Management and Monitoring in Protected Areas: A case study in The New Forest National Park

The effects upon Valley Mire and Heathland Communities

Alexander Thomas Lovegrove

A thesis submitted in partial fulfilment of the requirements of Bournemouth University for the degree of Doctor of Philosophy

October 2016

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Abstract

Conservation depends heavily on protected areas, and many of these must be intensively managed in order to maintain important habitats and species. However, few conservation actions can be properly described as evidence-based, and knowledge of the effects of different types of management is necessary to improve conservation success. The New Forest National Park, Hampshire, UK, has been subject to intensive management which has not been scientifically investigated. In this thesis I assess management and monitoring effectiveness within two important open habitats: valley mires and heathlands. Research aims are to determine the impact that long term management (including differences between cutting and burning) has had on heathland communities in the New Forest, uncover the effects of restoration programmes on both biotic and abiotic components of valley mires, and to identify whether the current monitoring practice through Common Standards Monitoring (CSM) can detect changes in both habitats and serve as a basis for adaptive management. Data were collected in the field for both heathland and mire habitats using space-for-time substitutions, where sites of different ages since management intervention were compared. The plant community and soil chemical properties were assessed using randomly distributed quadrats within plots set up at each site location. A total of 30 heathland plots and 60 mire plots were established.

Results showed that different heathland management resulted in great differences in the vegetation community, but with few changes as sites aged following management. The block nature of both management activities resulted in relatively uniform ericaceous age-structures within individual sites, and particularly low species diversity was recorded for burnt sites. CSM in heathland habitats is was poor at detecting changes in the habitats, but attribute sub-scores did demonstrate some differences. Analyses of mire restorations showed mixed results with abiotic conditions showing almost no changes between degraded and restored areas. Some minor differences emerged in the vegetation community, such as a decrease in *Carex panicea* following restoration and an increase in some *Sphagnum* species, such as *S. papillosum*. This work suggests that restoration has largely failed to achieve its goals. Techniques from the established monitoring program failed to show any changes following restoration and lacked the precision that detailed quantitative surveys showed. It was highly ineffective as a tool to monitor restoration work, calling into question its widespread use for such tasks in the New Forest. Current monitoring must be improved substantially if a move to effective and adaptive management is to be achieved.

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Author's Declaration

This thesis comprises only my original work and due acknowledgement has been made in the text to all other material used.

Acknowledgements

Many thanks go to my family: Mum, Dad, Luke and Ross for their support as I wrote this thesis. Thank you also to my supervisory team; Adrian Newton, Anita Diaz, and Elena Cantarello for the giving me the opportunity to work in the New Forest and for their help and advice throughout this project. Louise Pearson at Bournemouth Unviersity was also extremely helpful. Barbara Smith (GWCT), Paul Eggleton (NHM) gave assistance and advice in the early stages. Data collection would not have been possible without the assistance of Sarah Oakley (FC), Grace Hervé (FC), Jane Smith (FC), Jennifer Thomas (NE), Ian Barker (NFNPA), Julie Stubbs (NFNPA), Georgianna Watson (NFNPA) and Graham Ferris (Commoner's Defence Association) and Marianne Bergin (FC) who provided information about the Forest and permissions for research. Grateful thanks go to Iain Green for training and assistance with laboratory methods. Thanks also to my friends The Celestial Elks (Aaron Morris, Josie Pegg and Aralisa Shedden-Gonzalez) and also to Greg Murray, John Stewart and Katie Hess, for their encouragement during my time at BU. I would also like to thank Justine Cordingley, Becky Spake, Paul Evans and Tom Birkitt for their advice at various stages. Finally, I would like to thank my examiners, Anthony Davy and Phillipa Gillingham for their engaging discussion about my work!

This project was co-funded with the New Forest National Park Authority (NPA) and was organised as a joint project between the NPA and Bournemouth University.

Chapter 1: Conservation Management and the New Forest National Park

1.1 Introduction and Context

1.1.1 Protected Area Conservation in England and the United Kingdom

Globally, declines in biodiversity continue to take place resulting from human-caused changes the environment (Newbold *et al.,* 2016; Newbold *et al.,* 2015; Cardinale *et al.,* 2012). Protected areas are one of the most important methods currently utilised to attempt to stem the loss of biodiversity (Hobbs *et al.,* 2010; Lovejoy, 2006), and in many areas they have been effective at conserving biodiversity in wider landscapes (Gray *et al.,* 2016). However, there are still many concerns about the effectiveness of protected area networks (Geldmann *et al.,* 2013; Pressey *et al.,* 2015), including those in the UK (Gaston *et al.,* 2006) and Europe (Pullin *et al.,* 2009), and also how to best maximise the conservation value of these areas (Watson *et al.,* 2014). In 2010, a review of England's protected areas (Lawton *et al.,* 2010) found that, despite decades of conservation efforts, biodiversity losses have continued in the country. Lawton *et al.* described England's protected areas as 'unequivocally' not a coherent and resilient network, and suggest that they are likely to respond poorly to future challenges such as climate change. Lawton *et al.* advised that biodiversity, together with ecosystem functioning and resilience, should be re-established in a network capable of surviving continued environmental change. This work was quickly given policy backing in a subsequent Government White Paper (DEFRA, 2011).

Understanding the effectiveness of conservation measures within protected areas is critical if they are to succeed. There are a multitude of different actions that have been taken with the aim of conserving biodiversity inside protected areas. However, evidence of the effectiveness of management activity in protected areas is extremely limited (Geldmann *et al.,* 2013), with monitoring of management outcomes an especially weak area (Leverington *et al.,* 2010). Because of continuous declines in biodiversity and species abundance, these problems have been recognised, and are now an area of fervent research activity (Watson *et al.,* 2014). In many sites of conservation interest, it is necessary for management to actively intervene in local ecosystems – either to emulate natural disturbance regimes (Mori, 2011; Sibley *et al.,* 2012), or to maintain traditional human influences in rare modified ecosystems (Joffre *et al.,* 1999; Blondel, 2006). Where traditional land use practices have ceased, conservation management may aim to emulate this type of human disturbance as well (Goldammer & Bruce, 2004; Lovén & Äänismaa, 2004). Within heathland habitats (particularly in the UK), disturbance is considered necessary to prevent succession to woodlands, which would result in the loss of endemic and nationally rare species (Britton *et al*., 2000). In other situations, interventions are required to recover habitats from degraded states; the process of ecological restoration (Perrow & Davy, 2002). Wetland habitats that have previously been subject to human-caused drainage are one example where habitat restoration is often applied (e.g. Labadz *et al.,* 2002).

1.1.2 Developments in Conservation Management: Evidence-based Conservation

There have been several recent developments in conservation biology that have aimed to improve management effectiveness. One of these is evidence-based conservation. This development was inspired by a revolution in medicine, where researchers observed that a great deal of medical practice was influenced by professional and commercial concerns rather than objective assessments medical interventions (Sutherland *et al*., 2004). In medicine, observations from systematic reviews suggested that relying on traditional reviews, where expert judgement was involved, lead to serious errors in practice (Antman *et al*., 1992) because the original sources of that expert knowledge were not always known and may be unreliable. Subsequently, methods were developed to support 'evidence-based practice', and it is now part of medical training (Pullin & Knight, 2001; Sutherland *et al*., 2004). Traditional reviews in conservation have the same problems (Roberts *et al.,* 2006) and in the UK many decisions in conservation management have been based on anecdotal sources rather than reliable evidence (Pullin *et al.,* 2004; Sutherland *et al.,* 2004). At one UK SSSI, Sutherland *et al.* (2004) showed that only 2% of the sources used for conservation decisions were based on verifiable scientific evidence, and such patterns are also found outside the UK (Pullin & Knight, 2005; Cook *et al*., 2009). Conservation guidelines can easily be wrong if they have not been objectively analysed (Sutherland *et al.,* 2004), highlighting the importance of monitoring and evaluation in developing a sufficient evidence base for conservation. Outcomes of management have been rarely evaluated, with few results reviewed systematically or disseminated to other practitioners (Pullin & Knight, 2001; Sutherland *et al.,* 2004). Calls for knowledge sharing to address dissemination issues have resulted in a degree of development (Pullin & Knight, 2001; Sutherland *et al.,* 2004), some managers are already adopting more 'adaptive' strategies (Macgregor & van Dijk, 2014) and systematic reviews in the scientific literature have enhanced knowledge about several conservation practices (Pullin, 2012). However, despite the recognition that a fundamental shift in conservation practice towards monitoring and evaluation is necessary, recent evidence still suggests that management activity is still based largely on anecdotal evidence in many areas (Legge, 2015). In some situations, such as in heathland management (Newton *et al.,* 2009), there is remarkably little robust evidence about management effectiveness, so further research is required to build an effective basis for conservation management.

1.1.3 Developments in Conservation Management: Adaptive Management

Another approach to addressing uncertainties about the effectiveness of conservation managers is Adaptive Management (AM). This approach to managing natural resources was developed by C.S. Holling and Carl Walters, beginning in the late 1970s. As with evidence-based conservation, AM was developed as a result of increasing awareness of the extent of uncertainties around management decisions and practices (Rist *et al.,* 2012). AM was developed in order to produce a management approach that purposely increases knowledge and addresses these areas of uncertainty (Rist *et al.,* 2012), and its central idea is that management can be applied as experimental treatments (Walters, 1997). The application of AM has specific requirements (Rist *et al.,* 2012), but it has been the subject of much confusion in both scientific literature and management practice.

Adaptive management should not be considered equivalent to trial-and-error, or 'learning-by-doing', which it is often conflated with (Wilhere, 2002). For example, the experimental side of AM requires that practices that are not in widespread use should also be evaluated if there are reasons to think that conservation could be improved by using them. Approaches to AM can be passive or active (Walters & Hilborn, 1978). Passive adaptive management involves identifying conditions and threats at the habitat site, forming objectives and activities to address these, and developing a monitoring plan that tests assumptions (Salafsky *et al.,* 2002). This process is then adjusted as monitoring data becomes available. Active adaptive management involves undertaking management as a deliberate experiment, with alternative policies considered treatments, and applied through statistically rigorous experimental design (Wilhere, 2002). Monitoring is necessary as the data-collecting stage of the experiment. Active adaptive management, although more expensive and complex than the passive form, has become the focus of research in more recent years (Wilhere, 2002). The advantages of using AM mean that management activities themselves can quickly become part of a future evidence base for conservation effectiveness, and targeting management at specific questions enables this process to continually develop, improving evidence, over future management cycles. AM has gained wide recognition in the scientific literature and with managers (Walters, 2007), but its use in practice has been fraught with problems, particularly with insufficient monitoring (Walters, 2007). Building a suitable basis for its use in protected areas will require improved knowledge about the effectiveness of current management techniques, especially those used to monitor ecological responses to management.

1.1.4 Monitoring and Evaluation in Conservation

Monitoring is essential for evaluating management effectiveness and is central to the application of both adaptive management and evidence-based conservation. However, only very few assessments of management include ecological outcome monitoring (Leverington *et al*., 2010; Lindenmayer *et al.,* 2012a), and this is often in qualitative form (Hockings, 2003; Gaston *et al*., 2006; Hockings *et al*., 2009). Such monitoring is vulnerable to error (Cook *et al.,* 2009) and poor at addressing direct questions about management effectiveness (Pullin & Knight, 2005; Margoluis *et al*., 2009; Lindenmayer & Likens, 2009). A lack of outcome-focused monitoring is also likely to result in expenditure being targeted poorly (Legge, 2015). In order to be most useful for conservation, monitoring should be targeted at the desired outcomes and objectives, and test management-orientated hypothesis (Nichols & Williams, 2011; Lindenmayer *et al.,* 2012b). Targeted monitoring is more cost effective than blanket monitoring, especially for conservation and ecology (Nicholls & Williams, 2011). Measurement variables should be quantitative and carefully selected, emphasising important indicators or functions, and underpinned by good scientific practice (Lovett *et al.,* 2007). An ideal monitoring programme will also have the capacity to adapt in response to new information or improve techniques as they become available (Legge, 2015; Lindenmayer *et al.,* 2012a), with the caveat that this does not affect the integrity of long-term data sets (Lindenmayer *et al.,* 2012a). A close connection with management is essential, ideally with continual feedback so that errors or trends can be acted upon as quickly as possible (Lovett *et al.,* 2007; Legge, 2015). Integrating monitoring into wider scientific research helps to disseminate any findings and ensures that monitoring data has wider applications than the immediate management impacts (Lovett *et al.,* 2007). Monitoring programmes that meet these goals will be able to contribute effectively to scientific knowledge, the evidence base for conservation and to adaptive and effective management. Because monitoring of management is an area where many protected areas are weak, examinations of management effectiveness also should be accompanied by evaluations of the monitoring strategies used to support them.

1.2 Knowledge Gaps

There are significant knowledge gaps in our understanding of protected area effectiveness (Gaston *et al.,* 2006; Southworth *et al.,* 2006; Timko & Innes, 2009; Geldmann *et al.,* 2013; Pressey *et al.,* 2015). These could be improved with more information about the conservation efforts and the methods by which these are monitored within protected areas (Hockings *et al.,* 2006). Information is, however, also lacking about the effectiveness of many ecological techniques that are in widespread use by conservation managers (Sutherland *et al.,* 2004; Pullin & Knight, 2005; Cook et al., 2009; Newton *et al.,* 2009; Geldmann *et al.,* 2013), and an additional area of uncertainty is the effectiveness of current monitoring techniques (Saterson *et al.,* 2004; Brooks *et al.,* 2004; Leverington *et al.,* 2010). Their effectiveness for detecting changes and evaluating management are unknown in many areas (Geldmann *et al.,* 2013), although there are indications that there are widespread deficiencies in practice (Leverington *et al.,* 2010).

This thesis describes an investigation into these knowledge gaps, which are addressed by undertaking a case study in the New Forest National Park. As in several other protected areas, biodiversity in the New Forest is feared to be in decline despite extensive management activity (Tubbs 2001; Newton, 2010). However, it is currently very difficult to evaluate such trends (Newton, 2010). Undertaking a study in the New Forest allows the wider knowledge gaps about conservation effectiveness to be addressed, as there is considerable uncertainty about the current management techniques for heathland habitat (Newton *et al.,* 2009) and long-term responses to mire restorations (Haapalehto *et al.,* 2010). Improved knowledge about these management techniques would be useful for the widespread uptake or continued use of these techniques, as well as contributing to an evaluation of management effectiveness in the New Forest itself. In addition to the interventions themselves, there are questions around the monitoring and evaluation techniques used. The current principal monitoring technique in the UK is Common Standards Monitoring (CSM), and this is extensively used across the UK protected areas (Jackson & Gaston, 2008). CSM has been subject to remarkably little evaluation, and it has never previously been examined in relation to management, as it was primarily designed to quickly report on the status of SSSIs beginning in the late 1990s. There are suggestions CSM may be poor at relating to individual species and more quantitative techniques (Newton, 2010; Davies *et al.,* 2007), which have prompted this investigation. Linking an investigation of management effectiveness with monitoring effectiveness will also allow inferences about the effectiveness of the whole protected area to be made, which will contribute to global research on this subject.

Examination of management, monitoring, and restoration activity will take place using successional chronosequences – In this case, areas that have undergone management interventions in the past (including those subject to restoration), and are presently undergoing succession. In this way, a space for time substitution (Fukami & Wardle, 2005; Wardle & Zackrisson, 2005; Wardle *et al.,* 1997) can be made. This allows interpretation of the dynamics of the ecosystem following management, and also allows longer time periods to be investigated than traditional short-term studies. There is now a growing recognition that observational approaches and 'natural experiments' are useful for addressing ecological questions over greater spatial and temporal scales than conventional experiments (Wardle *et al*., 2012; Sagarin & Pauchard 2010; Fukami & Wardle, 2005). A larger scale is also relevant to interpret management actions, as there is increasing concern that present landscape management practices may affect ecosystem function in unknown ways (BESS, 2011). The New Forest provides a unique opportunity for this type of study because of the large number of separate locations available, both for heathland and mire habitat, that can be used as replicates in a larger dataset. This distinguishes this work from previous studies examining heathlands, which suffered from a lack of replication and short timescales (Newton *et al*., 2009) and mires, which are typically intensive studies of single sites (e.g. Large *et al*., 2007; Vasander *et al*., 2003; Klötzli & Grootjans, 2001).

1.3 Study Site: The New Forest National Park

1.3.1 Overview

The New Forest, situated in the South of England, is the largest semi natural landscape in lowland Britain (Tubbs, 2001). Covering a total of 57100ha, it contains extensive areas of ancient woodland, heathland and mire wetlands (Tubbs, 2001). The Forest has arguably been a protected area for over 900 years, since its establishment as a hunting ground by the King of England in 1079. Because of its history as a 'Royal Forest', and the continuation of common rights of pasture within it, the New Forest has largely escaped the agricultural intensification that has transformed the wider landscape in Britain and Europe. As a medieval relict, the forest is one of the few remaining pastoral landscapes in Europe, and due to its size, biodiversity and the presence of traditional practices, it is one of the most important. Many of the habitats within the forest were formerly widespread in Western Europe, but are now rare and highly fragmented (Wright & Westerhoff, 2001). Chatters (2010) noted that the importance of the Forest was not because the area had any unique qualities, but because its system of land use has survived largely intact from the Middle Ages to the present, unlike much of the rest of the country. The New Forest's importance for conservation and the environment is reflected in the designations it has received. Made a National Park in 2005, the Forest contains (at least partially) 20 SSSIs, six Natura 2000 sites, including the New Forest SAC and the New Forest SPA, and two Ramsar convention sites (Chatters, 2006). The National Park itself surrounds a greater extent of land than the SAC area (although some elements of the SAC lie outside the Park boundaries). The Forest is also subject to a widespread management regime, with a variety of interventions taking place in different habitats. As a result, a number of sites are available that allow a unique opportunity to examine the impact and dynamics of management and restoration techniques. Owing to this management regime, uncertainties about the effectiveness of conservation techniques, and its large area, the New Forest is an excellent location to examine protected area effectiveness and its management and monitoring approaches. Further detail of the history, habitats and biodiversity, and socio-economic system of the New Forest is available in Appendix I.

1.3.2 New Forest Moorlands

The New Forest features several different habitat types, but this research focuses on the open areas of the forest, which are subject to intensive management. These open habitats feature diverse species that are increasingly rare in other lowland locations (Rand & Chatters, 2010). The New Forest contains the largest contiguous areas of lowland heath in Britain (Webb, 1986), featuring diverse communities of both dry and wet heaths. Valley mires are also numerous in waterlogged areas (Tubbs, 2001) and the largest series of surviving lowland valley mires are also found within forest boundaries (Atkinson, 1984; Clarke, 1988). Mires and heathlands are closely associated in the New Forest and are important because of their international rarity and the unique species that they support (Clarke, 1988).

Dry heaths in the New Forest feature communities dominated by *Calluna vulgaris,* dwarf gorse *Ulex minor*, and bristle bent *Agrostis curtisii* (Webb, 1986). Common gorse *Ulex europaeus* is also common in many areas. Wet and 'humid' heaths are dominated by *C. vulgaris,* cross-leaved heath *Erica tetralix* and *Sphagnum compactum*. Beneath the heathlands, soils are humus-iron podzols, but are considered to be less nutrient deficient than equivalent heaths in Dorset (Webb, 1986). In the absence of grazing, burning, or other interventions, heaths are colonised by a wide range of woody species, including oak *Quercus* spp., holly *Ilex aquifolium,* hawthorn *Crataegus monogyna* and birch *Betula* spp. and pine *Pinus sylvestris* (Webb, 1986). New Forest heathlands are considered to be particularly important for reptiles, including smooth snake *Coronella austriaca* and sand lizard *Lacerta agilis* (Webb, 1986; Noble, 2010). They are also important habitats for breeding birds, including Dartford warbler *Sylvia undata*, Nightjar *Caprimulgus europaeus* and Woodlark *Lullula arborea* (Conway *et al*., 2010). Plants subject to decline in lowland communities, such as wild gladiolus *Gladiolus illyricus*, heath lobelia *Lobelia urens,* dodder *Cuscuta epithymum*, petty whin *Genista anglica,* heath rush *Juncus squarrosus* and mat-grass *Nardus stricta*, are also found (Rand & Chatters, 2010). Although heathland contains relatively few invertebrate species that exclusively use the habitat, physical, chemical and community characteristics result in a rich diversity of species (Webb, 1986).

Valley mires feature a variety of communities, often dominated by *Sphagnum* sp. moss (Atkinson, 1984; Rand & Chatters, 2010). Abundant plants include bog myrtle *Myrica gale,* black bog rush *Schoenus nigricans,* purple moor grass *Molina caerulea,* sundew *Drosera* spp., bog asphodel *Narthecium ossifragum*, cotton grass *Eriophorum angustifolium* and white-beak sedge *Rhynchospora alba* (Webb, 1986). Rare and declining taxa found in valley mires include bog orchid *Hammarbya paludosa,* great sundew *Drosera anglica,* galingale *Cyperus longus,* slender cotton grass *Eriophorum gracile,* bog pimpernel *Anagallis tenella, Carex* spp. (including *C. echinata, C. hostiana, C. lasiocarpa* and *C. rostrata*), common butterwort *Pinguicula vulgaris* and lesser bladderwort *Utricularia minor* (Rand & Chatters, 2010). The mires are particularly important for some rare dragonflies, including the endangered southern damselfly *Coenagrion mercurial* (Thompson & Watts, 2010). Breeding lapwings *Vanellus vanellus*, snipe *Gallinago gallinago*, curlews *Numenius arquata* and redshanks *Tringa totanus* also utilise mire habitat as nesting areas (Conway *et al.,* 2010). Although a series of early studies on lowland mire vegetation, and specifically in the New Forest, took place in the 1950s-60s (Rose, 1953; Newbould & Gorham, 1956; Newbould, 1960), there has been comparatively little research since then (although see Clark (1988) and Atkinson (1984) for more recent studies). A number of palynological studies exploring past environments have used pollen from mire locations in the Forest (e.g. Barber & Clarke, 1987; Clarke & Barber, 1987). The majority of valley mires are at least partially minerotrophic, with run-off from the surrounding habitat often draining into them (Tubbs, 2001). Surrounding catchments are predominantly heathland (Atkinson, 1984). Valley mires often show strong zonation of vegetation along chemical and hydrological lines, with *Sphagnum* sp. rich vegetation in base-poor areas, flushed vegetation where there is movement of water with intermediate base status, and alder carr in highly flushed areas (Rose, 1953; Newbould, 1960). Flushed communities can be highly variable, in some areas featuring *Myrica gale, Molina caerulea* and *Schoenus* and in others *Menyanthes trifoliata, Phragmites australis* and *Equisetum fluviatile* (Newbould, 1960). Areas of *Sphagnum* cover are not as acid as the normal ombrogenous bog peats of northern Britain (Newbould & Gorham, 1956), and peat depths are typically quite shallow (around 2m) although this has not been systematically studied (Tubbs, 2001).

1.3.3 Management of the New Forest

To maintain the persistence of heathland habitat, a programme of burning and cutting is carried out by Forest managers. Burns take place in late winter on areas of gorse and heath, and are closely controlled (Smith & Burke, 2010). Heaths are burnt by the Forestry Commission on rotation, typically on periods of six to 12 years but sometimes longer (Tubbs, 2001). Cutting (by swipes hauled by tractors) and subsequent heather baling typically takes places where winter burning is not practical. Management by burning in the New Forest began around the 1870s (Clarke, 1988) and the practice was probably not widespread before then (Webb, 1986). These interventions are aimed at maintaining the habitat as heathland, by preventing succession and encouraging secondary regrowth (Webb, 1986; Smith & Burke, 2010). There is also a perceived contribution to grazing for livestock (Smith & Burke, 2010), but there is actually little evidence to suggest this is valuable (Tubbs, 2001). The process of burning is considered by managers to encourage new growth and form mosaics of different vegetation structure, in addition to preventing more severe wildfires at other times of the year because of the removal of fuel in the form of dead heather (Smith & Burke, 2010). However, in the scientific literature, evidence is lacking for the effectiveness of burning (as well as cutting), and very few robust studies have been reported (Newton *et al.,* 2009). Very little monitoring of these activites has taken place, further reducing available information, and particular weaknesses are a lack of long-term and fully replicated studies (Newton *et al.,* 2009).

Management activities that affect valley mires include the clearance of shrubs and regulation of water through ditching (Atkinson, 1984). Traditional mire management involved small-scale removal of peat, which may have added to mire diversity by providing open pools (Atkinson, 1984). However, this practice has now completely died out (Tubbs, 2001). Removal of *Sphagnum* and rushes also took place (Clarke, 1988). Prior to more modern techniques that give better control, heathland burns likely affected the edges of mire habitat as well (Clarke, 1988). Drainage was undertaken throughtout the Forest to improve land for forestry and as a grazing resource. Three significant phases of drainage have taken place: the 19th Century, the 1920s-30s and the postwar period running well into the 1980s (Clarke, 1988). Drainage ditches were actively maintained until relatively recently, and deep ditches have caused deleterious effects on valley mires (Atkinson, 1984). The effects of more recent drainage operations have been thought to have been more severe, because of the advent of mechanical digging (Clarke, 1988). Following work by Atkinson (1984) and Clarke (1988) highlighting these threats, and owing to a growing recognition of the importance of these habitats, efforts have been made to undo the damage in recent years. Beginning in the mid-1990s, a series of projects have taken place aimed at restoring mires to their former status by blocking ditches and enhancing water retention on mire sites. Initial progress on these restorations was slow (Holzer & Elliot, 2010), with few initial interventions, because of concerns about the impact on livestock grazing. However, since then many mire locations have been subject to drain-blocking interventions, among them the use of clay plugs and heather bales (Cooch & Morris, 2001; Holzer & Elliot, 2010). Ecological restoration of mires and other wetland habitats continues today under the Higher Level Stewardship scheme for the New Forest (Natural England, 2010). Although these techniques have been reported as successful elsewhere (Carroll *et al.,* 2011), outcomes from these restorations have not previously been scientifically assessed, and in the wider scientific literature restorations of lowland valley mires have not received much attention. However, there has been some criticism of the heather bale method used in other mire habitats (Green *et al.,* 2009; Grand-Clement *et al.,* 2015).

A notable characteristic of almost all New Forest habitats is that they are and have been for some time subject to a very heavy grazing regime. This has been the subject of some research (Pratt *et al.,* 1986; Putman *et al.,* 1987; Langbein & Putman 1999). Recent estimates suggest approximately 2500 deer range across the Forest, which include red *Cervus elaphus*, sika *Cervus nippon*, fallow *Dama dama*, roe *Capreolus capreolus* and muntjac *Muntiacus reevesi*. (Putman, 2010). Fallow deer are by far the most numerous (Putman, 2010). Domestic animals, which are principally ponies and cattle, but also include donkeys, domestic pigs and occasionally sheep, are allowed to roam freely over the Forest. Their existence is a continuation of the rights of common, and all livestock are owned by individual commoners. The high biomass and diversity of herbivores over a long period is thought to have held significant contributions to the ecology of the Forest (Putman, 2010). New Forest heathlands feature reduced structural diversity in comparison to Dorset heaths, which has been attributed to grazing, particularly of *Molinia caerulea* (Putman, 2010). Mire communities are also subject to extensive grazing pressure in some locations (Atkinson, 1984), and the graminoid community from these habitats may be an important source of food for livestock (Clarke, 1988).

1.4 Research Aim & Objectives

Protected Areas such as the New Forest remain the most important method for combating global biodiversity loss, and conservation strategies are particularly dependent upon these areas. The aim of this project is to investigate the effectiveness of one protected area, the New Forest National Park, by examining the capabilities of its management approaches and the suitability of its monitoring system. The implications of this study will have widespread significance, as many of the management, restoration and monitoring techniques investigated here are used around the UK and elsewhere.

Chapter 2 contains an investigation of the impact that long term management of heathland ecosystems has on the vegetation community and abiotic conditions. Burning and cutting measures in the New Forest are investigated, with the emphasis on community ecology. The objectives of this research were to answer the following questions:

i. What are the differences in impact between management types (cutting and burning) on a) the plant community and b) abiotic conditions?
ii. What are the dynamics of the heathland community following management?
iii. To what extent do the management interventions account for the variation in the plant communities observed?

The purpose of Chapter 3 is to examine whether Common Standards Monitoring (CSM) in heathland environments is a sufficient approach to record management effectiveness, and whether the current practices are suitable for an adaptive management approach. Specifically, the research aimed to answer the following questions:

i. Do habitat condition scores show measurable differences following management activity?
ii. How well do habitat condition scores and their constituent attributes reflect community and structural changes observed in the habitat community?
iii. How do CSM scores compare to Habitat Suitability Indices when there are changes in heathland habitat?

Chapter 4 documents an examination of restoration efforts in lowland valley mires, which in particular investigated how the plant community changed following restoration work and how abiotic conditions may have changed. The objectives of this research were to answer the following questions:

i. How has the plant community changed following restoration work?
ii. How have abiotic and plant structural conditions changed as a result of restoration?
iii. To what extent do the restoration interventions account for the variation in the plant communities observed?
iv. What are the pathways where restoration success could be improved, and what are the potential barriers to successful restoration?

The final data chapter, Chapter 5, aims to identify if CSM techniques were sufficient to detect whether restoration work has been successful in mire habitats, and if it is a suitable tool to monitor the recovery of mire sites. This research was designed to answer the following questions:

i. Have mire restoration projects resulted in changes to habitat condition, based on CSM techniques?
ii. Is the CSM approach sufficient to identify whether restoration work has been successful, and is it a suitable tool to monitor the recovery of sites?
iii. How do Habitat Suitability Indices compare with CSM Scores in mire habitats?

The outcomes of these studies are then explored in Chapter 6, examining the implications of management in the New Forest and for wider knowledge about the conservation techniques used and protected area effectiveness. Suggested changes and future work that could enhance both knowledge and effectiveness of future conservation efforts are also described.

1.5 References

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Chapter 2: Management of a Heathland Ecosystem in the New Forest: Effects of cutting and burning

2.1 Introduction

Lowland heathland ecosystems, owing to their unique flora and fauna and cultural value, have considerable conservation value (Alonso *et al*. 2001). The vegetation community in these ecosystems are dominated by heathers (*Calluna*, *Erica* and other *Ericaceae*) together with leguminous shrubs such as gorse (*Ulex* sp.), grasses and other plants (Loidi *et al*., 2010). Heathlands are often considered to have significant cultural value because of their long history of traditional use, including grazing and fuel collection (Bullock & Webb, 1995; Mallik, 1995; Calvo *et al*., 2005; Yallop *et al*., 2006). Lowland heathland ecosystems are distributed across Western Europe in Atlantic coastal territories, including the UK, Denmark, western France, North-west Germany and the NW of Iberia (Loidi *et al*. 2010), with approximately 20% of lowland heathland located in the UK (Webb, 1986). They are protected under EU (Habitats Directive, 92/43/EEC) and UK law (SSSI sites under the Wildlife and Countryside Act, 1981) and are important for many high conservation value species including Dartford warbler, *Sylvia undata*, woodlark *Lullula arborea*, nightjar *Caprimulgus europaeus,* smooth snake *Coronella austriaca* and many specialist invertebrates (Dolman & Land, 1992).

Lowland heaths have been subjected to considerable declines in the past two centuries (Aerts & Heil, 1993; Rose *et al.,* 2000; Fagúndez, 2012; Ramil Rego *et al.,* 2013, Pedley *et al.,* 2013; Vogels *et al.,* 2013) which has led to increasing conservation concern. Much of this decline, in both the UK and mainland Europe, has been the result of changes of land use, particularly agricultural intensification, afforestation and urban expansion, in addition to succession to other habitats (Marrs & Britton, 2000; Fagúndez, 2012; Ramil Rego *et al.,* 2013). As well as losses of heathland extent, changes in the quality of heathland habitats (community composition and vegetation structure) have also occurred in recent years across Europe (Cordingley *et al.,* 2015; Diaz *et al.,* 2013; Mohamed *et al*., 2006; Milligan *et al*., 2004; Rose *et al.,* 2000). These have also been associated with a decline in traditional use, including grazing by livestock, cutting of vegetation for fuel and animal fodder, burning and cutting of turf and peat (Webb, 1986). Vegetation dynamics have been altered as a result of elevated soil nutrient concentrations in an ecosystem that is normally very low in nutrients (Alonso *et al*., 2001), with evidence that atmospheric nutrient deposition has been contributing to invasion of heathland by grasses (Diaz *et al.,* 2013; Mohamed *et al*., 2006) and a loss of species in UK heathland communities (Southon *et al.,* 2013).

Heathlands are anthropogenic in nature (Diemont *et al.,* 2013; Mücher *et al.,* 2000) and must be continuously managed in order to prevent loss of the community and species through succession (Britton *et al*., 2000a; Marrs & Britton, 2000). Management of lowland heathlands is therefore aimed at preventing succession to other habitats and maintaining the structure and community of heath communities (Newton *et al*., 2009), and in recent years an increasing emphasis has been given to reducing nutrient levels on heathlands to ensure their long-term survival (Power *et al*., 2001). Intensive management is often seen as being necessary owing to the fragmented and small nature of most heathlands (Webb, 1998a). These practices include cutting, burning, the application of herbicides and grazing (Dolman & Land, 1992). With the exception of herbicide application, these practices were a widespread element of traditional heathland use (Webb, 1998a), and are now an established part of heathland management, with grazing becoming more prominent in recent times (Newton *et al*., 2009). Vegetation cutting and grazing are used to maintain the growth of *Calluna* and supress competing grasses, and to reduce populations of colonising shrubs and trees (Gimingham *et al*., 1992; Le Duc *et al.,* 2007). Burning is also seen as an effective way of preventing succession and reducing nutrient concentrations (Ascoli *et al.,* 2013; Webb, 1998a), although if it is used over large areas it could potentially create large expanses of even aged stands (Webb, 1998a; Velle *et al.,* 2014) and it could negatively impact heathland fauna if too frequent (Pereoglou *et al.,* 2016). There may also be significant geographic variation in its effects (Velle & Vandvik, 2014). Cutting is perceived by managers to be less effective in controlling nutrient concentrations compared to burning (Webb, 1998a).

Whilst management practices are well established, evidence regarding their effectiveness is lacking. There have been a number of studies on heathland cutting and burning, but a systematic review by Newton *et al*. (2009) found that few studies have been undertaken previously with robust experimental designs, with only three addressing heathland burns (Brian *et al*., 1976; Bullock & Webb, 1995; Lippe *et al*., 1985), three heathland cutting (Britton *et al*., 2000a; 2000b; Hallam & Hallam, 1981), and a further three examining burning or cutting with other management techniques (Barker *et al*., 2004; Froment, 1981; Vandvik *et al*., 2005). Many previous studies suffered from a lack of replication, short timescales or lack of control or comparators (Newton *et al*., 2009). Evidence supports the position that burning removes most of the nutrients from standing plants and leaf litter (Webb, 1998) but relatively little is known about the impact on nutrient availability in soil and in the structure of plant communities (Marcos *et al.,* 2015; Mohamed *et al*., 2007; Adams, 2003). Evidence from short term studies suggests that burning lacks effectiveness in terms of reducing N availability (Niemeyer *et al.,* 2009). Many heathland plants are vulnerable to increases in N (Damgaard *et al.,* 2014; Meyer-Grünefeldt *et al.,* 2016), so changes in this feature are important. Very few studies have examined management dynamics both above and below ground (although see de la Pena *et al*., 2012). One example is provided by Härdtle *et al*. (2009) who investigated above- and below-ground nutrient dynamics in heathlands in Germany, but only over the medium term (10 years). Regeneration of heathland following fire has been documented (Velle *et al.,* 2012), but there are gaps in understanding the dynamics of vegetation following fire, as high variability is often observed, highlighting the need for management to be monitored and evaluated (Davies *et al*., 2010). There is some evidence that fire temporarily increases nitrogen availability in soil, and there are indications that different initial communities can respond differently to burning independently of other conditions (Hobbs & Gimingham 1987; Vandvik *et al*., 2005), which may limit its effectiveness as a management intervention (Mohamed *et al.*, 2007). Longer term dynamics of heathland following cutting and burning have not been investigated, and effects on plant diversity, habitat structure and macroinvertebrate communities of heathland are largely unknown.

This research aims to investigate the impact that long term management (cutting and burning) has had on both the biotic and abiotic components of a heathland ecosystem in the New Forest. The objectives of this research are to answer the following questions:

i. What are the differences between impacts of management types (cutting and burning) on a) the plant community and b) abiotic conditions?
ii. What are the dynamics of the heathland community following management?
iii. To what extent do the management interventions account for the variation in the plant communities observed?

In order to assess the impact of heathland burning and cutting on the vegetation community, soil properties and grazing resource, a chronosequence of heathland habitat was surveyed. The chronosequence allows interpretation of the effects of management on both biodiversity and ecosystem properties to be investigated in a space-for-time substitution (Fukami & Wardle, 2005; Harmon *et al.* 1999).

2.2 Method

2.2.1 Study Site

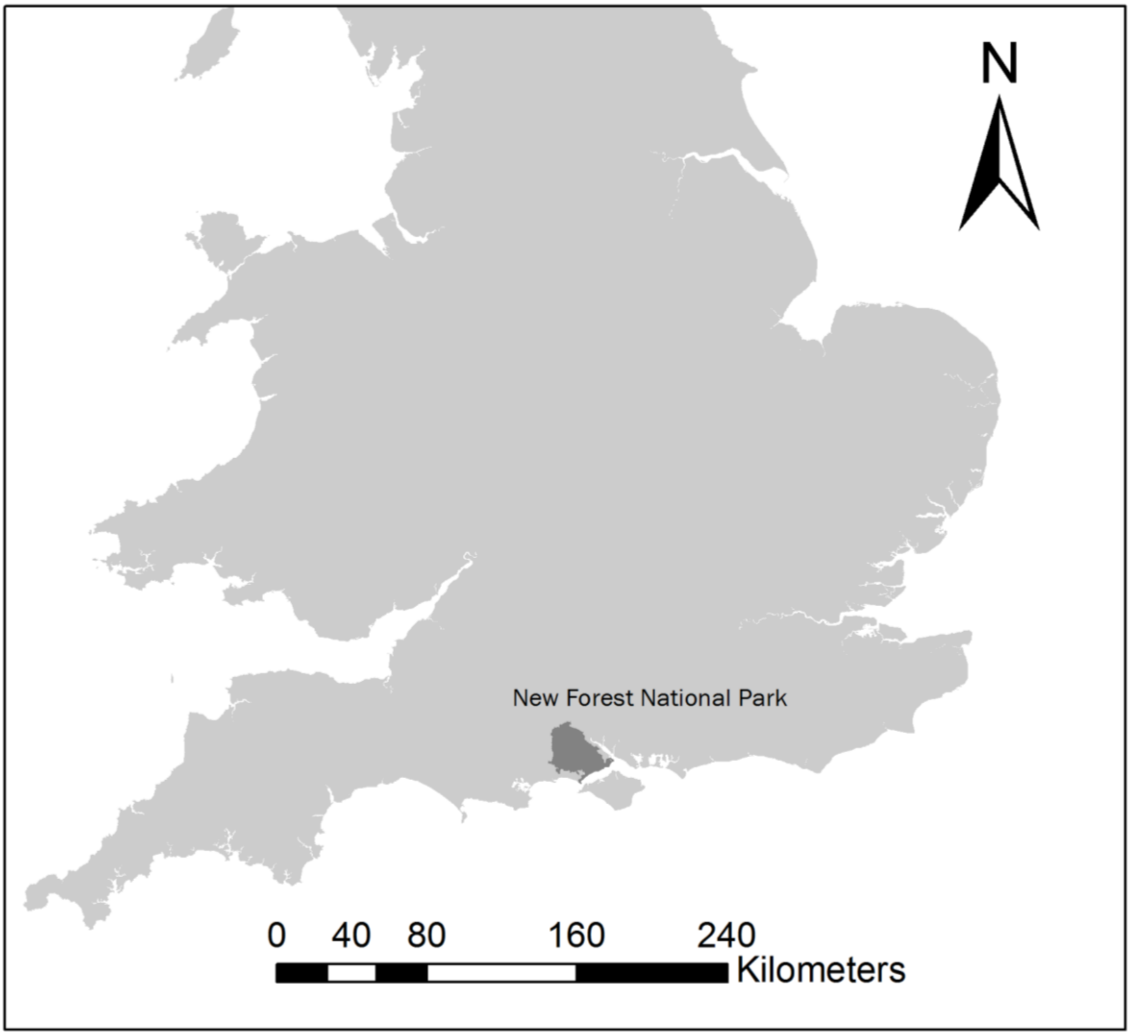


Figure 2.1 The location and boundary of the New Forest National Park within the United Kingdom.

Research was undertaken in the New Forest National Park (Hampshire, UK). The New Forest contains the UK's largest area of lowland heath and is has a number of conservation designations (see Chapter 1). New Forest heathlands are frequently characterised as 'humid heath', containing species associated with both wet heaths and dry heaths. The objective of heathland management is to keep the heathland in good condition, based on Common Standards Monitoring (CSM) requirements, and also to fulfil the requirement to keep open areas of the forest free from scrub under the New Forest Act 1949 (Forestry Commission, 2008). Burning takes place in the winter months (November to March) using a low temperature fire, whereas cutting is managed by tractor-mounted swipe year round (Smith & Burke, 2010). The size of the burn and cut areas is limited to a relatively small (2-

5 ha) area, and rotated approximately every 20 years, leading to a patchwork mosaic of heathland ages across the Forest.

2.2.2 Sampling Design

The effects of cutting and burning on the heaths were measured in a factorial design. Sites were selected from a database from the Game and Wildlife Conservation Trust (GWCT) based on Forestry Commission data (Smith *et al.,* 2013). Within three contiguous areas of heathland ("management blocks"), management has been undertaken on a rotational basis on a mosaic of heathland patches. Sites within each block were burnt or cut at different times, providing a chronosequence since last disturbance in each block (0, 1, 6, 10, 20 years), which were used to select sites for study. Replication was achieved by selecting one site of each age from each management block, reducing the impact of local conditions on the study. A total of five burned and five cut sites were selected randomly from each management block, resulting in a total of 15 burnt sites and 15 cut sites. Within each site, a survey plot of 50 x 50 m (0.25 ha) was established. As in Baker *et al.* (2004) no unmanaged (control) sites were used, as non-management is not an option that is currently pursued for maintaining lowland heathland in the New Forest. Management practices were therefore compared to each other rather than with unmanaged plots. A stratified random sampling approach was taken, whereby sites of each age were selected randomly from each block (where more than one site was available). Sites were between 2-5 ha in size. To generate the location of the survey plots, points were randomly generated within the boundary of each site using within ArcGIS (ESRI, Redlands, California) using the Create Random Points function in Arc Toolbox. Each generated point was then used as the SW corner of a survey plot within which data was collected. The corners of the plot were measured out using tape from the generated point, with the geographic location recorded with a handheld GPS device (Garmin GPSMap 62, Garmin, Kansas, USA).

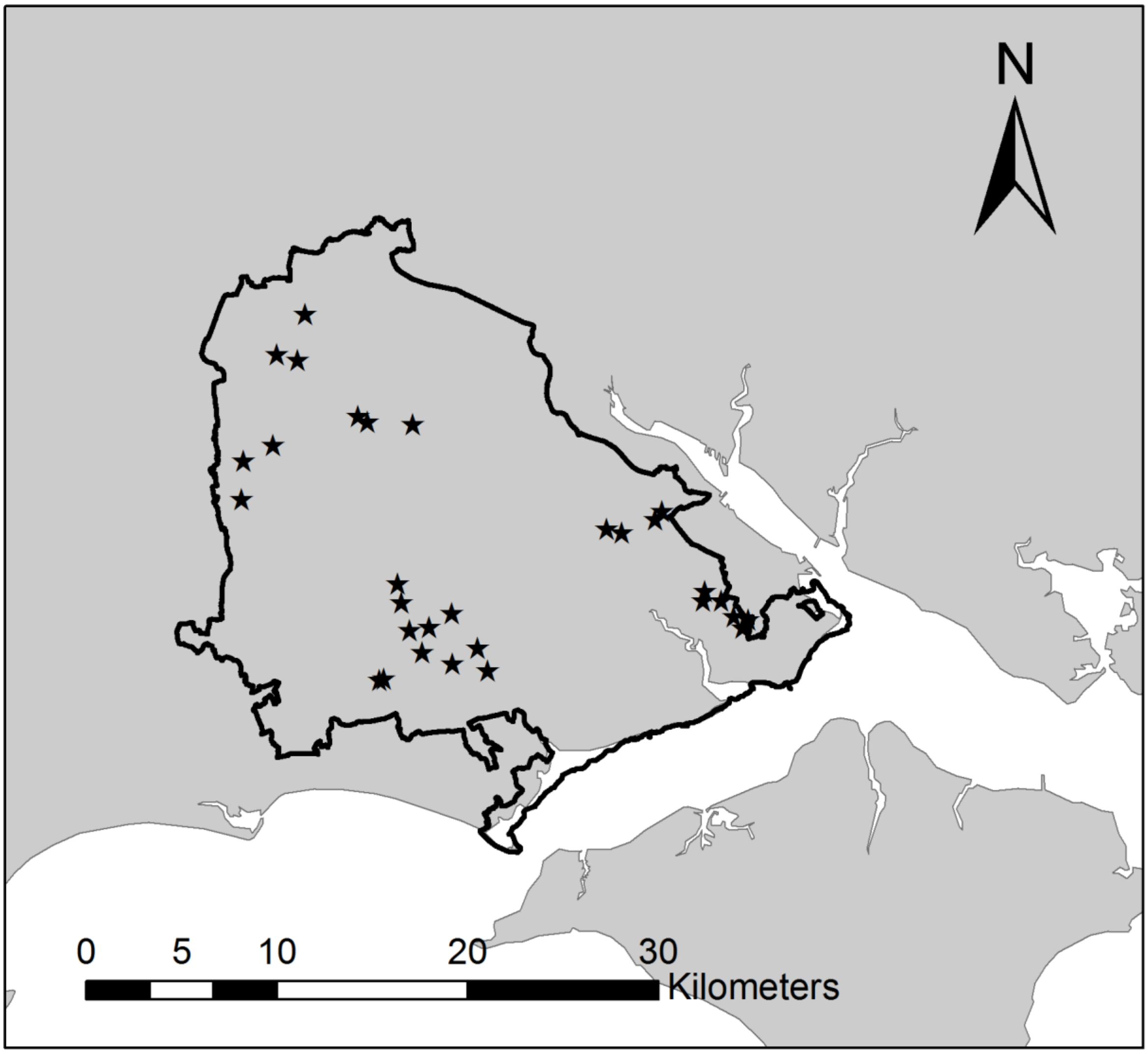


Figure 2.2 Location of heathland survey sites (black stars) within the New Forest National Park.

