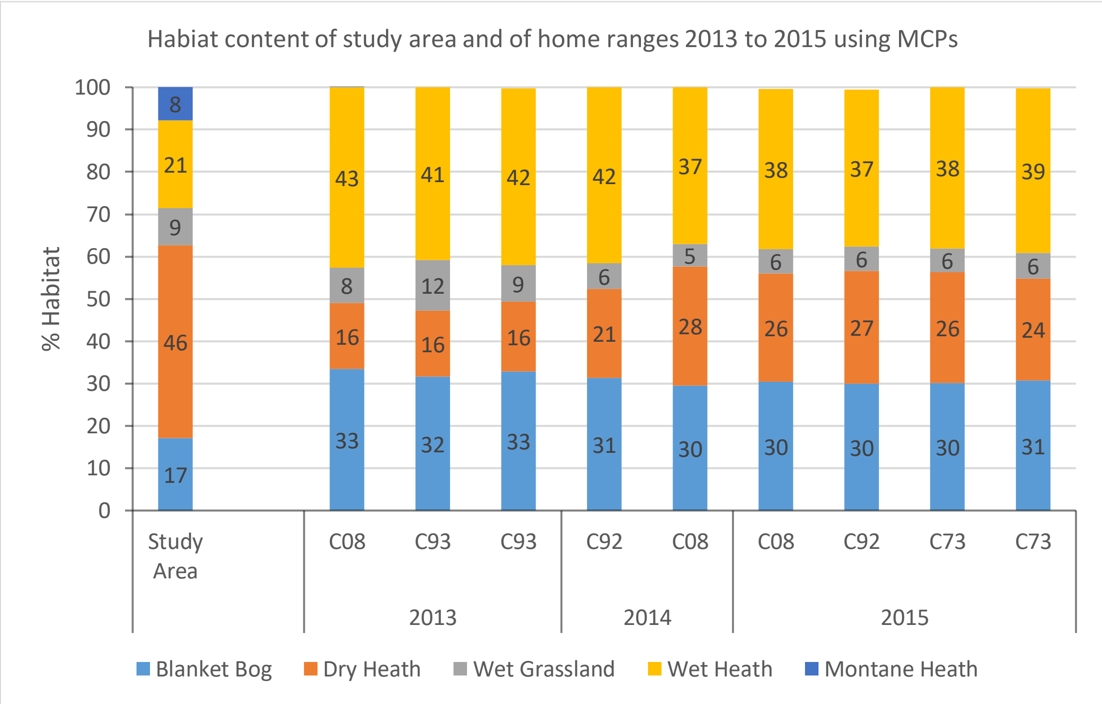
2.3.2 Habitat Selection

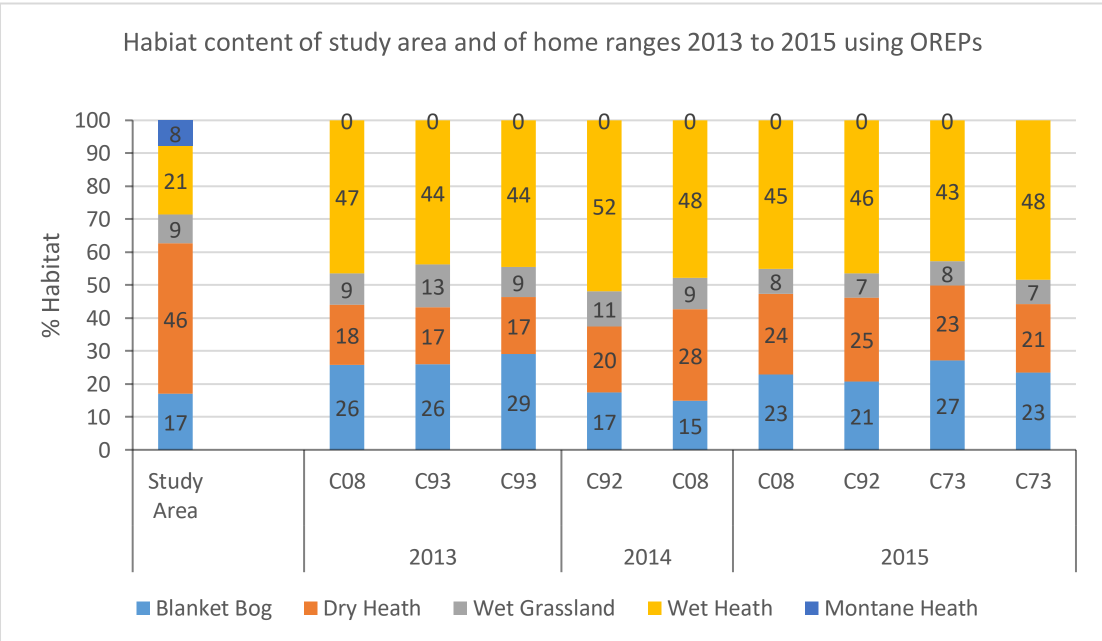
The habitat content of the study area (SA) (the 462-ha reserve), and of each home range was assessed to measure the proportions available to the herd, the proportions in home ranges (broad selection level) and proportions used at locations (detailed selection level). For the purposes of these analyses certain habitats were excluded; sea cliffs, siliceous alpine and boreal grasslands, alkaline fens, siliceous scree and rocky slopes. These either make up a minor proportion of the site (all combined = 3.4%) and/or they are unavailable to cattle (e.g. steep scree slopes and cliffs). Habitats that are rarely visited may need to be omitted from compositional analysis as too many zeros prohibits analysis (Smith, 2005; Williams et al., 2012a).

Habitat availability and selection within home range are presented in (Figures 6-9), along with maps of habitat use. Table 12 & 14 and Figure 11 Habitat selection by cattle in Mt Brandon Nature Reserve 2013 – 2015 using Jacobs' index (Jacobs, 1974). present data on habitat selection preferences detailed here.

Figure 6 shows the habitat proportions within the study area and those within the home ranges. Using both methods of estimation wet heath made up the largest proportion of the home ranges (MCP $\bar{x}$ = 45%, s = 2%, N = 9; OREP $\bar{x}$ = 46%, s = 3%, N = 9). Blanket bog made up 31% using MCP estimation (s = 1%, N = 9) and 23% using OREP (s = 4%, N = 9) and dry heath was 22% using both estimates (s = 5% for MCP and s = 4% for OREP and N = 9). Wet grassland made up 7% of the home ranges using MCP (s = 2%, N = 9) and 9% using OREP (s = 2%, N = 9). Figure 7–9 show the home range estimates and GPS positions overlaid on orthophotos and habitat maps.

Figure 6 Habitat content of Study Area (462 ha) and of home range 2013 to 2015 using MCP and OREP estimation.





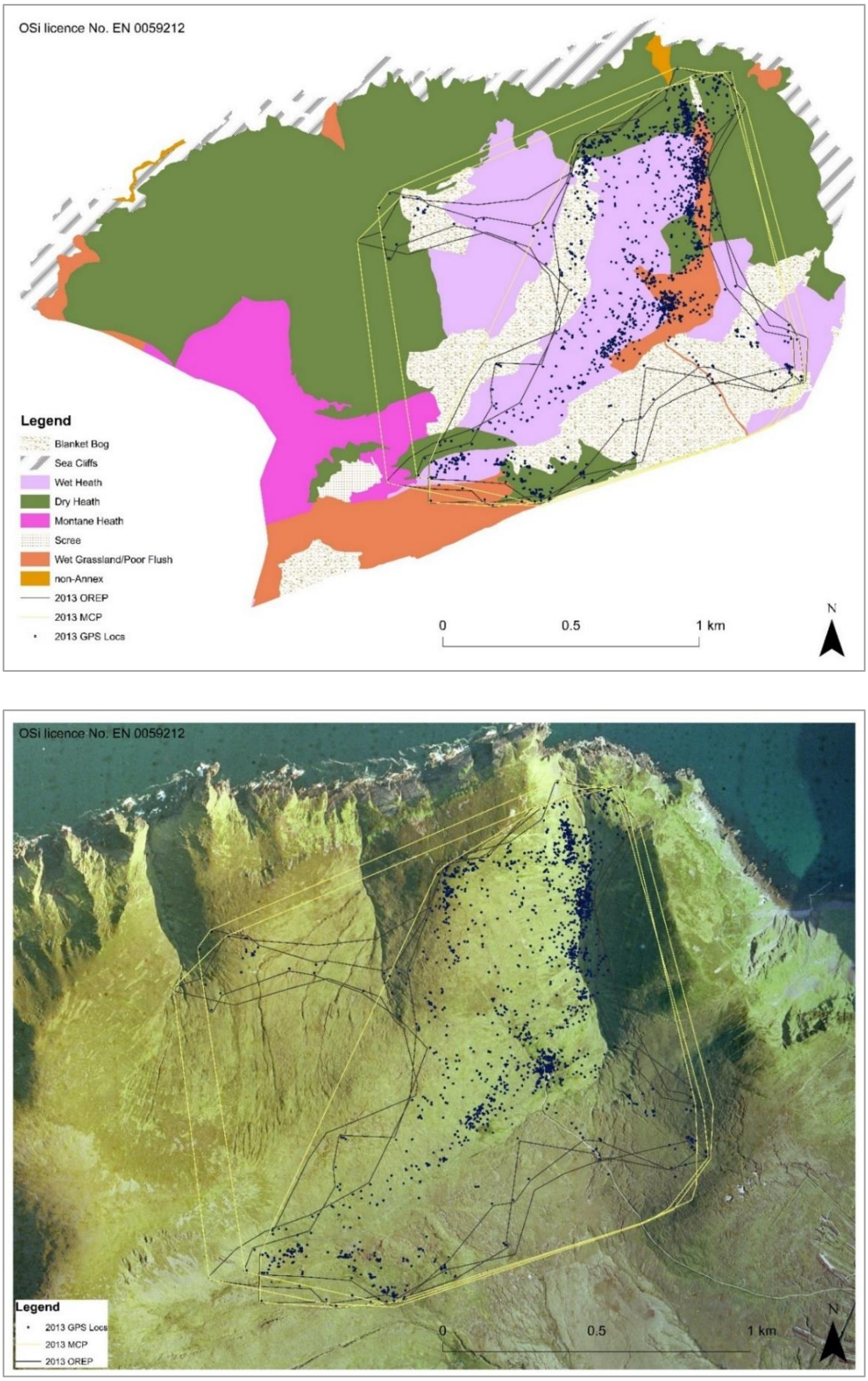


Figure 7 MCP and OREP home range estimates for Dexter cattle in 2013 on Mt Brandon habitat map and on OS*i* a2005 orthophotos

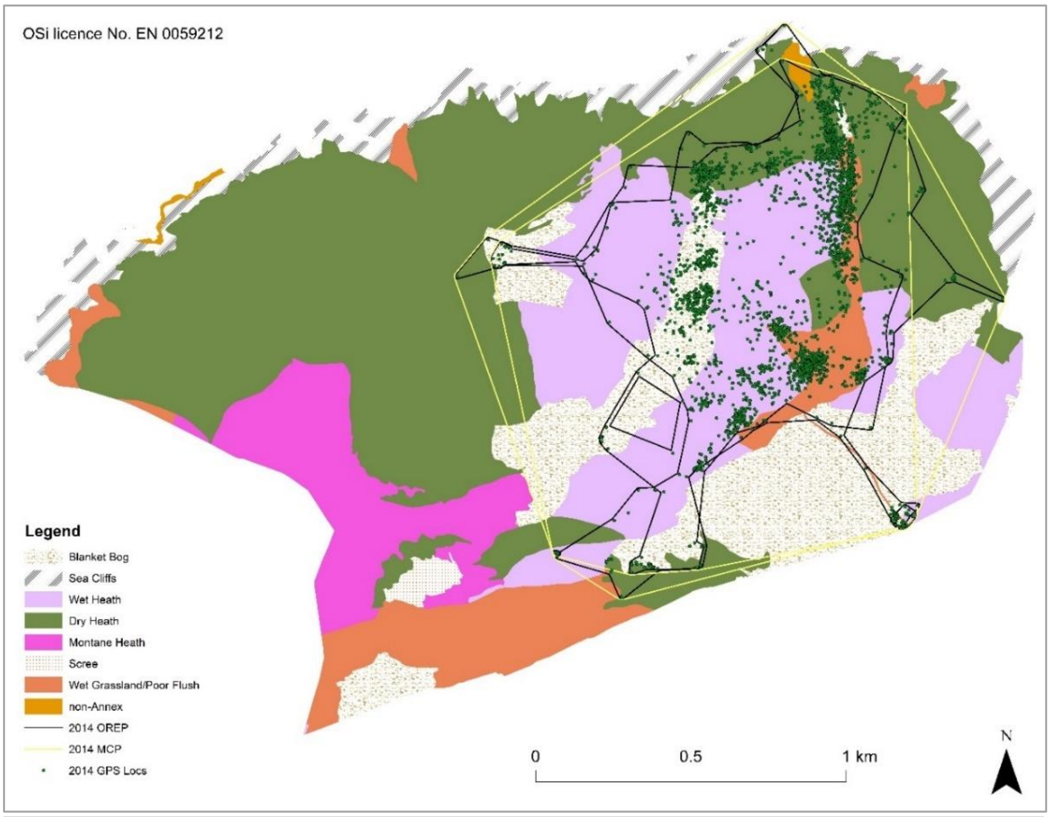


Figure 8 MCP and OREP home range estimates for Dexter cattle in 2014 on the Mt Brandon habitat map and on OS*i* a2005 orthophotos.

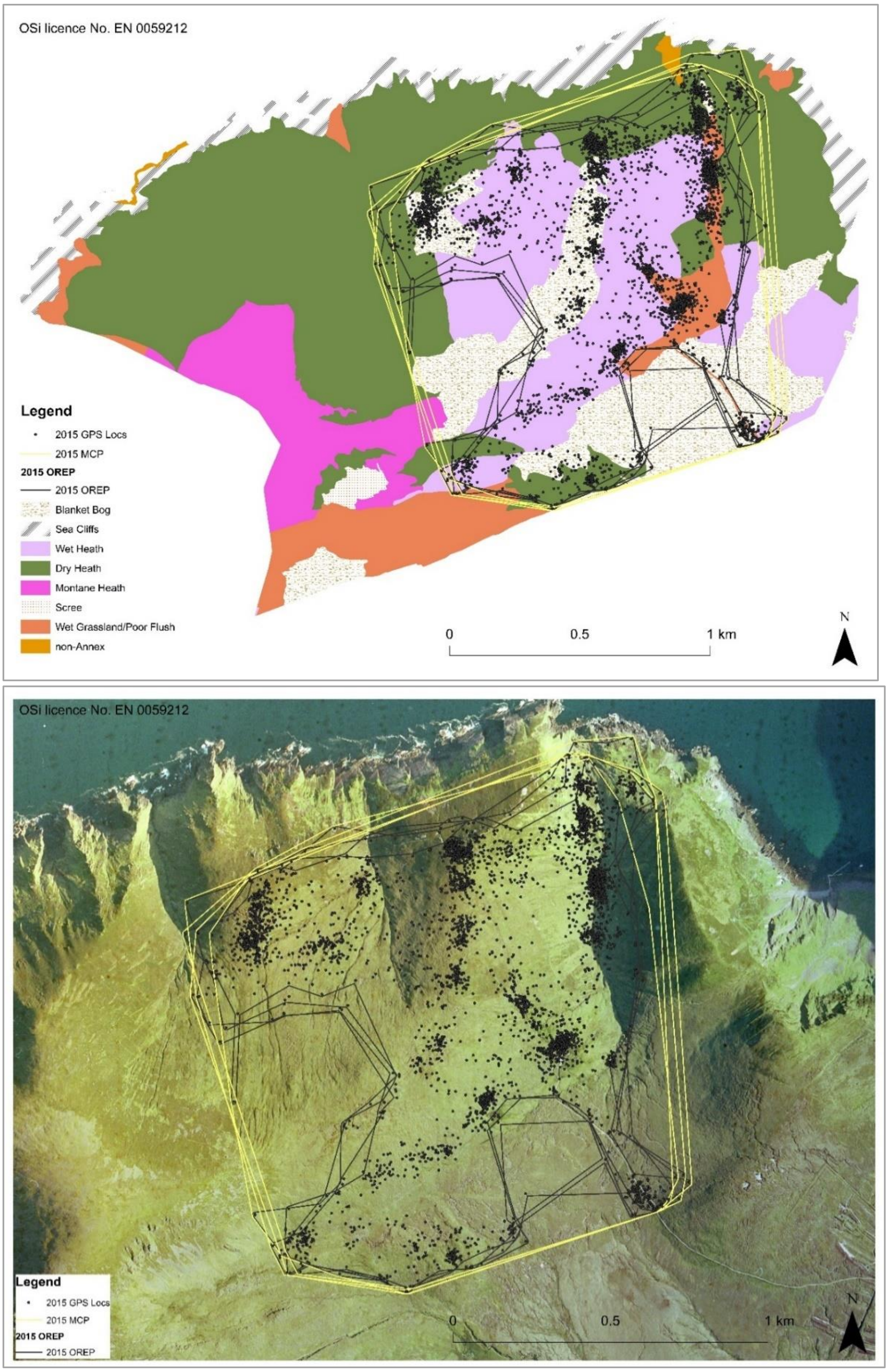


Figure 9 MCP and OREP home range estimates for Dexter cattle in 2015 on the Mt Brandon habitat map and on OS*i* a2005 orthophotos

Wet grassland accounted for 8% of the study area (Figure 10) and the cattle showed a preference for this habitat type, with 9% used in home range (OREP) and 21% used at location. Dry heath made up the largest proportion of the study of the study area (46%) and cattle showed avoidance at a broad selection level with 22% used in the home range. However, at a more detailed selection level cattle showed preference for dry heath, with 28% used at location. Wet heath made up 21% of the study area and the cattle showed preference for it at a broad selection level, with 45% used in home range. At a detailed selection level, the cattle showed slight avoidance for wet heath, with 33% used at location. Blanket bog made up 17% of the study area and 23% of the cattle's home range, indicating some selection at a broad level. Cattle showed moderate levels of avoid avoidance for blanket bog, with 17% used at location. Table 13 shows the proportions of GPS locations in each habitat type.

Habitat selection based on Jacobs' index (Table 12 and Figure 11) showed that the cattle had most preference for wet grassland with D = 0.5 (s = 0.1, N = 9) and the least preference for wet heath with D = -0.3 (s = 0.1, N = 9). Blanket bog was avoided ($\bar{x}$ D = -0.2, s = 0.2, N = 9) and dry heath was used in proportion to its availability ($\bar{x}$ D = 0.1, s = 0.1, N = 9).

Table 12 Habitat preference/avoidance of Dexter cattle 2013 - 2015 using Jacobs Index. Values of -1 indicate complete avoidance and +1 indicates exclusive use. Values close to zero indicate that a habitat was used in proportion to its availability (A = Avoid, P = Prefer)

Year	Collar	Blanket Bog	Dry Heath	Wet grassland	Wet Heath
2013	C08	-0.43 (A)	0.16 (P)	0.43 (P)	-0.11 (A)
	C92	-0.61 (A)	0.29 (P)	0.40 (P)	-0.32 (A)
	C93	-0.41 (A)	0.21 (P)	0.40 (P)	-0.09 (A)
2014	C08	-0.06 (A)	0.06 (P)	0.40 (P)	-0.54 (A)
	C92	-0.08 (A)	- 0.05 (A)	0.47 (P)	-0.42 (A)
2015	C08	-0.06 (A)	0.15 (P)	0.48 (P)	-0.32 (A)
	C92	-0.03 (A)	0.13 (P)	0.47 (P)	-0.31 (A)
	C73	-0.18 (A)	0.18 (P)	0.55 (P)	-0.32 (A)
	C99	0.08 (P)	0.10 (P)	0.38 (P)	-0.30 (A)
Mean across years		x = -0.2 ± 0.23 Avoid	x = 0.14 ± 0.10 Prefer	0.44 ± 0.06 Prefer	x = - 0.30 ± 0.14 Avoid

Table 13 Proportions of GPS locations in different habitats

Year	ID	Blanket Bog (%)	Dry Heath (%)	Wet Grassland (%)	Wet Heath (%)	N-locs
2013	C_08	12	25	22	42	1199
	C_93	9	31	27	32	719
	C_92	16	25	19	40	1216
2014	C_92	19	27	25	30	1927
	C_08	14	29	25	31	1300
2015	C_08	21	31	19	30	1271
	C_92	20	31	18	31	1366
	C_73	21	30	22	28	1357
	C_99	27	24	15	34	826
	$\bar{x}$	**17%** **(s = 5.0)**	**28%** **(s = 2.7)**	**21%** **(s = 3.6)**	**33%** **(s = 4.4)**	

Compositional analysis of habitat use (Table 14 Compositional analysis of habitat use) showed that at a broad level the cattle selected wet heath and blanket bog over wet grassland and dry heath with variables ranked from most to least preferred: *Wet Heath>>>Blanket Bog>>>Wet Grassland>>>Dry Heath>>>Montane Heath* ('>>>' denotes a significant difference between two consecutively ranked variables). At a detailed selection level wet grassland was most preferred, with the variables ranked as *Wet Grassland>>>Dry Heath>>>Blanket Bog>Wet Heath*. There was no montane heath in the home ranges so it was not included in analysis at a detailed selection level.

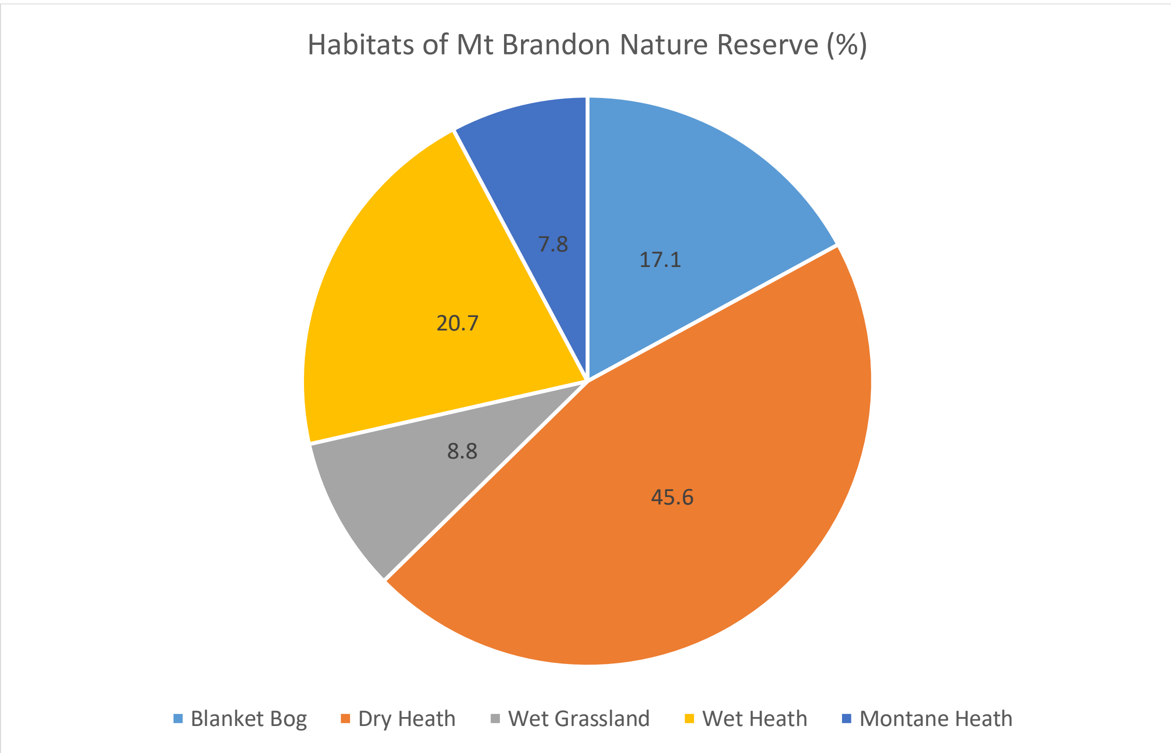


Figure 10 Habitat content of MBNR (data source: NPWS National Survey of Uplands Habitats, 2014).

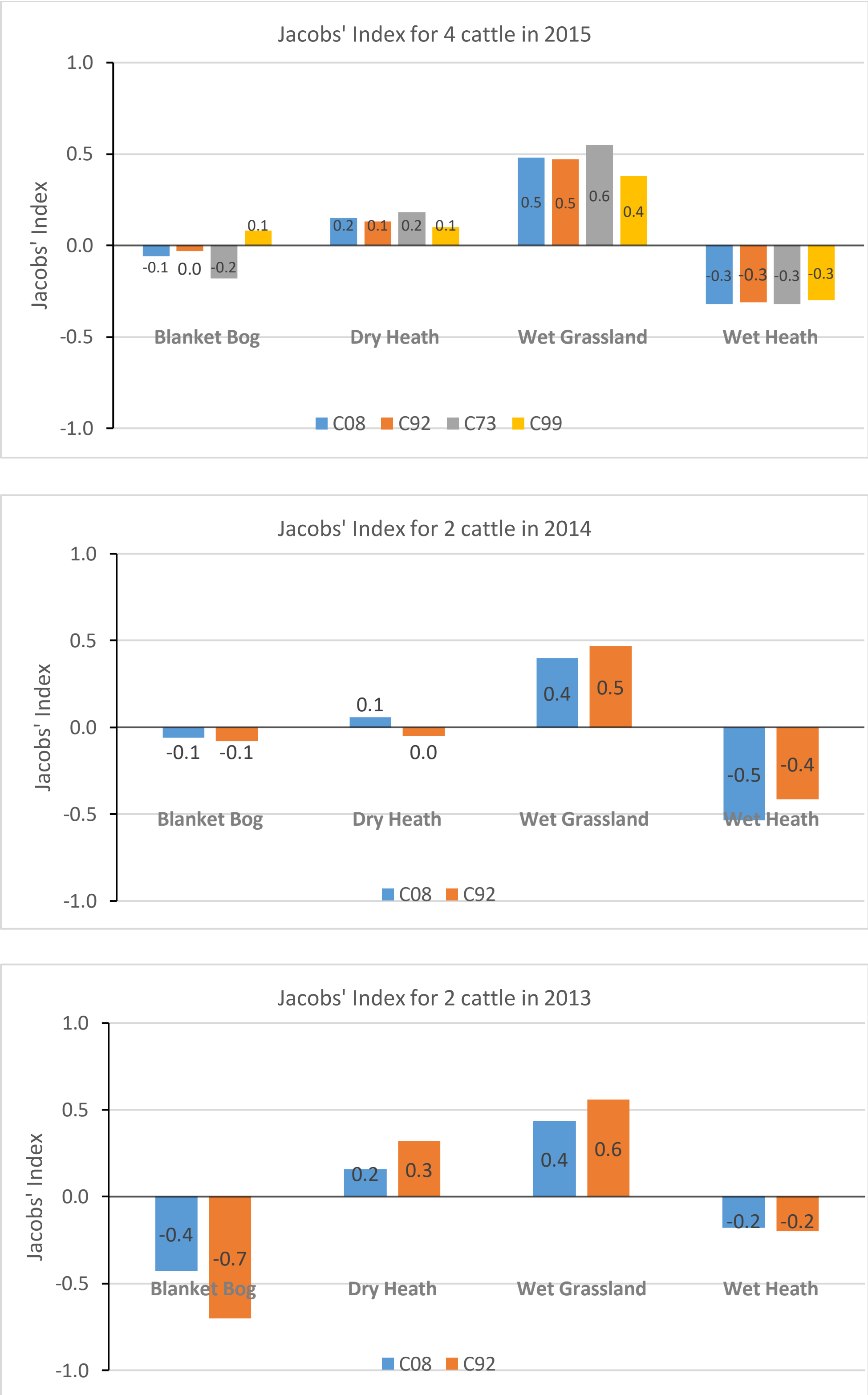


Figure 11 Habitat selection by cattle in Mt Brandon Nature Reserve 2013 – 2015 using Jacobs' index (Jacobs, 1974).

Table 14 Compositional analysis of habitat use

Multivariate (Wilk's lambda) MANOVA results.

1. **OREP Home Range (Used) vs. Study Area (462 ha) (available):**

Multivariate (Wilk's lambda) test: Lambda = 0.0000, Chi sq. = 127.37, d.f. = 4, p = <0.0001.

Matrix of means and standard errors					
	Blanket Bog	Dry Heath	Wet Grassland	Wet Heath	Mont. Heath
Blanket Bog		1.038 ± 0.120	0.268 ± 0.102	-0.527 ± 0.089	4.634 ± 0.073
Dry Heath	-1.038 ± 0.120		-0.771 ± 0.108	-1.565 ± 0.058	3.596 ± 0.058
Wet G'land	-0.268 ± 0.102	0.771 ± 0.108		-0.795 ± 0.066	4.366 ± 0.065
Wet Heath	0.527 ± 0.089	1.565 ± 0.058	0.795 ± 0.066		5.161 ± 0.020
Mont. Heath	-4.634 ± 0.073	-3.596 ± 0.058	-4.366 ± 0.065	-5.161 ± 0.020	

Ranked variable sequence (most to least used):

Wet Heath>>>Blanket Bog>>>Wet Grassland>>>Dry Heath>>>Mont. Heath

(>>> denotes a significant difference between two consecutively ranked variables)

Simplified ranking matrix						
	Blanket Bog	Dry Heath	Wet Grassland	Wet Heath	Mont. Heath	Rank
Blanket Bog		+++	+++	---	+++	3
Dry Heath	---		---	---	+++	1
Wet Grassland	---	+++		---	+++	2
Wet Heath	+++	+++	+++		+++	4
Mont. Heath	---	---	---	---		0

2. **Used at Loc. (used) vs. OREP Home Range (available):**

Multivariate (Wilk's lambda) test: Lambda = 0.0156, Chi sq. = 37.456, d.f. = 3, p = <0.0001.

Matrix of means and standard errors				
	Blanket Bog	**Dry Heath**	**Wet Grassland**	**Wet Heath**
Blanket Bog		-0.567 ± 0.184	-1.165 ± 0.130	0.057 ± 0.174
Dry Heath	0.567 ± 0.184		-0.597 ± 0.077	0.624 ± 0.063
Wet Grassland	1.165 ± 0.130	0.597 ± 0.077		1.222 ± 0.078
Wet Heath	-0.057 ± 0.174	-0.624 ± 0.063	-1.222 ± 0.078	

Ranked variable sequence (most to least used):

Wet Grassland>>>Dry Heath>>>Blanket Bog>Wet Heath

Simplified ranking matrix					
	Blanket Bog	**Dry Heath**	**Wet Grassland**	**Wet Heath**	**Rank**
Blanket Bog		---	---	+	1
Dry Heath	+++		---	+++	2
Wet Grassland	+++	+++		+++	3
Wet Heath	-	---	---		0

2.3.3 Activity Budgets of Upland and Lowland Cattle

The activity budgets of upland and lowland cattle were compared. The individual animal is the sample unit and observational data were pooled by day. Twelve sampling days were completed between August and October at each site. A second observer was available on several days (undergraduate student assigned to the project) and on these occasions extra samples were collected i.e. a second animal was randomly chosen for observation and samples were collected simultaneously. The sampling period was cancelled if an animal went out of view and the 3-7pm sessions after October 7th were shortened by 1 hour due to loss of daylight hours. Consequently, for focal animal sampling N=14 animals in the uplands and N=18 in the lowlands and for scan sampling N=12 for both study areas.

The upland cattle spent proportionately more time Grazing ($\bar{x}$ = 63% ± 21%) than the lowland cattle ($\bar{x}$= 50% ± 16%), as shown in Table 15 and Figure 12. To test the hypothesis that upland and lowland cattle were associated with significantly different grazing times, an independent samples Mann-Whitney *U* test was conducted. The test was associated with a statistically significant effect (U = 73.000, z = -2.013, p = 0.044, r = -0.35). The upland cattle spent proportionately more time walking ($\bar{x}$ = 6% ± 7%) than the lowland herd ($\bar{x}$ = 3% ± 4%), however no statistically significant effect was detected.

The lowland cattle were associated with more time spent ruminating (25% ± 13%) than the upland cattle (19% ± 18%). The lowland herd was also associated with more time spent standing, lying, interacting and engaged in 'other' activities than the upland cattle (Table 15) but no significant effect was found when tested.

Table 15 Focal Animal sampling of activity: time spent by upland and lowland cattle on 6 activities (mean % time, ± standard deviation).

Activity	Upland (*N* = 14)	Lowland (*N* = 18)
Grazing	63 (21)	50 (16)
Walking	6 (8)	3 (4)
Standing	4 (3)	7 (6)
Lying	4 (7)	9 (10)
Interacting	1 (3)	3 (4)
Other	2 (2)	3 (5)

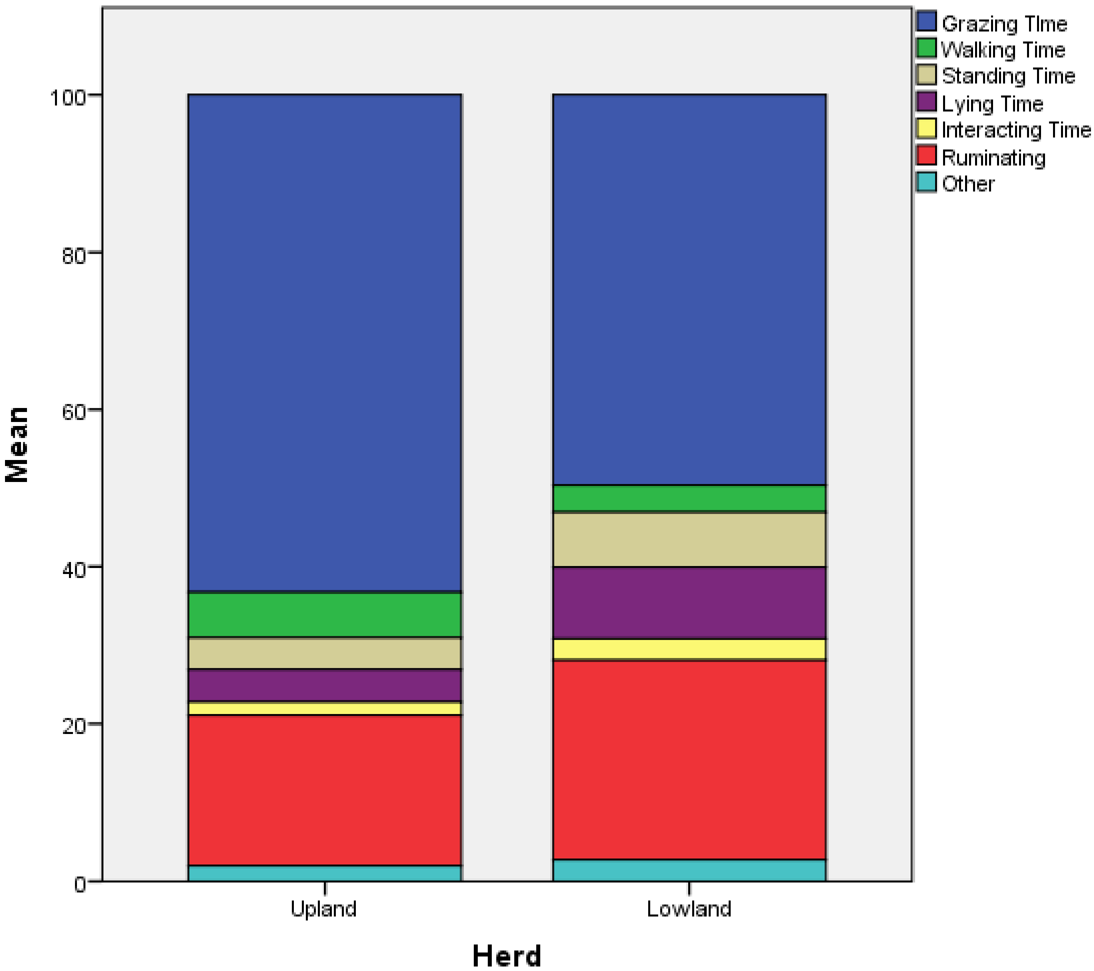


Figure 12 Comparison of time budgets of upland and lowland cattle in an upland and lowland herd of cattle.

During scan sampling the number of cattle ruminating was not recorded as it was not practical to assess every individual in the herd within the 15min window between focal animals' samples. The upland cattle were widely spaced and the topography of the landscape made it impractical for the observer to move around to examine every individual. The large numbers (55+) in the lowland herd made it impractical due to time constraints.

The results in Table 16 Scan sampling of activity: the mean percentage of animals engaged in each activity by upland and lowland cattle (mean number of cattle ± standard deviation). show that a higher mean proportion of the upland herd was observed grazing, walking and standing than the lowland herd, whereas the lowland herd was associated with higher proportions of animals lying, interacting and engaged in 'Other' activities. To test the hypotheses that the upland and lowland cattle were associated with statistically different proportions engaged in each of the activity categories, Mann-Whitney *U* tests were conducted but no significant effect was detected. Figure 13 shows the proportions of activities the upland and lowland herd were activities.

Table 16 Scan sampling of activity: the mean percentage of animals engaged in each activity by upland and lowland cattle (mean number of cattle ± standard deviation).

Activity	Upland (N = 12)	Lowland (N = 12)
Grazing	58 (24)	53 (22)
Walking	5 (11)	3 (4)
Standing	17 (13)	12 (9)
Lying	17 (17)	28 (18)
Interacting	1 (2)	1 (3)
Other	0.5 (1.4)	3 (6)

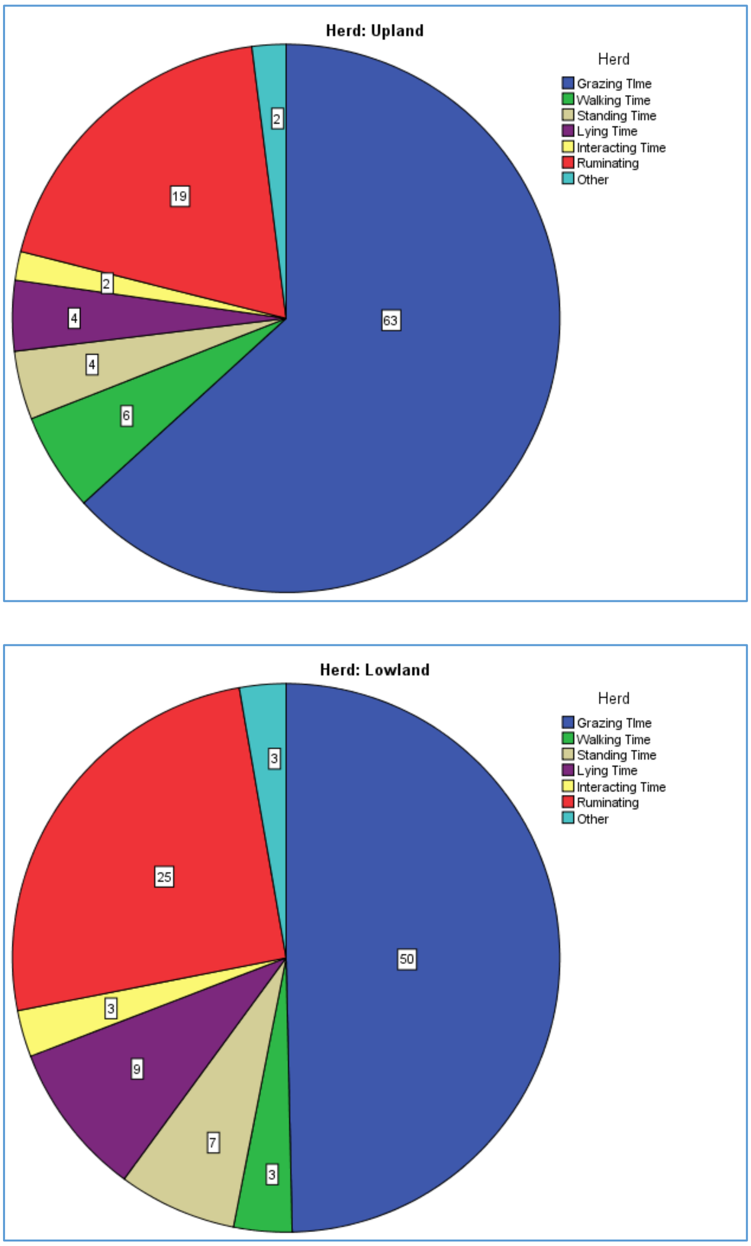


Figure 13 Proportions of activities that the upland and lowland herd were engaged in.

2.3.4 Stocking rates

Table 17 shows socking rates based on home range and habitat selection analysis of Dexter cattle in Mt Brandon NR between 2013 and 2015. Mean home range size and mean percentage of each habitat type within home range were used for analysis. For full home range results see Figure 11 & Table 11 section 2.3.1. The number of GPS locations (nLocs) and the percentage of locations in each habitat type (% nLocs) were used to derive coefficients for calculating the stocking rates (see Table 13, section 2.3.1 for full details of %nLocs per habitat type).

Stocking Rates (SR) are calculated by: *Livestock Units (LU)/ Home Range (HR),* adjusted for proportion of nLocs in each habitat. Worked example:

e.g. for Wet Heath: 21 Livestock Units, 32.8% of GPS locations in wet heath in an area of 56ha = (21 LU * 0.328)/56 ha = 0.1 LU/ha

Table 17 Stocking Rates based on home range and habitat selection in Mt Brandon Nature Reserve, Co Kerry.

	HR (ha)	% HR	LUs*	nLocs	% nLocs	**SR***
Full HR	122.7	100	21	1242.3	100	**0.17**
Wet Heath	56.0	45.6	21	408.1	32.8	**0.12**
Dry Heath	29.5	24.0	21	346.6	27.9	**0.20**
Blanket Bog	26.4	21.5	21	218.3	17.6	**0.14**
Wet Grassland	10.9	8.9	21	269.3	21.7	**0.42**

Note: stocking rate in Study Area = (21/462ha)/2 = 0.023 LU/ha.

*LUs based on 15 Dexter cows with 15 followers and 10 steers, adjusted for small size of cattle (30*0.7 = 21). Coefficient for small weight of cattle based on (Martin et al., 2013). Mean weight of cattle = 305.98 kg (weights based on July 2014 cattle weights, data available in (Dineen, 2016).

Note: the cattle were on-site for 3.5 months, which is 50% of the current requirement for stocking of upland habitats under the ANC scheme and Commonage management recommendations. The impact of this stocking rate on the habitats and species will be lower than if applied as per current recommendations.

2.4 Discussion

2.4.1 Home Range

Little variability was found in home range and habitat use between the tracked cattle with respect to home range size and shape, habitats available within home range and habitats used at location.

The results show that Minimum Convex Polygons (MCPs) provided the largest estimate of home range and included large areas of 'white space' unvisited by the animals. Convex polygons have external angles which are all greater than 180^0. MCPs are the smallest of such polygons which can be drawn around a set of locations, and have been widely used to define ranges (Harris et al., 1990; Kenward et al., 2008), but are strongly influenced by outlying locations (Kenward, 2001; Kenward et al., 2008).

Objective-Restricted-Edge Polygons (OREPs) provided the best fit to the location data and the best estimate of home range size in this study. This is consistent with the findings of Enright and Williams (2010) in a study of feral goats in the Burren and of Bevan (2008) in a study of cattle in Yorkshire, UK. The method eliminates outliers using outlier exclusions distances (OEDs) (Kenward et al., 2008). In this case, exclusion is of locations in the largest 5% of the nearest-neighbour distance distribution. Polygon edge distances are based on the distribution of mean distances from each location to all others, as similarly estimated in kernel analysis (Kenward, 2001). The Kernel Exclusion Distance (KED) is the default in RANGES8 (Kenward et al., 2008).

Few studies have examined the movements of free-ranging domestic animals (Williams et al., 2012a), particularly those involving cattle (Bevan, 2008), despite the general acceptance that they are less selective than sheep (Wright et al., 2006) and are a frequent choice for conservation purposes (English Nature, 2005; Bevan, 2008).

Kie and Boroski (1996) used GPS tracking collars to determine the home range of cattle in the Sierra Nevada of California (40 cattle, grazed between May and September). Out of 657 ha potentially available to the cattle, their home range was determined to be 162.80 ha (SD = 57.81, n = 4) in year 1, and 278.83 ha (SD = 54.03, n = 6) in year 2. These findings show that 24.6% was used out of the available area in year 1 and 42% in year 2. The results of the Brandon study are accordant with these findings, where mean range size using similar estimators was 115.40 ha (SD = 12.78, n = 3). This equates to 24.9% of the total area of the reserve, and 21.5% of what can be considered available i.e. 115 ha mean range size, out of 535 ha of available habitat (table 2 in study site description). Mean home range sizes observed in the present study were larger than those observed by Hall (1988b) in free-ranging cattle in Chillingham Park in the UK, where home range size was 5.7 – 53.56 ha (Hall, 1988a).

Using MCP estimation Moyo et al. (2013) found that cattle had a home range size of 103.8 ha (± 1.0) out of 797 ha available (n = 4 cattle, with a total of 370 GPS locations over 4 seasons) in the Eastern Cape of South Africa i.e. 13% used of what was available. Using the same estimator, the results show that the mean range size of the Brandon cattle was 185.7 ha (SD = 28.6, n = 3) i.e. 34% used of what was available. Cross-bred cattle of Hereford, Angus and Shorthorn backgrounds were found to utilise 35% of the area available to them in southern Oregon (Roath and Kreuger, 1982).

Kaufmann et al. (2013a) explain that as animals endeavour to optimise nutritional intake while minimising effort, the main factors affecting home range size, apart from accessibility, are forage quality and quantity. The factors affecting forage characteristics include community composition, plant morphology, palatability and growing conditions (Chapman et al., 2007; Kaufmann et al., 2013). How cattle are distributed across a landscape, and their use of habitats, is further influenced by temperature and topography (Roath, Kreuger and Krueger, 1982; Kie and Boroski, 1996; Kaufmann et al., 2013), protection from pests (Owens, Launchbaugh and Hollway, 1991; Beaver and Olson, 1997) and water availability. A combination of these factors offers likely explanations for the findings of the current study. Site-specific factors affecting the distribution of cattle and the patterns of their movements potentially include disturbance by recreational site users (walkers, occasional campers etc.) and other visitors to the reserve (NPWS staff, fieldworkers, farmers and caretakers etc.), shelter, and drier areas for lying up.

The results show that there was some inter-month variability of home range size, and there was range expansion from July through to September which was also reported by (Linnane, Brereton and Giller, 2001; Hejcmanová et al., 2009) and cattle utilised higher slopes of the reserve in these months. Range size, shape and structure changed dramatically in the month of October. Cattle shifted to lower elevations of the study site into Arraglen valley, which is dominated by grassland and heath/grassland transition habitat. This pattern was repeated in all years by the collared cattle. There is an established relationship between home range size and resource availability for many animals (Harestad and Bunnell, 1979; Hodder and Low, 1978; Mitchell and Powell, 2007; Moyer, McCown and Oli, 2007), showing that as resource availability in an area increases, home range size decreases.

Local topographical differences and site-specific features were likely to have influenced cattle movements. Figure 7–9 highlight areas where GPS fixes were concentrated. They include the area surrounding the NPWS hut, the location of historic dwellings which may also have provided shelter for cattle, and points at which tracks and paths converge which cattle used to travel around the site.

The Arraglen valley is an area where GPS fixes are concentrated, which is a sheltered, steep sided valley, composed of wet grassland and grassland/heath mosaics. The cattle may have favoured this valley due to the availability of forage and because it offers shelter from the prevailing south-westerly winds. A small river flows through the valley which the cattle drank from. A third area frequented by the cattle was a flat col/saddle between Más an Tiompán and Carraig a Chin, a small peak at the northern perimeter of the reserve. This area provided shelter in south west wind and has patches of grassland in blanket bog and wet heath mosaics.

2.4.2 Habitat selection

The difference in proportions of habitats used at location compared to those available in home range and in the study area show that cattle exhibit habitat selection. At a broad level cattle selected wet heath and blanket bog and at a detailed level chose wet grassland. Examining habitat preference using Jacobs' index indicated that the cattle avoided blanket bog and wet heath and showed most preference for wet grassland. Dry heath was used in proportion to its availability. The results from the compositional analysis were similar, in that at a broad level cattle selected wet heath and blanket bog but at a detailed level showed significant preference for wet grassland.

Studies on habitat selection by cattle in heterogeneous upland landscapes are limited in number. Fraser et al., (2009) found that cattle (and sheep) consistently preferentially selected grasses over dwarf shrubs even when ratios of heather to grass were high (Fraser et al., 2009). Cattle are reported to preferentially select grasses when grazing heather moorland (Grant et al., 1987) and the findings of the present study concur with this, with cattle exhibiting significant preference for grassland. The avoidance of wet heath and blanket bog in preference for grass dominated habitat is consistent with the findings of Grant (1987) and Fraser et al. (2009).

Wet Grassland was the most selected habitat at both levels and results from Jacobs Index concur with these findings. This has consequences for the management of the blanket bog and heath habitats that are of international conservation importance.

McNaughton (1984) reports that grazers congregate where vegetation productivity is high, so preference of grassland habitat by the cattle is not surprising.

Hodgson et al. (1991) showed that digestibility is higher in grassy communities than on dwarf shrub communities and that digestibility values decline progressively from spring to late autumn and that intakes are highest in summer (Hodgson et al., 1991b). This may explain why cattle preferentially selected grassland communities in the Brandon study and decreased their home range size in October to focus on grassland swards. Grassland was preferentially selected in this study probably because of higher forage availability (in patches) and quality than the heath and bog communities, despite there being less of it available overall to the animals in the study area and within home range.

The heath communities presented more subtle and complex results with respect to selectivity, with cattle using these habitats at levels close to their proportional availability. At the broad level cattle selected wet heath over dry heath and selection was significant between the two ranked variables, whereas the dry heath was selected in preference over wet heath (and blanket bog) at the detailed level. Jacobs Index shows that preference/avoidance levels were low, with D = +0.1 for dry heath and D = -0.3 for wet heath.

The availability of nutrients in different habitats can be important in determining selection during foraging (Wallis de Vries and Schippers, 1994). Cattle may have avoided Blanket Bog habitat in the present study because it is deficient in nutrients (Van Eck, 1984; Williams et al., 2012c) and is the least digestible compared to the other habitats available (Hodgson et al., 1991b). Further, it can be extremely wet (Fossitt, 2000) with many pools, areas of deep peat and surface water, which cattle may find undesirable underfoot.

In complex habitat mosaics with patches of differing nutrient quality, animals may be thought to be distributed so each has an equal share of resources (Wallis de vries, 1998a), resulting in a free distribution. In cattle, Wallis and Shippers (1994) found a trade-off between riverine and heathland habitats, whereby cattle could get higher nutrient levels (sodium and phosphorous) in riverine habitats but a higher level of energy from grazing heathland, particularly in winter (Wallis de Vries and Schippers, 1994).

Senft, Rittenhouse and Woodmansee (1985) explain that forage quality and quantity are good predictors of community preference, along with proximity to water. In the Brandon study area, the wet grassland habitats are near the river that flows through the site. Cattle have been shown to concentrate in riparian habitats when offered a choice because of the proximity to water and nutritious forage (Bryant, 1982; Roath, Kreuger and Krueger, 1982; Schulz and Leininger, 1990; Smith et al., 1992). This, along with the higher forage quality of the grassland habitats offer a likely explanation of the selectivity of the cattle in the present study, as cattle will select plant communities of high nutritive value (Anderson and Kothmann, 1980; Roath, Kreuger and Krueger, 1982; Moyo et al., 2013).

The choice of home range estimator affects habitat selection analyses because it impacts on the proportions that are deemed 'used' or 'available' (Aebischer, Robertson and Kenward, 1993; Kauhala and Auttila, 2009; Enright, 2012). In this study the GPS fixes were taken every two hours, which was a trade-off between high resolution tracking and preserving battery. MCP estimators provide the largest home range and include large areas of white space that may not have been visited by the animals and may fail to detect preference or avoidance when compared to other methods (Kauhala and Auttila, 2010). OREP estimation provided the best fit to the data and offered the best choice for habitat preference analysis.

Radio telemetry is a powerful tool available to wildlife biology because it has potential for providing unbiased spatiotemporal data on an animal behaviour. However, robust statistical analysis of habitat use can be problematic. Areas of particular concern are sampling level, data pooling across individuals, differential habitat use of individuals, and arbitrary definitions of availability (Aebischer, Robertson and Kenward, 1993). Central to these issues is that avoidance of one habitat by an animal will lead to an apparent preference of another. It is recommended that a minimum of 6 samples (individually tracked animals) is required in order to adequately compare 'utilised' with 'available habitats' i.e. compositional analysis (Aebischer, Robertson and Kenward, 1993; Taulman and Smith, 2004; Williams et al., 2012c).

Compositional analysis is suitable for type II and III study designs described by (Manly, McDonald and Thomas, 1993; Manly et al., 2002) where individual estimates of habitat use have been collected (Pendleton et al., 1998) as was the case in this study.

Four potential problems exist with analysis of habitat use datasets, as discussed by Aebischer, Robertson and Kenward (1993): (1) inappropriate determination of the sample unit, (2) nonindependence of habitat proportions, (3) differential use patterns by identifiable groups of animals and (4) arbitrary definition of availability. Compositional analysis accounts for these problems but other issues need to be addressed, namely dealing with habitats that are not available, or habitats that are available but are not used by the animals (Pendleton et al., 1998). In the present study, the habitats that were unavailable (e.g. sea cliffs) were removed prior to analysis and no available habitats were unused during the study.

2.4.3 Activity budgets of cattle in upland and lowland settings

Time spent on different activities reflects the availability and quality of pasture as well as climate and physiological states (Hirata et al., 2002). Cattle exhibit a wide variety of behaviours in 40 different categories of which grazing is the most common, followed by ruminating and resting and these three categories combined typically make up 90-95% of an animal's day (Kilgour et al., 2012). The cattle in this study followed this pattern with grazing, ruminating and resting making up 90% of the cattle's day in both study areas. However, some variation in allocation to each activity was evident.

As reported, the upland herd spent significantly more time grazing than the lowland herd. Next to grazing, ruminating was the most common activity in both herds. The upland herd also spent more time walking than the lowland herd and it was ranked number three in the order of behaviours ahead of lying and standing. Lying and standing may correspond to resting or idling as described by Cory (1927) and Hall

(1998). In comparison, rumination was followed by lying and standing in the lowland herd i.e. they rested more than walked, which may reflect the spatial arrangement of the preferred habitats and concurs with the findings of the GPS tracking, whereby cattle travelled an average of 1.4km.day^{-1}.

In a study on Chillingham cattle, Hall (1989) found that cows spent 37% of their time grazing in summer and 53% in winter. Kropp et al. (1973) found that Hereford and Hereford x Holstein heifers spent 42% of their time allocated to grazing. In 6 studies reviewed by Kilgour (2012), the average time allocated to grazing by cattle was 37% (±5%), compared to 66% and 52% by the upland and lowland cattle in the present study respectively. In a study of cattle in the uplands of the Czech Republic Hejcmanová et al. (2009) found that cattle in an extensive setting apportioned 52% of their time to grazing, followed by ruminating (20%) and resting (25%). The results of the present study are comparable, except that in the uplands the cattle spent less time resting and more time walking, which may have been expected given the spatial arrangement of preferred habitats and possibly available shelter, water and pest insects.

The poorer nutritional quality of available forage in the upland habitats compared to the improved grassland of the lowland site may explain the 13% difference in the amount of time spent grazing between the two herds, yet the upland cattle spent less time ruminating than the lowland cattle. Longer feeding bouts are necessary to fill the rumen when available vegetation is of low quality may, which explain the longer grazing times of the upland cattle as well as the increasing home range as the season progressed (Bourgoin et al., 2008).

Ruminating is the second most important activity for cattle (and any ruminant) after grazing (Arnold and Dudzinski, 1978; Hejcmanová et al., 2009) and usually varies with diet (Hejcmanová et al., 2009). Rumination increases with increasing fibre levels in diet (Hejcmanová et al., 2009), so the upland cattle may have been expected to spend more time ruminating than the lowland cattle. However, grazing animals maximise energy intake per unit time, as per the 'optimal foraging strategy' (Stephens and Krebs, 1986), which may explain why the cattle in the uplands allocated more time to grazing, traded off against rumination.

The upland cattle spent more time walking than the lowland cattle, which may be related to the spatial distribution of preferred habitats. Landscape configuration is a dominant factor in determining home range size (Bevanda et al., 2015), which decreases with good forage availability (Tufto, Andersen and Linnell, 1996). The spatial arrangement of different habitats can influence the distribution of large mammals (Clutton-Brock and Harvey, 1978), which influences movement trajectories (Bevanda et al., 2015). These factors may explain the longer walking times of the upland cattle compared to the lowland herd in this study.

GPS fix success rates were high in this study, despite the remoteness and topography of the site. The loss of a cow, the GPS collar it was wearing and the data stored on board was an unfortunate and expensive loss. A weakness of studies involving GPS tracking is often small sample sizes brought about by expensive equipment

(Hebblewhite and Haydon, 2010), sometimes further weakened by catastrophic failures or losses. Despite the losses, the nine samples successfully obtained in this study over three grazing seasons offers an insight into cattle behaviour in the Irish uplands.

The system employed by the grazier is such that cattle grazing the upland site are predominantly female, either cows with calves or in-calf heifers. Only small numbers of steers grazed the upland site and none of these were collared. Hall (1989) showed differences in maintenance behaviours between male and female Chillingham cattle, so further research involving steers and bulls in free ranging upland settings is recommended. It has been shown that calves spend less time grazing and more time resting in intensive systems (Hirata et al., 2002), and this is likely to be the case in free ranging upland settings and further direct observation studies would elucidate this.

Indices of habitat preference may show that an animal occurs in a habitat more than would be expected by chance, yet disproportionate use may not necessarily indicate preference (Kenward, 2001). Relative avoidance of one habitat always creates relative preference for another due to the unit-sum constraint (Aebischer, Robertson and Kenward, 1993) and ranking variables from most to least preferred offers a solution (Kenward, 2001).

Compositional analysis offers a solution to the unit-sum constraint and pseudoreplication in habitat selection analysis (Aebischer, Robertson and Kenward, 1993; Kenward, 2001). In this study habitat selection was assessed using an index of preference (Jacobs Index (1974)) and compositional analysis (Aebischer, Robertson and Kenward, 1993). The results of these analyses agreed and thereby offer an insight into the habitat preferences of cattle in a free ranging upland setting. However, the results are only representative of one herd on one site and issues of pseudoreplication must be acknowledged. Furthermore, non-independence of samples is indicated within years by the results of the dynamic interaction analysis i.e. within years the collared animals were in the same place at the same time, thus are representative of the herd and so are potentially not independent samples. Future research should consider tracking animals in multiple herds that are spatially independent to avoid issues of non-independence. Furthermore, future studies should sample male animals and cattle in different age classes to assess between-group differences in home range and habitat selection.

The primary objective of this research was to study the home range behaviour and habitat preferences of cattle in a 462-hectare State-owned nature reserve. This study has identified the home range characteristics of native cattle in an area of international conservation importance in Ireland for the first time. The size, shape and structure of the home range of free ranging cattle has been described and quantified, the habitat preferences of the cattle have been identified and the activity budgets of upland and lowland cattle have been described.

Grazing prescriptions should include availability and spatial distribution of preferred habitats. This study's findings of preferential selectivity by cattle supports the argument that grazing prescriptions for upland habitats should go beyond stocking densities towards habitat availability and spatial distribution of grazers (Williams et al 2012; Hester and Baillie, 1998).

2.4.4 Stocking Rates

The stocking rate for the entire study area was 0.17 $LU.ha^{-1}$. As grazing animals do not forage evenly everywhere, analysis of home range and habitat selection behaviour allowed a more nuanced approach to be taken when calculating stocking rates. These rates were self-imposed by the cattle because they had unlimited access to the 462-ha site and established a home range within it i.e. Johnson's first-order selection (Johnson, 1980). Within their home range the cattle selected habitat components that met requirements of food, shelter etc., which is 'second-order selection' (Johnson, 1980). Analysing first and second order selection choices by the cattle in this study has allowed stocking rates in a free-ranging upland setting to be calculated.

Within the home range of the cattle (122.7 ha) the stocking rate was 0.17 $LU.ha^{-1}$. The home range results were also used to calculate rates for each habitat type. These rates were: 0.12 $LU.ha^{-1}$ for wet heath, 0.20 $LU.ha^{-1}$ for dry heath, 0.14 $LU.ha^{-1}$ in blanket bog and 0.42 $LU.ha^{-1}$ in wet grassland. Under Ireland's Areas of Natural Constraint (ANC) scheme, which encompasses most of the Irish uplands, the minimum stocking rates for *all* upland habitats not under commonage management is 0.15 $LU.ha^{-1}$.

Under Commonage Planning Framework planning (DAFM, 2015a) stocking rates (for undamaged habitats) are: 0.75-1.0 $ee.ha^{-1}$ for wet heath, 1.0-1.5 $ee.ha^{-1}$ for dry heath, 0-0.75 $ee.ha^{-1}$ for blanket bog, and 1.15-5.0 $ee.ha^{-1}$ for upland grassland. Where habitats are deemed to be damaged, habitat condition coefficients are applied and stocking rates may be adjusted up or down. The results of the present study are at variance with the prescriptions for commonages and would suggest that where habitats are in good condition, stocking rates should be lower than are currently prescribed.

Stocking rates under Ireland's ANC and under Commonage Planning Framework are higher than equivalent schemes in the Scotland, England, Wales and Northern Ireland. Table 18 shows compares stocking rates in Britain and Ireland, along with the calculated rates from this study (shaded).

Table 18 Current stocking rates in Britain and Ireland.

Habitat	SCOT	ENG	WAL	NI	IRL	MBNR study*
Blanket Bog	0.02 (0.00-0.05)	0.07	-	0.075	0.15 (ANC) 0-0.75 (CM)	0.14
Wet Heath	0.08 (0.05-0.10)	0.09	<0.4LU Apr-June <0.2LU July-Sept <0.1LU/ha Oct-Mar. Never below 0.2LU Apr-Sept 0.05LU July-Sept.	0.25	0.15 (ANC) 0.75-1.0 (CM)	0.12
Dry Heath	0.12 (0.10-0.2)	0.2	-		0.15 (ANC) 1.0-1.5 (CM)	0.2
Grasslands						0.42
Poor	0.25 (0.2-0.4)	-	-	-	0.15 (ANC)	
Moderate	0.50 (0.4-0.6)	-	-	-	1.15-5.0 (CCM)	
Good	0.7 (0.6-0.8)	0.2	-	-		
Semi-natural	0.8-1.0	0.2	-	-		

*Cattle were on site for 3.5 months (half the current requirement for stocking of upland habitats under ANC and Commonage Management recommendations.

2.5 Conclusions

The uplands of Britain and Ireland contain large areas semi-natural habitats, much of which are considered High Nature Value farmland that support high levels of biodiversity. These landscapes and the wildlife that they support are simultaneously under threat from both intensification and land abandonment (Lomba et al., 2014). Upland landscapes are primarily managed through extensive grazing, yet a strong evidence base for fine tuning grazing management is lacking (Wallis De Vries et al., 2016).

Selecting appropriate management prescriptions for upland HNV farmland is challenging, particularly with regard to choosing suitable stocking rates because many site-specific variables are influencing (Williams et al., 2012). Agri-environment schemes are often criticized for the broad application of stocking rates and there is a need for more specific prescriptions based on grazer behaviour (Hester and Baillie, (1998) in Williams et al., (2012)).

In this study, cattle were GPS tracked over three summers in a free-ranging upland setting and their home range and habitat selection behaviours were established. Mean home range size was 122.7 ha and habitat selection was statistically significant, with cattle showing most preference for grassland habitats and least preference for blanket bog. Cattle in this study showed a slight preference for dry heath over wet heath. Activity budgets for cattle in the study area were also established and showed that cattle in the uplands spend significantly more time grazing than cattle in the lowlands.

Upland cattle also spent less time ruminating and resting than lowland cattle in this study.

Recommended stocking rates for the Irish uplands are higher than those for Scotland, England, Wales and Northern Ireland. The home range and habitat selection results were used to calculate stocking rates for the cattle in the study area. The findings demonstrated that in this (free-choice system) the cattle self-imposed stocking rates were more in line with UK recommendations than current Irish schemes. The findings support the argument that management prescriptions should not just be based on broad stocking densities, but also consider habitat availability and behaviour of grazing animals (Williams et al., 2012a).

Habitat condition and conservation status in relation to current stocking rates in the study area are explored and presented in detail in Chapter 3.

3 Chapter 3: The impact of conservation grazing with Dexter cattle on upland habitats in south west Ireland

A view to Mas an Tiompán (762 m) in Mt Brandon Nature Reserve. Photo K.Kelly.

Dexter cattle grazing on wet grassland beside the Arraglen river, Reserve. Photo K.Kelly

3.1 Introduction

3.1.1 The uplands

The uplands cover almost a third of the land surface of both Ireland and Britain, and can be broadly classified as areas of unimproved lands that occur on hills and mountains (Fielding and Haworth, 1999; Perrin et al., 2011a). Although no formal definition of the uplands exist, they are described as being 'above limits of enclosed farmland' and generally above 200m by Ratcliffe (1977), or above 150m in Ireland by Perrin et al. (2014). Altitudinal zonation is usually defined by climatic effects on mountain vegetation. Climate, for which altitude is a proxy, influences vegetation gradients, yet this influence is mediated by topography and management practices such as improvement of land for agriculture. Thus, the boundary observed between uplands and lowlands may be sharp or less distinct depending on the intensity of agricultural activity (Fielding and Haworth, 1999).

Over 40% of the sites designated under the Habitats Directive (European Commission, 1992) in Ireland occur in the uplands. Irish upland habitats include heaths, bogs, semi-natural grasslands, bracken and areas of exposed rock and scree (Perrin et al., 2011a). These typically occur in complex mosaics because of variation in topography, drainage and management conditions. Upland habitats are of high conservation value with 14 types listed in Annex I of the Habitats Directive (Perrin *et al.* 2010).

Overstocking, drainage, uncontrolled burning, afforestation and more recently windfarm development, are the main activities that have led to large scale degradation of Ireland's upland habitats (Heritage Council, 1999). Despite conservation measures introduced under agri-environmental schemes, such as Rural Environmental Protection Scheme (REPS) and the Commonage Framework Plan (CFP), results have been ambiguous to date (Kleijn and Sutherland, 2003a; Whittingham, 2007; Finn and Ó hUallacháin, 2012a) and the status of the majority of EU Protected upland habitats in Ireland is currently assessed to be in 'poor' or 'bad' condition (NPWS, 2013, 2019a).

3.1.2 The Uplands: Geology and Climate

Geology plays an important role in the ecology of upland areas. Uplands, formed initially by tectonic processes such as faulting and folding, are generally found where the rock type is resistant to weathering and most will show evidence of erosion from the last glacial period which ended approximately 10 000 years BP (Fielding and Haworth, 1999). Geology influences the ecology of the uplands through the effect of rock type on soils and topography. Uplands are topographically complex, which can lead to a range of topoclimatic and microclimatic conditions occurring in a small area (Barry, 2008; Hodd, 2012). Rocks resistant to weathering are often deficient in calcium, resulting in the development of acidic soils (Fielding and Haworth, 1999) and high rainfall may exacerbate calcium deficiencies through the effects of leaching, thus further reducing pH. Acidic conditions combined with high rainfall can lead to an accumulation of organic material and peat development.

Climatic conditions are among the main factors affecting the composition of the vegetation in Ireland's uplands. In Ireland and Britain the uplands are characterised by low temperatures, high rainfall and humidity, high windspeed and near continual ground wetness with low evaporation rates (Fielding and Haworth, 1999). Ireland is on the western fringe of Europe and has a climate that is dominated by the influence of the Atlantic Ocean i.e. it is highly ‘oceanic’. ‘Oceanicity’ describes the conditions of temperature, humidity and other factors that result from maritime influences and alter the environment of oceanic regions (Crawford, 2008). The oceanic climate of Ireland means that it doesn’t have extremes in temperature but does have high altitudinal temperature lapse rates and high cloudiness with frequent rainfall and humidity (Ratcliffe, 1968; Grace and Unsworth, 1988)

Ireland’s mountainous western margin is subject to near constant high humidity as it is the first landfall for Atlantic weather systems (Sweeney, 1997). Temperatures in Ireland are high for its latitude due to the influence of the Gulf Stream and temperatures do not rise as high as continental regions of similar latitude in summer due to the cooling effects of the ocean. The topography of Ireland is such that the uplands are mainly coastal and there is high climatological contrast between the maritime margins and the interior of the country which is relatively continental (Sweeney, 1997). Temperature decreases with altitude (Barry, 2008) and is important because it is one of the major determinants of plant growth.

Precipitation rates and windspeed are usually higher in upland areas than in the lowlands (Brunsdon, McClatchey and Unwin, 2001; Barry, 2008). Mangerton mountain in county Kerry (808 m) receives 3,184 mm of precipitation per year, while a station one kilometre away at 58m in elevation receives only 1585mm (Carruthers, 1998). Furthermore, Mangerton received 1800 mm more *per annum* than nearby

Valentia (9m asl) between 1961 and 1990 (Sweeney, 1997; Hodd, 2012). In middle and high latitudes wind speed will increase with height, with isolated peaks and exposed ridges having high average and extreme wind speeds due to limited frictional effects of terrain on air movements (Barry, 2008).

3.1.3 Human Impact

In Europe, large open heathland complexes have been intimately related to human activities since their development around 6000 years ago and their conservation value is related to low intensity grazing over a long period (Hampton, 2008). The impact of humans on the Irish landscape over the last few centuries has resulted in rapid landscape alterations (Hall, 1997; Mitchell and Ryan, 1997). The main human influence on upland vegetation in Ireland is sheep grazing. Overgrazing has occurred in Ireland over the last 30 years as a result of increasing sheep numbers (Bleasdale and Sheehy-Skeffington, 1995) and grazing-related damage has been attributed to rural development schemes that gave rise to increased stocking levels (Bleasdale and Sheehy-Skeffington, 1995; Williams et al., 2012c).

More recent agri-environmental policies have led to a reduction in stocking rates (Holden et al., 2007; Williams et al., 2012c), yet damage is still evident. Grazing pressure varies depending on topography and slope, with less grazing occurring on steep slopes and cliffs (Hodd, 2012). In the uplands, overgrazing can result in changes to habitats, with grassland replacing heath and in severe cases the entire loss of vegetation from a slope (Bleasdale, 1998). Other factors may contribute to vegetation loss, such as drought, wind, rain and frost (Fenton, 1937) but overstocking facilitates erosion processes through excessive defoliation and trampling (Brigand and Bioret, 1994).

Ireland is one of the least forested countries in Europe, yet plantation forests have brought forest cover from less than 1% of the land area of Ireland to approximately 10.5% in the last century (Forest Service, 2012). Conifer plantations came to dominate the previously open spaces of the uplands in the 1950s and 1960s (Hall, 2011), particularly on blanket peat (Rodgers et al., 2010). Large areas of plantation forest in Ireland are reaching second rotation after harvesting and replanting (Oxbrough et al., 2010). The impact of forestry on biodiversity varies with the stage of afforestation. In early stages species from the original habitat persist and previous land uses are important influences on diversity, yet the biota associated with these habitats are unlikely to persist beyond canopy closure (Iremonger et al., 2006).

3.1.4 Vegetation history of the Dingle Peninsula

Vegetation histories are known from Dingle peninsula from direct evidence at Ballinloghig Lake (Barnosky, 1988), which is 9 km from the study area, and from the Lough Adoon valley (Dodson, 1990), which is 7 km away. Following deglaciation, *circa* 13,000 B.P., the pollen records show that grasses and *Rumex* were present, followed by a juniper-crowberry heath landscape (Barnosky, 1988). By 10,000 B.P. *Salix* spp., *Plantago maritima* and *Betula* species begin to appear. *Corylus* was present in the area from 9800 B.P., before it was known from anywhere else in Ireland, and it was later followed by woodlands that included *Betula, Pinus, Alnus, Quercus, Ilex* and *Hedera.* The decline of woodland in the area first commenced around 4100 B.P., when the

Pinus and *Alder* disappear from the records and an increase in peatland taxa was evident (Barnosky, 1988).

Further intensification of woodland clearance between 2750 and 2450 B.P., lead to the podsolization of soils and further spread of heathland and peat formation (Barnosky, 1988; Dodson, 1990). The pollen records of Barnosky's (1988) work show that following the elimination of *Pinus* and *Alnus* from the region, species of Ericaceae, *Myrica,* Gramineae, Cyperaceae and *Sphagnum* appear. This tracks the "*shift from woodland to blanket bog in the uplands*" (Barnosky, 1988). Thus, blanket bog and heath habitats have dominated the uplands of the Dingle peninsula for at least the last 2000 to 4000 years.

3.1.5 Irish Upland Habitats

Classification

The broad scale Fossitt (2000) habitat classification scheme (structure of which is presented in Appendix I) is widely used for surveying and mapping in Ireland and is intended as a first step for general habitat recording (Perrin et al., 2009). In 2008, the NPWS initiated a scoping study for the mapping of upland habitats in Ireland (Perrin et al., 2011a). This was a precursor to the National Survey of Upland Habitats (NSUH) which began in 2010. The objectives of the NSUH are to map and assess the conservation status of upland habitats in Ireland (Perrin et al., 2014a). This study follows the NSUH in that it focuses on habitats listed in Annex I habitats of the EU Habitats Directive (European Commission, 1992, 2007) (see Table 19).

A provisional classification scheme was developed by Perrin et al. (2014) on upland vegetation as part of the NSUH that is based on relevé data. Appendix I provides a breakdown of upland habitats, their provisional NSUH communities and codes, and the corresponding Annex I and Fossitt (2000) categories. Appendix I also provides an abridged version of Fossitt (2000) showing its basic structure.

Peatlands

The peatland habitats that occur in the Irish uplands are bogs (upland blanket bog, cutover bog and eroding blanket bog), fens and flushes (rich and poor) and transition mires (Fossitt, 2000). Of these, Blanket bog (7130), transition mires (7140), *Rhynchosporion* depressions (7150) and Alkaline fens (7230) are Annex I habitats. Blanket bogs are comprised of peat forming vegetation on deep peat (typically 1-2 m in the uplands) and occur on gentle slopes up to 20-25^{o} (Tallis, 1998). Blanket bogs are rare globally, making up 3% of the world's peatlands and in northern latitudes their distribution is restricted to temperate maritime latitudes (Lindsay et al., 1988). Blanket bogs can be classified into Montane (over 200m asl) and Atlantic (Foss, O' Connell and Crushell, 2000) and occur as complex systems of bogs, flushes and mires in close mosaics depending on local trophic status (Sottocornola et al., 2009). The variable vegetation types are usually characterised by *Eriophorum* spp. and *Sphagnum* spp., and seven categories are recognised under the provisional classification scheme described in the NSUH (Perrin et al., 2011a).

Active blanket bog (*7130) is a priority habitat under the Habitats Directive and in Ireland it covers an area of approximately 2287km^2, although producing accurate estimates are challenging due to the intimate mosaics it forms with wet and dry heath (NPWS, 2013). The main threats to blanket bog in the uplands are sheep grazing, afforestation, mechanical removal of peat and erosion, and its current conservation status under Article 17 is 'bad' with a declining trend (NPWS, 2013). The provisional classification scheme of the NSUH recognises four types of blanket bog in the Irish uplands depending on the plant community composition (Perrin et al., 2011a) (Appendix I).

Heathlands

Upland heaths in Britain and Ireland are of international conservation importance (Usher and Thompson, 1993; Thompson et al., 1995) and the main Annex I heath habitats that occur in the Irish uplands are Northern Atlantic Wet Heaths with *Erica tetralix* (4040), European Dry Heaths (4030) and Alpine and Boreal Heaths (4060).

Wet heaths are highly variable and occur on acidic peaty soils up to 0.5 m deep between at 200-600m (Critchley et al., 2008) and are usually dominated by dwarf shrubs (*Erica tetralix* and *Calluna vulgaris*), grasses (usually *Molinia caerulea*), sedges and mosses (Rodwell, 1992; Fossitt, 2000). Ireland has 1430 km^2 of wet heath, 23.4% of the EU Natura2000 network total (UK has 60.9%). It has a widespread distribution in Ireland and is especially prevalent in the wetter and more mountainous west. The conservation status of Wet Heath in Ireland is bad but stable (NPWS, 2013).

Pressures and threats to the conservation status of wet heaths include grazing pressure (primarily by sheep) peat cutting, afforestation and more recently renewable energy projects (Cummins, Peadar and Mee, 2012). High grazing pressure in the latter half of the 20th century resulted in the degradation of upland heaths in Britain and Ireland (Critchley et al., 2008), usually with dwarf shrubs being replaced by graminoid species (Bardgett, Marsden and Howard, 1995).

Widespread destocking of the Irish uplands *c.* 2002 under the Commonage Framework Plans may have a positive impact on the recovery of this habitat in Ireland (NPWS, 2008) and in some cases, the removal of grazing may allow dwarf shrub regeneration (Hulme et al., 2002). However, where dwarf shrubs have been lost completely, reduced stocking results in an increase of grasses such as *Molinia caerulea,* which prevents shrub recolonisation (Critchley et al., 2008; Marrs et al., 2004a). Cattle have been shown to selectively graze *M. caerulea* in summer (Grant et al., 1996;

Mandaluniz, Oregui and Aldezabal, 2005), and as summer grazing can reduce this species (Hulme et al., 2002), cattle may have a role to play in wet heath restoration (Critchley et al., 2008).

Dry heaths occur on flat to steeply sloping ground from sea level up to 400 m where they merge into Alpine and Boreal Heath (4040) (NPWS, 2013). They are dominated by ericaceous dwarf shrubs and usually occur on shallow free-draining peats of less than 50 cm deep (Fossitt, 2000; Perrin et al., 2011a). *Calluna vulgaris* is usually the dominant species with *Erica cinerea, Ulex gallii* and *Vaccinium myrtillus* often being important components. Dry heath is distributed widely, especially in the drier east, and it covers 1094 km^2 of the land area. The NSUH describes 6 communities in the provisional classification scheme, five of which are referable to Fossitt (2000) 'dry siliceous heath' and one to 'dry calcareous heath' (Perrin et al., 2011a). Impacts and pressures are similar to those affecting wet heath, with sheep grazing and burning considered to be threats of high importance (NPWS, 2013).

Montane heaths occur above 400m on the summits and upper slopes of mountains among loose rock and exposed bedrock, and are characterised by low shrubs (mainly *Calluna*), with *Racomitrium lanuginosum* being a key component (Perrin et al., 2011a). Much of the vegetation of the mountains above about 400 m western Ireland can be classified as montane heath. The majority occurs on peat and thus in many cases grades into blanket bog (Hodd, 2012). The NSUH divides montane heath into three groups depending on the community composition: 'montane heaths', 'montane grass-heath' and 'montane vegetation' and all refer to Fossitt (2000) MH4 Montane Heath.

Montane heaths have a dwarf shrub cover of at least 10% and the four types described in the NSUH refer to the Annex habitat 4060 Alpine and Boreal heath, which covers an area of 170km^2 in Ireland. Grass-heath is a relatively species poor, overgrazed version of other montane habitats that is characterised by *Nardus stricta* and, *Juncus squarrosus* and *Racomitrium lanuginosum* (Bleasdale and Sheehy-Skeffington, 1995; Hodd, 2012). Two of the montane vegetation communities described in the provisional classification scheme refer to the Annex I habitat 6150 Siliceous and boreal grasslands and the other is a non-Annex community. The main threats to montane heath plant communities are climate change (Hodd, 2012), sheep grazing and recreation (NPWS, 2013) and the overall conservation assessment under Article 17 is bad (NPWS, 2013).

Upland grasslands usually occur on shallow soils on gently sloping ground and are typically derived from heath habitats that have been modified by heavy grazing and/or burning (Perrin et al., 2011a). They are dominated by grass species and contain herbs such as *Potentilla erecta* and *Gallium saxatile.* The Fossitt (2000) classification scheme includes two grasslands that occur in the uplands, GS4 wet grassland and GS3 dry-humid acid grassland (see Appendix I).

'GS3 dry-humid acid grassland' occurs most frequently at the upper limit of enclosed farmland on free-draining acidic, mineral rich or peaty podzols and occasionally on siliceous sandy soils in the lowlands e.g. the Curragh in county Kildare. It frequently forms mosaics with heath and bog habitats, especially dry heath. It is characterised by *Agrostis* spp., *Festuca* spp, *Nardus stricta, Anthoxanthum odoratum* and *Deschampsia* *flexuosa,* as well as sedges e.g. *Carex binervis, Carex pilulifera* and herbs such as *Potentilla erecta, Succisa pratensis, Rumex acetosella* and *Viola riviniana.*

The NSUH describes 6 grassland habitats occurring in the Irish uplands and two of these are comparable to the Fossitt (2000) GS3 dry-humid acid grassland. One of these (UG1) is a close-cropped bright green productive sward dominated by *Agrostis capillaris* and the other (UG2) is a less palatable sward dominated by *Nardus stricta.* In UG2 *Nardus stricta* is usually abundant and is a relatively unpalatable coarse sward. Both have species-rich sub-communities referable to Annex I habitat **6230 Species-rich* Nardus *grassland in mountain areas*. In this case a relatively high cover of broadleaf herbs should occur, and species diversity will be high.

Wet grassland, Fossitt (2000) GS4 and NSUH UG4, is a non-annexed habitat that is often derived from over grazed wet heath in the uplands (Perrin et al., 2011a). Overgrazing may result in reduction of dwarf shrub cover and replacement with more resistant graminoid species, thus facilitating the expansion of grassland habitats (Backshall et al., 2001). Heather cover will decline if grazing animals utilise more than 40% of a season's growth, potentially allowing grasses to dominate (Thompson, Macdonald and Hudson, 1995; Backshall et al., 2001; Hampton, 2008).

GS4 occurs on wet or waterlogged mineral soils and community composition is variable but *Juncus* spp. will often be abundant. *Molinia caerulea* may dominate and other grasses such as *Nardus stricta, Anthoxanthum odoratum* and *Fescue* spp. also occur.

Table 19 Annex I habitats occurring in the Irish uplands (from Perrin et al., 2011a).

Habitat Code	Habitat Name
4010	Northern Atlantic wet heaths with *Erica tetralix*
4030	European dry heaths
4060	Alpine and Boreal heaths
6230	*Species-rich *Nardus* grassland
7130	Blanket bog (* if active)
7140	Transition mires and quaking bogs
7150	Depressions on peat substrates of the *Rhynchosporion*
7230	Alkaline fens
8110	Siliceous scree of the montane to snow levels
8120	Calcareous and calcshist screes of the montane to alpine levels
8210	Calcareous rocky slopes with chasmophytic vegetation
8220	Siliceous rocky slopes with chasmophytic vegetation
*Denotes a priority habitat under EU Habitats Directive	

3.1.6 Molinia caerulea

Molinia caerulea (hereafter *Molinia*), 'Purple Moor Grass' or 'Purple Heath Grass', is a tufted perennial grass native to north Africa and northwest Europe that can be locally abundant in the north and west of the British Isles. *Molinia* is common on uplands of Britain and Ireland, often dominating large areas at the expense of other flowering plants (Taylor, Rowland and Jones, 2001). *Molinia* is typically found in open sub-montane peatlands and grasslands on gentle to moderate slopes on gleys and deep peats. It can grow on many different soil types and has a bimodal pH distribution, with peaks on acidic soils of pH < 4, and calcareous soils of pH >7.0 (Taylor, Rowland and Jones, 2001).

Molinia abundance has increased since the industrial revolution at the expense of *Calluna vulgaris* in upland areas of the British Isles and its encroachment has been viewed as a major threat to moorland conservation (Marrs et al., 2004a). *Molinia* has also increased in other parts of Europe (e.g. Dutch heathlands) at the expense of dwarf shrub vegetation (Heil and Diemont, 1983; Diemont and Heil, 1984). Observed changes in vegetation from heath to *Molinia* have been attributed to inappropriate grazing and burning practices (Miller, Miles and Heal, 1984; Grant and Maxwell, 1988), and increased deposition of nitrogen and sulphur in association with industrialisation (Hogg, Squires and Fitter, 1995; Roem, Klees and Berendse, 2002; Marrs et al., 2004a).

Molinia caerulea exhibits high levels of genetic and morphological variation. Two subspecies are recognised within *Molinia caerulea sensu stricto* and many intermediates occur. *M. caerulea* spp. *caerulea* is tetraploid and forms clumps of several single-culmed plants of smaller stature (culms <65cm) and panicle size (<30cm). *M. caerulea* ssp. *arundinacea* is diploid and decaploid tussock-building plant with larger culms (65 – 125cm) and spreading panicles (30-60cm). Subspecies *caerulea* is widespread on moors, heaths, bogs, fens and uplands grasslands in Britain and Ireland. Subspecies *arundinacea* occurs in fens and fen-scrub or along rivers and canals on base-rich mineral soils. It has a scattered distribution in central Britain but is limited in Scotland and Ireland (Taylor, Rowland and Jones, 2001).

Molinia possesses a high phosphorous use efficiency (Aerts, 1989), and can translocate 75-85% of nitrogen and phosphorous from senescent leaves before abscission, storing them in root system internodes for use in the next growing season (Thornton and Millard, 1993; van Heerwaarden et al., 2005). This makes it a successful competitor in unmanaged swards, particularly where the availability of phosphorous and potassium is low (Hejcman, Češková and Pavlů, 2010).

Molinia is tolerant of grazing and burning, and in the uplands the history of these management practices on a site can determine whether heath or grassland habitats dominate. Burning on a 3-6 year rotation shifts dominance from heather to *Molinia* (Miles, 1988), and where grasses are already dominant, burning favours it over other grasses (Grant, Hunter and Cross, 1963). Heavy grazing has been shown to favour *Agrostis* spp. over *Molinia,* whereas *Molinia* will dominate in ungrazed or lightly grazed situations (Job and Taylor, 1978).

The grazing value of *Molinia* is intermediate between *Agrostis-Festuca* and *Nardus* grasslands and sheep will concentrate on *Agrostis-Festuca* areas in free choice or extensive grazing situations. Ungrazed leaves are shed in autumn and dead leaf litter accumulates. This impedes animal movements in the following winter and new growth of all grasses in the subsequent spring. The dead material reduces the attractiveness of *Molinia* communities for subsequent grazing and diminishes the quality of diet. However, when managed to reduce leaf litter accumulation, these communities can provide better-quality forage for grazing animals between June and August (Grant et al., 1985). The fast growth and height of *M. caerulea* in early summer may indicate that it is more suitable to cattle grazing than sheep, (Taylor, Rowland and Jones, 2001) as sheep generally graze closer to the ground.

In a study examining control measures for *Molinia*, Marrs et al., (2004) examined burning, grazing and herbicide treatments. Herbicide treatment was the only treatment to show consistent effects on *Molinia.* Spring burning reduced accumulated leaf litter cover, but effects were temporary and *Molinia* recovery was rapid. Three years of repeated light defoliation by cattle (33% of lamina length removal) reduced leaf production in a fourth uninterrupted season by 40% compared with ungrazed controls, and heavy defoliation reduced it by 78%. Floristic diversity was also increased on grazed sites compared with ungrazed sites (Marrs et al., 2004a)

3.1.7 Cattle as Conservation Grazers

Habitat conservation efforts are primarily concerned with 'natural' and 'semi-natural' landscapes. In natural landscapes, there is almost no human interference and they are governed by natural processes. In semi-natural landscapes the physiognomy of the landscape is altered by humans but the species occurring are typically native and spontaneous. Semi-natural landscapes are often the focus for nature conservation because they may be the only remaining source of wild plant and animal species in a region (Wallis de Vries, 1998). In Europe, centuries of extensive agricultural management has led to the creation of semi-natural landscapes which support a wide range of species (Kleijn et al., 2006).

Grazing can be defined as the 'grazing of grass dominated vegetation by large herbivores' (Walli de Vries, 1998), and it may be essential for the management of important wildlife habitats including grassland, heathland and woodland, as it maintains structural composition important for the survival of many plants and animals (English Nature, 2005). Grazing animals influence the structure and composition of habitats through processes of selective defoliation (Fenton, 1936), poaching (Nagy et al., 2002) and dunging (Bakker, 1998). Ecosystem processes such as productivity, turnover, and the distribution of nutrients may be modified by grazing and it can cause indirect cascading effects on the structure of entire ecosystems (Wallis de Vries, 1998).

Grazing and conservation have been argued as being incompatible and in the modern era livestock management is often so intensive that it has little value to the preservation of wild species (Wallis de Vries, 1998). However, grazing by sheep *Ovis aries*, cattle *Bos taurus,* and goats *Capra hircus* has helped to maintain the open character of semi-natural upland landscapes (Ratcliffe and Thompson, 1988; Usher and Gardner, 1988). 'Conservation grazing' is a term used to describe the use of grazing animals to maintain and enhance the biodiversity of semi-natural habitats (Small, 2003). Although it is not new, there has been an expansion of the practice in recent years and the Grazing Advice Partnership (GAP) reports that over 600 sites use grazing for conservation in the UK (Grazing Animals Project, 2017).

The European Forum for Nature Conservation and Pastoralism (EFNCP) is a network of over 60 organisations in 20 European countries focusing on the use of low-intensity livestock farming for conservation on High Nature Value (HNV) farmland (EFNCP, 2017). In Ireland, conservation grazing is now part of many agri-environment programmes e.g. the Burren and Programme in Co Clare, the AranLIFE Project in Co Galway, and the McGillycuddy Reeks EIP. It is also used by both state and non-government conservation organisations e.g. on the Shannon Callows in the midlands by Birdwatch Ireland (breeding wader and conservation), and in Pollardstown Fen in Co Kildare by the NPWS (grazed by highland cattle to improve habitat for Marsh Fritillary (*Euphydryas aurinia*).

Conservation management is interested in how diversity i.e. plant and animal species and community richness, can be enhanced or restored (Bakker, 1998). Studies have shown that herbivores often increase species and community diversity, yet their effects vary at different spatial and temporal scales (Olff and Ritchie, 1998), and with different stocking rates and grazing regimes (Grime, 1973; Vickery et al., 2001; Deng, Sweeney and Shangguan, 2013).

Large herbivores graze in different ways and have varying impacts on sward characteristics (Milne et al., 1998; Wright et al., 2006). Physiological differences contribute to differences in grazing and browsing behaviour which affects plant selectivity and the most important effect is that of body size. Small animals require more energy relative to body size than large animals and must select higher quality foods. Larger animals can retain food in the gut for longer and digest it more thoroughly (Rook et al., 2004). Hofmann (1989) classified ruminant species into three categories: grazers ('grass and roughage eaters'), browsers ('concentrate selectors') and intermediate types. The differences in the ability of herbivores to be selective, and the resulting impacts on sward characteristics and biodiversity, emerge when high quality or desirable components are rare or difficult to access (Rook et al., 2004).

Selective defoliation resulting from dietary choices is the most important mechanism by which grazing animals alter sward heterogeneity (Rook et al., 2004). In recent decades sheep have been the dominant domestic herbivores in the Irish uplands and they have small mouths and highly curved incisors, making them more selective feeders than cattle (Rook et al., 2004). They graze close to the ground and can bite off the portion of the plant that they are interested in. Thus, sheep have a greater ability than cattle to select high quality plant parts such as flowers, pods and young shoots (Oliván and Osoro, 1998).

CAP underwent reform in 1999 under ‘Agenda 2000’ and the Mid-term Review (2003 and 2004), with production-linked support and protection (‘decoupling’) being phased out. The Single Payment Scheme (SPS) was introduced in 2005 and it reduced the incentive to maintain high stocking rates and in some cases stocking density was reduced by up to 50% (Acs et al., 2010). Cattle and sheep numbers were greatly reduced in the uplands on sheep and beef farms.

3.1.8 The project

In 2011 a five-year grazing plan for Mount Brandon Nature Reserve was agreed between the National Parks and Wildlife Service (NPWS) of the Department of Arts, Heritage and Local Government (DAHLG) and Mr Paddy Fenton, an organic beef farmer in Ventry county Kerry. IT Tralee entered the agreement and carried out a baseline survey of vegetation and macroinvertebrates in the reserve prior to commencement of the grazing. The dominant habitats in the Reserve are Blanket Bog, European Dry Heath, Northern Atlantic Wet Heath and Wet Grassland, and these were the focus of the grazing trials. Control plots (grazing exclosures) were established in 2011 by the NPWS: one 50 x 50 m exclosure was built in each of the four habitats.

Thirty Dexter cattle, which included a small number of Dexter x Angus, ranged freely on the 462-ha site between July and October from 2011 to 2015 as per the grazing agreement. Cattle were tracked with GPS collars, and vegetation surveys were conducted each season to examine the impact of the grazing regime on the habitats of concern.

3.2 Aim and Objectives

Aim:

To examine the impact of conservation grazing with a traditional cattle breed on Annex I upland habitats.

Objectives:

1. To examine the impact of a low-density seasonal grazing regime on upland vegetation
2. To assess changes in the condition of three Annex I habitat types: Active Blanket Bog, Northern Atlantic Wet Heath with *Erica tetralix*, and European Dry Heath.

3.3 Methodology

3.3.1 Location and site description

Mount Brandon Nature Reserve is a 462-hectare statutory reserve located 35 km west of Tralee town on the northern edge of the Dingle Peninsula in county Kerry in south west Ireland (Figure 14). The Reserve was established in 1986 under Statutory Instrument No.420: (Nature Reserve (Mount Brandon) Establishment Order, 1986). The reserve makes up 3% of the Mount Brandon Candidate Special Area of Conservation (cSAC). Mount Brandon cSAC (Habitats Directive (92/43/EEC; site code 000375) is 14,355 hectares and is designated due to the presence of seven Annex I habitats including Blanket Bog, a priority habitat under the Directive, along with Northern Atlantic Wet Heath, Alpine and Boreal Heath, Vegetated Sea Cliffs, Chasmophytic vegetation and nutrient-poor lakes (NPWS, 2009). The site has further designation under Annex II due to the presence of freshwater pearl mussel *Margaritifera margaritifera* and Killarney Fern *Vandenboschia speciosa*. Más an Tiompán (763 m) is the highest peak in the reserve.

The underlying geology of the reserve is sandstone, conglomerates and siltstones of the Upper Devonian and Lower Carboniferous periods (Jackson, 1994; NPWS, 2009). The soils of the reserve are comprised of poorly drained peaty podzols with associated lithosols and blanket peat (NPWS, 2009). Sheep grazing predominated in the area until the Reserve was established in 1986 and since then it has been grazed intermittently by small numbers of stray sheep and by Kerry x Highland cattle during the late 1990s (Tim O' Donoghue, *pers. comm.,* 2013) but details on the number of animals and exact timing of the grazing are unclear. A herd of approximately 60 feral goats *Capra hircus* has been free-ranging in the Reserve since its establishment.

The habitats of the reserve (Figure 15) exist in intimate and complex mosaics but are dominated by a few primary habitat types; European Dry Heath (46%), Northern Atlantic Wet Heath (21%), Montane Heath (8%) and Blanket Bog (17%), all Annex 1 habitat types under the Habitats Directive (1992). Grasslands make up approximately 9% of the reserve, of which 2% is the Annex 1 habitat Siliceous Alpine and Boreal Grassland and the remainder is dry-humid acid grassland (4%) or wet grassland (1.4%).

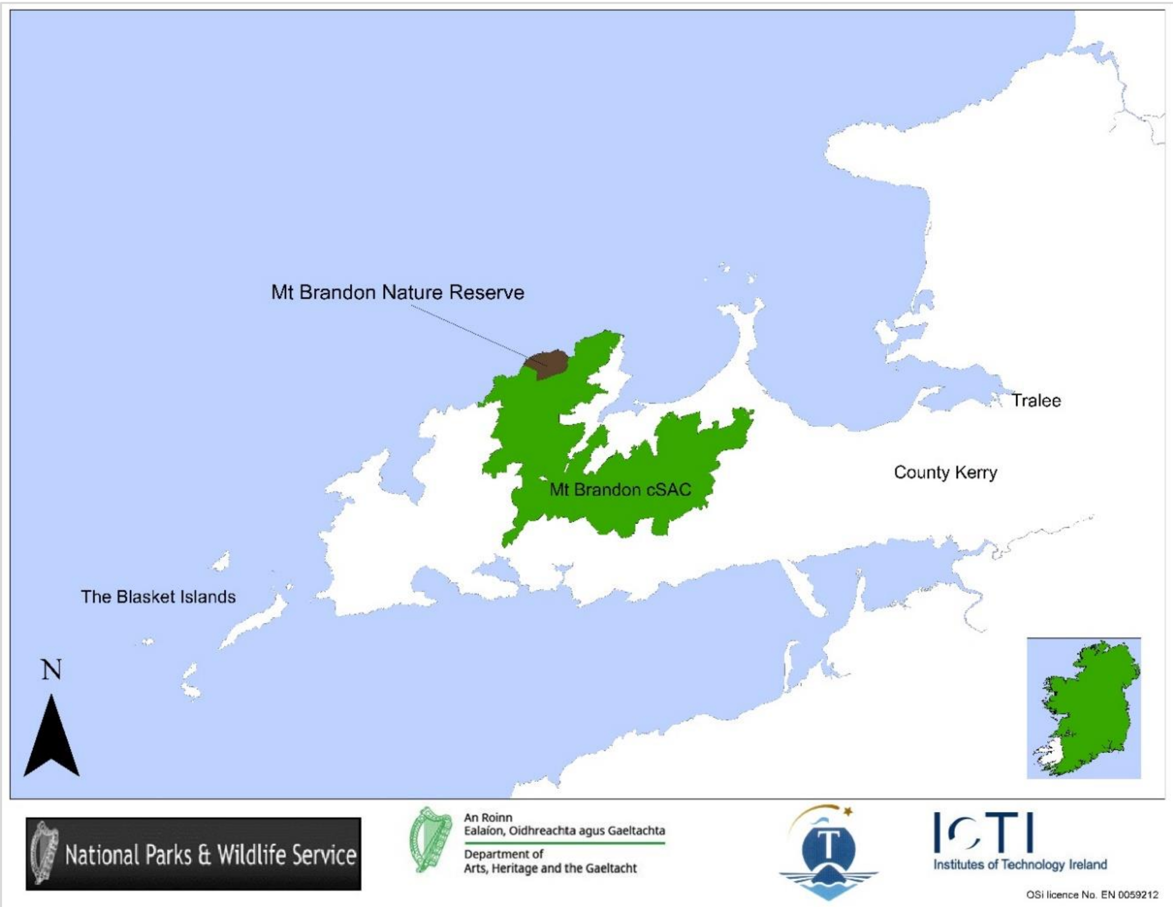


Figure 14 County Kerry, with Mt Brandon Nature Reserve and Mt Brandon cSAC

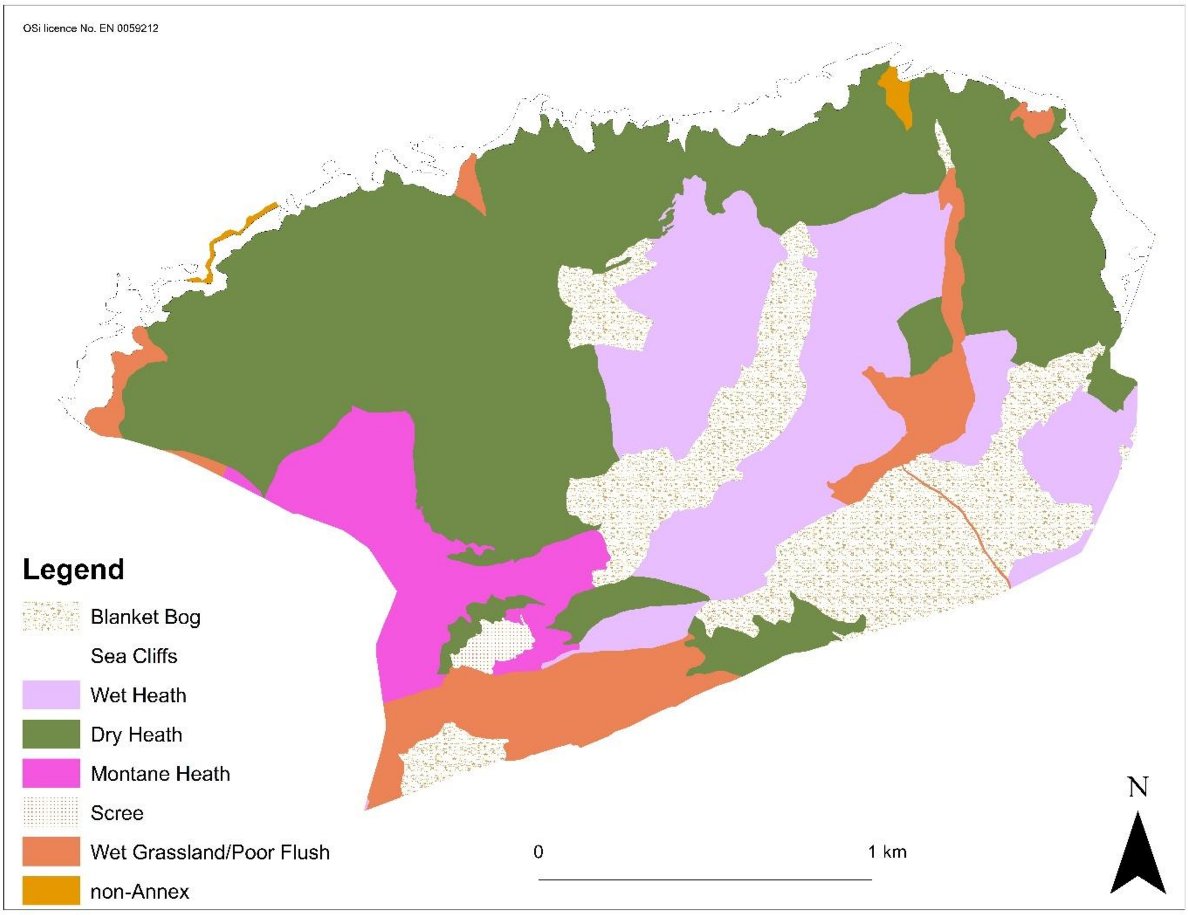


Figure 15 Habitats of Mt Brandon Nature Reserve

Figure 16 Selected images from Mt Brandon Nature Reserve.

3.3.2 Grazing Exclosures and Experimental Plots

Permanent 50 x 50 m control plots were established in 2011 by the NPWS, one in each of the four habitats under examination i.e. Dry Heath, Wet Heath, Blanket Bog and Wet Grassland (Figure 17). To examine the impact of grazing on the habitats experimental (grazed) plots of equal size were selected for vegetation sampling. A site assessment was carried out at the beginning of the projects first field season (February 2013) to delineate the maximum home range of the cattle from the previous season. The study area was walked, and presence/absence of the cattle was recorded based on evidence of defoliation, the presence of dung and poaching. Where evidence was noted, the GPS locations were recorded using and photographs were taken. GPS locations were imported into ArcMap 10.0 (ESRI, 2010) and the extent of the use of the site was determined by linking all the outermost GPS locations, thus producing a polygon approximating the home range from 2012. This was plotted over existing site maps and a 100 x 100 m grid was placed over it. The approximate home range was plotted and estimated to be 75 hectares for 2012.

Sixteen random plots (100 x 100 m) were then selected from within this home range for vegetation sampling. The selected plots were divided into four 50 x 50 m sub-plots to match the fenced controls and one of these was then selected at random for sampling. GPS coordinates for selected plots were extracted using GIS. Figure 18 shows the workflow for plot selection.

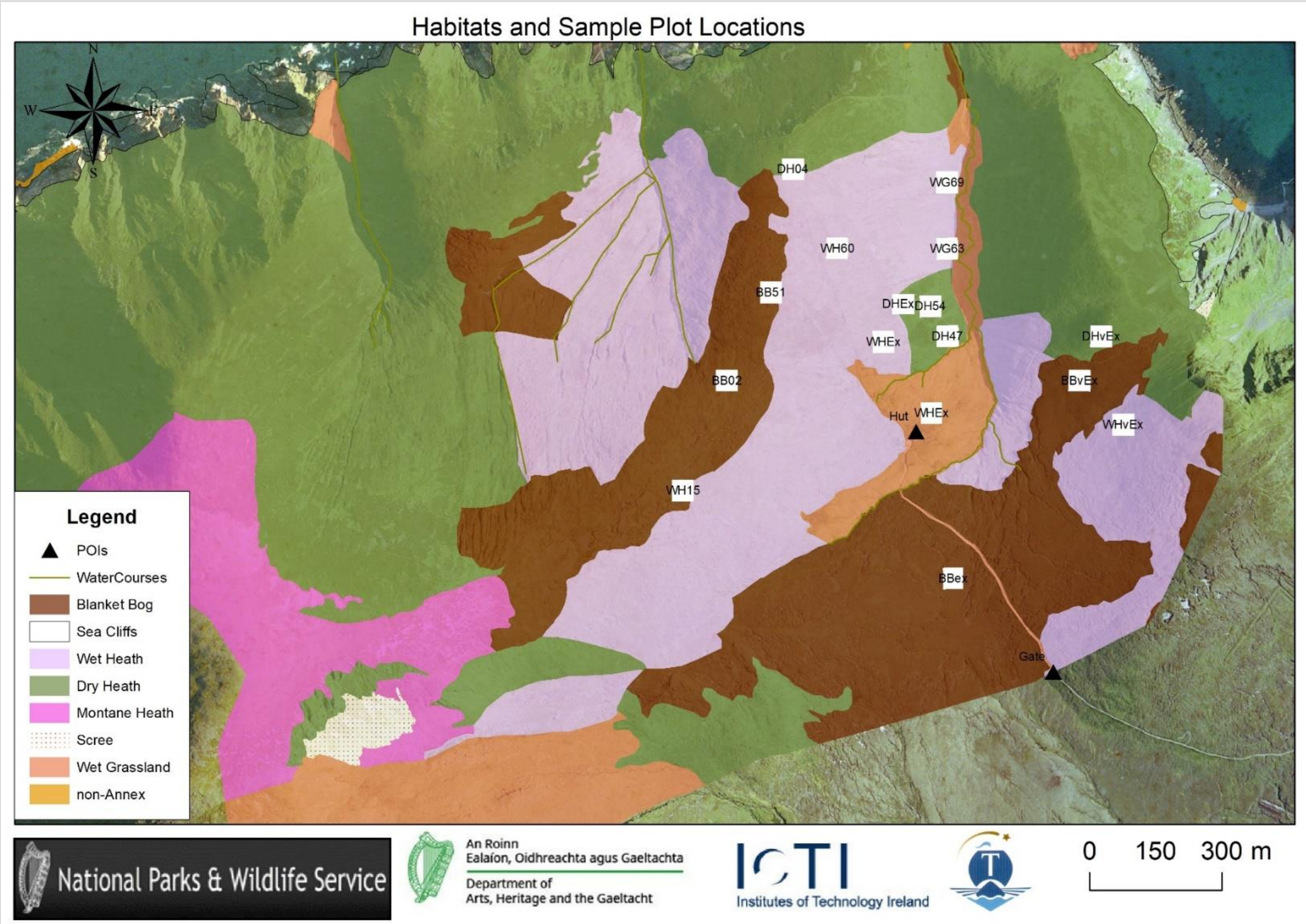


Figure 17 Location of sample plots and grazing exclosures in Mt Brandon Nature Reserve

The results of the GPS tracking in 2013 were used to verify that the selected plots were within the area grazed by the cattle. To increase the number of replicates of *un*grazed plots, the home range estimates derived from the GPS tracking were used to locate four additional 50 x 50 m plots, one in each of the habitats under examination, outside of the home range. Thus, for 2014 and 2015 a total of sixteen plots (2 grazed + 2 ungrazed x 4 habitats = 16) were selected for subsequent vegetation sampling.

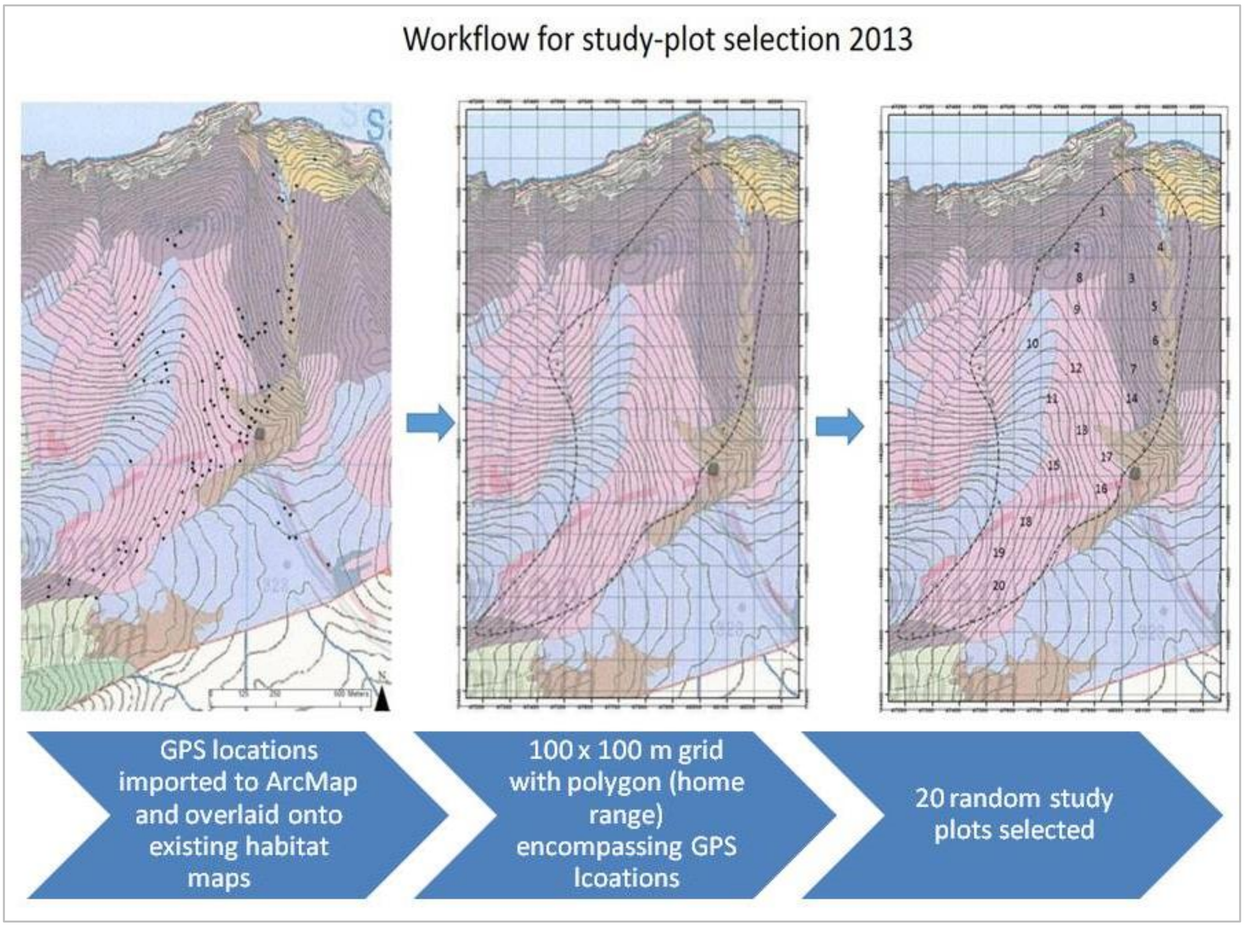
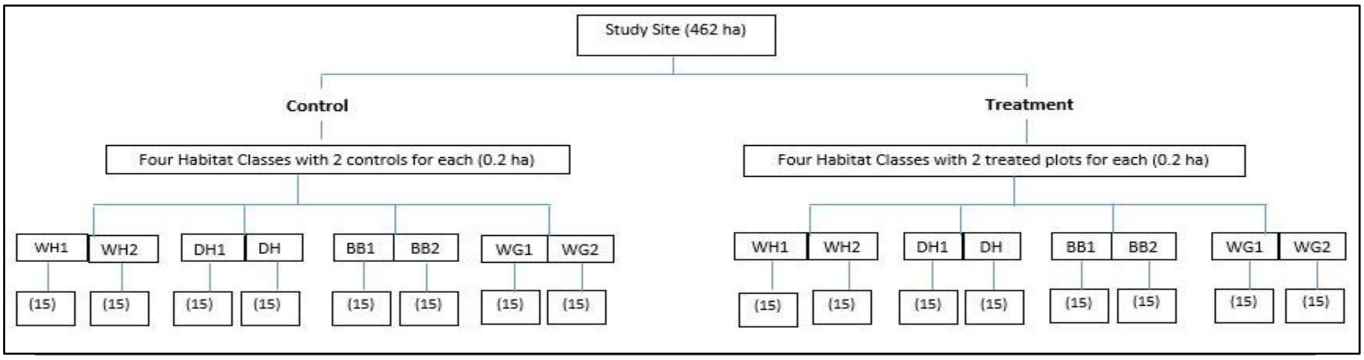


Figure 18 Illustration of workflow of random plot selection for vegetation sampling in 2013

Fifteen randomly located 2 x 2 m quadrats were completed in each plot in 2013 (i.e. 240 grazed and 180 ungrazed). Random number generating functions were used to designate the location of each quadrat, which were subsequently marked out in the field using GPS, compass and pacing techniques.

The number of plots was reduced in 2014 and 2015 to sixteen i.e. 2 grazed plots and 2 ungrazed controls for each of the four habitats. Fifteen quadrats were then completed in each plot (total = 240 quadrats in each year) (Figure 19).

Figure 19 Experimental design for vegetation sampling



Code	Explanation
Treatment	Experimental plots i.e. 50 x 50 m *grazed* plots
Control	Experimental controls i.e. 50 x 50 m *ungrazed* plots
WH	Wet Heath
DH	Dry Heath
BB	Blanket Bog
WG	Wet Grassland
(15)	The number of samples in the plot

The complex mosaic nature of the habitat patches in the study area was such that the 50 x 50m sampling plots were found to contain small patches of other habitat types e.g. a 50 x 50m wet heath plot typically contained smaller patches of blanket bog depending on local topography and soil depth. To capture such small-scale variation, all 2 x 2m quadrats were reclassified during data processing and completion, based primarily on their vegetation composition and soil depths. The habitat key and descriptions in the provisional scheme of the NSUH (Perrin et al., 2009) were used to reclassify every quadrat.

Due to the variability at small spatial scales, when quadrats within 50x50m habitat plots were reclassified, some were not indicative of the overall plot and were thus were pooled with the appropriate habitat samples).

Five wet heath plots that were sampled in 2013 were not sampled in 2014 or 2015 as they were outside the area of interest once home range had been established (i.e. vegetation sampling in 2013 was prior to the home range study). A total of 513 quadrats were suitable for analyses when the quadrats had been reclassified. Data from quadrats of the same habitat classification were then averaged for each 50 x 50 m plot.

Table 20 Habitats sampled, and the total number of quadrats completed in each 2013 – 2015.

	2013		**2014**		**2015**		
	Grazed	Ungrazed	Grazed	Ungrazed	Grazed	Ungrazed	Total
Wet Heath*	120	15	30	30	30	30	255
Dry Heath	15	30	30	30	30	30	165
Blanket Bog	15	30	30	30	30	30	165
Wet Grassland	15	30	15	30	15	30	135
Total	225	105	105	120	105	60	720

*Wet Heath made up 46% of the home range of the cattle over 9 samples, thus sampling was conducted to reflect this proportion (35% of all quadrats were wet heath).

Table 21 Number of samples in each habitat after reclassification and averaging of quadrats.

	Grazed	**Ungrazed**
Wet Heath	17	8
Dry Heath	4	5
Blanket Bog	4	5
Wet Grassland*	NA	NA

*when quadrats within the grassland plot were classified, they were found to be wet heath.

3.3.3 Sampling methods

Vegetation sampling was carried out for quadrats following the methods of Perrin et al. (2014b) and Dunne (2000). The methods of Perrin et al. (2009) have been developed for the National Survey of Upland Habitats (NSUH), while the methods developed by Dunne (2000) focus specifically on grazing impacts on upland habitats. These include measurements of vegetation structure as well as assessment of positive and negative indicator species for the specific habitats involved.

All vascular plants, bryophytes and macrolichens contributing cover within a quadrat were identified to species level in the field where possible, and bryophyte samples were collected for later verification when necessary. Nomenclature followed Stace (2010) for vascular plants, Smith (2004) for mosses, Paton (1999) for liverworts and Smith et al. (2009) for lichens. Percentage cover of each species was estimated to the nearest 5%, except for covers of less than 5%, which were recorded as 0.1, 0.3, 0.5, 0.7, 1, 3 or 5%. Percentage cover was recorded for leaf litter, bare soil, exposed rock, surface water and algal covered peat. The percentage cover of the vegetative layers was also recorded i.e. dwarf shrub layer, field layer and bryophyte layer. Soil depth, slope and the median height of the main vegetative layers was also recorded for each quadrat. Photographs, GPS location, aspect and altitude were also recorded for each quadrat.

3.3.4 Data Analysis

3.3.4.1 Habitat descriptions

Data for habitat descriptions were mined from a GIS dataset provided by the NPWS arising from the National Survey of Upland Habitats Site Report No7: Mount Brandon cSAC (00375), Co Kerry (Perrin et al., 2014b). The cSAC dataset was imported into ArcMap10.0 (ESRI, 2010) and clipped to the outline of Mt Brandon Nature Reserve (which makes up just 3% of the 14,355 ha cSAC). Attribute tables were queried to extract habitat data that were then used to describe the habitats of the Reserve. Detailed habitat descriptions based on these data are presented in section 3.4.2.