2.2.3 Sampling Method

In each 50x50m plot, a total of 10 sample points were established, using random number generation to locate samples within the plot. 2 x 2 m quadrats were used to assess vegetation (Bullock, 2006). At each quadrat point, the maximum height of the grass sward (at the centre of plot) was recorded, together with three measurements of vegetation height using drop disks, which were randomly placed inside the quadrat. The age structure of ericaceous shrubs was recorded (JNCC, 2004) where they were present, using the categories ‘pioneer’, ‘building/mature’, ‘degenerate’ and ‘dead’ and recording the total percentage cover of each of these categories. All vascular plant species were identified to species level and the respective percentage cover of each species within the quadrat was visually estimated, as in Diaz *et al*. (2011), with the addition of recording lichen and moss cover. Unidentified plants were recorded and photographed; these were subsequently identified outside of the field. Taxonomy of recorded species followed Stace (2010) with identifications using keys from Rose & O' Reilly (2006) and Rose (1989). The number of seedlings of tree species was also recorded, together with the amount of ground bare or covered by leaf litter. Observation bias was minimised by using the same researcher (the author) for all vegetation surveys that were undertaken.

Soil and quadrat samples were taken from the same location, recorded by GPS. The location of each sample within the plot was determined by using a random number sequence and using the plot edge as a grid. 10 soil cores of 5cm diameter (Rowell, 1994) were taken to 15 cm depth, comprising a composite of horizons, and were subsequently bulked to provide a per-plot value (Rowell, 1994; Goidts et al., 2009). Soil samples were analysed for extractable soil ammonium (NH_4^+) and extractable soil nitrate (NO_3^-) in addition to total N, C, P and K. Soil samples were bulked after collection to provide a value that would represent the whole plot, and were placed through a 2 mm sieve prior to analysis (Rowell, 1994). All samples were duplicated to ensure that representative values had been obtained alongside blank and standard samples to assess laboratory contamination. To measure plant available N, fresh soil was mixed with 50mL of 2M KCl, shaken on an orbital shaker, and then passed through medium grade filter paper. NH_4^+ was measured from this solution using an Ion-Sensitive Electrode (ISE, Vernier, Oregon, USA) as described by Molins-Legua *et al*. (2006). Extraction by vanadium (III) reduction (Rowell, 1994) was carried out to detect NO_3^-. Vanadium(III) in dilute acid solution causes reduction of nitrate to nitrite and/or nitric oxide, both of which are captured by Griess reagents (sulfanilamide and N-(1-naphthyl)-ethylenediamine) to produce a red dye. The reagent solution (200ml HCL, 0.5g Vanadium III, 0.2 g sulfanilamide and 0.01 g N-(1-naphthyl)-ethylenediamine dihydrochloride), was first prepared and then mixed with samples directly in cuvets. Samples were left for 24 hours to allow colour to develop, followed by colorimetric analysis using a UV spectrophotometer (Varian Vista Pro, Yarnton, UK).

For other elements, soil was dried at 60°C for 48 hours (until there was no measurable decrease in mass) prior to further analysis. P, K, Na, Ca and Mg were measured as follows: 0.1 g soil subsamples were weighed into microwave vessels, with 4.5ml HCl and 1.5ml HNO_3 subsequently added to each vessel. Microwave digestion was undertaken with a multiwave digester (Anton Paar GmbH, Graz, Austria). Samples were subsequently filtered into volumetric flasks before concentrations were determined by an Inductively Coupled Plasma Optical Emission Spectrometer (ICP-OES, Varian Vista Pro, Yarnton, UK). Additional soil analyses were conducted by the analytical laboratory at Forest Research, Alice Holt, Hampshire, UK. Soil pH was measured by suspension of 5 grams of soil with 25 ml of water, shaken on an orbital shaker for 15 min and rested for 45 min before being analysed by a Sentek pH electrode. Total Nitrogen, Carbon, Organic Carbon and Inorganic Carbon (TN, TC, TOC, TIC) were measured with Reference method ISO 10694 & 13878. This is a combustion method using a Carlo Erba CN analyser (Flash1112 series). Samples were ball milled before analysis, with 30 mg of milled soil weighed in tin capsules before being analysed simultaneously for total carbon and total nitrogen in the Flash analyser. For organic carbon, the same process was used with the exception that the capsule was silver and placed in a furnace at 500^{o}C for 2 hours to remove the organic carbon. The subsequent difference in measurement with total carbon allows the organic fraction to be calculated.

To evaluate plant nutrient composition, vegetation samples were taken from *Molinia caerulea* within each plot. Terminal shoots (<5 mm) were collected, with 6 shoots taken from each quadrat in order to sample multiple plants and multiple areas of each plot. Only leaf content was measured as this has the most relevance to forage quality for grazing animals. Samples were measured in duplicate to reduce experimental error. After collecting samples were dried at 60^{o}C for 48 hours (until a constant mass was achieved) and subsequently milled. Plant nutrients (P, K, Na, Ca and Mg), were measured using acid-digestion and ICP-OES following Moody & Green (2010). Samples of plant material (approximately 0.25g) were digested in 10ml of 70% nitric acid, subsequently evaporated to dryness and re-suspended in 25ml of 5% nitric acid prior to analysis in ICP-OES.

2.2.4 Statistical Analyses

Data were analysed in IBM SPSS Statistics 19.0 (IBM Corporation, New York USA), Primer 6.1.16 (Plymouth Marine Laboratory, Plymouth, UK), ECOM II (Pisces Conservation, Lymington, UK), and R 3.2.3 (R Core Team, 2016). As heathland features specialist species that are not found in other habitats, analysis examined both the impact on total species richness and also some of the key species found at the site. Data were initially examined in a correlation matrix in order to examine the relationships between the variables and their potential impact on the analyses performed. Spearman-rank correlation was carried out on untransformed data for this exploratory step. The high number of variables included necessitates a number of multiple comparisons within the correlation. Testing using confidence intervals was applied, using the bootstrapping technique in SPSS, to reduce the risk of type I error (Cumming, 2012). This was preferred to Bonferroni correction as the latter method was considered to risk a very high chance of type II error.

Confidence intervals for Spearman's Rho were calculated to 95%, and determined to be significant if the interval did not cross zero.

To detect differences between management types and to identify dynamics following restoration, 2 x 5 factorial ANOVA tests were performed to determine whether the measured variables differed between management types, between the age of the plots, or whether there was a significant interaction between the two factors. As running a number of ANOVA tests on different variables increases the risk of Type I error, some caution in interpreting the results is required. Due to the number of comparisons, it was felt that applying Bonferroni correction would increase the risk of type II error too substantially to be of use (Field, 2013). Therefore, eta-squared effect sizes (η^2) and cautious consideration of the actual differences between groups, were used to guide interpretation of significant test results. η^2 represents the proportion of the total variability in the dependent variable that is accounted for by variation in the independent variable (Levine & Hullett, 2002). η^2 is scaled from 0-1 and can be considered analogous to r^2 in correlation analyses (Levine & Hullett, 2002). Treatment means were examined in combination with Tukey's HSD post-hoc analysis to determine significance. Where an interaction was present, simple effects analysis was performed in SPSS instead of post-hoc tests. The assumptions for ANOVA were tested using Levene's test (homogeneity of variance), visually checking the normality of residuals using graphs, and examining boxplots for potential outliers. Normality tests were not used because the number of factors prohibited effective test sizes. The design is robust to samples showing unequal variances owing to the equal sample sizes for each factorial group, and ANOVA is relatively robust to violations of normality, but results should be considered with greater caution if these conditions are breached.

Additionally, Non-metric multidimensional scaling (MDS) was used to determine ecological distance in the habitat community. This was performed so that changes in the vegetation community as a whole could be examined, without having to test every variable with ANOVA. This was carried out on a Bray-Curtis similarity matrix, a popular matrix for abundances and suitable for habitat community data (Clarke & Gorley, 2006). An Analysis of Similarities (ANOSIM) was performed to statistically test whether *a priori* factors influenced the patterns detected in the MDS output. Similarity Percentages (SIMPER), based on the Bray-Curtis similarity, were used to follow-up the ordination to examine where plots shared similarities and where they differed.

To determine how much the variation in the vegetation community resulted from management, further analyses were undertaken. Canonical Correspondence Analysis was undertaken using habitat and environmental data in order to examine the potential relationship between the habitat community and possible explanatory variables, including management and soil conditions. CCA is considered the best constrained ordination method (McGarigal et al., 2000) and is robust to skewed data or data where there is considerable noise (Palmer, 1993). The collected data meets the assumptions that they are independent observations and are randomly selected owing to the sampling design. The dummy variables of site age (1= coded for 0 and 1 year old plots and 0 = all older plots) and management type (1= burn, 0 = cut) were added as explanatory variables, to determine whether dynamics following management or the type of disturbance has as great an effect on the vegetation community as local environmental conditions. Environmental data included was log10 (y + 1) transformed before analysis as because a 1-unit difference in nutrient concentration is probably much more important at low concentrations than it is at high concentrations (Palmer, 1993). Data were tested for multicollinearity, with some variables were removed so as to not bias the result. Variables that were removed from the analysis in order to prevent situations of multicollinearity included soil Na, K, C/N ratio, nitrate and ammonium and the slope and aspect of terrain. In particular, C/N ratio correlated with management activity, and soil ammonium with soil N. Multicollinear variables were removed based on assumptions of how important the variable was likely to be for the plant community. Monte-Carlo Permutations (1000 replications) were used to test the statistical significance of CCA output, as well as examination of the aforementioned correlation for explanatory factors (C:N ratio, total C, total N, total P, soil pH, total K, total Na, total Ca, total Mg, in addition to elevation, site age and management type).

2.3 Results

2.3.1 Exploratory Analysis

Spearman correlation showed that that many variables showed strong correlations with each other, although confidence intervals were generally very large, reflecting variability in the data. Of particular note are the strong negative correlations between ericaceous plants and some of the soil nutrients (principally N and P), and negative correlations between ericaceous species and overall plant species diversity measures. There is also a trend towards higher grass cover being associated with higher concentrations of nutrients. Further supporting the link between below ground nutrients and the vegetation community, high C/N ratios were associated with higher moss, lichen and ericaceous cover, as would be expected for more typical heathland soils of this type. A full correlation matrix can be seen in Appendix II, Figure II.1.

2.3.2 Effects of different Management Types (cutting and burning) on the plant community and abiotic conditions

A statistically significant main effect of management on the amount of bare earth was found (d.f. = 1, $F = 9.24$, $p = 0.006$, $\eta^2 = 0.16$), with burnt plots having higher cover of bare ground (mean ± SE of 7.6 % ± 1.2) compared to cut plots (2.6 % ± 1.2). The age of the plot following management work also showed differences in cover (d.f. = 4, $F = 3.97$, $p = 0.016$, $\eta^2 = 0.28$), with post-hoc analysis indicating that plots of 0 and 1 year of age were different from those of 6-20 years ($p < 0.05$ in both examples). There was a significant interaction between the type of intervention and the age of the plot following management on the cover of bare ground (d.f. = 4, $F = 2.97$, $p = 0.044$, $\eta^2 = 0.21$), and simple effects analysis was indicated that the cover of bare ground differed between management types at the ages of 0 and 1 years ($p = 0.001$ and $p = 0.026$ respectively) but not at older ages. Specifically, cover of bare ground was initially high in burnt plots (18.4% ± 6.3 for plots of 0 years, 10.5% ± 4.7 for plots of 1 year) before falling (2.9% ± 0.3, 3.8% ± 2.1% and 2.6 ± 0.3 for burnt plots of 6, 10 and 20 years), while cut plots maintained a low cover throughout (sites of 0, 1, 6, 10 and 20 years were 3.9% ± 0.7, 1.6% ± 0.7, 1.8% ± 0.7, 2.8% ± 1.0 and 2.7% ± 0.2 respectively). Although some of the assumptions of ANOVA were violated (normality), the test is reasonably robust against violations of normality so the differences over age and management type are well supported.

The percentage cover of all Ericaceous species also showed differences between plots of different management activity (d.f. = 1, *F* = 17.67, *p* = 0.001, η^2 = 0.30) and differences across plot age after intervention (d.f. = 4, *F* = 4.43, *p* = 0.01, η^2 = 0.30). The effects of interaction were not significant (d.f. = 4, F = 0.80, *p* = 0.538), indicating that ericaceous cover responded in the same way, regardless of the intervention type. Burnt plots featured ericaceous cover in greater abundance (mean of 48.9% ± 5.6 S.E) than cut plots, (24.0 % ± 4.5). Post-hoc tests for site age revealed significant differences only between sites of 10 years following management and early sites (0 years, *p* = 0.043, and 1 year marginal with *p* = 0.059). The composite nature of 'ericaceous cover' may contribute to this low precision. Mean cover for plots of 0, 1, 6, 10 and 20 years were 21.1% ± 6.6, 22.4% ± 6.6, 41.1% ± 6.6, 51.2% ± 6.6 and 46.6% ± 6.6 respectively). This pattern seems to indicate an increase in cover over time, followed by a potential decrease in ericaceous cover in the oldest plots, as sites of 20 years are not statistically distinguished from sites of 0-6 years. However, the wide confidence intervals and low statistical significance mean these differences are difficult to state with confidence. The change in ericaceous cover over age and management type is shown in figure 2.3. All ANOVA assumptions were met for this test.

For the individual dwarf shrub species, *Calluna vulgaris* showed significant differences across intervention type (d.f. = 1, F = 5.40, *p* = 0.031, η^2 = 0.11) and also across plot age (d.f. = 4. F = 5.19, *p* = 0.005, η^2 = 0.41). There was no interactive effect (d.f. = 4, F = 1.19, *p* = 0.345), indicating that *Calluna* cover showed a similar shaped response to management whether it was cut or burnt, despite diffences in the relative cover between management. Post hoc tests revealed significant differences between plots of 0 and 1 years with those of 10 years of age (p <0.05 in both cases). This shows a strong association with the ericaceous cover and building heath age class, as would be expected. Generally, *Calluna vulgaris* was present in greater percentage cover on burnt sites (mean cover of 28.4% ± 5.0 SE) than on cut sites (17.1 ± 3.7, fig 2.3). Following management, *Calluna* appeared to increase in abundance until declining between the oldest plots, although only differences between the earliest plots and those with greatest abundance at 10 years could be statistically separated (fig 2.3). All ANOVA assumptions were met for this test. Tests on *Erica tetralix* showed highly significant differences between management type (d.f. 1, F = 29.33, *p* = 0.001, η^2 = 0.52), but no differences over the age of the plots (d.f. = 4, F = 0.54, *p* = 0.707) and no interactive effect (d.f. = 4, F = 1.31, *p* = 0.301). *Erica tetralix* showed considerably higher abundance on burnt plots (16.6% ± 2.7) than on cut plots (1.5% ± 0.6). All ANOVA assumptions were met, and the large effect size inspires confidence in the result. *Erica cinerea* did not register any significant differences in cover relative to the factors in the study (see Appendix II for ANOVA tables), possibly as a result of its overall low abundance.

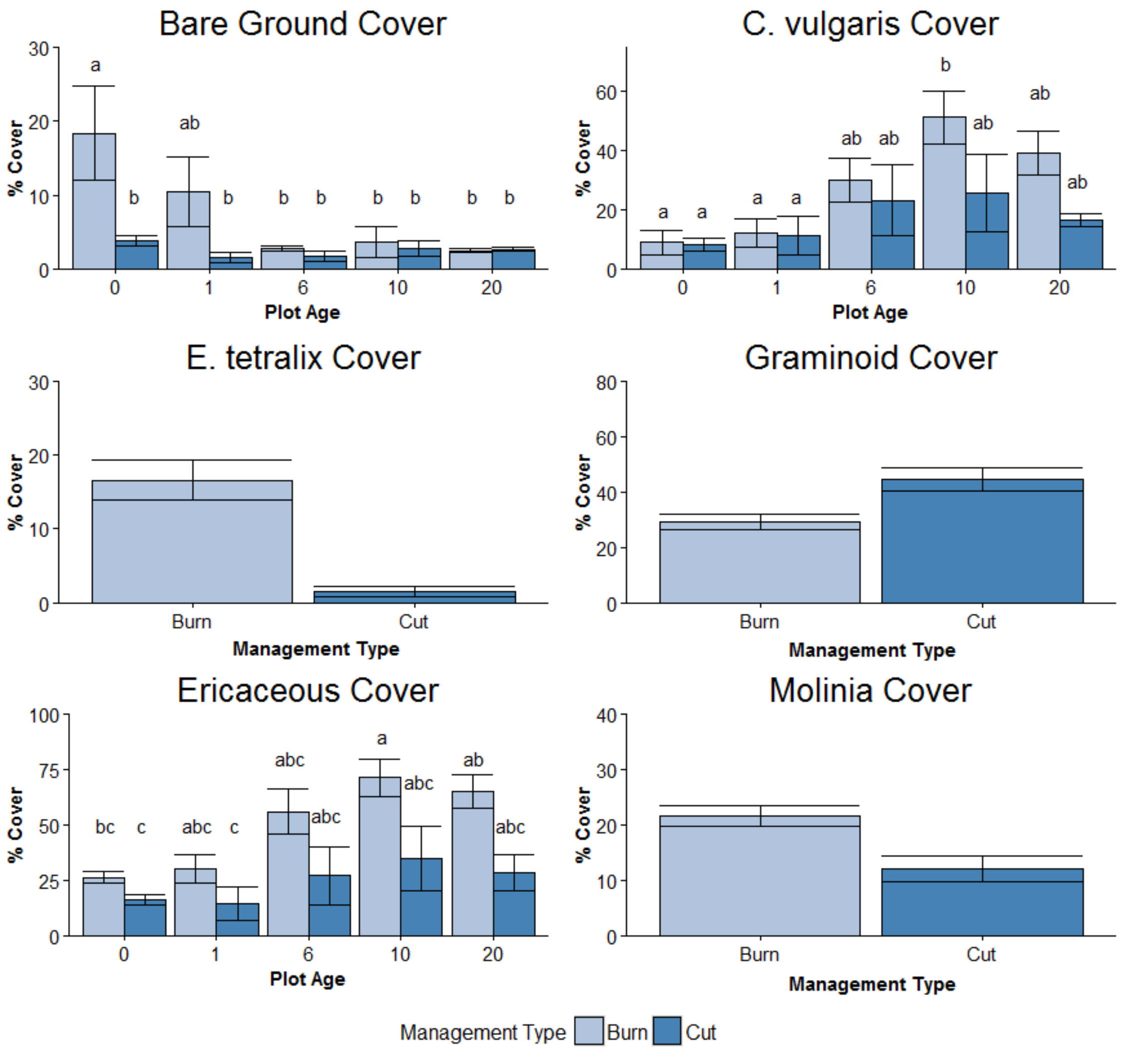


Figure 2.3. Graph illustrating differences in key variables and species across management types and age after ANOVA analysis. All error bars represent standard error. Bars that share the same letter are not significantly different from each other in Tukey HSD post hoc tests (*p* < 0.05).

Tests of the total cover of grasses or Graminoids showed a significant difference between plots of different interventions (d.f. =1, *F* = 8.36, *p* = 0.009, η^2 = 0.26), but no differences over plot age (d.f. = 4, *F* = 0.52, *p* = 0.72) or any interactive effect (d.f. = 4, *F* =0.42, *p* = 0.795). Some caution is required in interpreting these results as plotting the residuals revealed a slightly non-normal distribution, and there were some outliers in the burnt sites. The outliers are likely to be a result of high variability in grass cover on the burnt sites. Generally, it does appear that grass cover was lower in burnt sites (Figure 2.3), and this is likely connected to the greater cover of Ericaceous species in these areas. An initial spike in grass cover was not detected. Percentage cover values (mean ± S.E.) were 28.1% ± 2.8 and 43.6% ± 4.1 for burnt and cut sites respectively. Of the constituent species of grass cover, *Molinia caerulea* showed significant changes between plots undergoing different management (d.f. = 1, $F = 12.83$, $p = 0.002$, $\eta^2 = 0.27$), but no differences between the age of the plots (d.f. = 4, $F = 0.87$, $p = 0.499$). The was also a significant interaction between the two factors (d.f. = 4, $F = 3.01$, $p = 0.043$, $\eta^2 = 0.25$), indicating that there were potentially some differences in how *Molinia* changed over time depending on the management used. However, the relatively high *p* value at only 30 samples, and low precision of mean percentage cover, makes the interaction difficult to interpret, and it may be a false positive. Separated by management, cut plots featured less cover of *Molinia* (12.2% ± 2.3) than burnt plots (21.7% ± 1.9), but in general there is high cover of this species across the study (figure 2.4). All ANOVA assumptions were met for these tests. Tests run on *Agrostis curtisii*, the second most abundant graminoid, also differed over management (d.f. = 1, F = 19.79, $p = 0.001$, $\eta^2 = 0.39$), but not over age (d.f. = 4, F = 1.32, $p = 0.295$) or interaction (d.f. = 4, F = 1.30, $p = 0.305$). Cover was much greater in cut plots (6.85 % ± 1.6) compared to burnt plots (0.09% ± 0.05), with a large effect size.

Examination of the ericaceous age structure revealed several differences between plots. Pioneer heath showed differences in cover between management intervention (d.f. = 1, F = 5.99, $p = 0.024$, $\eta^2 = 0.09$) and the age of the plot since intervention (d.f. = 4, F = 4.09, $p = 0.014$, $\eta^2 = 0.25$). Post-hoc tests indicate that significant differences are only found here between the sites of 0 years and those of 10 and 20 years ($p < 0.05$ in both cases). A significant interaction between management type and plot age (d.f. = 4, F = 60.4, $p = 0.002$, $\eta^2 = 0.36$) was also found. It appears that pioneer cover responds differently to the management activity used, with an initially high proportion of cover in burnt sites (mean ± SE of 22.2% ± 1.9 for 0 years and 12.7% ± 4.9 for 1 year) subsequently declining (9.9% ± 4.3, 2.5% ± 0.3 and 2.83% ± 0.6 for sites of 6, 10 and 20 years). In cut sites a low initial cover remains relatively constant throughout (sites of 0, 1, 6, 10 and 20 years are 5.8% ± 1.0, 4.7% ± 1.5, 4.7% ± 2.9, 6.7% ± 0.9 and 8.1% ± 2.9 respectively). This pattern becomes quite apparent in Figure 2.4. The test met all ANOVA assumptions.

Building and mature heath made up the majority of ericaceous cover recorded, but no significant differences were detected between plots subject to different management (d.f. = 1, F = 2.89, $p = 0.105$). Plots did significantly differ over age, however (d.f. = 4, F = 3.70, $p = 0.021$, $\eta^2 = 0.34$). No interactive effects were detected (d.f. = 4, F = 1.27, $p = 0.314$). Post-hoc analysis found differences in cover between plots of 10 years and sites of 0 years after intervention (mean ± SE of 36.5% ± 10.0 and 6.4% ± 1.9 respectively, $p < 0.05$). This indicates that building and mature heath must increase in cover after management, but also that low precision in determining the mean makes it difficult to distinguish until it reaches its maximum cover. The pattern here strongly resembles that for total ericaceous cover, of which it is a large constituent part. The oldest 20 year plots could not be separated statistically from the youngest, which may indicate a decline in cover between the 10 and 20 year plots. All ANOVA assumptions were met for this analysis, although there were some outliers in the 20 year sites showing very high and very low cover. Both of these outliers were in cut plots, and were considered to be reflective of high habitat variability in these areas.

Degenerate heath cover showed highly significant differences between plots of different management (d.f. = 1, F = 17.29, $p = 0.001$, $\eta^2 = 0.17$) and over plot age (d.f. = 4, F = 7.58, $p = 0.001$, $\eta^2 = 0.29$).). Post hoc tests showed significant differences between sites of 20 and sites of 0, 1 and 6 years since management (p = 0.001 for 0 and 1 years, $p = 0.01$ for 6 years). A highly significant interaction was also present (d.f. = 4, F = 9.06, $p = 0.001$, $\eta^2 = 0.35$), indicating that the cover of degenerate heath showed different responses after management, depending on the intervention carried out. Simple effects analysis found that the differences in degenerate cover between burnt and cut plots occurred in those of 10 and 20 years ($p = 0.04$ and $p = 0.001$), figure 2.4. Mean cover in early burnt plots of 0 (1.0 %± 0.8 S.E.) and 1 years (0.5% ± 0.5) was very similar to that in cut sites of the same age (0 years = 0.5% ± 0.3, 1 year = 0.6% ± 0.5), showed little difference after 6 years (burnt plots = 2.9 % ± 1.9, cut plots = 6.2% ± 5.7) but then diverged, remaining low in cut plots (10 years = 0.5 ± 0.5, 20 years = 0.7 ± 0.6) but dramatically increasing in burnt plots (10 years = 20.0% ± 11.3, 20 years = 40% ± 4.0). No outliers were present with these data and there was some minor derivation from normal distribution in a plot of the residuals. As previously stated, the test should be relatively robust to these violations, but results should be interpreted with some caution.

The percentage cover of dead heath showed highly significant differences between management interventions (d.f. = 1, F = 14.46, $p = 0.001$, $\eta^2 = 0.23$) and significant differences between age since management (d.f. = 4, F = 4.02, $p = 0.015$, $\eta^2 = 0.25$). Post-hoc tests showed that the differences occurred between 20 year plots and plots of 0, 1 and 6 years since intervention ($p < 0.05$). An interaction effect was also detected (d.f. = 4, F = 3.22, $p = 0.034$, $\eta^2 = 0.20$), with simple effects analysis indicating that the difference between managements occurred in plots of 20 years age ($p = 0.001$). Plots of both management types of early age show a low, variable amount of dead heath cover, before this appears to rise in burnt sites of 20 years (fig 2.4). No rise in the cover of dead heath occurs in cut plots. All ANOVA assumptions were met. The cover of dead heath remains very low throughout all plots, so caution should be applied in interpreting this result as these small changes would be difficult to detect by visual observation.

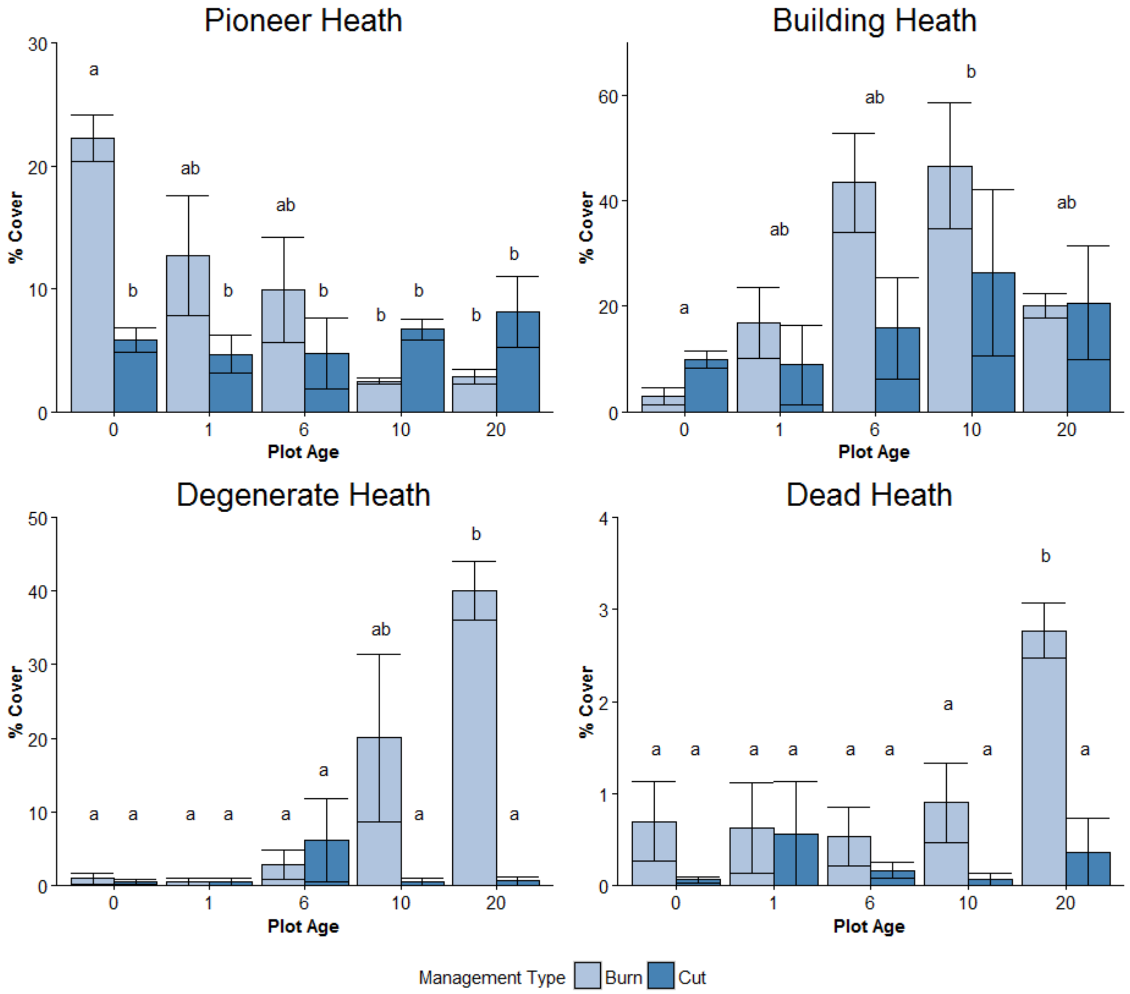


Figure 2.4 Differences over age and management type for different heathland growth stages following ANOVA analysis. All error bars represent standard error. Bars that share the same letter are not significantly different from each other based on Tukey HSD ($p > 0.05$). Building heath did not show an interactive effect, but differered in both management and age, and therefore is shown across all plots.

Species richness showed highly significant differences between different management types (d.f. = 1, F = 38.43, $p = 0.001$, $\eta^2 = 0.63$), but no differences across the age of plots (d.f. = 4, F = 0.55, $p = 0.701$) or any interaction between factors (d.f. = 4, F = 0.05, $p = 0.994$). The difference between the interventions is readily apparent with species richness much lower on burnt plots (mean of 12.9 ± 1.1 SE) than on cut plots (31.9 ± 2.5). However, detailed investigation is needed to examine why this is the case, and whether the species changes are a result of non-heathland or even undesirable species being present in the cut plots. The Shannon-Wiener H' diversity index showed the same pattern, with highly significant differences between management (d.f = 1, F = 43.12, *p* =0.001, η^2 = 0.63) but not age (d.f.. = 4, F = 0.881, *p* = 0.857) or interaction (d.f. = 4, F = 0.88, *p* = 0.493). ANOVA assumptions were met, although there was one outlier showing exceptionally high diversity in the cut interventions. Reflecting the species richness changes, mean diversity (S-W H') was also lower for burnt plots (1.48 ± 0.09) than in cut plots (2.55 ± 0.12). With these differences suggesting a different vegetation community present on burnt versus cut plots, a more detailed breakdown of the main components of the respective communities was examined through the use of ordination analysis.

The pH values of soil were lower on burnt plots (d.f. = 1, *F* = 4.73, *p* = 0.042, η^2 = 0.15), but showed no difference over age (d.f. = 4, *F* = 0.99, *p* = 0.435) or interactive effects between factors (d.f. = 4, *F* = 0.69, *p* = 0.607). The normality of residual assumptions were not met in this test. All plots featured acidic soils typical of heathland communities, with very similar mean pH (mean ± S.E.) of 4.3 ± 0.05 and 4.4 ± 0.07 for burnt and cut plots respectively (figure 2.5). The total concentration of N in soil showed significant differences across management types (d.f. = 1, *F* = 8.33, *p* = 0.009), but not for age (d.f. = 4, *F* = 1.01, *p* = 0.425) or interactive effects (d.f. = 4, *F* = 0.76, *p* = 0.566). All ANOVA assumptions were met for this test. In general, the mean concentrations of N were exceptionally low in both burnt (0.36% ± 0.03 SE) and cut plots (0.47% ± 0.02 SE, figure 2.5). Plant available nitrogen, in the form of ammonium/ammonia, showed significant differences between the management (d.f. = 1, *F* = 17.20, *p* = 0.001, η^2 = 0.40), with an average in burnt plots of 1250 mg kg^{-1} ± 100 and 770 ± 50 for cut plots. Differences over age (d.f. = 4, *F* = 0.17, *p* = 0.952) and interaction (d.f. = 4, *F* = 1.26, *p* = 0.317) were not significant. Plant available nitrates in the soil did not show any significant changes across management type (d.f. = 1, *F* = 0.73, *p* = 0.404) or plot age (d.f. = 4, *F* = 1.06, *p* = 0.402), but did show an interaction effect (d.f. = 4, *F* = 3.47, *p* = 0.026, η^2 = 0.36). However, assumptions of ANOVA were violated, making it difficult to confidently state that the difference was not merely a result of error, and Tukey HSD post-hoc failed to detect differences. Differences for the measures of plant available N are shown in Figure 2.6. Soil C showed no significant changes (Appendix II), although it was important to note that essentially all detected carbon in the soil was in organic form. C/N ratios in the soil did show significant differences between management types (d.f. = 1, *F* = 13.00, *p* = 0.002, η^2 = 0.36), but again, as with other soil attributes, not for plot age (d.f. = 4, *F* = 0.29, *p* = 0.883) or interaction between factors (d.f. = 4, *F* = 0.46, *p* = 0.767). All ANOVA assumptions were met. Mean C/N ratios were (25.4 ± 1.4 SE) for burnt plots and (18.9 ± 0.9 SE) for cut plots (fig 2.5). As carbon ratios were correlated with many other vegetation attributes, the significant difference found here could be one of the pathways in which the different management causes changes in the vegetation community.

For other soil nutrients, ANOVA testing showed differences in management type with Ca (d.f. = 1, $F = 4.40$, $p = 0.049$, $\eta^2 = 0.14$), although with a value should be treated with considerable caution. Differences over age were not significant (d.f. = 4, $F = 0.77$, $p = 0.557$); nor were interactive effects (d.f. = 4, $F = 1.03$, $p = 0.413$). Mean concentrations of Ca in soil were 461.9 mg kg-1 ± 124.3 SE for burnt plots and 797.5 mg kg-1 ± 97.3 for cut plots (fig 2.5). Soil Mg also showed significant differences across management type (d.f. = 1, $F = 5.97$, $p = 0.024$, $\eta^2 = 0.17$), but again not for plot age (d.f. = 4, $F = 0.85$, $p = 0.509$) or the interaction between factors (d.f. = 4, $F = 1.19$, $p = 0.348$). All ANOVA assumptions were met for these tests. Concentrations of Mg were higher in cut plots (754.6 mg kg-1 ± 134.5 SE) than in burnt plots (392.0 mg kg-1 ± 63.6), despite the relatively high amount of variability shown within them (fig 2.5). Soil P showed the same pattern, with highly significant differences across management type (d.f. = 1, $F = 16.03$, $p = 0.001$, $\eta^2 = 0.39$), but again none over plot age (d.f. = 4, $F = 1.16$, $p = 356$) or the interaction between factors (d.f. = 4, $F = 0.11$, $p = 0.979$). All ANOVA assumptions were met. The mean concentrations of soil P were higher in cut plots (278.6 mg kg-1 ± 13.4 SE) than in burnt plots (177.6 mg kg-1 ± 17.4), as illustrated by figure 2.5. Other soil variables, including K and Na in soil, did not show any significant differences across the plots when tested (Appendix II, table II.4).

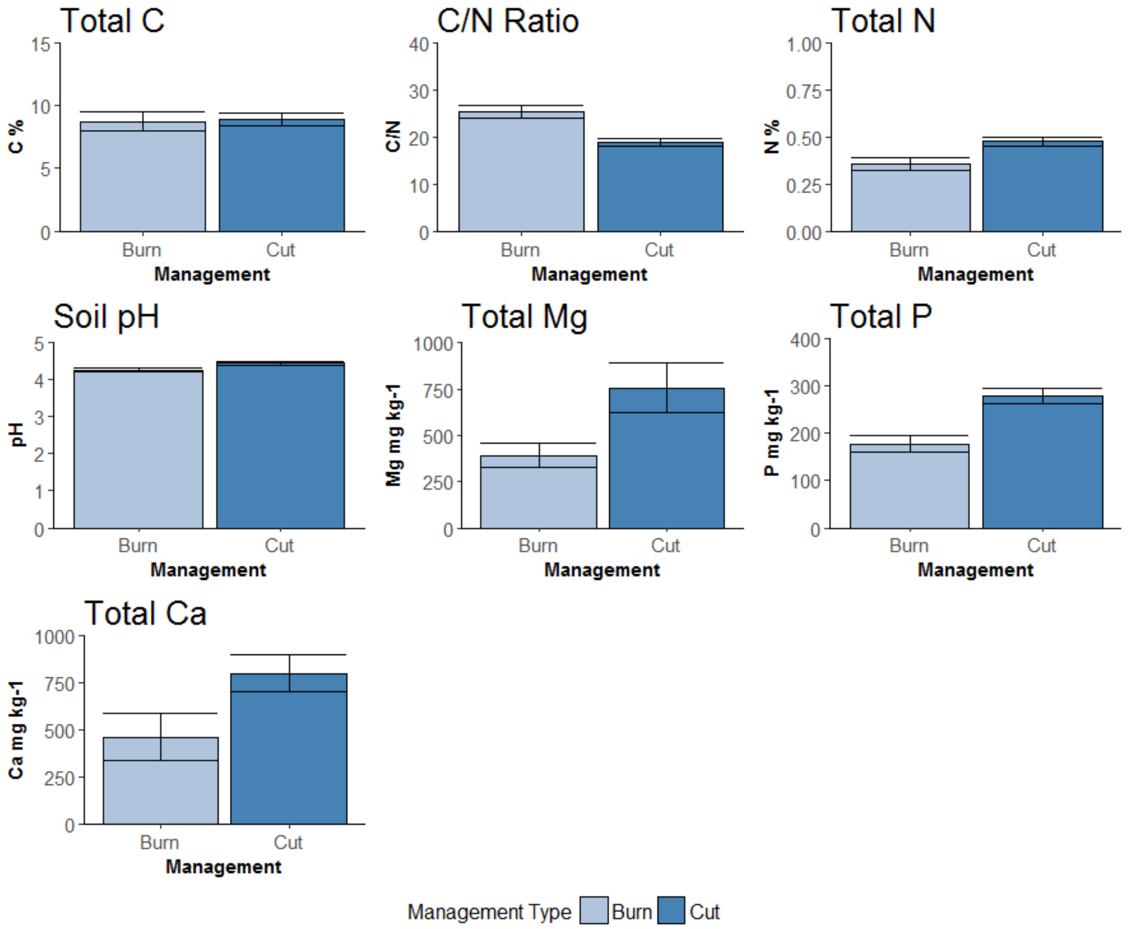


Figure 2.5 Differences in measured soil chemical properties between sites subject to different management types. Soils were sampled to 20cm depth and bulked. No significant differences were detected for soil carbon, but all other variables shown here were significantly differentiated with ANOVA analysis ($P < 0.05$). Error bars represent standard error.

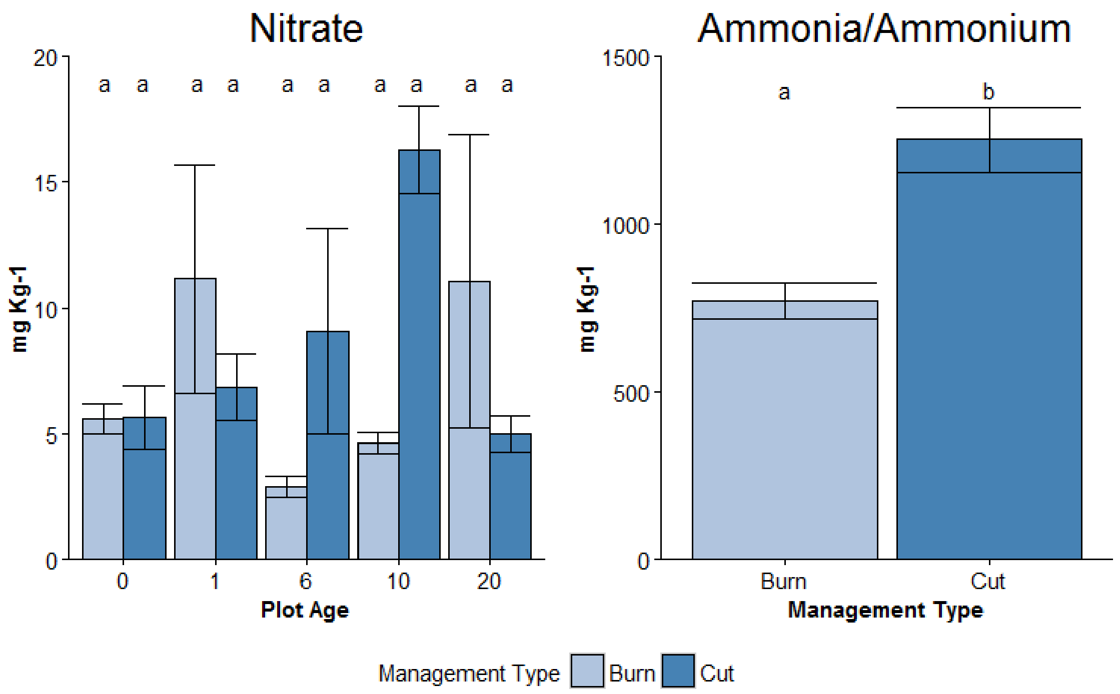


Figure 2.6 Differences in plant available N between study plots after ANOVA analysis. Nitrate showed differences based on ANOVA tests, but Tukey HSD post-hoc failed to identify these. Ammonium is a much greater constituent of available N and showed differences between management types. All error bars represent standard error. Bars that share the same letter are not significantly different from each other based on Tukey HSD.

Nutrients in *Molinia* leaf tissue were also examined. Leaf P showed highly significant differences across plots of different management for type of intervention (d.f. =1, *F* =40.93, *p* = 0.001, η^2 = 0.61) but only slight differences across plot age (d.f. =4, F = 0.98, p = 0.44). No interactive effects were detected (d.f. = 4, *F* = 0.50, *p* = 0.737). Differences between management types are readily apparent with P showing much higher concentrations on cut plots (mean concentration of 5690mg kg-1 ± 208 S.E.) than on burnt sites (3577mg kg-1 ± 240), shown in figure 2.7. The pattern shown here reflects differences observed in soil concentrations. ANOVA assumptions were met for this test. *Molinia* K content showed significant differences in leaf tissue in plots of different management type (d.f. = 1, *F* = 4.51, *p* = 0.046, η^2 = 0.13), but not for plot age (d.f. = 4, *F* = 0.92, *p* = 0.473) or any interactive effects (d.f. = 4, *F* = 1.33, *p* = 0.293). The differences here should be interpreted with caution, as the values shown for cut sites could potentially be anomalous because of their exceptionally high values. There were a number of extreme outliers, and in addition there was a significant departure from normal distribution of the residuals. A Kruskal-Wallis test showed highly significant differences between management type (*p* = 0.001), but caution is still necessary in determining the actual changes as a result of the data. Mean K concentration was 5034 mg kg-1 ± 774 SE within *Molinia* in burnt plots, and 12881 mg kg-1 ± 3677 in cut plots. Mg content in *Molinia* tissue showed a statistically significant difference across plots of different management type (d.f. = 1, F = 9.38, p = 0.006, η^2 = 0.27), but not over plot age (d.f. = 4, F = 0.84, p = 0.517) or any interactive effect (d.f. = 4, F = 0.39, p = 0.811), and all assumptions were met. Higher Mg concentrations were in cut plots (5258mg kg-1 ± 159) relative to burnt plots (4393mg kg-1 ± 213 SE). No statistically significant differences were found for Na or Ca within *Molinia* leaf tissue (Appendix II, table II.5). Overall, these tests showed that nutrients within *Molinia* leaf tissue showed similar patterns to those for nutrients within the soil.