3.3.4.2 Multivariate Analysis

Analysis was carried out using PC-Ord version 6.17 (McCune and Mefford, 2011) and SPSS Statistics 24 (IBM Corp., 2013). Non-metric Multi-dimensional Scaling (NMS) was carried out in PC-Ord. NMS is primarily used for displaying relationships between quadrats and environmental variables. NMS avoids the assumption of linearity between variables, is suitable for non-normal data and allows any distance measure to be used. It is also less prone to outliers than other ordination methods as it is based on rank distances (McCune, Grace and Urban, 2002).

An initial autopilot NMS using Quantitative Sørensen (Bray Curtis) as a distance measure was conducted. It was run with 250 runs of real data and 250 runs of randomised data, stepwise reduction in dimensionality with each cycle, a stability criterion of 1×10^{-7} standard deviations in stress over 10 iterations and a maximum of 500 iterations.

The final ordination was run using 2 axes, stepwise reduction in dimensionality (step length = 0.20), a stable criterion of $1x10^{-7}$ standard deviations in stress over 10 iterations and a maximum of 500 iterations. The best starting configuration and optimal number of axes was found based on the autopilot run. The optimal number of axes was determined as the number of axes beyond which reduction in stress was small.

Environmental variables, including grazing intensity as a variable, were overlain on NMS ordinations. The correlation co-efficient was calculated between environmental variables and axes of the NMS plots to determine the relationship between community composition and environmental factors (McCune, Grace and Urban, 2002). The variables were mainly non-parametric, so Spearman Correlation was used to analyse correlations between environmental variables.

3.3.4.3 Univariate Analysis

Modelling

The approach for modelling was to explore the factors that influence various measures of quality appropriate to each habitat (e.g. species richness, cover of positive indicator species, cover of bare ground etc.), and to examine the influence of Utilisation (cattle use) as a factor in these outcomes. Two approaches were taken:

1. General Linear Models (GLMs) in R (RStudio Team, 2018)
2. Spearman rank correlations and stepwise multiple regressions were carried out in SPSS v.25 (IBM Corp., 2017) to explore the relationships between predictor variables and selected monitoring criteria.

1. **GLM in R** (RStudio Team, 2018)

GLMs were constructed using the 'lme4' package (Bates et al., 2015) in R Studio (RStudio Team, 2018). These models were used to examine the effects of environmental variables on species richness and 'applicable conservation status monitoring criteria' for the habitats under consideration. Before proceeding, autocorrelation between variables was examined with Spearman rank correlation coefficient (ρ) using the R package 'Hmisc' (Harrell, 2016). Highly correlated factors were removed from analysis. Where necessary, Bonferroni corrections were used to adjust p values to counteract issues with multiple comparisons.

Autocorrelated factors that were removed from analysis:

- ‘Graze’ and ‘UI’ (rho = 0.72, $p<0.001$): decision - remove ‘Graze’. Consider ‘utilisation’ as ‘graze’ when interpreting results.
- For wet heath: ‘PosIndCov’ and ‘PosBryoCov’ (rho = 0.72, $p<0.001$): decision – remove ‘PosIndCov’. Closely aligned species list. Consider ‘PosBryoCov’ as a proxy for ‘PosIndCov’ when interpreting results.
- ‘PosBryoCov’ with ‘BryoCov’ (rho = 0.72, $p<0.001$): decision – keep ‘BryoCov’.
- ‘H’ with ‘Evenness’ (rho = 0.941, $p<0.001$), ‘Simpsons’ with ‘Evenness’ (rho=0.95, $p<0.001$), and ‘Simpsons’ with ‘Shannons’ (rho=0.96, $p<0.001$). All Biodiversity indices all highly autocorrelated. Decision: remove Simpson’s and Evenness from models, keep Shannons.

Models were checked for homogeneity by plotting standardised residuals against fitted values. Probability-Probability (P-P) plots and partial regression scatter plots were examined to assess normality of standardised residuals, linearity and homoscedasticity.

2. **Spearman Rank Correlations and Stepwise Multiple Regression in SPSS**

Applicable conservation status monitoring criteria that are used to assess the status of Annex I habitats were selected for each habitat. The appropriate criteria for each habitat were taken from the Guidelines for the National Survey of Upland Habitats (NSUH) (Perrin et al., 2014a). Species richness was also included in the analysis. A full description of these criteria follows in section 3.3.4.5 ‘Conservation Status

Assessments'. Table 23 provides a list of criteria for each habitat. For the purposes of modelling these are the 'outcome variables.

Spearman rank correlations and stepwise multiple regressions were carried out to explore the relationships between predictor variables (slope, Utilisation etc.) and selected criteria (outcome variables). Criteria that were not applicable were discounted. For example, no negative indicator species were recorded in wet heath during the study, so the criterion '*cover of negative indicators*' was excluded from analysis.

Predictors under examination included environmental variables measured in the field (slope, bare ground and soil depth), plus Utilisation scores generated from home range analysis of GPS data (described in Chapter 2, section 2.2.4).

Since no *a priori* hypotheses had been made to determine the order of entry, stepwise backward multiple linear regression was used, which is suitable for exploratory analysis (Field, 2013, p.232). Mahalanobis' distance was used to identify extreme outliers which were removed if necessary. Mahalanobis' distance measures the distance from the mean of a predictor variable to individual cases and it assesses the influence of outliers on a fitted model. Values greater than 15 can be problematic and cases should be examined (Barnett and Lewis, 1974).

Variance Inflation Factors (*VIFs*) were examined for collinearity among explanatory variables and were considered within acceptable limits if individual values were below

10 and average values were close to 1 (Bowerman and O'Connell, 1990; Myers, 1990; Menard, 1995).

Standardised residuals were examined to ensure at least 95% of cases were within ± 2 standard deviations. Cook's distances were inspected to check for influential cases and were considered acceptable if within the recommended boundary of three times the average (Cook and Weisberg, 1982; Stevens, 2002). Models were checked for homogeneity by plotting standardised residuals against fitted values.

Probability-Probability (P-P) plots, partial regression scatter plots and histograms were used to assess for normality of standardised residuals, linearity and homoscedasticity. Where potential violations were identified, regressions were rerun based on 1000 bootstrapped samples with 95% bias corrected and accelerated confidence intervals, which do not rely on assumptions of normality or homoscedasticity (Field, 2013, p.352). The Durbin-Watson statistic was used to assess the assumption of independent errors in each model (Durbin and Watson, in Field, 2013).

3.3.4.4 Rank abundance curves

Rank abundance curves (Whittaker plots) were constructed to visualise changes in diversity and dominance patterns at a plot level over multiple years. Rank abundance plots can be used for representing community structure, whereby species are ranked by abundance and the proportion of the total number of individuals belonging to each species is plotted against the species rank.

Rank abundance plots visualise relative species abundances in a sample. They also depict species richness and evenness. The x-axis reflects the abundance rank, with the most abundant species given a value of 1, the second most abundant a rank of 2 etc. The y-axis indicates the relative abundance and the length of the plotted line shows the species richness of the sample. The slope of the plotted line indicates evenness, whereby a steep slope demonstrates low evenness, with high-ranking species having much higher abundances than the low-ranking species. Shallow gradients on the plotted lines indicate high evenness, whereby the abundances of different species is similar (Whittaker, 1965; Clarke, 1990; Wilson, 1991). Rank abundance plots were prepared using MS Excel.

Note that some plots were sampled in 2013, 2014 and 2015, while others were only sampled in 2014 and 2015. For the purposes of producing rank abundance curves samples from 2013 and 2014 were pooled and labelled 'early' and samples from 2015 were pooled and labelled 'late' in the rank abundance figures.

3.3.4.5 Conservation Status Assessments

Habitat conservation status assessments were carried out following the guidelines of National Survey of Upland Habitats (NSUH) (Perrin et al., 2014a). These assessments examine four areas: (1) Range, (2) Area, (3) Structure and Functions and (4) Future Prospects. In this study 'Range' and 'Area' are limited to the site boundaries so were not assessed.

The 'Structure and Functions' criterion relates to vegetation composition (presence and cover of positive and negative indicator species, cover of characteristic species etc), vegetation structure and physical structure of the habitat at particular points in space and time. Assessments were carried out on data from all sampling seasons to reveal trends over time. Conservation Assessments were carried out on all quadrats using NSUH guidelines (Perrin et al., 2014a) and adapted where appropriate to reflect the study site. The full conservation status monitoring criteria for Northern Atlantic wet heaths with *Erica tetralix* (4010), European dry heaths (4030) and Blanket bogs (9130) are described Perrin et al (2014). Table 22 shows applicable criteria for each habitat in this study.

The NSUH methods recommend that 4 monitoring stops be conducted for an area 0.04 – 10 ha (Perrin et al., 2011a). Sample plots in this study were 0.25 ha, therefore 4 stops per plot would be required. However, 15 quadrats were completed in each plot and all of these were used to assess conservation status. Quadrats were randomly located within each plot, which is in line with NSUH methods. Monitoring stops are usually conducted in the field. In this study conservation assessments were done *post-hoc* in Excel in order to save time in the field.

As per NSUH guidelines, monitoring stops with no failed criteria were automatically passed. Stops with 3 or more failed criteria were deemed to have failed. Intermediate stops with one or two failed criteria were judged on an individual basis and passed if deemed acceptable ecologically. For example, in wet heath the cover of positive indicator species must be ≥ 50%. Due to the subjective nature of assessing per cent cover, 10% margin of error was given on this criterion.

To monitor change over the period of the study, values for each criterion were calculated (rather than just presence/absence) for all quadrats and averaged at the plot level to give values for each sample area. As an example, Table 23 illustrates this for the Wet Heath exclosure.

Table 22 Applicable monitoring criteria for Annex I upland habitats.

Northern Atlantic Wet Heath with *Erica tetralix*.

	Criterion	Scale	Used (√)
1	*Erica tetralix* present	20 m radius	N/A
2	Cover of positive indicator species* ≥ 50%	4 m^2	√
3	Total cover of: *Cladonia* sp. *Sphagnum* sp., *R. lanuginosum* and pleurocarp mosses ≥ 10%	4 m^2	√
4	Cover of ericoid species ≥ 15%	4 m^2	√
5	Cover of dwarf shrub species < 75%	4 m^2	√
6	Cover of neg. indicators: *A. capillaris*, *H. lanatus*, *R. repens*, collectively < 1%	4 m^2	N/A
7	Cover of non-native species < 1%	4 m^2	N/A
8	Cover of non-native species < 1%	Vicinity	N/A
9	Cover of *Pteridium aquilinium* < 10%	Vicinity	N/A
10	Cover of *Juncus effusus* < 10%	Vicinity	N/A
11	Cover of disturbed, bare ground < 10%	4 m^2	N/A

**B. chrysocoma, C. vulgaris, Carex* sp., *D. albicans, Drosera* sp., *E. tetralix, E. angustifolium, N. ossifragum,* non-crustose lichens, *P. sylvatica, P. purpurea, P. serpyllifolia, P. erecta, Sphagnum* spp., *S. pratensis, T. germanicum.*

European Dry Heath.

1	No. of bryophyte or non-crustose lichen species present excl. *Campylopus* spp. and *Polytrichum* spp. ≥ 3	4 m^2	√
2	No. of positive indicator species* present ≥ 2	4 m^2	√
3	Cover of positive indicator species* ≥ 50	4 m^2	√
4	Cover of weedy neg. indicator sp. (*Cirsium* sp., *R. repens, R. acetosa, U. dioica, S. jacobea* collectively < 1%	4 m^2	N/A
5	Prop. of dwarf shrub cover composed of *Myrica gale, Salix repens and Ulex gallii* collectively < 50%		N/A
5	Cover of non-native species < 1%	Vicinity	N/A
6	Cover of non-native species < 1%	Vicinity	N/A
7	Cover of *Pteridium aquilinium* <10%	4 m^2	N/A
8	Total cover of the negative indicator species	4 m^2	N/A
9	Cover of *Juncus effusus* < 10%	4 m^2	N/A
10	Cover of disturbed, bare ground < 10%	4 m^2	N/A
11	Cover of disturbed, bare ground < 10%	Vicinity	N/A

**C. vulgaris, E. cinerea, U. gallii, V. myrtillus.*

Active Blanket Bog.

1	Number of positive indicator species* ≥ 7	4 m^2	√
2	Cover of bryophyte or lichen species > 10%	4 m^2	√
3	Cover of bryophyte or lichen species, excluding *S. fallax* ≥ 10%	4 m^2	N/A
4	Cover of each of following < 75%: *C. vulgaris, E. vaginatum, M. caerulea* and *T. germanicum*	4 m^2	√
5	Total cover of neg. indicator spp. (*A. capillaris, H. lanatus, P. aquilinium*) < 1%	4 m^2	N/A
6	Cover of non-native species < 1%	4 m^2	N/A
7	Cover of non-native species < 1%	4 m^2	N/A
9	Cover of bare ground < 10%	4 m^2	N/A
9	Cover of bare ground < 10%	Vicinity	√

**B. chrysocoma, C. vulgaris, D. albicans, D. rotundifolia, E. tetralix, E. angustifolium, E. vaginatum, N. ossifragum,* non-crustose lichens, *P. sylvatica, P. lusitanica, P. purpurea, P. serpyllifolia, R. languinosum, Sphagnum* species (count separately and ignore *S. fallax*), *V. myrtillus.*

Table 23 An example conservation status monitoring table for Wet Heath Exclosure 2013-2015.

Plot 1: u1 (Wet Heath Exclosure)				
		2013	**2014**	**2015**
Criterion		**Mean (±*s*)**	**Mean (±*s*)**	**Mean(±*s*)**
1	*Erica tetralix* present	0.9 (0.2)	0.9 (0.2)	1.0 (0.0)
2	Cover of positive indicator species ≥ 50%	78.6 (38.2)	92.2 (21.1)	73.7 (35.2)
3	Total cover of: *Cladonia* sp. *Sphagnum* sp. *R. lanuginosum* and pleurocarp mosses ≥ 10%	47.7 (29.6)	29.4 (10.2)	30.2 (22.1)
4	...			
5	...			
6	...			

For criterion 1 '*Erica tetralix* present', a value of 0.9 means that *E. tetralix* was present in 13/15 quadrats. Criteria 2 shows the average cover of positive indicator species for the plot. Table 22 lists the positive indicator species for each habitat.

3.4 Results

Presentation and structure of results:

- An overview of the general trends is presented in section 3.4.1
- Section 3.4.2 gives a detailed description of the habitats in the study area from data provided by NPWS from the National Survey of Upland Habitats (Perrin et al., 2013)
- Section 3.4.3 shows results from the analysis of community data (multivariate statistics).
- Regression modelling using conservation status criteria is in section 3.4.5
- Results of conservation assessments are given in section 3.4.6

3.4.1 Overview and general trends

A total of 675 quadrats was completed over three sampling seasons. Overall, 119 plant species were recorded: 5 shrubs, 37 herbs, 4 ferns, 35 graminoids, 28 bryophytes, 4 lichens and 6 liverworts. A species list is provided in Appendix III

The most frequent and abundant species are presented in Table 24.

Table 24 Most frequent and abundant pant species in Mt Brandon Nature Reserve 2013-2015

	Most frequent species	(% freq.)	Most Abundant Species	(% cover)
1	*Potentilla erecta*	89	*Molinia caerulea*	27.2
2	*Calluna vulgaris*	87	*Sphagnum* sp.	22.2
3	*Molinia caerulea*	85	*Calluna vulgaris*	21.2
4	*Sphagnum* sp.	77	*Trichophorum germanicum*	5.2
5	*Erica tetralix*	67	*Potentilla erecta*	4.0
6	*Rhytidiadelphus loreus*	56	*Eriophorum angustifolium*	3.9
7	*Festuca vivipara*	54	*Nardus stricta*	3.4
8	*Nardus stricta*	54	*Hylocomium splendens*	3.3
9	*Carex panicea*	53	*Festuca vivipara*	2.8
10	*Hylocomium splendens*	51	*Erica tetralix*	2.3

Species richness was greatest in wet heath ($\bar{x} = 16.6$, $s = 4$), followed by blanket bog ($\bar{x} = 16.0$, s = 4) and dry heath ($\bar{x} = 14.1$, s = 3.5). For both wet and dry heath species richness was largest in the ungrazed samples. In blanket bog, grazed samples had slightly more species (Table 25)

Diversity values using Shannon's H' was 1.8 in both grazed and ungrazed wet heath, and 1.8 and 1.9 in grazed and ungrazed dry heath respectively. Diversity values were 1.8 in grazed blanket bog and 1.6 in ungrazed. Table 25 and Figure 20 & 21 summarise the species richness and diversity results.

Table 25 Species Richness and Diversity (Grazed v Ungrazed by habitat)

Species Richness and Diversity (Grazed v Ungrazed by habitat)					
Habitat			N	Mean	Std. Dev.
Wet Heath	Grazed	Sp.Rich	91	16.1	3.8
		Diversity (S-W)	91	1.8	0.4
		N	91		
	Ungrazed	Sp.Rich	205	16.9	4.0
		Diversity (S-W)	205	1.8	0.4
		N	205		
Dry Heath	Grazed	Sp.Rich	50	13.8	3.2
		Diversity (S-W)	50	1.8	0.2
		N	50		
	Ungrazed	Sp.Rich	55	14.4	3.8
		Diversity (S-W)	55	1.9	0.2
		N	55		
Blanket Bog	Grazed	Sp.Rich	75	16.5	3.9
		Diversity (S-W)	75	1.8	0.4
		N	75		
	Ungrazed	Sp.Rich	29	14.7	4.0
		Diversity (S-W)	29	1.6	0.5
		N	29		

Univariate statistical tests between variables was not carried out. Linear modelling techniques were used to explore relationships between variables.

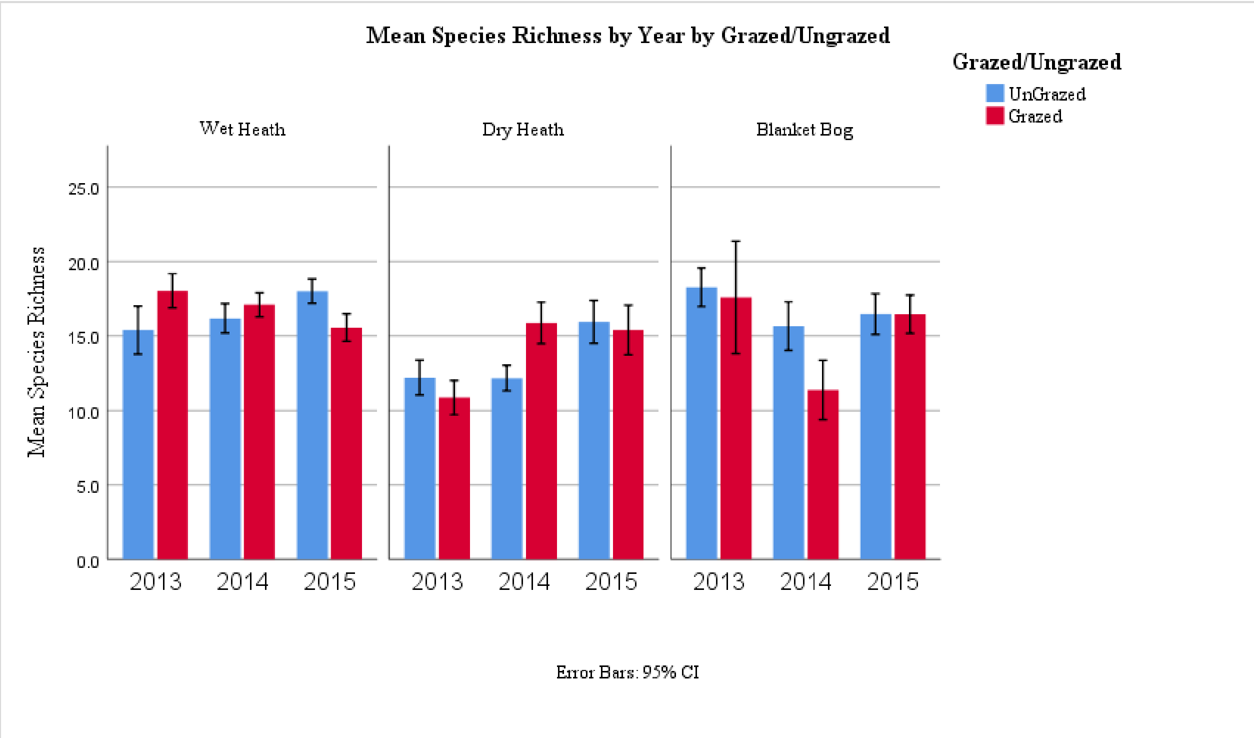


Figure 20 Mean Species Richness by habitat, year and grazing status

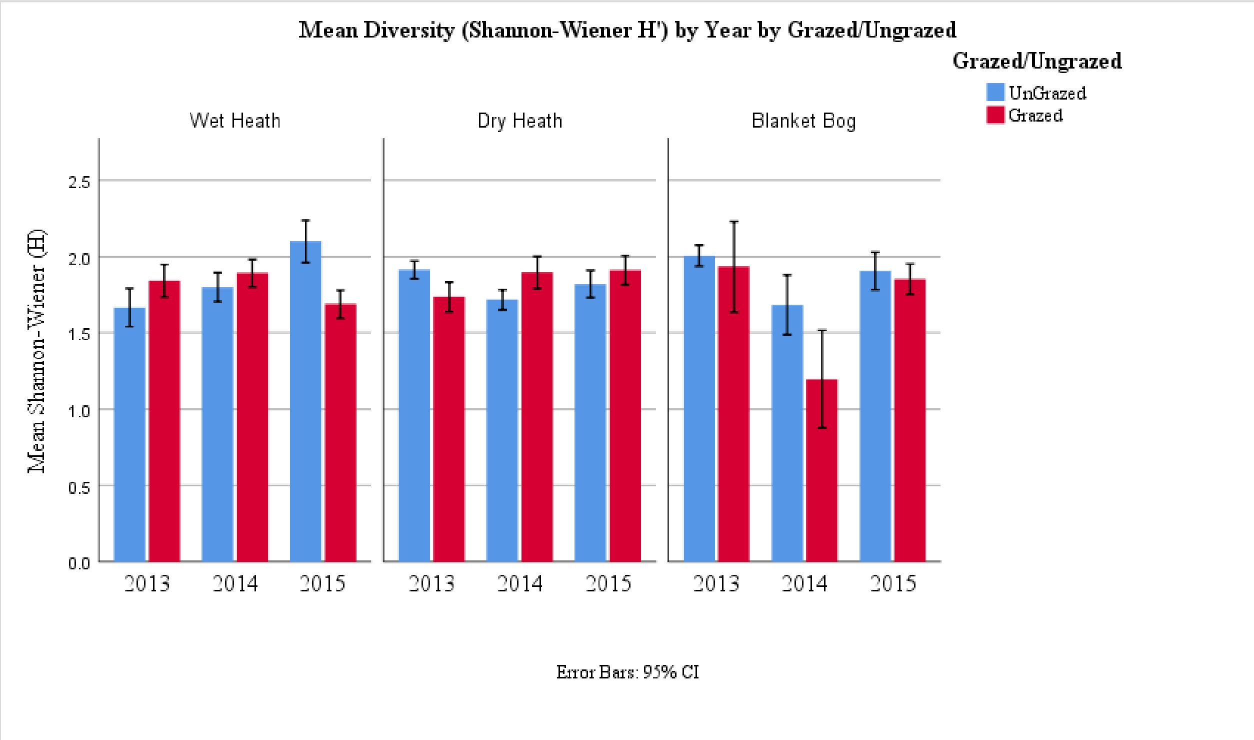


Figure 21 Mean plant species diversity (Shannon-Wiener) by year and Grazing Status

The percentage cover of bare soil by habitat and treatment type was low, with mean per cent cover of less than < 3% overall. There was marginally more bare soil in grazed areas compared to ungrazed areas. In wet heath, the average amount of bare soil declined from 2013 to 2015 in both grazed and ungrazed treatments. There was no change in ungrazed dry heath, with values of <0.1 % cover of bare ground. In the grazed dry heath, it declined from 1% to 0.3% between 2013 and 2014 and increased to 0.4% in 2015. No bare soil was recorded in ungrazed blanket bog. On the grazed sites it increased from 0% (2013 and 2014) to 1.3% in 2015. Figure 22 shows the percentage bare soil by year and treatment for each habitat.

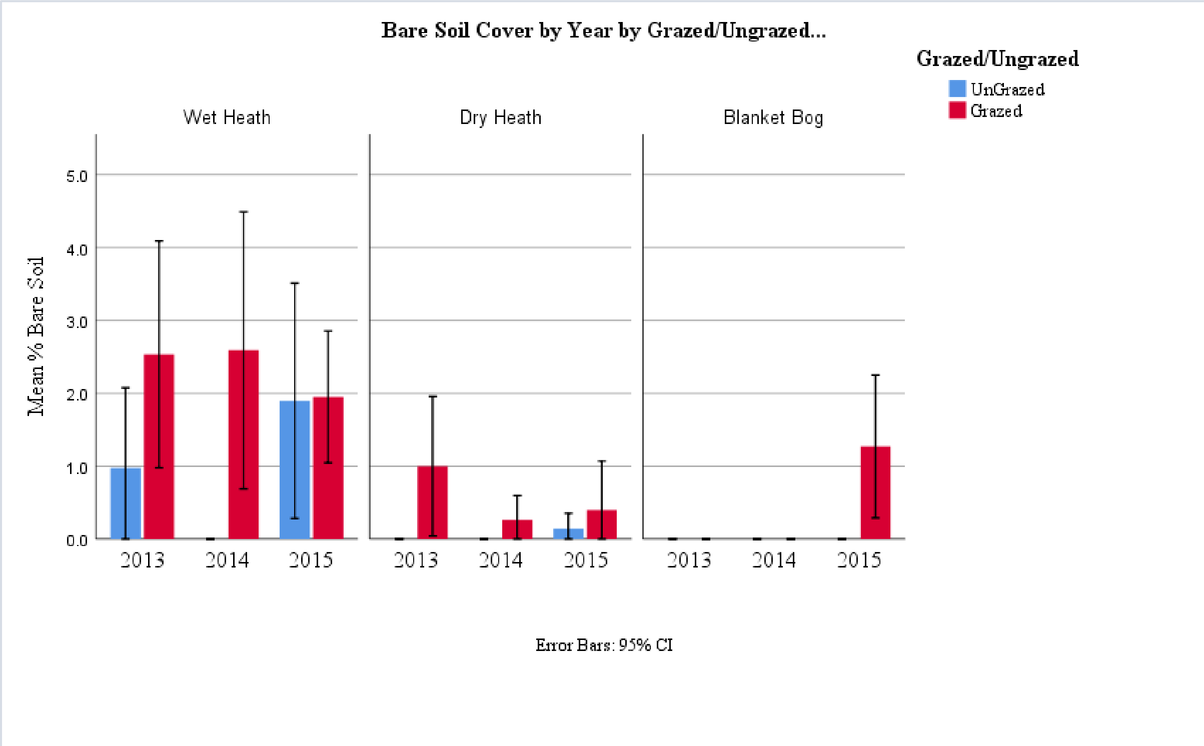


Figure 22 Mean percentage cover of bare soil by habitat, year and treatment

The ground layer cover (Figure 23) was higher in ungrazed than in grazed for blanket bog in each year. It was more variable for wet and dry heath but generally cover was greater in ungrazed samples. Similar patterns were found with the field layer cover (Figure 24), with ungrazed areas having more field cover than grazed areas.

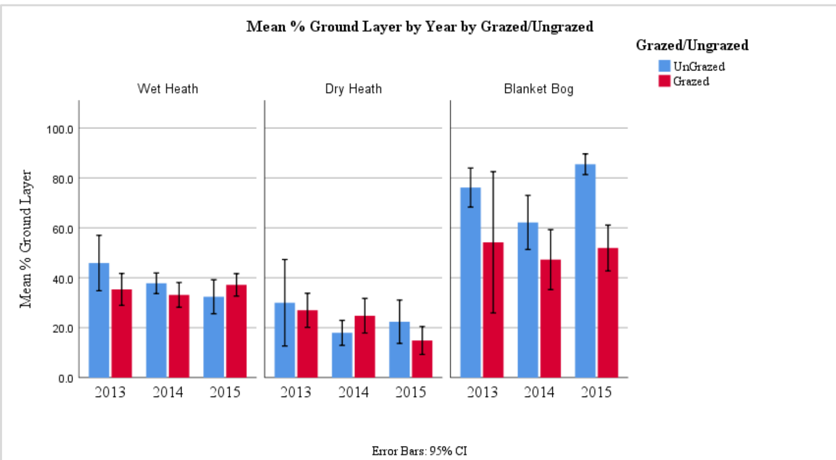


Figure 23 Percentage cover of ground layer for three habitats by year and by treatment.

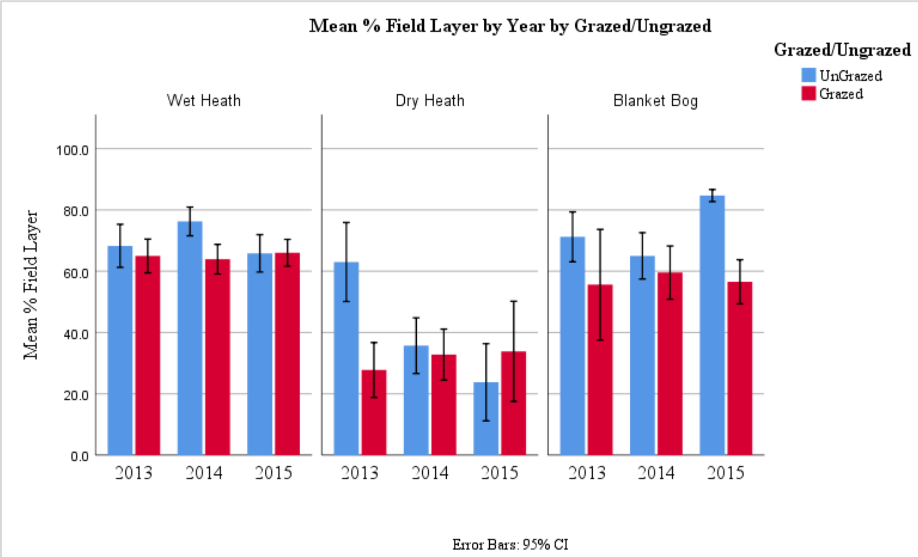


Figure 24 Percentage cover of field layer for three habitats by year and by treatment.

In wet and dry heath, the percentage cover of the shrub layer (Figure 25) was higher in ungrazed samples than in grazed. An exception to this trend was observed in 2013 in dry heath, where the grazed site had a higher percentage cover of shrubs than the ungrazed. The percentage cover of the shrub layer was higher in grazed blanket bog compared to ungrazed (Figure 25).

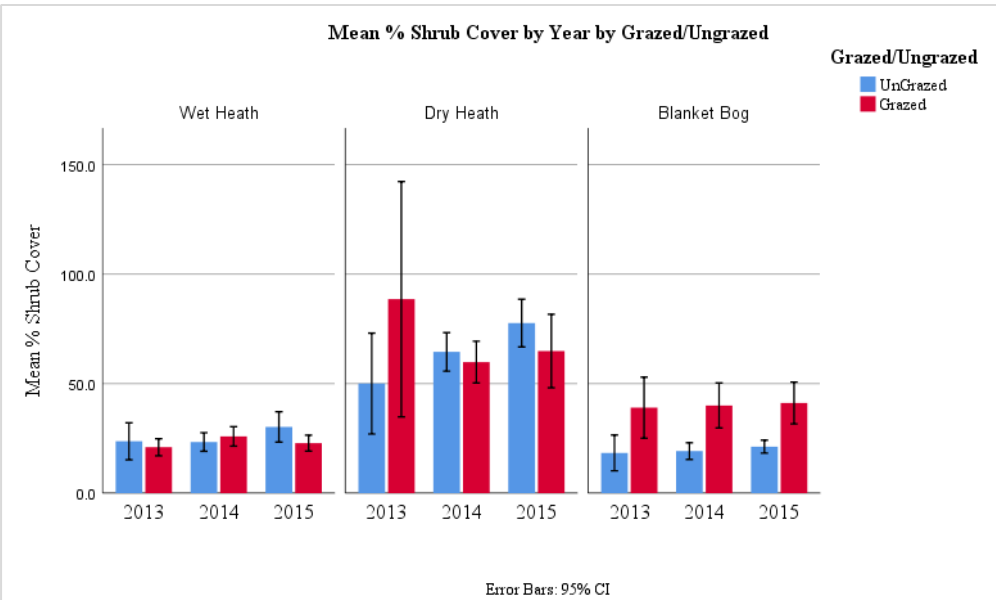


Figure 25 Percentage cover of shrub layer vegetation cover 2013-2015

Leaf litter cover was higher in ungrazed areas for all habitats across all years (Figure 26). Percentage cover of *Molinia caerulea* was higher in ungrazed samples for all habitats and years (Figure 27). Cover of *Molinia* in blanket bog and wet heath was high compared to dry heath. Values of 40% and 44% were recorded in ungrazed wet heath and blanket bog respectively, while grazed areas had values of 34.8% and 25%. Dry heath had an average cover of 13.7% in ungrazed areas and 1.7% in grazed areas.

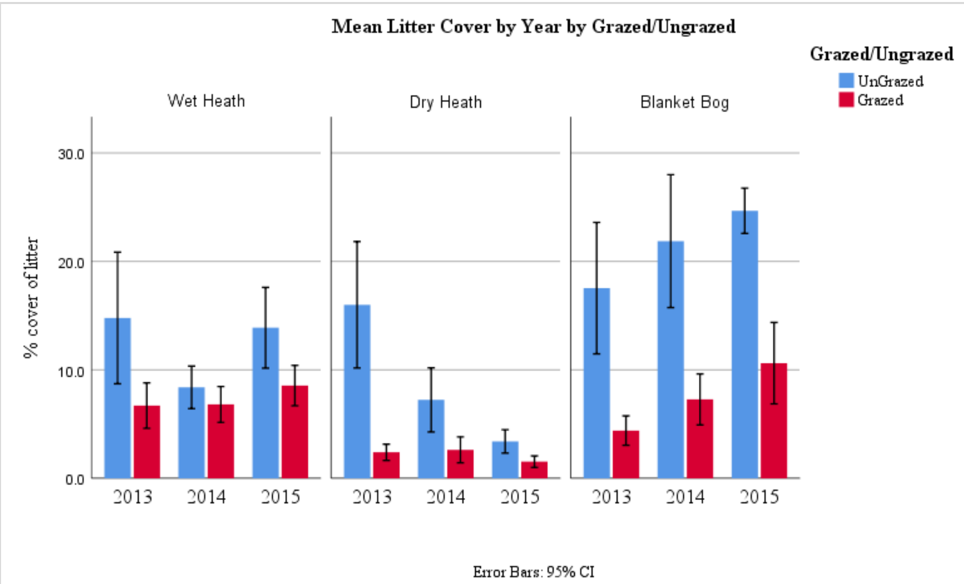


Figure 26 Percentage cover of leaf litter 2013-2015

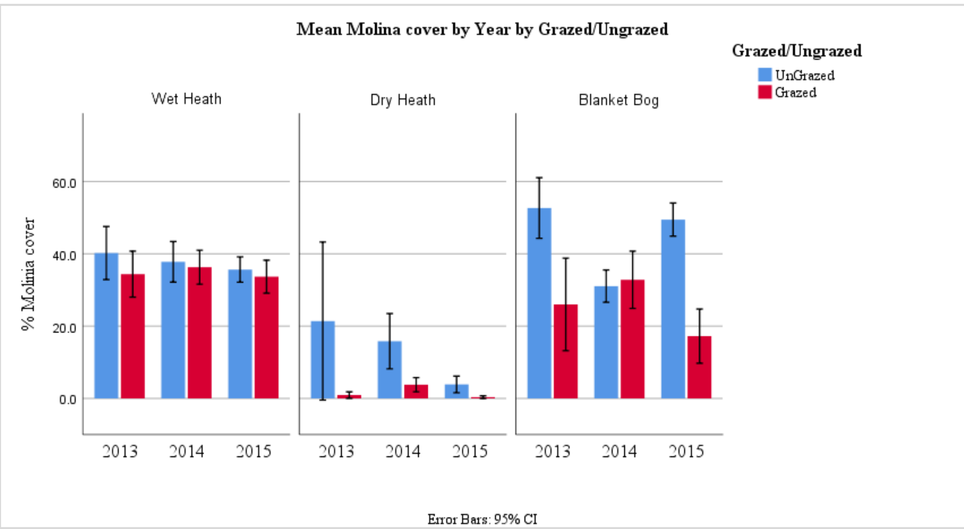


Figure 27 Percentage cover of *Molinia caerulea* 2013-2015

The height of the field and shrub layers (Figure 28 and Figure 29) correspond to the percentage cover values, with ungrazed areas having taller vegetation. The only exception is grazed dry heath in 2013 (discussed later).

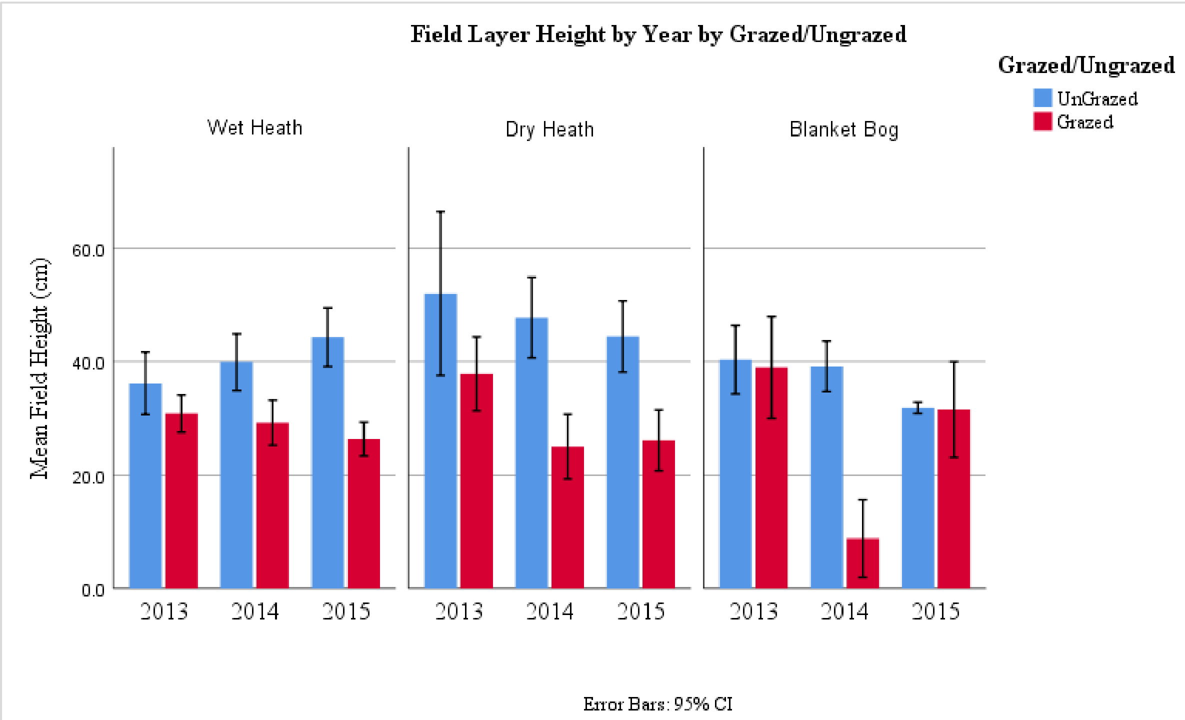


Figure 28 Average height of the field layer 2013-2015

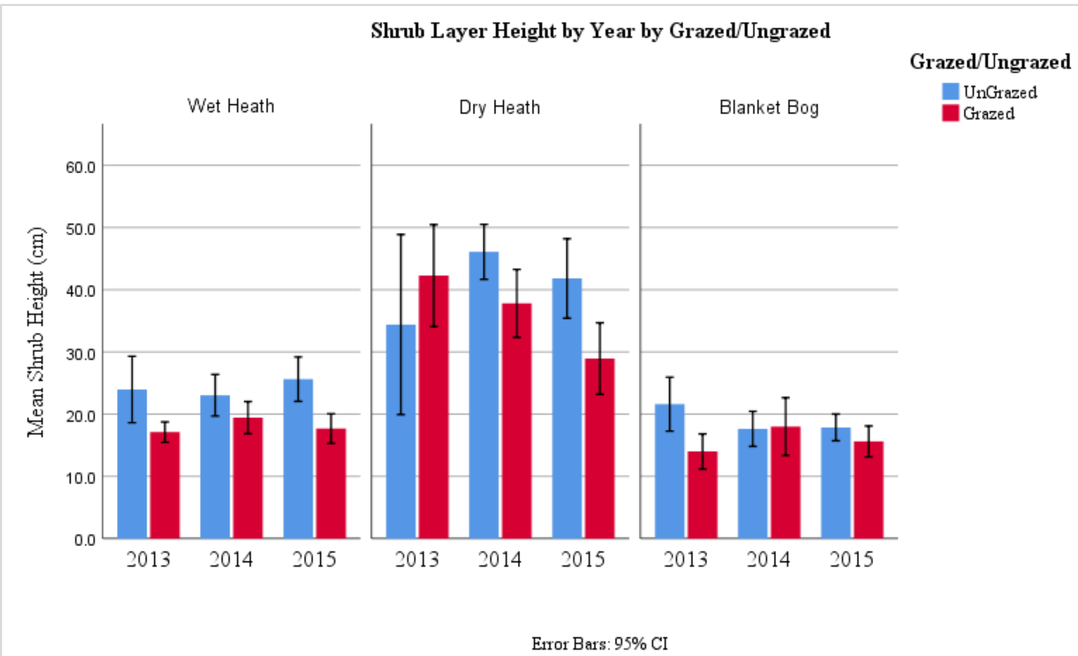


Figure 29 Average height of the shrub layer 2013-2015

3.4.2 Habitat descriptions

Dry Heath

The heath habitats occur on the shallow soils and peat of the Reserve's slopes. The European Dry Heath, Annex 1 category 4030 and Fossitt code HH1 Dry Siliceous Heath in the study area is aligned with DH1 of the NSUH *"Dry heath vegetation with* Ulex gallii, *accompanied by* Erica cinerea *and/or* Calluna vulgaris*; typically occurring in coastal areas"* (Perrin et al., 2011a). The soils of the dry heath are an average depth of 14 cm, the ground is sloping at approximately 25° and it has a dwarf shrub component of on average 61%. Frequently occurring dwarf shrubs are *C. vulgaris, E. cinerea and U. gallii* and the only commonly occurring broadleaf herb is *Potentilla erecta*. The sedge species occurring include *Carex binervis, Carex panacea, and Carex pulicaris,* and of the grasses, *Agrostis capillaris, Nardus stricta, Festuca vivipara* occur most frequently. *Molinia caerulea* does occur in this habitat but is usually not frequent or abundant. *Thuidium tamariscinum, Hylocomium splendens, Rhytidiadelphus loreus* and *R. squarrosus* occur frequently in the bryophyte layer and on average contribute 20% to cover on the ground layer.

Wet Heath

The wet heath of the reserve can be categorised as Northern Atlantic Wet Heath, Annex 1 code 4010 or Fossitt HH3. It is variable, as is typical of wet heaths (Perrin et al., 2011a), and grades between dry heath and blanket bog. It occurs on soils with a typical depth of 43 cm and quite variable slopes averaging 21 degrees. It resembles WH3 and WH4 of the provisional classification under the NSUH, with *C. vulgaris, M. caerulea* and *Sphagnum* spp. being the dominant species, typical of WH3. The often abundant *Trichophorum germanicum* brings some patches more in-line with the sub-community WH4. Typical species in the bryophyte layer of the reserve's wet heaths are *Breutelia chrysocoma, Hypnum jutlandicum* and *H. cupressiforme, Racomitrium languinosum, Rhytidiadelphus loreus, Sphagnum* spp. and *Thuidium tamariscinum.*

Blanket Bog

Blanket Bog occurs on flatter slopes of the reserve (0 to 12°) on an average soil depth of 67 cm and frequently greater than 1 m. The vegetation cover is variable depending both on these factors and on the level of grazing or erosion experienced by the patch. However, it is characterised by the peat forming species of *Sphagnum* spp., *Eriophorum* spp., *R. languinosum* and *M. caerulea*. All the blanket bog in the reserve is classified as Annex I *7130 and Fossitt PB2 but sub-classes (*sensu* NSUH classification) occur, depending on elevation and slope.

Approximately 50% of it is closest to community BB4 *Trichophorum germanicum – Eriophorum angustifolium* bog. A further 40% of the blanket bog in the reserve is sub-community BB5a, *Calluna vulgaris – Eriophorum* spp. bog, which is high altitude bog that can be drier than BB4 and lacking in *Sphagnum* cover.

Sphagnum cuspidatum/denticulatum bog hollows occur (upland variant HW1i), as do *E. angustifolium – Sphagnum fallax* hollows (upland variant HW2i) but not frequently and were not encountered during vegetation sampling for this work.

Grasslands

The grasslands of the reserve occur on ground with an average slope of approximately 18° and on soils of about 27 cm. The dwarf shrub component is reduced compared to the heaths and bogs with cover of 10 to 25%. Grass species dominate, with *Potentilla erecta* and *Gallium saxatile* as the main herb component. The Fossitt (2000) classifications for the grasslands of the reserve are Wet Grassland GS4 which amounts to 10 ha, dry-humid acid grassland GS3 (12 ha) and 1.6 ha of Dry calcareous and neutral grassland.

Almost 50 % of the wet grassland (non-annexed, Fossitt GS4) poor flush PFLU3 *Juncus acutiflorus/effusus – Calliergonella cuspidata* flush and around 15% is PFLU2 *Juncus effusus – Sphagnum cuspidatum/palustre* flush. Some 10% of the wet grassland is *Pteridium aquilinium* community BK1.

Under the provisional NSUH classification scheme, UG2a '*Nardus-stricta – Galium saxatile* upland grassland' makes up 20% of the wet grassland patches, with *Juncus squarrosus* or *Sphagnum* spp. being prevalent and include *Festuca ovina, Anthoxanthum odoratum, Danthonia decumbens, Rhytidiadelphus squarrosus* and *Hylocomium splendens.*

There are 12 hectares of acid grassland (Fossitt GS3) in the reserve and it is classified as either UG2A or UG2B *Nardus stricta – Galium saxatile* upland grassland with small patches of *Vaccinium myrtillus – Rhytidiadelphus squarrosus – Anthoxanthum odoratum* montane heath (MH3).

The Annex I habitat '6150 Siliceous alpine and boreal grasslands' exists within the reserve at high elevations around the peak of Más an Tiompán above the 550 m contour and are components of the montane heath *Carex bigelowii – Racomitrium languinosum/Dicranum fuscescens* communities.

Montane Heath

There are almost 46 hectares of montane heath in the reserve (Fossitt HH4, Annex I Alpine and boreal heath 4060) and it is characterised by short, wind-clipped vegetation and "*the presences of plants indicative of high altitude*" (Perrin et al., 2011a). The most frequently occurring and making up almost half (47%) of the montane heath community is MH1a *Calluna vulgaris – Racomitrium lanuginosum,* with 10% of it as the MH1b *Juncus squarrosus* sub-community. *Vaccinium myrtillus – Rhytidiadelphus loreus – Herbertus aduncus* community MH2 covers 15% of the montane heath and over one third is MH3 *Vaccinium myrtillus – Rhytidiadelphus loreus – Anthoxanthum odoratum*.

Siliceous scree SC1 makes up 33% of the montane heath area, characterised by block scree with species such as *Festuca vivipara,* ferns and *Saxifraga spathularis,* as well as *Racomitrium lanuginosum*. There is about 0.5 ha of the sub-community MH6a *Carex bigelowii – Racomitrium lanuginosum* also occurring.

3.4.3 Analysis of community data by habitat and treatment

The best solution presented by NMS was 2-dimensional, with a stress value of 11.192. Stress values of around 10 are considered interpretable and reliable ordination axes (McCune and Grace, 2002; Peck, 2010). The two axes represent a total of 93.1% of the variance in the dataset. Axis 1 explains 68.9% of the variance and axis 2 explains 24.1% (Table 26 and Figure 30).

Table 26 Coefficients of determination for the correlations between ordination distances and distances in the original n-dimensional space

Axis	Increment (r^2)	Cumulative
1	0.689	0.689
2	0.241	0.931

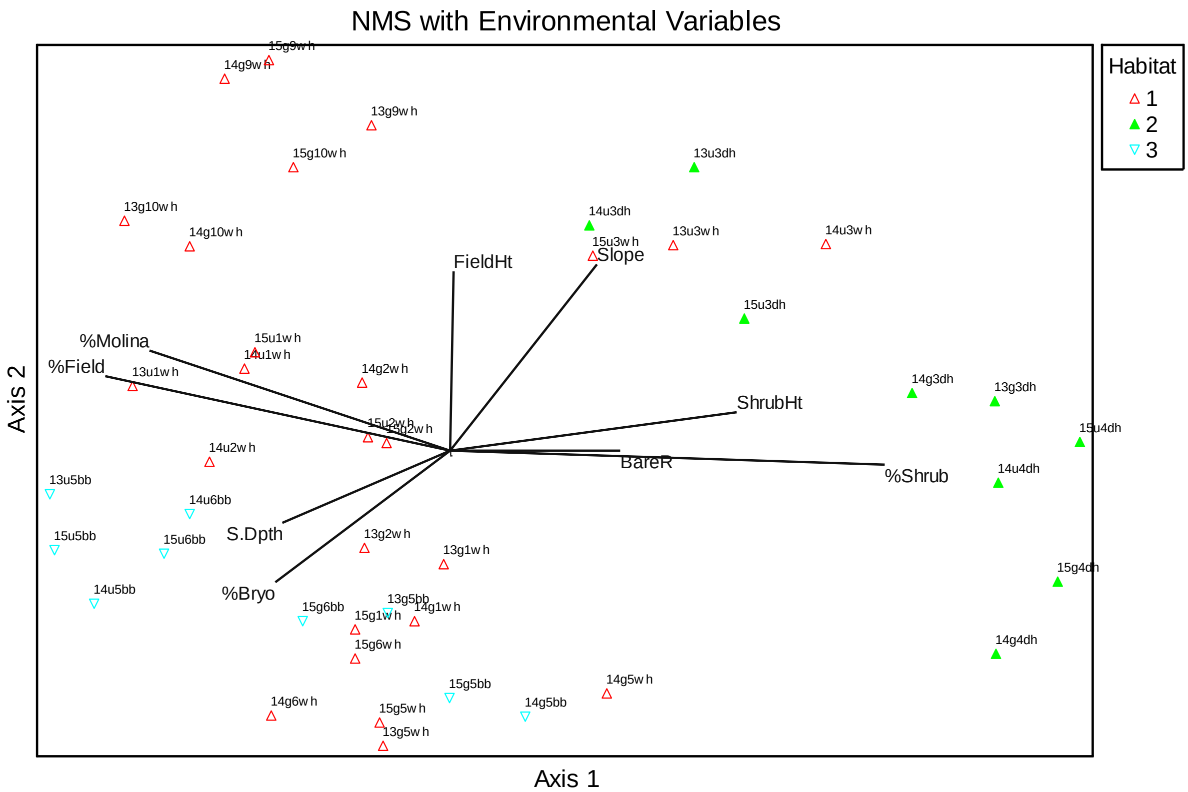


Figure 30 NMS of plots grouped by habitat Lines show the direction of significant environmental variables, with longer lines representing higher levels of significance.

The samples from the different habitats are generally well-differentiated in the NMS plot (Figure 30). Grazing does not appear on the plot as it was not found to be a significant variable affecting the vegetation composition when all three habitats are considered together. There is a strong differentiation between wet heath samples that have high field layer cover and percentage cover of *Molinia caerulea* on the left of the plot, and dry heath plots on the right with high dwarf shrub cover, steeper slopes, bare rock and greater heights of the dwarf shrub layer. Soil depth and percentage cover of the bryophyte layer define the blanket bog plots on the bottom left of the ordination (e.g. "14u5bb" = ungrazed blanket bog plot 2014 sample).

Shrub Layer cover and Shrub Layer Height had the strongest (negative) correlations with axis 1, indicating a gradient from dry heath plots with high heather cover, to blanket bog and wet heath plots. Soil depth, Percentage Field Layer and *Molinia caerulea* cover had the strongest (positive) correlations with axis 2. Wet heath samples are widely scattered on the plot, reflecting the variability of this habitat type, both generally, and observed at this site. Table 27 shows the Pearson correlation coefficients between NMS axes and environmental variables; and Table 28, 29 and 30 show average species abundances across treatments and habitats.

Table 27 Pearson correlation coefficient between NMS axes and environmental variables. Correlations $r^2 > 0.2$ are bold). It is indicated whether the correlation is positive or negative.

Axis		1		2
		r^2		r^2
Grazing intensity	(-)	0.005	(+)	0.019
Slope	(-)	**0.284**	(+)	**0.363**
Soil Depth	(+)	**0.326**	(-)	0.14
Bare Soil	(+)	0.063	(-)	0.006
Bare Rock	(-)	**0.33**	(-)	0
% Leaf litter cover	(+)	0.167	(-)	0.011
% Bryophyte cover	(+)	**0.34**	(-)	**0.254**
% Field layer cover	(+)	**0.669**	(+)	0.145
% Shrub layer cover	(-)	**0.845**	(-)	0.028
% *Molinia caerulea* cover	(+)	**0.584**	(+)	0.195
Field Layer Height	(-)	0.006	(+)	**0.349**
Shrub Layer Height	(-)	**0.558**	(+)	0.076
Ground Layer Height	(+)	0.094	(-)	0.113
S	(+)	0.171	(-)	0
H	(+)	0.001	(-)	0.003

Table 28 Average abundance of selected species in grazed and ungrazed wet heath.

	Ungrazed						**Grazed**					
	2013			2015			2013			2015		
	N	**Mean**	SE	N	**Mean**	SE	N	**Mean**	SE	N	**Mean**	SE
C .vulgaris	23	**19.5**	3.8	34	**25.3**	2.9	62	**18.3**	1.8	68	**19.8**	1.8
E. cinerea	23	**0.2**	0.1	34	**0.7**	0.5	62	**0.2**	0.1	68	**0.2**	0.1
E. tetralix	23	**3.6**	0.7	34	**3.9**	0.6	62	**2.1**	0.2	68	**1.7**	0.3
D. rotundifolia	23	**0.1**	0.0	34	**0.0**	0.0	62	**0.0**	0.0	68	**0.0**	0.0
N. ossifragum	23	**2.4**	0.7	34	**3.7**	0.9	62	**2.1**	0.6	68	**1.2**	0.2
P. erecta	23	**2.7**	0.6	34	**2.6**	0.5	62	**3.0**	0.4	68	**2.5**	0.3
J. bulbosus	23	**0.0**	0.0	34	**0.0**	0.0	62	**0.2**	0.1	68	**0.1**	0.1
J. effusus	23	**0.0**	0.0	34	**0.0**	0.0	62	**0.4**	0.2	68	**0.9**	0.5
J. squarrosus	23	**0.1**	0.1	34	**1.3**	0.5	62	**2.9**	0.7	68	**2.4**	0.5
C. panicea	23	**2.1**	0.4	34	**0.9**	0.2	62	**2.7**	0.4	68	**1.8**	0.3
E. angustifolium	23	**0.4**	0.2	34	**1.4**	0.3	62	**2.9**	0.7	68	**7.0**	1.1
T. germanicum	23	**3.7**	1.5	34	**6.8**	1.4	62	**4.9**	1.1	68	**6.6**	1.2
A. stolonifera	23	**0.0**	0.0	34	**0.1**	0.1	62	**0.4**	0.1	68	**0.4**	0.2
F. vivipara	23	**3.2**	1.3	34	**2.6**	0.7	62	**1.6**	0.4	68	**3.8**	0.5
N. stricta	23	**4.5**	1.3	34	**1.2**	0.4	62	**7.3**	1.0	68	**3.2**	0.6
M. caerulea	23	**40.2**	3.7	34	**42.0**	2.5	62	**34.4**	3.2	68	**31.8**	2.6
B. chrysocoma	23	**1.7**	0.5	34	**1.6**	0.4	62	**0.1**	0.1	68	**0.4**	0.1
H. jutlandicum	23	**0.0**	0.0	34	**1.6**	0.4	62	**0.5**	0.1	68	**1.7**	0.5
R. loreus	23	**0.5**	0.3	34	**2.8**	0.8	62	**0.5**	0.2	68	**1.6**	0.5
Sphagnum sp.	23	**27.9**	6.6	34	**20.0**	3.2	62	**26.0**	3.2	68	**25.2**	2.3
C. portentosa	23	**0.2**	0.1	34	**0.1**	0.1	62	**0.2**	0.1	68	**0.2**	0.1

Table 29 Average abundance of selected species in grazed and ungrazed dry heath.

	Grazed						**Ungrazed**					
	2013			**2015**			**2013**			**2015**		
	N	**Mean**	SE	N	**Mean**	SE	N	**Mean**	SE	N	**Mean**	SE
C. vulgaris	5	**25.4**	7.2	21	**48.9**	6.3	15	**62.0**	6.1	15	**62.3**	8.4
E. cinerea	5	**4.8**	1.7	21	**5.0**	1.0	15	**2.2**	0.5	15	**1.6**	0.7
E. tetralix	5	**4.2**	1.6	21	**0.7**	0.4	15	**0.1**	0.1	15	**0.0**	0.0
P. erecta	5	**6.3**	1.6	21	**2.1**	0.6	15	**2.6**	0.5	15	**6.8**	1.0
J. squarrosus	5	**0.0**	0.0	21	**0.1**	0.1	15	**0.0**	0.0	15	**2.7**	1.4
A. capillaris	5	**1.2**	1.0	21	**2.1**	1.0	15	**5.0**	1.6	15	**6.1**	1.3
M. caerulea	5	**21.4**	10.9	21	**6.9**	3.4	15	**0.9**	0.5	15	**0.4**	0.2
N. stricta	5	**11.2**	6.1	21	**0.5**	0.3	15	**2.0**	0.6	15	**0.1**	0.1
Sphagnum sp.	5	**5.3**	3.8	21	**1.6**	1.0	15	**4.2**	2.3	15	**0.3**	0.2
T. tamariscinum	5	**8.0**	2.3	21	**6.1**	1.6	15	**19.0**	2.2	15	**1.2**	0.3

Table 30 Average abundance of selected species in grazed and ungrazed blanket bog

	Ungrazed						**Grazed**					
	2013			**2015**			**2013**			**2015**		
	N	**Mean**	SE	N	**Mean**	SE	N	**Mean**	SE	N	**Mean**	SE
C. vulgaris	15	**11.8**	2.1	30	**15.0**	1.6	5	**37.0**	8	14.0	**38.8**	5.1
N. ossifragum	15	**4.0**	0.8	30	**4.4**	0.6	5	**1.7**	1	14.0	**2.6**	0.6
D. rotundifolia	15	**0.1**	0.0	30	**0.1**	0.0	5	**0.0**	0	14.0	**0.0**	0.0
J. squarrosus	15	**0.0**	0.0	30	**0.1**	0.1	5	**2.4**	2	14.0	**1.9**	1.1
C. panicea	15	**0.0**	0.0	30	**0.9**	0.2	5	**0.0**	0	14.0	**0.3**	0.2
F. vivipara	15	**0.0**	0.0	30	**1.4**	0.3	5	**0.0**	0	14.0	**6.1**	1.9
T. germanicum	15	**7.9**	1.8	30	**12.7**	2.1	5	**12.0**	4	14.0	**4.5**	1.5
Sphagnum sp.	15	**57.9**	8.0	30	**63.0**	5.1	5	**52.7**	14	14.0	**45.2**	4.8

3.4.4 Species diversity responses

Ungrazed wet heath showed an increase in biodiversity over time, with the curve shifting up and to the right i.e. more species (from 43 to 51) and higher evenness indicated by the change in slope of the line (Figure 31). In grazed wet heath species richness showed little change (45 species to 43) and almost no change in evenness.

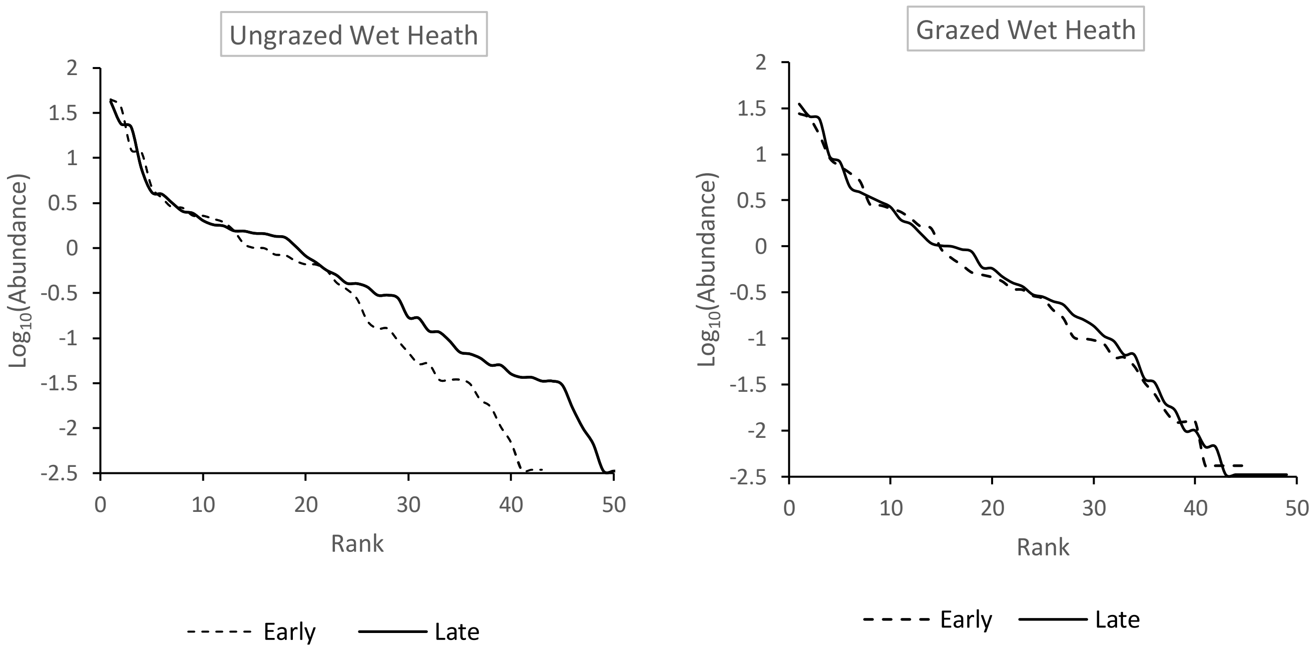


Figure 31 Rank abundance plots for ungrazed and grazed wet heath

Dry heath (Figure 32) showed little change in biodiversity over time, regardless of whether it was grazed or not. A small number of species had high abundance (*C. vulgaris, U. gallii* and *A. capillaris*) and the remainder of the community was relatively even as illustrated by the slopes of the lines. Species richness was very similar; ungrazed dry heath had 40 species in the early samples and 34 in the late, whilst grazed dry heath had 37 in the early samples and 42 in the late. The slope of the line for grazed dry heath indicates a marginally more even community.

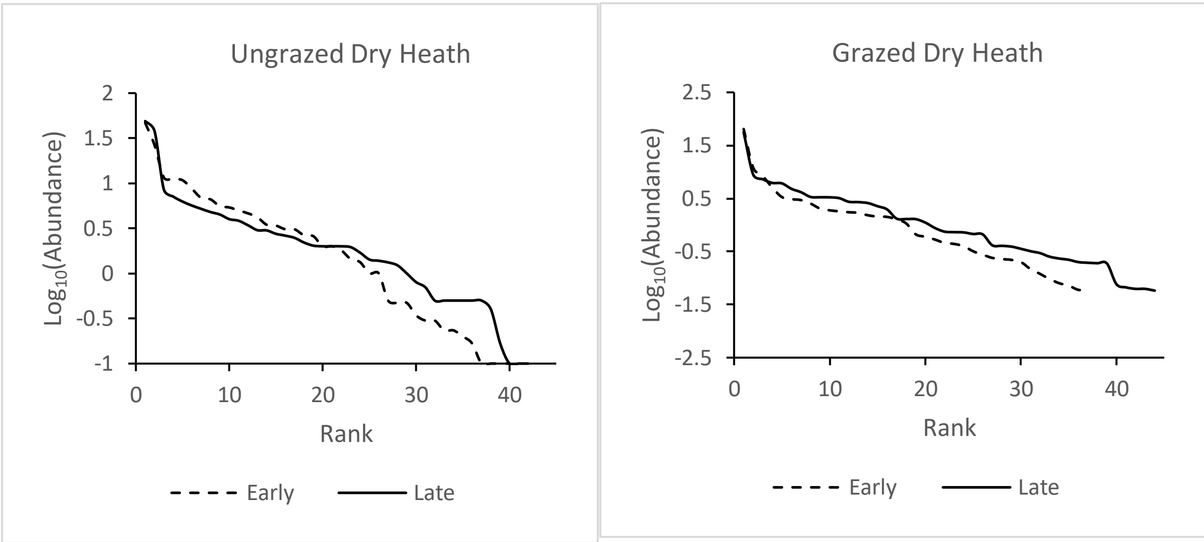


Figure 32 Rank abundance curves for ungrazed and grazed dry heath

Ungrazed blanket bog (Figure 33) was a less even community than either of the heath communities and contained less species. Twenty-nine species were recorded in the early samples and 28 in the late. As expected, a small number of species had high abundances i.e. *C. vulgaris, T. germanicum, M. caerulea, R. lanuginosum* and *Sphagnum* spp. Grazed blanket bog had a slightly more even community, as indicated by the slopes of the lines. It also contained more species, with 27 recorded in 2013 and 32 in 2015.

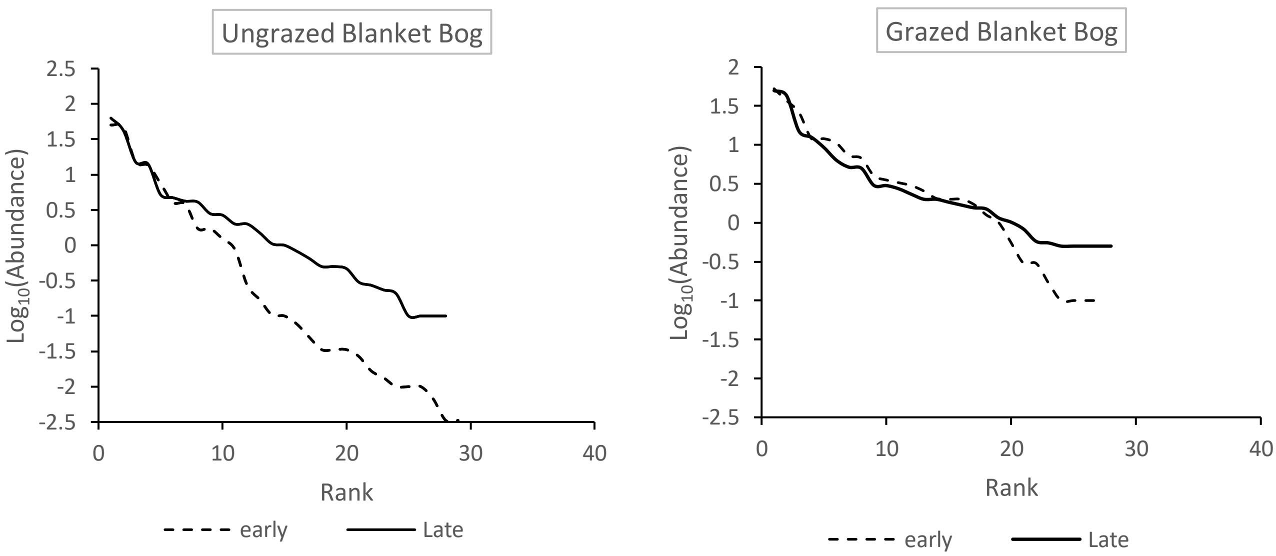


Figure 33 Rank abundance curves for ungrazed and grazed blanket bog

3.4.5 Relationships between utilisation scores and conservation criteria

Table 31 presents results from general linear regression modelling (GLM). The table shows the model type, the link function, the outcome variable being modelled e.g. 'species richness', 'cover of positive indicators', and each predictor in the model.

Utilisation (χ^2 = 0.002, z = 22.6, df = 89, p = 0.0247) and slope (χ^2 = 0.007263, z = -2.098, df = 89, p = 0.0359) were significant predictors for species richness. No factors were identified as being significant predictors for the 'cover of positive indicators' and utilisation was identified as being a significant negative predictor of the 'number of positive indicators' (χ^2 = -0.006099, z = -4.571, df = 89, p = 0.001). None of the variables examined had a significant influence on the cover of *Molinia caerulea*.

For bryophyte cover, utilisation was a significant predictor in the model (χ^2 = -0.007985, z = -4.382, df = 89, p < 0.001). Soil depth was a positive predictor for field layer cover (χ^2 = 0.0681148, z = 2.774, df = 89, p = 0.00674). For shrub cover, utilisation (χ^2 = -4.029578, z = -17.394, df = 89, p < 0.001) and litter (χ^2 = -0.006825, z = -4.25, df = 89, p < 0.001) were significant predictors in the model.

Multiple regression (presented in Table 32) identified relationships between variables within habitat types (Table 32). In **wet heath**, the cover of positive indicator species was significantly negatively related to utilisation (grazing) but was not a significant predictor. Altitude, %ground, % field, % shrub, *Molinia*, and field height were positively related. A significant negative correlation was found between the cover of bryophytes and utilisation but was not identified as a predictor in the models. Altitude, bare soil and height of the field layer were significant predictors of the cover of bryophytes and lichens, accounting for 34% of the variation. There was a significant negative relationship between the cover of ericoid species and utilisation (plus *Molinia* and % field layer). Cover of dwarf shrub species: Utilisation, bare soil, %field layer, %*Molinia,* were significantly negatively related to the cover of dwarf shrubs. Utilisation was a significant predictor in the model (β = -0.63 p < .001).

In **Dry Heath,** utilisation was negatively related to the number of positive indicator species and it was a significant predictor in the model (β = -0.46 p = 0.01). As expected, the cover of litter, field layer, ground layer and *Molinia* were negatively related to the cover of positive indicators (*C. vulgaris, E. cinerea*, *U. gallii* and *V. myrtillus*).

In **Blanket bog,** the number of positive indicator species was negatively related to slope and positively correlated to % litter, % field layer, % ground layer and % *Molinia* cover. Slope was the only significant predictor of the number of positive indicator species in blanket bog (β = -0.42, p = 0.01). The cover of bryophyte or lichen species was negatively related to utilisation, slope and bare soil, and positively related to soil depth and cover of field layer. Slope (-) and bare soil (-) were significant predictors in the model. The cover of *Calluna* was positively related to utilisation. Soil depth (+) and utilisation (+) were significant predictors. No significant relationships were discovered for the cover of *Eriophorum.*

Molinia cover was positively correlated to litter cover, soil depth, and ground layer cover. Utilisation and bare soil were negatively related. Litter was the only significant predictor (β = 0.34, p = 0.03).

Table 31 GLM of selected criteria for grazed samples in MBNR (habitats=all, year=2015). Exponential family and link fn indicated.

Species Richness					**Bryophyte Cover**				
Family: Poisson, link=logit					Family: Quasi-binomial, link=logit				
	Estimate	**SE**	***z*-value**	***p*-value**		**Estimate**	**SE**	***z*-value**	**p-value**
(Intercept)	2.868863	0.127015	22.587	0***	(Intercept)	-4.162154	0.249272	-16.697	< 0.001 ***
Utilisation	0.002077	0.000925	2.246	**0.0247** **	Utilisation	-0.007985	0.001822	-4.382	< **0.001** ***
Slope	-0.007263	0.003462	-2.098	**0.0359** **	Slope	-0.011577	0.006791	-1.705	0.0917
Soil Depth	-0.019929	0.015811	-1.26	0.2075	Soil Depth	0.032918	0.031162	1.056	0.2937
Bare Soil	0.010087	0.007496	1.346	0.1784	Bare Soil	-0.015587	0.017136	-0.91	0.3655
Cover of Positive Indicators					**Field Layer Cover**				
Family: Binomial, link=logit					Family: Quasi-binomial				
	Estimate	**SE**	***z*-value**	***p*-value**		**Estimate**	**SE**	***z*-value**	***p*-value**
(Intercept)	-4.127724	4.88335	-0.845	0.398	(Intercept)	-5.0746031	0.1990805	-25.49	< **0.001** ***
Utilisation	-0.004529	0.035345	-0.128	0.898	Utilisation	0.0002551	0.0014434	0.177	0.86011
Slope	-0.003131	0.129054	-0.024	0.981	Slope	0.0068146	0.004942	1.379	0.17137
Soil Depth	-0.018521	0.618023	-0.03	0.976	Soil Depth	0.0681148	0.0245536	2.774	**0.00674** **
Bare Soil	-0.017615	0.33734	-0.052	0.958	Bare Soil	0.0010821	0.0126218	0.086	0.93187
Number of Positive Indicators					**Shrub Layer Cover**				
Family: Poisson, link=log					Family: Quasi-binomial				
	Estimate	**SE**	***z*-value**	***p*-value**	(Intercept)	**Estimate**	**SE**	***z*-value**	***p*-value**
(Intercept)	2.001459	0.182027	10.995	< **0.001** ***	Utilisation	-4.029578	0.231663	-17.394	< **0.001** ***
Utilisation	-0.006099	0.001335	-4.571	< **0.001** ***	Slope	-0.00138	0.001661	-0.831	0.4083
Slope	0.005124	0.00456	1.124	0.2612	Soil Depth	-0.001383	0.006109	-0.226	0.8214
Soil Depth	0.026469	0.023039	1.149	0.2506	Bare Soil	-0.060727	0.029389	-2.066	**0.0417** *
Bare Soil	0.02123	0.010384	2.044	0.0409*	(Intercept)	-0.036021	0.017174	-2.097	**0.0388** *
***Molinia caerulea* cover**					**Litter Cover**				
Family: Binomial, link=logit					Family: Quasi-Binomial, link=logit				
(Intercept)	**Estimate**	**SE**	***z*-value**	***p*-value**		**Estimate**	**SE**	***z*-value**	***p*-value**
Utilisation	-5.2553017	4.8845893	-1.076	0.282	(Intercept)	-4.663929	0.219396	-21.258	< 0.001
Slope	-0.0001248	0.0354448	-0.004	0.997	Utilisation	-0.006825	0.001606	-4.25	< **0.001** ***
Soil Depth	0.0113362	0.1178496	0.096	0.923	Slope	0.002472	0.005536	0.447	0.656
Bare Soil	0.0854306	0.6014167	0.142	0.887	Soil Depth	0.06803	0.027439	2.479	**0.015** *
(Intercept)	0.0096092	0.298645	0.032	0.974	Bare Soil	-0.006321	0.014592	-0.433	0.666

* indicates significance at $P \leq 0.05$, ** significant at ≤ 0.01, and ***=significant at ≤ 0.001.

Table 32 Multiple Regression linear model predictors of conservation status monitoring criteria for three habitats.

WET HEATH						
PosIndCover	**b**	**SE b**	**β**	**Sig.**	**LB**	**UB**
(Constant)	-35.88	12.05		0.00	-59.93	-11.83
Altitude	0.12	0.02	0.34	0.00	0.07	0.16
% Ground Layer	1.01	0.10	0.54	0.00	0.80	1.22
% Field Layer	0.66	0.14	0.33	0.00	0.37	0.94
% Shrub Cover	0.89	0.13	0.41	0.00	0.62	1.15
Molinia cover	-0.37	0.13	-0.19	0.01	-0.62	-0.11
Field Height	-0.29	0.13	-0.12	0.03	-0.56	-0.03
PosBryoCover						
(Constant)	29.03	9.88		0.00	9.32	48.74
Altitude	0.06	0.02	0.32	0.01	0.02	0.11
Bare Soil	-1.22	0.57	-0.22	0.04	-2.36	-0.08
Field Height	-0.42	0.16	-0.29	0.01	-0.73	-0.10
Shrub Cover						
Utilisation Score	-0.32	0.10	-0.63	0.00	-0.51	-0.13
Altitude	-0.06	0.03	-0.38	0.05	-0.12	0.00
Litter	0.53	0.22	0.24	0.02	0.09	0.97
% Field Layer	-0.31	0.10	-0.33	0.00	-0.51	-0.12
DRY HEATH						
No.PosInd						
(Constant)	7.24	0.94		0.00	5.33	9.15
Slope	-0.06	0.02	-0.42	0.01	-0.11	-0.02
% Field Layer	0.02	0.01	0.25	0.10	0.00	0.04
PosIndCover						
(Constant)	1.79	14.72		0.90	-28.42	31.99
Litter	0.92	1.01	0.09	0.37	-1.16	3.01
% Ground Layer	-0.12	0.15	-0.07	0.43	-0.43	0.19
% Field Layer	0.12	0.13	0.13	0.37	-0.15	0.39
% Shrub Cover	0.98	0.14	1.01	0.00	0.68	1.27
Molinia cover	-0.48	0.45	-0.09	0.29	-1.41	0.44

BLANKET BOG						
No.PosInd	**b**	**SE b**	**β**	**Sig.**	**LB**	**UB**
(Constant)	7.24	0.94		0.00	5.33	9.15
Slope	-0.06	0.02	-0.42	0.01	-0.11	-0.02
% Field Layer	0.02	0.01	0.25	0.10	0.00	0.04
BryoCover						
(Constant)	50.01	21.74		0.03	5.96	94.07
Utilisation Score	0.07	0.21	0.07	0.75	-0.36	0.50
Slope	-0.90	0.38	-0.34	0.02	-1.67	-0.12
Soil Depth	0.06	0.22	0.06	0.79	-0.38	0.50
Bare Soil	-3.37	1.11	-0.38	0.00	-5.62	-1.12
% Field Layer	0.31	0.24	0.23	0.20	-0.17	0.79
***Calluna* cover**						
(Constant)	5.48	13.09		0.68	-20.98	31.95
Utilisation Score	0.61	0.15	0.82	0.00	0.31	0.90
Soil Depth	0.11	0.14	0.16	0.43	-0.17	0.39
***Eriophorum* cov.**						
(Constant)	4.51	4.82		0.35	-5.25	14.27
Slope	-0.21	0.13	-0.33	0.11	-0.47	0.05
Bare Soil	-0.20	0.36	-0.09	0.57	-0.93	0.52
Litter	-0.01	0.10	-0.02	0.90	-0.21	0.18
% Ground Layer	0.02	0.04	0.09	0.64	-0.07	0.11
Ground Height	0.29	0.27	0.17	0.30	-0.26	0.84
***Molinia* Cover**						
(Constant)	5.21	16.81		0.76	-28.89	39.31
Utilisation Score	-0.14	0.17	-0.16	0.42	-0.47	0.20
Slope	-0.15	0.35	-0.07	0.67	-0.85	0.55
Soil Depth	0.23	0.17	0.29	0.19	-0.12	0.58
Bare Soil	-0.61	0.95	-0.08	0.52	-2.53	1.31
Litter	0.70	0.31	0.34	0.03	0.07	1.33
% Ground Layer	0.04	0.13	0.05	0.77	-0.22	0.29

3.4.6 Conservation status assessments

A summary of the conservation status assessments of Annex I habitats in Mt Brandon Nature Reserve is provided here in 'traffic light format', as per NSUH monitoring (monitoring criteria for each habitat are in Table 22, section 1.3.4.5). Appendix II provides the plot-level data for the assessment trends over the time for all criteria. An example, for the wet heath exclosure, is given in Table 33 below.

Table 33 Conservation status assessment values for an ungrazed wet heath plot

Plot 1: u1 (Wet Heath Exclosure)						
	2013		2014		2015	
	Mean	*s*	Mean	*s*	Mean	*s*
Erica tetralix present (1 = present, 0 = not)	0.9	0.2	0.9	0.2	1.0	0.0
Cover of positive indicator species ≥ 50%	78.6	38.2	92.2	21.1	73.7	35.2
Tot. cover of: *Cladonia* sp. *Sphagnum* sp. *R. lanuginosum* and pleurocarp mosses ≥ 10%	47.7	29.6	29.4	10.2	30.2	22.1
Cover of ericoid species ≥ 15%	13.4	7.6	24.4	11.2	21.9	16.5
Cover of dwarf shrub species < 75%	13.4	7.6	24.4	11.2	22.3	16.9
Cover of neg. indicators: *A. capillaris*, *H. lanatus*, *R. repens*, collectively < 1%	0.1	0.5	0.0	0.0	0.1	0.5
Cover of non-native species < 1%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of non-native species < 1%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of *Pteridium aquilinium* < 10%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of *J. effusus* < 10%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of disturbed, bare ground < 10%	1.5	3.1	0.0	0.0	0.7	1.4

For criterion 1 '*Erica tetralix* present', a value of 0.9 means that *E. tetralix* was present in 13/15 quadrats. Criteria 2 shows the average cover of positive indicator species for the plot. Table 22 (page 171) lists the positive indicator species for each habitat.

Traffic light system for conservation status assessments of Annex I habitats.

	Favourable (F)	**Unfavourable Inadequate (U-I)**	**Unfavourable Bad (U-B)**
Criteria	No stop failures	1 – 25% of stops failed	> 25% of stops failed

3.4.6.1.1 Wet Heath

Results from the first year of sampling (2013) show that 2/7 plots were in Unfavourable-Bad (UB) conservation status, 4/7 were Unfavourable-Inadequate (US), and 1/7 Favourable (F). Table 34 shows the results for all wet heath plots from 2013 to 2015. It shows that the trend in seven of these plots is towards Favourable status. Two plots remained in unfavourable status. Table 34 provides details for each plot sampled, showing conservation status, level of utilisation, the number of stop fails, and criteria that resulted in stop failures.

Table 34 Conservation status assessments for Northern Atlantic Wet Heath with *Erica tetralix*

Plot	Plot code	Utilisation	2013	2014	2015	Comment
1	u1/whex. Exclosure.	0 (none)	U-I: 13% (2/15 fails)	F	U-I: 13% (2/15 fails)	• Cover of positive indicator sp., Bryophyte Cover, Ericoid cover
2	u2/whvex. Ungrazed	0 (none)	Not sampled	U-I: 13% (2/15 fails)	F	• Cover of negative indicators (*A. capillaris* = 5%)
3	g1/wh15. Grazed plot	5% (low)	U-I: 20% (2/10 fails)	F	F	• Bryophyte cover • Cover *A. capillaris* (negative indicator sp.) • Ericoid cover
4	g2/wh60 Grazed	44% (med)	U-I: 20% (2/14 fails)	F	F	• Cover of bare ground = 20% in two 2013 plots
5	u3/dhvex. Reclassified from DHex. Ungrazed	0 (none)	F	F	F	
11	g5/bb51. Reclassified as WH quadrats from a BB plot. Grazed.	55% (medium)	U-I: 11% (1/9 fails)	F	F	• One stop fail for bare soil and ericoid cover
12	g6/bb02. Reclassified WH Grazed	16% (low)	Not sampled	U-I: 13% (2/15 fails)	F	• Bare ground • Bryophyte cover
16	g9/wg69 Transition habitat Valley floor. Grazed.	90% (high)	U-B: 36% (5/14 fails)	U-B: 50% (7/14 stop fails)	U-B: 43% (6/14 stop fails)	• Inadequate ericoid cover • Negative indicator species (*A. capillaris*) • Bryophyte cover
17	g10/wg63 Transition habitat from WG to WH to DH. Valley floor. Grazed.	90% (high)	U-I: 53% (8/15 stop fails)	U-B: 35% (3/8 stop fails).	U-I: 13% (1/8 stop fails)	• Cover of negative indicators (*A. capillaris*) • Cover of positive indicators inadequate • Cover of ericoid inadequate • Inadequate bryophyte cover

3.4.6.1.2 Dry Heath

The results indicate that dry heaths of the reserve are in good conservation status. The status remained constant from 2013-2015. Table 35 shows the conservations status, level of Utilisation, and reasons for any stop failures.

Table 35 Conservation status assessments of European Dry Heath

Plot	Plot code	Utilisation	2013	2014	2015	Comment
5	u3/dhex. Exclosure. Ungrazed	0% None	F (n = 4 stops)	F (n = 9 stops)	F (n = 6 stops)	
7	g3/dh47 Grazed	53% (medium)	F (n = 15 stops)	F (n = 15 stops)	Not sampled	
6	u4/dhvex Ungrazed	0%	Not sampled	F (n = 15 stops)	F (n = 15 stops)	
8	g4/dh04 Grazed	82% (high)	Not sampled	F	F	Note 2 stop failures occurred in this plot, but they keyed as acid grassland and were thus ignored for assessment.

3.4.6.1.3 Blanket bog

The ungrazed blanket bog exclosure (plot no9) had no stop failures in any year and was in good conservation status throughout this study. The ungrazed (but not fenced) plot no.10 did not have any stop failures in any year and was also in good status at the time of assessment. Plot no.11 had moderate levels of Utilisation (55%) and plot no12 had low levels (17%). Both these plots were also in good conservation status at the time of assessment. Table 36 provides assessment details for blanket bog plots.

Table 36 Conservation status assessments for Blanket Bog

Plot	Plot code	Utilisation	2013	2014	2015	Comment
9	u5/bbex Exclosure. Ungrazed	0% (none)	F	F	F	
11	g5/bb51 Grazed.	55% (medium)	F	F	F	
10	u6/bbvex Not fenced, but Ungrazed.	0% (none)	Not sampled	F	F	
12	g6/bb02 Grazed.	17% (low)	Not sampled	F	F	

3.5 Discussion

3.5.1 Introduction and Management Perspective

'Conservation grazing' is a term used to describe the use of grazing animals to maintain and enhance the biodiversity of semi-natural habitats (Small, 2003). Grazing with domestic livestock has impacts on vegetation from a botanical diversity and a structural perspective. Outcomes from studies examining the impact of grazing animals have the potential to inform management decisions and research direction for biodiversity conservation.

In the present study, grazing with cattle maintained or enhanced the conservation status of Annex I habitats in Mt Brandon Nature Reserve. Species richness and biodiversity measures varied little between years, within habitat type, or between grazing treatments. Current stocking densities in this study had a positive impact on Annex I habitats from a structure and function perspective and trends were favourable.

However, extensive grazing studies of this kind may take many years to demonstrate longer term trends and impacts, so a cautious approach is recommended. Improvement habitats exclusively from a vegetation perspective may not be enough to meet all site conservation requirements. Within sites, specific vegetation management may be required for the benefit of animal taxa e.g. bird or invertebrate groups. It has been illustrated that invertebrate groups require both botanical diversity and vegetation structural diversity to meet multiple needs. Invertebrate needs vary between multiple taxonomic levels, and at varying stages of lifecycle at taxon level.

3.5.2 General Findings

The present study found that species richness and biodiversity values varied little over the course of the study within habitat type or between grazing treatments. Specific plots and outcome variables for each habitat will be discussed further in the following section (3.5.3). More generally, where the site had low to moderate levels of Utilisation, outcomes regarding conservation measurables were found to be stable or improving e.g. habitat conservation status, species richness, cover of positive indicator species, and ericoid cover. Where grazing levels were high, outcomes were poorer.

Species richness is increasing in ungrazed plots. This is possibly due to the relaxation of grazing pressure once the resident goats and occasional trespassing sheep were completely excluded by fencing. This is likely to continue in the short-term until heather cover increases to the point of shading out species (Bokdam and Gleichman, 2000). However, continued observation is recommended to reveal long term trends, given that the recovery after complete exclusion of grazing is slow, relative to the imposition of grazing by large herbivores (Milchunas, 2011).

In blanket bog, the grazed areas were more species rich than the ungrazed areas. However, the character of the grazed plot was quite different to the ungrazed plot. The grazed plot (PB 51) received moderate levels of grazing (US = 55). It is a narrow strip (100 m wide) of blanket bog that follows a ridge at a slope of 5 – 10 ° (Figure 17). This strip of blanket bog grades into wet heath on either side. Where the blanket bog has slipped to form steps in the break of slope it is degraded. Here the area tends to have shallower soils (56 cm versus 200+cm), peat hags and patches of bare ground (mean bare ground cover = 4% and frequency of bare patches = 40%). In these areas and at the edges of the ridge of blanket bog, heath species become more frequent and abundant e.g. *C. vulgaris, J. squarrosus, T. germanicum and B. chrysocoma*. The grazed site also has substantially less litter cover (3% versus 19%), more shrub cover (33% versus 18%), less field (42% versus 72%) cover and less *Molinia* cover (15% v 53%).

These factors, along with the moderate levels of disturbance caused by grazing probably explain greater species richness, biodiversity values, and a more even community as indicated by the rank abundance plots.

In contrast, the ungrazed blanket bog plot is situated in a large, flat expanse of stable blanket bog on deep peat. In this study, cattle avoided it. Stray sheep that occasionally ventured into the study site were rarely observed using this area of bog. The plant community is dominated by *Sphagnum* spp., *M. caerulea, Eriophorum* spp., *T. germanicum*. The rank abundance plots showed that it was an uneven community and this changed little during the study. This plot is characterised by a stable, climax blanket bog plant community.

Scimone et al., (2007) found that the level of grazing had an impact on vegetation structure but not plant diversity over three years. In the present study, the patterns for vegetation structure (measured by per cent cover of shrub, field and ground layers, per cent cover of leaf litter, and field and shrub layer heights) were consistent for all habitats. The cover of these layers and the heights of shrub and field layers were greater in ungrazed treatments compared to grazed. In this study cattle utilised patches of vegetation with varying intensity, leading to structural diversity at the site level. Sward structure is a useful indication of sward condition (Mills et al., 2007) and where habitat and site-level conservation objectives are not just vegetation focused, vegetation structure should be considered. Invertebrate taxa such as butterflies and moths (Pöyry et al., 2006), grasshoppers (Spalinger et al., 2012) and ground beetles require structural heterogeneity to meet multiple needs.

Levels of bare soil across all habitats and treatments was very low, varying little between grazed and ungrazed sites. In the grazed blanket bog plot, bare soil cover was higher. This plot was discussed above. It had moderate levels of grazing, slopes were relatively high (average slope value for the plot was 10.5°) and it was degraded in nature. The ungrazed bog plots were characterised by high field layer and ground layer cover compared to grazed plots.

In the present study, the cover of *Molinia* was lower in grazed sites compared to ungrazed. In wet and dry heath, which had higher utilisation levels than the bog, *Molinia* decreased over time. *Molinia* is common on uplands of Britain and Ireland, and can dominate at the expense of other plants (Taylor, Rowland and Jones, 2001). The abundance of *Molinia* has increased since the industrial revolution at the expense of *Calluna vulgaris* in upland areas of the British Isles, and its encroachment has been viewed as a major threat to moorland conservation (Marrs et al., 2004a). The fast growth and height of *M. caerulea* in early summer may indicate that it is more suitable to cattle grazing than sheep (Taylor, Rowland and Jones, 2001), as sheep generally graze closer to the ground. Marrs et al., (2004b) found that three years of repeated defoliation by cattle reduced leaf production 40% compared to ungrazed controls. Floristic diversity also increased on grazed sites in that study.

3.5.3 Habitats of Mt Brandon Nature Reserve and Conservation Status Assessments

Examination of the habitat data from the NSUH revealed that the habitats within the reserve are typical of those described for the wider Mt Brandon cSAC. They compare well with those described in the provisional classification for upland habitats Ireland (NSUH; Perrin et al., 2014a). The habitats in the reserve exist in complex mosaics. Within the 50 x 50 m plots there are patches of other habitat types i.e. at the 2 x 2 m level. To capture such small-scale variation, all 2 x 2m quadrats were reclassified during data processing, based on their vegetation composition, slope and soil depths. The habitat key and descriptions in the provisional scheme of the NSUH were used to reclassify every quadrat (Perrin et al., 2014a).

Eighty five percent of Ireland's EU protected habitats are in unfavourable conservation status. Of these, 46% of them have declining trends (NPWS, 2019b). Nationally, the habitats considered in this study are also in unfavourable conservation status (Inadequate or Bad): Wet Heath is 'Bad' and 'deteriorating', European Dry Heath is 'Bad' but 'stable', and Blanket Bog is 'Bad' and 'Deteriorating (NPWS, 2019c).

The habitats of the reserve were generally in good conservation status at the beginning of the study. At current stocking densities utilisation by the cattle did not negatively impact on the conservation status of the habitats. The status of dry heaths and blanket bogs were favourable and did not change over the period of the study. However, blanket bog plot 51 received moderate levels of grazing (55%). Species richness was higher in this plot and increasing over time and the community is becoming more even (as shown by the rank abundance plots). As Blanket bog is a stable, climax plant community these trends are not desirable.

The results from wet heath are more nuanced. Where grazing levels were low to medium (in the context of this study), the trends showed improvement in status towards 'favourable', suggesting that stocking densities were appropriate. Plot no.3 (g1/wh15), a plot with utilisation levels of 5% is an example. This plot had two stop failures in 2013, and in 2014 and 2015 there were none. This pattern is repeated in plot 4 (US = 44%), plot 11 (US = 55%) and plot 12 (US = 16%), where trends towards favourable status occurred.

Where Utilisation levels were high in plot 16 (US = 90%) and plot 17 (US= 90%), there was a relatively high proportion of stop failures (43% and 34% respectively), and the overall status of these plots is bad. However, the data show that these plots were in bad conservation status at the beginning of the study. Despite the initial bad conservation status, plot 17 went from 8/15 failures in 2013 to 1/15 in 2015, showing a trend towards favourable status. It failed on the lack of ericoid cover and positive indicator cover and had a component of *A. capillaris* (a negative indicator). On closer examination, the results show that *A. capillaris* cover declined from 1.4% to 0.4% between 2013 and 2015. The cover of ericoid species increased from 6.3% to 9.6%. The total cover of dwarf shrubs increased from 6.3 to 11.2%. If trends continue under the current grazing regime this plot will achieve favourable status (raw plot-level data from conservation monitoring assessments are available in Appendix II.

Wet heath plot no16 did not display the improving trends of plot 17, even though they were adjoining. Stop failures remained constant in this plot (16% in 2013, 50% in 2014 and 43% in 2015). Failures were due to inadequate bryophyte and ericoid cover, and cover of negative indicator (*A. capillaris*). The trends in the data show a negative trend in these criteria for this plot. The heather cover in this plot is patchy (average cover 35%, ranging from 58% down to 7%. This plot is very close to the river, so has a wet grassland influence, as indicated by the 1-4% cover of *J. effusus*. The cattle also drink from the river at along the edge of this plot.

These two plots (16 and 17) are in the bottom of Arraglen, a flat bottomed but steep sided valley in the reserve. Shelter, isolation and a high cover of graminoids relative to other areas of the reserve, may explain the high levels of utilisation. The cattle move to this area of the reserve late in the grazing season (September to mid-October). It has been suggested that this may be related to avoidance of *Culicoides* biting midges, although this was not considered here. Even within these plots and utilisation bands, there were more heavily used areas than others (e.g. close to the river), that the resolution of the utilisation scores could not capture.

The Arraglen valley where these plots are situated has had a long history of use by people and animals, as evidenced by ruined buildings and lazy beds. People were living and farming in the valley up until around 1900 (M. Neville, *pers. comm.,* 2018). The results from chapter 2 (home range and habitat selection) showed that the cattle preferentially selected wet grassland plots in this part of the study area. This should be a consideration if more sensitive and 'higher value' habitats are adjoining or existing in a mosaic setting.

The wet heath exclosure had 2 stop failures in 2013 and 2 in 2015 due to insufficient cover of positive indicator species, low bryophyte cover and insufficient ericoid cover. However, with 13 stops passing, the overall status of Unfavourable-Inadequate is marginal. Plot no2 is the other ungrazed (though not fenced) wet heath plot and it had 2 stop failures in 2013 and none in 2015. In these plots *M. caerulea* and *J. effusus* were dominant and heather species did not meet the pass percentage. However, in the plot overall there is an average of 88% heather cover, so these stops are not representative.

3.5.4 Community response to utilisation

Habitats were well differentiated, with wet heath and blanket bog being characterised by field layer cover, and greater soil depths. However, grazing was not a significant factor in the differentiation of habitats in ordination space. This indicates that the underlying geographical factors such as slope and soil depth are stronger drivers of community composition than imposed grazing regimes. Furthermore, in the present study, stocking densities are low and three years is a short timescale to detect change.

3.5.5 Species diversity responses

The rank abundance plots show an apparent increase in biodiversity in ungrazed **wet heath**. Species richness went from 43 to 51. The additional species were *Dicranum majus, Pleurozia purpurea, Polytrichum commune, Plagiothecium undulatum, Cladonia uncialis, Agrostis stolonifera, Luzula multiflora,* and *Prunella vulgaris*. The bryophytes are potentially explained by variations in rainfall prior to and during sampling. *Prunella vulgaris* is a species that is easily transported by people and animals, particularly when wet (Clark and Wilson, 2003). It is recorded in the study site beside tracks and trails. It is likely that this spread into WH60 (a grazed WH plot close to the track) is via the cattle, the grazier or field workers. It may also be possible that the randomised quadrat locations can explain some of the differences in species recorded.

The rank abundance curves for **blanket bog** show that there was little change in the ungrazed sites. As previously described, this is a very stable, expansive area of blanket bog that receives little disturbance. It is dominated by a few species (*Sphagnum* spp, *Molinia, T. germanicum*, *C. vulgaris*, and *Racomitrium lanuginosum*).

The grazed site ('BB51') is a thin strip of blanket bog that runs along a ridge (described in section 3.5.2). It is becoming more species rich over time, and the community more even. Some of the quadrats in this plot keyed out at wet heath. The increased species richness in the bog quadrats is possibly an indication that the entire plot is becoming more like wet heath, potentially due to the grazing pressure, evidence that grazing pressure is not desirable for this habitat type.

3.5.6 Relationships between utilisation scores and conservation criteria

GLMs indicated that utilisation has a significant positive influence on species richness but a negative relationship with the number of positive indicator species. In wet heath models showed that utilisation had a negative relationship with the number of positive indicator species. Models indicated that utilisation was a negative predictor for *Molinia* cover although not significantly so in either model. Utilisation was a significant negative predictor for bryophyte cover and shrub cover. Although trampling can have a negative impact on bryophytes, a moderate level of grazing is required to reduce competition of grasses in favour of bryophytes, so management will require balance (Bullock and Pakeman, 1997; Stewart and Pullin, 2008).

In wet heath, utilisation was a significant predictor for the cover of dwarf shrubs and was significantly correlated with the cover of ericoid species. As expected, the cover of litter, field layer, ground layer and *Molinia* were negatively correlated to the cover of positive indicators (*C. vulgaris, E. cinerea*, *U. gallii* and *V. myrtillus*). The replacement of shrubs by graminoids is consistent with other findings (Olofsson et al., 2001).

3.5.7 Comments on sampling and the exclosures

The vegetation sampling was conducted over three seasons and some caveats should be considered in relation to the results. Longevity of grazing trials is dependent on the maintenance of both boundary fences and grazing exclosures. This is important in relation to cattle husbandry, field worker safety and the quality of the work long term. For example, on a few occasions the cattle broke loose from the site and one (with a collar) was lost completely. Maintenance of the exclosures is crucial to prevent animals (goats, stray sheep and cattle) from getting into them. Experimental grazing exclosures created by Dunne (2000) in the Brandon study site were unfortunately in poor condition by 2013 and grazing animals were able to access them freely. These issues and costs must be considerations for stakeholders in this type of study. The value of studies such as this one is increased greatly if exclosures can be maintained, so that long term effects of grazing and exclusion can be measured.

Extensive grazing leads to slow changes to plant community structure and composition, thus regimes may take years to demonstrate effects (Marriott et al., 2009). Three years is not a long time in ecological terms (Rook et al., 2007), so results from this study should be considered in this context.

3.6 Conclusions

This study explored the impact of conservation grazing on Annex I upland habitats in south west Ireland. At the beginning of the study they were in already in favourable conservation status, so when implementing the grazing trials, the aim was to maintain this status overall and improve or enhance conservation criteria (*sensu* structure and functions) where possible. The results have shown that at the current stocking densities favourable status is being maintained.

However, the time scale is still short considering the approach was cautious and the grazing regime relatively light and summer only. The trends in plots that do have failures on some criteria is towards improvement in all but one plot. Continued monitoring is recommended for that plot. Furthermore, careful monitoring of blanket bog plots is essential in the context of a changing plant community. Longer term monitoring is required to establish more certain trends.

The monitoring of paths is recommended. Cattle used established paths to access different parts of the reserve (section 2.4.1); for desired grazing, water, shelter, and moving due to disturbance by hikers and walking groups. Thus, habitat patches accessible from the paths require careful monitoring for long term damage or habitat change. Although not explicitly examined here, there is also scope for investigating the use of access to resources (e.g. path management, use of licks and or troughs) in order to encourage (or discourage) free grazers into particular vegetation patches. This would be particularly useful in large commonage areas, where separation into smaller plots by fencing is neither feasible nor desirable.

Dexters on blanket bog, Mt Brandon Nature Reserve. Photo K.Kelly.

4 Chapter 4: Preliminary investigation into the response of ground beetle communities (Coleoptera: Carabidae) to cattle grazing in Mt Brandon Nature Reserve

Carabus granulatus with prey, Mt Brandon Nature Reserve. Photo K. Kelly.

4.1 Introduction

4.1.1 Background

Invertebrates make up approximately 80% of earth's biodiversity and over two-thirds of all terrestrial species (Braby, 2018; Lister and Garcia, 2018). Insect populations are in global decline (Wagner, 2020). The alarming reductions of insect populations have been noted among entomologists for at least a decade, but global media exposure of high profile studies (e.g. Hallmann et al., 2017; Sánchez-Bayo and Wyckhuys, 2019) has brought the plight of the insects into the public eye, even though some of these claims have come under question (Didham et al., 2020). Insect population dynamics are complex and care is needed when evaluating population trends and identifying drivers. Inter-annual population variability is the norm among insect species, which poses challenges when establishing baselines (Didham et al., 2020).

The main pressures on insect taxa are habitat loss and conversion to intensive agriculture, pollution by fertilisers and pesticides, biological factors (introduced species and pathogens), and climate change (Sánchez-Bayo and Wyckhuys, 2019). Insects are intrinsically important components of biodiversity, and play critical roles in the functioning of ecosystems (Walpole et al., 2009). Exploring the reasons for declines and developing conservation measures is therefore of paramount importance.

Extensive grazing practices in upland settings, typically with low stocking densities and seasonal regimes, promote structurally diverse swards and habitat heterogeneity (Dennis, Young and Gordon, 1998). Extensive grazing is typical of upland landscapes and in the creation of a patchy network of habitats conducive to multiple invertebrate needs. The promotion of heterogeneous habitats in the uplands through grazing has impacts on invertebrate communities through the effects of selective grazing, trampling and defecating (Cole et al., 2010).

Grazing can change invertebrate communities through alterations of plant growth, architecture and diversity (Kruess and Tscharntke, 2002a). Van Klink et al. (2015) identified three main factors by which grazing impacts on arthropod communities (i) disturbance and unintentional predation (direct effects), (ii) decreases in resource availability (direct effects), and (iii) changes in plant diversity (vegetation-community-mediated effects), structure (vegetation-structure-mediated effects) and abiotic conditions (soil-mediated effects). Although the first two are detrimental, the third factor can be either detrimental or beneficial depending on the circumstance. In some scenarios grazing can increase resource availability by suppressing competitive plant species or by improving climatic conditions through changes in vegetation structure (Wallis de Vries et al., 2016).

Implementing a grazing regime on upland habitats has implications for invertebrate communities. Distinct communities can be encountered under different grazing regimes (Woodcock et al., 2005). Changes to invertebrate communities can be brought about through resource depletion or provision e.g. dung and carrion, and indirect effects via changes in floral community composition and vegetation structure (Dennis, Young and Bentley, 2001; Vickery et al., 2001; Woodcock et al., 2005; Dennis et al., 2015).

Indicators of Change

The conservation status of upland habitats is primarily assessed from a vegetation perspective, yet the sensitivity of invertebrates to fine-scale habitat change makes them valuable indicators in a broader context. Many families of Coleoptera are emerging as useful bioindicators (Rainio and Niemelä, 2003; Avgın and Luff, 2010). They contain some of the best-known taxa in entomology and fill a variety of trophic levels, including scavengers, granivores, herbivores and omnivores (Lövei and Sunderland, 1996). They are globally abundant and a diverse animal group, making them suitable for ecological research. Koivula (2011) reviewed the evidence for using carabids as 'indicators' from a range of perspectives; as indicators of richness and abundance of other taxa, as keystone species, as indicators of human-altered conditions (pollution), indicators of environmental conditions (through biomass dominance), as early warning indicators and as indicators of disturbance and management.

Although carabids are useful model organisms, carabid species richness is a poor indicator for richness and abundance of other taxa (Koivula, 2011). Carabids are sensitive to human induced environmental change (e.g. pollution, pesticide use) and are thus potentially useful indicators of ecosystem health. They also have the potential to reflect soils, wetness and habitat type variation, however they cannot compete with plants as indictors of these factors (Koivula, 2011). Carabid communities host species that are characteristic of habitat types, making them useful dominance indicators (Koivula, 2011).

As early warning indicators, carabids are believed to serve some function e.g. as indicators of climate change (Pizzolotto, Gobbi and Brandmayr, 2014). Although carabid communities respond directly to drastic habitat change such as forestry management (Niemelä and Kotze, 2007) and prescribed burning (Nunes et al., 2006), in such contexts they do not indicate change until after it has become visually obvious, thus not acting as true early warning indicators (Koivula, 2011). However, carabids show a range of responses to grazing regimes (Kotze et al., 2011), are useful indicators of habitat change (Rainio and Niemelä, 2003) and some studies suggest that carabids are good indicators of grazing pressure at assemblage level (Kaltsas et al., 2013).

Carabid beetles are taxonomically and functionally diverse and in the uplands they constitute a significant part of the faunal biodiversity and are of considerable ecological importance as part of the food-web (Dennis, 2003; Pearce-Higgins, 2010). Much work has been done in the British uplands concerning the spatial and functional displacement of beetles in relation to grazer impact (Cole et al., 2005, 2006; Ribera et al., 1999; Cole et al., 2010). In general, these studies have found that large species with poor dispersal abilities (brachypterous/apterous species), will decrease in richness and abundance under increased grazing pressure, while smaller generalist species increase.

The intensification of agriculture is a primary driver of biodiversity declines across the globe. The ongoing removal of natural habitats and persistent use of pesticides and fertilisers associated with intensive agriculture negatively affects multiple unrelated components of farmland biodiversity, including birds, plants and insects (Newton, 2004; Sánchez-Bayo and Wyckhuys, 2019).

Evidence linking grazing management to biodiversity responses in High Nature Value landscapes is particularly lacking in relation to invertebrates (Wallis De Vries et al., 2016). Literature reviewed by Van Klink et al. (2015) revealed that relatively few studies focus on arthropod responses to grazing compared to plants. Therefore, a study on the response of arthropods to grazing is timely, particularly in an upland context.

Carabid beetles occupy a range of trophic levels and fulfil a variety of ecosystem functions (e.g. as predators, food resource, scavengers) and they are sensitive to changes in habitat quality, particularly at the larval stage (Kromp, 1999). Long-term studies of carabids have shown declining trends in biomass and species numbers at local and country scales in the UK (Brooks et al., 2012) and across Europe (Kotze et al., 2011; Hallmann et al., 2017; Homburg et al., 2019). This study explores the response of this important insect group to habitat management in an Irish context.

4.1.2 Carabid beetles

Carabids beetles (Coleoptera; Carabidae) are a diverse group of arthropods that occur in most terrestrial ecosystems (Spake et al., 2016). Carabids or 'ground beetles' predominantly live at ground level during the adult phase (Kromp, 1999) and there are an estimated 40,000 extant species worldwide, approximately 350 of which occur in Britain and Ireland (Luff, 2007). Carabids have long been the focus of entomological studies and their ecology is now well described (Ribera et al., 1999). They are generally very active, with long legs that facilitate rapid travel (Kromp, 1999) and body forms adapted to either running or wedge-pushing (Evans and Forsythe, 1984; Forsythe, 1981, 1987).

An annual lifecycle is most common within Carabidae, although a few larger species in the *Carabus* genus may exist for two or more seasons with both adults and larvae overwintering (Luff, 2007). Female carabids lay 30-600 eggs, usually in soil. Eggs may then be abandoned or provided with a protective capsule, or in some cases shown parental care. There are usually three larval instars before a final rapid pupa stage, all of which occur primarily in soil. Once the adults emerge they may remain *in situ* for some time before becoming active, a strategy which is understood be effective in synchronising emergence post-diapause to achieve greatest breeding success (Luff, 2007).

Carabid beetles are mostly predatory, although they occupy a variety of trophic levels as scavengers, granivores, herbivores and omnivores (Lövei and Sunderland, 1996). In ecosystems they fill multiple functional roles (as predators, prey and granivores) and so provide ecosystem services such pest control, weed control and food for other taxa (Lyons, 2017). Habitat selection in carabids is influenced by microclimatic conditions such as moisture, temperature and light (Thiele, 1977), which makes them sensitive to changes in vegetation management. Their chitinised larvae are particularly sensitive to changes in microclimate (Kromp, 1999). Their role in ecosystem function, coupled with their sensitivity to changes in habitat condition, highlights the importance of understanding their responses to management in the uplands.

Grazing influences carabids at both species and assemblage level in the uplands (Dennis et al., 1997). Carabids can differ in habitat preference depending on geographical location, so direct comparison at species level between different geographical locations is problematic (Cole et al., 2006). Biodiversity indices have been widely used in carabid studies and are regarded as being more reliable as they do not consider the actual species present. However, much ecological insight can be lost by simply looking at community diversity and the use of ecological groupings can allow for more reliable comparisons (Cole et al., 2006).

4.1.3 Carabid Traits and Functional Classification

Members of the Carabid family display the same basic morphology and relatively few habitat-specific adaptations are found, with most variations instead relating to differences in habit (Cole et al., 2002). Although many carabids have the ability to fly (Luff, 2007), they usually move around on foot (Kromp, 1999). Wing development is variable, from fully winged to wingless (Zalewski and Ulrich, 2006). Furthermore, wing-dimorphism occurs within species, where only part of the population is fully winged (Zalewski and Ulrich, 2006).

Apterous species have no wings and brachypterous species have wings shorter than the elytrae. They have lower dispersal ability (Gobbi et al., 2007) and are known as '*per pedes*' colonists (Brandmayr, 2005 in Gobbi et al., 2007). Temporally stable (undisturbed) habitats tend to be characterised by large wingless species (e.g. *Carabus* sp.), whereas macropterous species tend to be associated with fragmented or unstable habitats (Kromp, 1999; Ribera et al., 2001).

Environmental quality and resource availability influence metabolic activity, generation times and body size of organisms (West, Brown and Enquist, cited in Langraf et al., (2017). Changes in the patterns of species distributions and body size can be linked to the level of environmental burden. Body size ('biovolume' or 'ellipsoid biovolume') and wing form of carabids are linked to levels of disturbance, which makes them useful as bioindicators. Large-bodied, wingless and brachypterous species have lower dispersal ability and are associated with less disturbed systems. Conversely, smaller-bodied, winged species are associated with systems that are subject to disturbance. Szyszko, cited in Langraf et al., (2017), showed that a decrease in environmental disturbance allows for larger than average body size.

Carabids were classified into distinct ecological groups by Cole et al., (2002). The influence of agricultural land use and management intensity was then investigated. The study introduced a method of classifying carabids based on their ecology rather than taxonomy. The influence of management practices across ecological groups can be investigated using this approach, rather than relying on individual indicator species (Cole et al., 2002). Furthermore, it alleviates the reliance on diversity indices which can mask underlying ecological nuance.

Cole et al., (2002) examined 10 ecological traits (size, overwintering state, lifecycle, food of adult, diel activity, breeding season, emergence, main activity period, wing morphology and locomotion). The separation of species in ordination space was primarily related to size (and associated attributes e.g. 2-year lifecycles and overwintering as larvae) and diet (generalist/specialist predators, phytophagous species). Seven ecological groupings were identified using fuzzy clustering. For example, large wingless predatory species grouped together (all *Carabus* species), small diurnal plant feeders grouped together (also included two nocturnal Collembola specialists *in* the *Trechus* genus), and small nocturnal predators grouped together (e.g. *P. rhaeticus*) (Cole et al., 2002).

In agricultural landscapes, larger brachypterous or apterous carabid species tend to be associated with less intensively managed (often) upland sites, whereas macropterous and dimorphic species tend to be more associated with disturbed, typically lowland sites (Ribera et al., 2001). Large wingless carabid species such as *Carabus glabratus* and *C. problematicus* have been shown to be associated with less disturbed upland habitats (Dennis et al., 1997; Cole et al., 2002).

Strong relationships exist between morphological and life trait characteristics of carabids and the environmental characteristics of their habitats, as first described and predicted by habitat templet theory (Southwood, 1977). Summer breeding species are associated with less disturbed sites and high elevation while spring and autumn-winter breeders are associated with lowland landscapes, which can be related to the limitations of climate at high elevation or the increased disturbance in lowland agriculture during summer months (Ribera et al., 2001).

Carabids have been successfully linked to habitat assessment and to agricultural management due to their sensitivity to disturbance and their relationship with vegetation structure (reviewed in Koivula (2011)). Carabids of upland, extensively managed landscapes are typically generalist predators (e.g. *Carabus problematicus*), while specialist predators (e.g. of Collembola) or species that incorporate plant material to their diet tend to be associated with lowland, intensively managed landscapes.

4.1.4 Functional Classification by Locomotion and Shape

Many contrasting adaptations for locomotion exist within Coleoptera. For example, *Cicindela campestris* is a predator that occurs in open sandy habitats (heaths and dunes) in Britain and Ireland that is adapted for short flights and sprinting (runs at 0.6 $m.s^{-1}$). *C. campestris* has long thin legs that allow for rapid acceleration and deceleration (Evans and Forsythe, 1984). At the other end of the 'speed/force' spectrum is *Geotrupes stercorarius*, a dung beetle evolved for pushing and excavating. However, with max speeds of 0.04 $m.s^{-1}$, it is a very slow walker (Evans and Forsythe, 1984).

Families of beetles within Coleoptera that are adapted for speed are usually predators, including Carabidae and Staphylinidae. 'Force-adapted' families include Geotrupidae, Scarabaeidae, Lucanidae, and Histeridae. Rapid runners occur at one end of a speed-force (S/F) spectrum, powerful burrowers at the other, and many species occur in-between (Evans and Forsythe, 1984; Evans, 1990). Adaptations in form and function allow functional groups to be defined, with specialisations reflecting different habits. These habits may further fit these functional groups into a particular habitat (Sharova, in Evans and Forstythe 1984).

Carabids have a wedge-push ability that facilitates moving through leaf litter and soil. The head and thorax are wedge shaped and can enlarge spaces moving forward, and vertical oscillations of the hind body enhances the wedge effect. Forsythe (1981) examined locomotion and function in ground beetles based on leg structure, adopting the classification formulated by Erwin, Whitehead and Ball (1977) and Kryzhanovskii

(1975), both cited in Forsythe (1981). These arranged Carabidae into the following groups (Table 37 Groupings of Carabidae based on leg structure, from Forsythe (1981):

Table 37 Groupings of Carabidae based on leg structure, from Forsythe (1981)

Carabinae group 1	Carabinae group 2	Carabinae group 3
Nebriitae	Broscitae	Scarititae
Loriceritae	Trechitae	
Elaphritae	Patrobitae	
Carabitae	Pterostichitae	
Cincindelitae	Callistitae	
	Harpalitae	

Group 1 (above) are fast runners but weak horizontal pushers, group 2 are stronger horizontal pushers but slower runners than group 1, and group 3 are very strong horizontal pushers but slow runners (Forsythe, 1981). However, Forsythe states that Carabini (including Cychrini) (supertribe Carabitae (Kryzhanovskii, 1976a)) resemble group II in their running and pushing abilities, even though their legs are like Group I (long, slender). Carabitae therefore, show characteristics of both groups; they are fast walkers and strong pushers. This allows them to tackle large, slower moving prey (molluscs, worms, caterpillars) compared to other species in the group (Forsythe, 1981).

The large body form of Carabitae allows species in this supertribe to move through environments that offer more 'resistance' (after Heydemann (1957) and Thiele (1977), cited in Evans and Forsythe, 1984). Dense habitats such as meadows, grasslands and woodland-pasture are more difficult to traverse than open woodland floor or sandy substrates. The physical build of Carabitae allows them to cope with the higher levels of resistance offered by such habitats compared to other group 1 species. Long, strong legs and large bodies allow these species to push though grass and litter that offers moderate resistance (compared to high resistance offered by soil) i.e. they are formidable 'wedge-pushers' that combine group I and group II characteristics (Evans and Forsythe, 1984).