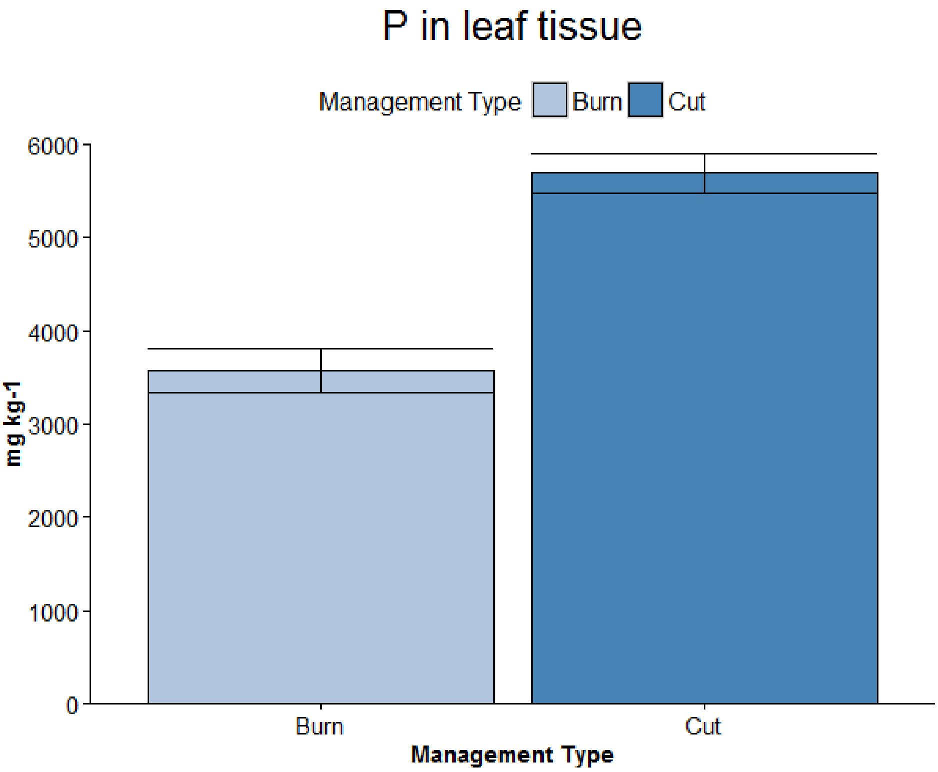


Figure 2.7 Significant differences in P content in *Molinia caerulea* leaf tissue between cut and burnt plots, detected with ANOVA analysis. The higher amount in cut plots suggests that nutrients are more readily available in this environment. Error bars show standard error.

2.3.3 Differences in the Community through Ordination Analyses

Non metric Multi-Dimensional scaling (MDS) showed striking differences between management types, with plots generally positioned in groups based on management activity (figure 2.8). This indicates the habitat community was very difference based on what management type had been used on the plots. There is some overlap in the middle of the diagram, and plots positioned here are likely to share features characteristic of both management types. The age of the plots did not appear to result in any obvious changes to the habitat community, with little discernible pattern in similarity between plots of similar ages. However, the oldest plots did generally appear to cluster with others of the same age and management type more closely than younger plots, which showed greater differences and a wider scatter across the ordination diagram. This suggests that younger plots have a greater variability in their habitat community than older plots. The MDS representation was considered an accurate depiction of the similarity of plots, as there was a relatively low stress value (0.1), the Shepard diagrams showed a low scatter (Appendix II Figure II.2), and superimposed groups from a Hierarchical cluster analysis (at the similarity level of 40%) roughly matched the position and number of groups in the ordination graph Fig 2.8, see Appendix II, Figures II.3 and II.4 for cluster analysis output). Bubble plots of some of the species recorded (Appendix II, Figures II.5-9) allowed the differences in vegetation community to be examined in detail. It can be seen that many species are almost exclusive to plots of particular management types – for example *Erica tetralix* on burnt sites, and *Agrostis capillaris* on cut sites, whereas others are present on most plots but show changes in their abundance, such as *Calluna* and *Molina*. Not all of these changes, particularly in *Molinia*, appear to be directly relatable to management, indicating that there are other important causes of variation within the plots.

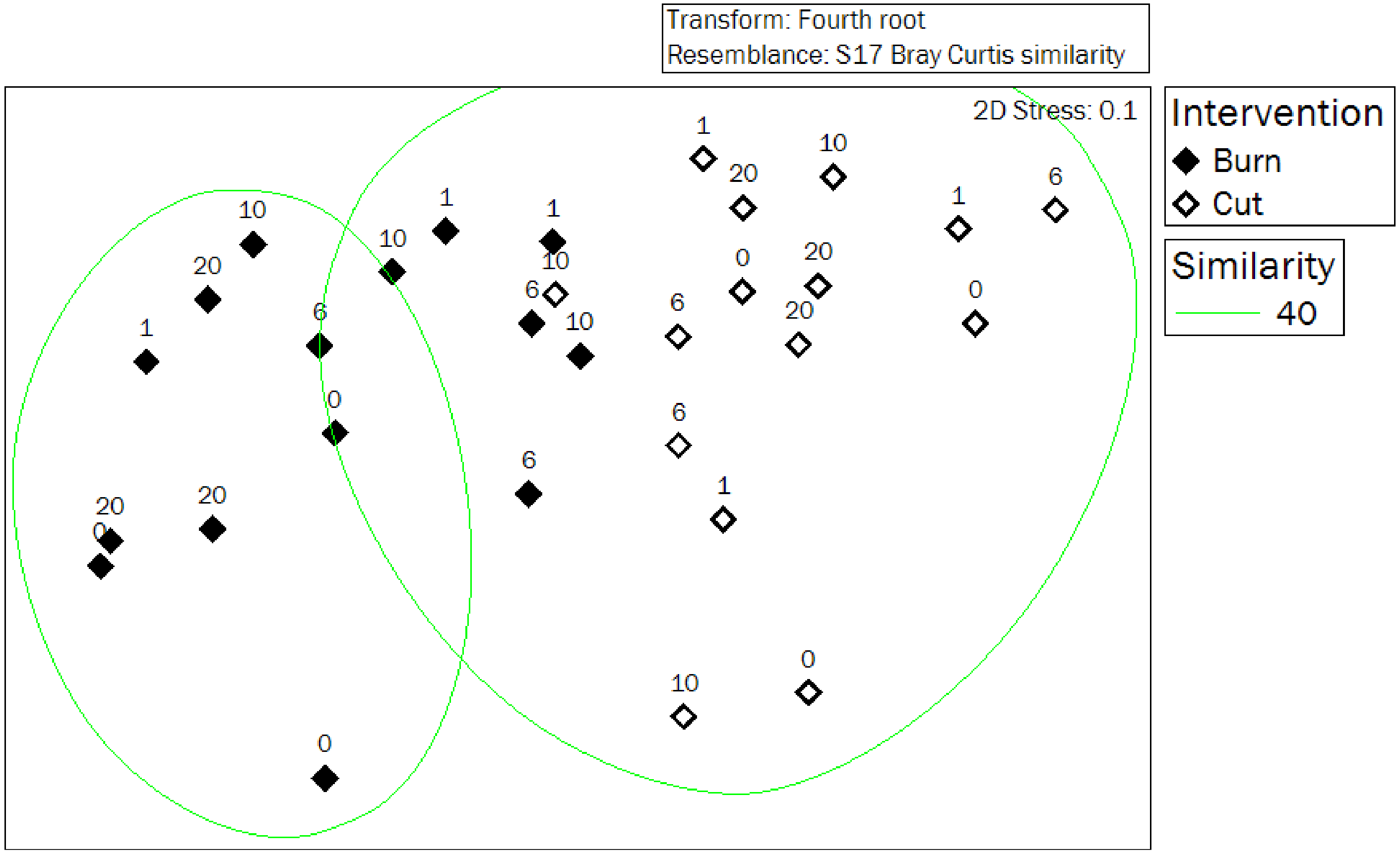
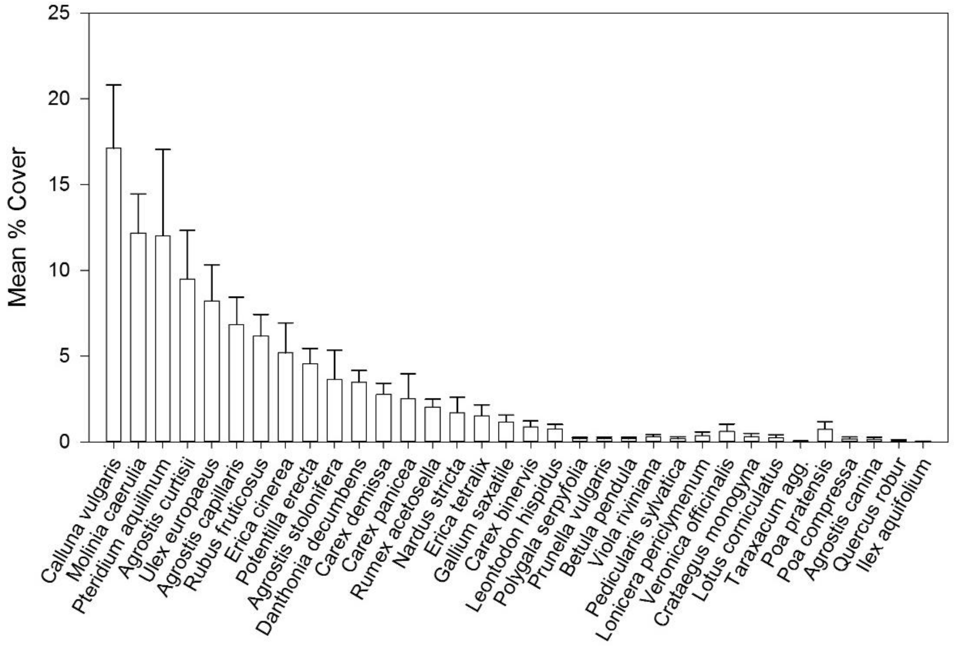
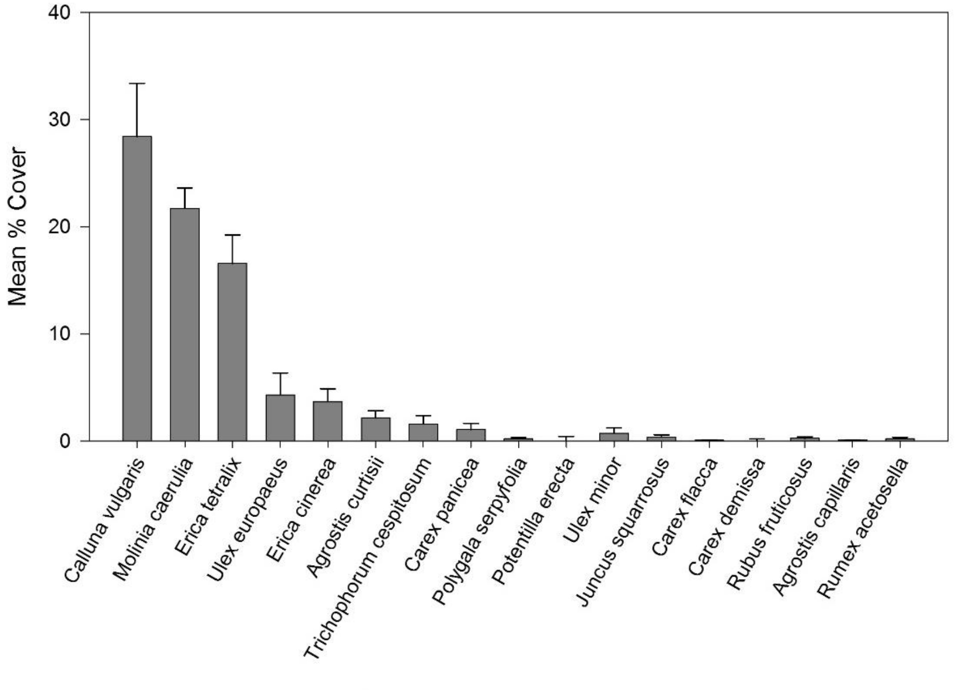


Figure 2.8 MDS Plot for habitat community data. Filled diamonds indicate burnt plots and open diamonds cut plots. Numbers indicate the age of the plot following management activity. The clear division of burnt and cut plots is visible. An arbitrary similarity level of 40% superimposed on the MDS is consistent with the cluster analysis, but also indicates that, despite the differences between cut and burnt plots, there is some overlap between groups.

Two-way crossed ANOSIM was undertaken on the Bray-Curtis similarity matrix that underpinned the MDS. ANOSIM detected statistically significant differences in the habitat community across management intervention (R = 0.563, p = 0.001) but none across age groups within the management interventions (R = 0.015, p = 0.42), supporting the general observations made from the MDS ordination that there were large differences between management types, but no detectable differences over the age of the plots. SIMPER analysis was performed to examine the role of individual plant species in contributing to the differences between management types and to identify where the community structure differed. The average dissimilarity between burnt and cut plots was 63.9%, based on the Bray-Curtis measure. A large number of species with relatively small individual abundances contributed to this pattern, with *Erica tetralix* (Bray-Curtis score and % contribution of 2.59, 4.06%), *Agrostis capillaris* (2.58, 4.04%), *Rubus fruticosus* (2.26, 3.78%), *Ulex europaeus* (2.35, 3.68%), *Potentilla erecta* (2.05, 1.78%), *Agrostis curtisii* (1.93, 3.01%), *Danthonia decumbens* (1.92, 3.01%), *Carex demissa* (1.92, 3.01%) and *Pteridium aquilinum* (1.92, 3.0%) providing about 31% of the dissimilarity. Other species contributed less than 3% each, but there were still many contributing species, and can be seen in Appendix II Table II.6. Additionally, all of these species were more abundant in cut plots, with the exception of *Erica tetralix,* which was more abundant on burnt plots. The average similarity score between burnt plots was 51.9%, with *Calluna vulgaris* (12.99, 25%), *Molinia caerulea* (12.6, 24.3%) and *Erica tetralix* (11.1, 21.4%) the typical species found in burnt plots. Cut plots featured a larger number of species with comparatively small contributions; the Bray-Curtis similarity score was 52.4%, with the species typical of cut sites including *Calluna vulgaris* (4.98, 9.5%), *Molinia caerulea* (4.53, 8.64%), *Ulex europeaus* (3.79, 7.23%), *Rubus fruticosus* (3.71, 7.08%), *Agrostis capillaris* (3.37, 6.43%), *Agrostis curtisii* (3.26, 6.21%), *Danthonia decumbens* (3.25, 6.20%) and *Potentilla erecta* (3.16, 6.02%). It is interesting to note that the two key species of *Calluna* and *Molinia* are abundant in both plots, while elsewhere there were considerable differences. The mean abundance of the most frequently occurring species in each group is shown in fig 2.9, highlighting these differences. Many species were present in the cut plots that were absent from the burnt plots, with *Pteridium*, *Agrostis* (multiple species), and *Ulex* featuring in the cut community to a large extent. In burnt plots, *Erica tetralix* was also an important component of the flora, while it was relatively uncommon in the cut plots. Many species associated with grassland communities were found in the cut plots, as well as some potentially ‘invasive’ species such as *Rubus fruticosus*,



a. frequent species on cut plots



b. frequent species on burnt plots

Figure 2.9 Variation in cover for vascular plant species that accounted for the majority of cover across cut (a) and burnt (b) plots. The graph shows all species covering more than one per-cent of the respective plots. Error bars show standard error.

2.3.4 To what extent do the management interventions account for the variation in the plant communities observed?

Outcomes from Constrained Correspondence Analysis (CCA) showed that the first two axes of the had eigenvalues of 0.46 (axis 1) and 0.26 (axis 2) explaining 28% of the species variation, and the overall solution was found to be statistically significant with Monte Carlo permutations ($p = 0.005$, n = 1000). Management activity was closely associated with axis 1, with plots clearly distributed based on the type of management (fig 2.10). Total N, P and Ca also showed an association with axis 1, suggesting, as ANOVA tests on abiotic conditions did, that management is associated with different concentrations of nutrients. Plots were also clearly separated, based on management type, within the ordination. Despite this separation, plots did show a great deal of variation in their position along the axis, which could potentially be attributable to unmeasured variables. Plots subject to cut management also showed greater variation along axis 2 than burnt plots. As axis 2 appeared to be related to both pH and vegetation height, some of the cut plots were associated with higher pH scores and low vegetation height, but others lacked these characteristics. The distribution of species is immediately noticeable (Figure 2.11), with the majority of the species clustered at the right hand side of the plot. A great deal of the position of the species appears to be related to axis 1, indicating that the majority of plant species were associated with higher soil concentrations of N, P and Ca and were also associated with cutting rather than burning. A number of species, however, clustered in association with burning and low concentrations of these nutrients Examining where individual species plot allows some additional characteristics of the plots to be inferred, based on the accepted life strategies of those species. There appear to be many species characteristic of mire or wet heath in the far left, typical heath centre-left, species-rich heath centre-right, and essentially acid grassland species on the far right (fig 2.11, see Appendix II table II.9 for abbreviations of species names). Such a vegetation gradient would be expected examining plots across the Forest, but it is interesting that cut and burnt sites appeared to be separated, both in the MDS and the constrained ordination shown here. As explanatory variables, plot age, elevation (in meters above sea level) and total soil carbon appeared to have little influence over the position of either species or plots.

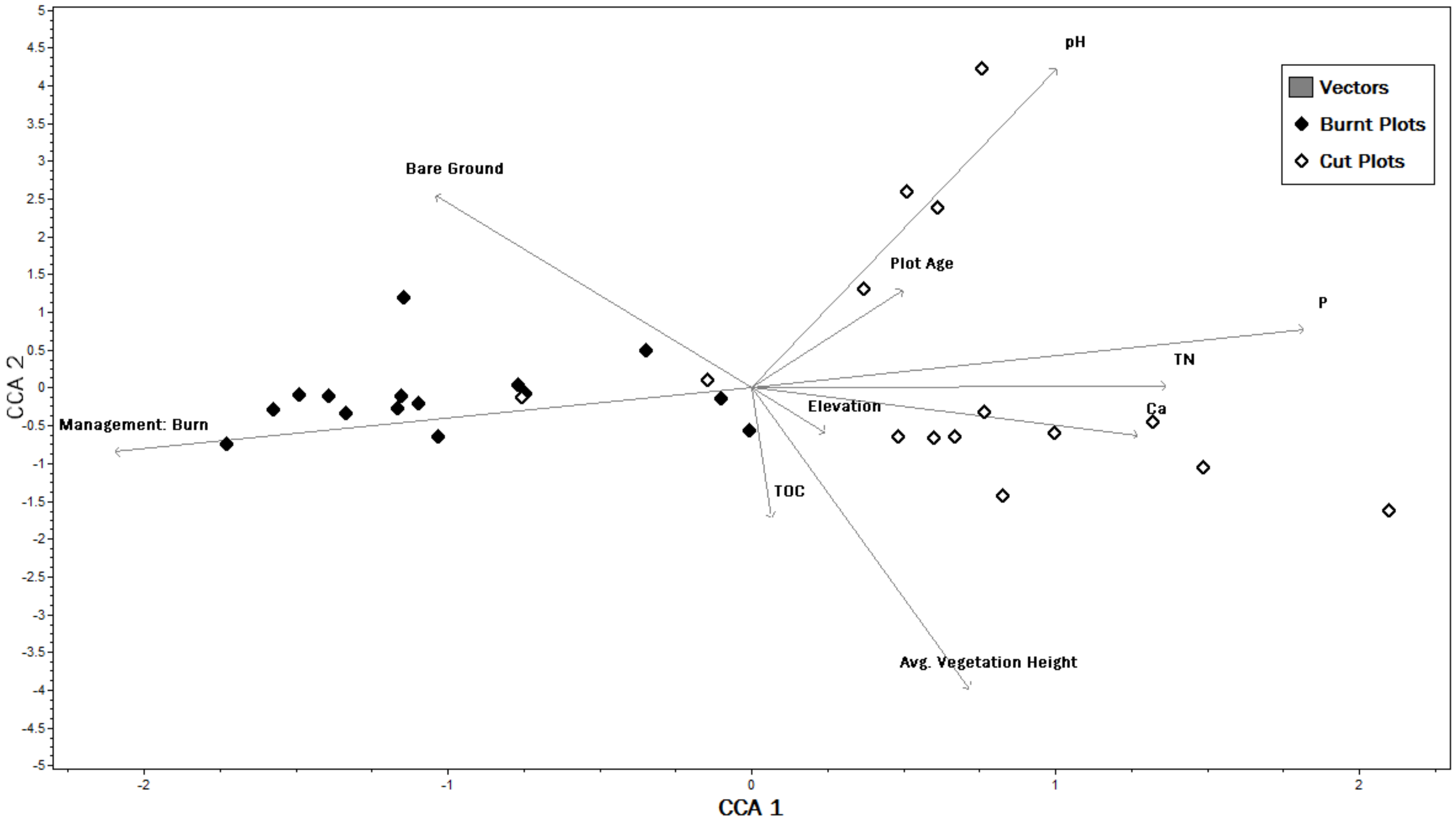


Figure 2.10 CCA biplot showing plots constrained by predictive variables. Species are not shown for clarity. Axis 1 (CCA 1) explains approximately 20% of the variation found, with axis 2 (CCA 2) explaining about 10%. Clear separation between burnt and cut plots is shown, most closely associated with management activity and soil nutrients. Soil pH also shows an association with vegetation on a small number of the cut plots.

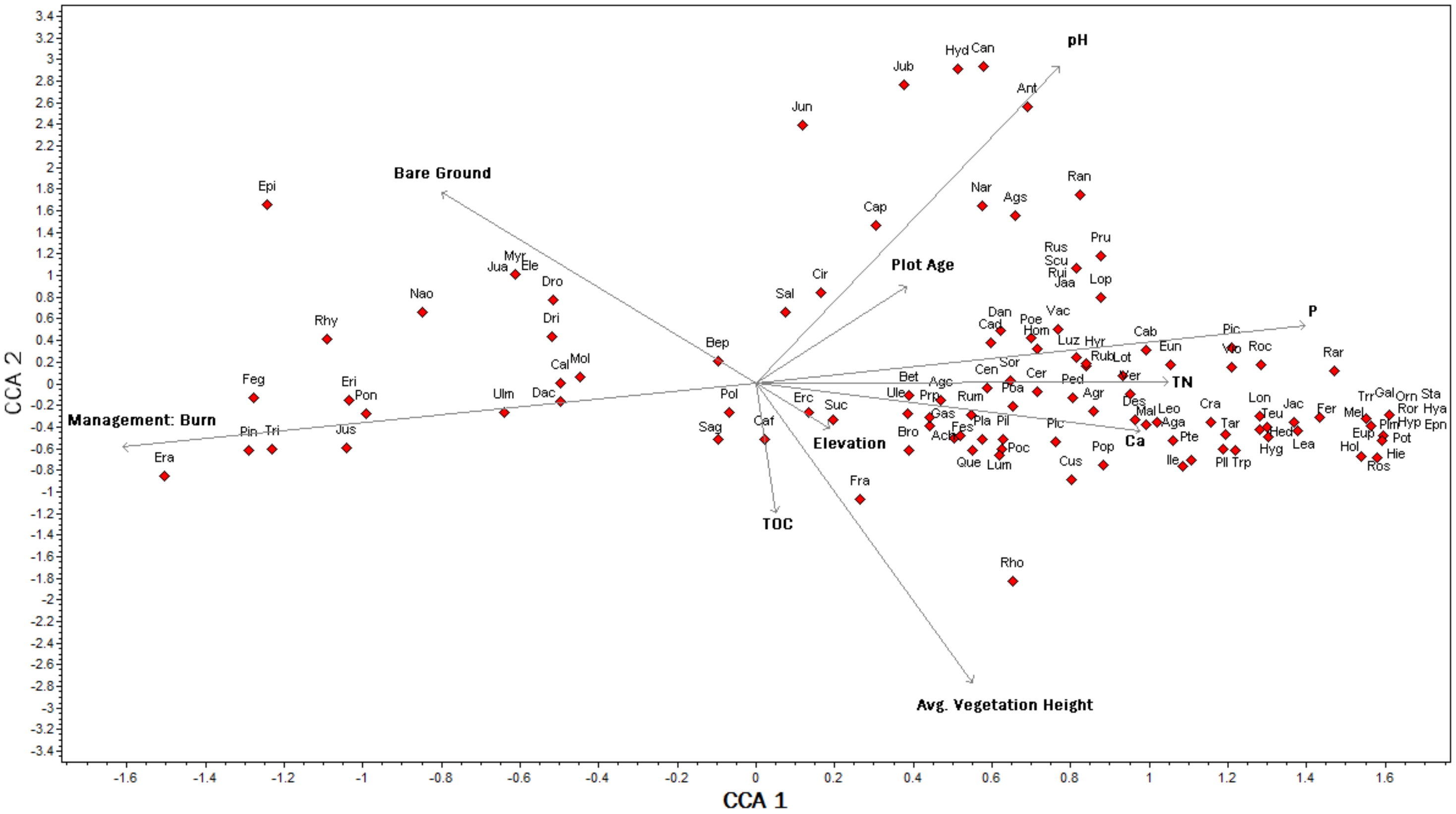


Figure 2.11 CCA biplot showing the position of vascular plant species constrained by predictor variables. Labels for the species are available in Appendix II, table II.9. The majority of recorded species are associated with higher nutrient concentrations, but several heathland specialists are placed closer to burning management activity and lower concentrations of soil nutrients.

2.4 Discussion

Because many aspects of heathland management are still unknown (Newton *et al.,* 2009), the aim of this research was to quantify the impacts that long-term management, through both cutting and burning, have had on heathland ecosystems. Unlike previous studies, replication is provided by separate heaths undergoing the same management interventions. Results have shown that there are large differences between management types for both the plant community and abiotic conditions, but that dynamics following management interventions were poorly resolved other than for ericaceous structure. Management appeared to be responsible for a large proportion of habitat variation, but there are also several other influences that may also influence the heathland community.

2.4.1 What are the differences between management types (cutting and burning) on a) the plant community, b) abiotic conditions and c) suitability for grazing?

Some striking differences in the vegetation community were observed between management types, both with ANOVA and MDS ordination analyses. In particular, ericaceous cover was substantially higher in burnt plots as opposed to cut plots, suggesting that burning is more effective in maintaining a high cover of these species than cutting. Both *Calluna vulgaris* and *Erica tetralix*, key plants of the New Forest 'humid heath', showed this pattern individually. Several studies examining management burns have reported effective regeneration of heather, as with the results here (Mallik & Girmingham, 1983; Marrs, 1988; Bakker *et al.,* 2004; von Plettenburg *et al.,* 2004; Nilsen *et al.,* 2005). In contrast to the ericaceous species, grasses showed greater abundance in cut plots. Similar results of low-*Calluna* and high grass cover have been reported after attempts at *Pteridium* control, including cutting (Marrs & De Luc, 2000), with high spikes in grass cover following cutting compared to burning in central European heaths (Sedlákova & Chytrý, 1999), and poor regeneration of *Calluna* in exclusively cut sites (Britton *et al.,* 2000c). Growth of *C. vulgaris* can be inhibited under high-intensity mowing (Bakker *et al.,* 2004), whereas grasses can compete effectively after cutting has removed the wood or shrubby canopy (Britton *et al.,* 2000a; Calvo *et al.,* 2007). Collectively the results in this study suggest that burning may provide a habitat where ericaceous species have a competitive advantage over grasses, when compared to cutting, as supported by prior research (Alonso *et al.,* 2001; Mohamed *et al.,* 2007). An exception is the cover of *Molina caerulea*, which was less abundant in cut plots. Several other studies suggest that cutting inhibits the spread of this species, which the evidence here would appear to support (Grant *et al.,* 1996; Jacquemyn *et al.,* 2005; Critchley *et al.,* 2008; Hejcman *et al.,* 2010).

Potentially negative influences of burning were also detected by this study, as species richness and S-W H' diversity showed higher values for cut plots. High diversity scores have previously been reported for grass-dominated habitats in comparison to *Calluna* dominated habitats elsewhere (Bartolomé *et al.,* 2005). There are also suggestions that there is a risk that burning may lead to species-poor communities with the loss of rare heathland plants, as found in some studies (i.e. Velle *et al.,* 2014). In this study, some of these rare plants, including *Nardus stricta*, *Pedicularis sylvatica, Cuscuta epithymum* and *Euphrasia* sp., were all more frequent in cut plots. However, some others, among them *Trichophorum cespitosum, Ulex minor,* and *Juncus squarrosus*, were much more likely to be found in burnt plots. Much of the increased diversity in cut plots is attributable to non-heathland species (including some scrub species), and restoration of areas with high densities of these species could be problematic (Mitchell *et al.,* 1998; Calvo *et al.,* 2007). Increases of these species have previously been observed following cutting of heaths elsewhere (Calvo *et al.,* 2007). Overall, the evidence here and in other studies (Velle *et al.*, 2014) suggests that important heathland specialists are probably maintained by burning despite reductions in diversity indices. However, some caution should remain about this conclusion, because of the lack of some species in the burnt plots, and a general community similarity across the burnt areas, which is not ideal for fauna that require diverse heathland environments (Webb, 1986).

As with the vegetation community, there were clear contrasts between management types when examining abiotic features. Soil pH scores showed a significant but small difference between the burnt and cut sites. These were both slightly less acidic than managed heaths in Dorset (Mitchell *et al.,* 1997; Mitchell *et al.,* 1999) but are similar to some mainland European heaths (Jansen *et al.,* 1996; Marcos *et al.,* 2009). Extractable nitrogen, both in the form of ammonium and nitrate, appeared to be much greater than in Dorset heaths (Mitchell *et al.,* 1999), and was mostly ammonium, as would be expected in heathland soil (Troelstra *et al.,* 1995). All plots featured elevated nutrients compared to Dorset heathlands, supports observations by Tubbs (2001) suggesting that New Forest heaths were not nutrient deficient. The important nutrients N (including plant-available N) and P (and also Ca and Mg) showed higher concentrations in soil beneath cut plots, compared to burnt plots, and this was also reflected with soil C/N ratios, which were considerably greater on burnt plots. However, even C/N ratios on cut plots were greater than some mainland European heaths where nutrient enrichment has occurred (Jansen *et al.,* 1996). Although burning does remove nutrients from the ecosystem (Hardtle *et al.,* 2007; Mohamed *et al.,* 2007), such changes are often relatively small or absent in low-intensity burns (Marcos *et al.,* 2009; Green *et al.,* 2013), so the stark differences here are relatively unusual. Results here also differ from evidence by Härdtle *et al.* (2009) that mowing resulted in a higher loss of P than prescribed burning. Such a contrast between the two management types could be a result of repeated management over many cycles, as most prior studies, such as the one previously mentioned, have only examined one cycle on short timescales. Because high N concentrations are associated with *Molinia* encroachment on some heathland communities (Falk *et al.,* 2010; Aerts & Berendse, 1988), and higher P concentrations may also allow encroachment of grass (Falk *et al.,* 2010), the findings would appear to give support for assertions that burning maintains typical abiotic conditions more effectively than cutting.

Because of the measured differences in grasses between management types, cut plots were likely to be more suitable for grazing than burnt plots. Grasses are favoured by horses (Menard *et al.,* 2002), and studies in the New Forest have supported these observations (Putman *et al.,* 1987). Interestingly, the main food source for ponies is believed to be *Molinia caerulea* (Putman, 2010), so the greater cover of *Molinia* within burnt plots actually complicates conclusions about the provision of grazing. One explanation is that, because grazing reduces the cover of *Molinia* (Milligan *et al.,* 2004), it may be replaced by other species if grazing pressure is particularly intense, such as *Agrostis capillaris*. *Molinia* was still very abundant in cut plots, and additionally, the concentrations of nutrients within *Molinia* tissue were greater in cut plots for P, Mg and K. These differences closely followed those observed in the soil, and suggest that grass in the cut sites was a richer resource for grazing animals. Concentrations were, however, within the range reported by Aerts (1989) for other heathlands. Clarke (1988) has suggested that management for grazing puts an unnecessary focus on seasonal productivity, which, together with focus on *Molinia,* may create the impression that burning is better among managers, despite the observations made here.

2.4.2 What are the dynamics of the heathland community following management?

Following management, relatively few of the variables measured showed a consistent relationship with the age of the plots. Differences in the ericaceous age structure, however, were particularly noticeable, and this showed a complex, interactive pattern between management and age following the interventions. Pioneer cover of heath was especially abundant in the earliest burnt plots, whereas it featured with a low but consistent cover in cut plots. The consistent cover of pioneer heath is similar to some other cutting studies, where *C. vulgaris* was kept in an immature state (Le Duc *et al.,* 2007). Building heath showed an increase in cover as plots aged, and a decline in the oldest plots. This closely follows the model of *Calluna* regeneration described by Watt (1955), which was also reported by Bullock and Webb (1995) following a destructive fire. Although building heath cover was, overall, greater in burnt plots, the pattern is similar for both interventions. Degenerate heath showed an increase as burnt sites reach older stages (10-20 years), but is negligible in other plots. Degenerate heath is thought to recover poorly when it is burned (Allchin *et al.,* 1996; Velle *et al.,* 2012), so the increase in the very oldest plots would suggest that burning intervals are timed to maximise ericaceous recovery. However, the low cover of degenerate heath, except in the very oldest plots, indicates poor conditions for several reptiles and invertebrates, which utilise this habitat structure (Spellerberg & Phelps, 1977; Edgar *et al*., 2010). Dead heath cover was very low in all plots, with a slight increase in the very oldest burnt plots only, so interventions appear to prevent the appearance of this structural component.

The cover of bare ground, in addition to its use by a number of heathland species (Webb, 1986), may also help the establishment of *Calluna* following disturbance to the habitat (Britton et al., 2000b). In this study, cover showed a dynamic effect, with early-stage burnt plots featuring the highest amounts of bare ground. In cut plots, cover was consistently low, and a similar pattern can be observed in the older burnt plots. This suggests that only the active intervention of burning is effective at creating this cover type, which supports findings by Britton *et al* (2000b) that cutting is poor in providing this habitat feature. A potential mechanism proposed for this result is the destruction of mosses and litter cover by burning (Velle *et al.,* 2012). However, both mosses and litter cover did not vary between sites or age in this study, and bare ground decreases very quickly in burnt plots towards similar cover levels as that found in cut plots. Such a short lived response may not actually be beneficial for the wildlife that utilise bare ground in heathland environments, and it may be that the isolated conditions found in later plots, with low but persistent presence of this feature, are more suitable.

Within the rest of the plant community, including diversity measures, very little variation was observed over plot age, both from ANOVA tests of variables and ordination analyses of the community structure. This suggests that plots remained similar to their initial communities throughout the recovery from disturbance. Although the initial community structure is often considered the prime driver in determining the dynamics of succession (Vandvik *et al*., 2005), especially for heaths (Mohamed *et al.,* 2007), this result was very different to other studies which showed strong successional trends in vegetation following fire (Mallik & Girmingham, 1983; Vandvik *et al.,* 2005; Britton & Fisher, 2007; Borghesio, 2009). These studies looked at shorter timescales, however, which may be a reason for the difference in observations. The age of the ericaceous structure prior to intervention can also have an influence on the outcome (Velle *et al.,* 2012), which could affect how the species responded. Vandvik *et al*. (2005) found that trends in succession following fire were not affected by grazing, but it may be the case that with very high pressure in the New Forest (Putman, 2010) such interactions are different. Support for this view is provided by the lack of change in vegetation height over site age, in contrast to other studies examining regeneration on heathlands (e.g. Nilsen *et al.,* 2005; Vandvik *et al*., 2005). If high grazing pressure reduced vegetation height at enough of the older plots to make statistical comparisons difficult, and this appears to be the case, it is also likely to impact on many of the other measures. Potentially, the rapid growth of grasses on some of the younger plots likely also added variability to this measure, as plots were not examined immediately following winter burning but several months later. Soil chemistry is also affected by the pre-treatment species composition (Härdtle *et al.,* 2009), and repeated management over many years may have reduced the likelihood of obvious trends being observed over the 20-year timescale investigated here. Unfortunately, the lack of site history records makes this potential connection difficult to prove.

2.4.3 To what extent do the management interventions account for the variation in the plant communities observed?

About 30% of vegetation variance was explained through CCA, and the results suggested that the differences in interventions and their associated soil conditions were the most important influences on the vegetation community (approximately 20% of variance), despite the lack of successional trends observed. The majority of recorded species were associated with cutting and high nutrient concentrations, including *Agrostis capillaris* and *A. curtisii,* but many other contributing species were typical of grassland or scrub communities. Some typical heathland species were not associated with either management type, including *Erica cinerea* and *Ulex europaeus*, which are found in other locations in the New Forest. Associated with burning were *Calluna vulgaris* and *Molinia caerulea,* with wet heath species most strongly associated with this management type. Other important influences on the vegetation community included vegetation height and pH. Vegetation height may be an indicator of the grazing pressure that plots have been subjected to, although it did not appear to be as significant as the management activity itself. High pH values were associated with some species typical of flushed, wet conditions (*Hydrocotyle vulgaris* and *Ranunculus repens* among them). The observed trends would provide support for assertions that the management interventions here are the principal cause of structural changes, and that grazing has different effects to burning and cutting. However, direct measures of grazing intensity (and probably further investigation of the impacts) would be required to confirm these observations. Several characteristic wet heath species (most notably *Erica tetralix*) were associated with burning. This probably a result of wet heath areas being avoided for cutting management, as they cannot easily be mown. Other than some change in species between management areas observed here, there do not seem to be any other effects of this potential bias, but results should be considered cautiously. Better understanding of the land-use history of each plot location would help to address these concerns.

2.4.4 Management Recommendations

This work was initiated by concerns about the effectiveness of heathland management, particularly burning. The substantial differences in community between cut and burnt sites suggest that the practices are not interchangeable, and considerable variation in the resulting community is to be expected depending on the type of management applied. While difficult to ascertain whether these differences were a direct result of management or whether they are influenced by site selection or other variables, it does demonstrate that the techniques cannot be substituted for each other. Some important heathland species appeared to be exclusive to either burnt or cut sites, so perhaps, as concluded by several other heathland researchers (Marrs & Britton, 2000; Vandvik *et al.* 2005; Newton *et al.,* 2009), a general management proscription is the wrong approach for conservation in a complex habitat. In this case, a mixture of the two techniques may be necessary for conserving the full diversity of heathland habitat in the Forest, utilising site-specific characteristics as suggested by Britton *et al.* (2000b) and Velle and Vandvik (2014), and carefully preserving important landscape features as suggested by Pereoglou *et al.* (2016). A good way of determining precisely how to adjust management would be to adopt an adaptive strategy, examining combinations of the two interventions as well as other alternatives. If structured together with scientists and managers, any problems with local site conditions or their spatial location could be addressed. While designing an experimental approach would be incredibly valuable, there would be serious barriers to its implementation, likely at the level of the regulatory authority. This is particularly the case where there is an obligation to protect habitat, and the perception of failure if experimental treatments do not succeed. Such concerns have been highlighted elsewhere (Lindenmayer *et al.,* 2012), demonstrating the important role that policy has in determining conservation efforts and the methods used to achieve them. The lack of knowledge about the effectiveness of heathland management, addressed in part by this research, underlines the need for monitoring to verify positive and negative impacts of management. Effective monitoring could contribute to the evidence base available to heathland managers and improve future conservation efforts, but the effectiveness of Common Standards Monitoring (CSM) in determining management impacts has not previously been assessed.

2.4.5 Conclusion

This study compared the effectiveness of different heathland management activities which have taken place in a protected area for decades, in an effort to address the lack of long-term, fully replicated studies. In a novel approach, a 20-year space-for-time substitution has shown management replications over several different heathlands. Results have shown that the current practices in the New Forest have significantly different outcomes, but both have advantages and disadvantages. Controlled burning appears to maintain the superficial structure and species for the heath, but concerns remain about its potential impact on rare species and its ability to control *Molinia*. Cutting, although successfully delaying succession, suffered from high cover of grasses and an influx of non-heathland species. Below ground comparisons also indicated differences in soil chemistry, although whether this was attributable to the management cycle examined here or multiple cycles was difficult to say. Grazing provision was complex, but it appears that cutting provided more extensive browsing compared to burning. Finally, this work also provides a basis to assess the monitoring approach to heathland management, and examine whether the changes detected here are replicated by the monitoring approach.

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Chapter 3: Monitoring of Heathland sites in the New Forest and the suitability of habitat for key taxa

3.1 Introduction

3.1.1 *Protected Area Monitoring in the UK*

Protected areas are an important cornerstone of global efforts to conserve biodiversity, but must be appropriately managed in order to meet their goals (Chape *et al.,* 2005). In recent years, concerns over the effectiveness of protected area management have spurred greater efforts to monitor and evaluate conservation success (Jackson and Gaston, 2008). Questions around how to improve conservation efforts, and how to most effectively measure progress towards conservation goals, have only recently become the focus of scientific research (Salafsky *et al.,* 2002). One way in which conservation practices are now being refined and improved is that of adaptive management, where testing the effectiveness of management activities helps to shape the management strategies themselves (Sutherland *et al.,* 2004). However, in order for this method to be successful, detailed monitoring and evaluation of interventions are necessary (Salafsky and Margoluis, 2003; Sutherland *et al.,* 2004; Stem *et al.,* 2005). Monitoring must also take a quantitative, not qualitative, approach if it is to be useful for evidence-based management (Legge, 2015).

In order to assess the effectiveness of conservation in the UK's protected areas, the Joint Nature Conservation Committee (JNCC) implemented the practice of Common Standards Monitoring (CSM). This technique was piloted in 1998 and fully implemented across the UK in 1999 (JNCC, 2006). CSM was designed in response to extensive criticism of the state of the UK's protected areas, particularly Sites of Special Scientific Interest (SSSI), which appeared to be suffering continuous damage, and in some cases designated features had been completely destroyed or lost (Alexander and Rowell, 1999). Much of this damage had been attributed to a lack of knowledge, and so CSM was intended to serve as a rapid and robust method for assessing the general condition of internationally and nationally designated sites (Williams, 2006). An additional purpose was to point out where future action or more detailed monitoring may be needed (Williams, 2006). CSM now also serves as a reporting tool for progress on international commitments, such as the European Directives.

CSM requires objectives for conservation at the site level together with management plans (Rowell, 1991), which are important as they determine what is specifically monitored. CSM examines conservation features for which sites were designated but not necessarily all features present on the site. Conservation features include species, habitats and other environmental variables, and these are assessed at least once within a 6 year period. In England this work is undertaken by Natural England, with each SSSI divided into units and assessed against a set of targets. Each site is also required to have a management plan that determines the targets to be assessed, a key feature of CSM and its link to evidence-based management. After the assessment is carried out, sites are categorised as being in 'favourable' or 'unfavourable' condition, with a number of subcategories. These are intended to help determine the general direction of progress (or degradation) on the site and identify where future management or monitoring may be needed. In practice, an assessment by a surveyor is carried out by a visual assessment (Cantarello & Newton, 2008). The assessor conducts a structured walk across the habitat feature and completes a questionnaire by recording scores based on criteria specific to the NVC community represented at the site (JNCC, 2004). Williams (2006) compares this to giving a quantifiable 'expert opinion'. However, the categorisation of sites into favourable or unfavourable status has been considered subjective (Gaston *et al*., 2006), and in particular the subcategory 'unfavourable recovering' is highly contentious as this is often given to sites in poor condition solely because they have been subject to management, without any reference to the outcome of that management (Williams, 2006).

While the role of CSM is understandably important, there has been very little critical evaluation of the method, especially in the scientific literature. Ross & Bealey (2005) performed validation studies examining the CSM approach with more quantitative techniques, but did not have any component of management activity in their study. Cantarello and Newton (2008) provided one critical study, demonstrating that the visual assessment compared poorly in comparison to more quantitative approaches in forested habitats. Another by Davies *et al*. (2007) suggested that CSM lacked enough sensitivity to identify more complex habitat conditions that important species may need, and questioned its effectiveness in these areas. While CSM may be useful for an overview of habitat status and reporting on policy commitments, its sufficiency for evidence-based conservation is unknown, despite its link to management being described as its 'great advantage' (Alexander and Rowell, 1999). In many areas, such as the New Forest, it is the *only* method used for monitoring management effectiveness. In these situations, the CSM approach is largely untested.

3.1.2 New Forest management goals

The New Forest is protected as a SSSI (Site of Special Scientific Interest, under the Wildlife and Countryside Act 1981 and Countryside and Rights of Way Act 2000) European Special Protection Area (SPA) and Special Area of Conservation (SAC), designated under the Birds Directive 79/409/EEC and Habitats Directive 92/43/EEC, in addition to its designation as a Ramsar Site (under The Convention of Wetlands of International Importance, Ramsar, Iran, 1971. Despite extensive management activities, a recent study of the New Forest by Cantarello *et al*. (2010) demonstrated that a low percentage of habitats were in favourable condition according to CSM scores. However, some descriptions of the New Forest have emphasised gains in 'favourable' area when compared to other sites in the UK (Natural England, 2012).