4.1.5 Functional Traits and Environmental Change

Changes to the environment can influence species differently depending on morphological traits (Ng et al., 2018). In general, large wingless species are more prone to extinction. Small species have high dispersal ability and are selected for in habitats that experience high levels of disturbance. Less disturbed habitats select for large species with low dispersal ability (Lövei and Sunderland, 1996; Ribera et al., 2001; Cole et al., 2002; Ng et al., 2018), for example those in the *Carabus* genus as described by functional group 2 in Cole et al., (2002). Agricultural land use and management has been found to influence composition of carabid communities in Scotland, with the *Carabus* genus being most sensitive to change (Cole et al., 2002). Alterations to the composition of carabids may have knock on consequences for other taxa and ecosystem functioning (Cole et al., 2002).

Studies of insects have shown alarming reductions in this class of invertebrates. Insects are important in ecosystem functioning and declines are likely to have serious consequences for natural processes (Walpole et al., 2009). Studies of carabid beetles in mainland Europe have demonstrated population declines, likely due to anthropogenic disturbance and climate change (Kotze and O'Hara, 2003). Substantial declines have also been shown in the UK (Brooks et al., 2012). Comparable studies in

Ireland are few, particularly in relation to HNV environments and in the context of carabids as indicators of change.

4.1.6 The Irish Uplands and Carabids

In Ireland, most studies of arthropods have been conducted at community level. McFerran et al (1994) investigated the impact of grazing on ground beetles in the uplands of Northern Ireland and found that areas subjected to the highest grazing intensity were characterised by species indicative of disturbed land, and areas where grazing had ceased had characteristic coloniser species (McFerran et al., 1994). Woodcock et al. (2004) examined the effects of grazing and turf cutting on carabids of oceanic blanket bog on the Beara peninsula in county Cork (with focus on *C. clatratus*). The study found that while there was no negative effect on *C. clatratus*, distinct communities were observed on disturbed sites compared to 'pristine' sites.

Williams and Gormally (2010) studied the effects of blanket bog management on ground beetles at a community level with focus on *Carabus clatratus* (and in parallel conducted the first national survey of this species). The study found that afforestation was a greater driver of ground beetle community change than was overgrazing (or hand turf cutting). The results also indicated that hand turf cutting sites are more like pristine sites than overgrazed or eroded sites. *C. clatratus* abundance was positively related to ground temperature (suggesting that areas of bare peat are important for this species) and that it did not show any preference for intact blanket bog, with cut-over and raised bogs being equally important (Williams and Gormally, 2010).

Anderson (2013) examined the effects of mixed grazing management and altitude on carabids and ecosystem services. The aim of this work was to describe the carabid communities of upland habitats in county Kerry and investigate their response to grazing in an HNV farming context. The study found that generalist predators and phytophages were more abundant in the lowlands and that specialist predators were abundant in the uplands. Specialist predators appeared to prefer shorter vegetation, while generalist abundance and richness was dependent upon the interaction between altitude and grazing 'state' (categories of low, moderate, high). *Carabus clatratus, C. problematicus, Pterostichus melanarius* and *Nebria brevicollis*, were (surprisingly) abundant in heavily grazed areas and *Abax parallelepipidus, Carabus granulatus* and *Pterostichus niger* were more abundant where grazing was less intensive (Anderson, 2013). The study sites of Anderson (2013) were improved grassland, lowland blanket bog, upland blanket bog and acid grassland. Field indicators such as leaf litter, cover of bare ground, selectivity of grazing and presence of dung were used to classify grazing intensity into Low (0-.02 $LU.ha^{-1}$), Medium and High (up to 0.48 $LU.ha^{-1}$ at her upland sites). In the present study, the habitats were different (heaths and upland blanket bog), were in relatively good condition, and were grazed at a low level by comparison. Therefore, care must be taken when comparing results with the present study.

4.2 Aim and Objectives

The aim of this study was to describe the Carabid assemblages (Coleoptera: Carabidae) of Mount Brandon Nature Reserve (MBNR) in Co Kerry and explore the relationship between ground beetles and cattle grazing.

Objectives:

1. To sample ground beetles from three upland habitats in MBNR: Northern Atlantic Wet Heath with *Erica tetralix*; European Dry Heath and Active Blanket Bog
2. To compare the carabid communities from grazed and ungrazed treatments in each habitat under examination
3. To examine the potential for using large carabids as functional indicators/ early warning indicators for grazing disturbance in sensitive upland habitat

4.3 Methodology

4.3.1 Overview

Thirty Dexter cattle grazed Mount Brandon Nature reserve between 2011 and 2015 as per agreement between the grazier (Mr Paddy Fenton) and the NPWS. A study on the home range, habitat selection and behaviour of the cattle was conducted between 2013 and 2015, as detailed in Chapter 2. Chapter 3 describes the vegetation sampling that took place between 2013 and 2015. In parallel with these investigations, sampling of ground beetles (Coleoptera, Carabidae and Staphylinidae) and spiders (Araneae) took place in 2013 and 2015. Spider data were analysed separately for an undergraduate thesis at IT Tralee by Geraghty (2014).

4.3.2 Location and Site

The site and its sub-plots (grazed plots, exclosures) for the ground beetle study were the same as those used for the home range (chapter 2) and vegetation (chapter 3) components of the work. A detailed site description and design of the grazing experimental plots is provided in section Chapter 3 section 3.3.1.

4.3.3 Invertebrate sampling

In 2013 and 2015 invertebrate sampling was conducted once per month using pitfall traps from May to September. In addition to the 4 control plots, seven 50 x 50 m experimental plots (grazed sites) were randomly selected: x1 blanket bog, x2 dry heath, x2 wet heath and x2 wet grassland (giving a total of 11 plots). The 50 x 50 m fenced exclosures were the control sites for each habitat. Fifteen pitfall trap samples were collected from each of these plots once per month. Traps were placed in association with the (randomly placed) vegetation quadrats to maintain the link between beetle and vegetation data (Dennis et al., 1997; Bhriain, Sheehy Skeffington and Gormally, 2002).

Pitfall traps were active (left open) between day 1 and day 15 of each month from May to September. Thereby, 825 samples were collected in 2013. In order to lessen the impact on beetle populations in the study area, sampling was reduced in 2015. In each of the four habitats, one grazed and one control plot was sampled each month, giving a total of 600 samples for that year.

Standard pitfall trapping methods were followed, such as those described by Luff (1975) and reviewed by Skvarla, Larson and Dowling (2014). Polypropylene cups 7.5 cm in diameter and 10 cm deep were dug-in, and polypropylene covers were pinned over each trap using wooded skewers to provide cover. Approximately 30 mL of 100% propylene glycol was placed as a killing agent and preservative in each trap, with 2-3 mL of detergent added to reduce surface tension. Undiluted propylene glycol was used in this case due to the very wet nature of the study site and the widespread occurrence of trap flooding and dilution of the preservation agent. GPS locations, photographs and paced coordinates were recorded to assist with finding traps at collection time. After collection, samples were preserved in methylated spirits to prevent deterioration of material. Samples were sorted, and beetles were identified in the laboratory using microscopes. Identification and nomenclature followed Luff (2007).

Pitfall traps are commonly used to catch surface active invertebrates in heathland (Luff, 1975; Gardner et al., 1997; Cameron and Leather, 2012). They provide relative abundance data based on invertebrate density and activity rather than absolute abundance (Greenslade, 1964) and are unlikely to catch inactive species. However, pitfall traps are easy to set-up and service, are repeatable between sites and studies, are cost effective and have high catch rates. Thus they are appropriate in studies of large sites, with multiple sampling locations and time periods (Oxbrough et al., 2012; Lyons et al., 2017).

4.3.4 Data Analysis

4.3.4.1 General and Community

Trap data were pooled for the 50 x 50 m plots by month. For community analysis, data were standardised by trap day. Data were standardised by calculating the abundance of each species at each location, dividing it by the number of actual trap days at that location and then multiplying it by the maximum number of traps days across all locations (475). Standardising by trap day is a method commonly used in studies using pitfall trapping data (Bergeron et al., 2013; Blanchet et al., 2013). Statistical analyses were carried out using SPSS Statistics, version 25 (IBM Corp., 2017) and PC-Ord version 6.17 (McCune and Mefford, 2011).

Ordinations were carried out using Non-metric Multi-dimensional Scaling (NMS) in PC-Ord. NMS is primarily used for displaying relationships between samples and environmental variables. NMS avoids the assumption of linearity between variables, is suitable for non-normal data and allows any distance measure to be used. It is also less prone to outliers than other ordination methods as it is based on rank distances (McCune, Grace and Urban, 2002).

An initial autopilot NMS run was conducted using Sørensen's (Bray-Curtis) as a distance measure. It was conducted with 250 runs of real data and 250 runs of randomised data, stepwise reduction in dimensionality with each cycle, a stability criterion of 1×10^{-7} standard deviations in stress over 10 iterations and a maximum of 200 iterations. The final ordination was run using 3 axes, stepwise reduction in dimensionality (step length = 0.20), a stability criterion of 1×10^{-7} standard deviations in stress over 10 iterations and a maximum of 500 iterations. The best starting configuration and optimal number of axes was found based on the autopilot run. The optimal number of axes was determined as the number of axes beyond which reduction in stress was small.

Environmental variables were overlain on NMS ordinations and correlation co-efficients were calculated between variables and the axes of the ordination plots to determine the relationship between community composition and environmental factors (McCune, Grace and Urban, 2002). Most variables were not normally distributed, so Kendall's tau (τ) was used, a rank correlation coefficient similar to Spearman's but more suited to smaller datasets with tied ranks (Field, 2013, p.278).

Multi-Response Permutation Procedure (MRPP) (Zimmerman, Goetz and Mielke, 2006) was used to test the differences between carabid communities among habitat types and grazing treatment. MRPP is a non-parametric method of testing differences between or among multiple groups (Peck, 2010). Community data do not usually meet assumptions of normality required for equivalent parametric tests such as Discriminant Analysis (DA) (McCune and Grace, 2002). The *p*-value produced by MRPP expresses the likelihood of occurrence of an expected delta (weighted mean within-group distance) being smaller or equal to the observed delta. 'A', the chance-corrected within-group agreement is also calculated, which compares the within-group heterogeneity with expected values from randomised data. It serves as a measure of how well defined the groups are, where A=1 if samples within group are identical; A=0 if within group heterogeneity is equal to that expected by chance and A<0 if there is less agreement than expected by chance. In community ecology, values are usually less than 0.1 (Bruce and Grace, 2002). MRPP is a technique that compares the mean ecological distance between predefined groups, against the distance in groups created at random from the same dataset (Steinauer and Collins, 2001). It is similar in purpose to the *t* test but has more relaxed requirements on data structure (Zimmerman, Goetz and Mielke, 2006).

To assess differences in carabid community compositions between grazing treatments, two-factor Permutational Multivariate Analysis of Variance (PerMANOVA) (Anderson, 2001) was used with 4999 randomisations, Sorensen's (Bray-Curtis) distance measure and groups defined by 'habitat' and 'grazing'. PerMANOVA provides statistical analysis based on dissimilarity measures (Euclidean or non-Euclidean-embeddable measures). The technique mirrors ANOVA, with *p*-values obtained using distribution free techniques. It allows for robust analysis of multi-variate systems, even where data are over dispersed or non-normal in distribution (Anderson, 2017). In ecology

PerMANOVA facilitates the analysis of variation in community structure (beta diversity) across multiple spatial or temporal scales (Anderson, 2017).

4.3.4.2 Wing State, Locomotion and Body Size

For carabid species, as habitat becomes more stable and as time since colonisation increases, the proportions of apterous/brachypterous, macropterous and dimorphic species is predicted to change (Lövei and Sunderland, 1996; Riley and Browne, 2011).

Species were classified according to (1) wing morphology (apterous/brachypterous ('wingless') and macropterous/dimorphic ('winged')); and (2) locomotion (pushers and runners). For the purposes of analyses and presentation, apterous/brachypterous species are referred to as 'wingless' and macropterous/dimorphic species are referred to as 'winged'. Classifications for locomotion follow Erwin, Whitehead and Ball, (1977); Forsythe, (1983;) Evans and Forsythe, (1984). Group 1 (runners) are large-bodied winged species that include Carabini and Cychrin (Carabitae), and Group 2 (pushers) includes Pterostichini, Nebriini, Harpalini and Platynini.

The proportions and frequencies of the two groups were tabulated and examined in relation to Utilisation Score using linear regression. This was conducted with a view to developing a more simplified and practical assessment method for field-based application, particularly in farmland settings. Table 38 shows the carabid species captured during sampling, along with ecological traits.

The proportions and frequencies of the two groups were tabulated and examined in relation to Utilisation Score using linear regression.

The ratios within and between different groups were examined to explore the magnitude of change over time. For example, within group I, the ratio of Carabitae to Brachypterous was examined to detect any changes.

Table 38 Carabid species from MBNR, their feeding groups, wing morphology and locomotion. Information on locomotion is taken from (Forsythe, 1983; Cole et al., 2002).

Species	Group	Locomotion	Wing Morphology
Abax parallelepipedus	Generalist predator (Purtauf, Dauber and Wolters, 2005; Harvey et al., 2008)	Pusher	Apterous/Brachypterous (Mazzei et al., 2015)
Carabus clatratus	Generalist predator (Huk and Kühne, 1999)	Runner	Dimorphic (McFerran, McAdam and Montgomery, 1995)
Carabus granulatus	Generalist predator (Purtauf, Dauber and Wolters, 2005)	Runner	Apterous/Brachypterous (Ribera et al., 1999)
Carabus problematicus	Generalist predator (Cole et al., 2002; Harvey et al., 2008)	Runner	Apterous/Brachypterous (Cole et al., 2002)
Cychrus caraboides	Generalist predator (Cole et al., 2002)	Runner	Apterous/Brachypterous (Cole et al., 2002)
Harpalus latus	Phytophagus (Cole et al., 2002)	Pusher	Macropterous (Cole et al., 2002)
Leistus terminatus	Specialist predator (collembola) (Cole et al., 2002; Bauer (1985), cited in Ribera et al., 2001)	Runner	Macropterous (Cole et al., 2002)
Nebria brevicollis	Generalist predator (Cole et al., 2002; Haysom et al., 2004)	Runner	Macropterous (Cole et al., 2002)
Olisthopus rotundatus	Generalist predator (Cole et al., 2002)	Pusher	Dimorphic (Cole et al., 2002)
Pterostichus diligens	Generalist predator (Ribera et al., 2001; Cole et al., 2002)	Pusher	Dimorphic (Cole et al., 2002)
Pterostichus melanarius	Generalist predator (Cole et al., 2002; Haysom et al., 2004)	Pusher	Dimorphic (Cole et al., 2002)
Pterostichus niger	Generalist predator (Cole et al., 2002; Haysom et al., 2004)	Pusher	Macropterous (Cole et al., 2002)

4.4 Results

4.4.1 Overview

A total of 2435 carabid beetles were caught from 16 species in the habitats under examination. A singleton of *Notiophilus* sp. was discounted from the analysis. 1435 beetles were caught in 2013, compared to 1000 in 2015. Sampling was scaled down in 2015. It was reduced in the second year due to more limited resources and to lessen the potential impact on beetle populations on the study site. Furthermore, a higher proportion of traps were lost in 2015 (22%) compared to 2013 (14%) due to flooding and interference by cattle. Of the three habitats, 60.4% of all beetles were caught in dry heath, with 23% in blanket bog and 16% in wet heath. Singletons of *Agonum fugilinosum* and *Olisthous rotundatus* were also recorded. *O. rotundatus* was included in the ordinations in error. The inclusion of rare species in ordinations may influence the model summarising the underlying data (Poos and Jackson, 2012).

Abax parallelepipedus was the most abundant species, making up 40% of the total catch (1038 individuals). *Pterostichus rhaeticus* accounted for 18% of the total catch (441) and *Carabus problematicus* made up 17% (412). *Pterostichus melanarius* was the next most abundant carabid, making up 11% of the total (270 individuals) and the rest of the carabid species accounted for less than 10% of the catch each. Table 39 presents summary statistics for all species.

Table 39: Summary statistics for Carabid species in MBNR: CV% (coefficient of variation, as a percentage) and V/M (variance to mean ratio)/. V/M indicates deviation from a Poisson distribution, with large numbers indicating aggregation and small values indicating dispersion) (McCune and Mefford, 2011).

	RankAbun	Sum	RankFreq	Freq	Mean	S.Dev.	CV%	V/M
Abax parallelepipedus	1	1038	1	51	17.30	24.8	143.1	35.4
Pterostichus rhaeticus	2	441	3	35	7.35	16.9	230.4	39.0
Carabus problematicus	3	412	2	50	6.87	10.6	154.0	16.3
Pterostichus melanarius	4	270	5	28	4.50	8.1	180.6	14.7
Pterostichus niger	5	136	4	31	2.27	5.2	228.0	11.8
Carabus granulatus	6	64	6	25	1.07	1.8	164.5	2.9
Cychrus caraboides	7	29	7	17	0.48	1.0	199.7	1.9
Pterostichus diligens	8	27	8	15	0.45	1.1	246.9	2.7
Nebria brevicollis	9	19	9	7	0.32	1.1	356.0	4.0
Pterostichus strenuus	10	9	12	4	0.15	0.7	439.6	2.9
Harpalus latus	11	5	11	4	0.08	0.3	400.8	1.3
Pterostichus madidus	12	5	13	3	0.08	0.4	457.7	1.7
Leistus terminatus	13	4	10	4	0.07	0.3	377.3	0.9
Carabus clatratus	14	2	14	2	0.03	0.2	543.1	1.0
Olisthopus rotundatus	15	1	15	1	0.02	0.1	774.6	1.0
Agonum fuliginosum	15	1	15	1	0.02	0.1	774.6	1.0
Notiophilus sp.	15	1	15	1	0.02	0.1	774.6	1.0

Dry heath had the highest abundance of beetles caught (1487), followed by blanket bog (541) and wet heath (407). Large carabid species were dominant in dry heath, with *A. parallelepipedus* making up 56% of the catch, and *C. problematicus* and *C. melanarius* accounting for 17% each.

Blanket bog was dominated by *P. rhaeticus* (62%), with *A. parallelepipedus* and *C. problematics* accounting for 11% each. In terms of abundance, wet heath was intermediate between dry heath and blanket bog, and *A. parallelepipedus* was the most abundant species (34%), followed by *C. problematics* (24%) and *P. rhaeticus* (22%) (Table 39).

Table 40 Carabid beetle abundance (raw data) in three habitats in MBNR (UG = ungrazed, G = Grazed).

	Wet Heath				Dry Heath				Blanket Bog					
	2013		2015		2013		2015		2013		2015			
	UG	G	UG	G	UG	G	UG	G	UG	G	UG	G	Tot.	%
Abax parallelepipedus	31	75	17	16	257	280	131	168	15	12	11	25	1038	42.6
Pterostichus rhaeticus	13	27	20	27	0	1	1	0	15	145	24	168	441	18.1
Carabus problematicus	23	41	22	11	91	92	32	34	2	s	9	29	386	15.9
Pterostichus melanarius	5	1	2	1	84	43	40	85	1	0	1	7	270	11.1
Pterostichus niger	3	24	2	0	26	32	6	21	5	8	6	3	136	5.6
Carabus granulatus	1	8	1	5	0	7	15	14	4	3	3	3	64	2.6
Cychrus caraboides	1	3	3	0	6	2	5	2	3	1	3	0	29	1.2
Pterostichus diligens	1	2	0	7	1	0	2	0	1	8	2	3	27	1.1
Nebria brevicollis	1	7	0	0	0	0	0	0	0	10	0	1	19	0.8
Pterostichus strenuus	0	0	1	1	0	0	0	0	0	0	4	3	9	0.4
Harpalus latus	0	0	0	0	3	1	1	0	0	0	0	0	5	0.2
Leistus terminatus	0	1	0	0	0	3	0	0	0	0	0	0	4	0.2
Pterostichus madidus	0	2	0	0	0	0	1	0	0	0	0	0	3	0.1
Carabus clatratus	1	0	0	0	0	0	0	0	1	0	0	0	2	0.1
Olisthopus rotundatus	0	0	0	0	0	0	0	0	0	1	0	0	1	0.04
Agonum fuliginosum	0	0	0	0	0	0	0	0	0	0	0	1	1	0.04
Total Abundance	**80**	**191**	**68**	**68**	**468**	**461**	**234**	**324**	**47**	**188**	**63**	**243**	**2435**	
Species Richness	**13**				**11**				**12**				**16**	

Species richness varied little between habitats. Thirteen species were recorded in wet heath, 11 blanket bog and 12 species in dry heath. *Harpalus latus* was not present in wet heath, *Nebria brevicollis*, *Pterostichus strenuus* and *Carabus clatratus* were not present in dry heath, and *Leistus terminatus*, *Pterostichus madidus* and *Harpalus latus* were not recorded in blanket bog. Singles of *Olisthopus rotundatus* and *Agonum fuliginosum* were recorded in blanket bog. Two individuals of *C. clatratus* were caught, one each in blanket bog and wet heath.

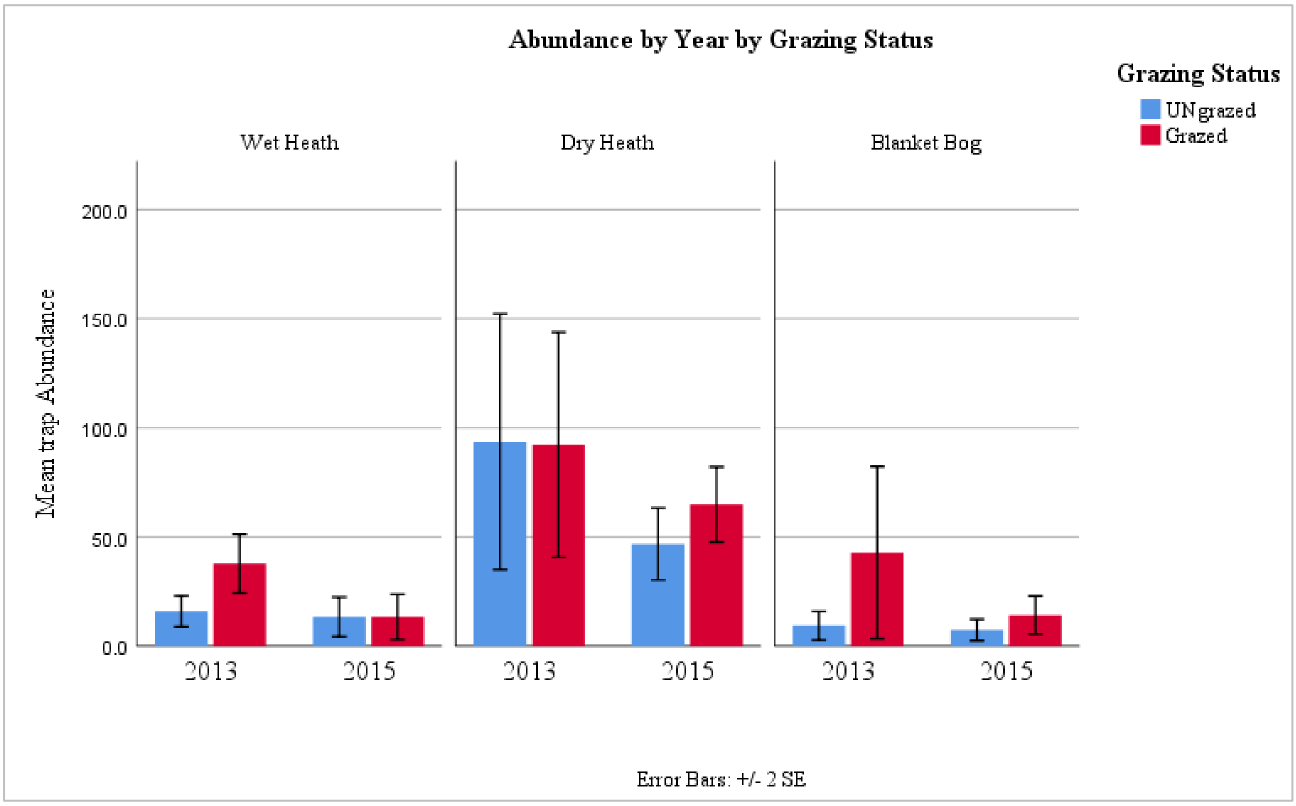


Figure 34 Carabid abundance by year and grazing status (raw data).

4.4.2 Community response

For the full dataset (2013 and 2015), the best solution presented by NMS was 3-dimensional, with a stress value of 11.6. Values of close to 10 are considered interpretable and reliable (McCune and Grace, 2002; Peck, 2010). The three axes represent a total of 91.0% of the variance in the dataset. Axis 1 explains 60.5% of the variance and axes 2 and 3 explain 18.6% and 11.6% respectively. The ordination was plotted for axes 1 and 2 (Figure 35) as they represent most of the variation and are ecologically interpretable. The habitat groups were reasonably well differentiated, with dry heath samples on left of the plot, characterised by large carabid species such as *Abax parallelepipedus*, *C. problematicus*, *Pterostichus niger* and *P. melanarius*. These plots are associated with high shrub cover and steeper slopes.

The blanket bog samples are on the right of the ordination space, are characterised by *P. rhaeticus, C. clatratus, P. diligens* and *P. strenuus*, and are associated with high field layer cover and deep soils. The wet heath plots are widely scattered, depicting the variable nature of the habitat in the study area and an even community structure, with species of both wet and dry habitats being represented e.g. *P. rhaeticus* (wet), and *C. problematicus* (dry). The wet heath and blanket bog plots were correlated with deeper soils, higher field cover and more leaf litter.

Exploration of subsets of the data by year and by treatment revealed similar patterns to the full dataset. Figure 36 and 37

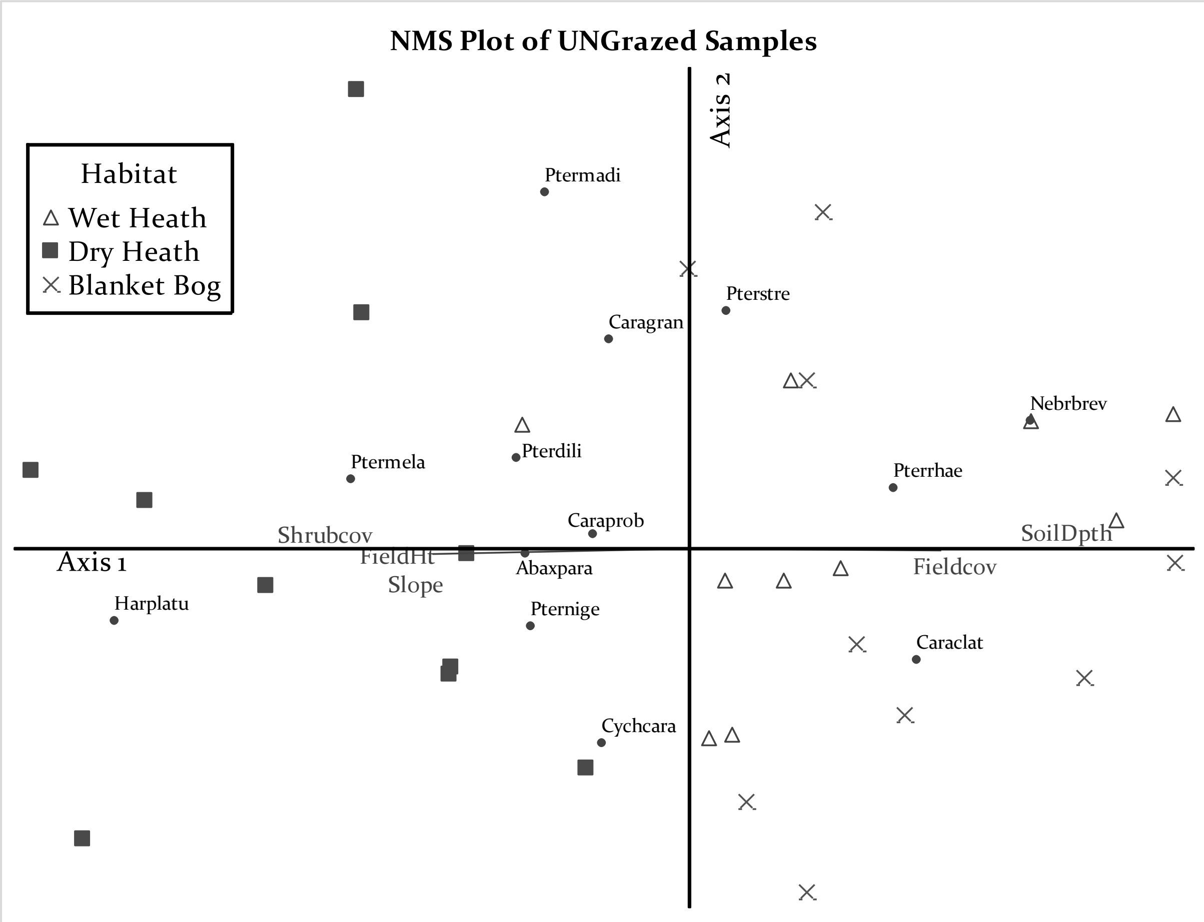


Figure 37 show the ordination space for grazed and ungrazed plots separately. The blanket bog plots were again characterised by small carabid species such as *P. rhaeticus* and *P. diligens*, while large carabids such as *C. niger* and *P. melanarius* were characteristic of dry heath, with wet heath plots being more widely scattered in ordination space.

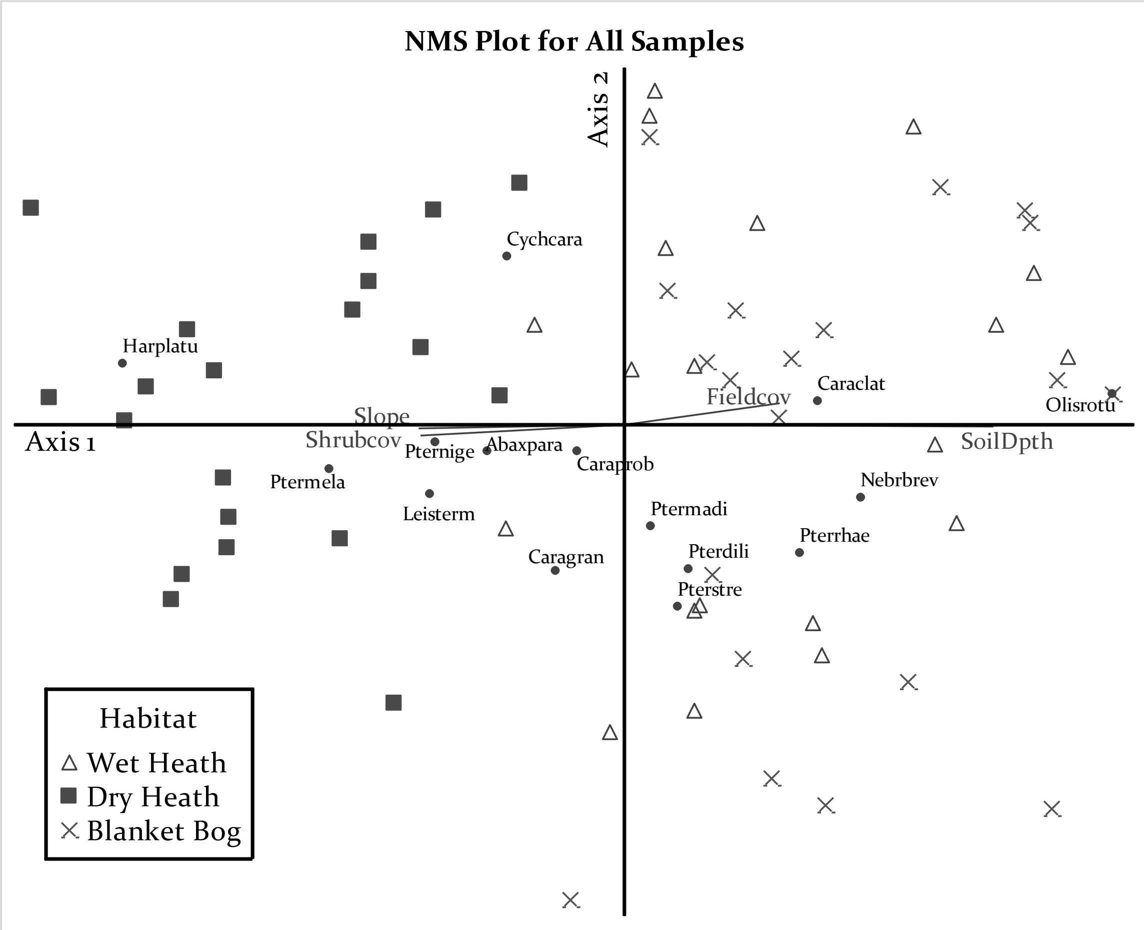


Figure 35 NMS plot of carabid samples, grouped by habitat. Environmental variables which are significantly correlated with the axes are also plotted.

Coefficients of determination for the correlations between ordination distances and distances in the original n-dimensional space for the full dataset

Axis	Increment (r^2)	Cumulative
1	0.605	0.605
2	0.186	0.791
3	0.116	0.907

Kendall's tau correlation coefficients between NMS axes and environmental variables. Coefficients τ > 0.2 are in **bold**).

	Axis 1	Axis 2	Axis 3
Util	-0.043	-0.029	-0.165
BareS	-0.027	-0.177	0.137
Litter	**0.24**	0.147	-0.156
Fieldcov	**0.351**	0.213	-0.124
Shrubcov	**-0.376**	-0.144	0.182
FieldHt	0.024	0.196	0.004
Slope	**-0.448**	-0.085	0.15
SoilDpth	**0.55**	-0.042	-0.152

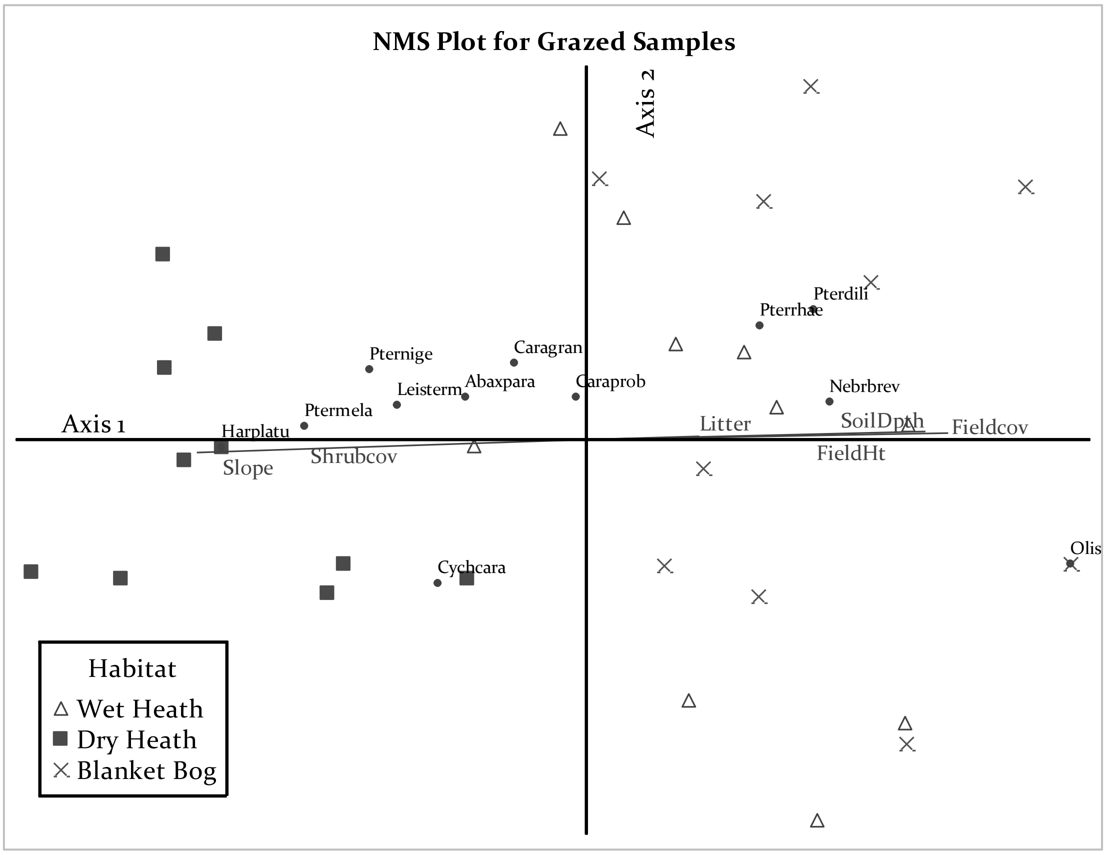


Figure 36 NMS plot of carabid samples from grazed plots, grouped by environmental variables which are significantly correlated with the axes are also plotted.

Coefficients of determination for the correlations between ordination distances and distances in the original n-dimensional space for the full dataset:

Axis	Increment (r^2)	Cumulative
1	0.61	0.61
2	0.13	0.743
3	0.068	0.810

Kendall's tau correlation coefficients between NMS axes and environmental variables.

	Axis 1	**Axis 2**	**Axis 3**
	tau	tau	tau
Util	-0.094	0.066	0.212
BareS	0.051	0.226	-0.072
Litter	**0.285**	0.067	0.092
Fieldcov	**0.641**	-0.022	0.181
Shrubcov	**-0.379**	-0.032	-0.087
FieldHt	**0.692**	0.092	0.144
Slope	**-0.508**	-0.102	-0.017
SoilDpth	**0.557**	0.151	0.047

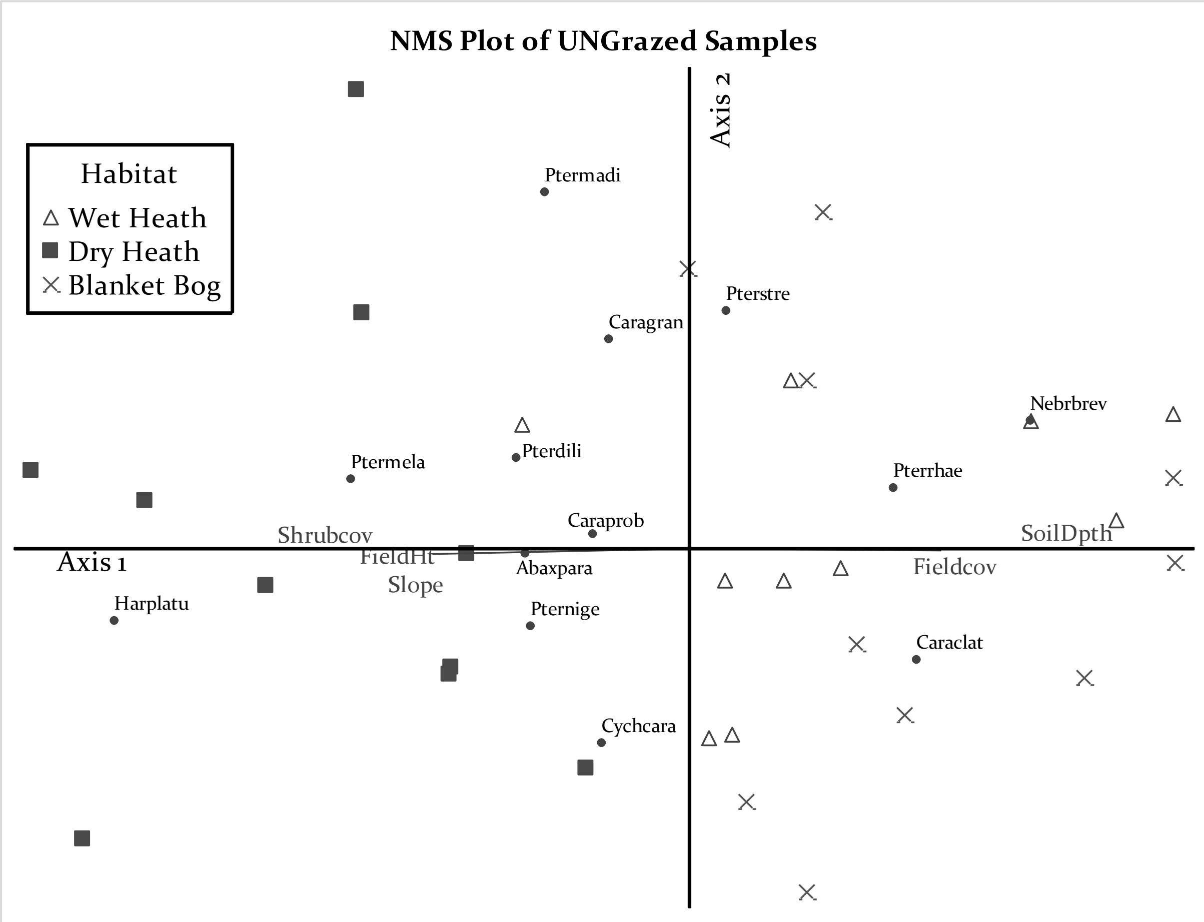


Figure 37 NMS plot of carabid samples from ungrazed plots, grouped by habitat (WH = wet heath, DH = dry heath, BB = Blanket Bog). Environmental variables which are significantly correlated with the axes are also plotted.

Coefficients of determination for the correlations between ordination distances and distances in the original n-dimensional space for the full dataset:

Axis	Increment (r^2)	Cumulative
1	0.74	0.674
2	0.150	0.824
3	0.112	0.936

Kendall's tau correlation coefficients between NMS axes and environmental variables.

	Axis 1	Axis 2	Axis 3
Util	0.004	-0.187	-0.268
BareS	**-0.231**	0.031	0.241
Litter	**0.29**	-0.087	-0.072
Fieldcov	**0.493**	0.007	0.012
Shrubcov	**-0.451**	0.017	0.027
FieldHt	**-0.517**	-0.052	0.022
Slope	**-0.527**	0.027	0.072
SoilDpth	**0.569**	0	-0.149
PropApt	**-0.562**	0.042	0.077

Results of two-way PerMANOVA revealed that carabid community assemblages varied significantly by habitat (pseudo-F=10.566, df=2, P(perm)=0.002), but not by grazing state (pseudo-F=1.6716, df=1, P(perm)=0.159) and there was no significant interaction between the two factors (pseudo-F=0.5320, df=2, P(perm)=0.8306).

MRPP results show significant differences between habitat types and indicate that the chance-corrected within-group agreement (within group homogeneity) was relatively high (observed delta = 0.4524, expected delta = 0.5302; A = 0.1467, $p < 0.001$). No significant differences were found between grazed and ungrazed samples and within group homogeneity was low (observed delta = 0.5278, expected delta = 0.5301; A = 0.004, $p = 0.2106$).

Table 41 Permutation based nonparametric MANOVA (perMANOVA) (method follwing Anderson, 2001), evaluating differences in species composition between groups. Design: two-way factorial, with groups defined by 'Habitat' and 'Grazing' (grazed or ungrazed). Distance measure = Sorensen (Bray-Curtis). Randomisation test of significance of pseudo F values with 4999 randomisations.

Source	**d.f.**	**SS**	**MS**	**F**	***p*** *
Habitat	2	2.5821	1.2911	10.566	**0.0002**
Grazing	1	0.20426	0.20426	1.6716	0.159
Interaction	2	0.13002	6.50E-02	0.53202	0.8306
Residual	54	6.5985	0.12219		
Total	59	9.5149			

Statistics for randomisations

		F from randomised groups				
Source	**F Obs.**	**Mean**	**Max**	**S.Dev.**	**No. > or = to observed F**	***p****
Habitat	10.56570	1.00785	6.11692	0.55652	0	**0.0002**
Grazing	1.67163	1.02225	6.0706	0.79049	794	0.1590
Interaction	0.53202	1.0143	5.21368	0.5463	4152	0.8306

*proportion of randomized trials with indicator value equal to or exceeding the observed indicator value. p = (1 + number of runs >= observed)/(1 + number of randomized runs)

4.4.3 Functional group responses: wing state and locomotion

Combining all habitats, wingless carabids were more abundant than winged species in both treatments (Figure 38), however the difference was not significant. Overall the abundance of wingless species showed a weak positive relationship (R^2 = 0.0011, p = 0.7175, n = 120) with Utilisation Score (Figure 39 and Table 42 below).

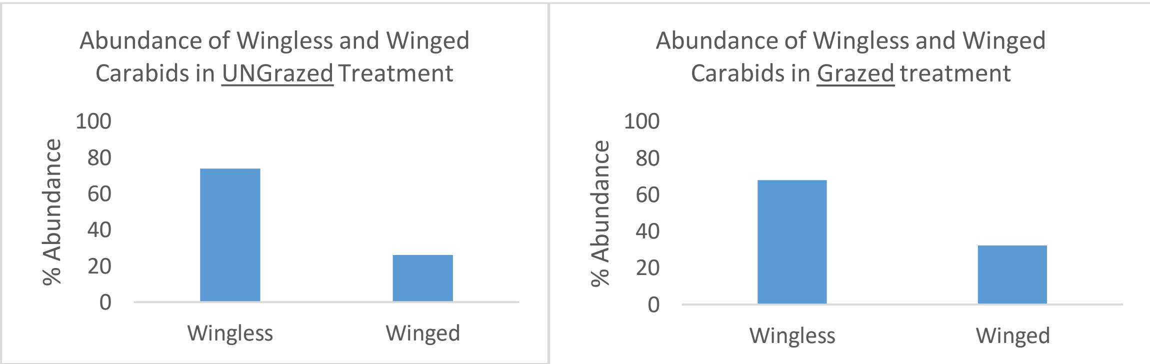


Figure 38 Relative abundance of Wingless and Winged carabids in MBNR (all years).

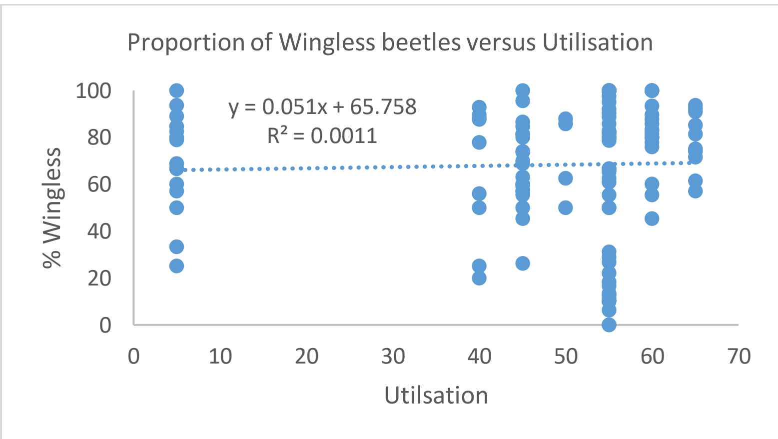


Figure 39 Proportion of Wingless beetles versus Utilisation Score (all habitats and years).

Table 42 Proportion and per cent frequency of wingless carabids in MBNR.

	All Years	
<u>All Habitats</u>	**Grazed**	**Ungrazed**
Mean % Wingless	68.2	62.2
% Frequency	98.3	84.4
Regression	R^2= 0.0011, p = 0.7175	
t-test	df=136, t = 1.3495, p = 0.1794	

Group 2 carabids (pushers) were relatively more abundant than group 1 (runners) in both treatments when all habitats were combined. Group 1 carabids had a weak negative relationship with Utilisation Score (R^2 = 0.0058, p = 0.4065, n = 120) with Utilisation Score (Figure 40 & Figure 41 and Table 43 below).

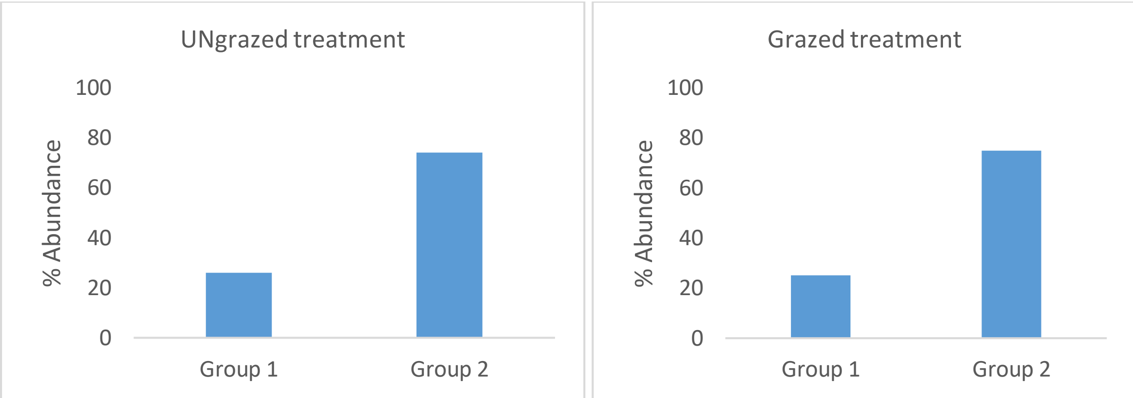


Figure 40 Abundance of Group 1 and Group 2 carabids in Ungrazed and Grazed treatments in MBNR.

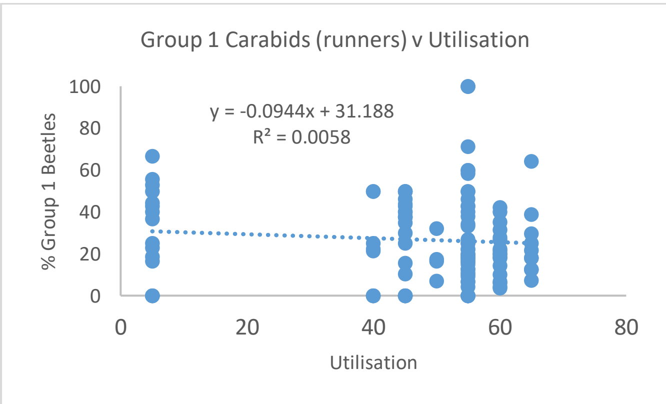


Figure 41 Abundance of Group 1 carabids (runners) versus Utilisation. All habitats, all years.

Table 43 Proportion and per cent frequency of Group 1 carabids (runners) in MBNR.

	All Years	
All Habitats	**Grazed**	**Ungrazed**
Mean % Group 1	26.7	29.3
% Frequency	58.8	75
Regression	R^2= 0.0058, p = 0.4065	
t-test	df=141, t = -1.236, p = 0.2184	

Wet Heath (wing state)

In 2013 a higher proportion of wingless species was recorded in ungrazed sites compared to grazed sites (68.5% versus 67%), however the difference was not significant. Wingless species occurred with the same frequency in both treatments. There was a weak negative relationship between the proportion of wingless species and Utilisation. In 2015, a higher proportion of wingless species was recorded in ungrazed samples compared to grazed sites (64.9% versus 42.6%), however the difference was not significant. Wingless beetles were also more frequent in ungrazed samples (86.7%) compared to grazed (66.7%) and a stronger negative relationship was recorded, yet not significant (Table 44 and figures figure 42).

Table 44 Proportion and frequencies of wingless beetles in grazed and ungrazed wet heath in 2013 and 2015.

	2013		2015	
Wet Heath	**Grazed**	**Ungrazed**	**Grazed**	**Ungrazed**
Mean % Wingless	67.0	68.5	42.6	64.9
% Frequency	100	100	66.7	86.7
Regression	$R^2 = 0.003, p = 0.793$		$R^2 = 0.025, p = 0.573$	
t-test (two-tailed)	df=36, $t = -0.258, p = 0.798$		df=28, $t = -1.648, p = 0.110$	

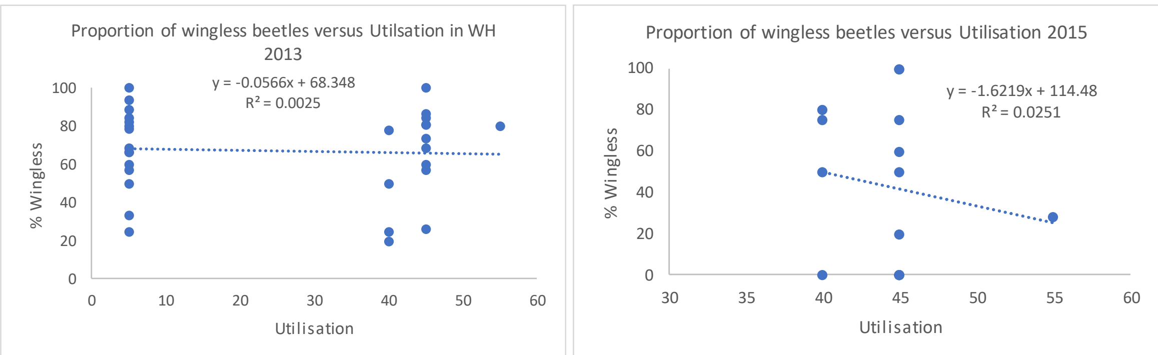


Figure 42 Proportion of wingless beetles versus Utilisation Score in Wet Heath.

Wet Heath (locomotion)

In wet heath, Group 1 carabids (runners) were relatively less abundant and less frequent than Group 2 carabids (pushers) in the grazed plots compared to ungrazed plots (Table 45). Group 1 carabids showed a negative relationship with Utilisation Score in both years (Figure 43), although the relationship was not significant.

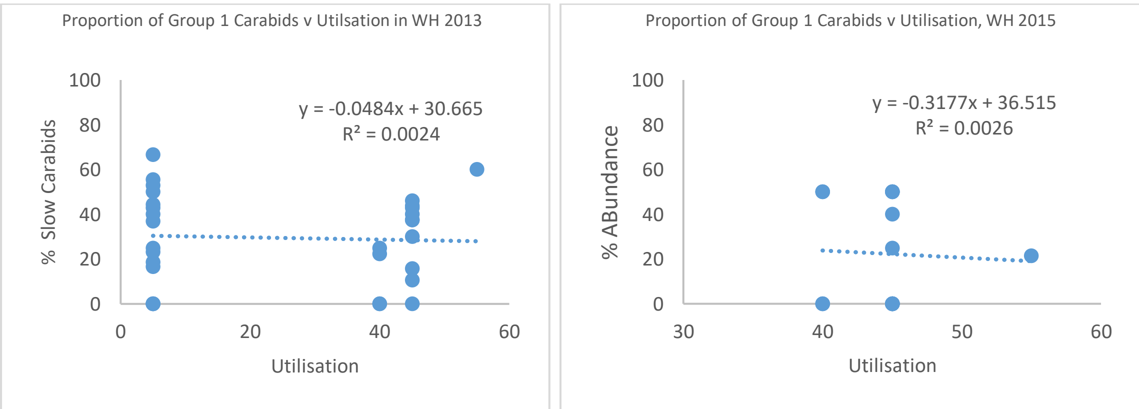


Figure 43 Proportion of Group 1 carabids versus Utilisation Score in Wet Heath in 2013 and 2015.

Table 45 Proportions and frequencies of Group 1 beetles in grazed and ungrazed wet heath.

	2013		2015	
Wet Heath	**Grazed**	**Ungrazed**	**Grazed**	**Ungrazed**
Mean % Group 1	29.5	39.9	22.4	38.6
% Frequency	80	93	53.3	73.3
Regression	$R^2 = 0.00$, $p = 0.798$		$R^2 = 0.025$, $p = 0.858$	
t-test (two-tailed)	df=25, $t = -1.509$, $p = 0.144$		df=26, t=-0.1620, p=0.117	

Blanket Bog (wing-state)

In blanket bog a higher proportion of wingless species was recorded in ungrazed sites compared to grazed sites in 2013 (38.5% versus 18.4%), however absolute numbers were low in ungrazed samples overall (47 beetles, compared with 231 in grazed samples). Of the ungrazed samples, *Abax parallelepipedus* was the most abundant and frequent species. Wingless species occurred with a higher frequency in grazed sites (92.9%) compared to ungrazed (60%). Wingless beetles made up a higher proportion of samples and were more frequent in grazed plots in 2015 compared to ungrazed plots (Table 46, Figure 44). A weak negative relationship was found between Utilisation Score and the proportion of wingless species in 2013, however a weak positive relationship was observed in 2015.

Table 46 Proportion and frequencies of wingless beetles in grazed and ungrazed blanket bog in 2013 and 2015.

	2013		2015	
Blanket Bog	**Grazed**	**Ungrazed**	**Grazed**	**Ungrazed**
Mean % Wingless	18.4	38.5	79.3	62.5
% Frequency	92.9	60.0	93.3	80.0
Regression	$R^2 = 0.005$, $p = 0.072$		$R^2 = 0.003$, $p = 0.845$	
t-test (two-tailed)	df=56, $t = 0.107$, $p = 0.915$		df=17, $t = -1.8832$, $p = 0.0769$	

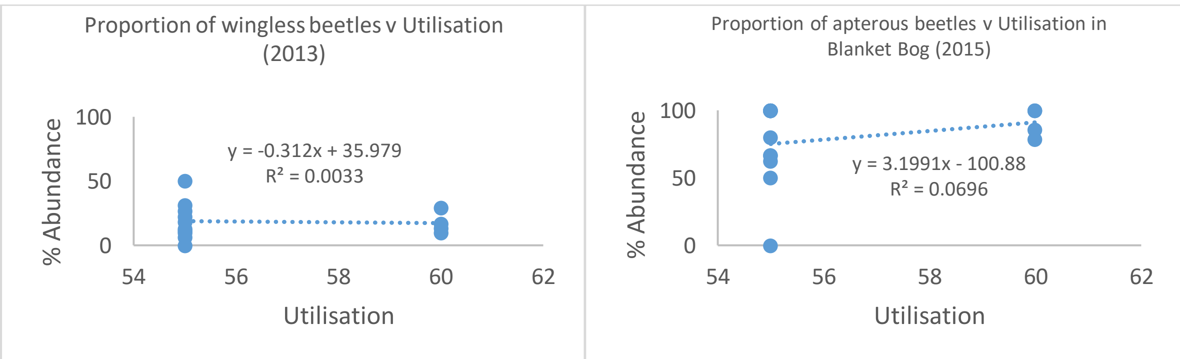


Figure 44 Proportion of wingless beetles versus Utilisation Score in blanket bog 2013 and 2015.

Blanket Bog (locomotion)

In blanket bog, Group 1 carabids (runners) were relatively more abundant and more frequent than Group 2 carabids (pushers) in the grazed plots compared to ungrazed plots Table 47. Group 1 carabids showed a negative relationship with Utilisation in 2013 and a positive relationship with Utilisation in 2015 (Figure 45). Similar relationships were found between utilisation and locomotion, as between utilisation and wing state. A negative relationship was observed in 2013, yet a positive relationship was observed in 2015.