The New Forest has several plans and agreements in place to determine management goals, interventions, and monitoring. The New Forest SAC Management Plan (Wright and Westerhoff, 2001), provides as an overview of management goals for SAC habitats, setting the goals of CSM. All conservation designations are included as part of the SAC plan, forming a unified approach to management and monitoring. The conservation objective for New Forest heathlands is to 'maintain' habitats in favourable condition and 'restore' habitats in unfavourable condition. The condition assessment criteria are identified as the national guidelines (i.e. those by the JNCC), with extra reference to breeding bird populations, for which the SPA designation is given. These management objectives are also reflected in the HLS Stewardship Agreement for the New Forest (2010), which repeats the goal of maintaining and restoring habitat to favourable condition, and refers to the SAC management plan. Management of the Forest's habitats is therefore very closely related to CSM and condition assessment, and management strategies are based on the results of CSM. Monitoring is undertaken on a unit-by-unit basis by Natural England, including for the larger designations which overlap the SSSI. There are a total of 582 units (Cantarello *et al.* 2010. Additional monitoring for breeding birds consists of population surveys, but not measures of the extent or condition of available habitat (HLS Agreement, 2010; Goater *et al.,* 2004). Because management planning is directly related to the monitoring technique through the use of favourable or unfavourable designations, the effectiveness of CSM is of even greater importance than it would otherwise be.

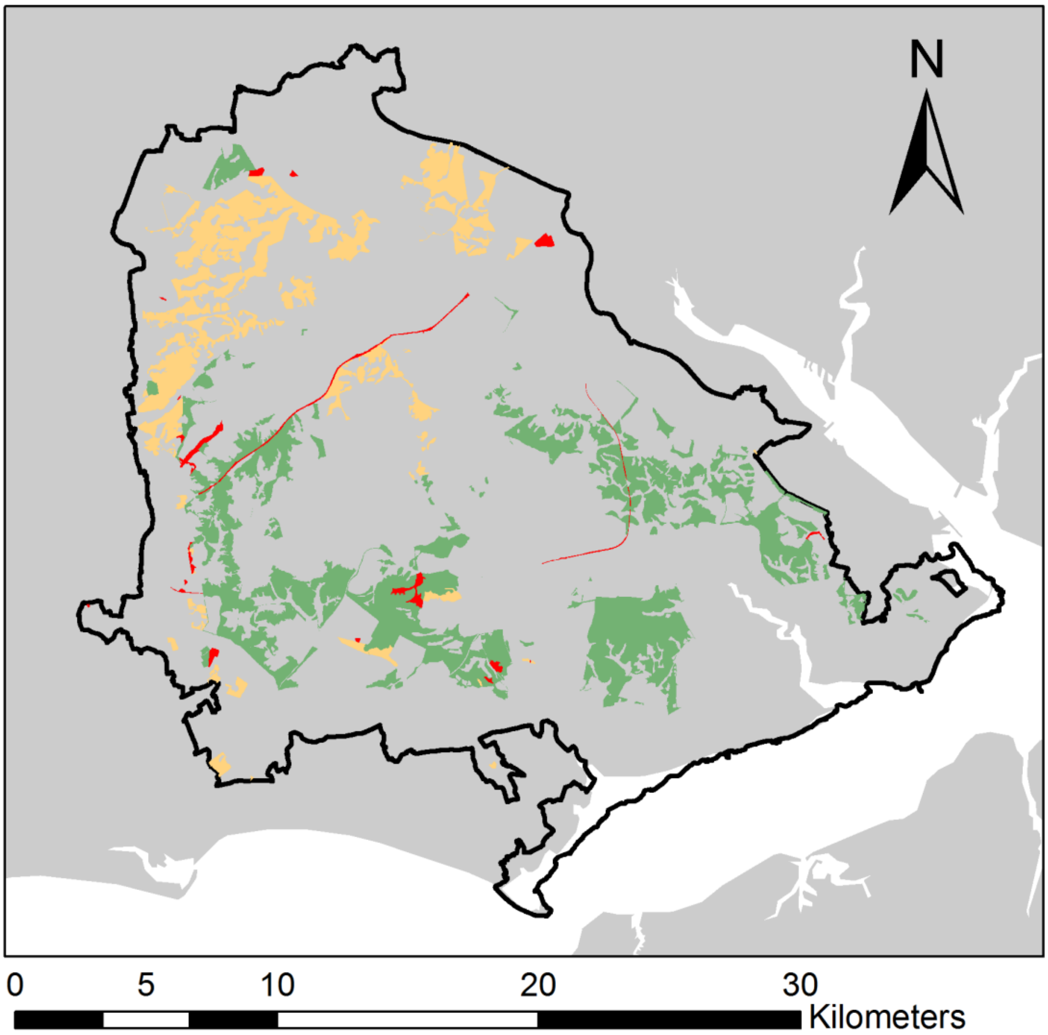

Figure 3.1 The locations of Heathland SSSI sub-units in the New Forest, showing those in "Favourable" status (Green), "Unfavourable recovering" (Yellow), and all Unfavourable (Red) in the 2013 assessment. The black outline indicates the National Park boundary. The two noticeable red lines are small patches of heathland alongside major roads through the forest.

3.1.3 Species Monitoring and Habitat Suitability Indices

Although a potential weakness of CSM is its exclusive focus on plant communities (Davies *et al.,* 2007), one of the advantages of this approach is this focus can potentially provide an insight into the provision of habitat for species of other taxonomic groups. However, its effectiveness in this regard is largely untested. Habitat Suitability Indices (HSIs) offer a possible way to test whether CSM records important habitat characteristics that are useful for fauna. Specifically developed to determine the effects of management on wildlife habitat (Roloff & Kernohan, 1999), HSIs quantify an organisms life requirements using the composition, structure and spatial components of its habitat (United States Fish and Wildlife Service, 1981). HSIs usually take the shape of simple mathematical equations for calculating habitat quality based on habitat components that reflect life requirements, which are combined to form a rating of habitat suitability (Brown *et al.,* 2000; Brooks 1997). Component variables typically include feeding, water and shelter requirements (Brooks, 1997). HSIs originated in the United States for use by the Fish and Wildlife service (United States Fish and Wildlife Service, 1981), but similar approaches have gained acceptance in Europe (Dettki *et al.,* 2003; Glenz *et al.,* 2001). Such models are supported by ecological theory based on habitat selection, niche partitioning and limiting factors (Verner *et al.,* 1986; Morrison *et al.,* 1992), but require detailed information to produce reliable results (Roloff and Kernohan, 1999).

Many heathland species are appropriate candidates for HSIs, and data collected from a HSI approach may give information about how management modifies the provision of habitat suitable for these taxa. To test CSM effectively, it is important to include a range of taxonomic groups, as important habitat characteristics can differ considerably. Additionally, a frequent criticism of environmental monitoring is that it focusses on few taxonomic groups, particularly plants (Perring *et al.,* 2015). In this study, three species of diverse taxonomic affiliation were examined: smooth snake *Coronella austriaca,* silver-studded blue butterfly *Plebejus argus*, and Dartford warbler *Sylvia undata*. Smooth snake and Dartford warbler are exclusive to heathland in the UK and nationally important populations are present in the New Forest (Tubbs, 2001). Dartford warblers are categorised as near threatened by the IUCN and are declining globally (BirdLife International, 2012). Silver studded blues have experienced a great decline in range across the UK (Ravenscroft & Warren, 1996), but can be locally abundant. These techniques could also help to address potential ecological concerns with burning, such as the loss of reptile communities or structural diversity that current CSM monitoring may not detect.

3.1.4 Research Aim and Objectives

Understanding monitoring effectiveness is critically important to ensure success in meeting conservation goals. It is therefore important to determine whether past management work has influenced condition, whether such changes are desirable, and how CSM assessment compares to more robust measures of habitat composition and structure. Answering these questions will uncover whether improvements in favourable status, which have recently been described for the New Forest (Natural England, 2012), really mean that management has been effective. In particular, heathland habitats are of great importance for several reasons: a highly restricted distribution, the presence of endemic species, and that it was found to be in the worst condition of all UK habitats after the initial 6 years of CSM (Williams, 2006).

The overall aim of this research is to determine whether CSM monitoring of heathland is an effective approach for monitoring the impact of management interventions. Specifically, the research aims to answer the following questions:

i. Do habitat condition scores show measurable differences following management activity?
ii. How well do habitat condition scores and their constituent attributes reflect community and structural changes observed in the habitat community?
iii. How do CSM scores compare to Habitat Suitability Indices when there are changes in the habitat?

3.2 Method

3.2.1 Study Design

The research design from Chapter 2 was used as a basis for examining CSM in heathland habitats. Condition scores were calculated for each site based on the Common Standards Monitoring Guidance for lowland heathland habitats (JNCC, 2004). The principal NVC habitat types studied here are H2 *Calluna vulgaris – Ulex minor* heath and H3 *Ulex minor - Agrostis curtisii* heath, which are identified by Wright and Westerhoff (2001) as the important heathland communities to be protected by management. H2 is found on drier habitats; M16 *Ericetum tetralicis* wet heath community may also be present in waterlogged areas but is more characteristic of mire locations. H3 is intermediate between the H2 and M16 communities (Rodwell, 1991). A detailed description of these habitat types is available in Appendix III. The JNCC outline a scoring system that differs between 'wet' and 'dry' heath for lowland communities (JNCC, 2004), outlined in tables 3.1 and 3.2. Generally the heath habitats examined here were typical of dry conditions, although some wet heath species were present in several of the plots. The dry heath test was therefore used as the main assessment of condition, but wet heath tests were also carried out for comparative purposes and are shown in Appendix III.

There were some differences between how CSM would be conducted in practice and how it was assessed here. This was because of the need to record quantitatively so that scores could be assessed statistically, and the changes here represent a reduction in subjectivity so that differences could be clearly shown and the methods examined without bias. In CSM, several stops would be made along a structured walk where the requirements would be visually assessed, but in this study observation points were randomly distributed, and visual assessment was aided with 2x2 m quadrats. Scores of 1 (conditions met) and 0 (conditions not met) were calculated for each quadrat based on the attributes provided by the JNCC advice. Some of these attributes had multiple conditions to be met in order to give a positive score; causes of failure in these attributes were examined further. Scores were then summed to give a 'total condition score' indicating how many of the attributes were met. These total scores are representative of the final output from CSM, where a category would be assigned based on meeting all attributes, but differ as they are numerical and less subjective than the assessment in practice. One attribute, changes in habitat extent, was not calculated for this study as a result of a lack of baseline data. It will be possible to measure future changes, however, as suitable aerial photography of the sites is now available. Differences between the approach used here and in practice are investigated by comparing final condition scores with the categories for sites from the 2013 assessment and explored further. Unfortunately, data on individual attribute scores was not available from Natural England, so specific comparisons with these could not be made.

Habitat suitability scores were calculated based on a literature search for the selected species. Several important habitat characteristics were identified, and an appropriate threshold identified to score these attributes. Information about the habitat requirements, scoring system and sources is available in Appendix IV, together with a detailed view of the attributes and the sources of data. HSIs were subsequently compared to CSM condition scores to examine how the two related to each other. A number of habitat characteristics, measured in the second chapter, were also compared with condition scores. These included vegetation height, the total cover of ericaceous species, graminoids and bryophytes, and species diversity and richness. Measures were recorded for each plot and correlated against condition and HSI scores. These characteristics were also examined through further correlation analysis to all of the other measured variables to examine how informative they were as ecological descriptors of the sites.

3.2.2 Statistical Analysis

Data were analysed with R 3.2.3 (R Core Team, 2016), using the following packages: multcomp (Hothorn *et al.,* 2008), plyr (Wickham, 2011), reshape2 (Wickham, 2007), and agricolae (de Mendiburu, 2016). Additionally, ggplot2 (Wickham, 2009), gridExtra (Auguie, 2016) and ggrepel (Slowikowski, 2016) were used to construct graphical output. 2 x 5 factorial ANOVA was used to determine whether condition differed between management types, between the age of the plots, or whether there was a significant interaction between the two factors. Age (0 years, 1 year, 6 years, 10 years and 20 years since intervention) and management type (burning or cutting of the plots) were used as factors. These tests were performed on overall condition score and for individual attributes which contributed to this score. Assumptions of normality were tested with Q-Q plotting of residuals, and homogeneity of variance with Levene's test; outliers were examined using boxplots. Tukey's HSD post-hoc was performed where ANOVA results were considered worthy of further investigation (p = ~ <0.05). These were followed up with post-hoc tests. Eta-squared effect sizes for ANOVA tests were calculated using the lsr package (Navarro, 2015). Where correlations were used, Kendall's Tau was the preferred test, as this is robust compared to other measures (Croux & Dehon, 2010).

Table 3.1 JNCC Scoring system for Dry heath habitats (JNCC, 2004)

Category	Attribute	Scoring
Habitat Extent	Bare Ground	1-10% Cover
Vegetation Structure	% Cover of dwarf shrubs	Cover 25-90% Species include: *Arctostaphylos uva-ursi, Calluna vulgaris, Empetrum nigrum, Erica ciliaris, E. cinerea, E. tetralix, E. vagans, Genista anglica, G. pilosa, Ulex gallii, U. minor, Vaccinium myrtillus, V. vitis-idaea* (and hybrids).
	Cover of *Ulex* sp.	Total cover of *Ulex* and/or *Genista* spp. <50%. *Ulex europaeus* <25%
	Growth phase composition of ericaceous cover	Pioneer phase (including pseudo-pioneer): 10-40% Building/mature phase: 20-80%; Degenerate phase: <30% Dead: less than 10% of ericaceous cover
Vegetation Composition	Dwarf Shrub Species	At least two species present Frequent abundance
	Graminoids	At least one species frequent 2 species occasional *Deschampsia flexuosa* and *Nardus stricta* <25% cover, no more than occasional Species include: *Agrostis* spp.*, Ammophila arenaria, Carex spp., Danthonia decumbens, Deschampsia flexuosa, Festuca* spp.*, Molinia caerulea, Nardus stricta, Trichophorum cespitosum*.
	Desirable forb species	At least 2 species occasional Species include: *Armeria maritima, Galium saxatile, Genista anglica, Hypochaeris radicata, Lotus corniculatus, Plantago lanceolata, Plantago maritima, Polygala serpyllifolia, Potentilla erecta, Rumex acetosella, Scilla verna, Serratula tinctoria, Thymus praecox, Viola riviniana*
	Bryophytes and Lichens	% cover maintained or increased
Negative Indicators	Signs of disturbance	<1% of habitat heavily eroded.
	Negative indicator species	<1% exotic species exotics include: *Rhododendron ponticum, Gaultheria shallon, Fallopia japonica*. < 1 % ragwort, nettle, thistles and other herbaceous spp other herbaceous spp include*: Cirsium arvense, Digitalis purpurea, Epilobium* spp. (excluding *E. palustre), Chamerion angustifolium, Juncus effusus, J. squarrosus, Ranunculus* spp.*, Senecio* spp.*, Rumex obtusifolius, Urtica dioica*, 'coarse grasses'. < 15% trees & scrub Tree and scrub spp include: *Betula* spp.*, Prunus spinosa, Pinus* spp.*, Rubus* spp., *Sarothamnus scoparius, Quercus* spp., *Hippophae rhamnoides*. <10% bracken (dense canopy) Acrocarpous mosses less than occasional

Table 3.2 JNCC Scoring system for Wet heath habitats (JNCC, 2004)

Category	Attribute	Scoring
Habitat Extent		
	Bare Ground	1-10% Cover
Vegetation Structure	% Cover of dwarf shrubs	Cover 25-90%
		Dwarf-shrubs include: *Calluna vulgaris, Erica ciliaris, E. cinerea, E. tetralix, E. vagans, Ulex gallii, U. minor, Vaccinium* spp.
	Growth phase composition of ericaceous cover	All stages of growth present, no one stage dominant
Vegetation Composition	Dwarf Shrub Species	At least two species present
		Frequent abundance
	Graminoids	At least one species frequent
		2 species occasional Graminoids include: *Carex panicea, Carex pulicaris, Eleocharis* spp., *Eriophorum angustifolium, Juncus acutiflorus, Juncus articulatus, Molinia caerulea, Rhynchospora alba, Schoenus nigricans, Trichophorum cespitosum*.
		Molinia no more than occasional
	Desirable forb species	At least 2 species occasional Desirable forbs include: *Anagallis tenella, Drosera* spp., *Galium saxatile, Genista anglica, Myrica gale, Narthecium ossifragum, Pinguicula* spp., *Polygala serpyllifolia,* *Potentilla erecta, Serratula tinctoria, Succisa pratensis.* *serpyllifolia, Potentilla erecta, Rumex acetosella, Scilla verna, Serratula tinctoria, Thymus praecox, Viola riviniana,*
	Bryophytes and Lichens	>10% cover of Sphagna (if naturally present)
		>5% cover of lichens (if naturally present)
Negative Indicators	Signs of disturbance	No artificial functioning drains
		<1% of habitat showing signs of trampling/paths
		No silt or leachate
	Negative indicator species	<1% exotic species exotics include: *Rhododendron ponticum, Gaultheria shallon, Fallopia japonica*.
		< 1 % ragwort, nettle, thistles and other herbaceous spp other herbaceous species include: *Apium nodiflorum, Cirsium arvense, Digitalis purpurea, Epilobium* spp. (excl. *E. palustre), Glyceria fluitans, Juncus effusus, J. squarrosus, Oenanthe crocata, Phragmites* spp., *Ranunculus repens, Fallopia japonica, Senecio jacobaea, Rumex obtusifolius, Typha* spp., *Urtica* spp.
		< 15% trees & scrub Tree and scrub species include: *Alnus glutinosa, Betula* spp., *Pinus* spp., *Prunus spinosa, Quercus* spp., *Rubus* spp., *Salix* spp..
		<5% bracken
		<10% *Ulex europaeus*
		Acrocarpous mosses <occasional

3.3 Results

3.3.1 Do habitat condition scores show measurable differences following management activity?

Overall the total condition scores, shown in Figure 3.2, varied little between plots of different age and management type. No statistical differences were detected for age (df = 4, $F = 1.58$, $p = 0.22$), management type (df = 1, $F = 0.09$, $p = 0.77$) or interactive effects (df = 4, $F = 1.34$, $p = 0.29$). Condition scores were also generally high, with a mean score of 12.7 ± 0.22 standard error, but consistently did not meet the maximum favourable score of 16. One of the most frequent causes of failure was a low cover of pioneer heath across all sites, followed by building or mature heath cover. The attributes of tree and shrub cover, and the presence of dead heath and degenerate heath failed only infrequently (Figure 3.3), the relatively low cover of these latter characteristics may be responsible for the failure to distinguish between sites.

Several differences were detected for attribute scores between different management types with ANOVA analysis, illustrated in Figure 3.4. Those that differed by management type included dwarf shrub cover (df = 1, $F = 16.22$, $p = 0.001$), dwarf shrub species composition (df = 1, $F = 21.63$, $p = 0.001$), graminoids (df = 1, $F = 6.37$, $p = 0.02$), desirable forbs (df = 1, $F = 26.72$, $p = 0.001$), and tree and shrub cover (df = 1, $F = 10.05$, $p = 0.005$); these showed no statistically significant differences over the age of the plots or for interactive effects (Appendix III). The dwarf shrub cover and species attributes were much higher for burnt plots (0.79 ± 0.06 and 0.91 ± 0.03; mean ± standard error) than for cut plots (0.39 ± 0.08 and 0.59 ± 0.06), showing that burnt plots better matched ideal condition for these characteristics. The effect size of $\eta^2 = 0.38$ indicates a large effect of this factor. In contrast, graminoid scores were higher in cut plots (0.91 ± 0.03) compared to burnt plots (0.69 ± 0.07, $\eta^2 = 0.22$), although the scores were more closely matched in this example. Desirable forbs showed a huge difference, with a score of 0.06 ± 0.03 for burnt plots against 0.5 ± 0.09 for cut plots ($\eta^2 = 0.45$). Both management types have poor results here, but notably burnt plots almost never meet the requirement for favourable condition. In the case of the tree and shrub cover attribute, burnt plots had higher scores (0.99 ± 0.01) than cut plots (0.80 ± 0.05, $\eta^2 = 0.29$) but both scores were high. It is likely that in practice, without quadrats being used to score the results, both management types would be recorded as being in overall favourable condition as a result of larger areas being considered.

Relatively few attributes showed age-related changes, and these are also shown in Figure 3.4. The building heath attribute differed over the age of the plots (df = 4, *F* = 3.80, *p* = 0.02, $\eta^2 = 0.36$), with Tukey's HSD post-hoc tests indicating that differences between sites of 10 years (mean score ± standard error of 0.66 ± 0.16) and 0 years (0.13 ± 0.07) were responsible (*p* = 0.03). Other age-groups did not show statistically distinguishable differences, but plots of 6 years and 0 years were close to significance (*p* = 0.07). Non-significant test results are shown in Appendix III.

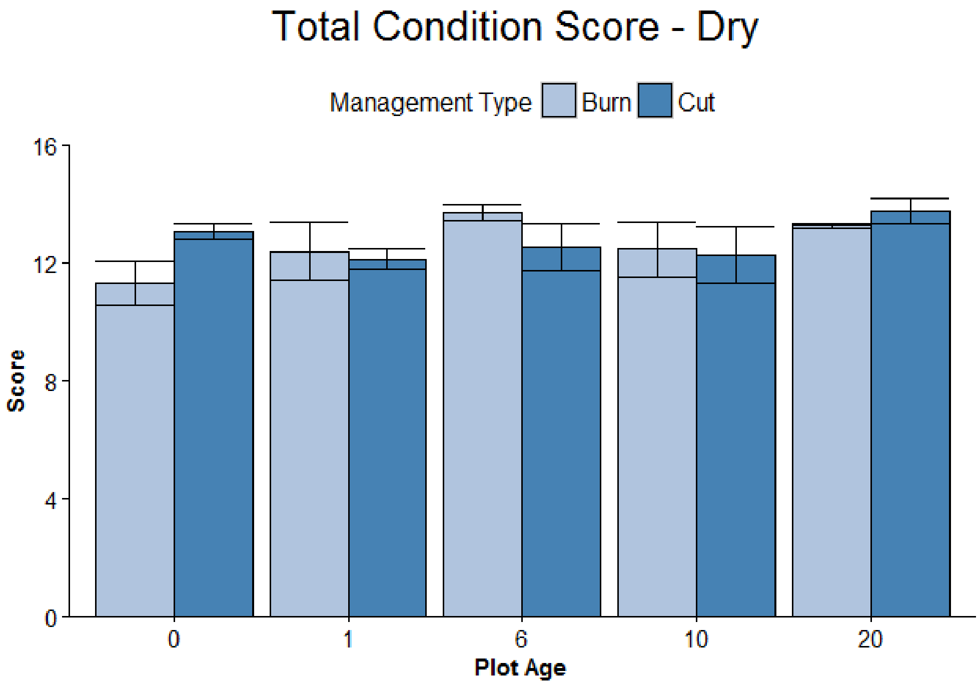

Figure 3.2 Total condition scores for the study plots, based on dry heath condition requirements. No significant differences were detected with ANOVA analysis (*p* <0.05). Error bars indicate standard error.

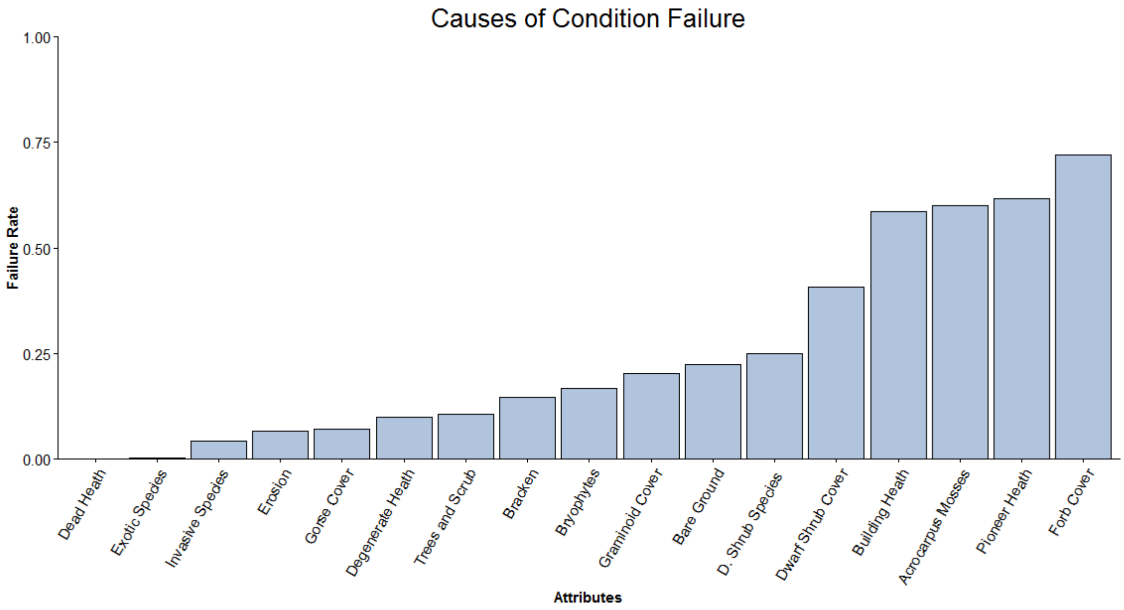

Figure 3.3 The proportion of samples not meeting condition scores by attribute. Failure rate refers to the proportion of recorded quadrats that failed the CSM assessment.

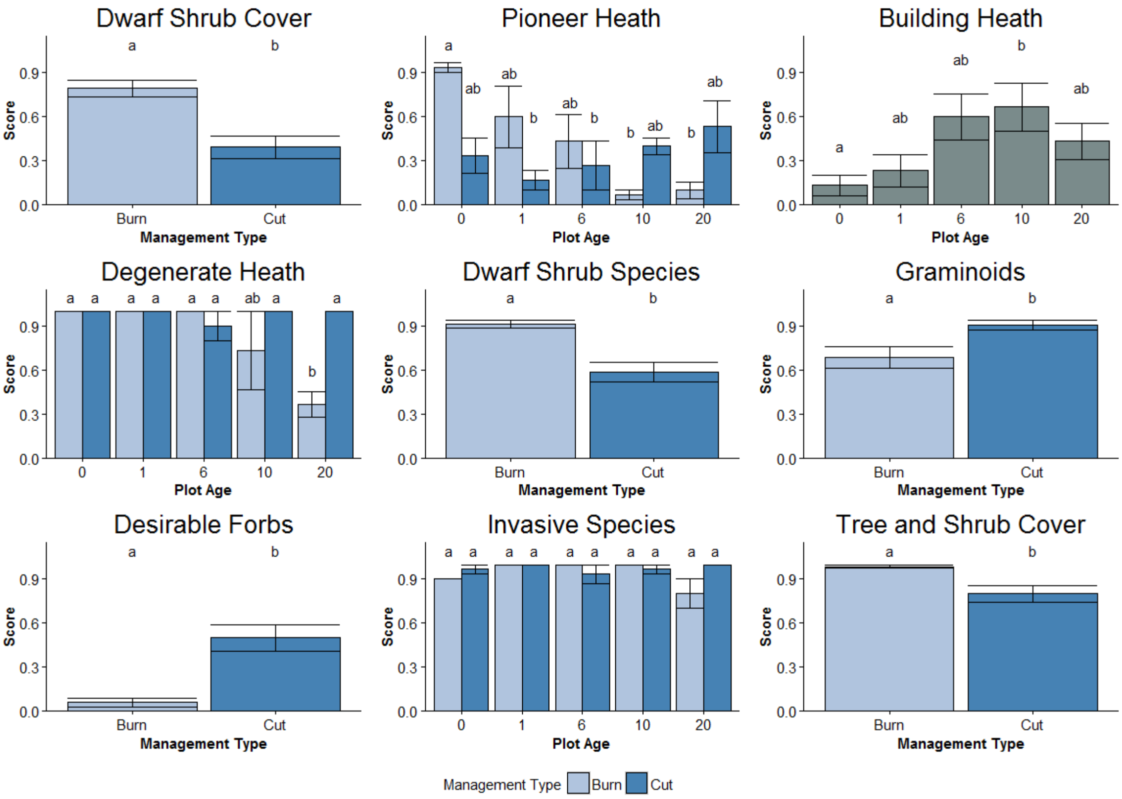

Figure 3.4 Attributes that showed differences between different plots with ANOVA tests (based on dry heath condition scores). Means grouped by the same letter are not significantly different (p < 0.05, Tukey test and p <0.05 ANOVA where management type is assessed). Error bars indicate standard error.

The degenerate heath (df = 4, F = 4.981, p = 0.006, η^2 = 0.31) and invasive species attributes (df = 4, F = 3.33, p = 0.03, η^2 = 0.31) showed interactive effects between age and management. Significant differences in invasive species were not detected with post-hoc Tukey's HSD, indicating that changes are relatively minor, probably as a result of considerable noise in the data and infrequent detection of these species, despite a large effect size. For degenerate heath, post-hoc tests pointed to differences between the oldest (20 year) burnt plots and almost all others (p = 0.004 in these cases), with the exception of 10 year old burnt plots (p = 0.22). In fact, the majority of the plots, with the exceptions of 10 and 20 year burnt plots (0.73 ± 0.27 SE and 0.37 ± 0.09 respectively) and 6 year cut plots (0.9 ± 0.1), met perfect condition scores here.

3.3.2 Condition Compared to the Vegetation Community

Generally, some similar changes were picked up with the condition attributes compared to the Chapter 2. Some differences were also evident as there was a failure to detect some observed changes using condition attributes. Based on habitat data, the cover of bare ground showed significant differences between management types and age, including an interactive effect (d.f. = 4, F = 2.97, p = 0.044) but attribute scores showed no differences when examined in the same way (d.f. =4, F = 1.21, p = 0.34). Correlation tests using Kendall's tau show that scores were not linearly correlated with the cover (τ = 0.05, p = 0.70), probably because the score must meet both a minimum and maximum cover value. In other instances, attribute scores appeared at first to match cover scores relatively well, such as with ericaceous cover and species composition (Figure 3.5), but changes detected as the plots aged on the original cover data could not be detected with the attribute scores. The age structure of ericaceous plants (Figure 3.6) was a closer match, with detected changes matching with both methods. In the case of dead heath, changes in cover were not shown by the attribute, but this is likely because such changes were very small in percentage cover terms (i.e. 1-3%). Attributes examining graminoids and desirable forbs closely matched the total cover of these species groups (Figures 3.7, 3.9) but a great deal of information about the constituent species was lost through this method. This is demonstrated when examining correlations of individual species and graminoid scores (Figure 3.8). While *Danthonia decumbens* (τ = 0.39, p = 0.06), *Carex demissa* (τ = 0.43, p = 0.004), *Agrostis curtisii* (τ = 0.43, p = 0.003) and *Agrostis capillaris* (τ = 0.27, p = 0.06) were correlated with the attribute, *Molinia caerulea* (τ = -0.13, p = 0.36), *Nardus stricta* (τ = 0.24, p = 0.12), *Trichophorum cespitosum* (τ = - 0.06, p = 0.71), *Agrostis stolonifera* (τ = 0.10, p = 0.53) and *Carex panicea* (τ = 0.18, p = 0.24) were not. Of particular interest is that *Molinia caerulea* is of greater abundance on burnt plots compared to cut plots, which runs contrary to the graminoid attribute and is not detected by condition scoring. *Molinia* is also the most frequent and most abundant grass, highlighting its importance. Forb species were generally correlated with the forb attribute, with *Galium saxatile* (τ = 0.65, *p* = 0.001), *Lotus corniculatus* (τ = 0.34, *p* = 0.04), *Plantago lanceolata* (τ = 0.35, *p* = 0.03), *Potentilla erecta* (τ = 0.59, *p* = 0. 001), *Rumex acetosella* (τ = 0.54, *p* = 0. 001) and *Viola riviniana* (τ = 0.61, *p* = 0.001) all highly correlated. *Hypochaeris radicata* (τ = 0.29, *p* = 0.08) and *Polygala serpyllifolia* (τ = 0.26, *p* = 0.07), the latter a characteristic heathland species, were not correlated with the attribute. It can be clearly seen that a great deal of information about the relative abundance of these species is lost by using the attribute measure (Figure 3.10). Several forb species that are also part of the attribute (*Armeria maritima, Genista anglica, Plantago maritima, Scilla verna, Serratula tinctoria* and *Thymus praecox*) were not not recorded and are likely absent from the study area.

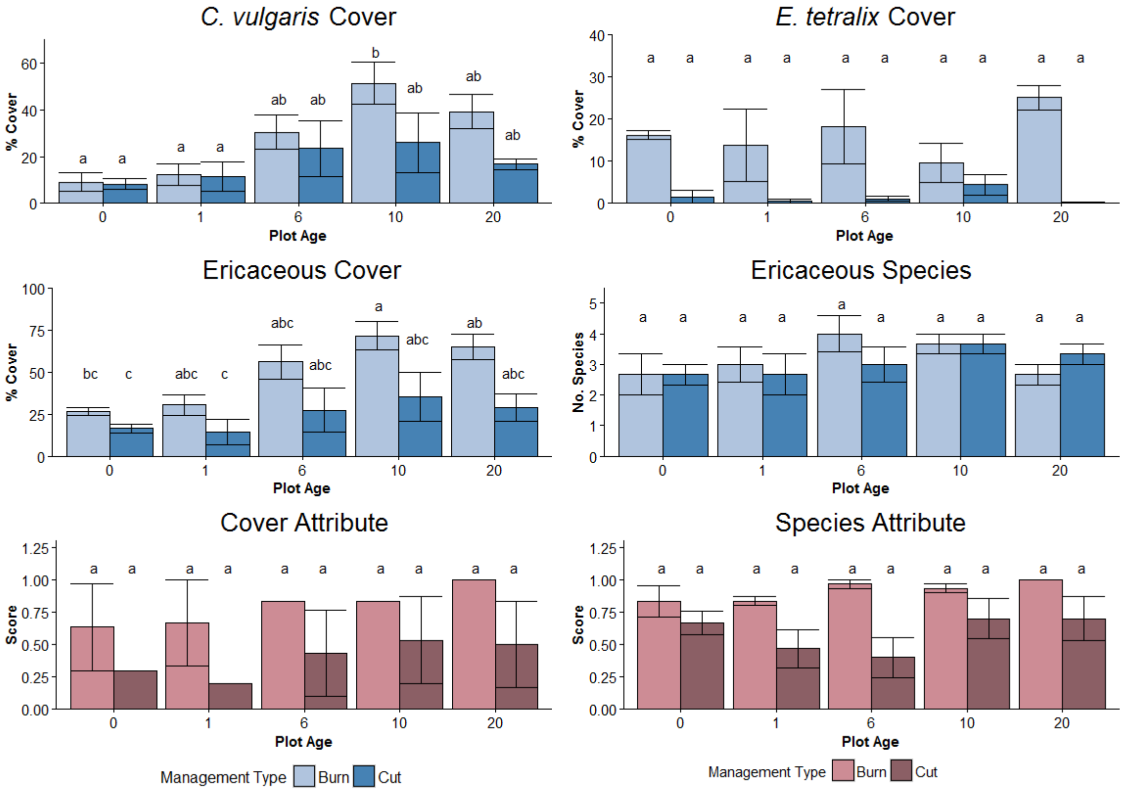

Figure 3.5. Results of ANOVA tests on ericaceous variables recorded by the vegetation survey compared to attributes recorded by CSM. Error bars indicate standard error, and means grouped by the same letter are not significantly different ($p < 0.05$, Tukey test). Attribute scores only differed by management, not age (see Fig 3.4).

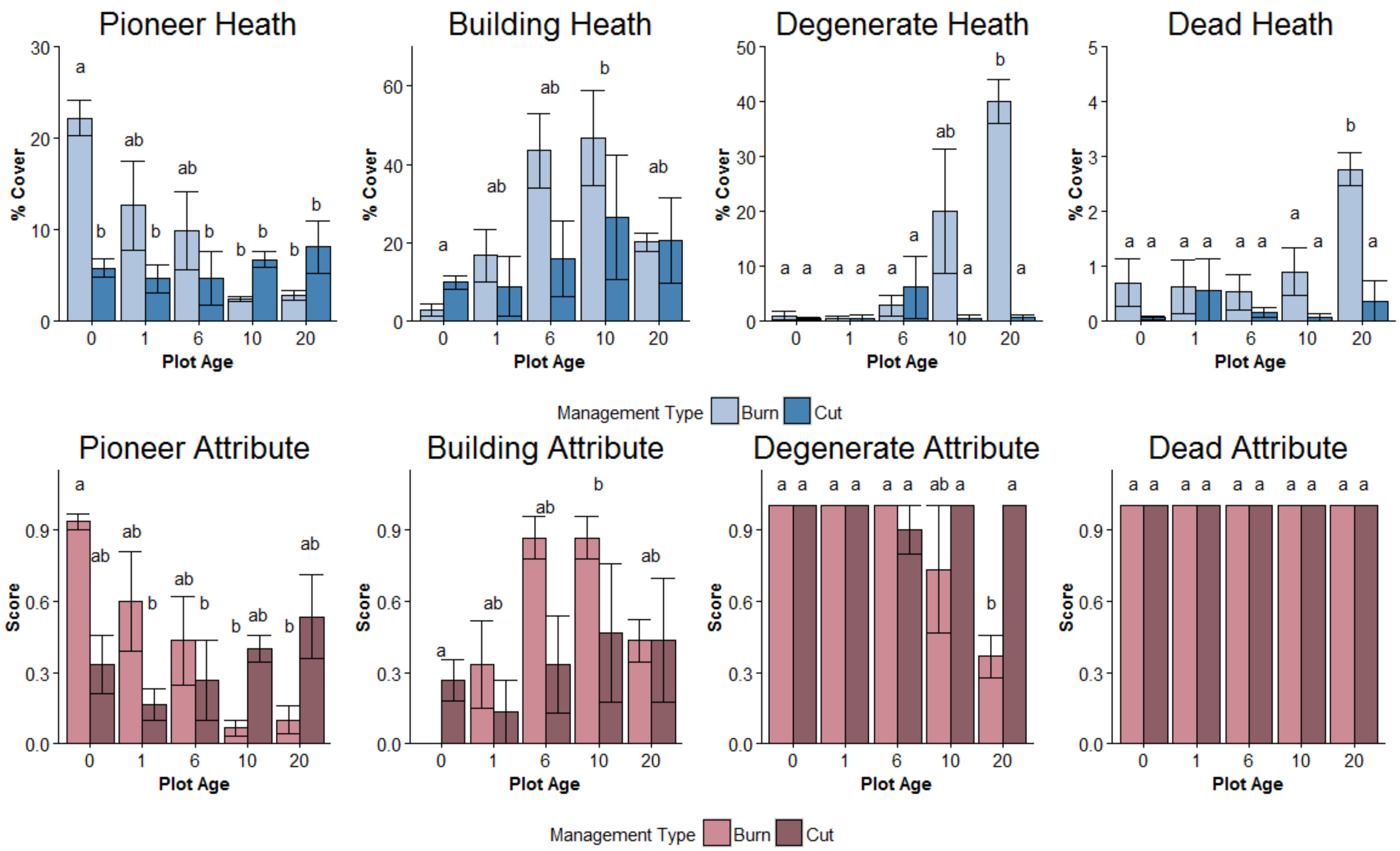

Figure 3.6 Age structure based on habitat data (above) and condition attributes (below) following ANOVA analysis. A close match can be observed, with degenerate heath a negative indicator at high cover and dead heath cover below cut-off values for the attribute. Error bars indicate standard error, and means grouped by the same letter are not significantly different ($p < 0.05$, Tukey test).

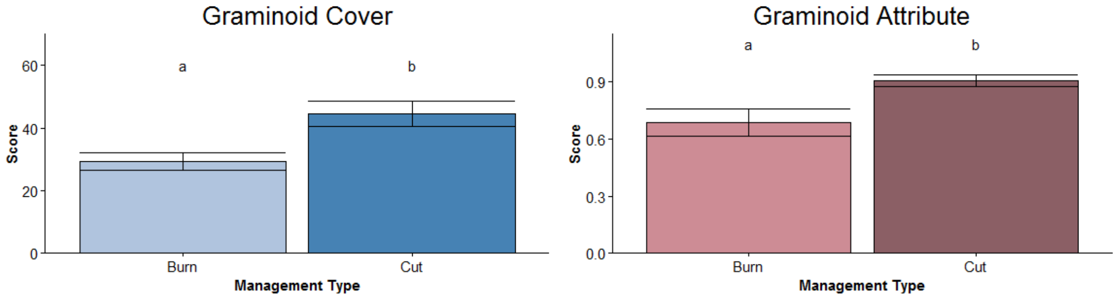

Figure 3.7 Significant differences ($p < 0.05$, ANOVA) in the cover of important graminoid species and the attribute score given by condition assessment. Error bars indicate standard error. The attribute score shows a similar difference to the raw cover values.

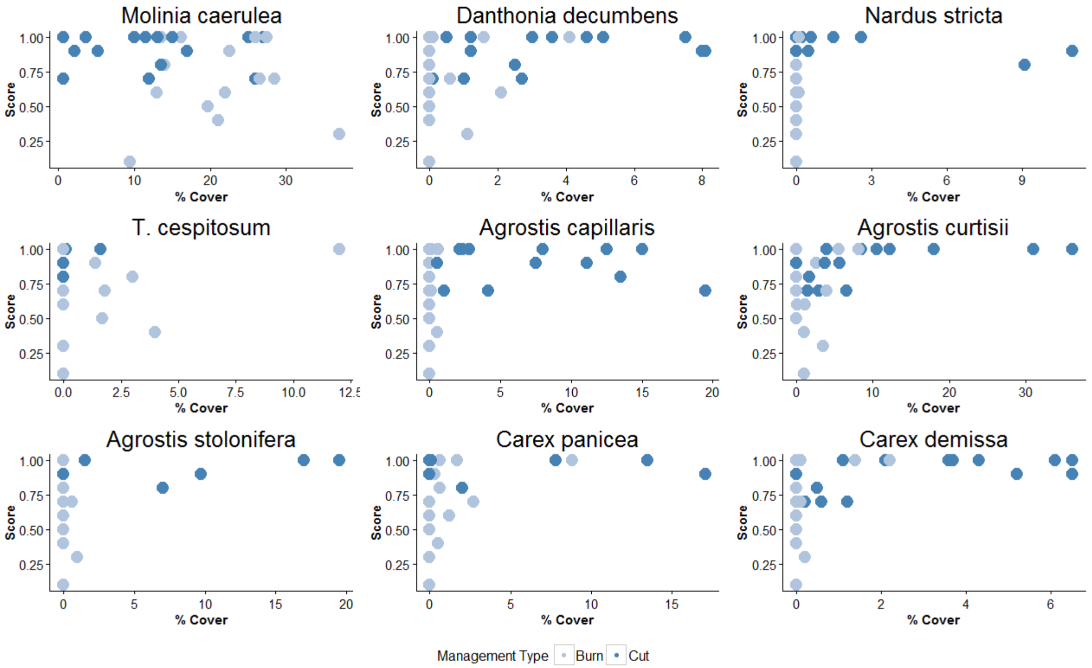

Figure 3.8 Scatterplots showing the attribute score for graminoid species plotted against the cover of individual species making up the attribute. Considerable variation in scores and cover values can be observed, with some species showing a high variation in cover with no apparent association in cover score. Darker circles indicate cut plots.

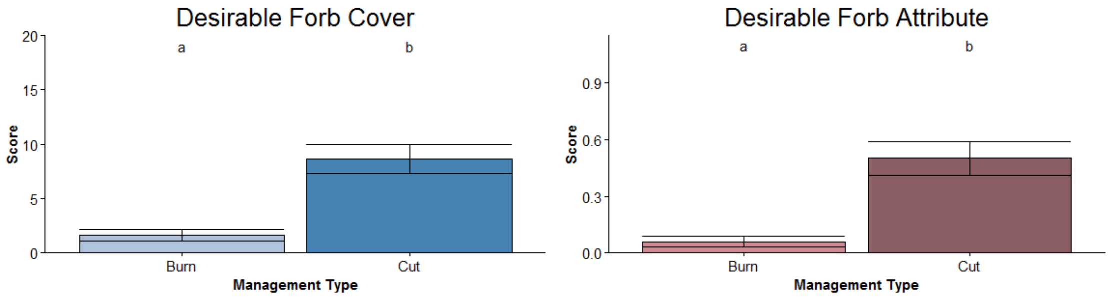

Figure 3.9 Significant differences ($p < 0.05$, ANOVA) in the cover of important herbaceous species and the attribute score given by condition assessment. Error bars indicate standard error. A close match can be observed between the two measurements.