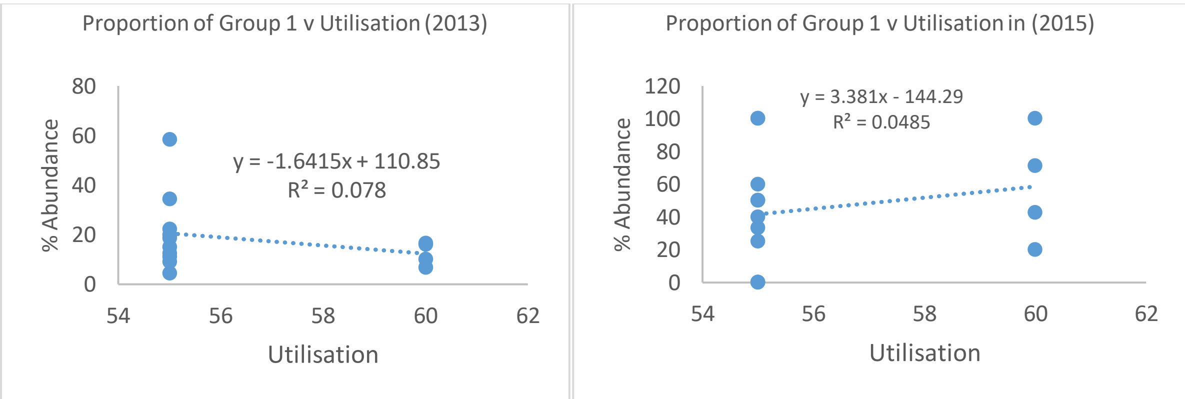


Figure 45 Proportion of Group 1 carabids versus Utilisation in Blanket bog, 2013 and 2015.

Table 47 Proportion and per cent frequency of Group 1 carabids in grazed and ungrazed blanket bog in 2013 and 2015.

	2013		2015	
Blanket Bog	**Grazed**	**Ungrazed**	**Grazed**	**Ungrazed**
Mean % Group 1	18.2	15.9	46.2	37.2
% Frequency	100	33.3	80.0	60.0
Regression	$R^2 = 0.0780$, $p = 0.333$		$R^2 = 0.0485$, $p = 0.4304$	
t-test (two-tailed)	df=21, $t = 0.2833$, $p = 0.7797$		df=28, t=0.6634, p=0.5125	

Dry Heath (wing-state)

In dry heath in 2013 wingless beetles were proportionately more abundant in grazed samples compared to ungrazed (83.7% versus 75%) and occurred with equal frequency in the two treatments. Differences were not significant when tested. In 2015 wingless beetles were again equally frequent in grazed and ungrazed plots, occurring in 100% of samples. Wingless beetles were proportionately more abundant in ungrazed plots (76.6%) compared to grazed plots (67.7%). Results show a weak positive relationship between Utilisation and the proportion of wingless beetles in dry heath although results were not significant (Table 48 and Figure 46).

Table 48 Proportion and frequencies of wingless beetles in grazed and ungrazed dry heath in 2013 and 2015.

	2013		2015	
Dry Heath	**Grazed**	**Ungrazed**	**Grazed**	**Ungrazed**
Mean % Wingless	83.7	75.0	67.7	76.6
% Frequency	100	100	100	100
Regression	$R^2 = 0.004$, $p = 0.757$		$R^2 = 0.024$, $p = 0.583$	
t-test (two-tailed)	df=31, $t = 2.7684$, $p = 0.0094$		df=28, $t = -1.473$, $p = 0.1519$	

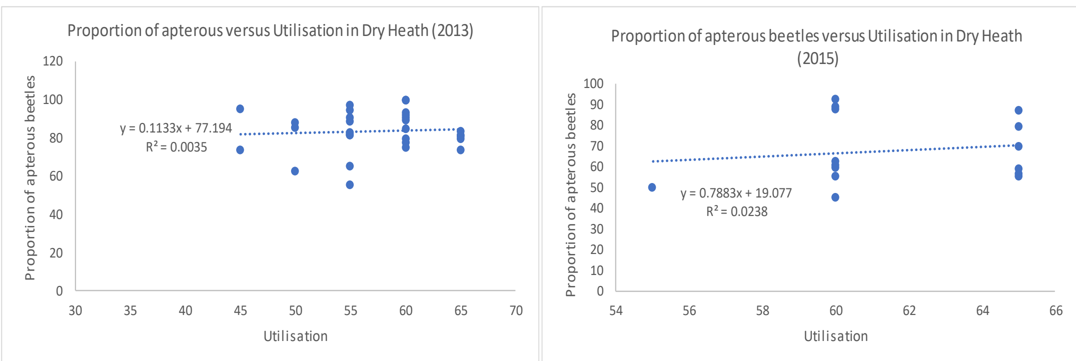


Figure 46 Proportion of wingless beetles versus Utilisation in dry heath 2013 and 2015.

Dry Heath (locomotion):

In dry heath, group 1 carabids (runners) were slightly more abundant and frequent than Group 2 carabids in grazed plots in 2013 compared to plots that were not grazed. However, the opposite was the case in 2015, when Group 1 carabids were more abundant and frequent in ungrazed plots compared to grazed. A weak positive correlation was found between Utilisation and abundance of Group 1 carabids in 2013, however a moderate negative correlation was found in 2015. Figure 47 and table 49 show the results

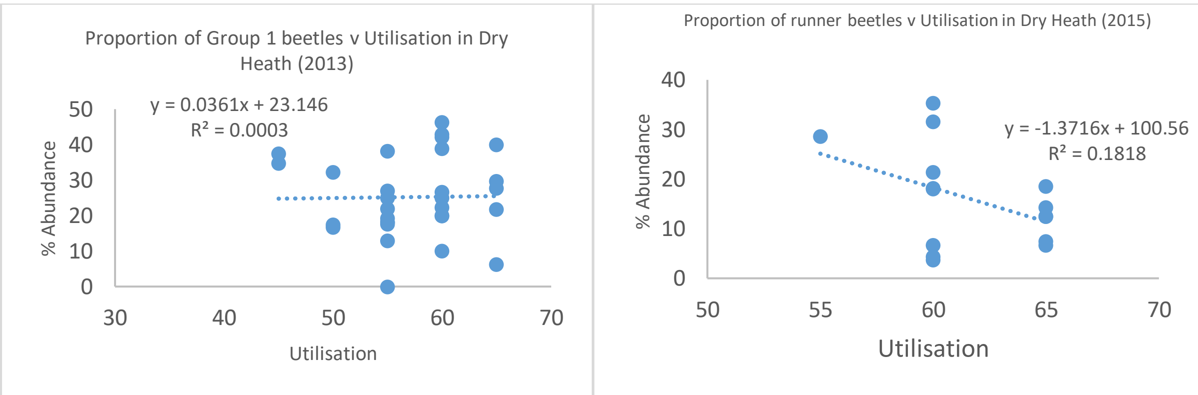


Figure 47 Proportion of Group 1 beetles versus Utilisation in dry heath, 2013 and 2015.

Table 49 Abundance and frequency of Group 1 carabids in Dry Heath, 2013 and 2015

	2013		2015	
Dry Heath	**Grazed**	**Ungrazed**	**Grazed**	**Ungrazed**
Mean % Group 1	25.5	20.3	16.0	21.5
% Frequency	96.7	93.3	50.3	93.3
Regression	$R^2 = 0.003$, $p = 0.927$		$R^2 = 0.1818$, $p = 0.1130$	
t-test (two-tailed)	df = 31, $t = 1.470$, $p = 1518$		df = 27, $t = -1.4122$, $p = 0.1693$	

4.4.4 Functional group responses: body size and wing form

Table 50, 51, 52 present the raw data and calculated proportions of carabids in relation to wing form and size. Table 50 shows carabid abundance with the proportions per habitat and treatment (ungrazed and grazed). Table 51 and 52 show the total abundance and proportion for all habitat types combined. The brachypterous group includes all the *Carabus* species (*C. granulatus, C. problematicus, C. clatratus*), plus *Abax parallelepipedus* and *Cychrus caraboides*. The Carabitae group is the same but without *A. parallelepipedus*.

Table 50 shows that the proportions of the Carabitae group is consistently smaller in grazed habitats. The patterns are very similar when looking at wing form, with the large wingless species being less abundant in grazed areas. Dry heath is the exception; with *A. parallelepipedus* included there is a greater proportion of large carabids in the grazed area. Table 53 shows that when all habitats are combined, the proportion of large beetles is greater in grazed habitats. The contrast between treatments is more obvious regarding small, macropterous beetles. Seventy percent of the macropterous group was captured in grazed areas.

Table 50 Carabid beetles from MBNR 2013 and 2015, showing three habitats (G = grazed plots, UG = Ungrazed). Proportions of brachypterous and Carabitae are given.

	Wet Heath				Dry Heath				Blanket Bog				
	2013		2015		2013		2015		2013		2015		
	UG	G	UG	G	UG	G	UG	G	UG	G	UG	G	Tot.
Abax parallelepipedus	31	75	17	16	257	280	131	168	15	12	11	25	1038
Pterostichus rhaeticus	13	27	20	27	0	1	1	0	15	145	24	168	441
Carabus problematicus	23	41	22	11	91	92	32	34	2	0	9	29	386
Pterostichus melanarius	5	1	2	1	84	43	40	85	1	0	1	7	270
Pterostichus niger	3	24	2	0	26	32	6	21	5	8	6	3	136
Carabus granulatus	1	8	1	5	0	7	15	14	4	3	3	3	64
Cychrus caraboides	1	3	3	0	6	2	5	2	3	1	3	0	29
Pterostichus diligens	1	2	0	7	1	0	2	0	1	8	2	3	27
Nebria brevicollis	1	7	0	0	0	0	0	0	0	10	0	1	19
Pterostichus strenuus	0	0	1	1	0	0	0	0	0	0	4	3	9
Harpalus latus	0	0	0	0	3	1	1	0	0	0	0	0	5
Leistus terminatus	0	1	0	0	0	3	0	0	0	0	0	0	4
Pterostichus madidus	0	2	0	0	0	0	1	0	0	0	0	0	3
Carabus clatratus	1	0	0	0	0	0	0	0	1	0	0	0	2
Olisthopus rotundatus	0	0	0	0	0	0	0	0	0	1	0	0	1
Agonum fuliginosum	0	0	0	0	0	0	0	0	0	0	0	1	1
Total Abundance	80	191	68	68	468	461	234	324	47	188	63	243	2435
Brachypterous abun.	56	127	43	32	354	381	183	218	24	16	26	57	
Brachypterous prop.*	**0.70**	**0.66**	**0.63**	**0.47**	**0.6**	**0.83**	**0.78**	**0.67**	**0.51**	**0.09**	**0.41**	**0.23**	
Carabitae (size) abun.	26	52	26	16	97	101	52	50	10	4	15	32	
Carabitae prop.**	**0.464**	**0.409**	**0.605**	**0.500**	**0.274**	**0.265**	**0.284**	**0.229**	**0.417**	**0.250**	**0.577**	**0.561**	
Macropterous	24	64	25	36	114	80	51	106	23	172	37	186	
Macropterous prop.***	**0.3**	**0.34**	**0.37**	**0.53**	**0.24**	**0.17**	**0.22**	**0.33**	**0.49**	**0.91**	**0.59**	**0.77**	

*Proportion of brachypterous beetles out of total. **The proportions of the Carabitae in the above table are given in relation to the Brachypterous group. *** Proportion of macropterous out of total abundance.

Table 51 Total abundance of brachypterous, Carabitae and Macropterous carabids in ungrazed and grazed treatments (all habitats)

Wing Form	Ungrazed	Grazed
Brachypterous	686 (1517)	831 (1517)
	0.45	0.55
Carabitae	226 (481)	255 (481)
	0.47	0.53
Macropterous	274 (918)	644 (918)
	0.30	0.70

Table 52 Total abundance and proportions of carabids by size (Carabitae) in ungrazed and ungrazed treatments (all habitats).

Size	Ungrazed	Grazed
Carabitae	226 (481)	255 (481)
	0.47	0.53
Smalls	734 (1954)	1220 (1954)
	0.38	0.62

4.5 Discussion

4.5.1 Overview

Carabid assemblages host species characteristic of habitat types and can reflect variation in natural conditions, offering a view on the structure of the environment and acting as indicators of change (Koivula, 2011). The intensity of agricultural management can influence carabid beetle assemblage in terms of species traits regarding size and dispersal ability (Eyre, McMillan and Critchley, 2016).

Investigation into the ground beetles are generally restricted to time-limited sampling regimes, due to constraints of resources (Eyre, McMillan and Critchley, 2016). Conclusions drawn from time-series snapshots between two sampling points need to be considered with caution as they rarely reflect local abundances through time. Insect populations fluctuate widely. Populations or interacting groups of carabids fluctuate in numbers in space and time (Kotze et al., 2011). Even with baselines, care is needed when assessing temporal variation, as considerable changes occur in the abundances of species collected between successive years (Didham et al., 2020). Nonetheless the results presented here, and the tentative conclusions drawn on the grazing treatment, will serve as a useful indication and a baseline for future studies.

4.5.2 Community assemblage and species response

In this study carabid communities varied between habitat types and between grazing treatments within habitat type, though not significantly. Species richness differed little between habitats, with wet heath having the largest number of species, followed by blanket bog and dry heath. There were only small variations in species richness across habitats and treatments over the period of the study. Abundance was highest in dry heath, dominated by three large carabids: *C. problematicus, P. melanarius* and *A. parallelepipedus*. This was followed by blanket bog, dominated by the small, winged *P. rhaeticus.*

In terms of species present, the findings of the study were broadly comparable with those of McFerran et al., (1994), where 18 species of carabid were collected in 1989 in Northern Ireland. *Carabus glabratus, Loricera pilicornis, Carabus nitens, Carabus arvensis, Leistus rufescens, Nebria salina, Patrobus assimilis, Notiophilus germinyi* and *Pterstichus minor* were recorded by McFerran et al., (1994), but were absent from the present study. By contrast, *Nebria brevicollis, Harpalus latus, Pterostichus madidus, Leistus terminatus, Carabus clatratus and Olisthopus rotundatus* were recorded in this study but not by McFerran et al. (1994).

Anderson (2013) recorded 42 species of carabid in south west Kerry. However, in that study, sampling was conducted over a range of geographic areas, farms, habitat types and management types (mainly sheep grazed at various stocking rates), and altitudes a broader altitude range (400 m – 800 m).

In this study, community assemblage is likely explained by the habitats present, the structure and composition of the plant communities, and the grazing treatment. However, the remote location and near pristine character of the study site must be considered. There is no improved grassland adjoining the site and it is bounded by wide expanses of blanket bog and sea cliffs. The nearest improved grassland is 2.8 km from the eastern perimeter of the site. Although the lowest elevation in the reserve is

18m asl, the sampling sites were all within a relatively narrow altitudinal range (approx. 250m to 400 m asl). The habitats of the reserve are also in Good Conservation Status (Chapter 3) and have received little management intervention in recent decades. Given the remoteness and near-pristine condition of the site, a carabid assemblage dominated by large, poorly dispersing, wingless species was to be expected.

The differences in species richness between habitats in this study are as a result of a few species that were recorded in low abundance e.g. *Carabus clatratus* occurred only once in blanket bog and once in dry heath. *Harpalus latus* was only recorded in dry heath (five individuals) and four individuals of *Leistus terminatus* were trapped, three in dry heath and one in wet heath. Singles of *Olisthopus rotundatus* and *Agonum fuliginosum* were recorded in blanket bog.

4.5.3 Functional group responses

Wing state and locomotion

Large apterous/ brachypterous species can be negatively impacted by intensive management, including livestock grazing (Dennis et al., 1997; Ribera et al., 2001; Cole et al., 2002, 2005). The results of this study show that dry heaths were dominated by the large carabid species *C. problematicus*, *P. melanarius* and *A. parallelepipedus*. *Carabus* species depend on soil crevices for refugia to avoid predation and desiccation (Dennis, 2003). Studies have shown that carabids in this genus can be negatively affected by intensive management, especially cattle grazing (Butterfield, Luff and Baines, 1995). *C. problematicus* made up 24% of total abundance in wet heath and 17%

in dry heath in this study. In dry heath there was no difference in the abundance of this species between grazed and ungrazed samples, indicating that the grazing intensity applied here is likely to be appropriate for the continued persistence of these communities (limitations of the study notwithstanding).

In wet heath, the relative abundance of *C. problematicus* increased in ungrazed samples between 2013 and 2015 (28% to 32%) and decreased in grazed plots from 22% to 16%. It is difficult to draw firm conclusions here. It is possible that the changes represent dispersal of beetles to more favourable areas, rather than genuine declines. The size and mosaic nature of the study site means that there is a lot of available habitat for beetles to disperse to. In other settings with less habitat availability this could be a concern. Given that insect populations fluctuate widely under normal circumstances (Didham et al., 2020) and in the context of concerning global declines, further replicates are required before drawing conclusions from these trends.

In blanket bog, *C. problematicus* was more abundant in grazed plots than in ungrazed plots, which was unexpected. However, this is likely to reflect significant differences in habitat structure and the location of sampling site, rather than being an effect of cattle utilisation.

Abax parallelepipedus is a flightless carabid that is often reported to be a forest specialist in other regions (e.g. Petit and Burel, (1998); Jopp and Reuter (2005)) and it has been suggested that forest carabid species have adapted to living in dwarf shrub communities in Britain (Anderson, McFerran & Cameron, cited in Lyons et al., 2017). Ireland is one of the least forested countries in Europe (Cross, 2012), with total cover of approximately 11% (DAFM, 2017a) and only 2% of that as native woodland (Perrin et al., 2008), the remainder having its origins only in the 20th century. In this study, *A. parallelepipedus* accounted for 40% of total beetle abundance, 56% of abundance in dry heath and 35% of abundance in wet heath, indicating the importance of heath habitats for this species in Ireland, at least.

A. parallelepipedus is a poorly dispersing species (Lövei and Sunderland, 1996) and, as a flightless species (Marcus et al., 2015), can also be impacted by cattle grazing. In the present study, *A. parallelepipedus* increased in ungrazed dry heath between years (52% to 56%), whereas a decrease in relative abundance was found in grazed samples (61% to 52%). In wet heath relative abundance also decreased in grazed samples from 40% to 24%. However, similar decreases were also observed in ungrazed samples (39% to 25%), which cannot be explained by cattle utilisation. Anderson (2013) found that *A. parallelepipedus* was more common under 'low' intensity management in sheep farms on the Iveragh peninsula; however, the habitats covered in her study included significant amounts of acid grassland, and also experienced a higher level of grazing and disturbance (0.2 - 0.8 LU/ha) than that which occurred in the present study; thus Anderson's 'low' grazing level may be comparable with the 'grazed' areas sampled here. Background fluctuations in populations, differences in weather from one sampling time point to another, changes in microclimate, or potentially the impact of the first trapping season are other possible explanations.

P. melanarius tends to occur in areas of more intensive grazing (Dennis et al., 2004; Vanbergen et al., 2005; Anderson, 2013). In the present study *P. melanarius* was found almost exclusively in dry heath (182 were caught in dry heath, compared to 9 individuals in both blanket bog and wet heath). The relative abundance of the species did not change in ungrazed samples (17% in both 2013 and 2015) but it did increase from 7% to 26% in grazed areas, consistent with the findings of Dennis et al., 2004; Vanbergen et al., 2005; Anderson, 2013). *Nebria brevicollis* is also a species which occurs more frequently in heavily grazed areas and in this study, it was most abundant in grazed blanket bog and wet heath (although numbers were low overall at 19 individuals). Thus, as expected, small and mobile carabids dominated in more disturbed areas.

Pterostichus rhaeticus is a hygrophilic species indicative of blanket bogs (Williams and Gormally, 2010), and is a species more typical of upland areas (Anderson, 2013). So, it was unsurprising that it made up 63% of total beetle abundance in blanket bog in this study. It seemed to favour the grazed blanket bog site, with 70% of *P. rhaeticus* caught from the grazed plot. Furthermore, it increased from 68% to 71% in the grazed treatment. Williams and Gormally (2010) found it to be a significant indicator of disturbed bog sites, so the high abundance in disturbed bog in this study is not surprising. Exploring the results species by species reveals similar patterns as the trends for the overall groups.

Body size and wing form

Due to the varying dispersal abilities of carabids, undisturbed habitats tend to be characterised by large wingless species (e.g. *Carabus* sp.), whereas macropterous/dimorphic species tend to be associated with fragmented or unstable habitats (Kromp, 1999; Ribera et al., 2001; Jelaska and Durbešić, 2009; Langraf et al., 2017, 2019). In this study, differences in the proportions of wingless beetles between grazed and ungrazed treatments were not significant. This was not surprising given the extensive grazing regime in place, the short timescale of the study (in terms of habitat change), and natural temporal fluctuations in the abundance of carabid species. The study site is of high conservation importance nationally and the Annex I habitats contained within in it are in good conservation status. Therefore, a cautious approach was taken to the application of a grazing regime. The observed differences in this study must therefore be considered preliminary and form a basis for future research on the study site to investigate cumulative impact.

In wet heath higher proportions of wingless carabids were found in ungrazed samples, yet results were not significant and patterns were not repeated in dry heath or blanket bog. In blanket bog wingless beetles were more abundant and frequent on the more disturbed (utilised) site. This may have been unexpected; however, it probably reflects differences in the character of habitats at each site (as discussed further below) and may ultimately be a consequence of randomly chosen sampling sites. Although both (grazed and ungrazed) blanket bog sites are classified as active blanket bog ('PB1' per Fossitt (2000) and Habitats Directive code *4130), the sites are quite different in character and composition, as described in section 3.5.2.

Observed differences in the carabid communities of the grazed and ungrazed blanket bog probably reflect differences in the character and composition of the habitat, rather than being an effect of cattle use. Invertebrates are influenced by microclimate, so fine-scale habitat heterogeneity is important, with heterogeneity of vegetation being particularly important (Cole et al., 2010). The requirements of many invertebrate species vary with season and circadian activity patterns, and as a consequence many species favour heterogeneous habitats (Cole et al., 2010). Care must be taken when interpreting the impact grazing regimes have on invertebrate communities because underlying differences between plots may also influence assemblage (Cole et al., 2010).

Shape and Wing Form

Morphological traits strongly influence how invertebrates interact with their environment. Changes in environmental conditions, e.g. as brought about by agricultural management, can influence the ease with which species traverse the habitat, the cover and camouflage that is available to them, and the availability and abundance of prey dependent on that managed environment. Large brachypterous carabid species have been shown to be negatively impacted by disturbance, whereas small macropterous species that have good dispersal abilities are favoured (Ng et al., 2018).

In this study macropterous beetles were more abundant in grazed treatments, regardless of habitat. The results were not as clear regarding large flightless species in general, although they generally favoured ungrazed areas in all habitats. In wet heath and blanket bog, the large brachypterous species were more abundant in ungrazed areas but that was not repeated in dry heath.

Interestingly, narrowing the group further to just Carabitae and standardising this group against brachypterous individuals, results in a consistent drop in the proportion of the Carabitae across all grazed sites, possibly demonstrating a sensitivity to the presence of cattle in all plots. There is ample evidence showing that Carabitae form a guild with features that differentiate them from other Carabid beetles, most notably their larger and deeper body form than compared to other flatter and light members. Their speed/force compromise indicates their slower gait and greater relative pushing strength in their natural environment (Evans and Forsythe, 1984). Whether or not such apparent sensitivity would be consistent across future studies is difficult to predict. Future sampling is recommended to explore this further.

In this study, trends suggest that large wingless poorly dispersing carabids were impacted by the grazing treatment. However, further sampling is required, as studies with just two time points are limited, given background fluctuations in populations, local variations and the context of broader insect declines.

Limitations

Sampling effort was reduced in 2015 due to more limited resources and to lessen the potential impact on local carabid populations. One grazed and one ungrazed sample from each habitat, repeated over 5 months was all that was practical given the nature of the site and the available resources.

A higher proportion of traps were lost in 2015 (22%) compared to 2013 (14%), due to flooding and interference by cattle. Data from Met Éireann's Valentia Observatory (36 km south of the study site) show that 681.9 mm of rainfall fell in the 2013 study period

(Met Eireann, 2018). In 2015 a weather station (Davis Instruments 6250 Vantage Vue) was set up on the study site. Data show that 886.6 mm of rain fell between May and September of 2015 on the study area, with June, July, and August all experiencing over 200 mm each. Pitfall traps were covered with fluted polypropylene covers to prevent rainwater getting in, however due to the nature of these habitats (blanket bog and wet heath in particular), many traps were flooded by surface water. The traps were not covered with wire mesh or caging, which would have reduced interference by cattle, however it was deemed that making 600 cages and transporting them into the remote sampling sites was not workable.

A lack of replication at site-level, of treatment type and at plot level was a limiting factor in this study, as it is with many studies in ecology, particularly in upland settings where sites are difficult to access. If possible, grazing regimes should be replicated across multiple sites to avoid any site-specific effects (Marriott et al., 2003). However, replication of grazing treatments is costly in studies where cattle are required to express natural behaviours in large sites (Lenoir and Lennartsson, 2010).

Treating individual pitfall traps as independent introduces pseudo-replication (*sensu* Hurlbert (1984). This is a common issue in ecology and prevalent in many studies using pitfall trapping for sampling invertebrates (Koivula, 2011). Fewer pitfall traps per plot, and more sites and treatment types could be considered for future work. Pitfall trapping, although widely used and efficient method for beetles, is dependent on a species' density and activity, thus not a measure of absolute density (Oxbrough et al., 2010). However, pitfall traps are easy to set-up, are repeatable, cost effective and have high catch rates.

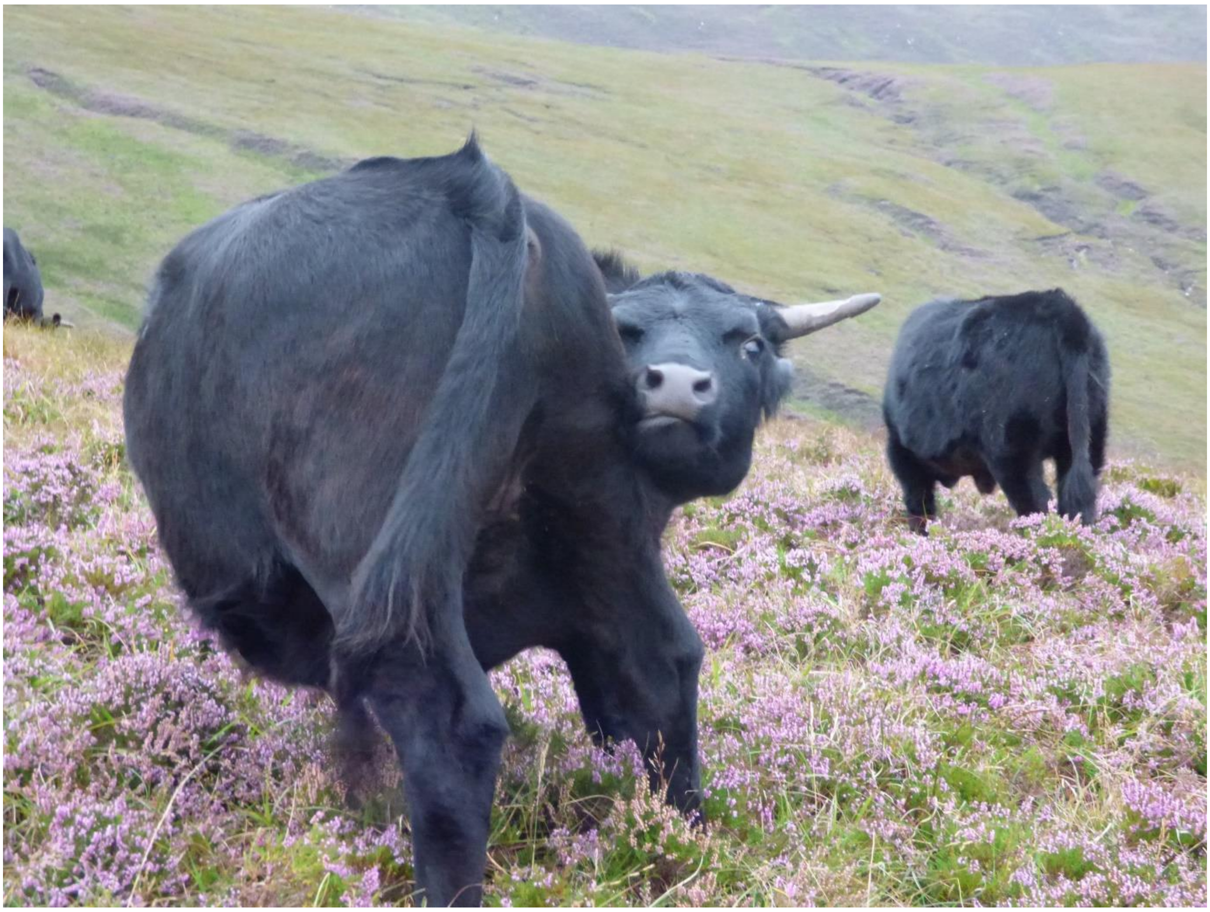
The results here should be considered baseline. Time series snapshots between two time points may not accurately reflect local abundance trends through time (Didham et al., 2020), so future repeated measures are strongly recommended.

4.6 Conclusions

Semi-natural habitats subjected to light-moderate grazing are believed to provide the greatest habitat heterogeneity (Fuller and Gough, 1999). They provide suitable conditions for multiple invertebrate needs through greater provision of resources (Cole et al., 2010). Stocking density and the inclusion of cattle in upland habitats (two cows with one suckler each, mixed with two ewes in 3.3 ha) has been shown to increase the numbers and biomass of foliar arthropods in Scottish upland grasslands (Dennis et al., 2008), with consequences for the conservation of arthropods and more broadly for birds of conservation concern.

Mobile arthropods show a strong association with grazing intensity and body size (Blake et al., 1994; Cole et al., 2005, 2010), and species richness and diversity are often maximal under moderate levels of grazing (Kaltsas et al., 2013), as predicted by the intermediate disturbance hypothesis (Connell, 1978). Large immobile species (e.g. *Carabus* spp.) are adversely affected by intensive grazing, and (typically smaller) mobile species are favoured (Dennis et al., 2004). Large carabids have been shown to occur in low abundances under intensive grazing (Dennis et al., 1997; Cole et al., 2006).

In the present study, large carabids appeared not to be adversely impacted under the current grazing regime. However, three years is a relatively short timescale to detect change. Summer grazing at the current stocking densities (presented in Chapter 2, section 2.3.4, p.123) appear appropriate for a range of carabid species. However, as grazing animals do not forage uniformly everywhere and habitat selection was shown to be significant in this study, care must be taken in applying broad stocking rates. Management plans for upland areas should therefore consider the home range and behaviour of grazing animals, as well as the availability and connectivity of preferred habitats.



5 Chapter 5: Final conclusions and Recommendations

Dexter heifer 0184, Mt Brandon Nature Reserve. Photo K.Kelly.

Biodiversity loss is a concerning global issue and anthropogenic causes of biodiversity declines and ecosystem collapse need to be addressed immediately (Ceballos, Ehrlich and Dirzo, 2017). Agricultural intensification has led to farmland biodiversity declines among many taxa (Benton, Vickery and Wilson, 2003), from direct habitat loss, habitat fragmentation and habitat conversion. The sobering declines of insect populations is a particularly pressing conservation concern (Dibner et al., 2009) and are very likely to cause cascade effects up the food chain.

The European landscape reflects centuries of interaction between people and the environment. Much of what is valued from a natural and cultural history perspective has been created and is maintained by agricultural systems (Bignal and McCracken, 2000). Half of the European landscape is under agricultural management and has been for 7-10,000 years. Agricultural intensification is a main driver of global biodiversity declines, yet in Europe farming has also long been recognised as being part of the solution (Batáry et al., 2015).

Concern over biodiversity declines in Europe led to the development of agri-environment schemes (Kleijn and Sutherland, 2003a), and a recognition that biodiversity conservation is dependent on maintenance of High Nature Value (HNV) farming systems (Beaufoy, 2008). Agricultural intensity in Europe ranges from high-input intensive farming on fertile land, to extensive low-input farming on marginal land that has high biodiversity value; that is, HNV farmland. HNV landscapes play a vital role in biodiversity conservation, clean water provision, carbon storage, climate regulation, and cultural and aesthetic landscapes (Sullivan et al., 2017). Recent EU

Habitat's Directive Article 17 reports from Ireland have revealed that the majority of upland habitats in Ireland are in Unfavourable conservation status, and with deteriorating trends (NPWS, 2013, 2019a).

Conservation grazing is the use of grazing animals for the maintenance and enhancement of biodiversity. Grazing is commonly used as a tool for maintenance of diversity within plant communities (Lyons et al., 2017), yet grazing management for biodiversity still lacks a strong evidence base (Wallis de vries et al., 2016).

Historically, cattle grazed the uplands of Ireland (O'Rourke et al., 2012) and Scotland (Dennis et al., 2015), but cattle numbers have declined since the 1970s due to European subsidies, resulting in a shift to sheep-dominated farming systems. Many protected habitats in the uplands have developed in tandem with farming and cessation of grazing practices could compromised their status (Costello, 2020). This research is the first in Ireland to examine the impact of cattle from a conservation grazing perspective, focusing on the maintenance and enhancement of Annex I upland habitats.

The research explored home range and habitat selection behaviours of free-ranging cattle (chapter 2). It studied the impact of conservation grazing with cattle on upland habitats from a vegetation (chapter 3) and ground beetle (chapter 4) perspective. The approach was novel in this research. Studies often involve fenced plots of fixed size and stocking density. This study was unique in that cattle were given free access to large landscape with a mosaic of habitats; a situation which describes much designated upland and commonage.

Chapter 2 examined the home range and habitat preferences of Dexter cattle in Mt Brandon Nature Reserve. Data from GPS collars were used to establish the home range of the cattle. Orthophotos, ordnance maps and habitat data were used to explore how they utilised space. Mean home range size was 122.7 ha and habitat selection by the cattle was statistically significant. The animals showed most preference for grassland habitats and least preference for blanket bog. Cattle in this study showed a slight preference for dry heath over wet heath. Home range estimates were used to establish stocking densities for each habitat and later these were examined in relation to conservation outcomes (chapter 3). Stocking rates for were 0.17 LU.ha^{-1} for the whole study area, 0.12 LU.ha^{-1} on wet heath, 0.20 LU.ha^{-1} in dry heath, 0.14 LU.ha^{-1} on blanket bog and 0.42 LU.ha^{-1} on wet grassland.

In this study the cattle were only on-site for 3.5 months, which is half of the recommended minimum stocking period under ANC and commonage management in Ireland. Due consideration must be given to this in the context of prescribing grazing regimes for sensitive upland habitats.

Blanket adjustments to stocking rates for biodiversity are often too blunt (Mills et al., 2007). In this study, cattle did not use the site evenly, showing preference for select components in the landscape. This suggests that a variety of site-specific solutions may be required to encourage cattle to graze target patches or habitats within a free choice system (e.g. in a commonage). Flexible fencing solutions, mineral licks, supplementary feed, and provision of water are possibilities. In this way animals could be stimulated to utilise areas that are identified as targets for conservation or be attracted away from areas that should be avoided (sensitive habitats, rare species, damaged areas). This approach would be especially desirable if very sensitive habitat patches adjoin less sensitive but more palatable vegetation patches.

Activity budgets of the cattle were compared between an upland and lowland herd using direct observation studies. The study established cattle in the uplands spend significantly more time grazing than cattle in the lowlands. This illustrates that the animals must forage for longer to meet requirements, which may have consequences for the use of paths and tracks by cattle to access different grazing areas. As expected, the upland cattle spent more time walking than the lowland herd, possibly to access more favourable areas. The lowland herd spent more time standing, lying and (observably) interacting with each other than the upland herd.

Conservation objectives are sometimes conflicting. Measures for one taxon may not be compatible with or may be detrimental to others (Mills et al., 2007; Bonari et al., 2017). Changes to grazing patterns have been found to produce different results for vegetation and invertebrate diversity (Rook et al., 2007; Scimone et al., 2007). Increasing grazing intensity has been shown to have no impact on plants but negative impacts on invertebrates (Kruess and Tscharntke, 2002b). Therefore, one strategy for conservation prescriptions may not optimize all outcomes (Mills et al., 2007), and management plans should match desired outcomes to predicted grazer impacts.

Chapter 3 explored the impact of conservation grazing with cattle on upland habitats. When the project was initiated the application of the grazing regime was done with caution because the habitats of the reserve were in good conservation status and the site is pristine relative to many geographically similar upland sites. The aim was to maintain the good status of the habitats overall and to examine the response of the vegetation to an extensive grazing regime.

Current stocking densities in the reserve are maintaining the Annex I habitats in favourable conservation status. However, the cattle were only on the site for 3.5 months, which is half of the current minimum period. Application of grazing presciptions for similar upland areas must suitably account for this, by either halving the stocking rate or adjusting the grazing period.

Vegetation change can be slow for upland habitats under extensive grazing regimes, so a cautious approach and long-term monitoring is recommended here. Furthermore, this good conservation status is rare in the Irish uplands (NPWS, 2013, 2019c). Where sites have poor conservation status due to overgrazing, due care should be taken when in the implementation of grazing regimes.

Grazing regimes should focus on proportions of habitat patches within an area, rather than applying stocking rates based on area alone. In the present study, cattle did not use the habitats evenly, selecting preferred habitat patches, and hardly using other patches if at all. Active management of the animals on-site is necessary, both from a husbandry and animal care perspective but also if parts of a site are to be targeted or avoided for conservation purposes in order to achieve conservation aims.

A preliminary examination of ground beetle response to the grazing treatment was presented in **Chapter 4.** The results indicate that small mobile carabids were abundant in disturbed plots. Large, wingless and poorly dispersing carabids were less abundant in disturbed sites compared to undisturbed areas, in keeping with the literature. The abundance of beetles in the supertribe Carabitae appear to be consistently depressed in cattle grazed areas in all habitats between 2013 and 2015. However, as discussed, these findings should be treated with caution because populations fluctuate naturally, and there is a broader context of declining insect populations.

5.1 Practical considerations: the farmer's experience

If conservation grazing with traditional breeds of cattle is to be used for management of upland habitats it must be flexible and adaptable in its approach, particularly regarding stocking rates, animal selection, and timing. It should be both ecologically sustainable and economically viable. Farmers are willing to adjust practices to suit conservation grazing management practices, but arrangements and supports must ensure that they are a good fit to the farming system.

There are numerous considerations from the farmer's perspective, a few of which are captured here from discussions with the farmer, Mr Paddy Fenton. In the present study the relationship between the farmer, the research team, land managers (NPWS), the funding bodies, and the local people involved was excellent. Such relationships are not to be taken for granted. It requires good communication and relationship building skills to foster and maintain partnerships and these are essential for the continuance of farming and the long-term ecosystem research (LTER) opportunities.

The cattle were usually turned out on the first or second week in July and returned by the first week of October. The grazing agreement was June-October but cattle were not ready for the mountain until July. In the current system, calving is in March/April, so a 60-day period is required to ensure the cows are back in calf before going to the mountain (P. Fenton, *pers. comm.*, August 2020).

The cattle fared very well most seasons and were in good condition coming off the mountain. However, it was important to get them down by late September/early October. They begin to lose condition if left out any later due to excess metabolic load on cows with calves (P. Fenton, *pers. comm*., August 2020). The cattle were weighed before and after the grazing season in 2014 and gained an average of 0.5kg.day^{-1} (Dineen, 2016).

From an animal rearing perspective, it would be preferable to have bullocks over two years old grazing the mountain i.e. 'second summer' animals. This would mean finishing at 3 years. However, markets for beef are such that farmers currently aim to finish animals under 24 months, so it would be inefficient to keep animals longer (P. Fenton, *pers. comm*., 2020).

The mountain site is remote, which makes animal care very challenging and there are no facilities on-site (i.e. a working crush or a cattle-safe yard) (P. Fenton, *pers. comm.,* August 2020). Any issues that arose with the animals involved bringing them down to a neighbour's shed for checking and handling. Foot injuries, selling animals, and fitting/checking GPS collars were some of the jobs that arose. It is/was a 2.5km journey down a rough track to the nearest neighbour. The whole herd had to be brought down each time because they behaved as a tight unit and splitting them on the hill was taxing for both animals and people, and potentially dangerous on such difficult terrain. The return journey with in-calf cows with calves at foot had to be taken slowly due to the steep ground and warm weather. All going smoothly, these operations, which would be relatively easy jobs on a lowland farm, took almost a full day and at least five people. Long-term continuance of grazing trials at this site would benefit from the creation of proper handling facilities on site.

Observations from the farmer, the local grazier, and the research team suggest that the biting midges were strongly influencing the cattle movement. The ‘good grazing’ for the cattle was down in Arraglen, the deep river valley at the centre of the reserve. There is also a constant supply of water in the river at the bottom of the valley, yet the cattle rarely went down until late August or early September. This, very site-specific factor, was advantageous because the grassland in the valley floor was left alone until late in the season, ensuring forage availability (P. Fenton, *pers. comm*., 2020).

It is essential that future research and conservation grazing arrangements consider the above issues carefully, and constant communication with stakeholders is essential.

5.2 Conclusions

The research presented here investigated the home range and habitat selection by a traditional cattle breed in a free-ranging setting in county Kerry, south west Ireland. It explored the impact of the cattle grazing on EU protected Annex I habitats and carabid beetles in the state-owned Mt Brandon Nature Reserve.

The study provided an opportunity to explore the impact of free-ranging cattle in an upland landscape. It was the first to study and elucidate the home range behaviour and habitat selection of free-ranging cattle. It has shown that grazing management influences plant community composition and structure, and ground beetle assemblage.

The cattle in this study established well defined home ranges and displayed significant habitat preferences. Conservation grazing maintained and improved the conservation status of Annex I habitats. Stocking rates varied considerably between habitats, as did specific positive outcome indicators of habitat quality. The findings may serve to inform agri-environment and conservation grazing measures, but plans should be local and site-specific where possible. Animals do not use space evenly and it is important to consider the range, distribution and status of habitats within sites (e.g. commonages, reserves). Furthermore, management goals may differ between taxa, and positive outcomes for one (or more broadly at the habitat level), may not necessarily be advantageous for species or groups of conservation concern.

Equipment inspection, Mt Brandon Nature Reserve. Photo K. Kelly.

6 References

Aalen, F.H.A., 1997. The Making of the Irish Landscape: The Irish rural landscape: synthesis of habitat and history. In: F.H.. Aalen, K. Whelan and M. Stout, eds. *Atlas of the Irish Rural Landscape*. Cork University Press.pp.4–30.

Acs, S., Hanley, N., Dallimer, M., Gaston, K.J., Robertson, P., Wilson, P. and Armsworth, P.R., 2010. The effect of decoupling on marginal agricultural systems:

Implications for farm incomes, land use and upland ecology. *Land Use Policy*, 27(2), pp.550–563.

Adrados, C., Girard, I., Gendner, J.-P.P. and Janeau, G., 2002. Global Positioning System (GPS) location accuracy improvement due to selective availability removal. *Comptes Rendus - Biologies*, 325(2), pp.165–170.

Aebischer, N.J., Robertson, P.A. and Kenward, R.E., 1993. Compositional analysis of habitat use from animal radio-tracking data. *Ecology*, 74(5), pp.1313–1325.

Aerts, R., 1989. Aboveground biomass and nutrient dynamics of Calluna vulgaris and Molinia caerulea in a dry heathland. *Oikos*, pp.31–38.

Al-Mufti, M.M., Sydes, C.L., Furness, S.B., Grime, J.P. and Band, S.R., 1977. A

Quantitative Analysis of Shoot Phenology and Dominance in Herbaceous Vegetation. *The Journal of Ecology*, 65(3), p.759.

Alderson, G.L.H., 1981. Celtic Cattle: value in ancestral traits. *The Ark*, 8(5), pp.161–166.

Allden, W. and McDWhittaker, I., 1970. The determinants of herbage intake by grazing sheep: The interrelationship of factors influencing herbage intake and availability. *Australian Journal of Agricultural Research*, 21(5), p.755.

Altmann, J., 1974. Observational study of behaviour: sampling methods. *Behaviour*, 49(3/4), pp.227–267.

Altmann, J. and Muruthi, P., 1988. Differences in daily life between semi provisioned and wild-feeding baboons. *American Journal of Primatology*, 15, pp.213–221.

Anderson, R. McFerran, D. Cameron, A., 2000. *The ground beetles of Northern Ireland (Coleoptera, Carabidae). Atlases of Northern Ireland Flora and Fauna Volume 1.* Belfast: Ulster Museum Publications.

Anderson, D.M. and Kothmann, M.M., 1980. Relationship of Distance Traveled with Diet and Weather for Hereford Heifers. *Journal of Range Management*, 33(3), p.217.

Anderson, M.J., 2001. A new method for non-parametric multivariate analysis of variance. *Austral Ecology*, 26(1), pp.32–46.

Anderson, M.J., 2017. Permutational Multivariate Analysis of Variance (PERMANOVA). *Wiley StatsRef: Statistics Reference*.

Anderson, R.M., 2013. Biodiversity change in the Irish uplands - the effects of grazing management. PhD thesis. UCC.

Antonelli, A., Kissling, W.D., Flantua, S.G.A., Bermúdez, M.A., Mulch, A., Muellner-Riehl, A.N., Kreft, H., Linder, H.P., Badgley, C., Fjeldså, J., Fritz, S.A., Rahbek, C., Herman, F., Hooghiemstra, H. and Hoorn, C., 2018. Geological and climatic influences on mountain biodiversity. *Nature Geoscience*.

Arnold, G.W., 1984. Comparison of the time budgets and circadian patterns of maintenance activities in sheep, cattle and horses grouped together. *Applied Animal Behaviour Science*, 13(1–2), pp.19–30.

Arnold, G.W. and Dudzinski, M.L., 1978. Ethology of free-ranging domestic animals. *Amsterdam, Elsevier scientific*, pp.1–125.

Augustine, B.C., Crowley, P.H. and Cox, J.J., 2011. A mechanistic model of GPS collar location data: Implications for analysis and bias mitigation. *Ecological Modelling*, 222(19), pp.3616–3625.

Augustine, D.J. and Derner, J.D., 2013. *Assessing herbivore foraging behaviour with GPS collars in a semiarid grassland*. *Sensors (Switzerland)*.

Avgın, S.S. and Luff, M.L., 2010. Ground beetles (Coleoptera: Carabidae) as bioindicators of human impact. *Mun Entomol Zool*, 5(1), pp.209–215.

Backshall, J., Manley, J., Rebane, M. and English Nature (Agency), 2001. *The upland management handbook*. English Nature.

Bailey, D.W., Gross, J.E., Laca, E.A., Rittenhouse, L.R., Coughenour, M.B., Swift, D.M. and Sims, P.L., 1996. Mechanisms that result in large herbivore grazing distribution patterns. *Journal of Range Management*, 49(5), p.386.

Bakker, J., Esselink, P., Van Der Wal, R. and Dijkema, K.., 1997. Options for restoration and management of coastal salt marshes in Europe. In: K.M. Urbanska, N.R. Webb and

P.J. Edwards, eds. *Restoration ecology and sustainable development*. Cambridge: Cambridge University Press.p.397.

Bakker, J.P., 1985. The impact of grazing on plant communities, plant populations and soil conditions on salt marshes. *Vegetatio*, 62(1–3), pp.391–398.

Bakker, J.P., 1989. *Nature management by grazing and cutting*. Kluwer Academic Publishers.

Bakker, J.P., 1998. The impact of grazing on plant communities. In: M.F.W. Wallis-deVries, J.P. Bakker and S.E. van. Wieren, eds. *Grazing and Conservation Management*. Kluwer Academic Publishers.pp.137–184.

Bardgett, R.D., Marsden, J.H. and Howard, D.C., 1995. The extent and condition of heather on moorland in the uplands of England and Wales. *Biological Conservation*, 71(2), pp.155–161.

Barnett, V. and Lewis, T., 1974. Outliers in statistical data. Available at :<http://cds.cern.ch/record/283291>

Barnosky, C.W., 1988. A Late-Glacial and Post-Glacial Pollen Record from the Dingle Peninsula, County Kerry. In: *Proceedings of the Royal Irish Academy. Section B: Biological, Geological, and Chemical Science*. JSTOR.pp.23–37.

Barry, R.G., 2008. *Mountain Weather and Climate*. Third ed. New York: Cambridge University Press.

Basarab, J. and Gould, R., 2001. *Understanding The Animal Unit Month (AUM)*. The Cattle Site.

Batáry, P., Dicks, L. V., Kleijn, D. and Sutherland, W.J., 2015. The role of agri-environment schemes in conservation and environmental management. *Conservation Biology*, 29(4), pp.1006–1016.

Bates, D., Mächler, M., Bolker, B. and Walker, S., 2015. Fitting Linear Mixed-Effects Models Using lme4. *Journal of Statistical Software*, 67(1), pp.1–48.

Bauer, T., 1985. Beetles which use a setal trap to hunt springtails: the hunting strategy and apparatus of Leistus (Coleoptera, Carabidae). *Pedobiologia*, 28, pp.275–287.

Beaufoy, G., 2008. *HNV Farming - Explaining the concept and interpreting EU and National policy commitments*. European Forum for Nature Conservation and Pastoralism.

Beaver, J.M. and Olson, B.E., 1997. Winter range use by cattle of different ages in southwestern Montana. *Applied Animal Behaviour Science*.

Bekker, D.L. and Bakker, J.P., 1989. Het Westerholt IX: veranderingen in vegetatiesamenstelling en-patronen na 15 jaar beweiden. *De Levende Natuur*, 90, pp.114–119.

Benton, T.G., Vickery, J.A. and Wilson, J.D., 2003. Farmland biodiversity: Is habitat heterogeneity the key? *Trends in Ecology and Evolution*, 18(4), pp.182–188.

Bergeron, C.J.A., Spence, J.R., Jan, W., Pinzon, J. and Hartley, D.J., 2013. Effect of habitat type and pitfall trap installation on captures of epigaeic arthropod assemblages in the boreal forest. *Canadian Entomologist*, 145(5), pp.547–565.

Bevan, R., 2008. *Monitoring Cattle Movement with GPS Collars. in The Limestone Country Project: Monitoring for Ecological Change.* Yorkshire Dales National Park Authority.

Bevanda, M., Fronhofer, E.A., Heurich, M., Müller, J. and Reineking, B., 2015. Landscape configuration is a major determinant of home range size variation. *Ecosphere*, 6(10), pp.1–12.

Bhriain, B.N., Sheehy Skeffington, M. and Gormally, M., 2002. Conservation implications of land use practices on the plant and carabid beetle communities of two turloughs in Co. Galway, Ireland. *Biological Conservation*, 105(1), pp.81–92.

Bignal, E.M. and McCracken, D.I., 2000. The nature conservation value of European traditional farming systems. *Environmental Reviews*, 8(3), pp.149–171.

BirdWatch Ireland, 2010. *Action Plan for Upland Birds in Ireland 2011-2020. BirdWatch Ireland's Group Action Plans for Irish Birds.*

Blake, S., Foster, G., Eyre, M. and Luff, M., 1994. Effects of habitat type and grassland management practices on the body size distribution of carabid beetles. *Pedobiologia*, 38, pp.502–512.

Blanchet, F.G., Bergeron, J.A.C., Spence, J.R. and He, F., 2013. Landscape effects of disturbance, habitat heterogeneity and spatial autocorrelation for a ground beetle (Carabidae) assemblage in mature boreal forest. *Ecography*, 36(5)(5), pp.636–647.

Bleasdale, A., 1998. Overgrazing in the west of Ireland–assessing solutions. *Towards a conservation strategy for the bogs of Ireland*, pp.67–78.

Bleasdale, A., 2018. *Result Result-based approaches: based approaches: experiences and reflections experiences and reflections of the National Parks and Wildlife Service of the National Parks and Wildlife Service*.

Bleasdale, A. and O'Donoghue, B., 2015. NPWS perspectives on Locally-led Agri-environment Schemes. Available at: <https://www.agriculture.gov.ie/media/migration/farmingschemesandpayments/locallyledschemes/AndyBleasdalepres16oct15221015.pdf>.

Bleasdale, A. and Sheehy-Skeffington, M., 1995. The upland vegetation of north-east Connemara in relation to sheep grazing Ed. Jefferey, D, Jones, M. and McAdam, J. *Irish Grasslands, their biology and management*.

Bleasedale, A. and O'Donoghue, B., 2015. NPWS Perspectives on Locally-led agri-environment schemes. DAHG.

Bobbink, R. and Willems, J., 1987. Increasing dominance of *Brachypodium pinnatum* (L.) beauv. in chalk grasslands: A threat to a species-rich ecosystem. *Biological Conservation*.

Bokdam, J.A.N. and Gleichman, J.M., 2000. Effects of grazing by free-ranging cattle on vegetation dynamics in a continental north-west European heathland. *Journal of*

Applied Ecology, 37(3), pp.415–431. Bonari, G., Fajmon, K., Malenovský, I., Zelený, D., Holuša, J., Jongepierová, I., Kočárek, P., Konvička, O., Uřičář, J. and Chytrý, M., 2017. Management of semi-natural grasslands benefiting both plant and insect diversity: The importance of heterogeneity and tradition. *Agriculture, Ecosystems and Environment*, 246, pp.243–252.

Bourgoin, G., Garel, M., Van Moorter, B., Dubray, D., Maillard, D., Marty, E. and Gaillad, J.-M., 2008. Determinants of seasonal variation in activity patterns of mouflon. *Canadian Journal of Zoology*, 86(12), pp.1410–1418.

Bowerman, B.L. and O'Connell, R.T., 1990. *Linear statistical models: an applied approach*. Duxbury.

Braby, M.F., 2018. Threatened species conservation of invertebrates in Australia: an overview. *Austral Entomology*, 57(2), pp.173–181.

Braghieri, A., Pacelli, C., Girolami, A. and Napolitano, F., 2011. Time budget, social and ingestive behaviours expressed by native beef cows in Mediterranean conditions. *Livestock Science*, 141(1), pp.47–52.

Brandmayr, P., 2005. I coleotteri carabidi per la valutazione ambientale e la conservazione della biodiversità: manuale operativo. Apat.

Brigand, L. and Bioret, F., 1994. Réflexions sur l'influence du mouton dans les modes de gestion des milieux insulaires: Ouessant et Clare (Irlande). *Norois*, 164(1), pp.559–564.

Brooks, D.R., Bater, J.E., Clark, S.J., Monteith, D.T., Andrews, C., Corbett, S.J., Beaumont, D.A. and Chapman, J.W., 2012. Large carabid beetle declines in a United Kingdom monitoring network increases evidence for a widespread loss in insect biodiversity. *Journal of Applied Ecology*, 49(5), pp.1009–1019.

Browne, S.J. and Aebischer, N.J., 2003. Habitat use, foraging ecology and diet of Turtle Doves *Streptopelia turtur* in Britain. *Ibis*, 145(4), pp.572–582.

Bruce, M. and Grace, J.B., 2002. Analysis of Ecological Communities. *Journal of Experimental Marine Biology and Ecology*.

Brunsdon, C., McClatchey, J. and Unwin, D.J., 2001. Spatial variations in the average rainfall-altitude relationship in Great Britain: an approach using geographically weighted regression. *International Journal of Climatology*, 21(4), pp.455-466.

Bryant, L.D., 1982. Response of Livestock to Riparian Zone Exclusion. *Journal of Range Management*, 35(6), p.780.

Management methods and their effects on heathland vegetation. *Biological Conservation*, 79(1), pp.1–13.

Burton, R.J.F. and Schwarz, G., 2013. Result-oriented agri-environmental schemes in Europe and their potential for promoting behavioural change. *Land Use Policy*, 30(1), pp.628–641.

Butterfield, J., Luff, M. and Baines, M., 1995. Carabid beetle communities as indicators of conservation potential in upland forests. *Forest Ecology and Management*, 79, pp.63–77.

Byrne, D., 2018. *Developing scoring systems*. [Conference publication] Available at: <https://rbapseu.files.wordpress.com/2018/06/byrne-d-session-1-scoring-rbaps.pdf> [Accessed 15 Aug. 2018].

Cameron, K.H. and Leather, S.R., 2012. Heathland management effects on carabid beetle communities: The relationship between bare ground patch size and carabid biodiversity. *Journal of Insect Conservation*.

Carruthers, T., 1998. *Kerry: A Natural History*. Cork: Collins Press.

Cavanagh, J.A.L., Tammen, I., Windsor, P.A., Bateman, J.F., Savarirayan, R., Nicholas, F.W. and Raadsma, H.W., 2007. Bulldog dwarfism in Dexter cattle is caused by mutations in ACAN. *Mammalian Genome*, 18(11), pp.808–814.

Ceballos, G., Ehrlich, P.R. and Dirzo, R., 2017. Biological annihilation via the ongoing sixth mass extinction signalled by vertebrate population losses and declines.

Proceedings of the National Academy of Sciences of the United States of America, 114(30), pp.E6089–E6096.

Chapman, D.F., Parsons, A.J., Cosgrove, G.P., Barker, D.J., Marotti, D.M., Venning, K.J., Rutter, S.M., Hill, J. and Thompson, A.N., 2007. Impacts of spatial patterns in pasture on animal grazing behaviour, intake, and performance. *Crop Science*, (47), pp.399–415.

Chesterton, C., Condliffe, I. and Peel, S., 2006. Revised calculation of livestock units for higher level stewardship agreements. Technical Advice Note 33 (Second ed.), Rural Development 1-3.

Clark, D.L. and Wilson, M.V., 2003. Post-dispersal seed fates of four prairie species. *American Journal of Botany*, 90(5), pp.730–735.

Clarke, K.R., 1990. Comparisons of dominance curves. *Journal of Experimental Marine Biology and Ecology*, 138(1–2), pp.143–157.

Clutton-Brock, T.H. and Harvey, P.H., 1978. Mammals, resources and reproductive strategies. *Nature*, 273(5659), pp.191–195.

Cole, L.J., McCracken, D.I., Downie, I.S., Dennis, P., Foster, G.N., Waterhouse, T., Murphy, K.J., Griffin, A.L. and Kennedy, M.P., 2005. Comparing the effects of farming practices on ground beetle (Coleoptera: Carabidae) and spider (Araneae) assemblages of Scottish farmland. *Biodiversity and Conservation*, 14(2), pp.441–460.

Cole, L.J., Pollock, M.L., Robertson, D., Holland, J.P., McCracken Cole, D.I., Cole, L.J., Robertson, D., McCracken, D.I., Pollack, M.L., Holland, J.P. and H., 2006.

Carabid (Coleoptera) assemblages in the Scottish uplands: the influence of sheep grazing on ecological structure, 17(3), pp.229-240.

Cole, L.J.L., Pollock, M.L.M., Robertson, D., Holland, J.P., McCracken, D.I., Harrison, W., Agriculture, J.H.-, ecosystems, undefined and 2010, undefined, 2010. The influence of fine-scale habitat heterogeneity on invertebrate assemblage structure in upland semi-natural grassland. *Agriculture, ecosystems & environment*, 136(1), pp.69–80.

Cole, L.L.J., McCracken, D.I. DI, Dennis, P., Downie, I.S., Griffin, A.L., Foster, G.N., Murphy, K.J., Waterhouse, T., Agriculture, I.D.-, Ecosystems, U. and 2002, U., 2002. Relationships between agricultural management and ecological groups of ground beetles (Coleoptera: Carabidae) on Scottish farmland. *Agriculture, Systems & Environment*, 93(1), pp.323–336.