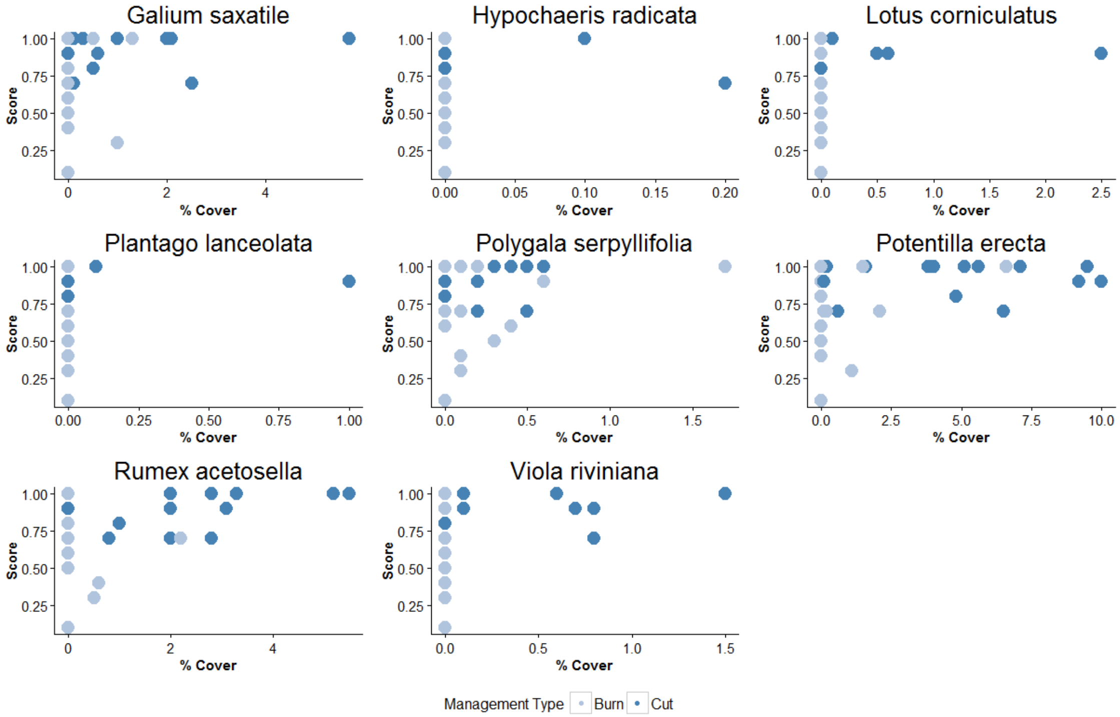

Figure 3.10 Scatterplots showing the attribute score for herbaceous species plotted against the cover of individual species making up the attribute. Considerable variation in the attribute score is present, and large variation in cover values can be observed with some species with little association with the attribute score. Cut plots are distinguished by the darker circles.

Habitat community data had also revealed significant differences in species richness and diversity between burnt and cut sites, which has no direct equivalent in condition assessment. These differences are not picked up by the use of condition scoring, the only connection being frequent failure of burnt sites to support a favourable forb community. But in fact, the cut site community is less representative of typical heathlands as, despite sharing many species with the driest heath communities, it is dominated by grasses and features a lower abundance of ericaceous plants (Chapter 2). Many of the species contributing to these richness and diversity values did not fall into any condition attributes, being neither positive or negative indicators. The overall condition scores in particular do not reflect the large difference in communities demonstrated by using more specific measures.

The overall condition score was also very poor at reflecting large vegetation and habitat gradients measured in the study. Plotted against these gradients (figure 3.11), condition score can be seen to vary very little over some large changes in habitat characteristics, such as vegetation height ranging from 20 to 600 mm and species richness from 5 to 49 vascular species. Of the examined gradients, none of grass or graminoid cover ($\tau = 0.07$, $p = 0.57$), average vegetation height ($\tau = -0.06$, $p = 0.65$), bryophyte cover ($\tau = 0.04$, $p = 0.76$), species richness ($\tau = -0.11$, $p = 0.41$) or Shannon-Wiener diversity scores (H' Index, $\tau = -0.06$, $p = 0.65$) showed any correlation with the condition score. Ericaceous cover, which was well described in its individual attribute scores, was positively correlated with overall condition score ($\tau = 0.30$, $p = 0.02$), but this influence was not enough to reflect changes in cover between management and age groups in the overall condition scores.

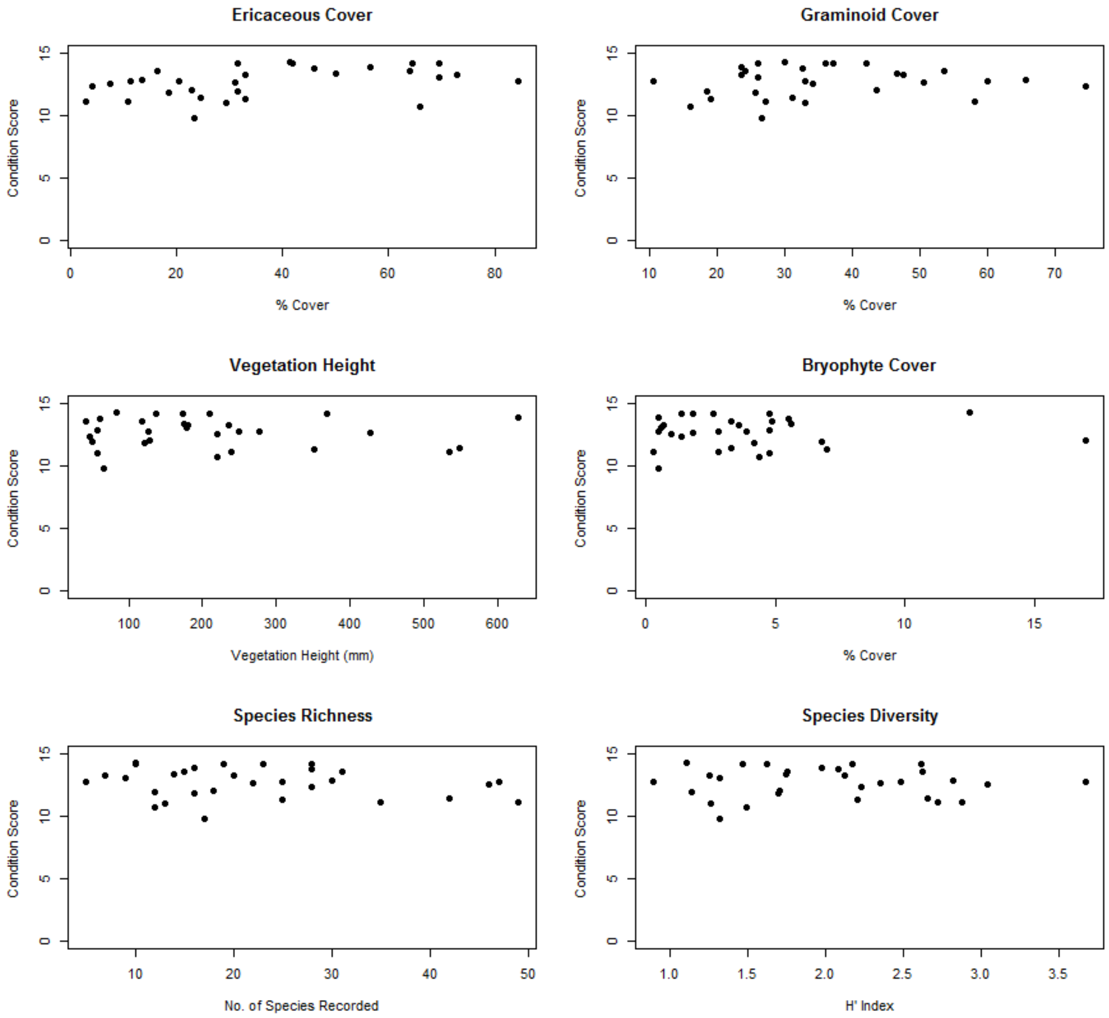

Figure 3.11 Scatterplots showing total condition score plotted against vegetation characteristics recorded throughout the study. The overall score shows poor association with these variables. Kendall's Tau showed no correlation between condition score and graminoid cover (τ = 0.07, *p* = 0.57), average vegetation height (τ = -0.06, *p* = 0.65), bryophyte cover (τ = 0.04, *p* = 0.76), species richness (τ = -0.11, *p* = 0.41) or Shannon-Wiener diversity scores (H' Index, τ = -0.06, *p* = 0.65). Ericaceous cover was weakly correlated with overall condition score (τ = 0.30, *p* = 0.02),

3.3.3 Condition Compared to Habitat Suitability

Habitat Suitability Indices, calculated for *C. austriaca,* and *S. undata*, showed no correlation with the total condition score ($\tau = -0.03$, $p = 0.82$ and $\tau = 0.14$, $p = 0.29$ respectively), showing that the final condition score does not reflect the suitability of habitat for these species. For *P. argus*, however, the HSI was positively correlated with condition score ($\tau = 0.31$, $p = 0.02$), suggesting that in this case the CSM score does partly reflect the habitat requirements of this species. Scatterplots for these correlations are shown in Figure 3.12. Some caution should be noted here, as the full range of HSI and condition scores were not recorded in the study, and there is still considerable scatter in the condition that does not appear to influence the *P. argus* score.

HSIs also showed differences between plots, shown in Figure 3.13. The Index for *C. austriaca* differed between management types (d.f. = 1, $F = 7.33$, $p = 0.014$), and was higher in burnt plots (0.52 ± 0.02; mean ± standard error) than in cut plots (0.41 ± 0.04). Effect size was relatively small at $\eta^2 = 0.17$. No differences were detected for these indices over the age of the plots (d.f. = 4, $F = 1.27$, $p = 0.317$) or for the interactive effect between age and management (d.f. = 4, $F = 2.54$, $p = 0.072$), although the latter was close to significance. A similar pattern was repeated for the *P. argus* HSI, with higher values in burnt plots (0.62 ± 0.02) compared to cut plots (0.45 ± 0.04; d.f. = 1, $F = 17.1$, $p = 0.001$), with a large effect size ($\eta^2 = 0.36$). As with *C. austriaca,* there were no differences over age (d.f. = 4, $F = 1.06$, $p = 0.403$) or interaction (d.f. = 4, $F = 1.60$, $p = 0.214$). The HSI for *S. undata* did not show any significant differences between management type (d.f. = 1, $F = 1.08$, $p = 0.312$), interactive effects (d.f. = 4, $F = 0.85$, $p = 0.513$), or age (d.f. = 4, $F = 2.54$, $p = 0.072$), although it was close to significance here. The mean index score for *S. undata* recorded across the study was 0.57 ± 0.03.

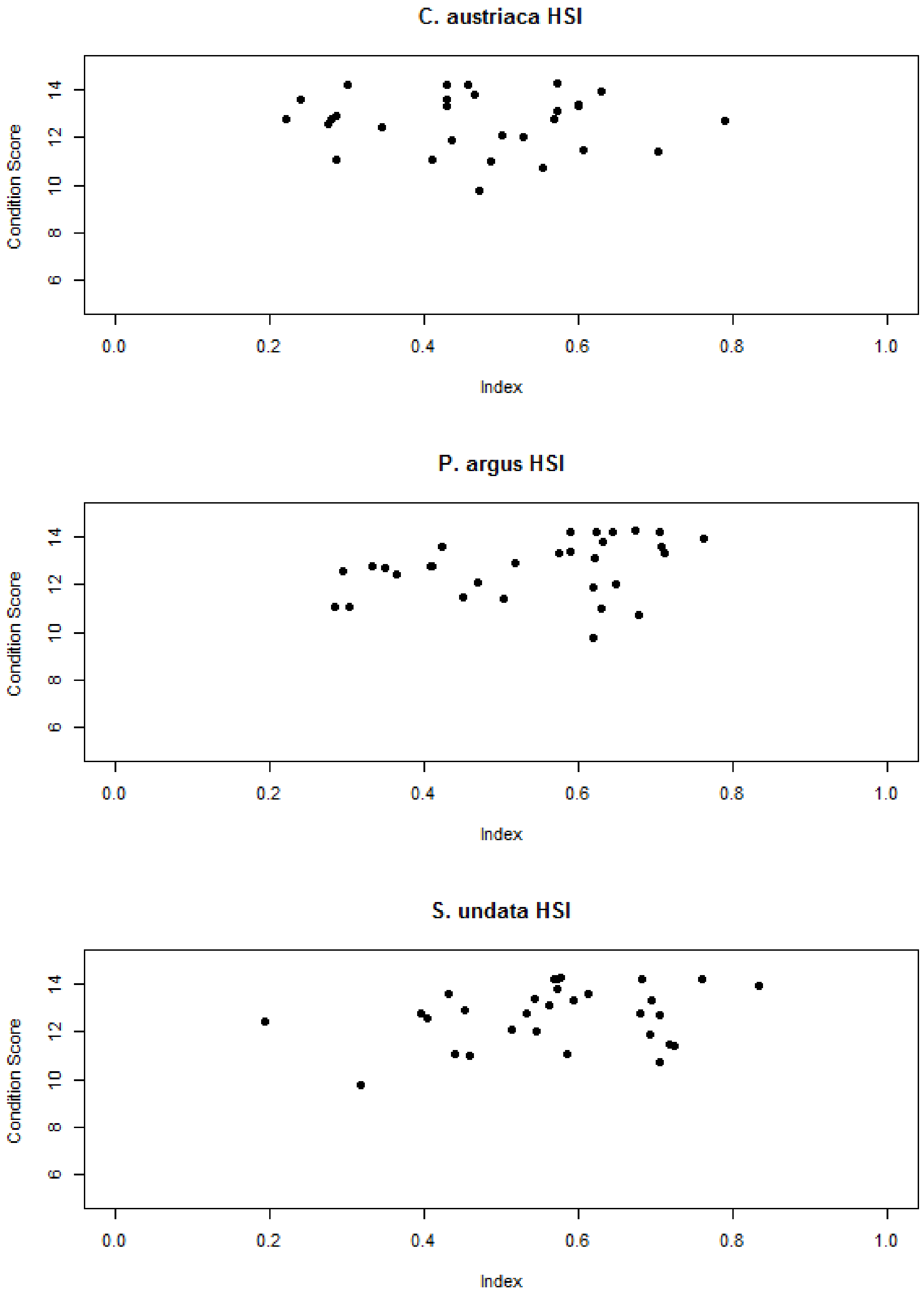

Figure 3.12 Scatterplots showing total condition score plotted against Habitat Suitability Indices for the three heathland species. Kendalls Tau analysis showed that Habitat Suitability Indices, calculated for *C. austriaca* (τ = -0.03, *p* = 0.82) and *S. undata* (τ = 0.14, *p* = 0.29), showed no correlation with the total condition score. The HSI for *P. argus* showed a weak correlation with condition score (τ = 0.31, *p* = 0.02).

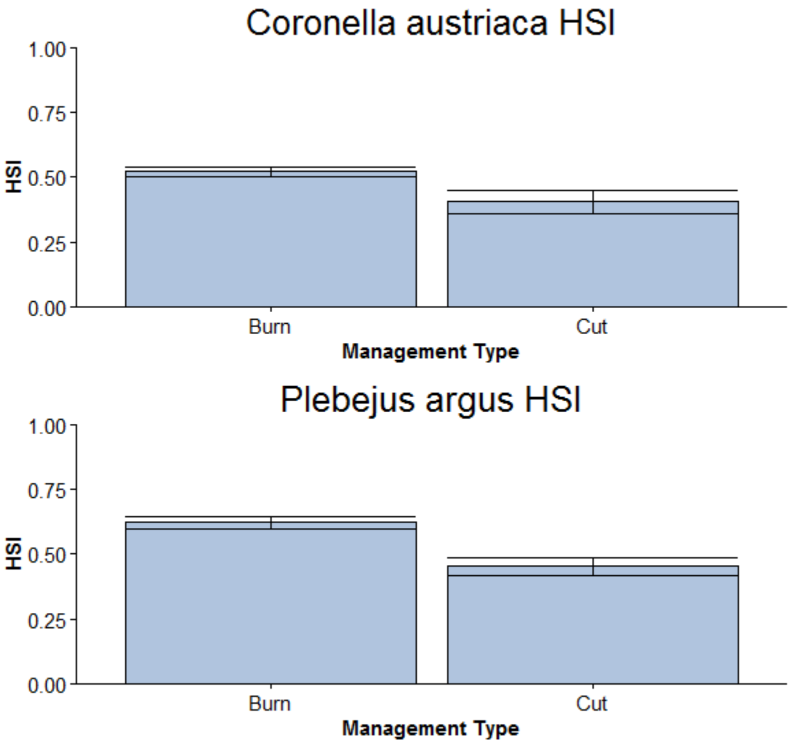

Figure 3.13 Graphs showing differences in HSI scores between plots. HSIs for *C. austriaca* and *P. argus* showed significant differences over management type, while *S. undata* did not show any significant differences and is not shown here. Error bars are standard error.

Unlike Condition Scores, HSIs showed stronger correlations with some of the vegetation gradients recorded in the study, which may suggest that they are better than condition at detecting or recording changes in the habitat. For *C. austriaca*, the suitability index was positively correlated with ericaceous cover (τ = 0.38, *p* = 0.003) and close to significance for vegetation height (τ = 0.24, *p* = 0.063). Graminoid cover was negatively correlated with the index value (τ = -0.37, *p* = 0.004), but there was no correlation with bryophyte cover (τ = 0.20, *p* = 0.133). Correlations for the *C. austriaca* index and habitat gradients are shown in Figure 3.14.

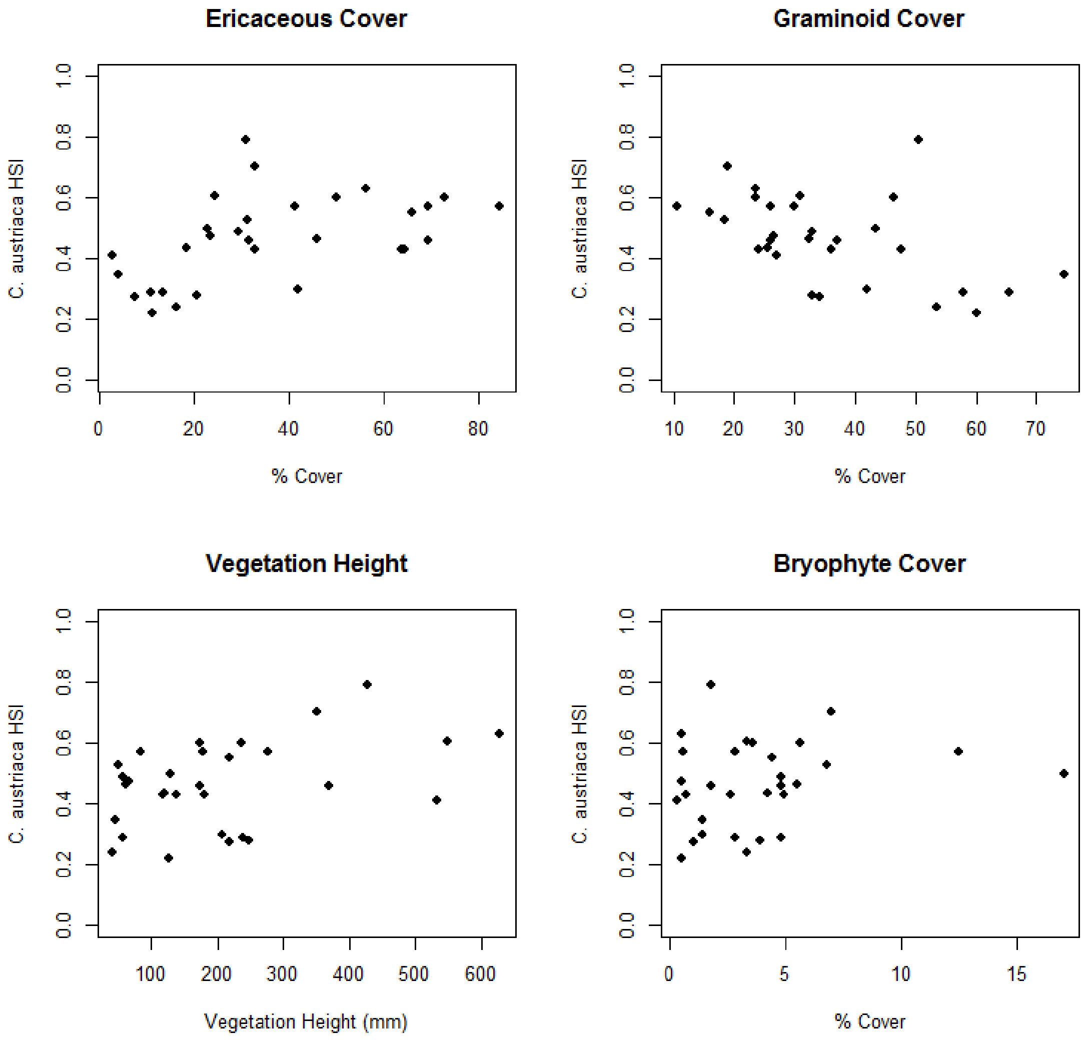

Figure 3.14 Scatterplots showing the HSI for Smooth Snake *Coronella austriaca* plotted against vegetation gradients recorded throughout the study. Ericaceous cover was positively associated with the HSI, graminoid cover negatively correlated, but associations with other variables were not detected (Kendall's Tau, $p < 0.05$).

P. argus Index values showed a very similar pattern with positive correlations with ericaceous cover ($\tau = 0.41$, $p = 0.002$) and negative correlations with graminoid cover ($\tau = -0.27$, $p = 0.040$), but none for either vegetation height ($\tau = -0.09$, $p = 0.454$) or bryophyte cover ($\tau = 0.19$, $p = 0.153$). *P. argus* index and gradient correlations are displayed in Figure 3.15. Finally, *S. undata* index values were positively correlated with ericaceous cover ($\tau = 0.35$, $p = 0.007$) and vegetation height ($\tau = 0.43$, $p = 0.001$), negatively correlated with graminoid cover ($\tau = -0.31$, $p = 0.015$), and not correlated with bryophyte cover ($\tau = -0.01$, $p = 0.957$), shown in Figure 3.16. It should be noted that these correlations are not necessarily indicative of the effect of particular vegetation groups or characteristics, but are specific to the plots measured. For example, in different environmental conditions it is possible that graminoid cover would be positively associated with the *C. austriaca* index; the negative association here could be caused by the cut plots, which feature high cover of graminoids at the expense of ericaceous cover.

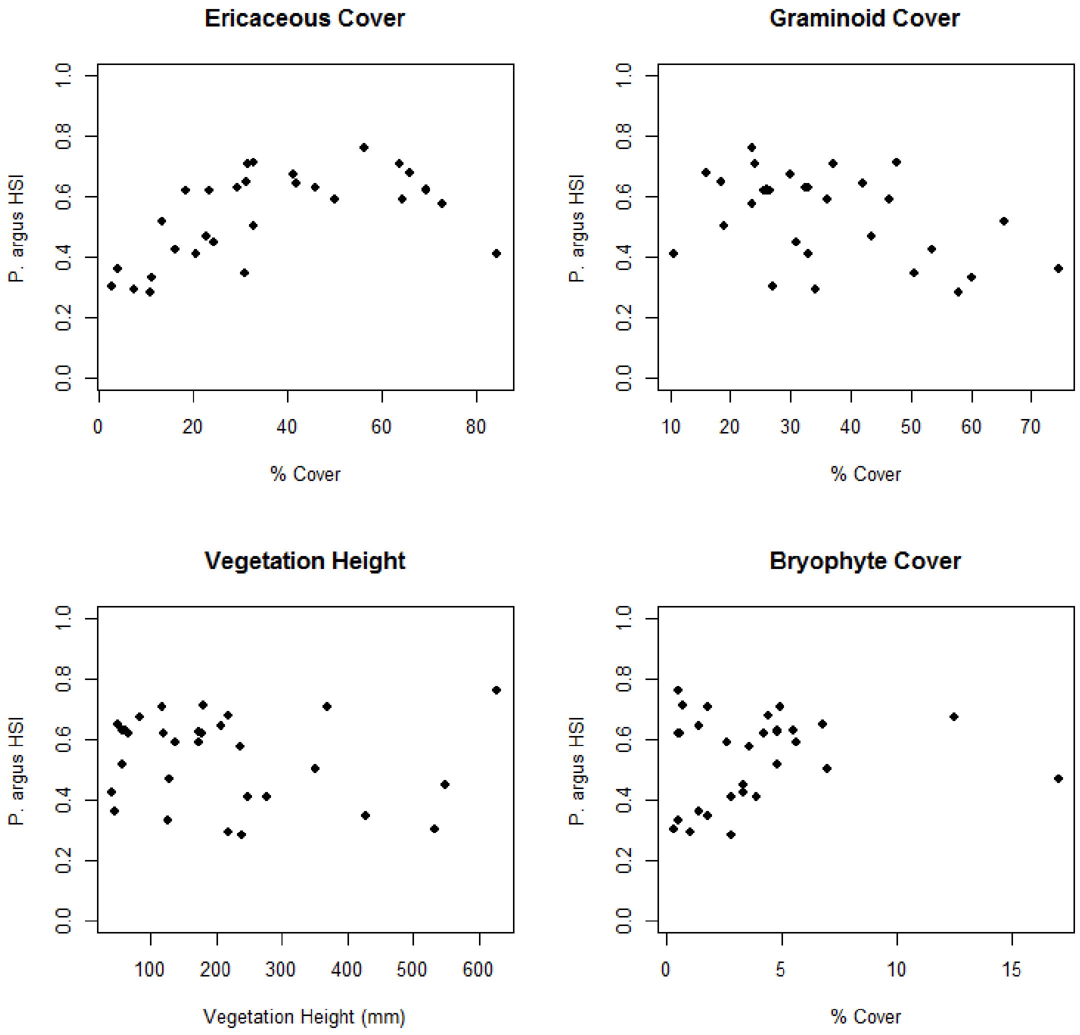

Figure 3.15. Scatterplots showing the HSI for Silver-Studded Blue Butterfly *Plebejus argus* plotted against vegetation gradients recorded throughout the study. The HSI score was positively correlated with ericaceous cover, negatively correlated with graminoid cover, but with associations with other variables (Kendall's Tau, $p < 0.05$).

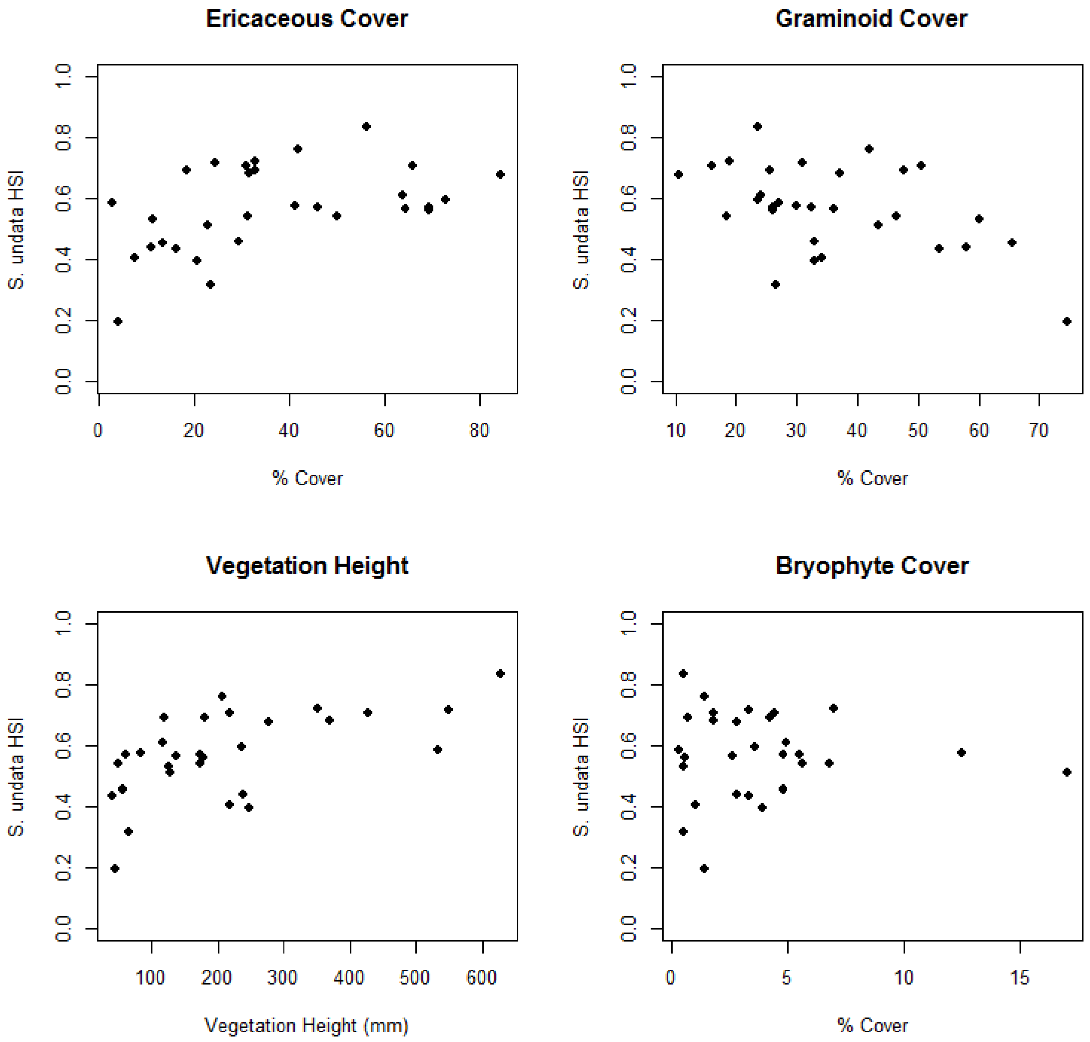

Figure 3.16 Scatterplots showing the HSI for Dartford Warbler *Sylvia undata* plotted against vegetation gradients recorded throughout the study. *S. undata* index values were positively correlated with ericaceous cover and vegetation height, negatively correlated with graminoid cover, and not correlated with bryophyte cover (Kendall's Tau, *p* >0.05).

3.4 Discussion

Critical assessment of Common Standards Monitoring (CSM) has been extremely limited, and little is known about its effectiveness at detecting change in heathland habitats. Jackson and Gaston (2008) recommended a full empirical validation of current CSM measures; here results from lowland heathland habitat are reported for the first time. In this study, overall CSM scores did not differ between different management types despite the considerable variation in vegetation composition and structure between these (demonstrated in Chapter 2). Individual attributes were much more successful than the combined score at reflecting changes observed in the habitat, but a great deal of information that could be useful for management is still lost with this technique. The CSM score itself was found to be of questionable ecological value; according to the evidence presented here, it does not correlate with some important environmental characteristics of the plots or with the suitability of habitat for species from a range of taxonomic groups. As feared by Gaston *et al.* (2006), condition categories are therefore ecologically dubious. The results here show clearly that CSM is not effective as a method to monitor the results of management activity or other important changes in New Forest heathlands, and likely elsewhere in lowland heathland habitat. Considerable information that would be useful for adaptive management approaches are never recorded or detected under a CSM approach, including small scale habitat changes and information about individual species, as has previously been suggested (Davies *et al.,* 2007).

3.4.1 Limitations

Although the methods used here to statistically assess CSM were constructed differently to how the approach is implemented in practice, having used more quantitative techniques such as random sampling, quadrat-based assessments and scores calculated for each quadrat, the implications of the results should not be affected. Assessing CSM in this way allows the effectiveness of the scoring system to be examined without the additional problem of having to interpret subjective scores or interpretations. Furthermore, the potential of subjectivity in CSM is acknowledged as one of its downsides (Williams, 2006; Gaston *et al.,* 2006). The study also examines a fairly limited number of sites at one geographical location; despite this, clear differences are seen when comparing condition scores to abundance data, and environmental characteristics showed wide ranges that were expected to influence scores. The use of Habitat Suitability Indices, as calculated here, could be considered controversial as these were essentially preliminary and would need to go through extensive testing before being deployed to support management (Brooks, 1997). But regardless of these limitations, it is immediately apparent that HSIs, emphasising more specific habitat attributes, showed different results to CSM, demonstrating that the broad approach used in CSM misses fine-scale habitat details that are the essential characteristics of many species habitat requirements.

3.4.2 Do habitat condition scores show measurable differences following management activity?

CSM failed to detect large changes in the vegetation community. This may be a result of the original intention of CSM to examine the state of SSSIs, making the requirements necessarily broad to cover a variety of different vegetation sub-types and geographical locations and serious degradation (Alonso *et al.,* 2003), rather than management aimed at maintaining the habitat. A consequence of this is that the assessment struggles to detect small responses to management that may not have an immediate impact on condition status but could cumulate over the long term. Unlike the overall condition scores, attribute scores showed more success in detecting changes when considered individually, which is not altogether surprising because they are targeted at important heathland features (JNCC, 2004; Alonso *et al.,* 2003). Nevertheless, if these attributes are overlooked, the overall score or classification will not be useful. Causes of condition failure observed here were surprisingly very different to those reported by Cantarello *et al.* (2010) in New Forest heathlands, suggesting that some of these attributes are indeed overlooked in practice. Failures recorded by Cantarello *et al.* were principally the excessive shrub and tree cover in addition to overgrazing by livestock, contrasting with low cover of pioneer, building heath and herbaceous plants in this study. While it could be argued that the failure of sites to meet the condition requirements for pioneer heath is a temporary result following intervention disturbance, the problem was prevalent across a range of sites. Condition attributes addressing a lack of management, such as excessive degenerate heath or increasing scrub, almost never failed, suggesting that the focus of management is in reducing these conditions. Potentially this could lead to negative consequences of management (i.e. too much disturbance) being overlooked when assessed subjectively, an especially negative circumstance when areas are determined to be recovering on the basis that management is being carried out. A focus of CSM on the effects of a lack of management can be traced back to its initial genesis, where essentially abandoned SSSIs had been lost (Alexander and Rowell, 1999).

3.4.3 How well do habitat condition scores and their constituent attributes reflect community and structural changes observed in the habitat community?

Attributes that had contributions from a number of species demonstrated a considerable loss of information when compared to their individual components, because the large number of contributing species masked individual information. This is not a surprising result, but a potentially important one as it means that the loss of some individual species could go unrecorded if monitoring strictly follows the CSM recording process. Overlooking individual species has previously been raised as a potential problem with CSM in this habitat (Ross & Bealey, 2005), and although in some cases the subjective approach may mean that land managers record individual species, this is not guaranteed. This is particularly apparent with the example of *Molinia caerulea*, an important heathland engineer (Marrs *et al.,* 2004), which was not associated with the attribute measuring grasses on the heath. While quantitatively measuring and analysing species abundance is perhaps more difficult, it is much more useful for monitoring owing to its increased precision, and will be more effective at recording rare species or species that are overlooked because of the attribute scoring. When assessing management, this usefulness is accentuated because cumulative small changes over time could become important. Some other attributes, particularly those relating to ericaceous structure and species, appeared to match abundance data closely, indicating that these performed well and could have some utility in assessing management. For attributes addressing ericaceous species, this is likely because such attributes are closely defined and measure specific habitat features rather than a combination of species.

In any case, despite the apparent utility of some attribute scores, overall condition scores were ecologically questionable. The very poor performance of the overall condition score in responding to large ecological gradients was surprising, but adds support to descriptions by Jackson and Gaston (2008) that condition assessment was insufficient to provide detailed information about habitat status, and fit with those of Cantarello & Newton (2008) who showed that visual assessment scores did not correlate with measurements using more quantitative techniques involving plots and point surveys in forested habitats. The reason scores appeared to be questionable is likely because of the large number of attributes, which mask any signal from individual attributes that may respond to ecological changes. Ross & Bealey (2005), in tests of CSM performance, claimed that vegetation height was associated with condition status, but their conclusions were not adequately statistically supported and run contrary to the findings here, where a large variation in vegetation height had no measurable change in score. One area where CSM scores were associated was with a gradient in ericaceous cover. Favourable plots were associated with higher ericaceous cover in validation tests (Ross & Bealey, 2005) of CSM performance, although in that study the categories of favourable and unfavourable were used rather than scores. However much more detail on the structural and community variation is necessary for effective monitoring, as small changes could occur in advance of habitat shifts, giving indications to managers while there is still time to react to them.

3.4.4 How do CSM scores compare to Habitat Suitability Indices when there are changes in the habitat?

HSIs appeared to be somewhat superior to overall condition score in detecting the differences between management types. For two species, HSI values were notably higher in burnt plots, but mean scores were still relatively low, suggesting that even these areas were not ideal habitat. Notably the HSIs were much more likely to be correlated with environmental gradients than condition score, showing that they respond to changes within the habitat. Measures of habitat structure were frequent contributors to the habitat suitability indices for most species, which is one area that CSM largely overlooks. In particular, associations with vegetation height and grazing pressure were noticeable, but CSM has little room for recording these features. One of the criticisms of habitat monitoring is its lack of applicability for fauna (Verberk *et al.,* 2010) and the lack of habitat structure measures, other than for heathland age structure, supports these assertions. This is despite the importance of these features being widely acknowledged, even in advice for condition assessment itself (Alonso *et al.,* 2003). A notable difference between CSM and HSIs is that HSI calculations involve fewer attributes or component scores, which likely aids the association with ecological changes in the plots.

Of the measured species, *Coronella austriaca* habitat is likely to be negatively affected by the burning rotations here, as little degenerate heath was available for them to exploit. Variable aging rates of heath (Alonso *et al.,* 2003) are supposed to be taken into account by management, but there is little evidence of the presence of older heath in this study. Extensive ground communities, including litter, lichen and moss, were also relatively low. Intensive grazing also likely affects the suitability of habitat in the New Forest for this species and could contribute to such structural problems. Repeated management also seems to reduce the amount of available scrub and gorse for *Sylvia undata*, negatively affecting its habitat indices in

New Forest heathlands. The HSI for *S. undata* may not have detected changes because of the spatial scale at which it was applied; at larger scales this may be a more effective measurement. Extensive gorse areas are present in the New Forest but were not part of this study; such areas would likely score highly. Total CSM scores correlated with the HSI for *Plebejus argus*, and this probably a result of the large component of ericaceous cover. However, the correlation seems to reveal relatively little, as CSM scores did not differ over management types while *P. argus* indices showed considerable differences. Previous assessments of this species that found higher abundance in unfavourable areas compared to favourable areas recorded by CSM (Davies *et al.,* 2007). Despite the association in scores, CSM cannot be considered to be able to indicate ideal habitat for this species. Concerns over the applicability of CSM for invertebrates, particularly butterflies (Davies *et al*., 2007) have not yet been addressed.

3.4.5 Implementation of Management and Recommendations

In addition to the problems identified here, implementations of condition assessment may be carried out in a variety of ways that may influence scores (Jackson & Gaston, 2008; Davies *et al.,* 2007). These effects have not been assessed here, but would further reduce CSM's effectiveness in monitoring management in practice because of the potential for subjective assumptions, and are inappropriate for evidence-based management. There are suggestions that attribute scores may be useful in reporting the status of habitat, and these could perhaps form the basis for more detailed reporting of habitat changes following management. However, because of the loss of precision these methods should be treated cautiously when examining management methods. Additionally, they require management goals to be much more specific than "favourable status", referring to thresholds among the attribute scores themselves. There seems to be little association of attribute scores with the management practice itself, because the overall condition classification into status is the main method by which habitat status is reported (JNCC, 2006; Natural England, 2012). Because desired outcomes of management cannot always be accurately predicted (Jackson & Gaston, 2008), reporting status as "recovering" because of management activity reports where action is taking place rather than if it is effective or not. It is precisely this circular argument that means that CSM measures should not be used to monitor management effectiveness. To properly conduct evidence-based and adaptive management, a different approach to monitoring will be necessary. This will require monitoring that quantitatively assesses key variables, including habitat structure, community structure, and where appropriate, abiotic conditions. Importantly, the monitoring must necessarily be tied to the locations and management taking place, because as shown above a broad approach may miss important changes that may occur. Based on the initial findings here, Habitat Suitability Indices could aid conservation methods, but would probably not be useful for directly measuring management unless improving the provision of habitat for that species was specifically the management goal.

3.4.6 Conclusion

This study found that condition assessment was poor at responding to environmental gradients recorded in the study area and did not demonstrate several differences between management approaches that were detected in vegetation community data. It also showed poor association with the habitat requirements of several important heathland species. With the exception of some attribute scores, CSM did not reflect community and structural changes in the habitat following management, and as a result of this cannot be considered useful as a tool for monitoring management. In the New Forest it is clear that it is an insufficient approach to monitoring management interventions, and greater and more directed monitoring effort is required if a move to evidence based management is required. This study marks one of only very few critical assessments of CSM in the UK, and suggests that the system may be deeply flawed if it is used as an exclusive monitoring programme in lowland heathlands. Results from other habitats await investigation.

3.5 References

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Chapter 4: The Recovery of vegetation following restoration work in New Forest wetlands.

4.1 Introduction

Freshwater wetlands are some of the most important ecosystems globally, providing both biodiversity and important ecosystem services through their functional processes (Gopal & Junk, 2000). They play a large role in cycling and storing carbon and other nutrients, support a diverse and endemic biology, and regulate water cycling and flow throughout the wider landscape (Moreno *et al*., 2012; Millenium Ecosystem Assessment, 2005). Globally, wetlands are increasingly under threat from a variety of sources, including river modification, pollution, exotic species introduction and particularly land-use change - with drainage, afforestation and river diversion being major contributors to decline (Gopal & Junk, 2001). Europe has suffered major losses in the extent of wetlands, particularly of marsh and mire habitats (Morre, 2002; Joosten, 1997). These declines have been greatest in North-West Europe, and in the UK up to 90% of mire habitat has been destroyed (Burns *et al.*, 2013). Wetlands such as mires and wet grassland are also low-nutrient habitats, so changes in nutrient inputs (i.e. through nitrogen deposition) also have damaging effects on ecosystem function and biodiversity (Bragazza et al. 2003; Bobbink & Roelofs 1995; Lamers et al. 2000). Despite these decines, study of wetland biodiversity has become the focus of widespread investigation relatively recently (Gopal & Junk, 2001), and wetlands are poorly understood with large gaps in knowledge of biodiversity, microbial communities and the relationships between biodiversity and ecosystem function.

As a result of these declines and in parallel with increasing awareness of their importance, wetlands have become a focus of conservation efforts, both through protection and restoration (Pfadenhauer & Grootjans, 1999). Awareness of the importance of wetlands has led to their protection under The Convention on Wetlands of International Importance (Ramsar, Iran, 1971), EU Birds Directive (79/409/EEC) and Habitats Directive (92/43/EEC). In Europe and North America, a number of restoration projects have been undertaken (Galatowitsch & van der Valk, 1996; Wheeler *et al.*, 1995), often with the ambition of restoring lost ecosystem services. However, many of these projects have focused on specific wetland functions, particularly flood control, and few addressed biodiversity issues or the combination of restoring biodiversity and wetland function (Gopal & Junk, 2001; Zedler, 2000). Restorations are often aimed at raising the water table (Grand-Clement *et al.*, 2013), removing successional or invasive species, or large-scale habitat recreation. Restorations of mires have recently become widespread as their value as carbon and methane sinks is recognised, leading to a number of studies (Grand-Clement *et al.*, 2013; Holden *et al.*, 2011; Haapalehto *et al.*, 2010; Worral *et al.*, 2007) though there are still relatively few that focus on biodiversity changes (Ramchunder *et al.* 2009), and few studies recording long-term change (Haapalehto *et al.*, 2010). Similar practices have also been undertaken for wet grassland (Schrautzer *et al.*, 2013; Kolos & Banaszuk, 2013; Acreman *et al.*, 2010).

Despite these studies, the practice of wetland restoration is still in its infancy, with many uncertainties remaining. Highly variable conditions make general proscriptions difficult, and in mire systems long-term changes have often been uncertain when compared to short term changes (Wheeler *et al.*, 2002). These uncertainties and limitations represent the main knowledge gaps within wetland restoration ecology. Often restoration projects have not been scientifically studied, with monitoring based on an intuitive or subjective habitat condition approach (Wheeler *et al.*, 2002). Additionally, restoration of abiotic conditions and ecosystem function has rarely been measured (Ruiz-Jaen & Aide, 2005; Wheeler *et al.*, 2002). Those studies that have assessed ecosystem functions have tended to detect a lack of recovery even when biological communities appear to be restored (Moreno *et al.*, 2012). Several mire studies (e.g. Haapalehto *et al.* 2010; Vasander *et al.*, 2003) found that species composition on restored sites can remain in a degraded state for long periods of time, making it very important to study at least some functional changes alongside standard approaches (Ruiz-Jaen & Aide, 2005; Morgan & Short, 2002). Measuring abiotic changes also allow the mechanisms behind the restoration to be understood (Pfeifer-Meister *et al.*, 2012) as restored areas may differ in function to those of 'natural' wetlands (Moreno *et al.,* 2012; Rey Benayas *et al.*, 2009; Aronson & Galatowitsh, 2008).

The New Forest contains a network of wetlands designated under the Ramsar Convention and the EU Habitat and Birds Directives (Cantarello *et al.*, 2010). These are considered of great importance to the elements of Forest's biodiversity, including breeding wading birds (Goater *et al.*, 2004) and rare, specialised flora (Atkinson, 1984). In the past, mires, which are often located within small valleys, have been subject to a variety of drainage measures, which were initially aimed at improving grazing provision for Commoners livestock and forestry (Tubbs, 2001). As a result of these changes, the biodiversity and function of the mires have been degraded (Clarke, 1984). Since 1997, these areas have been the focus of a number of restoration efforts aimed at restoring past watershed function and reducing drainage from mire habitat. Natural processes are not expected to recover without clear intervention (Van Seters & Price 2001 in Holden 2004) and past drainage may have caused irreversible changes in the physical properties of peat substrates in the Forest, as found by Holden *et. al.* (2004) in other locations. Such damage may reduce the likelihood of successful restoration, but also highlight the need for a rapid repair of the mire watershed in order to prevent further damage. The main restoration programmes in the Forest are (in chronological order), the Life II and Life III projects, the Rural Pathfinder Scheme and the current Higher Level Stewardship (HLS) Scheme (Cooch & Morris, 2001; New Forest Life Partnership, 2001; 2006; Smith, 2006; Natural England, 2010). Restoration has focused on a variety of habitat types, including mire and valley bogs, wet grassland, bog and riverine woodland and river channels, where past modifications had promoted drainage. Much of the restoration work has focused on removing sources of drainage through blocking or modifying channels, and removing scrub or trees where past drainage has allowed succession to non-wetland habitats (Holzer & Elliot, 2010). The management goal of this restoration work is to restore habitats to “favourable condition”, based on the JNCC’s guidelines for individual habitats present in mire areas (JNCC, 2004). Monitoring of restoration work has been limited, and restricted to condition assessment of SSSI units and photographic documentation, except in the case of river channel restorations where some hydrological studies have been undertaken (e.g. Millington *et al.*, 2007). Existing monitoring schemes, based on the vegetation cover of the sites, also do not give any indication of abiotic changes in the ecosystem that may have long-term implications of the success or failure of the restorations.