Colhoun, K. and Cummins, S., 2013. Birds of Conservation Concern in Ireland 2014 – 2019. *Irish Birds*, 9(1999), pp.523–544.

Compton, T.L. and Brundage, A.L., 1971. Cattle Behaviour on Subalpine Range in Southcentral Alaska. *Journal of Animal Science*, 32(2), p.339.

Connell, J.H., 1978. Diversity in Tropical Rain Forests and Coral Reefs. *Science*, 199(4335), pp.1302–1310.

Cook, R.D. and Weisberg, S., 1982. *Residuals and influence in regression*. Chapman and Hall.

Costello, E., 2020. Hill farmers, habitats and time: the potential of historical ecology in upland management and conservation. *Landscape Research*, 45(8), pp.951–965.

Council Regulation (EC) No 1099/2009 of 24 September 2009 on the protection of animals at the time of killing.

Crawford, R.M.M., 2008. *Plants at the margin: ecological limits and climate change*. Cambridge University Press.

Critchley, C.N.R., Adamson, H.F., McLean, B.M.L. and Davies, O.D., 2008. Vegetation dynamics and livestock performance in system-scale studies of sheep and cattle grazing on degraded upland wet heath. *Agriculture, Ecosystems and Environment*, 128(1–2), pp.59–67.

Cross, J.R., 2006. The potential natural vegetation of Ireland. Biology and Environment: Proceedings of the Royal Irish Academy. In *Biology and Environment: Proceedings of the Royal Irish Academy,* pp65-116. Royal Irish Academy.

Crowley, C., O'Keeffe, B. and O'Sullivan, S., 2016. *Irish Uplands Socio-Economic Profile: A report into Ireland's inhabited and farmed upland areas.* Report available at

http://irishuplandsforum.org/wp-content/uploads/2016/10/Irish-Uplands-Socio-Economic-Profile-2016-FINAL.pdf.

Cummins, S., Peadar, O.C. and Mee, A., 2012. Upland Heath and Blanket Bog. In: R. Richard Nairn and O.H. John, eds. *Bird Habitats in Ireland*. Cork: The Collins Press.

Curran, P.L., 1990. Kerry and Dexter Cattle and Other Ancient Irish breeds, A History. Royal Dublin Society.

DAFM, 2012. *AEOS 3: Commonage Grazing Plans (guidance doc.)*. Available at: <https://www.agriculture.gov.ie/farmerschemespayments/agri-environmentoptionsschemeaeos/>.

DAFM, 2015a. *A Guide to the completion of the GLAS Commonage Plan.* Available at file:///C:/Users/Killian/Downloads/96226_58003a72-7299-401e-a75f-20eb2a9e8e9.pdf.

DAFM, 2015b. *GLAS Structure*. Available at: <https://www.agriculture.gov.ie/media/migration/farmingschemesandpayments/glastranche1/GLASStructure240215.pdf>

DAFM, 2016. *GLAS Specification for GLAS Tranche 3*. Available at: <https://www.agriculture.gov.ie/media/migration/farmingschemesandpayments/glastranche3/GLAS3Specification031116.pdf>

DAFM, 2017a. *Forest Statistics - Ireland 2017*. Available at https://www.teagasc.ie/media/website/crops/forestry/advice/Forest-Statistics-Ireland-2019.pdf .

DAFM, 2017b. Nitrates Explanatory Handbook for Good Agricultural Practice for the Protection of Waters Regulations 2018.

DAFM, 2018a. Helpsheet/Terms and Conditions: Areas of Natural Constraints (ANC)

Scheme 2018.

DAFM, 2018b. *Ireland-Rural Development Programme*. Available at: <https://www.agriculture.gov.ie/media/migration/ruralenvironment/ruraldevelopment/ruraldevelopmentprogramme2014-2020/ApprovedFouthAmendment140818.pdf>

DAFM, 2019. *Helpsheet/Terms and Conditions. Areas of Natural Constraint (ANC) and Areas of Specific Constraint (Island Farming) Schemes 2019*.

Deane, P., 2018. *Discussion on visitor numbers to the Reeks based on unpublished data from people counters.* Personal communication, April 2018. McGillycuddy Reeks EIP.

Debano, S.J., 2006. Effects of livestock grazing on aboveground insect communities in semi-arid grasslands of south eastern Arizona. *Biodiversity and Conservation*, 15(8), pp.2547–2564.

Deng, L., Sweeney, S. and Shangguan, Z.-P., 2013. Grassland responses to grazing disturbance: plant diversity changes with grazing intensity in a desert steppe. *Grass and Forage Science*, 69(3), pp.524–533.

Dennis, P., 2003. Sensitivity of upland arthropod diversity to livestock grazing, vegetation structure and landform. *Journal of Food, Agriculture and Environment*, 1(2), pp.301–307.

Dennis, P., Doering, J., Stockan, J.A., Jones, J.R., Rees, M.E., Vale, J.E. and Sibbald, A.R., 2004. Consequences for biodiversity of reducing inputs to upland temperate pastures: effects on beetles (Coleoptera) of cessation of nitrogen fertilizer application and reductions in stocking rates of sheep. *Grass and Forage Science*,59(2), pp.121–135.

Dennis, P., Skartveit, J., Kunaver, A. and McCracken, D.I., 2015. The response of spider (Araneae) assemblages to structural heterogeneity and prey abundance in sub-

montane vegetation modified by conservation grazing. *Global Ecology and Conservation*, 3, pp.715–728.

Dennis, P., Skartveit, J., McCracken, D.I., Pakeman, R.J., Beaton, K., Kunaver, A. and Evans, D.M., 2008. The effects of livestock grazing on foliar arthropods associated with bird diet in upland grasslands of Scotland. *Journal of Applied Ecology*, 45(1), pp.279–287.

Dennis, P., Young, M.R. and Bentley, C., 2001. The effects of varied grazing management on epigeal spiders, harvestmen and pseudoscorpions of Nardus stricta grassland in upland Scotland. *Agriculture, Ecosystems and Environment*, 86(10), pp.39-57

Dennis, P., Young, M.R. and Gordon, I.J., 1998. Distribution and abundance of small insects and arachnids in relation to structural heterogeneity of grazed, indigenous grasslands. *Ecological Entomology*, 23(3), pp.253–264.

Dennis, P., Young, M.R.M., Howard, C.L.C.L., Gordon, I.J., Young, M.R.M. and Howard, C.L.C.L., 1997. The Response of Epigeal Beetles (Col.: Carabidae,

Staphylinidae) to Varied Grazing Regimes on Upland *Nardus stricta* Grasslands. *The Journal of Applied Ecology*, 34(2), p.433.

Dexter Cattle Society, 2014. *Dexter Cattle Society*. http://www.dextercattle.co.uk/

Didham, R.K., Basset, Y., Collins, C.M., Leather, S.R., Littlewood, N.A., Menz, M.H.M., Müller, J., Packer, L., Saunders, M.E., Schönrogge, K., Stewart, A.J.A., Yanoviak, S.P. and Hassall, C., 2020. Interpreting insect declines: seven challenges and a way forward. *Insect Conservation and Diversity*, 13(2), pp.103–114.

Diemont, W.H. and Heil, G.W., 1984. Some long-term observations on cyclical and seral processes in Dutch heathlands. *Biological Conservation*, 30(3), pp.283–290.

Dillon, E., Moran, B., Lennon, J. and Donnellan, T., 2017. *Teagasc National Farm Survey Preliminary Results 2017*. Available at: <https://www.teagasc.ie/media/website/publications/2018/NFS-Publication-2017.pdf>.

Dineen, N., 2016. *Prevalence and distribution of Gastrointestinal Nematoe Parasites and Dung Beetles Associated with Upland and Lowland Grazing of Irish Native Dexter Cattle*. Masters thesis, Institute of Technology, Tralee.

Dodson, J.R., 1990. The Holocene vegetation of a prehistorically inhabited valley, Dingle peninsula, Co. Kerry. In: *Proceedings of the Royal Irish Academy. Section B: Biological, Geological, and Chemical Science*.

Drew, D., 2018. *Karst of Ireland: Landscape Hydrogeology Methods*. Geological Survey of Ireland.

Dunford, B., 2015. Starting from scratch – the story of one locally- led programme: in Farmland Conservation with 2020 Vision. In: D. Ó hUallacháin and J. Finn, eds. Teagasc.pp.20–21.

Dunne, F., 2000. *Blanket Bog, Heath & Upland Grassland exclosures: Baseline Surveys & Monitoring Methodologies*. National Parks and Wildlife Service.

Durbin, J. and Watson, G.S., 1951. Testing for Serial Correlation in Least Squares Regression. II. *Biometrika*, 38(1/2), p.159.

Van Eck, H., 1984. *Irish bogs: a case for planning*. Catholic University.

Edwards, S., 2005. Managing the mountain ecosystems to meet human needs: a case for applying the ecosystem approach. In: D. Thompson, M. Price and C. Galbraith, eds. *Mountains of Northern Europe: Conservation, Management, People and Nature*.Edinburgh: Scottish Natural Heritage.

Eerden, Van, M., 1996. The response of Anatidae to changes in agricultural practice: long-term shifts in the carrying capacity of wintering waterfowl. *Gibier Faune Sauvage, Game Wildl.*, 13, pp.681–706.

EFNCP, 2017. *European Forum on Nature Conservation and Pastoralism*. Available at: <http://www.efncp.org/members/>.

Eireann, M., 2018. *Met Eireann*. Creative Commons Attribution 4.0 International (CC BY 4.0). https://creativecommons.org/licenses/by/4.0/.

Éireann, M., 2014. *Monthly Weather Data*. *Met Éireann*. Available at: <http://www.met.ie/climate/monthly-data.asp?Num=2275>.

Emerson, H.J. and Gillmor, D.A., 1999. The Rural Environment Protection Scheme of the Republic of Ireland. *Land Use Policy*, 16(4), pp235-245.

English Nature, 2005. *The importance of livestock grazing for wildlife conservation*. Available at: <http://publications.naturalengland.org.uk/publication/68026>.

Enright, R., 2012. Feral goats (*Capra hircus* L.) in the landscape: behaviour, home range and habitat use. PhD thesis. University College Cork.

Enright, R. and Williams, B., 2010. *Assessing the movements of Burren feral goats using GPS tracking.*

Erwin, T., Whitehead, D. and Ball, G., 1977. Family 4: Carabidae. The Ground Beetles. In: *Checklist of Beetles of Canada, United Sates, Mexioc, Central America and the West Indies*, Blackwelde. World Digest Publications. New York.

ESRI, 2016. ArcMap 10. *Environmental Systems Resource Institute, Redlands, California.*

European Commission, 1992. Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora. Council of the European Communities (CEC). *Official Journal of the European Communities*, 206, pp.7–50.

European Commission, 2007. Interpretation Manual of European Union Habitats. p.142.

European Commission, 2013. *Glossary: Livestock unit (LSU) - Statistics Explained*. Eurostat. Available at: <https://ec.europa.eu/eurostat/statistics-explained/index.php/Glossary:Livestock_unit.

European Commission, 2017. Agri-environment schemes: impacts on the agricultural environment Science for Environment Policy Agri-environmental schemes: impacts on the agricultural environment Keep up-to-date. Available at: http://ec.europa.eu/science-environment->

European Commission, 2018. *Natura 2000*. Available at: http://ec.europa.eu/environment/nature/natura2000/index_en.htm>

European Commission, 2018. *The common agricultural policy at a glance | European Commission*. Available at: <https://ec.europa.eu/info/food-farming-fisheries/key-policies/common-agricultural-policy/cap-glance_en>

Evans, G.M.E., 1990. Habits or habitats: Do carabid locomotor adaptations reflect habitats of lifestyles? In: N. Stork, ed. *The Role of Ground Beetles in Ecological and Environmental Studies*. Intercept, Hampshire.pp.295–305.

Evans, M.E.G. and Forsythe, T.G., 1984. A comparison of adaptations to running, pushing and burrowing in some adult Coleoptera: especially Carabidae. *Journal of Zoology*, 202(4), pp.513–534.

Eyre, M.D., McMillan, S.D. and Critchley, C.N.R., 2016. Ground beetles (Coleoptera, Carabidae) as indicators of change and pattern in the agroecosystem: Longer surveys improve understanding. *Ecological Indicators*, 68, pp.82–88.

Fáilte Ireland, 2017. Tourism Facts 2017. (June), pp.1–12.

Fan, P.-F., Ni, Q.-Y., Sun, G.-Z., Huang, B. and Jiang, X.-L., 2008. Seasonal Variations in the Activity Budget of *Nomascus concolor jingdongensis* at Mt. Wuliang, Central Yunnan, China: Effects of Diet and Temperature. *International Journal of Primatology*, 29(4), pp.1047–1057.

Farm Advisory Service, 2017. *Conservation Grazing for Semi-Natural Habitats. Technical Note: TN686*. Available at https://www.fas.scot/downloads/tn686-conservation-grazing-semi-natural-habitats/

Feehan, J., 2003. *Farming in Ireland: history, heritage and environment*. University College Dublin, Faculty of Agriculture.

Fenton, E.W., 1936. The problem of moor mat grass. *Scottish Journal of Agriculture*, 19(143), p.48.

Fenton, E.W., 1937. The Influence of Sheep on the Vegetation of Hill Grazing in Scotland. *The Journal of Ecology*, 25(2), p.424.

Festa-Bianchet, M., 1988. Seasonal range selection in bighorn sheep: conflicts between forage quality, forage quantity, and predator avoidance. *Oecologia*, 75(4), pp.580–586.

Fieberg, J., Matthiopoulos, J., Hebblewhite, M., Boyce, M.S. and Frair, J.L., 2010a. Correlation and studies of habitat selection: problem, red herring or opportunity? *Philosophical transactions of the Royal Society of London. Series B, Biological sciences*, 365(1550), pp.2233–44.

Fieberg, J., Matthiopoulos, J., Hebblewhite, M., Boyce, M.S. and Frair, J.L., 2010b. Correlation and studies of habitat selection: problem, red herring or opportunity? *Philosophical Transactions of the Royal Society B: Biological Sciences*, 365(1550), pp.2233–2244.

Field, A., 2013. *Discovering Statistics using IBM SPSS Statistics*. Sage.

Fielding, A. and Haworth, P.F., 1999. *Upland habitats*. Psychology Press.

Finn, J.A. and Ó hUallacháin, D., 2012a. A review of evidence on the environmental impact of Ireland's Rural Environment Protection Scheme (REPS). In: *Biology & Environment: Proceedings of the Royal Irish Academy*. The Royal Irish Academy.pp.1–24.

Finn, J.A. and Ó hUallacháin, D., 2012b. A review of Evidence on the Environmental Impact of Ireland's Rural Environment Protection Scheme (REPS). In *Biology and Environment: Proceedings of the Royal Irish Academy*, pp11-34.

Followit, 2012. *Tellus GPS System Manual*. Available from https://www.followit.se/wildlife/gps.

Forbes, T.D.A., 1988. Researching the Plant-Animal Interface: The investigation of Ingestive Behaviour in Grazing Animals. *Journal of Animal Science*, 66(9), p.2369.Forest Service, 2012. *The Second National Forest Inventory*. Dublin.

Forsythe, T., 1981. Running and pushing in relationship to hind leg structure in some Carabidae (Coleoptera). *The Coleopterists' Bulletin*. pp.353-378

Forsythe, T.G., 1983. Locomotion in ground beetles (Coleoptera carabidae): An interpretation of leg structure in functional terms. *Journal of Zoology*, 200(4), pp.493–507.

Forsythe, T.G., 1987. The relationship between body form and habit in some Carabidae (Coleoptera). *Journal of Zoology*, 211(4), pp.643–666.

Foss, P.J., O' Connell, C. and Crushell, P.H., 2000. *Bogs & fens of Ireland*. Internal report for NPWS.

Fossitt, J. a., 2000. *A guide to habitats in Ireland*. *Heritage Council of Ireland*. Heritage Council.

Fraser, M.D.D., Theobald, V.J.J., Griffiths, J.B.B., Morris, S.M.M. and Moorby, J.M.M., 2009. Comparative diet selection by cattle and sheep grazing two contrasting heathland communities. *Agriculture, Ecosystems and Environment*, 129(1–3), pp.182–192.

Fryxell, J.M., 1991. Forage Quality and Aggregation by Large Herbivores. *The American Naturalist*, 138(2), pp.478–498.

Fuller, R.J. and Gough, S.J., 1999. Changes in sheep numbers in Britain: implications for bird populations. *Biological Conservation*, 91(91), pp.73–89.

Gardner, S.M., Hartley, S.E., Davies, A. and Palmer, S.C.F., 1997. Carabid communities on heather moorlands in northeast Scotland: the consequences of grazing pressure for community diversity. *Biological conservation*, 81(3), pp.275–286.

Geraghty, K., 2014. *A study of the spider (Araneae) assemblages in dry and wet heath habitats within the Mount Brandon Nature Reserve*. Final year thesis. Institute of Technology, Tralee.

Getz, W.M. and Wilmers, C.C., 2004. A local nearest-neighbour convex-hull construction of home ranges and utilization distributions. *Ecography*, 27(4), pp.489–505.

Gobbi, M., Rossaro, B., Vater, A., De Bernardi, F., Pelfini, M. and Brandmayr, P., 2007. Environmental features influencing Carabid beetle (Coleoptera) assemblages along a recently deglaciated area in the Alpine region. *Ecological Entomology*, 32(6), pp.682–689.

González-Megías, A., Gómez, J.M. and Sánchez-PiÑero, F., 2004. Effects of ungulates on epigeal arthropods in Sierra Nevada National Park (southeast Spain). *Biodiversity and Conservation*, 13(4), pp.733–752.

Grace, J. and Unsworth, M.H., 1988. Climate and microclimate of the uplands. *Ecological change in the uplands*, pp.137–150.

Grant, S.A., Hunter, R.F. and Cross, C., 1963. The effects of muir burning *Molinia*-dominant communities. *Grass and Forage Science*, 18(3), pp.249–257.

Grant, S.A. and Maxwell, T.J., 1988. Hill vegetation and grazing by domesticated herbivores: the biology and definition of management options. *Ecological Change in the Uplands. Blackwell Scientific Publications, Oxford*, pp.201–214.

Grant, S.A., Suckling, D.E., Smith, H.K., Torvell, L., Forbes, T.D.A. and Hodgson, J., 1985. Comparative Studies of Diet Selection by Sheep and Cattle: The Hill Grasslands. *The Journal of Ecology*, 73(3), p.987.

Grant, S.A., Torvell, L., Common, T.G., Sim, E.M. and Small, J.L., 1996. Controlled Grazing Studies on Molinia Grassland: Effects of Different Seasonal Patterns and Levels of Defoliation on Molinia growth and Responses of Swards to Controlled Grazing by cattle. *The Journal of Applied Ecology*, 33(6), p.1267.

Grant, S.A., Torvell, L., Smith, H.K., Suckling, D.E., Forbes, T.D.A. and Hodgson, J., 1987. Comparative Studies of Diet Selection by Sheep and Cattle: Blanket Bog and Heather Moor. *The Journal of Ecology*, 75(4), p.947.

Greenslade, P.J.M., 1964. *Pitfall Trapping as a Method for Studying Populations of Carabidae (Coleoptera). Source: Journal of Animal Ecology*, pp.201-310.

Grime, J.P., 1973. Control of species density in herbaceous vegetation. *J Environ Manage*ment, pp 151-167.

Grime, J.P., 1979. *Plant Strategies and Vegetation Processes*. John Wiley & Sons. Chichester.

GSI, 2008. *Geological Survey of Ireland, Public Mapping. Geological Survey of Ireland*. Available at: <https://www.gsi.ie/Mapping.htm>.

Hall, S., 1988a. Chillingham Park and its herd of white cattle: relationships between vegetation classes and patterns of range use. *Journal of Applied Ecology*, 25(3), pp.777–789.

Hall, S.J.G., 1988b. Chillingham Park and its herd of white cattle: relationships between vegetation classes and patterns of range use. *Journal of Applied Ecology*, 25(3), pp.777–789.

Hall, S.J.G., 1989. Chillingham cattle: social and maintenance behaviour in an ungulate that breeds all year round. *Animal Behaviour*, 38(2), pp.215–225.

Hall, V.A., 1997. The development of the landscape of Ireland over the last two

thousand years; fresh evidence from historical and pollen analytical studies. *Chronion*, 1(1), pp.1–22.

Hall, V.A., 2011. *The Making of Ireland's Landscape Since the Ice Age*. Cork: Collins Press.

Hallmann, C.A., Sorg, M., Jongejans, E., Siepel, H., Hofland, N., Schwan, H., Stenmans, W., Müller, A., Sumser, H., Hörren, T., Goulson, D. and De Kroon, H., 2017. More than 75 percent decline over 27 years in total flying insect biomass in protected areas. *PLoS ONE*, 12(10).

Hampton, M., 2008. *Management of Natura 2000 habitats Northern Atlantic wet heaths with Erica tetralix 4010*. *European Commission*. Available at:

<http://scholar.google.com/scholar?hl=en&btnG=Search&q=intitle:MANAGEMENT+of+Natura+2000+habita+ts+Northern+Atlantic+wet+heaths+with+Erica+tetralix#0>.

Hanya, G., 2004. Seasonal variations in the activity budget of Japanese macaques in the coniferous forest of Yakushima: Effects of food and temperature. *American Journal of Primatology*, 63(3), pp.165–177.

Harestad, A.S. and Bunnell, F.L., 1979. Home range and body weight—a re-evaluation. *Ecology*, 60(2), pp.389–402.

Harper, P.A., Latter, M.R., Nicholas, F.W., Cook, R.W. and Gill, P.A., 1998. Chondrodysplasia in Australian Dexter cattle. *Australian veterinary journal*, 76(3), pp.199-202.

Harrell, F.J., 2016. *Hmisc: Harrell Miscellaneous*. The Hmisc Package. *R package version, 3(0-12), p.3.*

Harvey, J.A., van der Putten, W.H., Turin, H., Wagenaar, R. and Bezemer, T.M., 2008. Effects of changes in plant species richness and community traits on carabid assemblages and feeding guilds. *Agriculture, Ecosystems and Environment*, 127(1–2), pp.100–106.

Hasegawa, N. and Hidari, H., 2001. Relationships among behavior, physiological states and body weight gain in grazing Holstein heifers. Asian Australasian Journal of Animal Sciences, 14(6), pp.803–810.

Haysom, K.A., McCracken, D.I., Foster, G.N. and Sotherton, N.W., 2004. Developing grassland conservation headlands: Response of carabid assemblage to different cutting regimes in a silage field edge. *Agriculture, Ecosystems and Environment*, 102(3), pp.263-277.

Hebblewhite, M. and Haydon, D.T., 2010. Distinguishing technology from biology: ac ritical review of the use of GPS telemetry data in ecology. *Philosophical Transactions of the Royal Society B*, 365, pp.2303–2312.

van Heerwaarden, L.M., Toet, S., van Logtestijn, R.S.P. and Aerts, R., 2005. Internal nitrogen dynamics in the graminoid Molinia caerulea under higher N supply and elevated CO2 concentrations. *Plant and soil*, 277(1–2), pp.255–264.

Heil, G.W. and Diemont, W.H., 1983. Raised nutrient levels change heathland into grassland. *Vegetatio*, 53(2), pp.113–120.

Hejcman, M., Češková, M. and Pavlů, V., 2010. Control of Molinia caerulea by cutting management on sub-alpine grassland. *Flora: Morphology, Distribution, Functional Ecology of Plants*, 205(9), pp.577–582.

Hejcmanov, P., Stejskalov?, M., Pavl?, V. and Hejcman, M., 2009. Behavioural patterns of heifers under intensive and extensive continuous grazing on species-rich pasture in the Czech Republic. *Applied Animal Behaviour Science*, 117(3–4), pp.137–143.

Hejcmanová, P., Stejskalová, M., Pavlů, V. and Hejcman, M., 2009. Behavioural patterns of heifers under intensive and extensive continuous grazing on species-rich pasture in the Czech Republic. *Applied Animal Behaviour Science*, 117(3–4), pp.137–143.

Hepworth, K.W., Test, P.S., Hart, R.H., Waggoner, J.W., Smith, M.A. and Smith, M.A., 1991. Grazing Systems, Stocking Rates, and Cattle Behaviour in Southeastern Wyoming. *Journal of Range Management*, 44(3), p.259.

Heritage Council, 1999. *A Report on the impact of agricultural schemes and payments on aspects of Ireland's heritage*. Heritage Council.

Hessle, A., Rutter, M. and Wallin, K., 2008. Effect of breed, season and pasture moisture gradient on foraging behaviour in cattle on semi-natural grasslands. *Applied Animal Behaviour Science*, 111(1–2), pp.108–119.

Hester, A.J. and Baillie, G.J., 1998. Spatial and temporal patterns of heather use by sheep and red deer within natural heather/grass mosaics. *Journal of Applied Ecology*, 35(5), pp.772–784.

Higashiyama, M. and Hirata, M., 1995. Analysis of a Japanese black cattle rearing system utilizing a bahiagrass (*Paspalum notatum Fluegge*) pasture: Intake from pasture. *Grassland Science, 52(3), pp.117-126.*

Hirata, M., Iwamoto, T., Otozu, W. and Kiyota, D., 2002. The effects of recording interval on the estimation of grazing behaviour of cattle in a daytime grazing system. *Asian Australasian Journal of Animal Sciences, 15(5), pp.745–750.*

Hodd, R., 2012. A study of the ecology of the oceanic montane vegetation of western Ireland and its potential response to climate change. Unpublished PhD Thesis. University College Galway, Ireland.

Hodder, R.M. and Low, W.A., 1978. Grazing distribution of free-ranging cattle at three sites in the Alice Springs District, Central Australia. *The Rangeland Journal*, 1(2), pp.95–105.

Hodge, I. and Reader, M., 2010. The introduction of Entry Level Stewardship in England: Extension or dilution in agri-environment policy? *Land Use Policy*, 27(2), pp270-282.

Hodgson, J., Forbes, T.D.A., Amstrong, R.H., Beattie, M.. and Hunter, E.A., 1991a.Comparative studies of the ingestive behaviour and herbage intake of sheep and cattle grazing indigenous hill plant communities. *Journal of Applied Ecology, pp.205-227.*

Hodgson, J., Forbes, T.D.A., Armstrong, R.H., Beattie, M.M. and Hunter, E.A., 1991b. Comparative Studies of the Ingestive Behaviour and Herbage Intake of Sheep and Cattle Grazing Indigenous Hill Plant Communities. *The Journal of Applied Ecology*, 28(1), p.205.

Hofmann, R.R., 1989. Evolutionary steps of ecophysiological adaptation and diversification of ruminants: a comparative view of their digestive system. *Oecologia*, 78(4), pp.443–457.

Hogg, P., Squires, P. and Fitter, A.H., 1995. Acidification, nitrogen deposition and rapid vegetational change in a small valley mire in Yorkshire. *Biological Conservation*, 71(2), pp.143–153.

Holden, J., Shotbolt, L., Bonn, A., Burt, T.P., Chapman, P.J., Dougill, A.J., Fraser, E.D.G., Hubacek, K., Irvine, B., Kirkby, M.J., Reed, M.S., Prell, C., Stagl, S., Stringer, L.C., Turner, A. and Worrall, F., 2007. Environmental change in moorland landscapes. *Earth-Science Reviews, 82(1-2), pp.75-100.*

Holland, J.P., Pollock, M., Waterhouse, T., Morgan-Davies, C., Bibby, H., Stewart, S. and Armstrong, H.M., 2010. Developing guidance for managing extensive upland grazing where habitats have differing requirements. Nature Scotland commissioned report. Available at https://www.croftingyear.org.uk/userfiles/file/crops/fodder/402guide.pdf

Homburg, K., Drees, C., Boutaud, E., Nolte, D., Schuett, W., Zumstein, P., von Ruschkowski, E. and Assmann, T., 2019. Where have all the beetles gone? Long-term study reveals carabid species decline in a nature reserve in Northern Germany. *Insect Conservation and Diversity*, 12(4), pp.268–277.

Hughes, G.P. and Reid, D., 1951. Studies on the behaviour of cattle and sheep in relation to the utilization of grass. *The Journal of Agricultural Science*, 41(04), p.350.

Huk, T. and Kühne, B., 1999. Substrate selection by Carabus clatratus (Coleoptera, Carabidae) and its consequences for offspring development. *Oecologia*, 121(3), pp.348–354.

Hulme, P.., Merrell, B.., Torvell, L., Fisher, J.., Small, J.. and Pakeman, R.., 2002. Rehabilitation of degraded Calluna vulgaris (L.) Hull-dominated wet heath by controlled sheep grazing. *Biological Conservation*, 107(3), pp.351–363.

Hurlbert, S.H., 1984. Pseudoreplication and the Design of Ecological Field Experiments. *Ecological Monographs*, 54(2), pp.187–211.

Huston, M., 1979. A General Hypothesis of Species Diversity. *The American Naturalist*, 113(1), pp81-101.

IBM Corp., 2013 & 2017. *SPSS Statistics for Windows*.

Image, M., 2016. *Evaluation of the GLAS Phase 1*. Department of Agriculture, Food and Marine.

Indecon, 2017. *Ex-Post Evaluation of the Rural Development Programme Ireland (2007-2013). Final Report.* Produced by Indecon International EconomicConsultants. Department of Agriculture, Food and the Marine (DAFM).

Iremonger, S., O'Halloran, J., Kelly, D.., Wilson, M.., Smith, G.., Gittings, T., Giller, P.,Mitchell, F.J.., Oxbrough, A.G., Coote, L., Frnech, L., O'Donoghue, S., McKee, A.-., Pithon, J., O'Sullivan, A., Neville, P., O'Donnell, V., Cummins, V., Kelly, T.. andDowding, P., 2006. Biodiversity in Irish plantation forests. EPA and COFORD, Dublin.

Irish Wind Energy Association, 2018. *Facts and Stats*. Available at: <https://www.iwea.com/about-wind/facts-stats>

Jackson, P.W., 1994. *The Geology of Kerry: An exhibition devised by Patrick Wyse Jackson, Geological Museum, Trinity College, Dublin.*

Jacobs, J., 1974. International Association for Ecology Quantitative Measurement of Food Selection: A Modification of the Forage Ratio and Ivlev's Electivity Index. *Oecologia*, 14(4), pp.413–417.

Jelaska, L.S. and Durbesˇić, P., 2009. Comparison of the body size and wing form of carabid species (coleoptera: Carabidae) between isolated and continuous forest habitats. *Annales de la Societe Entomologique de France*, 45(3), pp.327–338.

Job, D.A. and Taylor, J.A., 1978. The production, utilization and management of upland grazings on Plynlimon, Wales. *Journal of Biogeography*, pp.173–191.

Johnson, C.J., Parker, K.L., C., H.D. and Gillingham, M.P., 2002. Movement parameters of ungulates and scale-specific responses to environment. *Journal of Animal Ecology*, 71, pp.225–235.

Johnson, D.H., 1980. The comparison of usage and availability measurements for evaluating resource preference. *Ecology*, 61(1), pp.65–71.

Jopp, F. and Reuter, H., 2005. Dispersal of carabid beetles - Emergence of distribution patterns. *Ecological Modelling*, 186(4), pp.389–405.

Kaltsas, D., Trichas, A., Kougioumoutzis, K. and Chatzaki, M., 2013. Ground beetles respond to grazing at assemblage level, rather than species-specifically: The case of Cretan shrublands. *Journal of Insect Conservation*, 17(4), pp.681–697.

Kaufmann, J., Bork, E.W., Blenis, P. V. and Alexander, M.J., 2013. Cattle habitat selection and associated habitat characteristics under free-range grazing within heterogeneous Montane rangelands of Alberta. *Applied Animal Behaviour Science*, 146(1–4), pp.1–10.

Kauhala, K. and Auttila, M., 2009. Estimating habitat selection of badgers—a test between different methods. *Folia Zoologica*, 59, pp.16–25.

Kauhala, K. and Auttila, M., 2010. Estimating habitat selection of badgers - A test between different methods. *Folia Zoologica*, 59(1), pp.16–25.

Kenward, R., 2001. *A manual for wildlife radio tagging*. London: Academic Press.

Kenward, R., Walls, S., South, A. and Casey, N., 2008. *Ranges 8: For the analysis of tracking and location data. manual*. Available at: <http://www.anatrack.com>.

Kenward, R.E., Clarke, R.T., Hodder, K.H. and Walls, S.S., 2001b. Density and linkage estimators of home range: Nearest-neighbour clustering defines multinuclear cores. *Ecology*, 82(7), pp.1905–1920.

Kie, J.G., Baldwin, J.A. and Evans, C.J., 1996. CALHOME: a program for estimating animal home ranges. *Wildlife Society Bulletin*, pp.342–344.

Kie, J.G. and Boroski, B.B., 1996. Cattle distribution, habitats, and diets in the Sioerra Nevada of California. *Journal of Range Management*, 49(6).

Kilgour, R.J., 2012. In pursuit of "normal": A review of the behaviour of cattle at pasture. *Applied Animal Behaviour Science*, 138(1–2), pp.1–11.

Kilgour, R.J., Uetake, K., Ishiwata, T. and Melville, G.J., 2012. The behaviour of beef cattle at pasture. *Applied Animal Behaviour Science*, 138(1–2), pp.12–17.

Kleijn, D., Baquero, R.A., Clough, Y., Díaz, M., De Esteban, J., Fernández, F., Gabriel, D., Herzog, F., Holzschuh, A., Jöhl, R., Knop, E., Kruess, A., Marshall, E.J.P., Steffan-Dewenter, I., Tscharntke, T., Verhulst, J., West, T.M. and Yela, J.L., 2006. Mixed biodiversity benefits of agri-environment schemes in five European countries. *Ecology Letters*, 9(3), pp.243–254.

Kleijn, D. and Sutherland, W.J., 2003a. How effective are European agri-environment schemes in conserving and promoting biodiversity? *Journal of applied ecology*, 40(6), pp.947–969.

Kleijn, D. and Sutherland, W.J., 2003b. How effective are European agri-environment schemes in conserving and promoting biodiversity? *Journal of applied ecology*, 40(6), pp.947–969.

Klink, R., Van Der Plas, F., Van Noordwijk, C.G.E., Wallis de Vries, M.F. and Olff, H., 2015. Effects of large herbivores on grassland arthropod diversity. 0031, pp.347–366.

Klink, R. van, Rickert, C., Vermeulen, R., Vorst, O., WallisDeVries, M.F. and Bakker, J.P., 2013. Grazed vegetation mosaics do not maximize arthropod diversity: Evidence from salt marshes. *Biological Conservation*, 164, pp.150–157.

Koivula, M.J., 2011. Useful model organisms, indicators, or both? Ground beetles (Coleoptera, Carabidae) reflecting environmental conditions. *ZooKeys*, (100), p.287. d

Kondo, S., 2011. *Recent progress in the study of behaviour and management in grazing cattle*. *Animal Science Journal*, 82(1), pp.26-35.

Körner, C., Jetz, W., Paulsen, J., Payne, D., Rudmann-Maurer, K. and M. Spehn, E., 2017. A global inventory of mountains for bio-geographical applications. *Alpine Botany*, 127(1), pp.1–15.

Körner, C., Paulsen, J. and Spehn, E., 2010. *A digital "Mountain Biodiversity Portal" for global biodiversity research*: A Global Assessment. Available at https://link.springer.com/article/10.1007/s00035-011-0094-4

Kotze, D.J., Brandmayr, P., Casale, A., Dauffy-Richard, E., Dekoninck, W., Koivula, M.J., Lövei, G.L., Mossakowski, D., Noordijk, J. and Paarmann, W., 2011. Forty years of carabid beetle research in Europe–from taxonomy, biology, ecology and population studies to bioindication, habitat assessment and conservation. *ZooKeys*, (100), p.55.

Kotze, D.J. and O'Hara, R.B., 2003. Species decline - But why? Explanations of carabid beetle (Coleoptera, Carabidae) declines in Europe. *Oecologia*, 135(1), pp.138–148.

Kromp, B., 1999. Carabid beetles in sustainable agriculture: a review on pest control efficacy, cultivation impacts and enhancement. *Agriculture, Ecosystems and Environment*, 74, pp.1987–228.

Kropp, J.R., Holloway, J.W., Stephens, D.F., Knori, L., Morrison, R.D. and Totusek, R., 1973. Range Behavior of Hereford, Hereford X Holstein and Holstein Non-Lactating Heifers. *Journal of Animal Science*, 36(4), p.797.

Kruess, A. and Tscharntke, T., 2002a. Contrasting responses of plant and insect diversity to variation in grazing intensity. *Biological Conservation*, 106(3), pp.293–302.

Kryzhanovskii, O.L., 1976a. An attempt at a revised classified classification of the family Carabidae (Coleoptera). *Entomological Review*, 55, pp.56–54.

Kryzhanovskii, O.L., 1976b. An attempt of revised classification of the family Carabidae (Coleoptera). *Entomological Reviews*, (55), pp.56–64.

Langraf, V., Petrovičová, K., David, S., Ábelová, M. and Schlarmannová, J., 2017. Body volume in ground beetles (Carabidae) reflects biotope disturbance. *Folia Oecologica*, 44(2), pp.114–120.

Langraf, V., Petrovičová, K., David, S., Nozdrovická, J., Petrovič, F. and Schlarmannová, J., 2019. The Bioindication Evaluation of Ground Beetles (Coleoptera: Carabidae) in Three Forest Biotopes in the Southern Part of Central Slovakia. *Ekologia Bratislava*, 38(1), pp.25–36.

Laver, P.N. and Kelly, M.J., 2008. A critical review of home range studies. *The Journal of Wildlife Management*, 72(1), pp.290–298.

Lenoir, L. and Lennartsson, T., 2010. Effects of timing of grazing on arthropod communities in semi-natural grasslands. *Journal of Insect Science*, 10(1), pp.1–24.

Lindsay, R., Charman, D.J., Everingham, F., Reilly, R.M.O., Palmer, M.A., Rowell, T.A. and Stroud, D.A., 1988. *The Flow Country- The peatlands of Caithness and Sutherland. JNCC Research Report*.

Linnane, M.I., Brereton, A.J. and Giller, P.S., 2001. Seasonal changes in circadian grazing patterns of Kerry cows (Bos Taurus) in semi-feral conditions in Killarney National Park, Co. Kerry, Ireland. *Applied Animal Behaviour Science*, 71(4), pp.277–292.

Lister, B.C. and Garcia, A., 2018. Climate-driven declines in arthropod abundance restructure a rainforest food web. *Proceedings of the National Academy of Sciences*, 115(44), pp.E10397–E10406.

Liu, T., Green, A.R., Rodríguez, L.F., Ramirez, B.C. and Shike, D.W., 2015. *Effects of number of animals monitored on representations of cattle group movement characteristics and spatial occupancy. PLoS one, 10(2), pp.e* 113117.

Liu, Z., Wang, X., Li, Z., Cui, D. and Li, X., 2005. Seasonal variation of diurnal activity budgets by blue sheep (*Pseudois nayaur*) with different age-sex classes in Helan Mountain. *Zoological Research*, 26(4), pp.350–357.

Lomba, A., Guerra, C., Alonso, J., Honrado, J.P., Jongman, R. and McCracken, D., 2014. Mapping and monitoring High Nature Value farmlands: Challenges in European landscapes. *Journal of Environmental Management*, 143, pp.140–150.

Loureiro, F. and Rosalino, L.M., 2009. Telemetry: An Ecologists Swiss-Army Knife. In:D.D. and J. Barculo, ed. *Telemetry: Research, Technology and Applications*. New ork: Nova Science Publishers.

Lövei, G.L. and Sunderland, K.D., 1996. Ecology and Behavior of Ground Beetles (Coleoptera: Carabidae). *Annual Review of Entomology*, 41(1), pp.231–256.

Low, D., 1845. *On the domesticated animals of this British Isles*. Longmans, Brown, Green & Longmans.

Low, W.A.A., Tweedie, R.L.L., Edwards, C.B.H.B.H., Hodder, R.M.M., Malafant,K.W.J.W.J. and Cunningham, R.B.B., 1981. The influence of environment on daily maintenance behaviour of free-ranging Shorthorn cows in central Australia. I. General introduction and descriptive analysis of day-long activities. *Applied Animal Ethology*, 7(1), pp.11–26.

Luff, M.L., 1975. Some features influencing the efficiency of pitfall traps. *Oecologia*, 19(4), pp.345–357.

Luff, M.L., 2007. *The Carabidae (ground Beetles) of Britain and Ireland (Handbooks for the Identification of British Insects).* 2nd ed. Royal Entomological Sociey.

Lyons, A., 2017. *The Impacts of Contrasting Grazing Management on Biodiversity in Upland Calcareous Grasslands*. PhD Thesis. Edge Hill University.

Lyons, A., Ashton, P.A., Powell, I. and Oxbrough, A., 2017. Impacts of contrasting conservation grazing management on plants and carabid beetles in upland calcareous grasslands. *Agriculture, Ecosystems and Environment*, 244, pp.22–31.

Lyons, A.J., Turner, W.C. and Getz, W.M., 2013. Home range plus: a space-time characterization of movement over real landscapes. *Movement Ecology*, 1(1), p.2.

Maestre, F.T., Quero, J.L., Gotelli, N.J., Escudero, A., Ochoa, V., Delgado-Baquerizo, M., García-Gómez, M., Bowker, M.A., Soliveres, S., Escolar, C., García-Palacios, P., Berdugo, M., Valencia, E., Gozalo, B., Gallardo, A., Aguilera, L., Arredondo, T., Blones, J., Boeken, B., Bran, D., Conceição, A.A., Cabrera, O., Chaieb, M., Derak, M., Eldridge, D.J., Espinosa, C.I., Florentino, A., Gaitán, J., Gabriel Gatica, M., Ghiloufi, W., Gómez-González, S., Gutiérrez, J.R., Hernández, R.M., Huang, X., Huber-Sannwald, E., Jankju, M., Miriti, M., Monerris, J., Mau, R.L., Morici, E., Naseri, K., Ospina, A., Polo, V., Prina, A., Pucheta, E., Ramírez-Collantes, D.A., Romão, R., Tighe, M., Torres-Díaz, C., Val, J., Veiga, J.P., Wang, D. and Zaady, E., 2012. Plant species richness and ecosystem multifunctionality in global drylands. *Science*, 335(6065), pp.214–218.

Mandaluniz, N., Oregui, L.M. and Aldezabal, A., 2005. Utilization of gorse and heather communities by cattle in mountain grazing. *Silvopastoralism and Sustainable Land Management. CABI Publishing, Oxford, United Kingdom*, pp.157–159.

Manly B.F.J., 1997. *Randomization, Bootstrap and Monte Carlo Methods in Biology*. Second ed. London: Chapman and Hall.

Manly, B.F.L., McDonald, L., Thomas, D.L., McDonald, T.L. and Erickson, W.P., 2007. *Resource selection by animals: statistical design and analysis for field studies*. Springer Science & Business Media.

Manly, B.F.J., McDonald, L.L., Thomas, D.L., McDonald, T.L. and Erickson, W.P.,2002. Resource selection by animals: statistical design and analysis for field studies. *Technology*, p.221.

Marcus, T., Boch, S., Durka, W., Fischer, M., Gossner, M.M., Müller, J., Schöning, I., Weisser, W.W., Drees, C. and Assmann, T., 2015. Living in heterogeneous woodlands – are habitat continuity or quality drivers of genetic variability in a flightless ground beetle? *PLoS One*, 10(12), pp.0144217.

Marriott, C., Fothergill, M., Jeangros, B., Scotton, M. and Louault, F., 2003. Long-term impacts of extensification of grassland management on biodiversity and productivity in upland areas. A review. *Agronomie*, 23, pp.407–418.

Marriott, C.A., Hood, K., Fisher, J.M. and Pakeman, R.J., 2009. Long-term impacts of extensive grazing and abandonment on the species composition, richness, diversity and productivity of agricultural grassland. *Agriculture, Ecosystems and Environment*, 134(3–4), pp.190–200.

Marrs, R.H., Phillips, J.D.P., Todd, P.A., Ghorbani, J. and Le Duc, M.G., 2004a. Control of Molinia caerulea on upland moors. *Journal of Applied Ecology*, 41(2), pp.398–411.

Marrs, R.H., Phillips, J.D.P., Todd, P.A., Ghorbani, J. and Le Duc, M.G., 2004b. Control of Molinia caerulea on upland moors. *Journal of Applied Ecology*, 41(2), pp.398–411.

Martin, D., Fraser, M.., Pakeman, R.J., Moffat, A.M., Frazer, M.D., Pakeman, R.J. and Moffat, A.M., 2013. *Impact of moorland grazing and stocking rates*. Natural England.

Mazzei, A., Bonacci, T., Gangale, C., Pizzolotto, R. and Brandmayr, P., 2015. Functional species traits of carabid beetles living in two riparian alder forests of the Sila plateau subject to different disturbance factors (Coleoptera: Carabidae). *Fragmenta Entomologica*, 47(1), p.37.

Mccormack, S., Regan, E., Nolan, M. and Anderson, R., 2009. Coleoptera from montane blanket bog in the Wicklow mountains. *Bulletin of the Irish Biogeographical Society*, 33(81–98).

McCune, B., Grace, J.B. and Urban, D.L., 2002. Analysis of Ecological Communities. *MjM Software Design*, p.300.

McCune, B. and Mefford, M.J., 2011. *PC-Ord. Mltivariate Analysis of Ecological Data*.

McFerran, D.M., McAdam, J.H. and Montgomery, W.I., 1995. The impact of burning and grazing on heathland plants and invertebrates in County Antrim. In *Biology and Environment: Proceedings of the Royal Irish Academy, pp.1-17.* Royal Irish Academy.

McFerran, D.M., Meharg, M.J., Montgomery, W.I. and McAdam, J.H., 1994. The impact of grazing on communities of ground-dwelling beetles (Coleoptera: Carabidae) in upland vegetation in north-east Ireland. In: *Carabid beetles: ecology and evolution*. Dordrecht: *Springer*.pp.325–330.

McGurn, P. and Moran, J., 2013. *A National, Outcome-based Agri-environment Programme Under Ireland's Rural Development Programme 2014-2020*. Available at: <http://www.efncp.org/download/AGRI_ENVIRONMENT-SCHEME- RDP-2014-2020_final12Dec.pdf>

McNaughton, S.J., 1984. Grazing lawns: Animals in herds, plant form and coevolution. *American Naturalist*, 124(6), pp.863–886.

Mech, L.D. and Barber, S.M., 2002. A critique of wildlife radio-tracking and its use in national parks. *A report to the US National Park Service*, pp.19–20.

Meehan, M., Sedivec, K.K., Printz, J. and Brummer, F., 2018. Determining carrying capacity and stocking rates for range and pasture in North Dakoa. NDSU Extension, North Dakota University.

Menard, S.W., 1995. *Applied logistic regression analysis*. Sage Publications.

Milchunas, D.G., 2011. Plant community dynamics in shortgrass steppe with grazing relaxation and imposition by large and small herbivores. *Israel Journal of Ecology and Evolution*, 57(1–2), pp.23–41.

Miles, J., 1988. Vegetation and soil change in the uplands. In: M.B. Usher and D.B. Thompson, eds. *Ecological change in the uplands*. Oxford: Blackwell Scientific Publications.pp.57–70.

Millenium Ecosystem Assessment, 2010. Ecosystems and Human Well-Being: Biodiversity Synthesis. *Ecosystems*, p.285.

Miller, G.R., Miles, J. and Heal, O.W., 1984. *Moorland management: a study of Exmoor.* Institute of Terrestrial Ecology.

Mills, J., Rook, A.J., Dumont, B., Isselstein, J., Scimone, M. and Wallis De Vries, M.F., 2007. Effect of livestock breed and grazing intensity on grazing systems: 5.

Management and policy implications. *Grass and Forage Science*, 62(4), pp.429–436.

Milne, J.A., Birch, C.P.D., Hester, A.J., Armstrong, H.M. and Robertson, A., 1998. *The impact of vertebrate herbivores on the natural heritage of the Scottish uplands-a review*. Scottish Natural Heritage Edinburgh.

Mitchell, F. and Ryan, M., 1997. *Reading the Irish landscape*. Town House Dublin.

Mitchell, M.S. and Powell, R.A., 2007. Optimal use of resources structures home ranges and spatial distribution of black bears. *Animal Behaviour*, 74(2), pp.219–230.

Mitchell, M.S. and Powell, R.A., 2008. Estimated home ranges can misrepresent habitat relationships on patchy landscapes. *Ecological Modelling*, 216(3–4), pp.409–414.

Mohr, C.O., 1947. Table of equivalent populations of North American small mammals. *American midland naturalist*, 37(1), pp.223–249.

Monaghan, F., 2012. *What is Commonage*. Available at: http://www.yourcommonage.ie/p/commonage_3.html> .

Moncorps, S., Boussès, P., Réale, D. and Chapuis, J.-L., 1997. Diurnal time budget of the mouflon (*Ovis musimon*) on the Kerguelen archipelago: influence of food resources, age, and sex. *Canadian Journal of Zoology*, 75(11), pp.1828–1834.

Moyer, M.A., McCown, J.W. and Oli, M.K., 2007. Factors influencing home-range size of female black bears. *Journal of Mammology*, 88(2), pp.468–476.

Moyo, B., Dube, S., Lesoli, M. and Masika, P., 2013. Seasonal habitat use and movement patterns of cattle grazing different rangeland types in the communal areas of the Eastern Cape, South Africa. *African Journal of Agricultural Research*, 8(1), pp.36–45.

Myers, R.H., 1990. *Classical and modern regression with applications*. Vol. 2. Belmont, CA. Duxbury Press.

Nagy, L., Nagy, J., Legg, C.J., Sales, D.I. and Horsfield, D., 2002. Monitoring vegetation change caused by trampling: a study in the Cairngorms, Scotland. *Botanical Journal of Scotland*, 54(2), pp.191–207.

Nature Reserve (Mount Brandon) Establishment Order. Available at: <http://www.irishstatutebook.ie/1976/en/act/pub/0039/index.html>.

Newman, J.A., Parsons, A.J., Thornley, J.H.M., Penning, P.D. and Krebs, J.R., 1995. Optimal Diet Selection by a Generalist Grazing Herbivore. *Functional Ecology*, 9(2), p.255.

Newton, I., 2004. The recent declines of farmland bird populations in Britain: An appraisal of causal factors and conservation actions. *Ibis*, 146(4), pp.579–600.

Ng, K., Barton, P.S., Blanchard, W., Evans, M.J., Lindenmayer, D.B., Macfadyen, S., McIntyre, S. and Driscoll, D.A., 2018. Disentangling the effects of farmland use, habitat edges, and vegetation structure on ground beetle morphological traits. 188(3), pp.645–657.

Niemelä, J.K. and Kotze, D.J., 2007. The effects of forestry on carabid beetles (Coleoptera: Carabidae) in boreal forests. *Journal of Insect Conservation*, (11), pp.5–18.

Nolan, W., 2009. *Geography of Ireland*. Available at: <https://web.archive.org/web/20091124012641/http://www.gov.ie/en/essays/geography.html>

NPWS, 2008. The status of EU protected habitats and species in Ireland. National Parks and Wildlife Service, Dublin. Available at: <http://www.npws.ie/publications/euconservationstatus/NPWS_2007_Conservation_Status_Report.pdf>.

NPWS, 2009. *Mount Brandon cSAC Conservation Statement*. Dublin: NPWS, Dept. of Environment, Heritage and Local Government.

NPWS, 2013. The status of EU protected habitats and species in Ireland. Habitat Assessments. National Parks and Wildlife Service, Department of Arts, Heritage and the Gaeltacht, Dublin.

NPWS, 2019a. The Status of EU Protected Habitats and Species in Ireland: Species Assessments. National Parks and Wildlife Service, Department of Arts, Heritage and the Gaeltacht, Dublin.

NPWS, 2019b. The Status of EU Protected Habitats and Species in Ireland: Vol. 1 Overview. *The Status of EU Protected Habitats and Species in Ireland*, 1: Summary, p.44.

NPWS, 2019c. The status of EU protected habitats and species in Ireland. Habitat Assessments. 2, p.149. National Parks and Wildlife Service, Department of Arts, Heritage and the Gaeltacht, Dublin.

Nunes, L., Silva, I., Pité, M., Rego, F., Leather, S. and Serrano, A., 2006. Carabid

(Coleoptera) Community Changes Following Prescribed Burning and the Potential Use of Carabids as Indicators Species to Evaluate the Effects of Fire Management in Mediterranean Regions. *Silva Lusitana*, 14(1), pp.85–100.

O'Dwyer, J.G., 2018. Rising number of walkers threatens Irish mountains. *The Irish Times ()*. Available at: <https://www.irishtimes.com/life-and-style/travel/ireland/rising-number-of-walkers-threatens-irish-mountains-1.3553601>.

O'Rourke, E., Kramm, N., Chisholm, N., O'Rourke, E., Kramm, N., Chisholm, N., O'Rourke, E., Kramm, N. and Chisholm, N., 2012. The influence of farming styles on the management of the Iveragh uplands, southwest Ireland. *Land Use Policy*, 29(4), pp.805–816.

Olff, H. and Ritchie, M.E., 1998. *Effects of herbivores on grassland plant diversity*. *Trends in Ecology and Evolution*, 13(7), pp.261-265.

Oliván, M. and Osoro, K., 1998. Foraging behaviour of grazing ruminants in rangelands. *Pasture ecology and animal intake (Keane MG, O'Riordan EG, eds). Teagasc, Meath, Ireland*, pp.110–126.

Oliver, T.H., Isaac, N.J.B., August, T.A., Woodcock, B.A., Roy, D.B. and Bullock, J.M., 2015. Declining resilience of ecosystem functions under biodiversity loss. *Nature Communications*, 6(1), p.10122.

Olofsson, J., Kitti, H., Rautiainen, P., Stark, S. and Oksanen, L., 2001. Effects of summer grazing by reindeer on composition of vegetation, productivity and nitrogen cycling. *Ecography*, 24(1), pp.13–24.

Orr, R.J., Tallowin, J.R.B., Griffith, B.A. and Rutter, S.M., 2014a. Effects of livestock breed and rearing experience on foraging behaviour of yearling beef cattle grazing unimproved grasslands. *Grass and Forage Science*, 69(1), pp.90–103.

Orr, R.J., Tallowin, J.R.B., Griffith, B.A. and Rutter, S.M., 2014b. Effects of livestock breed and rearing experience on foraging behaviour of yearling beef cattle grazing unimproved grasslands. *Grass and Forage Science*, 69(1), pp.90–103.

Orr, R.J., Tozer, K.N., Griffith, B.A., Champion, R.A., Cook, J.E. and Rutter, S.M., 2012. Foraging paths through vegetation patches for beef cattle in semi-natural pastures. *Applied Animal Behaviour Science*, 141(1–2), pp.1–8.

Owens, M.K., Launchbaugh, K.L. and Hollway, 1991. Pasture characteristics affecting characteristics affecting spatial distribution of utilization by cattle in mixed brush communities. *Journal of Range Management Archives*, 44(2), pp. 118-123.

Oxbrough, A., French, V., Irwin, S., Kelly, T.C., Smiddy, P. and O'Halloran, J., 2012. Can mixed species stands enhance arthropod diversity in plantation forests? *Forest Ecology and Management*, 270, pp.11-18.

Oxbrough, A., Irwin, S., Kelly, T.C. and O'Halloran, J., 2010. Ground-dwelling invertebrates in reforested conifer plantations. *Forest Ecology and Management*, 259(10), pp.2111–2121.

Parker, K.L., Gillingham, M.P., Hanley, T.A. and Robbins, C.T., 1999. Energy and Protein Balance of Free-Ranging Black-Tailed Deer in a Natural Forest Environment. *Wildlife Monographs*, pp.3–48.

Paton, J.A., 1999. *Liverwort flora of the British Isles*. Harley Books.

Pearce-Higgins, J.W., 2010. Using diet to assess the sensitivity of northern and upland birds to climate change. *Climate Research*, 45, pp.119–130.

Peck, J.E., 2010. *Multivariate analysis for community ecologists*. MjM Software Design. Pendleton, G.W., Titus, K., DeGayner, E., Flatten, C.J. and Lowell, R.E., 1998.

Compositional Analysis and GIS for Study of Habitat Selection by Goshawks in Southeast Alaska. *Journal of Agricultural, Biological, and Environmental Statistics*, 3(3), p.280.

Perrigo, A., Hoorn, C. and Antonelli, A., 2020. Why mountains matter for biodiversity. *Journal of Biogeography*, 47(2), pp.315–325.

Perrin, J.-B., Ducrot, C., Vinard, J.-L., Morignat, E., Gauffier, A., Calavas, D. and Hendrikx, P., 2010. Using the National Cattle Register to estimate the excess mortality during an epidemic: Application to an outbreak of Bluetongue serotype 8. *Epidemics*, 2(4), pp.207–214.

Perrin, P., Barron, S., Roche, J. and O Hanrahan, B., 2011a. Guidelines for a national survey and conservation assessment of upland vegetation and habitats in Ireland. Version 2.0. Irish Wildlfie Manuals, No. 79. *Irish Wildlife Manuals*, 79(79).

Perrin, P., Martin, J., Barron, S., O'Neil, F., McNutt, K. and Delaney, A., 2008. National survey of native woodland in Ireland. National Parks & Wildlife Service, Dublin.

Perrin, P., Roche, J., Daly, O. and Douglas, C., 2011b. The national survey of upland habitats. In: D. Ó hUallachái and D. Finn, eds. *Conserving Farmland Biodiversity*. Teagasc.pp.88–89.

Perrin, P.M., Barron, S.J., Roche, J.R. and O'Hanrahan, B., 2014a. *Guidelines for a national survey and conservation assessment of upland vegetation and habitats in Ireland. Version 2.0.* National Parks and Wildlife Service.

Perrin, P.M., O'Hanrahan, B., Roche, J.R. and Barron, S.J., 2009. *Scoping study and pilot survey for a national survey and conservation assessment of upland habitats and vegetation in Ireland.* Dept. Environment Heritage and Local Government.

Perrin, P.M., Roche, J.R., Barron, S.J. and Daly, O.H., 2013. *National Survey of Upland Habitats Site Report No. 7: Mount Brandon cSAC (000375), Co Kerry (DRAFT)*.

Perrin, P.M., Roche, J.R., Barron, S.J. and Daly, O.H., 2014b. *National Survey of Upland Habitats (Phase2, 2011-2012). Site Report No.7: Mt Brandon cSAC (00375), Co Kerry. (Revision)*.

Persson, S., 1984. Vegetation development after the exclusion of grazing cattle in a meadow area in the south of Sweden. *Vegetatio*, 55(2), pp.65–92.

Petit, S. and Burel, F., 1998. Connectivity in fragmented populations: *Abax parallelepipedus* in a hedgerow network landscape. *Comptes Rendus de l'Academie des Sciences-Series III-Sciences de la Vie*, *321*(1), pp.55-61.