The New Forest presents a unique opportunity for research into wetland restorations owing to the large number of essentially separate sites that have undergone restoration, over a variety of time periods. This allows the response following management to be investigated with a chronosequence approach. This is in contrast to many existing studies of wetland, particularly fen and mire restorations (e.g. Large *et al.*, 2007; Vasander *et al.*, 2003; Klötzli & Grootjans, 2001), which have focused on intensive studies of individual sites. These measurements will also allow a comparison of the mires in the New Forest with those elsewhere in Europe, which may aid future management and conservation. This has not previously been undertaken, and is particularly important when comparing monitoring results to other wetland systems (Haapalehto *et al.*, 2010).

This research aims to investigate the impact that restoration work has had on both the biotic and abiotic components of a wetland ecosystem (valley mires) in the New Forest. The objectives of this research are to answer the following questions:

i. How has the plant community changed following restoration work?
ii. How have abiotic and structural conditions changed as a result of restoration?
iii. To what extent do the restoration interventions account for the variation in the plant communities observed?
iv. What are the pathways where restoration success could be improved, and what are the potential barriers to successful restoration?

4.2 Method

4.2.1 Location of Restored Sites

In order to investigate the past effects of restoration, a number of sites of different ages were needed. As there is a lack of information about these restorations in published literature, it was necessary to quantify and map past restoration actions and their location before selecting sites. Data for the locations of HLS, Rural Pathfinder scheme (referred to as 2006-2009 restorations) and Life II and III programmes was provided directly by the Forestry Commission, who are responsible for the restoration work. The location of HLS restorations can be seen in figure 4.1 and the other schemes in figure 4.2. Proposed restoration work, labelled as red shapes in the figure, were included in the study to serve as a comparison to the restoration projects, to determine whether there were any differences between these areas and those that had experienced restoration. In addition, mires in favourable conservation status were identified from publically available Natural England data to essentially serve as a control group demonstrating the ideal outcome, based on management goals. These sites were selected from sites classified as favourable that had no documented restoration work. The Natural England data constituted the most recent condition assessment of the New Forest SSSI units at the time of the survey in 2013 (accessed from designatedsites.naturalengland.org.uk in 2014). Restoration of the mire areas consists of three actions: tree and shrub removal, in-filling of drainage ditches with heather bales, and blocking of drainage areas with clay plugs. Restoration often accompanied work on path areas to prevent erosion on the mires (personal observation). These actions were undertaken in concert or separately with each other depending on the location; unfortunately, there is a lack of detailed information regarding the specific locations of these interventions. Site visits were used to determine what restorations had been carried out in some areas where there was uncertainty about the approach.

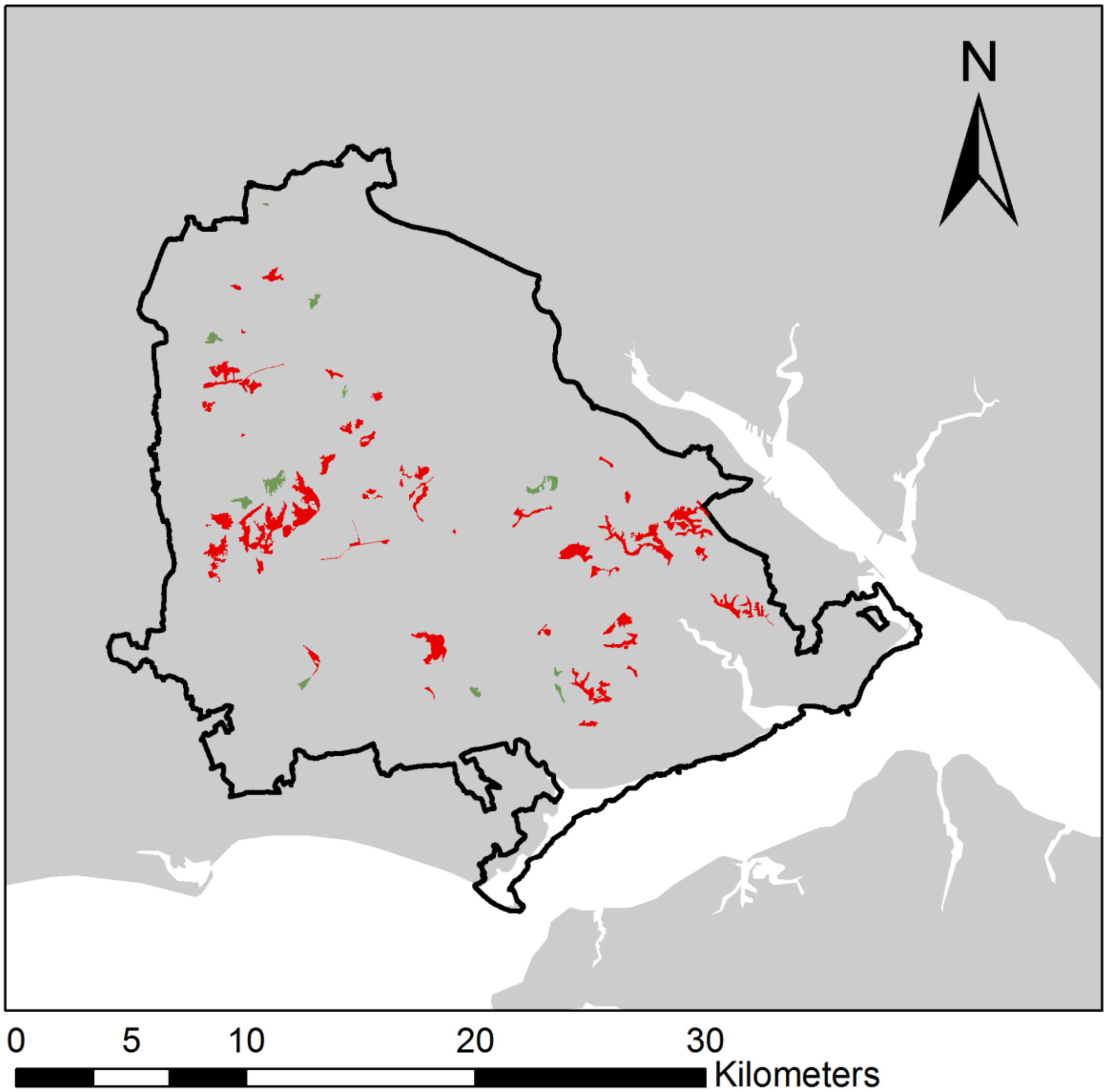

Figure 4.1 Location of wetland restorations carried out (green) and proposed (red) under the current HLS scheme, which began in 2010 and will run until 2020. Several of the proposed works are now underway, at the time of publication. Data supplied by the Forestry Commission.

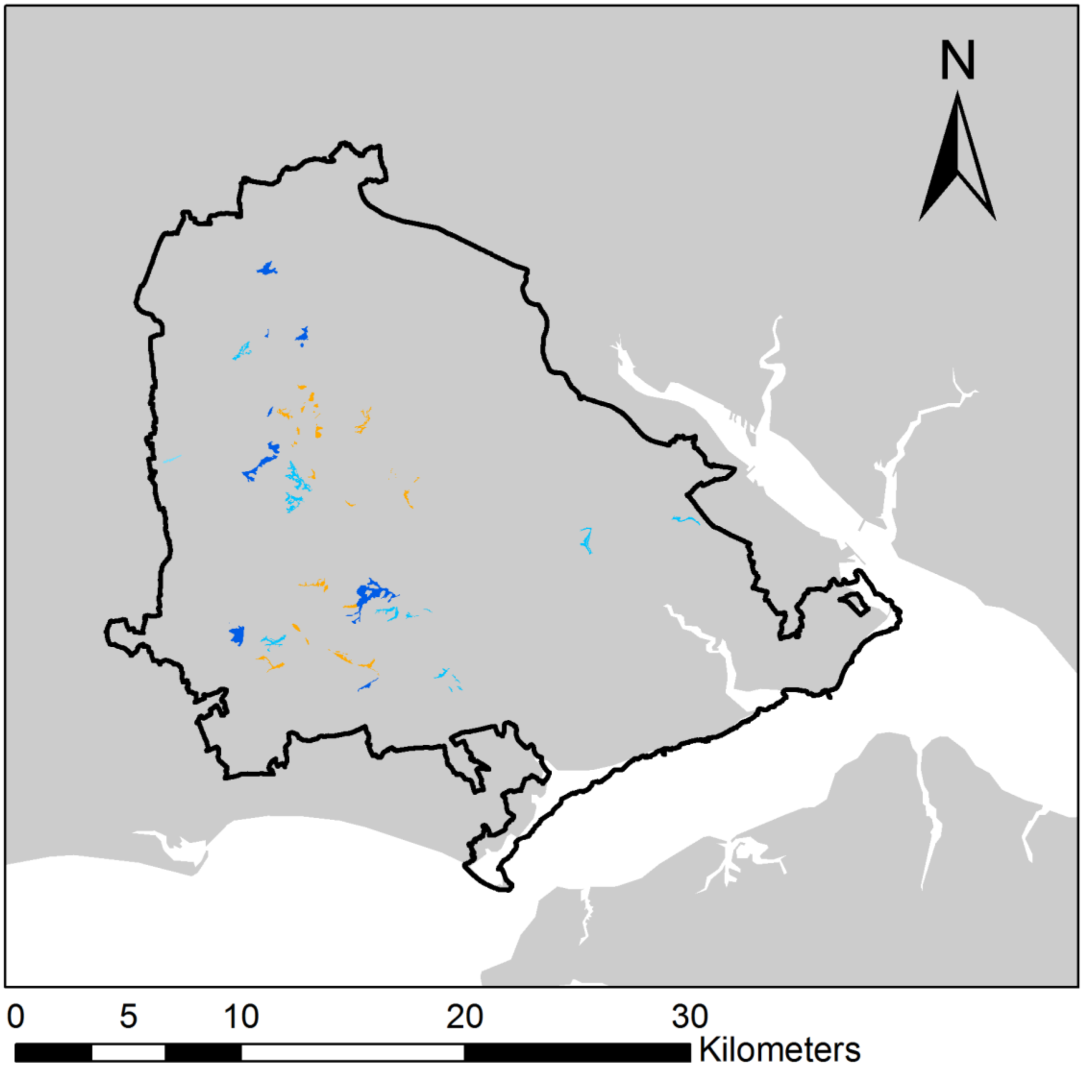

Figure 4.2 Location of wetland restorations carried out prior to the current scheme. From earliest to most recent: Life II scheme (Light Blue), Life III scheme (Orange) and the 2006-2009 Rural Pathfinder Scheme (Dark Blue). Data supplied by the Forestry Commission.

4.2.2 Site Selection

A space-for-time substitution approach was taken for the study, comparing vegetation changes across successive restoration programmes. Restorations included were the Life 2 (1997-2001), Life 3 (2002-2005), Rural Pathfinder Scheme (2006-2009) and HLS scheme (2010 – present). The study therefore shows a continuous pattern of different aged sites from 1997 to the time of the survey (2013). Additional mires were included, based on proposed future locations for restoration (considered 'degraded' sites) and those in favourable status without any documented restoration work. 10 sites of each type (referred to and analysed in the following text as "groups") were selected for study, resulting in a total of 60 mires to be surveyed. In order to minimise uncertainty from the use of different, independent locations, excessively small sites (those less than 0.5 ha), which are relatively frequent, were excluded from the study owing to potential edge effects, and areas of surface flowing water (fens, streams and rivers) were also excluded. The selected sites are shown in figure 4.3.

4.2.3 Quadrat survey

Plots were surveyed from July - September 2013. At each study location, a 50 x 50 m plot was set up in order to assess vegetation composition. The precise location was randomised within mires using the random point function in ArcGIS. If the centre point of the plot was located within 10 m of the mire edge or in non-mire vegetation or a stream channel, it was excluded and a different point used. This ensured that the location sampled included a similar habitat on all sites, but this approach may have reduced the full variability of site conditions being recorded.

Within each plot, a set of 10 2 m x 2 m quadrats was used to visually assess vegetation composition. The same researcher (the author) assessed all plots to minimize perception bias between plots. Vegetation composition was recorded by estimating the percentage cover of each plant species present, using five percent intervals, with a one percent score for flora of very low cover. Additional observations were made for the height of vegetation (using a drop disk method, Stewart *et al.,* 2001), the amount of surface water, soil or ground visible, and the number of seedlings of tree species. Vascular plants were identified to species level in the field where possible and were identified with keys from Rose and O' Reilly (2006) and Rose (1989). Taxonomy was based on Stace (2010). Where this was not possible, photographs were taken and a sample recovered for identification in the laboratory. Within quadrats, the total percentage of bryophyte cover was visually assessed, with the cover of *Sphagnum* sp. also recorded. The presence or absence of bryophyte species within each quadrat was then identified in the field, or based on photographic and sampled specimens. Identifications were based on Atherton *et al.* (2010) and Daniels and Eddy (1990).

4.2.4 Abiotic analysis of the substrate

Abiotic conditions were investigated by sampling the mire substrate at each location. A total of 10 samples were collected from each plot (where quadrats were sampled, located using GPS) using a 5cm diameter soil corer, to 20 cm depth. Bulk density was calculated using the volumetric method on fresh substrates (Rowell, 1994). Each of the soil samples was subsequently weighed, air-dried until there was no appreciable loss of weight (approximately 10-20 days), and re-weighed in order to calculate volumetric water content. Although air-drying in this manner is known to affect the soils (Kaiser *et al.,* 2015), all samples were treated the same way and stored in the same location, which should allow comparisons between sites to be made. Samples were then bulked and sieved (2 mm) prior to storage at low temperature (10^{o}C) in paper bags. After approximately 6 months of storage, substrate analyses were conducted by the analytical laboratory at Forest Research, Alice Holt, Hampshire, UK. Substrate pH was measured by suspension of 5 grams of soil with 25 ml of water, shaken on an orbital shaker for 15 min and rested for 45 min before being analysed by a Sentek pH electrode. Total Nitrogen, Carbon, Organic Carbon and Inorganic Carbon (TN, TC, TOC, TIC) were measured with Reference method ISO 10694 & 13878, a combustion method using a Carlo Erba CN analyser (Flash1112 series). Samples were ball milled before analysis, with 30 mg of milled soil weighed in tin capsules before being analysed simultaneously for total Carbon and total Nitrogen. Around 30 mg of soil were weighed in a silver capsule and placed in a furnace at 500^{o}C for 2 hours to remove organic carbon. These samples were then passed through the CN analyser in order to determine the inorganic fraction of carbon in soil. CN analysers work by combustion. Samples are loaded into the combustion by an autosampler, where each capsule is heated at 1600-1900 °C, and converted into elemental gases. A thermal conductivity detector (TCD) which produces an electrical signal proportional to the concentration of Nitrogen and Carbon, allows the element concentrations to be determined. Verado *et al.* (1990) provide a detailed description of this process.

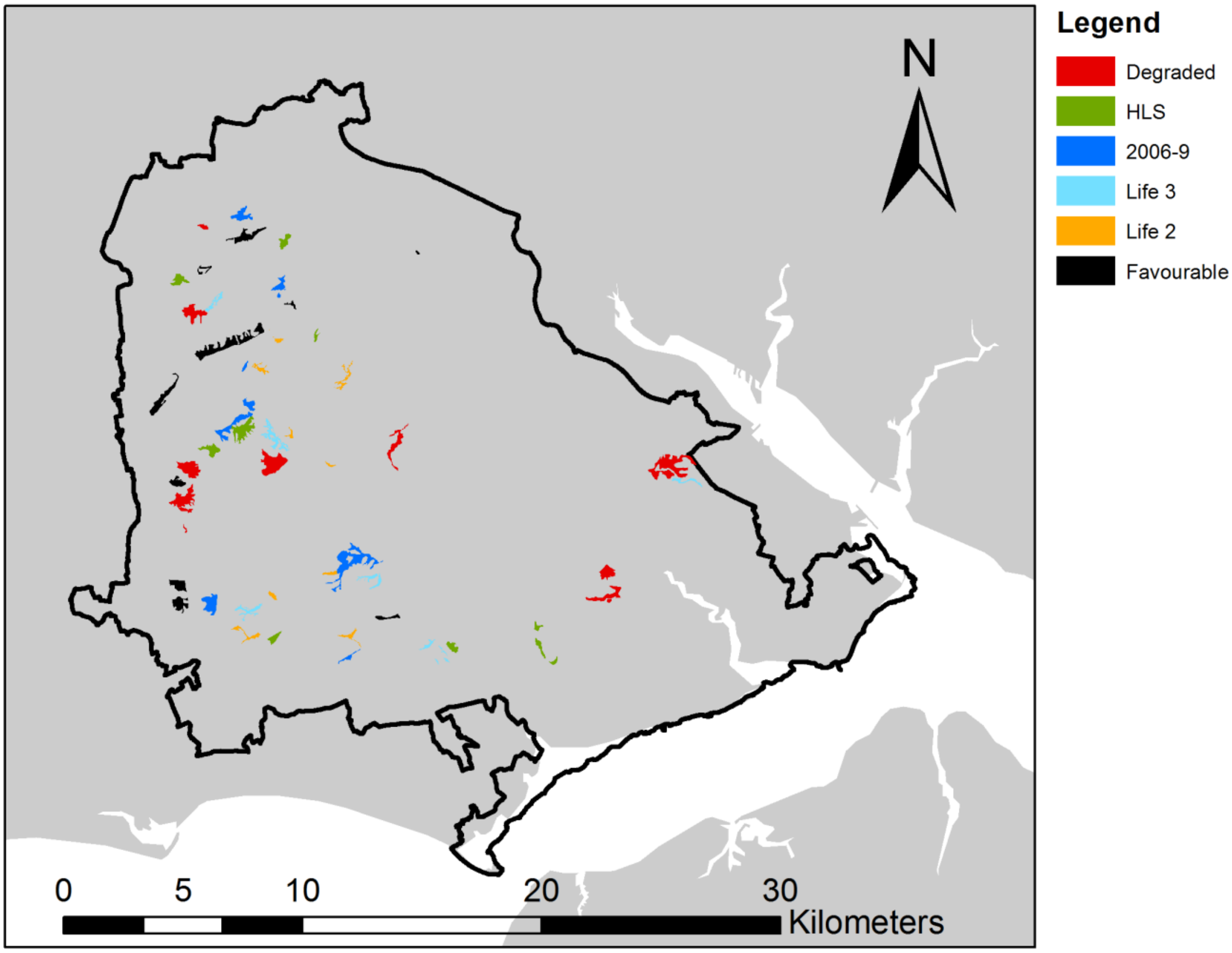

Figure 4.3 Location of the selected study sites and the boundary of the New Forest National Park. The legend indicates the groups of restorations, with 10 sites from each group included in the survey

4.2.5 Herbivory Index

Visual assessment of the amount of grazed vegetation was considered useful but potentially subjective, so in addition a 'grazing index' was created in order to give a score to different intensities of herbivory. The scoring method was based on Reimoser *et. al.* (1999), with some modifications to adapt the score to New Forest open habitat based on Bokdam and Gleichman, (2000); McNaughton, (1984) and Bakker *et. al*. (1984). Criteria are shown in table 4.1. Areas dominated by *Sphagnum* moss were not considered to be indicative or otherwise of grazing effect as they are unlikely to be browsed and can dominate the vegetation composition for other reasons. Signs of trampling however, did contribute to the score.

Table 4.1 Scoring system for Herbivory index on wetland sites

Score	Intensity of herbivory	Indicators
1	None	No signs of herbivory on all plants, extensive ground flora, no signs of tracks or dung, shrubs extensive in drier areas
2	Light	Extensive ground flora but some sign of browsed leaves, some tracks may be present
3	Moderate	Ground vegetation noticeably variable in height, shrubs patchy where they occur, some localised areas where grazing creates a lawn
4	Heavy	Ground flora shaped into lawn by grazing, but still with some variation in vegetation height, light tussock formation
5	Very Heavy	Ground vegetation <3cm tall, patches of bare soil present, extensive tracks and dung, shrubs absent or suppressed by grazing, herbaceous plants confined to inaccessible areas, Heavily tussocked vegetation

4.2.6 Statistical Analysis

Data were analysed with R 3.2.3, with the addition of the following packages: corrplot (Wei, 2013), lawstat (Gastwith *et al.,* 2015), multcomp (Hothorn *et al.,* 2008) and vegan (Oksanen *et al.,* 2015). The data were analysed by grouping sites restored under the same restoration program, in addition to comparator sites; this lead to six groups being compared: degraded sites (group 1), HLS sites (group 2; the most recent restorations), RP sites (group 3), Life III sites (group 4), Life II sites (group 5, the oldest restorations) and favourable sites (group 6). The structure of the data was initially examined using boxplots, and correlation of important variables (structural measurements and the most frequently recorded species) was investigated to examine their relationships and detect collinear variables. ANOVA tests were performed on *a priori* selected structural, abiotic and species variables. Assumptions were tested in R for normality (Q-Q plotting of residuals) and homogeneity of variance (Levene's test), and outliers were examined using boxplots. Tukey's HSD post-hoc was performed where ANOVA results were considered worthy of further investigation (p =~ <0.05). Where data could not be transformed satisfactorily, non-parametric Kruskal Wallis tests were used as an alternative. These were followed up with post-hoc tests. Eta-squared effect sizes for ANOVA tests were calculated using the lsr package (Navarro, 2015).

Non-metric multi-dimensional scaling (nMDS) was performed on the community data in order to examine any patterns in species distributions across the sites. Data were Hellinger transformed prior to analysis, as recommended by McGarigal (2000). The effectiveness of the nMDS ordination was determined using stress output and Shepard diagrams. Adonis analysis was used to test whether apparent differences between groups were supported by statistical tests. Constrained Correspondence Analysis (CCA), which constrains the placement of site and species points based on explanatory variables, was performed on the species data, with environmental and soil variables used as constraints. The final constraints were chosen by defining important influences a priori, investigation of co-linearity between variables (figure 4.12), and removal of data shown to have little influence. Species that occurred at less than five plots were excluded, as these are both unlikely to be correctly placed in ecological space and have an undue influence on the ordination (McGarigal, 2000). Species data were subsequently square-root transformed and standardised (using Decostand Range function). Environmental data was standardised to range from 0-1 so that different variables had comparable scales. The effectiveness of the constrained ordination was tested with Monte Carlo permutation on the axes and goodness-of-fit tests for species/samples diagnostics, with inertia and axes eigenvalues also examined.

4.3 Results

4.3.1 Response of the plant community following restoration

A selection of nine important vascular plants was examined in order to determine whether any of these showed significant differences between restoration groups. Of these, only *Carex panicea* and *Drosera rotundifolia* showed significant differences in ANOVA tests. *Carex panicea* (df = 5, F = 5.25, p = 0.001) showed a relatively complex structure in the post-hoc analysis, but the broad trend indicates that is was found at much lower abundance on older restorations and in favourable mire sites, with HLS and 2006-2009 restorations showing a cover of 4.5% ± 0.8 and 4.8% ± 1.0 respectively, against 1.4% ± 0.5 (Life III), 1.0% ± 0.4 (Life II) and 1.6% ± 0.4 (Favourable, mean ± SE for all groups). An effect size of η^2 = 0.33 indicated that substantial variation was attributable to the differences between groups. This species is often characteristic of edge habitat in typical valley mires, so may be indicative of slightly drier condition in the sites where it is found in higher abundance. Conversely, *Drosera rotundifolia* (df = 5, F = 2.62, p = 0.034) is a hydrophilous species, but one that would otherwise be expected to appear frequently in most, if not all of the surveyed sites. This species showed the highest abundance in late stage and favourable sites, although differences were only distinguished in post-hoc tests between early and middle stage restorations (figure 4.4). A relatively small effect size was found to be attributable to differences between restoration groups (η^2 =0.20). Cover in degraded and 2006-2009 restorations was 0.8% ± 0.2 and 0.6% ± 0.2 compared with 1.8 ± 0.2 for the highest recorded cover at Life III mires.

Of the other vascular species tested with ANOVA, *Molinia caerulea* (df = 5, F = 0.97, p = 0.443), *Erica tetralix* (df = 5, F = 0.85, p = 0.521), *Eriophorum angustifolium* (df = 5, F = 1.15, p = 0.346), *Rhyncospora alba* (df = 5, F = 0.81, p =0.549), *Juncus acutiflorus* (df = 5, F = 0.72, p = 0.609), *Narthercium ossifragum* (df = 5, F = 1.54, p = 0.195) and *Myrica gale* (df = 5, F = 1.06, p = 0.393) showed no significant statistical differences between restoration groups. These are important mire species but ubiquitous throughout the study, as such it may be the case that rarer species are more useful guides to vegetation community changes after restoration work. For this reason, more detailed ordination analyses were performed on the habitat community data in section 4.3.3. Despite the lack of differences between the restorations, the overall cover of *Molinia caerulea* (21.7% ± 1.2 SE) was very high for the type of habitat surveyed.

The total cover of *Sphagnum*, a very important component of the mire bryophyte community, showed no real differences between restorations (df = 5, *F* = 2.13, *p* = 0.076), with a mean cover recorded across all plots of 34.3% ± 2.5 SE. Individual *Sphagnum* species showed some differences. *Sphagnum papillosum*, a very important peat building species, showed statistically significant differences across restoration groups (df = 5, *F* = 4.22, *p* = 0.003), with post-hoc analysis highlighting the differences between late stage restorations and earlier ones (figure 4.5, present in all Life III, Life II and favourable plots compared with a probability of 0.5 ± 0.2 SE in 2006-2009 restorations). An effect size of $\eta^2 = 0.28$ showed that a fair proportion of the variance in the sample was attributable to between-group differences. *Sphagnum cuspidatum* showed a somewhat similar pattern (df = 5, *F* = 3.65, *p* = 0.006, figure 4.5, $\eta^2 = 0.25$) but lower abundance overall than *S. papillosum,* present with a probability of 0.1 ± 0.1 in 2006-2009 mire restorations, 0.8 ± 0.13 in Life II restorations and 0.7 ± 0.15 in favourable mires*.* *Sphagnum palustre* also showed differences between restorations (df = 5, *F* = 2.38, *p* = 0.051) but revealed a different pattern, with post-hoc tests showing the biggest differences between degraded sites (0.2 ± 0.13) and early restorations (0.8 ± 0.13 in HLS plots and 2006-2009 plots; see figure 4.5). An effect size of 0.18 suggests that most of the variation in the cover of *S. palustre* is not related to the restoration work. Of the other *Sphagnum* species examined, *S. fallax* (df = 5, *F* = 1.53, *p* = 0.195), *S. denticulatum* (df = 5, *F* = 0.51, *p* = 0.764), and *S. magellanicum* (df = 5, *F* = 1.84, *p* = 0.12) did not show any significant statistical differences between restorations. Changes in other *Sphagnum* species were examined as part of the vegetation community as a whole.

Relatively few measures of vegetation structure showed differences that could be distinguished statistically between restoration. The cover of bare ground showed evidence of differences in ANOVA (df = 5, *F* = 3.04, *p* = 0.017) between the oldest and youngest restorations (figure 4.6), but differences could only be determined with Tukey's HSD for the 2006-9 restorations (mean cover of 5.1% ± 1.7 SE) compared to the Life II (1.2% ± 0.5) and Life III (1.3% ± 0.5) restorations. The structure of the data may be partially responsible here, with bare ground showing unequal variances (Levene's test, *p* = 0.04) and a departure from the expected normal distribution in q-q plots. The ANOVA test itself should be fairly robust here, however, as the group sizes were equal. With an effect size of $\eta^2 = 0.22$, a fair proportion of the variance in the data could be attributed to differences between groups, but there were clearly several other sources. Trample damage also showed similar patterns, but with a marginally insignificant difference (df = 5, *F* = 2.35, *p* = 0.053), and not supported by post-hoc analysis (figure 4.6). Herbaceous plants, assessed as a group, showed significant differences across restorations (df = 5, *F* = 3.96, *p* = 0.04). Post-hoc analysis indicated that cover was highest at the 2006-2009 restorations (20.4% ± 4.9, figure 4.4), and lower at the Life III (8.8% ± 2.4), Life II (5.5% ± 1.1) and favourable (6.5% ± 1.8) mires. The data for herbaceous plants violated several of the ANOVA assumptions, however.

Lichen cover did not show any significant differences (df = 5, F = 0.60, *p* = 0.701) and there were relatively few detected in the survey. The lichens recorded, mostly *Cladonia* sp., were those more suited to drier heath conditions than those found in the mire sites. The cover of all recorded bryophyte species did not show significant differences between different restorations (df = 5, F = 2.19, *p* = 0.069). Total graminoid cover (df = 5, F = 0.36, *p* = 0.872) and dwarf shrub cover (df = 5, F = 1.33, *p* = 0.265) also failed to show any significant differences. Variables that were indicative of herbivorous pressure on the plots, including vegetation height (df = 5, F = 1.48, *p* = 0.213), the percentage of the plots showing evidence of herbivory (df = 5, F = 1.62, *p* = 0.171), and the calculated herbivory index (df = 5, F = 1.67, *p* = 0.157) did not show any differences between restoration groups but substantial variation was observed.

4.3.2. Abiotic conditions following restoration

Of the measured abiotic variables, soil pH was similar between all restorations (df = 5, F = 0.23, *p* = 0.946), with a mean of 4.74 ± 0.05. Total nitrogen and total organic carbon in the substrate were statistically distinguished, along with volumetric water content and bulk density. Together, these suggested that changes in soil characteristics occurred as restorations age. In soil nitrogen (df = 5, F = 3.33, *p* = 0.011), only middle-late stage (Life III) restorations could be distinguished statistically from the degraded sites (figure 4.6, Tukey's HSD). An effect size of $\eta^2 = 0.24$ shows that a significant portion of variance can be related to the restoration groups. Soil carbon (df = 5, F = 3.36, *p* = 0.01) showed a very similar pattern (figure 4.6, Tukey's HSD), with a similar effect size ($\eta^2 = 0.24$). Despite changes in nitrogen and carbon, C/N ratios did not show any differences between groups (df = 5, F = 1.44, *p* = 0.226), which may illustrate that substrate processes were somewhat similar across the study despite the restoration work.

Volumetric water content, a potentially critical measure for the success of restorations, did show differences between sample groups (df = 5, F = 2.64, p = 0.033), but the pattern was not quite the one that was expected. Generally, water content was relatively high across all sites (69% ± 0.02 SE), but Tukey post hoc tests could not distinguish between groups, despite the pattern that appeared (figure 4.7). This was likely a result of high variation in the data. Unexpectedly, sites not subject to restoration but considered favourable (by management records), did not show any difference from degraded sites. The effect size ($\eta^2 = 0.20$) suggested that any changes in the water content were relatively small. Surface water also failed to show differences determinable across management groups (df = 5, $F = 0.75$, $p = 0.585$), which could potentially be a result of variability in weather conditions during the survey period which could also affect volumetric content (i.e. after extended periods of rainfall). However, weather was relatively stable in the study period and examining the two water measures together, it appears that restoration has had little detectable effect on the presence of water in the habitat during the survey.

The bulk density of soil (df = 5, F = 4.61, $p = 0.001$) showed an interesting pattern with some similarities to water content, but with much higher and significant differences between groups. Both favourable and degraded sites showed similar, relatively high bulk density scores (0.36 g cm^{-3} ± 0.05 and 0.37 g cm^{-3} ± 0.6 respectively), whereas the oldest restorations showed low scores (Life III = 0.15 g cm-3 ± 0.03, Life II = 0.17 g cm-3 ± 0.4; see figure 4.7). Other restorations showed medium scores that could not be statistically distinguished from either the older sites or those not subject to management. This appears to show a pattern of lowering bulk density as sites age following restoration. However, the unexpected high density of the favourable sites again suggests that these areas may not be appropriate comparators for successful restorations. The effect size ($\eta^2 = 0.30$) indicates a moderately sized effect of restoration groups on the differences in bulk density.

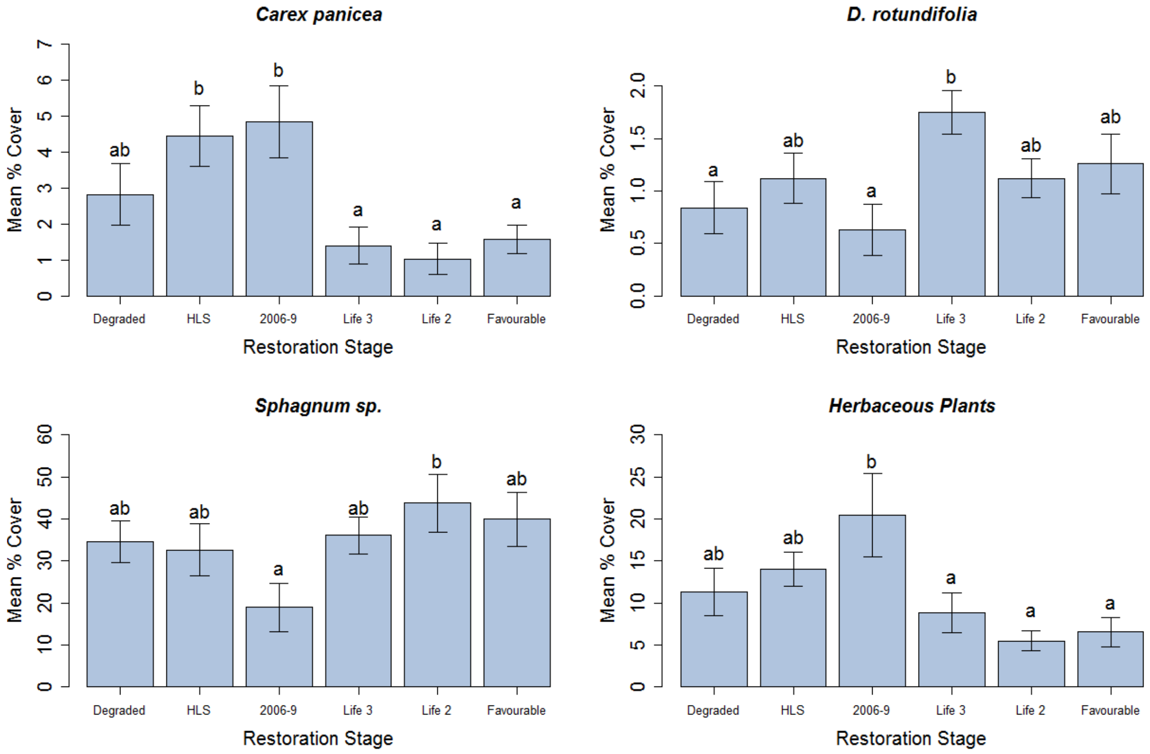

Figure 4.4 Community structure, including species, which differed between restoration groups following ANOVA analysis. Error bars show standard error. Values grouped by the same letter are not significantly different from each other at P ≤ 0.05 (Tukey's HSD). α was adjusted for family-wise false discovery rate.

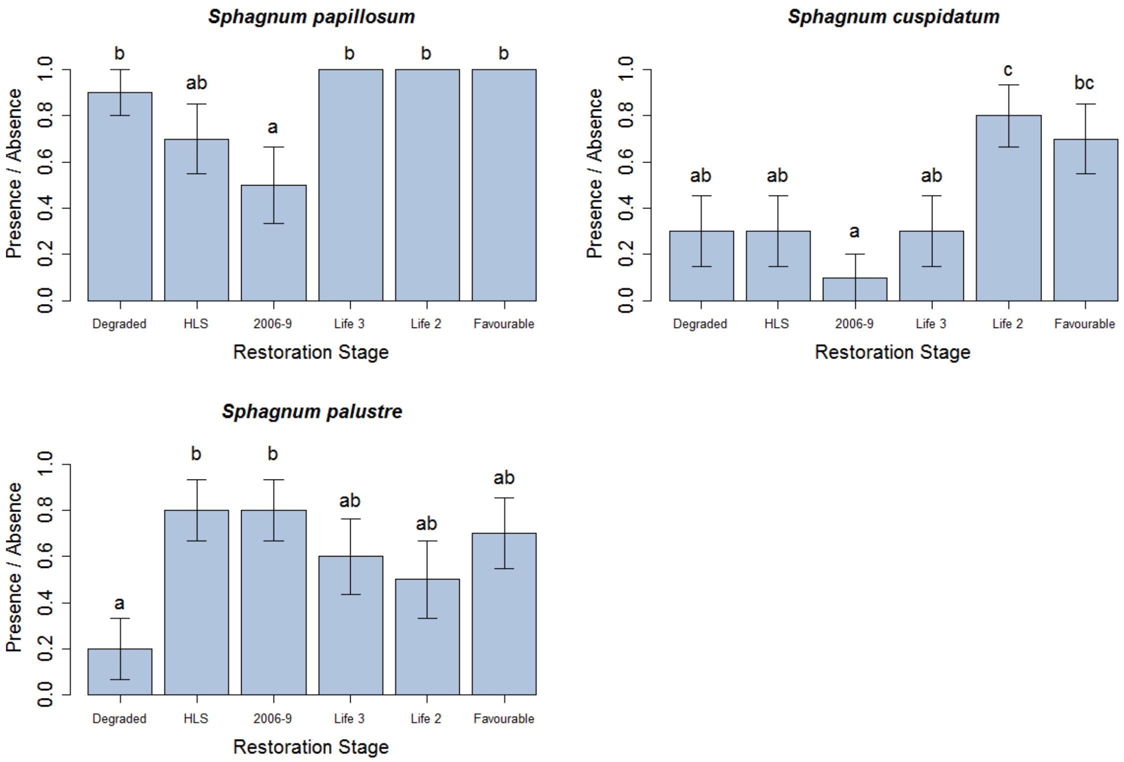

Figure 4.5 Important *Sphagnum* species that showed significant differences across restoration groups in ANOVA analysis. Error bars show standard error. Values grouped by the same letter are not significantly different from each other at $P \leq 0.05$ (Tukey's HSD). α was adjusted for family-wise false discovery rate.

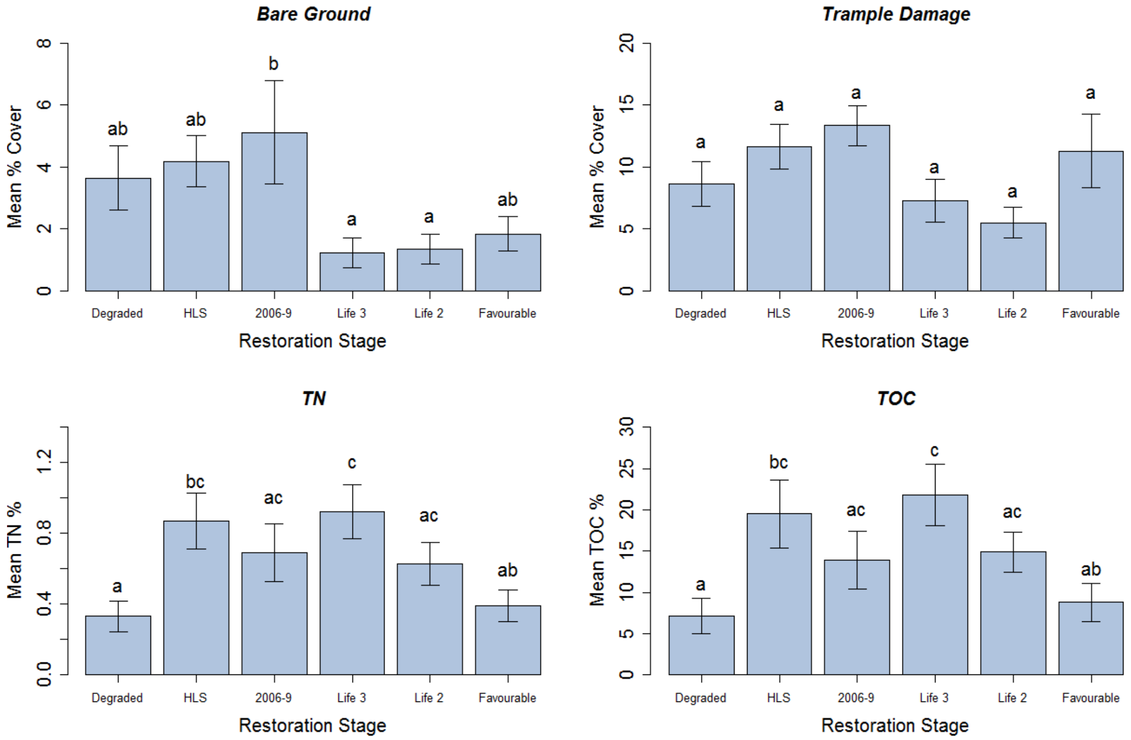

Figure 4.6 Abiotic conditions that significantly differed between restoration groups in ANOVA analysis. TN = Total Nitrogen content of soil, TOC = Total Organic Carbon in soil. Error bars show standard error. Values grouped by the same letter are not significantly different from each other at $P \leq 0.05$ (Tukey's HSD). α was adjusted for family-wise false discovery rate.

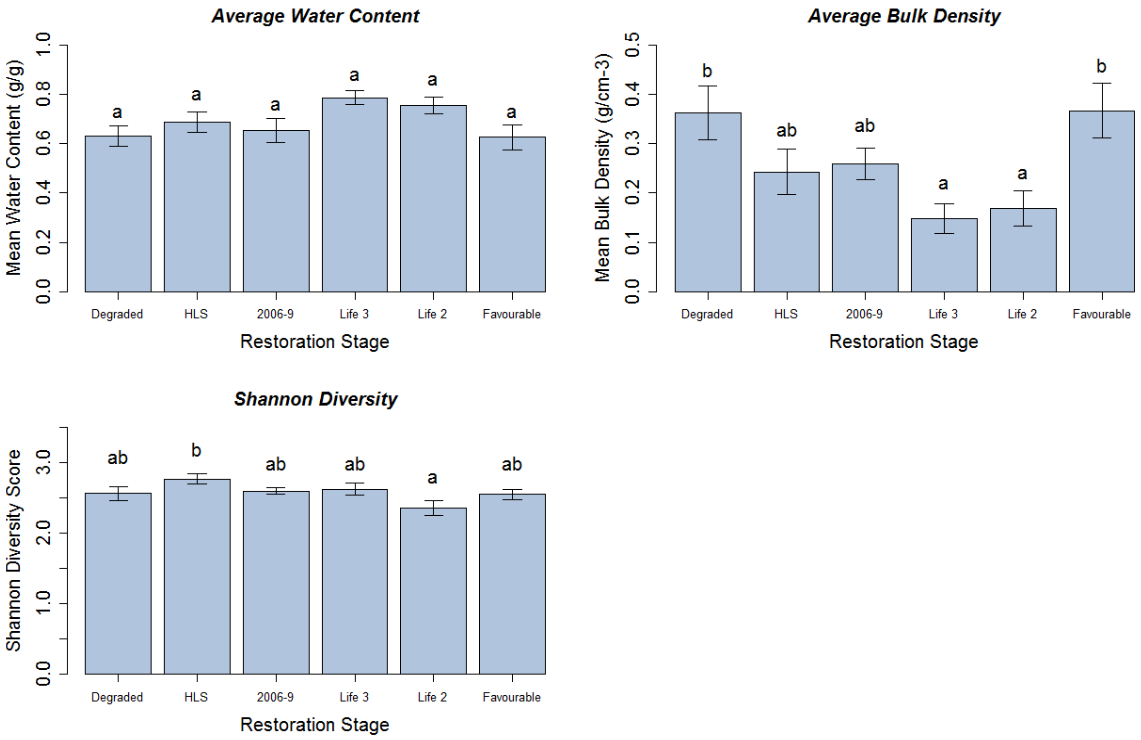

Figure 4.7 Abiotic and diversity conditions that differed between restoration groups following ANOVA analysis. Error bars show standard error. Values grouped by the same letter are not significantly different from each other at $P \leq 0.05$ (Tukey's HSD). α was adjusted for family-wise false discovery rate.

4.3.3 Patterns in community structure

The ordination plot produced by nMDS (figure 4.8) shows a scattered distribution of sites with considerable overlap, but the relatively high stress level of 0.23 means some caution should be applied to the interpretation, and that relatively limited conclusions can be drawn from the 2-dimensional output. There appears to be some grouping with the oldest sites (Life II and Life III restorations) being clustered with favourable areas, whereas earlier restorations and differ in both the location of their plots. There also appears to be a greater degree of scatter in the position of other groups, including those of favourable sites, suggestive of a dispersion effect. This demonstrates increased variability in the vegetation in these groups. Overall, the diagram indicates that there are some differences between the groups in ecological space, but that these are relatively subtle and with considerable overlap between groups. Species were placed in a scattered pattern, with those associated with mires in the centre of the plot, and non-mire species around the edges. Selected species that are important to valley mire habitats (particularly the M21 community), as indicated by JNCC (2004), showed an association with the older restorations and favourable sites. The placement of many early restorations and degraded sites away from these species is clearly shown. Adonis analysis, a robust test of the similarities between groups (Oksanen *et al.*, 2015), found a significant result (p = 0.005) indicating some differences between groups, suggesting that the restoration groups explain 13 % of the variance in the data. This supports the impression given by the ordination diagram that different stages of restoration did feature slightly different communities, but that there was substantial variation that was not attributable to the restoration groups. Shepard diagrams and Adonis output are shown in Appendix IV.

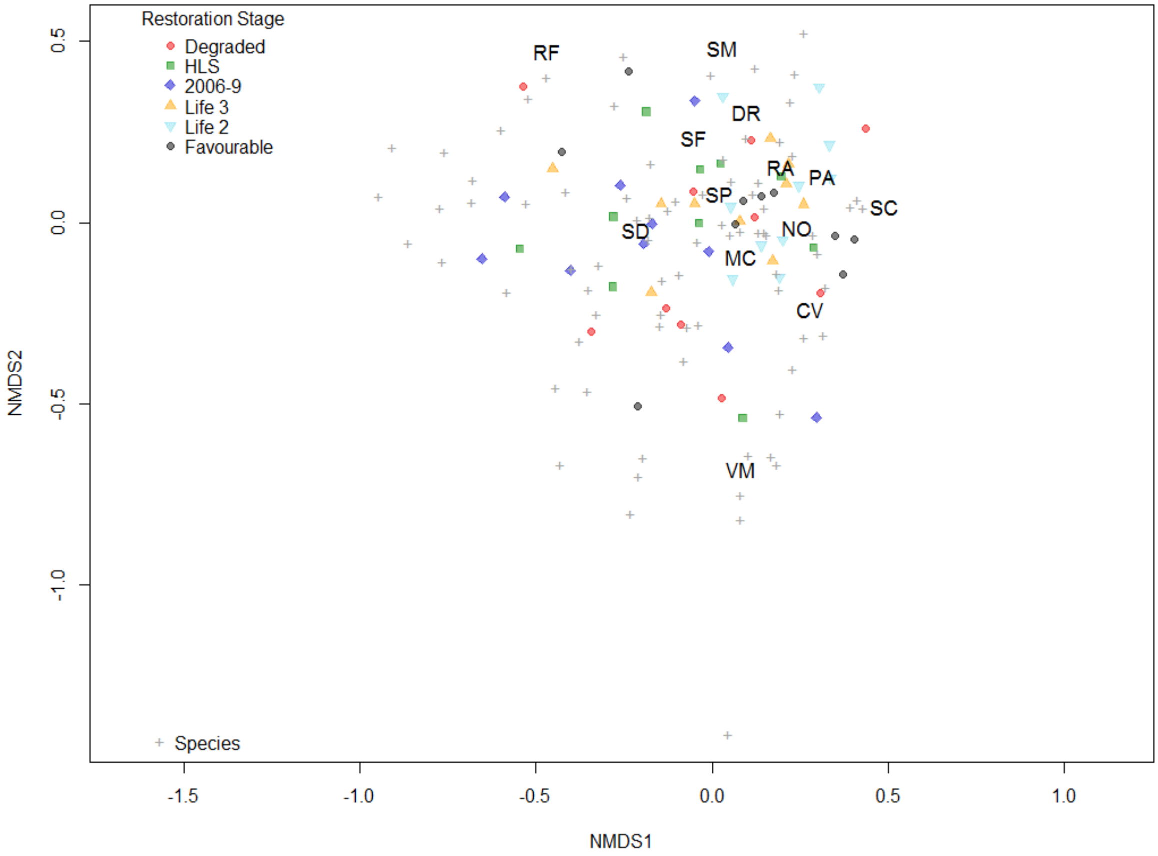

Figure 4.8 nMDS ordination output showing the location of important species (selected from JNCC guidelines) and plots. A partial association with these species and older restorations is seen, but generally there appears to be considerable variation within restoration groups as opposed to between them. A relatively high stress level (0.23) means that the ordination should be interpreted cautiously. Species are labelled as follows, RF = *Rubus fruticosa*, SM = *Sphagnum magellanicum*, DR = *Drosera rotundifolia*, RA = *Rhynchospora alba*, SF = *S. fallax*, SP = *S. papillosum*, PA = *Phragmites australis*, NO = *Narthecium ossifragum*, SD = *S. denticulatum*, SC = *S. cuspidatum*, MC = *Molinia caerulea*, CV = *Calluna vulgaris* and VM = *Vaccinium myrtillus*.

4.3.4 What influences variation in mire sites?

It is clear that considerable variability in the data is not directly attributable to restoration. Determining the other influences on community composition, and their relative importance, is essential for a successful management approach to restoration and to consider whether past restorations have succeeded. To address this, Figure 4.9 shows an ordination graph produced with CCA. In this ordination, the intensity of herbivory appears to have greater measured impact on the community than surface or soil moisture levels, indicated by the relative length of the vector arrows. However, there does seem to be a relationship between these variables with both highly browsed and trampled sites also those featuring less surface water and often less soil moisture. Generally, the oldest restorations cluster together, and there is more variability in the younger sites. This could be interpreted as showing successful restoration in older sites, but picture is somewhat confused by favourable sites also being highly variable, in broadly the same dimensions as younger sites. In fact, favourable and unfavourable sites seem to be poor comparisons with the restoration work, as they are both highly variable and appear to be more closely associated with herbivory than with other restorations.

Permutation tests of the overall ordination model showed a significant result (df = 5, $F = 1.68$, $p = 0.001$), and tests on the predictive variables showed significance for surface water ($F = 1.5033$, $p = 0.012$), trample damage ($F = 1.75$, $p = 0.002$), and the percentage of vegetation that was browsed ($F = 2.54$, $p = 0.001$). Soil moisture was relatively close to significance ($F = 1.32$, $p = 0.090$), but total nitrogen in soil was not ($F = 1.10$, $p = 0.351$) and is therefore unlikely to be the cause of vegetation variability in this study. Tests were also run on the plot axes; both of these were significant (CCA1, $F = 3.53$, $p = 0.001$; CCA2, $F = 1.87$, $p = 0.001$), and further axes scores were not significant. This suggests that the ordination, and the 2-D plot, are therefore good representations of the position of sites and species in environmental space. Eigenvalues, variance inflation factors and the proportion of variance explained by the test are given in Appendix VI.

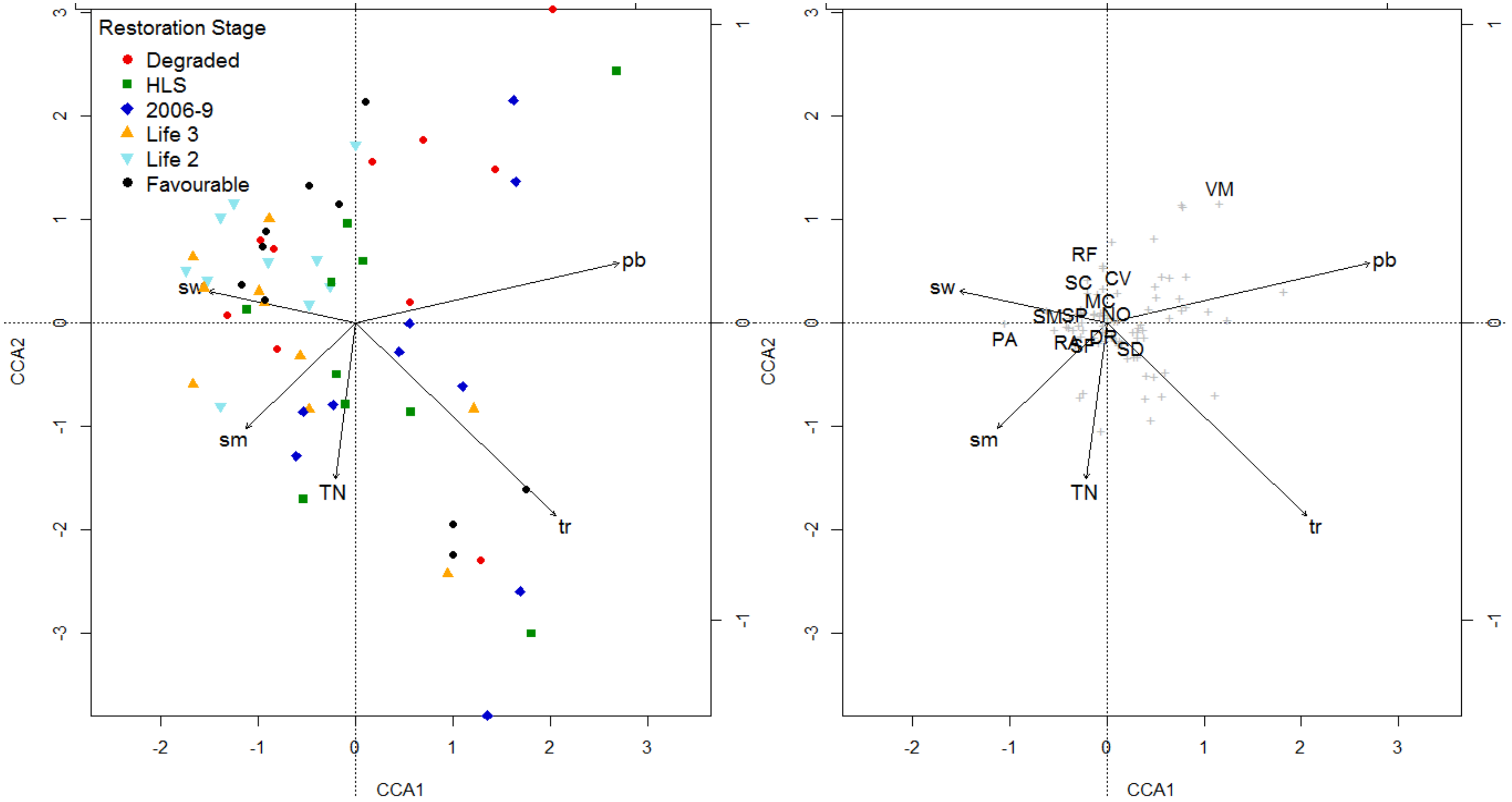

Figure 4.9 CCA ordination plots showing placement of groups (left) and species (right). Arrows indicate constraints; these are explanatory variables. Species occurring in <5 plots were dropped, and species data square root transformed. Environmental variables were standardised to range. For species, RF = *Rubus fruticosa*, SM = *Sphagnum magellanicum*, DR = *Drosera rotundifolia*, RA = *Rhynchospora alba*, SF = *S. fallax*, SP = *S. papillosum*, PA = *Phragmites australis*, NO = *Narthecium ossifragum*, SD = *S. denticulatum*, SC = *S. cuspidatum*, MC = *Molinia caerulea*, CV = *Calluna vulgaris* and VM = *Vaccinium myrtillus*. For explanatory variables, TN = Total N, tr = trample damage, pb = percentage browsed, sw = surface water and sm = soil moisture content.

Surface water was a better predictor of the vegetation community than volumetric content; the latter not a reliable predictor of habitat community according to test results. In terms of the interaction between explanatory variables, of particular note are the strong negative correlations between bulk density and the presence of water and nutrients in the soil (figure 4.12), suggestive of different soil characteristics. There was also a positive relationship between grazing and bare ground and the presence of some plant species, most notably *Carex panicea*. Surprisingly, there were few correlations with measures of water (both within the soil and on the surface of the plot) and plant species. Younger restorations (RP and HLS) and degraded sites appeared to feature a community driven by high herbivory and high trampling damage, as opposed to later restorations, and some favourable sites, where water retention appeared to be a bigger influence. The favourable surveyed sites also appeared to be subject to greater influence from herbivory than the oldest restorations.

Plotting the averaged positions of the restoration groups, and the control groups of favourable and degraded condition, allows the differences between groups to be investigated more easily (figure 4.10). It appears that recent restorations (HLS and RP) show similar, highly variable communities, and were associated with higher levels of herbivory but variable soil and surface water. The Life III restorations appear to be transitional between these and the Life II restorations. These are progressively less variable and increasingly associated with typical mire communities and higher amounts of water. Of the control groups, favourable mires show an overlap with the oldest mire restorations, but also some of the recent ones, and with degraded sites. The degraded group does not seem to show association with many of the restoration groups, but instead a particular set of communities and conditions that differs from both. The high variability of the vegetation community, both between and within sites, is likely to contribute to this overlap. A similar ordination overlay showing mire structure (figure 4.11), shows clearly defined groups, demonstrating the importance of structural conditions on the vegetation community and abiotic conditions.

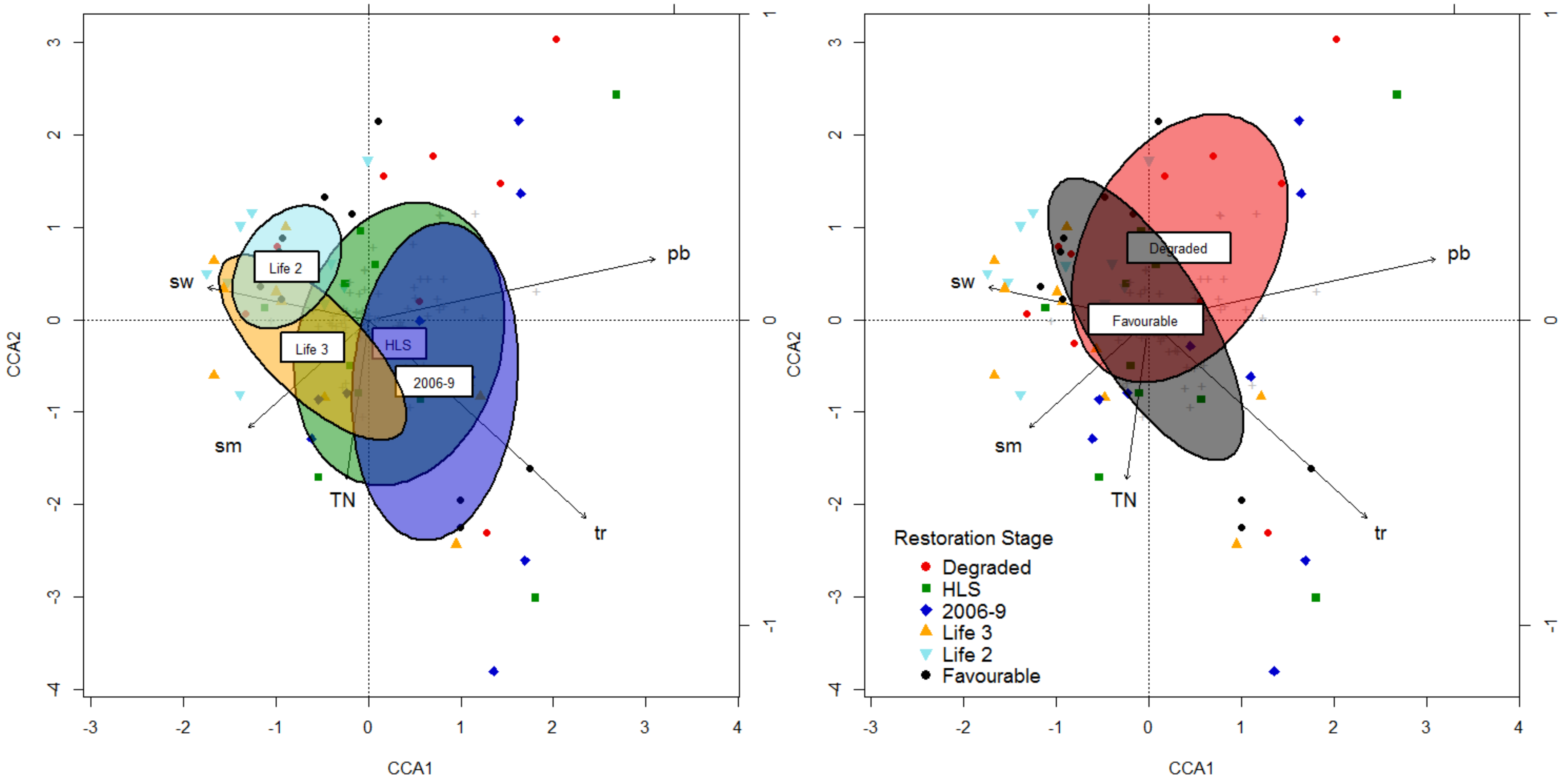

Figure 4.10 CCA ordination showing the placement of restoration plots (left) and control groups (right). The ellipses are drawn around the standard error of the factor means. For explanatory variables, TN = Total N, tr = trample damage, pb = percentage browsed, sw = surface water and sm = soil moisture content.

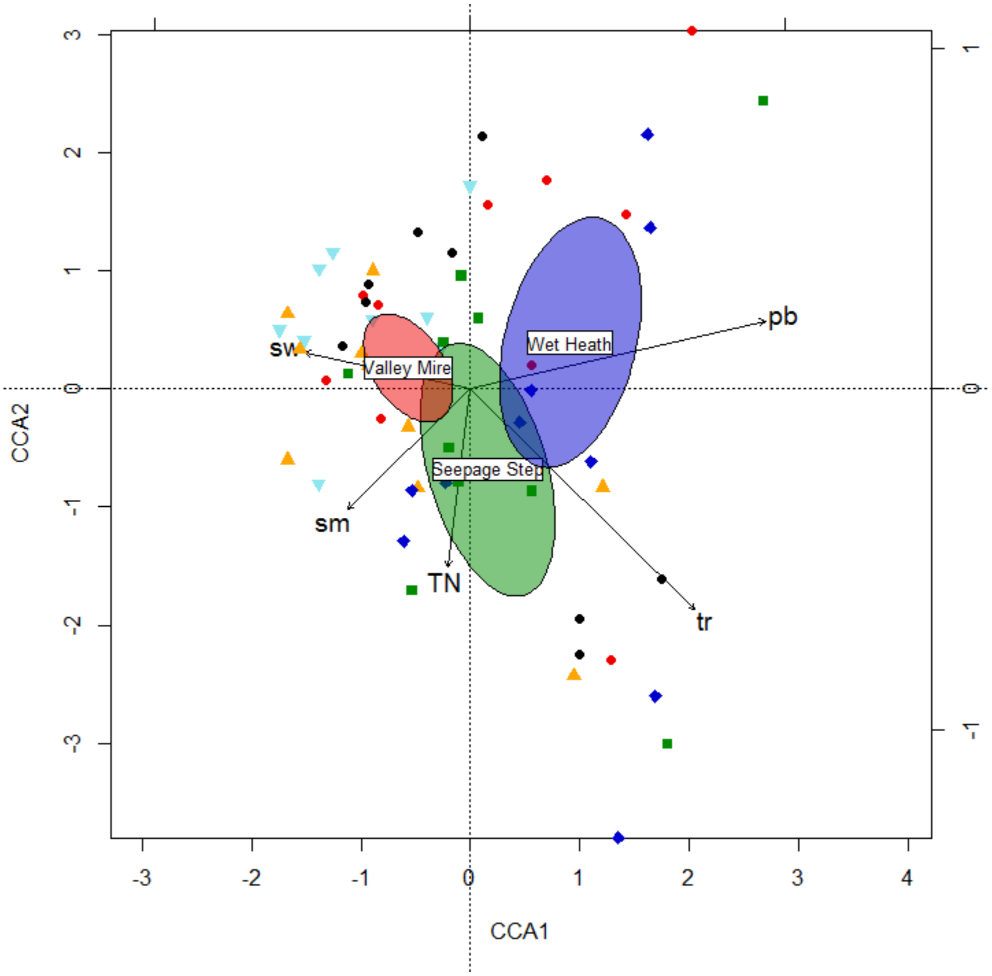

Figure 4.11 CCA showing ellipses around mire structures recorded during the survey. There appear to be important differences in the community based on the mire structure (note, this structure is easily affected by the amount of water present). For explanatory variables, TN = Total N, tr = trample damage, pb = percentage browsed, sw = surface water and sm = soil moisture content.

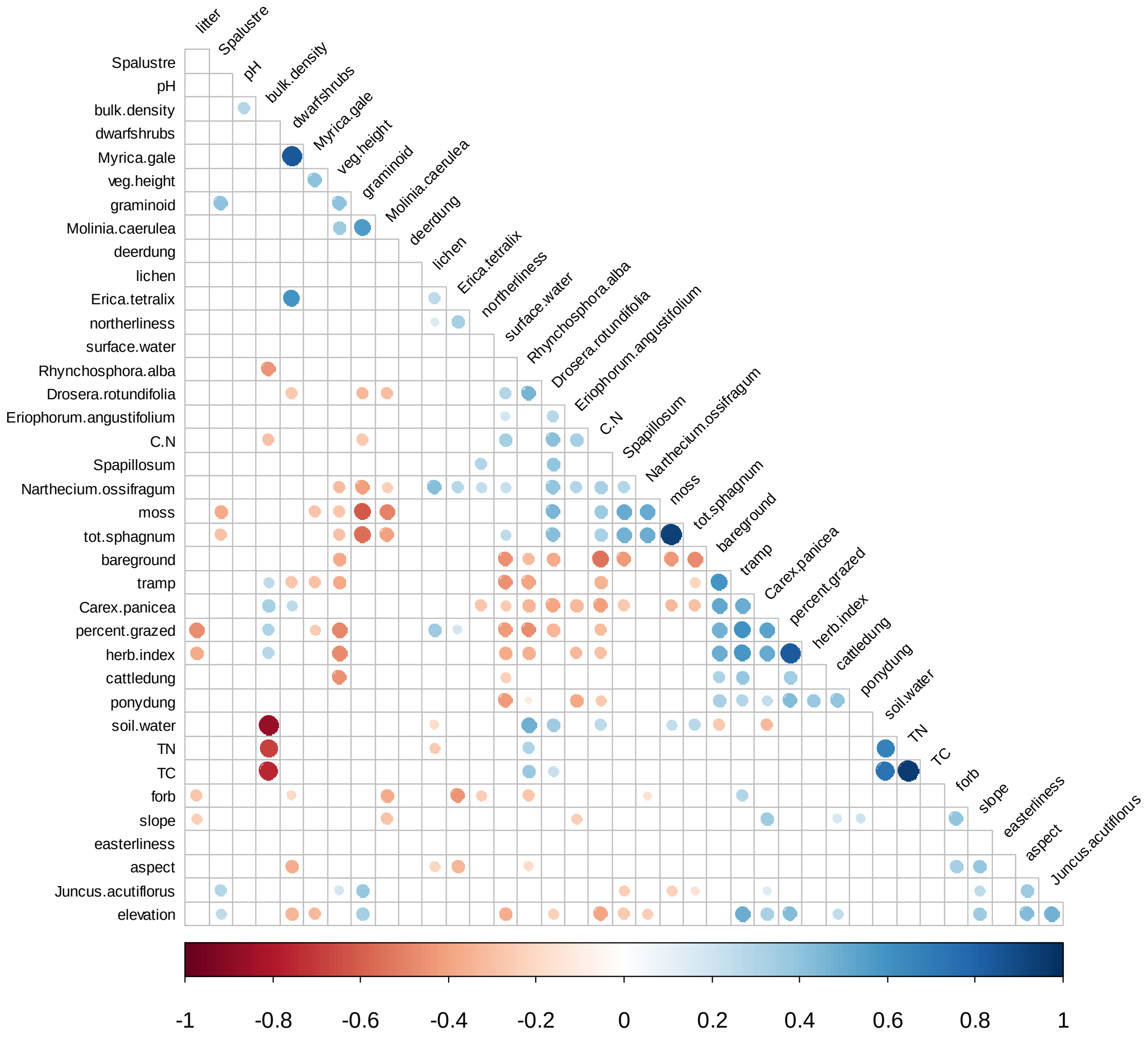

Figure 4.12 Spearman correlation matrix showing the strength of correlation coefficients between selected variables. Only those with $p < 0.05$ are shown. Points are shown in stronger colours and larger size relating to the strength of correlation (r^2), with red indicating negative correlations and blue positive correlations. Numerical values are available in the digital supplement.

4.4. Discussion

4.4.1 Changes in the plant community following restoration work

Change in valley mire communities in response to restoration are poorly understood, particularly in lowland situations such as the New Forest. This study has demonstrated a variety of different responses in the plant community. A few important *Sphagnum* species showed changes, but the overall cover of *Sphagnum* did not. *Sphagnum cuspidatum*, an indicator of water retention in mires (JNCC, 2004) and *S. papillosum*, an important peat building species (Daniels & Eddy, 1990) were more likely to be present on older restorations and in favourable sites. *S. papillosum* is intolerant of drought conditions (Clymo & Haywood, 1982) and has increased in abundance in other successful restoration projects (Gonzalez *et al.,* 2014). Peat building species including *S. papillosum* and *S. magellanicum* are typically slow to respond to management (Smolders *et al.* 2003), unlike other species of *Sphagnum* (Maanavilja *et al.,* 2015), highlighting the importance of examining long-term trends. The timescales involved in this study, and the initial vegetation communities, should have provided better circumstances for the recovery of these species, but *S. magellanicum* showed no changes following restoration work in this study. A particular increase was notable in *S. palustre* in early restorations when compared to degraded sites. In the UK, this species is often associated with high nutrient conditions relative to other *Sphagnum* species (Atherton *et al.*, 2010), and these conditions are expected to temporarily follow restoration work as a result of disturbance (Haapalehto *et al.,* 2014). A possible reason for a lack of change in overall *Sphagnum* cover is competitive exclusion by related species, described by Robroek *et al.* (2007).

Unlike several other studies (e.g. Komulainen *et al.,* 1997; Tuittila *et al.,* 2000), no changes in *Eriophorum* spp. were detected here, although increases in *Carex* spp. were similar to Tuittila *et al.* (2000; different individual species were responsible). In terms of species groups or guilds, herbaceous plants peaked in abundance in the middle restorations (2006-2009), which could be interpreted as showing temporary colonisation as mires are altered following restoration, although this is not supported by the conclusions of some studies (Sottocornola *et al.,* 2009). A low cover of lichen was found throughout all plots; abundant cover of lichen is considered a negative indicator of mire health owing to their adaptations to dry conditions (Gonzalez *et al.,* 2014). The cover of vascular plants, and in particular Graminoids, was generally quite high in proportion to the cover of bryophytes, but this is a pattern consistent with other valley mires where soligenous peat formation occurs (Malmer *et al.,* 2003).

However, *Molinia caerulea,* sometimes considered an invasive species in mires (Gogo *et al.,* 2011), showed high cover in all areas.

Ordination analysis of the vegetation community distinguished older restorations from more recent ones, and older sites also showed less variation in ecological space. Important mire species, identified from the JNCC (2004), were also more likely to be found in these older restorations than in sites of younger ages and in degraded sites. Section 4.4.3 examines the causes of this distribution in greater detail, with the addition of explanatory variables. Changes in vegetation composition caused by management are often expected to be relatively small in mire habitats (Haapalehto *et al.*, 2010), and high variability of the vegetation community often occurs naturally (Falk *et al.,* 2015) which reflect some of the findings here. The relative similarity of degraded and "favourable" sites supports observations by Wilson *et al.* (2011) that there is sometimes limited evidence linking drainage with community declines in peatlands. Overall, changes reflected the findings of some restoration studies in other mires showing a demonstrable impact on relatively few species (Haapalehto *et al.*, 2010), but also contrast with others that showed rapid and dramatic community changes in a relatively short space of time (Maanavilja *et al.*, 2015; Poulin *et al.,* 2013). There are notable similarities here with a study of grazed mires by Wilson *et al.* (2011), who found similar species and conditions in drained and undrained areas, although they did record differences in the dominant vegetation community where management was applied.

4.4.2 Abiotic and structural changes following restoration

The effect of restoration interventions on substrate conditions has not previously been examined in the New Forest, despite the importance for restoration success. This effect was small, although soil nitrogen, carbon, and bulk density showed changes between restoration groups. An increase of N and C in the early stages of restoration matches finds by Haapalehto *et al.* (2014) and is often expected (Vasander *et al.,* 2003). Elevated concentrations of N are often associated with increased decomposition and mineralisation caused by aeration of the mire substrate (Laiho *et al.,* 1999; Sundström *et al.,* 2000), suggesting that water retention has been poor in these areas, but could conceivably be related to disturbance (such as management interventions or herbivory). Carbon concentrations in the substrate were relatively low overall, as were the associated C/N ratios, when compared to other some other mires (eg over 50% C and C/N ratios of 30-47 in Tuittila *et al.,* 2000), but quite comparable to others that had experience drainage (Urbanová *et al.,* 2011). Lower C/N ratios would again be expected in areas with higher decomposition rates and low water retention. There may also have been some loss of C as a result of the sampling method (i.e. sieving the substrate) which would affect these ratios in comparison to other studies. Substrate bulk density was lowest in the oldest restoration sites (Life II and III) but surprisingly did not differ between favourable and unfavourable sites. Aside from these older restorations, the density of substrate was much higher in this study than in comparable ecosystems (eg Tuittila *et al.,* 2000 who found density of 0.1-0.2 g cm-^{3} and Bragazza & Gerdol, 1999 who found density of approximately 0.25 g cm-^{3}). Such values are associated with drainage (Minkkinen and Laine, 1998) and are similar to those in some drained mires (Urbanová *et al.,* 2011), but could also be related to the high concentration of large herbivores in the New Forest, which compact the substrate in and around the mires. High bulk density scores are indicative of poor conditions for *Sphagnum* development (Price, 1997), so this is a potential area of concern for the recovery of species.

There were few indications that restorations had increased either volumetric water content or the surface cover of water. Ditch blocking should lead to an immediate rise in the water retention and this has been found in many other studies (e.g. Haapalehto *et al.,* 2010; Verberk *et al.,* 2010; Ruseckas and Grigaliunas, 2008), although not all (Maanavilja *et al.*, 2015) and with sometimes inconsistent results (Jarasius *et al.,* 2015). The volumetric water content *was* broadly comparable to undisturbed mires in other studies (eg. 60-80% in Kellner & Halldin, 2002, who measured to the same depth of substrate as this study), potentially indicating that water conditions were adequate. However, there was high variability within the restoration groups which could have hampered any ability to detect changes. The failure of heather bales to stop drainage from the sites was directly observed (figure 4.13), and even where drains had been successfully blocked, permanent physical changes in peat substrates that reduce water retention may have occurred (Vasander *et al.,* 2003; Smolders *et al.,* 2003; Grootjans *et al.*, 2002; Holden & Burt 2002). Unexpectedly, in terms of water content, favourable status sites did not differ from those where restoration actions were proposed, a pattern reflecting that found in the vegetation community. There are a number of possibilities that could explain this pattern: favourable sites could have featured a wider range of natural conditions than restoration areas, favourable status may not adequately describe mire condition owing to inadequacies in condition assessment or alternatively that degraded sites had not lost significant amounts of water, the latter reflecting findings by Wilson *et al.* (2011) in blanket bog. Deleterious effects could still occur in this situation, however, because of the likelihood of increased water flow through the substrate. Restoration did not influence pH values, which were comparable to those of previous studies of the New Forest (Newbould and Gorham, 1956) and other Atlantic bog communities (Sottocornola *et al.,* 2009). As noted by Newbould and Gorham, the values suggest that there is some groundwater contribution to the water regime in addition to rainfall, as they are higher than typical ombrotrophic communities.

Structurally, there were also relatively few changes, and those that did occur may not be important changes. Areas of bare substrate or peat varied between restoration groups; these are often considered to be negative indicators of restoration success (Gonzalez *et al.,* 2013) but were not very numerous even in the degraded mires recorded, which may limit the use of bare substrate as an indicator. Bare ground was often associated with trampling damage from herbivores, and so the cause of such changes in the cover of bare ground is difficult to directly attribute to restoration work. Trampling and grazing intensity did not differ between sites, but the impact of grazing on both the vegetation community and restoration effectiveness is explored further in the following sections.

Figure 4.13 Failed heather bale restoration and erosion around a channel, downstream from one of the RP 2006-2009 mire sites. Note also the dry condition of the channel. The surrounding area is also heavily grazed but this is unlikely to be related to the failure of the heather bale. Photograph by the author.

4.4.3 Variation in mires and their ecological causes

Previous studies examining the causes of variation in New Forest mires were all carried out before any restoration work took place, so understanding the patterns in distribution is important to any assessment of restoration. Three main drivers in community diversity were identified with this study: herbivory, the presences of water, and mire structure. This study has shown like many others (e.g. Glaser *et al.,* 1990; Bubier *et al.,* 1996; Sottocornola *et al.,* 2009) that there was a strong association between water and the distribution of the plant community. CCA revealed that older restorations had a positive relationship with water levels (both surface water cover and volumetric content) while more recent restorations had a less strong and more variable association with water, which was somewhat unexpected. A number of explanations are plausible here, such as a slow increase in *Sphagnum* matter in the soil retaining water, but regardless of these it appears likely that older restorations have better retention of water than more recent ones. Restorations have had an influence on the developing habitat community through changes in the water regime, but it is subtle – both water and habitat are much more variable than expected, and restoration has not had consistent results.

The most important driver of variation in the vegetation community was herbivory. Many other studies of mires in the New Forest have not considered or described the impact of grazing (Newbould & Gorham, 1956; Newbould, 1960; Clarke and Allen, 1986) with the exceptions of Clarke (1984) and Atkinson (1984), but it clearly plays a huge role in determining the vegetation community. Clarke's 1984 work found grazed mires to be much more diverse than ungrazed mires, with a particular increase in bryophytes. Both Clarke (1984) and Atkinson (1984) state that grazing reduced *Molinia,* which was expected to dominate in the absence of grazing. In contrast, this study has shown that high grazing activity was associated with declines in some mire species, such as *Eriophorum* spp. and *Drosera rotundifolia*, but without any relation with the presence of bryophytes, *Sphagnum* species or *Molinia.* High herbivory was sometimes associated with poor mire communities, and was also correlated with the increases in bulk density of substrate and observed trampling damage, suggesting that soil compaction was taking place. Some studies of grazing in mire communities (Küchler *et al.*, 2009; Worrall *et al.,* 2007b) suggest that grazing can suppress dominant species, but there can also be deleterious effects through elevating nutrient content in the substrate and harming conditions for bog species (particularly mosses) through trampling (Falk *et al.*, 2015; Küchler *et al.*, 2009). Negative impacts of grazing have also been clearly demonstrated for other UK bogs (Wilson *et al.,* 2011). With the high densities of deer and livestock in the New Forest, the effect of herbivory is important research area and will need future study to compare different intensities of herbivory.

Mire structure is also an important driver of community composition and may also affect the success of particular restoration works. This is perhaps an obvious conclusion, but considerable variability in the community and abiotic variables is notable in some structures, but not in others. For example, mires present on slopes (“seepage steps”) are likely to experience a greater fluctuation in the water regime than those in valley bottoms (Clarke & Allen, 1986). It appears that mires in a more typical valley structure have experienced a greater degree of successful restoration than both seepage areas and mire and heath matrices (such mire and heath mixing may be indicative of tussock vegetation dominating in the presence of high grazing and trampling, and is sometimes considered indicative of strong drainage). Differences in the success of restorations based on local conditions have also been described in shallow peatlands in Exmoor National Park (Grand-Clement *et al.,* 2015). The New Forest does feature rich fens in addition to valley mires (Atkinson, 1984), and in these habitats a succession to *Sphagnum* dominated flora is considered undesirable (Kooijman *et al.,* 2016). However, none of the restorations surveyed here were representative of these habitats, as shown by consistently low pH records.

4.4.4 Potential barriers to successful restoration and pathways to successful restoration

There are several areas in which restorations in the New Forest have encountered barriers, and these both reflect and contrast with those found in other locations. Unlike those in many other studies, restorations appear to have had relatively little success in increasing water retention in the mires, which may be attributable to insufficient blocking and potentially the management of surrounding habitats. In upland peat habitats, burning has been found to reduce water tables (Worrall *et al.,* 2007b) and have other deleterious effects (Brown *et al.,* 2014), so the burning of heathland surrounding mires in the New Forest could influence the flow of water into mire watersheds. Controlled burns often occur very close to some mires (personal observation; Clarke 1984). Clarke and Allen (1986) and Newbould and Gorham (1956) assigned importance to sub-surface flow from exterior habitats into mires, which may not be considered by restoration that focusses on drain cuts only. Avoiding the use of burning around restoration areas could possibly impact restoration, but any such changes must be monitored. It is likely necessary that detailed hydrological monitoring is required before and after interventions take place. In addition, the maintenance of past restorations is clearly necessary, particularly where heather bales have been used. Heather bales, while frequent in restorations (Armstrong *et al.,* 2009) have been heavily criticised owing to a lack of durability (Green *et al.,* 2014; Grand-Clement *et al.,* 2015). Alternative, more durable methods, such as grip blocking using plastic or rock dams, may be more effective in this area (Armstrong *et al.,* 2009).

In some habitats, a high abundance of grasses may also be an area of concern for restoration work. *Molinia caerulea* showed relatively high abundance throughout and is considered to be of particular threat to moorland and mire communities (Limpens *et al.,* 2003; Gogo *et al.,* 2011; Marrs *et al.* 2004; Aerts & Berendse, 1988). Hummock species such as *Molinia* are suspected to benefit from drainage compared to those requiring constant water (Heikkilä & Lindholm 1995). Moss cover, including *Sphagnum*, was negatively correlated with graminoid cover and in particular negatively correlated with *Molinia* (figure 4.12), suggesting that areas of increased *Molinia* cover are poor for other important mire species. However, this could be related to spatial distributions in the mires, as Newbould (1960) found *Molinia* dominated and *Sphagnum* dominated vegetation to occupy different areas of Cranesmoor mire in the New Forest. A high frequency of *Molinia* was also reported by Newbould (1960) and Atkinson (1984), so this pattern has been present in New Forest mires for a long time. Quite how high cover of *Molinia* persists in the presence of grazing is something of a mystery, but an interpretation by Atkinson (1984) could be informative, in which he suggests that grazing intensity is influenced more by the density of animals in mire catchments than the suitability of habitat, as livestock (ponies in particular) did not move far from their home range.

Compared to many other studied mires, the New Forest is unique in its high density of large herbivores. As previously noted, grazing intensity could therefore have considerable influence on the outcome of restoration. Herbivory and trampling were associated with poorer mire communities in ordination analysis. However, it is not certain whether this structure is attributable to the herbivory, or whether herbivores are only taking advantage of better access to palatable vegetation in these areas. Wilson *et al.* (2011) demonstrated that drained areas did not improve forage for herbivores, and Clarke (1984) suggested that mires were a more significant grazing resource than lawn environments, so if successful, restoration work may enhance grazing. The impact of grazing on vegetation should be considered on future restoration work, and is a potential area for productive study.

4.4.5 Management Recommendations

This study has identified several areas where management and restoration success could be improved. The most important area is improved water retention; this should be accomplished by reviewing the success of drain blocking for individual mires together with a watershed approach including the surrounding management in restoration, potentially avoiding burning at mire edges. Heather bales in particular should be replaced by alternative methods owing to results in both this and other studies (e.g. Green *et al.,* 2014; Grand-Clement *et al.,* 2015). Monitoring must be carried out regularly to ensure that management goals are being met and that the desired effect of restoration is taking place; this should include at a minimum quantitative records of the plant community and water status (water content, water table depth and surface pools) and should be carried out before interventions begin and at regular periods afterwards. Quantitative monitoring will allow trends in particular species, such as *Molinia*, to be observed, and management can then be tailored to the observations in an adaptive management approach. At the moment, it appears that condition monitoring is too broad for an adaptive approach, and some long term trends in species are uncertain. Finally, although grazing was shown to have a large influence on the vegetation community, there is currently no information to support changes in grazing management. However, grazing processes should be observed and monitored, as this could change in the future.

4.4.6 Conclusion

Restorations have had limited success, with little evidence for the recovery of abiotic conditions and little detectable change in water retention but some limited evidence for positive community change towards more typical mire communities. This is restricted to relatively few species, but some of these, such as *Sphagnum cuspidatum*, *S. papillosum* and *Drosera rotundifolia,* are important mire components. A clear need for enhanced monitoring has emerged from this study, as sites considered to be favourable mires were highly variable and often different from any of the restored areas, but not sufficiently distinguished from degraded ones. A similar pattern was observed in blanket bogs by Wilson *et al.* (2011). Work by Rochefort *et al.* (2013) has shown that different survey techniques can give substantially different measures of abundance for mire plants, a conclusion supported by Cantarello & Newton (2008) in forest communities, so this is an area of concern. Although there are many uncertainties with the chronosequence approach, there is realistically no other way that the breadth of the restorations can be assessed, as a result of the lack of baseline data before restorations commenced. Potential threats to the restorations include a lack of water, possible invasion by *Molinia*, and trampling from large herbivores. These concerns will need to be addressed to ensure a long-term future for restored areas.

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Chapter 5: Condition Assessment, Monitoring and Habitat Suitability of the New Forest Mires

5.1 Introduction

5.1.1 Wetland Monitoring

The protection of wetland habitats and communities has become a conservation priority in recent years, following recognition of their important contributions to global biodiversity and ecosystem functioning (Moreno et al., 2012; Millenium Ecosystem Assessment, 2005). Despite this, many wetland habitats are still under threat, and improvements in conservation practice will be required to conserve these valuable environments and their processes into the future. Adaptive management, where the results of monitoring inform future interventions, is a rapidly developing area of conservation practice (Wilhere, 2002; Conroy *et al.,* 2011; Keith *et al.,* 2011; Rist *et al.,* 2012). However, this process requires extensive and effective monitoring in order to fully understand the implications of different management interventions (Duncan and Wintle, 2008; Conroy *et al.,* 2011; Nichols and Williams, 2011). Many uncertainties still remain about best practice for wetland management (Wheeler *et al.*, 2002), and these must be resolved if a transition towards effective, evidence-based conservation is to be achieved (Salafsky *et al.,* 2002). Despite these needs, monitoring has often been lacking even as wetland conservation becomes more widespread, because conservation practitioners focus on management actions rather than monitoring (Gaston *et al.,* 2006). Where monitoring has taken place, a lack of certainty remains about the effectiveness of some monitoring measurements (Stem *et al.*, 2005; Holl and Cairns, 2002; Lovett *et al.,* 2007).

One area where monitoring takes on great importance is that of habitat restoration. As attempts to restore degraded wetlands have increased in recent years, monitoring and evaluation techniques must also be adapted to these management changes (Holl and Cairns, 2002). Unfortunately, many restoration projects have shown a similar approach to other forms of conservation management, where monitoring has been very limited (Wheeler *et al.,* 2002; Pander & Geist, 2013; Morandi *et al.,* 2014). For example, from 1990 to 2005, $14-15 billion were spent in the US on river restoration projects, but only 10% of projects featured any monitoring, and little of that was useful for examining the restorations in terms of ecological effectiveness (Bernhardt *et al.,* 2005). Similar findings have been reported from Germany from 1994-2011 (Pander & Geist, 2013).

As discussed in the previous chapter, extensive restorations of mire habitats have taken place in the New Forest since 1997. Currently, mire and fen habitats in the New Forest are monitored by Common Standards Monitoring (CSM; often referred to as Condition Assessment). Scientific studies do not form part of the monitoring scheme, but may influence management to a limited extent. CSM is a rapid survey technique, predominantly visually assessed, and is designed to provide national level information on habitat condition, to aid ongoing management and help place targeted management and policy where they are necessary (Jackson and Gaston, 2008; Williams, 2006). CSM is described in detail in Chapter 3. In the New Forest, CSM has been used, in addition to its national function, to identify habitats requiring management focus (Cantarello *et al.,* 2010). The favourable status of sites assessed by CSM is also described as a specific management goal for restoration activity in the New Forest (Wright and Westerhoff, 2001). CSM has also been used as the principal method to investigate mire restoration in the Forest (Smith, 2006a).

Despite the widespread use of CSM, there has been surprisingly little study of its effectiveness in practice. Although CSM has many advantages, particularly at providing a snapshot of the state of the UK's SSSIs (Everett, 2004), there has been criticism of the approach. The categorisation of sites in CSM is considered highly subjective (Jackson and Gaston, 2008; Everett, 2004), and CSM style measurements have compared poorly with more quantitative approaches (Cantarello and Newton, 2008). Additionally, vegetation-only monitoring, as used in CSM, may not detect poor conditions for invertebrates and other species (Davies *et al.,* 2007; Verberk *et al.,* 2010), and is particularly poor regarding fine-scale heterogeneity (Davies *et al.,* 2007; Jackson and Gaston, 2008). These features are particularly important for mire biodiversity value (Verberk *et al.,* 2010; Beadle *et al.,* 2015). Whether CSM can give any indication of abiotic changes, such as an increase in water supply, is uncertain, despite measures in the CSM attributes to address these. The extent to which condition scores actually reflect the ecological viability of sites is also uncertain (Gaston *et al.,* 2006), which is of particular concern if CSM is used to evaluate restoration projects. Other concerns include the limited reporting of detailed information collected during the assessment process (Gaston *et al*., 2006) and the infrequency of assessments carried out under CSM (every 6 years) given the potential for rapid change in restoration outcomes. As a result of these limitations, there is doubt about the effectiveness of CSM in monitoring restoration interventions or providing a basis for adaptive management, especially at the scale of individual protected areas. Testing its effectiveness is therefore necessary to determine whether it should continue to be used for the purposes of monitoring restoration.