Pizzolotto, R., Gobbi, M. and Brandmayr, P., 2014. Changes in ground beetle assemblages above and below the treeline of the Dolomites after almost 30 years (1980/2009). *Ecology and Evolution*, 4(8), pp.1284–1294.

Pokorná, P., Hejcmanová, P., Hejcman, M. and Pavlu, V., 2013. Activity time budget patterns of sheep and goats co-grazing on semi-natural species-rich dry grassland. *Czech Journal of Animal Science*, 58(5), pp.208–216.

Pollak, S., 2017. Star Wars and Game of Thrones linked to growth in tourist visits. *The Irish Times* (, available at https://www.irishtimes.com/news/consumer/star- wars-and-game-of-thrones-linked-to-growth-in-tourist-visits-1.2953715).

Poos, M. S., and Jackson, D. A. (2012). Addressing the removal or rare species in multivariate bioassessments. The impact of methodological choices. *Ecological Indicators, 18, 82-90.*

Pöyry, J., Luoto, M., Paukkunen, J., Pykälä, J., Raatikainen, K. and Kuussaari, M., 2006. Different responses of plants and herbivore insects to a gradient of vegetation height: an indicator of the vertebrate grazing intensity and successional age. *Oikos*, 115(3), pp.401–412.

Prather, C.M., Pelini, S.L., Laws, A., Rivest, E., Woltz, M., Bloch, C.P., Toro, I. Del, Ho, C.-K., Kominoski, J., Newbold, T.A.S., Parsons, S., Joern, A. and Kellogg, W.K., 2013. *Biological Reviews*, 88, pp.327–348.

Purtauf, T., Dauber, J. and Wolters, V., 2005. The response of carabids to landscape simplification differs between trophic groups. *Oecologia*, 142(3), pp.458–464.

Rainio, J. and Niemelä, J., 2003. Ground beetles (Coleoptera: Carabidae) as bioindicators. *Biodiversity & Conservation*, 12(3), pp.487–506.

Ratcliffe, D.A., 1968. An ecological account of Atlantic bryophytes in the British Isles. *The New Phytologist*, 67(2), pp.365–439.

Ratcliffe, D.A., 1977. *A nature conservation review: the selection of biological sites of national importance* to nature conservation in Britain (vol 1). Cambridge University Press.

Ratcliffe, D.A. and Thompson, D.B.A., 1988. The British uplands: their ecological character and international significance. *Ecological change in the uplands*, pp.9–36.

Van Rensburg, T.M., Murphy, E. and Rocks, P., 2009. Commonage land and farmer uptake of the rural environment protection scheme in Ireland. *Land Use Policy*, 26(2), pp.345–355.

Reppert, J.N., 1960. Forage Preference and Grazing Habits of Cattle at the Eastern Colorado Range Station. *Journal of Range Management*, 13(2), p.58.

Ribera, I., Dolé Dec, S., Downie, I.S. and Foster, G.N., 2001. Effect of land disturbance and stress on species traits of ground beetle assemblages. *Ecology*, 82(4), pp.1112–1129.

Ribera, I., McCracken, D.I., Foster, G.N., Downie, I.S. and Abernethy, V.J 1999. Morphological diversity of ground beetles (Coleoptera: Carabidae) in Scottish agricultural land. *Journal of Zoology*, 247(1), pp.1–18.

Riley, K.N. and Browne, R.A., 2011. Changes in ground beetle diversity and community composition in age structured forests (Coleoptera, Carabidae). *ZooKeys* (147), pp.601–21.

Roath, L.R. and Kreuger, W.C., 1982. Cattle Grazing and Behavior on a Forested area. *Journal of Rangeland Management*, 35(3).

Roath, L.R., Kreuger, W.C. and Krueger, W.C., 1982. Cattle Grazing and Behaviour on a Forested area. *Journal of Range*, 35(3), p.332.

Rodgers, M., O'Connor, M., Healy, M.G., O'Driscoll, C., Asam, Z.U.Z., Nieminen, M., Poole, R., Müller, M. and Xiao, L., 2010. Phosphorus release from forest harvesting on an upland blanket peat catchment. *Forest Ecology and Management*, 260(12), pp.2241–2248.

Rodwell, J.S., 1992. *Grasslands and Montane Communities. British Plant Communities vol. 3.* Cambridge: Cambridge University Press.

Roem, W.J., Klees, H. and Berendse, F., 2002. Effects of nutrient addition and acidification on plant species diversity and seed germination in heathland. *Journal of Applied Ecology*, 39(6), pp.937–948.

Rook, A.J., Dumont, B., Isselstein, J., Mills, J., Osoro, K., Scimone, M. and Wallis de Vries M.F, 2007. Effects of livestock breed and grazing intensity on grazing systems: 4. Effects on animal diversity. *Grass and Forage Science*, 62(2), pp.185–197.

Rook, A.J.J., Dumont, B., Isselstein, J., Osoro, K., WallisDeVries, M.F.F., Parente, G. and Mills, J., 2004. Matching type of livestock to desired biodiversity outcomes in pastures - A review. *Biological Conservation*, 119(2), pp.137–150.

Rooney, S.M., Wolfe, A. and Hayden, T.J., 1998. Autocorrelated data in telemetry studies: time to independence and the problem of behavioural effects. *Mammal Review*, 28(2), pp.89–98.

RStudio Team, 2018. *RStudio: Integrated Development Environment for R*. Boston, M 770.

Ruckstuhl, K.E. and Neuhaus, P., 2009. Activity budgets and sociality in a monomorphic ungulate: the African oryx (*Oryx gazella*). *Canadian Journal of Zoology*, 87(2), pp.165–174.

S.I. No 605 of 2017, 2017. Statutory Instruments. S.I. No. 605 of 2017 European Union (Good Agricultural Practice for Protection of Waters) Regulations 2017. (605).

Sánchez-Bayo, F. and Wyckhuys, K.A.G., 2019. Worldwide decline of the entomofauna: A review of its drivers. *Biological Conservation*, 232, pp.8–27.

Schreiber, K.-F., 1997. Grundzüge der Sukzession in 20-jährigen Grünland-Bracheversuchen in Baden-Württemberg. *Forstwissenschaftliches Centralblatt*, 116(1–6), pp.243–258.

Schulz, T.T. and Leininger, W.C., 1990. Differences in Riparian Vegetation Structure between Grazed Areas and Exclosures. *Journal of Range Management*, 43(4), p.295.

Science for Environment Policy, 2017. *Agri-environmental schemes: how to enhance the agriculture-environment relationship*. *Thematic Isssue 57,* European Commission.

Scimone, M., Rook, A.J., Garel, J.P. and Sahin, N., 2007. Effects of livestock breed and grazing intensity on grazing systems: 3. Effects on diversity of vegetation. *Grass and Forage Science*, 62(2), pp.172–184.

Seaman, D.E. and Powell, R.A., 1996. An evaluation of the accuracy of kernel density estimators for home range analysis. *Ecology*, 77(7), pp.2075–2085.

Senft, R.L., Coughenour, M.B., Bailey, D.W., Rittenhouse, L.R., Sala, O.E. and Swift, D.M., 1987. Large Herbivore Foraging and Ecological Hierarchies. *BioScience*, 37(11), pp.789–799.

Senft, R.L., Rittenhouse, L.R. and Woodmansee, R.G., 1985. Factors influencing patterns of cattle grazing behavior on shortgrass steppe. *Journal of Range Management*, 38(1), pp.82–87.

Shi, J., Dunbar, R.I.M., Buckland, D. and Miller, D., 2003. Daytime activity budgets of feral goats (Capra hircus) on the Isle of Rum: influence of season, age, and sex. *Canadian Journal of Zoology*, 81(May), pp.803–815.

Silverman, B., 1986. Density estimation for statistics and data analysis. *Chapman and Hall*, 37(1), pp.1–22.

Skvarla, M.J., Larson, J.L. and Dowling, A.P.G., 2014. Pitfalls and Preservatives: A review. *Journal of the Entomological Society of Ontario*, 145.

Sleeman, A., McConnell, B. and Gatley, S., 2004. *Understanding Earth Processes, Rocks and the Geological History of Ireland*. Dublin: Geological Survey of Ireland.

Small, R.W., 2003. Conservation Grazing: delivering habitat management for conservation with livestock. 171, pp.38–44.

Smith, C.W., Aproot, A., Coppins, B.J., Fletcher, A., Gilbert, O.L., James, P.W. and Wosely, P.A., 2009. *Lichens of Great Britain and Ireland*. British Lichen Society.

Smith, M.A., Dodd, J.L., Rodgers, J.D. and Skinner, Q.D., 1992. Habitat selection by cattle along an ephemeral channel. *Journal Article Journal of Range Management J. Range Manage*, 45(4), pp.385-390

Smith, P.G., 2005. *Compos Analysis, version 6.2 standard [software]*.

Sottocornola, M., Laine, A., Kiely, G., Byrne, K.A. and Tuittila, E.S., 2009. Vegetation and environmental variation in an Atlantic blanket bog in South-western Ireland. *Plant Ecology*, 203(1), pp.69–81.

Southwood, T.R.E., 1977. Habitat, the templet for ecological strategies. *J. Anim. Ecol*, 46, pp.337–365.

Spake, R., Barsoum, N., Newton, A.C. and Doncaster, C.P., 2016. Drivers of the composition and diversity of carabid functional traits in UK coniferous plantations. *Forest Ecology and Management*, 359, pp.300–308.

Spalinger, L.C., Haynes, A.G., Schutz, M. and Risch, A.C., 2012. Impact of wild ungulate grazing on Orthoptera abundance and diversity in subalpine grasslands. *Insect Conservation and Diversity*, 5(6), pp.444–452.

Spehn, E., Rudmann-Maurer, K. and Korner, C., and Maselli, 2012 (eds.). *Mountain Biodiversity and global change*. GMBA - DIVERSITAS, Basel.

Stace, C., 2010. *New flora of the British Isles*. Third ed. Cambridge University Press.

Steinauer, E.M. and Collins, S.L., 2001. Feedback loops in ecological hierarchies following urine deposition in tallgrass prairie. *Ecology*, 82(5), pp.1319–1329.

Stephens, D.W. and Krebs, J.R., 1986. Foraging theory. Vol. 1. Princeton University Press.

Stephenson, M.B. and Bailey, D.W., 2017. *Do movement patterns of GPS-tracked cattle on extensive rangelands suggest independence among individuals? Agriculture, 7(7), p58.*

Stevens, J. (James P., 2002. *Applied multivariate statistics for the social sciences*. Routledge.

Stewart, G.B. and Pullin, A.S., 2008. The relative importance of grazing stock type and grazing intensity for conservation of mesotrophic 'old meadow' pasture. *Journal for Nature Conservation*, 16(3), pp.175–185.

Sullivan, C.A., Finn, J.A., Ó hÚallacháin, D., Green, S., Matin, S., Meredith, D., Clifford, B. and Moran, J., 2017. The development of a national typology for High Nature Value farmland in Ireland based on farm-scale characteristics. *Land Use Policy*, 67, pp.401–414.

Sweeney, J., 1997. Ireland. In: D. Wheeler and J. Mayes, eds. *Regional Climates of the British Isles*. London.pp.254–275.

Swihart, R.K. and Slade, N.A., 1985. Testing For Independence of Observations in Animal Movements. *Ecology*, 66(4), pp.1176–1184.

Szyszko, J., 1983. *State of Carabidae (Col.) fauna in fresh pine forest and tentative valorisation of this environment*. Wyndawn. SGGW-AR.

Tallis, J., 1998. Growth and degradation of British and Irish blanket mires. *Environmental Reviews*, 6, pp.81–122.

Tallowin, J.R.B., Rook, A.J. and Rutter, S.M., 2005. Impact of grazing management on biodiversity of grasslands. *Animal Science*, 81(2), pp.193–198.

Taulman, J.F. and Smith, K.G., 2004. Home range and habitat selection of southern flying squirrels in fragmented forests. *Mammalian Biology (69(1), pp.11-27.*

Taylor, K., Rowland, A.P. and Jones, H.E., 2001. Molinia caerulea (L.) Moench. *Journal of Ecology*, 89(1), pp.126–144.

Thiele, H.-U., 1977. *Carabid Beetles in Their Environments: a Study on Habitat Selection by Adaptations in Physiology and Behaviour*. Springer Berlin Heidelberg.

Thompson, D.B.A., Macdonald, A.J. and Hudson, P.J., 1995. Upland moors and heaths. In: *Managing Habitats for Conservation*. Cambridge: Cambridge University Press.pp.292–326.

Thompson, D.B.A., MacDonald, A.J., Marsden, J.H. and Galbraith, C.A., 1995. Upland heather moorland in Great Britain: A review of international importance, vegetation change and some objectives for nature conservation. *Biological Conservation*, 71(2), pp.163–178.

Thornton, B. and Millard, P., 1993. The effects of nitrogen supply and defoliation on the seasonal internal cycling of nitrogen in Molinia caerulea. *Journal of Experimental Botany*, 44(3), pp.531–536.

Tilman, D. and Downing, J.A., 1994. Biodiversity and stability in grasslands. *Nature,* 367(6461), pp.363-365

Tresset, A., 2003. French connections II: of cows and men. *Neolithic settlement in Ireland and Western Britain*, pp.18–30.

Tubridy, M., 2013. A Study to Identify Best Management of Upland Habitats in County Wicklow. Roundwood, Wicklow. Uplands Council.

Tufto, J., Andersen, R. and Linnell, J., 1996. Habitat Use and Ecological Correlates of Home Range Size in a Small Cervid: The Roe Deer. *The Journal of Animal Ecology*, 65(6), p.715.

United Nations, 1992. Convention on biological diversity. Available at: <https://www.cbd.int/doc/legal/cbd-en.pdf>

Usher, M.B. and Gardner, S.M., 1988. Animal communities in the uplands: how is naturalness influenced by management. *Ecological Change in the uplands*, pp.75–92.

Usher, M.B. and Thompson, D.B.A., 1993. Variation in the upland heathlands of Great Britain: Conservation importance. *Biological Conservation*, 66(1), pp.69–81.

Vanbergen, A.J., Woodcock, B.A., Watt, A.D. and Niemelii, J., 2005. Effect of land-use heterogeneity on carabid communities at the landscape scale. *Ecography*, (28), pp.3–16.

Vasey, N., 2004. Circadian rhythms in diet and habitat use in red ruffed lemurs (Varecia rubra) and white-fronted brown lemurs (*Eulemur fulvus albifrons*). *American Journal of Physical Anthropology*, 124(4), pp.353–363.

Vasey, N., 2005. Activity budgets and activity rhythms in red ruffed lemurs (Varecia rubra) on the Masoala Peninsula, Madagascar: seasonality and reproductive energetics. *American Journal of Primatology*, 66(1), pp.23–44.

Verrill, L. and Tipping, R., 2010. Use and abandonment of a Neolithic field system at Belderrig, Co. Mayo, Ireland: Evidence for economic marginality. *Holocene*, 20(7), pp.1011–1021.

Vickery, J.A., Tallowin, J.R., Feber, R.E., Asteraki, E.J., Atkinson, P.W., Fuller, R.J. and Brown, V.K., 2001. The management of lowland neutral grasslands in Britain: effects of agricultural practices on birds and their food resources. *Journal of Applied Ecology*, 38(3), pp.647–664.

Wagner, D.L., 2020. Insect Declines in the Anthropocene. *Annual Review of Entomology*, 65(1), pp.457–480.

Wahren, C.H.A., Papst, W.A. and Williams, R.J., 1994. Long-term vegetation change in relation to cattle grazing in sub-alpine grassland and heathland on the bogong high-

plains: An analysis of vegetation records from 1945 to 1994. *Australian Journal of Botany*, 42(6), pp.607–639.

Wallis-deVries, M.F.W., 1998. Large herbivores as key factors for nature conservation. In: M.F.W. Wallis-deVries, J.P. Bakker and S.E. Van Wieren, eds. *Grazing andn Conservation Management*. Dordrecht: Kluwer Academic Publishers.pp.1–20.

Wallis de Vries, M.F. and Schippers, P., 1994. Foraging in a landscape mosaic: selection for energy and minerals in free-ranging cattle. *Oecologia*, 100–100(1–2), pp.107–117.

WallisDeVries, M.F., 1998a. Habitat Quality and the Performance of Large Herbivores. In: M.F. WallisDeVries, J.P. Bakker and S.E. Van Wieren, eds. *Grazing and Conservation Management*. Kluwer Academic Publishers. p.304.

WallisDeVries, M.F., 1998b. Large herbivores as key factors for nature conservation. In: *Grazing and Conservation Management*. Dordrecht: Springer Netherlands.pp.1–20.

WallisDeVries, M.F., Noordijk, J., Colijn, E.O., Smit, J.T. and Veling, K., 2016. Contrasting responses of insect communities to grazing intensity in lowland heathlands. *Agriculture, Ecosystems and Environment*, 234, pp.72–80.

Walpole, M., Almond, R.E., Besançon, C., Butchart, S.H., Campbell-Lendrum, D., Carr, G.M., Collen, B., Collette, L., Davidson, N.C., Dulloo, E. and Fazel, A.M., 2009.

Tracking progress toward the 2010 biodiversity target and beyond. *Science*, *325*(5947), pp.1503-1504.

Walsh, G., 2017. *Cattle Breeds in Ireland: A History*. Wexford: The Borie Press

Welch, D., 1984. Studies in the Grazing of Heather Moorland in North-East Scotland. III. Floristics. *The Journal of Applied Ecology*, pp.209-225.

Welch, D. and Scott, D., 1995. Studies in the Grazing of Heather Moorland in Northeast Scotland. VI.20-Year Trends in Botanical Composition. *The Journal of Applied Ecology*, 32(3), p.596.

West, G.B., Brown, J.H. and Enquist, B.J., 1997. A general model for the origin of allometric scaling laws in biology. *Science*, 276(5309), pp.122–126.

Whittaker, R., 1965. Dominance and diversity in land plant communities: numerical relations of species express the importance of competition in community function and evolution. *Science,* 147(3655), pp.250-260.

Whittingham, M.J., 2007. Will agri-environment schemes deliver substantial biodiversity gain, and if not why not? *Journal of Applied Ecology*, 44(1), pp.1–5.

Van Wieren, S.E. and Bakker, J.P., 1998. Grazing for conservation in the twenty-first century. In: *Grazing and Conservation Management*. Dordrecht: Springer Netherlands.pp.349–363.

Wilde, W., 1862. On the ancient and modern races of Oxen in Ireland. *Proceedings of the Royal Irish Academy*, 1(7), pp.64-78,178.

Williams, B., Walls, S., Walsh, M. and Gormally, M., 2012b. Habitat Selection Byn Grazing Animals in Heterogeneous Environments: the Case of Hill Sheep in western

Ireland. In *Biology and Environment: Proceedings of the Royal Irish Academy* (pp. 267-283). Royal Irish Academy.

Williams, B., Walls, S., Walsh, M., Gormally, M., Bleasdale, A. and others, 2009. Proposing An Efficient Indicator Of Grazer Distribution On Heterogeneous Hill Vegetation. *Applied Ecology and Environmental Research*, 7(4), pp.341–358.

Williams, B., Walsh, M., Gormally, M., Walls, S. and Sheahan, J., 2010. Resource selection by hill sheep: direct flock observations versus GPS tracking. *Applied Ecology and Environmental Research*, 8(4), pp.279–300.

Williams, C.D. and Gormally, M.J., 2010. The effects of blanket bog management on ground beetles (Carabidae) with particular reference to the threatened Carabus clatratus L. *Irish Wildlife Manuals*, 47(47), pp.1–74.

Wilson, J., 1909. The origin of the Dexter-Kerry breed of cattle. In: *Scientific Proceedings of the Royal Dublin Society*. pp.1–17. Wilson, J.B., 1991. Methods for fitting dominance/diversity curves. *Journal of Vegetation Science*, 2(1), pp.35–46.

Woodcock, B.A., McCauseland, K.A., Mann, D.J. and McGavin, G.C., 2004. Management of Irish oceanic blanket bog and its effect on ground beetles: implications for the conservation of the threatened *Carabus clathratus* L. (Coleoptera: Carabidae). *Bulletin Irish biogeographical Society*, (28), pp.63–84.

Woodcock, B.A., Pywell, R.F., Roy, D.B., Rose, R.J. and Bell, D., 2005. Grazing management of calcareous grasslands and its implications for the conservation of beetle communities. *Biological Conservation*, 125(2), pp.193–202.

Woodman, P., 2016. The Introduction of prehistoric cattle into Ireland: Fresh Perspectives. In: M. O' Connell, F. Kelly and J. McAdam, eds. *Cattle in Ancient and Modern Ireland: Farming Practicies, Environment and Economy*. Newcastle upon Tyne: Cambridge Scholars Publishing.

Woodman, P., Mccarthy, M. and Monaghan, N., 1997. The Irish quaternary fauna project. *Quaternary Science Reviews*, 16(2), pp.129–159.

Worton, B.J., 1989a. Kernel methods for estimating the utilisation distribution in home-range studies. *Ecology*, 70(1), pp.164–168.

Worton, B.J., 1989b. Kernel methods for estimating the utilization distribution in home-range studies. *Ecology*, 70(1), pp.164–168.

Worton, B.J., 1995. Using Monte Carlo simulation to evaluate kernel-based home range estimators. *The Journal of wildlife management*, pp.794–800.

Wright, I.A.A., Heritage, S.N., Pakeman, R.J., Dennis, P., Dalziel, A.J. and Milne, J.A., 2006. The effects of cattle on the natural heritage of Scotland. *Scottish Natural Heritage Commissioned Report*, 203(203).

Zalewski, M. and Ulrich, W., 2006. Dispersal as a key element of community structure: the case of ground beetles on lake islands. *Diversity and Distibutions,* 12(6), pp.767–775.

Zhang, E., 2000. Daytime activity budgets of the Chinese water deer. *Mammalia*, 64(2), pp.163–172.

Ziesemer, F. and Meyburg, B.U., 2015. Home range, habitat use and diet of Honey-buzzards during the breeding season. *British Birds*, 108(8), pp.467–481.

Zimmerman, G.M., Goetz, H. and Mielke, P.W., 2006. Use of an Improved Statistical Method for Group Comparisons to Study Effects of Prairie Fire. *Ecology*, 66(2), pp.606–611.

Appendices

Appendix I:

Structure of Fossitt (2000) Habitat Classification for Ireland

Level 1		Level 2		Level 3	
F	Freshwater	FL	Lakes and Ponds	FL1	Dystrophic Lakes
				FL2	Acid oligotrophic lakes
				FL3	Limestone/Marl lakes
				FL4	Mesotrophic lakes
				FL5	Eutrophic lakes
				FL6	Turloughs
				FL7	Reservoirs
				FL8	Other artificial lakes and ponds
		FW	Watercourses	FW1	Eroding/upland rivers
				FW2	Depositing/lowland rivers
				FW3	Canals
				FW4	Drainage ditches
		FP	Springs	FP1	Calcareous springs
				FP2	Non-calcareous springs
		FS	Swamps	FS1	Reed and large sedge swamps
				FS2	Tall-herb swamps
G	Grassland and Marsh	GA	Improved Grassland	GA1	Improved agricultural grassland
				GA2	Amenity grassland (improved
		GS	Semi-natural grassland	GS1	Dry calcareous and neutral grassland
				GS2	Dry meadows and grassy verges
				GS3	Dry-humid acid grassland
				GS4	Wet grassland
		GM	Freshwater Marsh	GM1	Marsh
H	Heath and Dense Bracken	HH	Heath	HH1	Dry Siliceous heath
				HH2	Dry Calcareous heath
				HH3	Wet heath
				HH4	Montane Heath
		HD	Dense bracken	DH1	Dense Bracken
P	Peatlands	PB	Bogs	PB1	Raised bog
				PB2	Upland blanket bog
				PB3	Lowland blanket bog
				PB4	Cutover bog
				PB5	Eroding blanket bog

Provisional classification of upland habitats (Perrin et al., 2014a)

Habitats	Provisional communities and sub-communities	Code	Annex I	Fossitt
Lakes and pools	*Menyanthes trifoliata - Carex limosa* pool community			
	- infilling pool sub-community	PO1a	7140	PF3
	- aquatic sub-community	PO1b	3160	FL1
	Littorella uniflora – Lobelia dortmanna lake community			
	- upland variant	PO2i	3130	FL2
	- lowland variant	PO211	3110	FL2
Soakaways	*Potamogeton polygonifolius* soakway	SW1	-	PF2
Springs	*Philonotis fontana - Saxifraga stellaris* spring			
	- typical sub-community	SPG1a	-	FP2
	- species-poor *Sphagnum denticulatum* sub-community	SPG1b	-	FP2
	Palustriella commutata spring			
	- Annex I variant	SPG2i	7220	FP1
	- non-Annex I variant	SPG2ii	-	FP1
	Anthelia julacea - Sphagnum inundatum spring	SPG3	-	FP2
Poor flushes	*Carex nigra/echinata - Sphagnum denticulatum* flush	PFLU1	-	PF2
	Juncus effusus - Sphagnum cuspidatum/palustre flush	PFLU2	-	PF3
	Juncus acutiflorus/effusus - Calliergonella cuspidata flush	PFLU3	-	GS4
	Molinia caerulea - Sphagnum palustre flush			
	- typical sub-community	PFLU4a	-	PF2
	- *Erica erigena* sub-community	PFLU4b	-	PF2
	Carex rostrata – Sphagnum spp. flush	PFLU5	-	PF3
Calcareous or mineral-rich flushes and fens	*Carex viridula oedocarpa - Pinguicula vulgaris - Juncus bulbosus* flush	RFLU1a		
	brown moss sub-community	RFLU1b	7230	PF1
	species-poor-sub-community	RFLU2	-	PF1
	Eleocharis quinqueflora - Carex viridula flush	RFLU3	7230	PF1
	Carex panicea - Carex viridula subsp. oedocarpa flush	RFLU4	-	PF1
	Schoenus nigricans – Scorpidium scorpioides flush		7230	PF1
	Carex rostrata fen			
	brown moss sub-community	RFEN1a	7230	PF1
	species-poor sub-community	RFEB1b	7140	PF3
Upland Grasslands	*Agrostis capillaris - ovina upland* grassland			
	typical sub-community	UG1a	-	GS3
	Sphagnum spp. sub-community	UG1b	-	GS3
	species-rich calcareous sub-community	UG1c	*6230	GS3
	Juncus squarrosus sub-community	UG1d	-	GS3
	species-rich non-calcareous sub-community	UG1e	*6230	GS3
	Nardus stricta - Galium saxatile upland grassland			
	typical sub-community	UG2a	-	GS3
	Sphagnum spp. sub-community	UG2b	-	GS3
	species-rich sub-community	UG2c	*6230	GS3

	Juncus squarrosus sub-community	UG2d	-	GS3
	species-rich non-calcareous sub-community	UG2e	*6230	GS3
	Silene acaulis alpine grassland	UG3	6170	GS1
	Molinia caerulea – Anthoxanthum odoratum wet grassland	UG4	-	GS4
	Festuca ovina – Agrostis capillaris - Thymus praecox calcareous grassland			
	herb-rich sub-community	UG5a		GS1
	herb-poor sub-community	UG5b		GS1
	Sesleria caerulea – Carex flacca calcareous grassland	UG6		GS1
Bracken	*Pteridium aquilinum* community	BK1	-	GS1
Dry Heaths	*Ulex gallii - Erica cinerea* dry heath	DH1	4030	HH1
	Calluna vulgaris - Erica erigena - Molinia caerulea dry heath	DH2	4030	HH1
	Calluna vulgaris - Erica cinerea dry heath	DH3	4030	HH1
	Calluna vulgaris - Sphagnum capillifolium dry /damp heath	DH4	4030	HH1
	Calluna vulgaris – Antennaria dioica dry heath	DH5	4030	HH2
	Calluna vulgaris -*Vaccinium myrtillus* dry heath	DH6	4030	HH1
Wet Heath	*Schoenus nigricans - Erica tetralix* wet heath	WH1A	4010	HH3
	continuous cover sub-community	WH1B	4010	HH3
	open sub-community	WH2	4010	HH3
	Trichophorum germanicum - Cladonia spp. - Racomitrium lanuginosum wet heath	WH3	4010	HH3
	Calluna vulgaris - Molinia caerulea - Sphagnum capillifolium wet/damp heath		4010	HH3
	Trichophorum germanicum- Eriophorum angustifolium wet heath		4010	HH3
	typical sub-community	WH4a	4010	HH3
	Calluna vulgaris sub-community	WH4b	4010	HH3
	Juncus squarrosus sub-community	WH4C	4010	HH3
	Trichophorum germanicum - Nardus stricta - Racomitrium lanuginosum montane wet heath	WH5	4010	HH3
	Schoenus nigricans – Molinia caerulea – Myrica gale wet heath	WH6	4010	HH3
	Molinia caerulea – Ulex gallii wet heath	WH7	4010	HH3
Montane Heaths	*Calluna vulgaris - Racomitrium lanuginosum* montane heath			
	typical sub-community	MH1a	4060	HH4
	Juncus squarrosus sub-community	MH1b	4061	HH4
	Vaccinium myrtillus - Racomitrium lanuginosum - Herbertus aduncus montane heath	MH2	4062	HH4
	Vaccinium myrtillus - Rhytidiadelphus loreus - Anthoxanthum odoratum montane heath	MH3	4063	HH4
	Calluna vulgaris - Juniperus communis subsp. *nana* montane heath	MH4	4064	HH4
	Nardus stricta - Carex binervis - Racomitrium lanuginosum montane grass-heath	MH5	-	HH4
	Carex bigelowii - Racomitrium lanuginosum montane vegetation			HH4
	typical sub-community	MH6A	6150	HH4
	Dicranum fuscescens sub-community	MG6B	6150	HH4
	Juncus squarrosus sub-community	MH6C	6150	HH4
	Deschampsia flexuosa sub-community	MH6D	6150	HH4

	Nardus stricta - Carex bigelowii montane vegetation		6150	HH4
	typical sub-community	MGH7A	6150	HH4
	Anthoxanthum odoratum sub-community	MH7B	6150	HH4
	Juncus squarrosus sub-community	MH7C	6150	HH4
	Festuca vivipara – Thymus polytrichus – Galium saxatile montane vegetation	MH8	-	HH4
Blanket Bogs	*Schoenus nigricans - Eriophorum angustifolium* bog			
	continuous cover sub-community	BB1A	*7130	PB3
	open sub-community	BB1B	*7130	PB3
	Schoenus nigricans – Sphagnum spp. bog	BB2	*7130	PB3
	Eriophorum vaginatum – Sphagnum papillosum bog	BB3	*7130	PB2
	Trichophorum germanicum - Eriophorum angustifolium bog	BB4	*7130	PB2
	Calluna vulgaris - Eriophorum spp. bog			
	typical sub-community	BB5A	*7130	PB2
	Juncus squarrosus sub-community	BB5B	*7130	PB2
	Eriophorum angustifolium - Juncus squarrosus bog			
	typical sub-community	BB6A	*7130	PB2
	arctic-alpine sub-community	BB6B	*7130	PB2
	Eriophorum angustifolium – Sphagnum austinii bog	BB7	*7130	PB3
Hollows	*Sphagnum denticulatum/cuspidatum* hollow	HW1i		
	upland variant	HW1ii	*7130	PB2
	lowland variant	HW1ii	*7130	PB3
	flush variant	HW1iii	-	PF2
	Eriophorum angustifolium - Sphagnum fallax hollow			
	upland variant	HW2i	7130	PB2
	lowland variant	HW2ii	7130	PB3
	Rhynchospora alba hollow	HW3	7150	PB3
	Eleocharis multicaulis hollow			
	bog variant	HW4II	-	PB3
	flush variant	HW4ii	-	PF2
Degraded peat	*Campylopus introflexus - Polytrichum* spp. degraded peat community	DP1	-	ED3
	Nardus stricta – Eriophorum angustifolium degraded peat community	DP2	-	PB5
Tall herbs	*Luzula sylvatica - Vaccinium myrtillus* tall herb vegetation			
	rock face variant	TH1i	-	ER1
	dry heath variant	TH1ii	-	HH1
	Cochlearia pyrenaica tall herb vegetation	TH2	6430	ER2
	Sedum rosea - Angelica sylvestris tall herb vegetation	TH3	6430	ER2
Siliceous scree community	Siliceous scree community	SC1	8110	ER3
	Calcareous scree community	SC2	8120	ER4
Rock clefts and rocky slopes	*Saxifraga spathularis - Asplenium adiantum-nigrum* rock cleft community	RS1	8220	ER1
	Saxifraga aizoides - Asplenium spp. - Orthothecium rufescens rock cleft community	RS2	8210	ER2
Hepatic mats	*Calluna vulgaris - Scapania gracilis* hepatic mat			
	non-Annex I grassland variant	HM1i	-	GS3

	Annex I grassland variant	HM1ii	6230	GS3
	dry heath variant	HM1iii	4030	HH1
	wet heath variant	HM1iv	4010	HH3
	montane heath variant	HM1v	4060	HH4
	non-Annex I siliceous rock face variant	HM1vi	-	ER1
	Annex I rock face variant	HM1vii	8220	ER1
	siliceous scree variant	HM1viii	8110	ER3
	upland blanket bog variant	HM1ix	*7130	PB2
	lowland blanket bog variant	HM1x	*7130	PB3
Non-vegetation cover types				
Loose rock	- siliceous	SilcLoose	-	ER3
	- calcareous	CalcLoose	-	ER4
Scree	- siliceous	SilcScree	8110	ER3
	- calcareous	CalcScree	8120	ER4
Bed rock	- siliceous non-Annex	SilcRockN	-	ER1
	- calcareous non-Annex	CalcRockN	-	ER2
	- siliceous Annex	SilcRockA	8220	ER1
	- calcareous Annex	CalcRockA	8210	ER2
	- limestone pavement	LimePave	*8240	ER2
Bare peat	- eroding bog	BarePeatB	-	PB5
	- other	BarePeatO	-	ED2
Open water	- dystrophic lakes and pools non-Annex	OpenDN	-	FL2
	- dystrophic lakes and pools Annex	OpenDA	3160	FL1
	- lowland oligotrophic lakes and pools non-Annex	OpenLN	-	FL2
	- lowland oligotrophic lakes and pools Annex	OpenLA	3110	FL2
	- upland oligotrophic lakes and pools non-Annex	OpenUN	-	FL2
	- upland oligotrophic lakes and pools Annex	OpenUA	3130	FL2
Running water	- upland non-Annex	RunUN	-	FW1
	- upland Annex	RunUA	3260	FW1
	- lowland non-Annex	RunLN	-	FW2
	- lowland Annex	RunLA	3260	FW2
Gravel		Gravel	-	ED1
Sand		Sand	-	ED1
Till		Till	-	ED1
Road		Road	-	BL3
Made ground		Made	-	BL3
Stone wall		Stonewall	-	BL1
Bare soil		Baresoil	-	ED2

Appendix II: Conservation status assessment data

Wet Heath

Plot 1: u1 (Wet Heath Exclosure)						
	2013		2014		2015	
	Mean	*s*	Mean	*s*	Mean	*s*
Erica tetralix present	0.9	0.2	0.9	0.2	1.0	0.0
Cover of positive indicator species ≥ 50%	78.6	38.2	92.2	21.1	73.7	35.2
Tot. cover of: *Cladonia* sp. *Sphagnum* sp. *R. lanuginosum* and pleurocarp mosses ≥ 10%	47.7	29.6	29.4	10.2	30.2	22.1
Cover of ericoid species ≥ 15%	13.4	7.6	24.4	11.2	21.9	16.5
Cover of dwarf shurb species < 75%	13.4	7.6	24.4	11.2	22.3	16.9
Cover of neg. indicators: *A. capillaris, H. lanatus, R. repens*, collectively < 1%	0.1	0.5	0.0	0.0	0.1	0.5
Cover of non-native species < 1%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of non-native species < 1%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of *Pteridium aquilinium* < 10%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of *J. effusus* < 10%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of disturbed, bare ground < 10%	1.5	3.1	0.0	0.0	0.7	1.4

Plot 3: g3 (Wet Heath 15)						
	2013		2014		2015	
Erica tetralix present	1.0	0.0	1.0	0.0	1.0	0.0
Cover of positive indicator species ≥ 50%	85.7	36.2	119.3	18.4	86.8	23.6
Tot. cover of: *Cladonia* sp. *Sphagnum* sp. *R. lanuginosum* and pleurocarp mosses ≥ 10%	21.7	27.6	52.0	16.5	32.8	17.0
Cover of ericoid species ≥ 15%	27.9	14.0	35.6	14.2	34.1	15.5
Cover of dwarf shurb species < 75%	27.9	14.0	35.6	14.2	34.1	15.5
Cover of neg. indicators: *A. capillaris, H. lanatus, R. repens*, collectively < 1%	0.6	1.5	0.0	0.0	0.0	0.0
Cover of non-native species < 1%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of non-native species < 1%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of *Pteridium aquilinium* < 10%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of *J. effusus* < 10%	0.7	2.1	0.0	0.0	0.0	0.0
Cover of disturbed, bare ground < 10%	0.0	0.0	0.1	0.2	3.6	6.4

Plot 4: g2 (Wet Heath 60)						
	2013		2014		2015	
	Mean	*s*	Mean	*s*	Mean	*s*
Erica tetralix present	1.0	0.0	1.0	0.0	1.0	0.0
Cover of positive indicator species ≥ 50%	87.1	26.0	73.6	20.6	75.9	16.2
Tot. cover of: *Cladonia* sp. *Sphagnum* sp. *R. lanuginosum* and pleurocarp mosses ≥ 10%	39.1	20.4	24.0	14.1	33.3	13.3
Cover of ericoid species ≥ 15%	25.9	12.1	29.7	13.4	27.9	9.1
Cover of dwarf shurb species < 75%	26.6	12.6	32.2	13.1	29.9	10.1
Cover of neg. indicators: *A. capillaris, H. lanatus, R. repens*, collectively < 1%	0.0	0.1	0.0	0.0	0.0	0.0
Cover of non-native species < 1%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of non-native species < 1%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of *Pteridium aquilinium* < 10%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of *J. effusus* < 10%	0.0	0.0	0.5	1.3	0.1	0.5
Cover of disturbed, bare ground < 10%	5.3	8.8	0.7	1.5	1.0	2.6

Plot 2: u2 (Wet Heath Virtual Exclosure)						
	2013		2014		2015	
	Mean	*s*	Mean	*s*	Mean	*s*
Erica tetralix present	na	na	0.9	0.2	1.0	0.0
Cover of positive indicator species ≥ 50%	na	na	92.2	21.1	90.1	19.7
Tot. cover of: *Cladonia* sp. *Sphagnum* sp. *R. lanuginosum* and pleurocarp mosses ≥ 10%	na	na	29.4	10.2	41.0	14.1
Cover of ericoid species ≥ 15%	na	na	24.4	11.2	17.1	5.6
Cover of dwarf shurb species < 75%	na	na	24.4	11.2	17.1	5.6
Cover of neg. indicators: *A. capillaris, H. lanatus, R. repens*, collectively < 1%	na	na	0.0	0.0	0.7	1.7
Cover of non-native species < 1%	na	na	0.0	0.0	0.0	0.0
Cover of non-native species < 1%	na	na	0.0	0.0	0.0	0.0
Cover of *Pteridium aquilinium* < 10%	na	na	0.0	0.0	0.0	0.0
Cover of *J. effusus* < 10%	na	na	0.0	0.0	0.0	0.0
Cover of disturbed, bare ground < 10%	na	na	0.0	0.0	0.0	0.0

Plot 5: u3 (Dry Heath Exclosure reclassified samples)						
	2013		2014		2015	
	Mean	*s*	Mean	*s*	Mean	*s*
Erica tetralix present	1.0	0.0	1.0	0.0	1.0	0.0
Cover of positive indicator species ≥ 50%	92.1	18.7	84.2	19.7	65.2	15.0
Tot. cover of: *Cladonia* sp. *Sphagnum* sp. *R. lanuginosum* and pleurocarp mosses ≥ 10%	33.9	19.7	32.0	11.8	17.1	9.5
Cover of ericoid species ≥ 15%	41.9	18.9	40.1	8.2	43.8	10.8
Cover of dwarf shurb species < 75%	44.1	20.7	42.1	8.0	51.3	15.2
Cover of neg. indicators: *A. capillaris, H. lanatus, R. repens*, collectively < 1%	1.5	3.2	6.2	5.5	4.3	3.6
Cover of non-native species < 1%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of non-native species < 1%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of *Pteridium aquilinium* < 10%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of *J. effusus* < 10%	0.0	0.0	0.2	0.4	0.0	0.0
Cover of disturbed, bare ground < 10%	0.0	0.0	0.0	0.0	0.0	0.0

Plot 11: g5 (BB 51 reclassified samples)						
	2013		2014		2015	
	Mean	s	Mean	s	Mean	s
Erica tetralix present	1.0	0.0	1.0	0.0	1.0	0.0
Cover of positive indicator species ≥ 50%	86.6	25.9	115.4	34.4	100.6	16.8
Tot. cover of: *Cladonia* sp. *Sphagnum* sp. *R. lanuginosum* and pleurocarp mosses ≥ 10%	41.7	20.1	38.6	31.7	45.6	19.4
Cover of ericoid species ≥ 15%	28.5	11.5	50.8	13.9	31.6	9.2
Cover of dwarf shurb species < 75%	28.5	11.5	50.8	13.9	31.6	9.2
Cover of neg. indicators: *A. capillaris, H. lanatus, R. repens*, collectively < 1%	0.0	0.0	1.3	2.2	0.0	0.0
Cover of non-native species < 1%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of non-native species < 1%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of *Pteridium aquilinium* < 10%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of *J. effusus* < 10%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of disturbed, bare ground < 10%	6.7	8.7	0.0	0.0	2.4	1.6

Plot 12: g6 (BB 02 reclassified samples)						
	2013		2014		2015	
	Mean	s	Mean	s	Mean	s
Erica tetralix present	na	na	1.0	0.0	1.0	0.0
Cover of positive indicator species ≥ 50%	na	na	92.2	23.4	97.9	30.0
Tot. cover of: *Cladonia* sp. *Sphagnum* sp. *R. lanuginosum* and pleurocarp mosses ≥ 10%	na	na	31.3	19.6	29.0	17.7
Cover of ericoid species ≥ 15%	na	na	21.4	13.9	24.2	19.4
Cover of dwarf shurb species < 75%	na	na	21.4	13.9	24.2	19.4
Cover of neg. indicators: *A. capillaris, H. lanatus, R. repens*, collectively < 1%	na	na	0.0	0.0	0.2	0.6
Cover of non-native species < 1%	na	na	0.0	0.0	0.0	0.0
Cover of non-native species < 1%	na	na	0.0	0.0	0.0	0.0
Cover of *Pteridium aquilinium* < 10%	na	na	0.0	0.0	0.0	0.0
Cover of *J. effusus* < 10%	na	na	0.0	0.0	0.0	0.0
Cover of disturbed, bare ground < 10%	na	na	7.6	15.6	2.6	5.4

Plot 16: g9 (WG 69 reclassified samples)						
	2013		2014		2015	
	Mean	s	Mean	s	Mean	s
Erica tetralix present	1.0	0.0	0.9	0.3	0.6	0.5
Cover of positive indicator species ≥ 50%	46.1	16.9	33.2	11.5	35.7	16.3
Tot. cover of: *Cladonia* sp. *Sphagnum* sp. *R. lanuginosum* and pleurocarp mosses ≥ 10%	12.0	5.8	16.8	6.8	18.3	13.9
Cover of ericoid species ≥ 15%	20.0	12.9	6.3	6.7	8.9	10.7
Cover of dwarf shurb species < 75%	20.0	12.9	6.3	6.7	8.9	10.8
Cover of neg. indicators: *A. capillaris, H. lanatus, R. repens*, collectively < 1%	1.6	2.5	0.9	1.8	0.4	1.3
Cover of non-native species < 1%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of non-native species < 1%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of *Pteridium aquilinium* < 10%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of *J. effusus* < 10%	1.0	2.6	1.1	2.8	4.0	7.7
Cover of disturbed, bare ground < 10%	0.4	0.6	1.3	1.5	2.4	2.7

Plot 17: g10 (WG 63 reclassified samples)						
	2013		2014		2015	
	Mean	s	Mean	s	Mean	s
Erica tetralix present	1.0	0.0	0.9	0.3	0.9	0.3
Cover of positive indicator species ≥ 50%	54.6	24.3	52.4	13.5	53.8	22.9
Tot. cover of: *Cladonia* sp. *Sphagnum* sp. *R. lanuginosum* and pleurocarp mosses ≥ 10%	40.1	24.8	31.4	11.4	31.6	21.7
Cover of ericoid species ≥ 15%	6.3	6.1	7.2	7.3	9.6	6.5
Cover of dwarf shurb species < 75%	6.3	6.1	7.2	7.3	11.2	8.3
Cover of neg. indicators: *A. capillaris, H. lanatus, R. repens*, collectively < 1%	1.4	2.1	1.0	1.5	0.4	0.7
Cover of non-native species < 1%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of non-native species < 1%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of *Pteridium aquilinium* < 10%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of *J. effusus* < 10%	0.4	1.2	1.4	1.7	0.1	0.2
Cover of disturbed, bare ground < 10%	1.2	2.5	6.0	7.8	5.3	5.4

Dry Heath

Plot 5: u3 (DH Exclosure)						
	2013		2014		2015	
	Mean	s	Mean	s	Mean	s
Number of bryophyte or non-crustose lichen species present excl. *Campylopus* spp. and *Polytrichum* spp. ≥ 3	7.0	0.8	7.6	1.5	5.3	2.2
Number of positive indicator species present ≥ 2	2.8	0.5	2.6	0.5	2.5	0.5
Cover of positive indicator species ≥ 50%	60.0	16.8	51.1	12.9	52.8	29.1
Proportion of dwarf shrub cover composed of: *M. gale, S. repens* and *U. gallii* collectively < 50%	25.3	29.7	11.1	13.4	10.2	18.1
Total cover of weedy neg. indicator sp.: *Cirsium* sp., *R. repens, R. acetosa, U. dioica, S. jacobea* < 1%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of non-native species < 1%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of non-native species < 1%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of scattered native trees and scrub < 20%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of *P. aquilinium* <10%	0.0	0.0	0.0	0.0	0.0	0.0
Total cover of the negative indicator species	0.0	0.0	0.0	0.0	0.0	0.0
Cover of *J. effusus* < 10%	0.0	0.0	0.2	0.7	0.0	0.0
Cover of disturbed, bare ground < 10%	0.0	0.0	0.0	0.0	0.2	0.4

Plot 7: u4 (DH 47)				
	2013		2014	
	Mean	s	Mean	s
Number of bryophyte or non-crustose lichen species present excl. *Campylopus* spp. and *Polytrichum* spp. ≥ 3	7.9	1.4	5.3	1.3
Number of positive indicator species present ≥ 2	1.9	0.3	2.0	0.0
Cover of positive indicator species ≥ 50%	64.2	23.1	54.8	26.1
Proportion of dwarf shrub cover composed of: *M. gale, S. repens* and *U. gallii* collectively < 50%	0.0	0.0	0.0	0.0
Total cover of weedy neg. indicator sp.: *Cirsium* sp., *R. repens, R. acetosa, U. dioica, S. jacobea* < 1%	0.0	0.0	0.0	0.0
Cover of non-native species < 1%	0.0	0.0	0.0	0.0
Cover of non-native species < 1%	0.0	0.0	0.0	0.0
Cover of scattered native trees and scrub < 20%	0.0	0.0	0.0	0.0
Cover of *P. aquilinium* <10%	0.0	0.0	0.0	0.0
Total cover of the negative indicator species	0.0	0.0	0.0	0.0
Cover of *J. effusus* < 10%	0.2	0.5	0.1	0.3
Cover of disturbed, bare ground < 10%	1.0	1.9	0.4	1.0

Plot 6: u4 (DH Virtural Exclosure)						
	2013		2014		2015	
	Mean	s	Mean	s	Mean	s
Number of bryophyte or non-crustose lichen species present excl. *Campylopus* spp. and *Polytrichum* spp. ≥ 3	na	na	3.3	1.2	5.0	1.4
Number of positive indicator species present ≥ 2	na	na	2.7	0.5	2.7	0.5
Cover of positive indicator species ≥ 50%	na	na	75.3	21.2	92.2	5.4
Proportion of dwarf shrub cover composed of: *M. gale, S. repens* and *U. gallii* collectively < 50%	na	na	18.3	25.1	33.9	33.0
Total cover of weedy neg. indicator sp.: *Cirsium* sp., *R. repens, R. acetosa, U. dioica, S. jacobea* < 1%	na	na	0.0	0.0	0.0	0.0
Cover of non-native species < 1%	na	na	0.0	0.0	0.0	0.0
Cover of non-native species < 1%	na	na	0.0	0.0	0.0	0.0
Cover of scattered native trees and scrub < 20%	na	na	0.0	0.0	0.0	0.0
Cover of *P. aquilinium* <10%	na	na	0.0	0.0	0.0	0.0
Total cover of the negative indicator species	na	na	0.0	0.0	0.0	0.0
Cover of *J. effusus* < 10%	na	na	0.3	0.7	0.0	0.0
Cover of disturbed, bare ground < 10%	na	na	0.0	0.0	0.1	0.5

Plot 8: g4 (DH 04)						
	2013		2014		2015	
	Mean	s	Mean	s	Mean	s
Number of bryophyte or non-crustose lichen species present excl. *Campylopus* spp. and *Polytrichum* spp. ≥ 3	na	na	3.3	1.2	5.0	1.4
Number of positive indicator species present ≥ 2	na	na	2.7	0.5	2.7	0.5
Cover of positive indicator species ≥ 50%	na	na	75.3	21.2	92.2	5.4
Proportion of dwarf shrub cover composed of: *M. gale, S. repens* and *U. gallii* collectively < 50%	na	na	18.3	25.1	33.9	33.0
Total cover of weedy neg. indicator sp.: *Cirsium* sp., *R. repens, R. acetosa, U. dioica, S. jacobea* < 1%	na	na	0.0	0.0	0.0	0.0
Cover of non-native species < 1%	na	na	0.0	0.0	0.0	0.0
Cover of non-native species < 1%	na	na	0.0	0.0	0.0	0.0
Cover of scattered native trees and scrub < 20%	na	na	0.0	0.0	0.0	0.0
Cover of *P. aquilinium* <10%	na	na	0.0	0.0	0.0	0.0
Total cover of the negative indicator species	na	na	0.0	0.0	0.0	0.0
Cover of *J. effusus* < 10%	na	na	0.3	0.7	0.0	0.0
Cover of disturbed, bare ground < 10%	na	na	0.0	0.0	0.1	0.5

Plot 9: u5 (BB Exclosure)						
	2013		2014		2015	
	Mean	s	Mean	s	Mean	s
Number of positive indicator species ≥ 7	8.7	0.7	7.5	0.9	8.9	0.7
Cover of bryophyte or lichen species > 10%	76.4	16.2	84.9	11.9	86.2	11.3
Cover of bryophyte or lichen species, excl. *S. fallax* ≥ 10%	76.4	16.2	84.9	11.9	86.2	11.3
Cover of each of the following spp. < 75%:						
C. vulgaris	11.8	8.2	15.8	7.3	16.9	9.5
E. vaginatum	0.0	0.0	1.8	0.8	6.9	2.7
M. caerulea	52.7	16.2	36.7	6.5	49.5	12.8
T. germanicum	7.9	6.8	8.0	7.1	10.9	7.4
Total cover of neg. indicator spp.: (*A. capillaris, H. lanatus, P. aquilinium, R. repens*) < 1%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of non-native species < 1%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of non-native species < 1%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of bare ground < 10% (relevé)	0.0	0.0	0.0	0.0	0.0	0.0
Cover of bare ground < 10% (vicinity)	0.0	0.0	0.0	0.0	0.0	0.0

Plot 11: g5 (BB 51)						
	2013		2014		2015	
	Mean	s	Mean	s	Mean	s
Number of positive indicator species ≥ 7	8.6	1.8	6.4	1.6	7.5	1.4
Cover of bryophyte or lichen species > 10%	66.0	36.1	40.7	24.4	55.5	16.0
Cover of bryophyte or lichen species, excl. *S. fallax* ≥ 10%	65.8	36.3	40.7	24.4	55.5	16.0
Cover of each of the following spp. < 75%:						
C. vulgaris	37.0	17.2	40.5	16.5	43.0	19.3
E. vaginatum	10.5	10.6	5.4	3.4	5.0	3.1
M. caerulea	26.0	14.3	8.8	11.3	14.9	14.2
T. germanicum	12.0	10.0	14.9	16.8	5.2	6.3
Total cover of neg. indicator spp.: (*A. capillaris, H. lanatus, P. aquilinium, R. repens*) < 1%	0.0	0.0	0.4	0.8	0.0	0.0
Cover of non-native species < 1%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of non-native species < 1%	0.0	0.0	0.0	0.0	0.0	0.0
Cover of bare ground < 10% (relevé)	0.0	0.0	0.0	0.0	1.7	1.9
Cover of bare ground < 10% (vicinity)	0.0	0.0	0.0	0.0	1.7	1.9

Plot 10: u6 (BB Virtual Exlosure)						
	2013		2014		2015	
	Mean	s	Mean	s	Mean	s
Number of positive indicator species ≥ 7	na	na	6.3	0.8	7.5	1.2
Cover of bryophyte or lichen species > 10%	na	na	46.5	23.7	51.5	21.9
Cover of bryophyte or lichen species, excl. *S. fallax* ≥ 10%	na	na	46.5	23.7	51.5	21.9
Cover of each of the following spp. < 75%:	na	na				
C. vulgaris	na	na	17.9	8.7	13.0	7.7
E. vaginatum	na	na	2.6	2.0	2.5	1.8
M. caerulea	na	na	41.7	15.9	34.9	16.8
T. germanicum	na	na	22.3	13.0	17.6	13.4
Total cover of neg. indicator spp.: (*A. capillaris, H. lanatus, P. aquilinium, R. repens*) < 1%	na	na	0.2	0.8	0.0	0.0
Cover of non-native species < 1%	na	na	0.0	0.0	0.0	0.0
Cover of non-native species < 1%	na	na	0.0	0.0	0.0	0.0
Cover of bare ground < 10% (relevé)	na	na	0.0	0.0	0.0	0.0
Cover of bare ground < 10% (vicinity)	na	na	0.0	0.0	0.0	0.0

Plot 12: g6 (BB 02)						
	2013		2014		2015	
	Mean	s	Mean	s	Mean	s
Number of positive indicator species ≥ 7	na	na	na	na	7.0	1.0
Cover of bryophyte or lichen species > 10%	na	na	na	na	55.8	10.1
Cover of bryophyte or lichen species, excl. *S. fallax* ≥ 10%	na	na	na	na	55.8	10.1
Cover of each of the following spp. < 75%:	na	na	na	na		
C. vulgaris	na	na	na	na	34.3	14.0
E. vaginatum	na	na	na	na	15.7	14.0
M. caerulea	na	na	na	na	25.0	8.7
T. germanicum	na	na	na	na	7.5	3.5
Total cover of neg. indicator spp.: (*A. capillaris, H. lanatus, P. aquilinium, R. repens*) < 1%	na	na	na	na	0.0	0.0
Cover of non-native species < 1%	na	na	na	na	0.0	0.0
Cover of non-native species < 1%	na	na	na	na	0.0	0.0
Cover of bare ground < 10% (relevé)	na	na	na	na	0.0	0.0
Cover of bare ground < 10% (vicinity)	na	na	na	na	0.0	0.0

Appendix III

Species list for Mt Brandon Nature Reserve.

Shrub species	**Graminoids**	**Bryophytes, Liverworts and Lichens**
Calluna vulgaris	*Juncus acutiflorus*	*Breutelia chrysocoma*
Erica cinerea	*Juncus articulatus*	*Calliergonella cuspidata*
Erica tetralix	*Juncus bulbosus*	*Campylium stellatum*
Ulex gallii	*Juncus conglomeratus*	*Campylopus atrovirens*
Vaccinium myrtillus	*Juncus effusus*	*Campylopus flexuosus*
	Juncus squarrosus	*Campylopus gracillis*
Herbs	*Luzula campestris*	*Campylopus introflexus*
Alchemilla mollis	*Luzula multiflora*	*Dricranum majus*
Anagalis tenella	*Luzula sylvatica*	*Dicranum scoparium*
Bellis perennis	*Carex binervis*	*Dicranum* sp.
Campanula rotundifolia	*Carex echinata*	*Frullania teneriffae*
Cardamine flexuosa	*Carex nigra*	*Hylocomium splendens*
Cardamine pratensis	*Carex panicea*	*Hypnum cupressiforme*
Chrysosplenium oppositifolium	*Carex pilulifera*	*Hypnum jutlandicum*
Dactylorhiza sp.	*Carex pulicaris*	*Kindbergia praelonga*
Drosera rotundifolia	*Carex viridula*	*Leucobryum glaucum*
Epilobium brunnescens	*Eliocaris multicaulis*	*Mnium hornum*
Euphrasia officinalis agg.	*Eriophorum angustifolium*	*Plagiothesium undulatum*
Galium palustre	*Eriophorum vaginatum*	*Pleurozium scherberi*
Galium saxatile	*Trichophorum germanicum*	*Polytrichum commune*
Hypericum pulchrum	*Agrostis canina*	*Racomitrium fasciculare*
Hypochaeris radicata	*Agrostis capillaris*	*Racomitrium languinosum*
Lathryus linifolius	*Agrostis stolonifera*	*Rhytidiadelphus loreus*
Lysimachia nemorum	*Anthoxanthum odoratum*	*Rhytidiadelphus squarrosus*
Nathecium ossifragum	*Cynosurus cristatus*	*Scleropodium purum*
Pedicularis sylvatica	*Danthonia decombens*	*Scorpidium cossonii*
Pinguicula grandiflora	*Deschampisa flexuosa*	*Sphagnum* sp.
Plantago lanceolata	*Festuca ovina*	*Thuidium tamariscinum*
Polygala serpyllifolia	*Festuca rubra*	*Cladonia arbuscula*
Potentilla erecta	*Festuca sp.*	*Cladonia portentosa*
Prunella vulgaris	*Festuca vivipara*	*Cladonia sp.*
Ranunculus acris	*Holcus lanatus*	*Cladonia uncialis*
Ranunculus flammula	*Lolium perenne*	*Diplophyllum albicans*
Ranunculus repens	*Molinia caerulea*	*Liverwort sp.*
Rosa arvensis	*Nardus stricta*	*Lophocolea bidentata*
Rumex acetosella		*Pellia epiphylla*
Stellaria alsine	**Ferns**	*Pleurozia purpurea*
Succisa pratensis	*Blechnum spicant*	*Scapania* sp.
Taraxacum off. agg.	*Dryopteris aemula*	
Trifolium pratensis	*Hymenophyllum wilsonii*	
Trifolium repens	*Pteridium aquilinium*	
Viola palustris		
Viola riviniana		
Viola sp.		

[END]