5.1.2 New Forest management goals

Monitoring effectiveness depends heavily on the management goals set for ecological sites (Lovett *et al.,* 2007). In the New Forest, management goals for mire restorations are based on several planning documents, most notably the SAC Management Plan (Wright and Westerhoff, 2001) and the New Forest Wetland Management Plan (Smith, 2006a). Restoration work has fallen under several different programs, but the techniques for both intervention and monitoring have remained relatively consistent. From the SAC plan, the conservation objective for New Forest mires is to 'maintain' habitats in favourable condition and 'restore' habitats in unfavourable condition (Wright and Westerhoff, 2001). The condition assessment criteria are identified as the national guidelines (i.e. those by the JNCC), with extra reference to breeding bird populations, for which the SPA Designation is given. Under the current restoration and management programme, the HLS Scheme, the goal of maintaining and restoring habitat to favourable condition is repeated, and the scheme documents also refer to the SAC management plan (HLS Agreement, 2010).

The Life II project aims that are relevant to the restorations include halting erosion and addressing hydrological problems with 580 hectares of mire habitat, as well as establishing a continuous monitoring programme (New Forest Life Partnership, 2001). The Life III project goals include restoration of priority interest features of the New Forest SAC (in accordance with the SAC management plan) and the creation of suitable conditions for habitat recovery (New Forest Life Partnership, 2006). Management goals and aims are generalised, with the project aiming to improve 184 ha of mire habitat to "favourable" or "unfavourable recovering" condition (New Forest Life Partnership, 2006). The New Forest Wetland Management Plan, covering the period 2006-2016, also has general goals, citing condition status, drying, and erosion of the mires as ecological problems that require attention (Smith, 2006a). The expected outcomes of this work are not described beyond "habitat restoration", with no targets given other than carrying out the interventions. The rural pathfinder scheme (Smith, 2006b), another past phase of restoration in this study, follows the approach laid out in the Wetland Management Plan very closely. Such goals may hamper effective monitoring, as general, non-specific aims often lead to "surveillance monitoring" (Nichols & Williams, 2006) rather than monitoring of defined outcomes of management activity. The effectiveness of CSM is of considerable importance in the New Forest owing to its categories being used as the management goals. The use of habitat condition as a *restoration goal* (in particular the use of "unfavourable recovering") is also of concern because of the potential subjective nature of such an assessment.

5.1.3 Study Focus

Because of the limited success recorded in mire restoration projects (Chapter 4), it has become necessary to evaluate the use of CSM in the New Forest and determine whether its limitations prevent it from being used as an effective tool to monitor restoration work. Study of New Forest restorations has been extremely limited, and apart from some hydrological studies (Millington *et al.*, 2007), and the previous chapter in this thesis, has been restricted to CSM-based visual assessment or subjective assessment of the sites (e.g. Cooch & Morris, 2001; Gifford and Heritage 2014a; Gifford and Heritage 2014b). To examine the effectiveness of CSM as a monitoring tool, it must be tested against detailed vegetation abundance and community data, to see if the same patterns are detected.

An additional test is to compare CSM with Habitat Suitability Indices (HSIs) for species of conservation concern. This is aimed at determining whether changes in CSM score reflect changes in the availability of habitat for these species. Such indices are widely used in the United States (Warren *et al.,* 2016), and have seen extensive use in wetland habitats (e.g. Chapman & Howard, 1984; Newsom *et al.,* 1987; Vana-Miller, 1987; McKenzie & Zwank, 1988; Palmer & Cordes, 1988); currently they are only used to monitor Great Crested Newt *Triturus cristatus* habitat in the UK (Williams & Biggs, 2012). HSIs involve identifying important habitat characteristics for the species under study (such as feeding, cover, and breeding requirements), based on scientific literature and expert judgement (Newton, 2007). The quality of habitat is then assessed by using an index, from 0 (very poor quality) to 1 (optimum quality), for each habitat characteristic. The overall habitat quality can then be expressed by combining the suitability indices (SI) of the individual components, typically on a scale of 0–1. HSIs keep the advantages of the CSM approach, such as the ability to conduct a rapid survey (only the habitat characteristics need to be assessed), while providing greater detail, scientific robustness and application to fauna as well as plants. Furthermore, the international designation of many New Forest habitats, in particular wetland communities, is based on species that are appropriate candidates for HSI monitoring, rather than the CSM typically used for national SSSIs. HSIs have also been used to guide restoration projects in the US (Warren *et al.,* 2016) and as such could potentially contribute to the monitoring of restoration projects. Comparison of these scores against CSM will help determine whether the condition approach manages to capture habitat features relevant for some important species, and show whether the two approaches are complementary or not.

The research presented here will offer a critical assessment of the application of CSM in valley mires by comparison to data collected from Chapter 4 and selected Habitat Suitability scores for important species. The aim of this research is to determine whether CSM condition assessment is an effective monitoring system to evaluate wetland restorations in the New Forest, with outcomes from this investigation leading to proposals for improved monitoring for New Forest mires.

This research is designed to answer the following questions:

i. Have New Forest wetland restorations resulted in changes to habitat condition, based on CSM techniques?
ii. Is the CSM approach sufficient to identify whether restoration work has been successful, and is it a suitable tool to monitor the recovery of sites?
iii. How do Habitat Suitability Indices compare with CSM Scores for New Forest mires?

5.2 Method

5.2.1 Site Selection

All mire locations investigated in Chapter 4 were used for this study. A chronosequence of successive restoration programmes was used to investigate changes following restoration work. Restorations included were the Life 2 (1997-2001), Life 3 (2002-2005), Rural Pathfinder Scheme (2006-2009) and HLS scheme (2010 – present). The study therefore shows a continuous pattern of different aged sites from 1997 to the time of the vegetation survey (2013). Additional mires were included, based on proposed future locations for restoration (considered 'degraded' sites) and those in favourable status without any documented restoration work. 10 sites of each type were selected for study, resulting in a total of 60 mires to be surveyed. In order to minimise uncertainty from the use of different, independent locations, sites were selected based on their similararity. Excessively small sites (those less than 0.5 ha), which are relatively frequent, were excluded from the study owing to potential edge effects, and areas of flowing water (fens, streams and rivers) were also excluded.

5.2.2 Condition Scores

Condition scores were calculated for each mire location based on the Common Standards Monitoring Guidance for lowland wetland habitats (JNCC, 2004). Each mire plot consisted of 10 quadrat samples of species abundance. Scores of 1 (conditions met) and 0 (conditions not met) were calculated for each quadrat based on the attributes provided by the JNCC advice. Some of these attributes had multiple conditions which had to be met to score 1; causes of failure in these attributes were examined further. Scores were also summed to give a 'total condition score' indicating how many of the attributes were met. These total scores are representative of the final output from CSM, where a category would be assigned based on meeting all attributes. Some important differences between the way that CSM would be undertaken in practice and that undertaken here must be noted; these were necessary in order to assess the effectiveness of the scores. Quadrats were used here instead of visual assessment as given in the CSM guidelines. This should essentially reduce the subjectivity of the measurements for percentage cover. Scoring was also numerical, with each stop point assessed for condition (where scores of 1 meet the condition requirement, and scores of 0 do not), with plot scores an average of these values. In practice, an overall impression of each attribute would be scored subjectively for the site by an assessor, which could result in highly variable scores. Comparisons of scores in this study with those by Natural England in the most recent (2013) assessment were, unfortunately, not possible, because only the final condition categories were provided (Figure 5.1); this is a frequent problem with CSM also noted by Gaston *et al.* (2006).

The section 'Lowland Fen', and in particular 'Valley Fen' under the JNCC and NVC classification are used for the calculation of condition scores (JNCC, 2004). This section specifically refers to features in the New Forest, and includes a range of wetland types that reflect variability on the sites. Vegetation condition scores were calculated as in CSM based on a series of general attributes in combination with habitat specific attributes. Within New Forest valley mires, several different community types are present, including the M14, M21, M29, M25 and M6 NVC communities (Rodwell *et al.,* 1991). In the following study, the M21 designation was used to calculate condition scores (Table 5.1). This community type was selected in order to standardise the assessment on one vegetation type, and as plots were stratified into this habitat because of its near-ubiquitous presence in New Forest mires. Although the mires are made up of a patchwork of many different habitat types, examining one vegetation type allows the investigation of the CSM technique to be carried out without interference from using several different condition scores for each respective community A description of the M21 community is given in Appendix VI. Finally, comparison was also made between the results of this study and descriptions by Cantarello *et al.* (2010) of New Forest condition assessments, including in mires, to examine differences between what CSM measures have recorded in practice and the investigation into its effectiveness here

A number of habitat characteristics, measured in Chapter 4, were compared with condition scores, to determine how well scores related with variation in environmental conditions measured across the study sites. These included vegetation height, the total surface cover of water, the total cover of *Sphagnum* and the percentage of browsed vegetation in the plots. Surface water and grazing pressure were found to be significant influences on the vegetation community in CCA analysis in Chapter 4 (Figure 4.9), and *Sphagnum* cover and vegetation height are important measures of vegetation structure for mire habitat. These characteristics also varied substantially over the study, with vegetation height ranging from 100-600mm, surface water cover from 0-25%, *Sphagnum* cover from 0-80% and browsed vegetation from 0-60% of plot cover. Measures were correlated against condition and HSI scores to examine whether there was any association present. These characteristics were also examined through further correlation analysis to other measured variables to examine how informative they were as ecological descriptors.

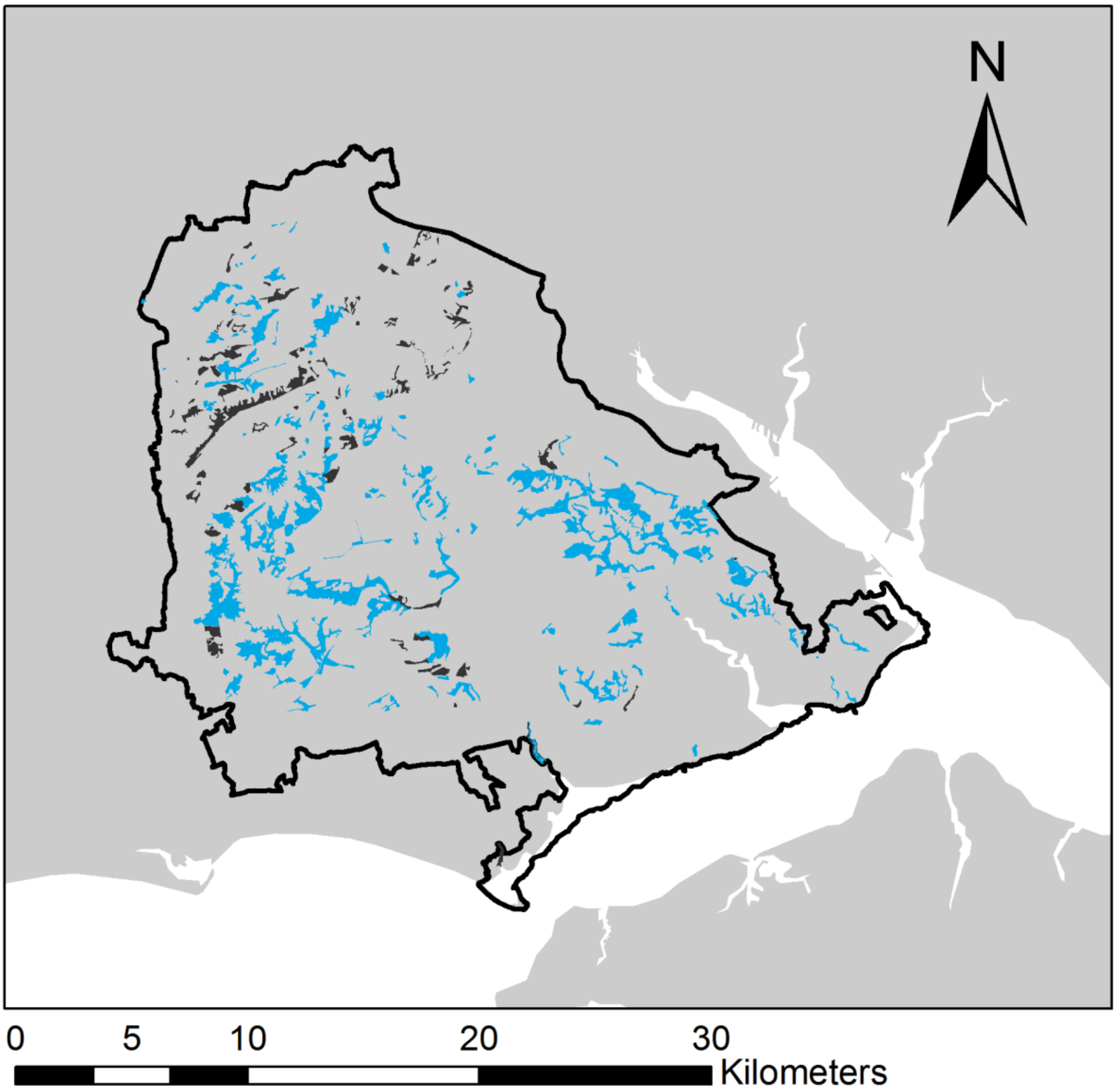

Figure 5.1 The locations of wetland SSSI sub-units in the New Forest, showing those in “Favourable” status (black, *not* including “Unfavourable recovering”) and all other categories (blue) in the 2013 assessment. The black outline indicates the National Park boundary.

Table 5.1 JNCC Scoring system for M21 mire habitats (JNCC, 2004)

Category	Attribute	Scoring
Habitat Extent		Not assessed here, measures change over time using aerial photographs or similar methods
Habitat Composition		
Habitat Structure	Bare Ground	0-10% Cover
	Leaf Litter	0-25% Cover
Vegetation Composition: Positive Indicators	M21 Sphagnum A	Sphagnum papillosum and/or S. magellanicum cover >70% in 80% of samples
	M21 Sphagnum B	Sphagnum denticulatum, S. cuspidatum,, S. fallax, Odontoschisma sphagni: at least one species at least occasional
	M21 Associated Species	Calluna vulgaris, Drosera rotundifolia, Erica tetralix, Eriophorum angustifolium, Molinia caerulea, Narthecium ossifragum, Rhynchospora alba, Vaccinium oxycoccos: at least 3 species constant
Negative Indicators	Invasive Species	*Crassula helmsii, Acorus calamus, Mimulus* spp.*, Impatiens glandulifera, Fallopia japonica, Heracleum antegazzianum* absent or rare
	M21 Negative Species A	*Phragmites australis, Phalaris arundinacea, Glyceria maxima, Epilobium hirsutum, Brachythecium rutabulum, Eurynchium praelongum* Not more than one species, <5% cover
	M21 Negative Species B	*Pteridium aquilinum, Rubus fruticosus, Molinia caerulea,* not more than one species, <5% cover
Negative Indicators - Woody Species	Trees / Shrubs	*Betula, Salix, Rhododendron, Pinus* and other woody species no more than scattered, mainly <1.5m. Cover <10%
	Seedlings / Saplings	Seedlings and saplings of woody species no more than rare
	Woody Species	No species present on flushes and springs, *Salix* acceptable >5m away

5.2.3 Habitat Suitability Indices

In order determine how CSM scores relate to important habitat requirements of mire fauna, it was necessary to produce HSIs from several different taxonomic groups. This was because habitat requirements are likely to show a wide range of variation between different taxonomic groups, and HSIs may therefore show a different response depending on which species are assessed. The following species were selected for the construction of Habitat Suitability Indices: Viviparous lizard *Zootoca vivipara*, Curlew *Numenius arquata,* and large marsh grasshopper *Stethophyma grossum*. These species were selected because they either extensively utilise or are exclusive to mire habitat, they represent different and diverse taxonomic groups, and are protected under national and international legislation. Curlew and viviparous lizards exploit a wide range of habitats (del Hoyo *et al.,* 1996, Edgar *et al*., 2010), but mires are a significant habitat for these species (del Hoyo *et al.,* 1996; Peñalver-Alcázar *et al*., 2016), especially for breeding habitat in the case of Curlew (Tubbs & Tubbs, 1996). Large marsh grasshoppers are restricted entirely to wetland environments (Keller *et al.,* 2012). Individual HSIs were produced following a literature search for the biological requirements of each species. A detailed description of the life requirements of each species is presented in Appendix VII, with their necessary habitat components used to construct three species-habitat models, shown in tables VII.1, VII.2 and VII.3 in the same appendix. HSI scores were subsequently tested with correlations against CSM scores to examine the relationships, and also tested against ecological characteristics of the sites to determine whether they responded to observed changes in these characteristics.

5.2.4 Statistical Analysis

Condition scores were calculated in Microsoft Excel before the data were analysed with R 3.2.3 (R Core Team, 2016). The following package extensions to R were used: multcomp (Hothorn *et al.,* 2008), plyr (Wickham, 2011), reshape2 (Wickham, 2007), and agricolae (de Mendiburu, 2016). Additionally, ggplot2 (Wickham, 2009), gridExtra (Auguie, 2016) and ggrepel (Slowikowski, 2016) were used to construct graphical output. ANOVA tests were used to determine whether condition differed between the age of the plots following restoration and whether restored plots differed from those not subject to restoration work. These tests were performed on overall condition score and for individual attributes which contributed to this score, and also on HSIs for each species. Assumptions were tested for normality (Q-Q plotting of residuals) and homogeneity of variance (Levene's test), and outliers were examined using boxplots. Tukey's HSD post-hoc was performed where ANOVA results were considered worthy of further investigation (p = <0.05). Eta-squared (η^2) effect sizes were used to determine the magnitude of significant test results, a measure that is relatively independent of sample size (Levine & Hullett, 2002). High effect sizes help to determine whether significant results were meaningful, reducing the chance of Type II error (Sullivan & Feinn, 2012). These were calculated using the lsr package in R (Navarro, 2015). Where variables were tested with correlations, Kendall's Tau was used, as this is a widespread and robust test (Croux & Dehon, 2010).

5.3 Results

5.3.1 Have New Forest wetland restorations resulted in changes to habitat condition, based on CSM techniques?

No changes in condition score were detected between different phases of restoration, with both the overall condition score and individual attributes failing to be distinguished statistically (Figure 5.2; Table 5.2). Despite the lack of differences, two attributes clearly stand out with particularly low scores in most of the sites: Sphagnum Attribute A (*Sphagnum papillosum* and/or *Sphagnum magellanicum* >70% in 80% of samples,) and Negative Species B (No more than 1 species, <5% cover of: *Pteridium aquilinum, Rubus fruticosa, Molinia caerulea*) The lack of difference between plots on habitats classified as favourable by Natural England in 2013 and the other plots also demonstrates the difference between the application of CSM in practice and that used here, suggesting that the accuracy of the method is reduced further by subjective assessment. Notably, a substantial proportion of the structural condition attributes relate to the presence of scrub and tree species (i.e. leaf litter, invasive species, tree and scrub cover, seedlings and saplings, and invasive woody plants). The baseline data lacked species that affect these scores to any great extent. This suggests that these scores have limited application in monitoring restoration work, as they address factors that do not appear to be affecting the sites. The two attributes that consistently failed, M21 *Sphagnum* attribute A and M21 Negative Species B, relate to the abundance of two important taxonomic groups. The *Sphagnum* attribute specifically addresses the amount of *Sphagnum* cover, which was frequently too low to meet a favourable score.

Table 5.2 ANOVA test results on total CSM score and attributes

Variable	d.f.	F	*p*
Total Condition Score	5	1.836	0.121
Bare Ground	5	1.863	0.116
Leaf Litter	-	-	N/A*
Invasive Species	-	-	N/A*
Trees / Shrubs	5	0.554	0.735
Seedlings / Saplings	-	-	N/A*
Woody Species	-	-	N/A*
M21 Sphagnum A	5	1.286	0.283
M21 Sphagnum B	5	0.8	0.555
M21 Associated Species	5	1.191	0.326
M21 Negative Species A	5	1.896	0.11
M21 Negative Species B	5	1.463	0.217

*attributes had the same score for all plots

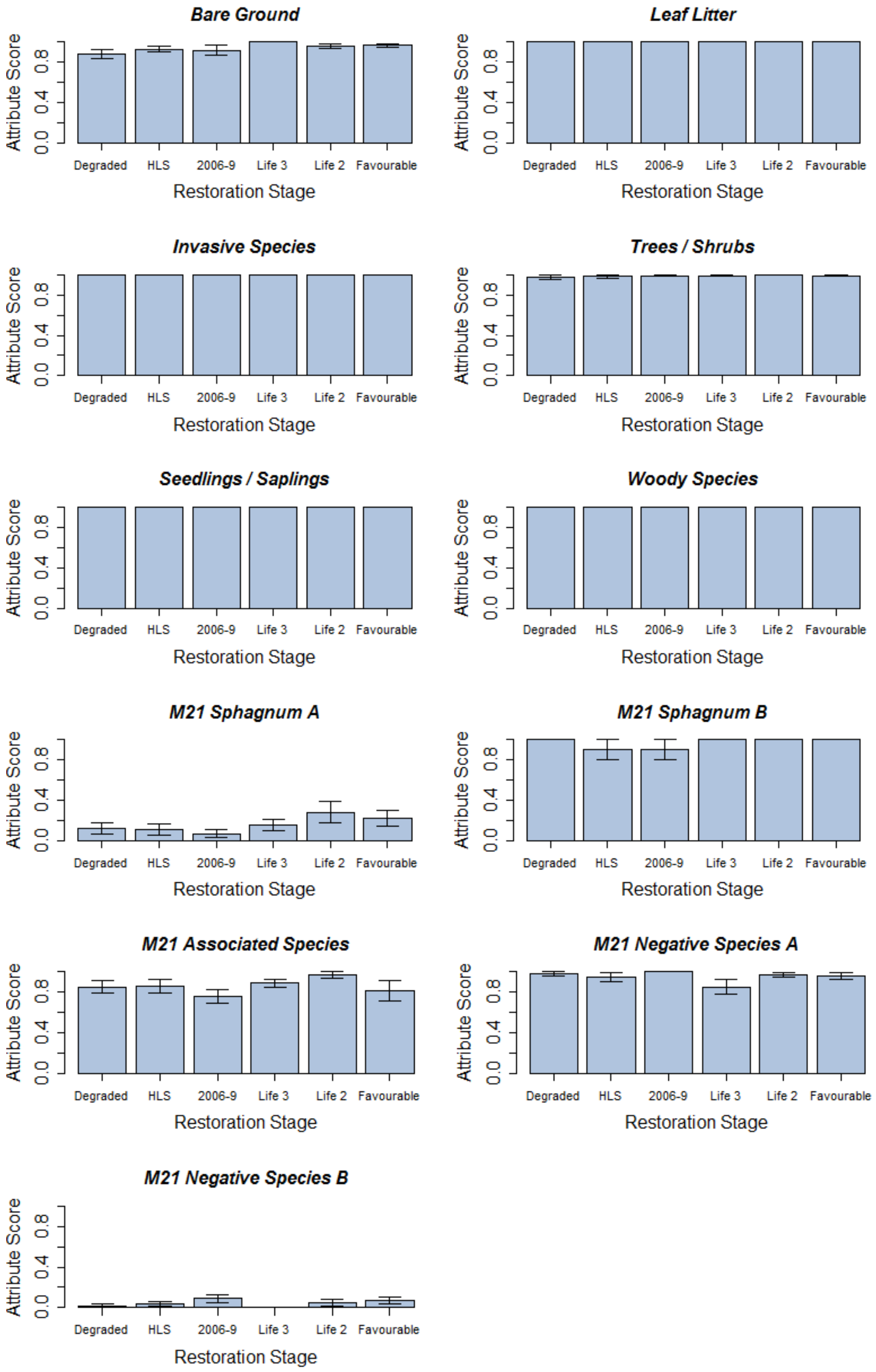

Figure 5.2 Attribute Scores for plots of different restoration ages. None of the attributes showed significant changes over the restoration stage. Error bars represent standard error. Some bars lack error graphs because the data was uniform (scores of 1) for all plots. Only two attributes (M21 *Sphagnum* A and M21 Negative Species B) regularly led to failures.

5.3.2 Can the CSM approach identify changes following restoration?

Condition scores, including individual attributes, frequently showed no change in areas where important component species of those scores were statistically distinguished, including attributes that showed favourable scores and those that showed poor scores (Fig 5.3, Table 5.3). For example, the two *Sphagnum* attributes, for M21 habitats, showed consistently low scores (for the first attribute) and consistently high scores (for the second), despite changes in the presence of several important *Sphagnum* species and in the overall cover of *Sphagnum* (Chapter 4). While the scores as shown may indicate that problems with *Sphagnum* cover have been detected (i.e. it appears to be too low), they do not demonstrate differences that may have ecological importance (i.e. changes in the abundance of individual species), a key criterion for effective restoration monitoring. Where individual attributes had requirements that were far below recorded cover values, very different sites appear similar using the attribute score. This is shown with the condition attribute addressing the cover of *Molina caerulea,* which consistently failed over the sites. This attribute shows the same score for sites that had relatively low cover (10-15%) and those with much higher cover of 30-40%. Valuable ecological data is therefore lost when using this attribute, and any changes at higher cover values will go unrecorded.

Other attributes showed better matches to the underlying data. The associated species attribute reflected the abundances of the component species, with areas of higher abundance often given a higher score (Figure 5.4). *Calluna vulgaris* ($\tau = 0.19$, $p = 0.05$), *Drosera rotundifolia* ($\tau = 0.27$, $p = 0.006$), *Erica tetralix* ($\tau = 0.24$, $p = 0.016$), *Eriophorum angustifolium* ($\tau = 0.23$, $p = 0.021$), *Narthecium ossifragum* ($\tau = 0.30$, $p = 0.003$) and *Rhynchospora alba* ($\tau = 0.45$, $p = 0.001$) all had significant correlations. However, changes in abundance beyond a certain level were simply not detected and associated score may not detect large changes in individual species if the other species remain constant (Figure 5.4). The bare ground attribute showed negative correlations with percentage cover browsed ($\tau = -0.26$, $p = 0.01$) and browsing intensity ($\tau = -0.27$, $p = 0.02$), trampling damage ($\tau = -0.34$, $p = 0.001$) and, as would be expected, bare ground cover itself (correlation = -0.56, $p = 0.001$). Despite these correlations, significant amounts of information appear to be lost with the attribute, with some areas of high trampling and browsing still given a high attribute score (Figure 5.5). The score seemed particularly inefficient at measuring grazing intensity, with the correlations relatively being weak for this measure.

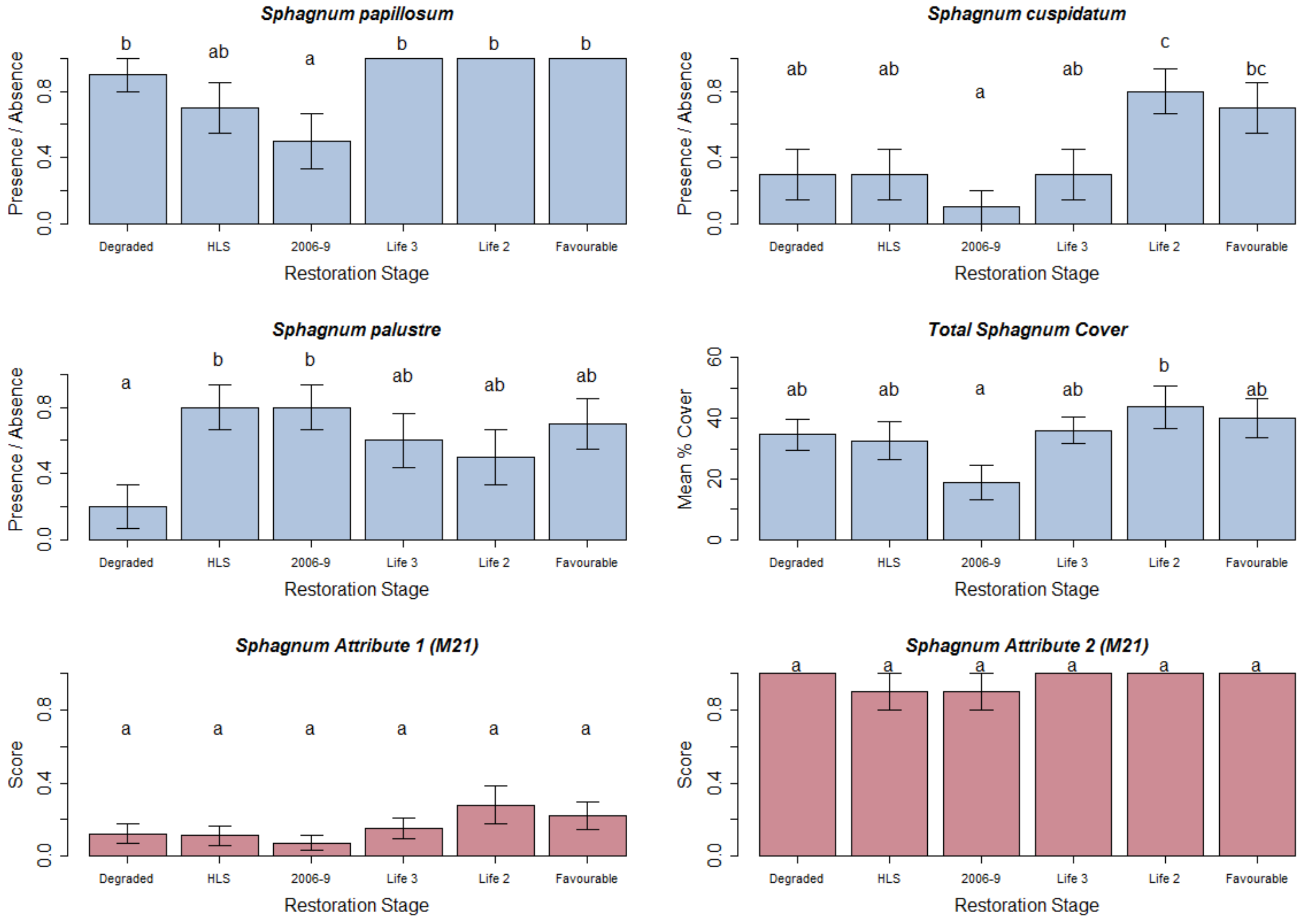

Figure 5.3 Recorded *Sphagnum* cover variables (blue) and *Sphagnum* attributes from the CSM assessment (pink) distinguished by ANOVA analysis. *Sphagnum* attribute scores for the M21 community do not reflect detected changes in distribution for several important *Sphagnum* species, losing valuable information about the sites. Bars with the same letter were not significantly different (p <0.05, Tukey HSD).

Table 5.3 ANOVA test results comparing *Sphagnum* attributes compared to *Sphagnum* cover on restoration groups

Variable	d.f.	F	*p*
Attributes:			
M21 *Sphagnum* A	5	1.286	0.283
M21 *Sphagnum* B	5	0.8	0.555
Cover and Species data:			
Sphagnum papillosum	5	4.22	0.0026*
S. cuspidatum	5	3.65	0.00645*
S. palustre	5	2.38	0.0505
Total *Sphagnum* Cover	5	2.126	0.0762

* significant at $p < 0.05$

Selected habitat characteristics, including some of those shown to influence the vegetation community in CCA ordination (Chapter 4, Figure 4.9), had almost no influence on the final condition scores, emphasising the poor application of CSM to monitoring restoration in mires. Vegetation height (τ = -0.12, p = 0.18), surface water cover (τ = -0.18, p = 0.07), and grazing pressure (τ = -0.01, p = 0.95) were not related to total condition score. This is despite the considerable variation in these environmental measures (100-600mm vegetation height, 0-25% surface water cover, and 0-60% grazing cover). Changes in *Sphagnum* cover, one of the most important components of mire ecosystems, did show influence on the final score (τ = 0.49, p = 0.001), but with the considerable variability in the amounts of cover (0-80%) poorly represented by overall score changes (Figure 5.6). These measures show that the Condition Monitoring process fails to reflect some significant causes of ecological variation present in the study. Correlation analyses performed on these characteristics show that total *Sphagnum* cover was associated with reduced cover of the tussock-forming grasses *Molinia caerulea* (τ = -0.29, p = 0.02) and *Carex rostrata* (τ = -0.28, p = 0.03), in addition to positive associations with the wetland plants *Narthecium ossifragum* (τ = 0.36, p = 0.001), *Drosera rotundifolia* (τ = 0.31, p = 0.02) and the

Sphagnum species that showed high cover: *S. capillifolium* (τ = 0.30, *p* = 0.02), *S.cuspidatum* (τ = 0.32, *p* = 0.01) and *S. papillosum* (τ = 0.39, *p* = 0.001). Vegetation height was associated with several species that are considered detrimental to the M21 community, including *Myrica gale* (τ = 0.28, *p* = 0.03), *Phragmites australis* (τ = 0.28, *p* = 0.03) and *Equisetum fluviatile* (τ = -0.28, *p* = 0.03). *Carex rostrata* (τ = 0.29, *p* = 0.03) was also associated. Surface water cover directly correlated with relatively few variables, but was a significant factor identified by permutation tests following CCA analysis (*F* = 1.5033, *p* = 0.012), as was the percentage of browsed vegetation (*F* = 2.54, *p* = 0.001).

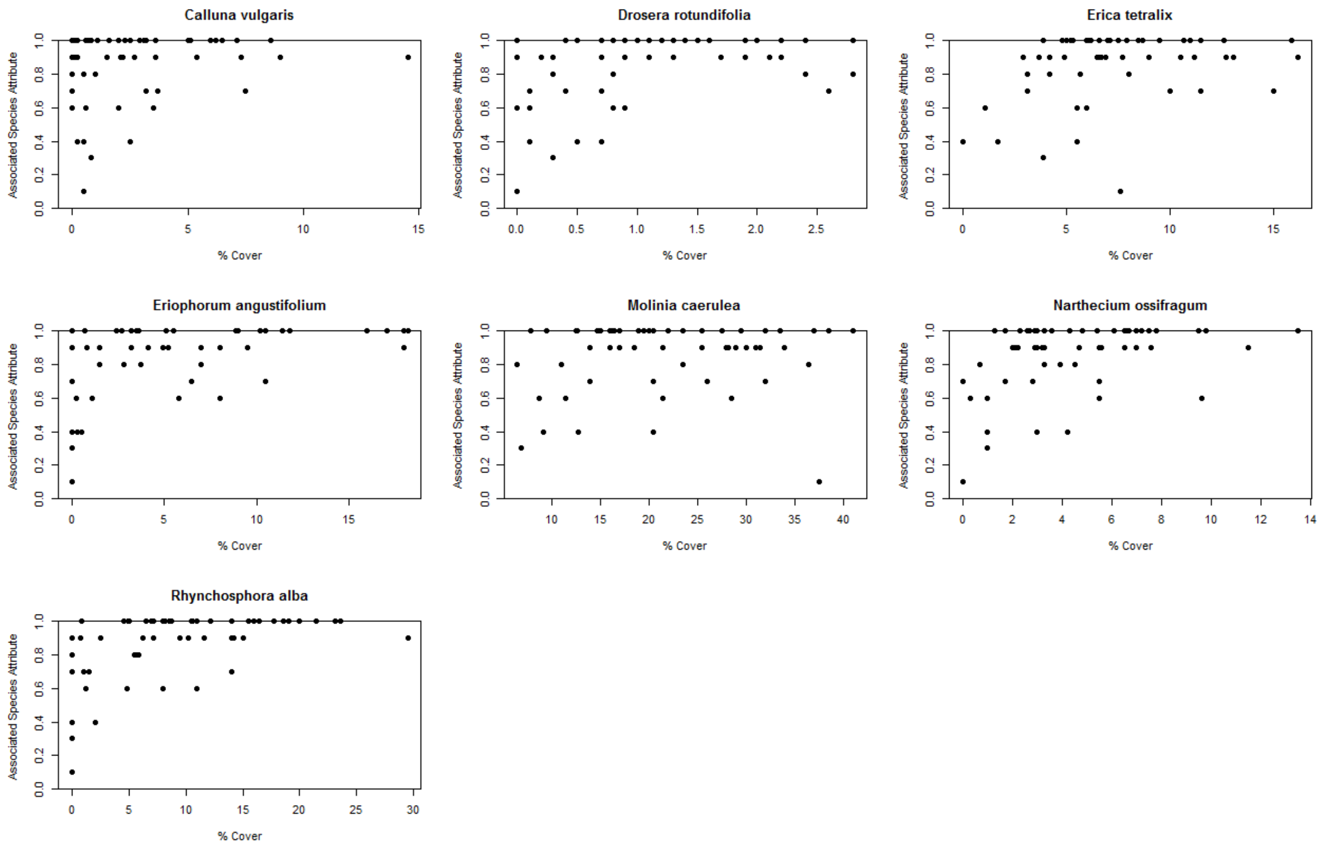

Figure 5.4 Scatter plot showing the Associated Species Attribute plotted against individual component species. Associated species scores show a positive relationship with the abundance of these component species. However, note the range of cover scores for individual species when the maximum attribute score is reached. Considerable variation is present in species cover.

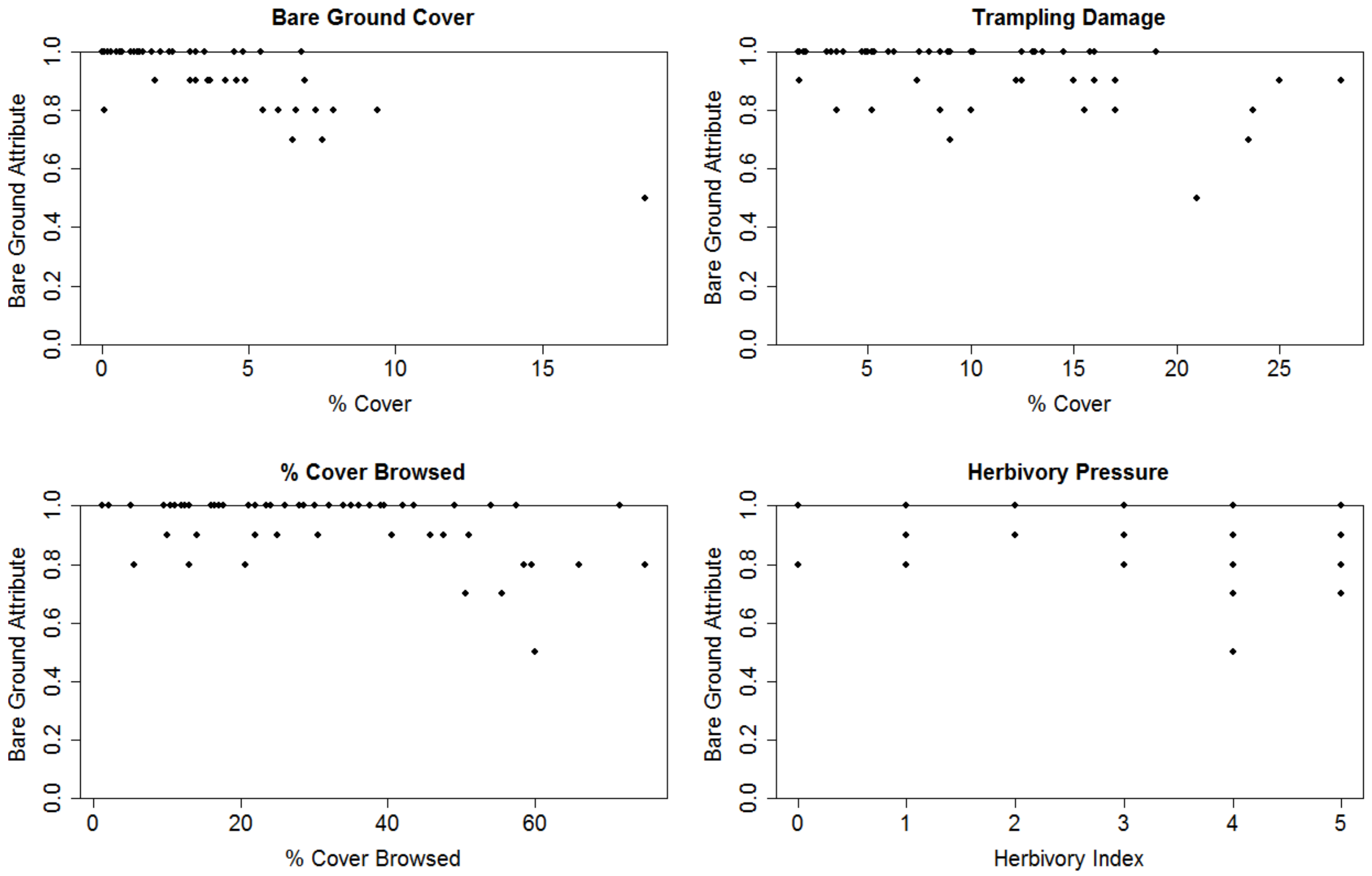

Figure 5.5 Bare ground attribute plotted against ecological characteristics that the attribute is intended to represent. Significant positive correlations were found with Kendall's Tau, $p<0.05$ in all cases.

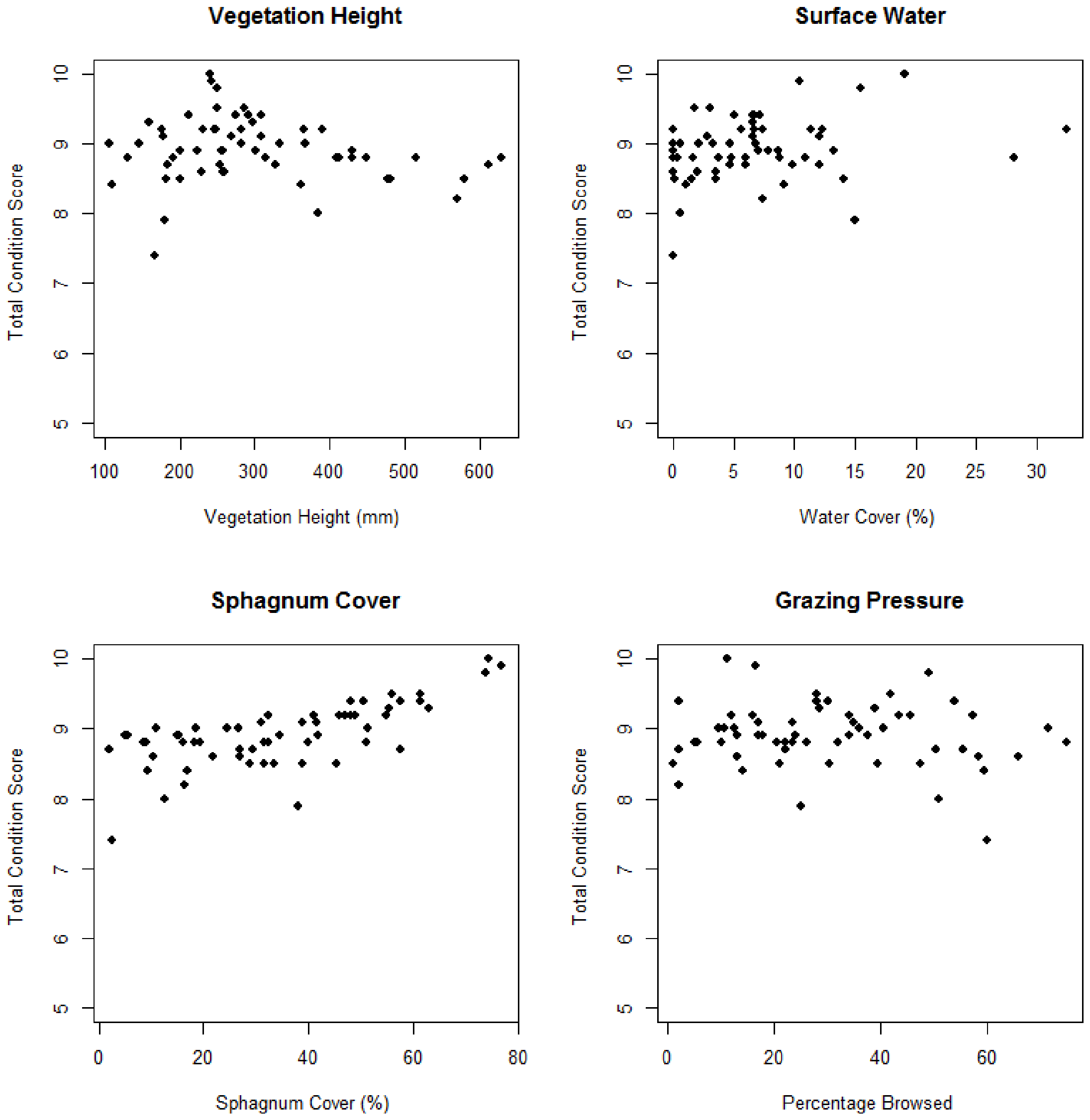

Figure 5.6. Total CSM scores plotted against habitat characteristics. Despite large variations in habitat structure, condition showed few changes over a range of variables that have important ecological significance. Only *Sphagnum* cover was found to influence the total condition score (τ = 0.49, p = 0.001; p>0.05 in all other cases)

5.3.3 How do Habitat Suitability Indices compare with CSM Scores for New Forest mires?

HSIs, calculated for *Zootoca vivipara, Stethophyma grossum* and *Numenius arquata*, showed no correlation with the total condition score for the plots (τ = -0.07, *p* = 0.415; τ = -0.02, *p* = 0.853; and τ = 0.17, *p* = 0.070 respectively), suggesting that the final condition score does not reflect the suitability of habitat for these species. Scatterplots showing these correlations are displayed in Figure 5.7. Generally, the habitat indices were scattered towards the higher end of the suitability scale, with mean values of 0.71 ± 0.02 S.E. for *Z. vivipara*, 0.76 ± 0.02 for *S. grossum* and 0.64 ± 0.02 for *N. arquata.* No differences were detected in index values between the stage of restoration for *N. arquata* (d.f. = 5, *F* = 2.06, *p* = 0.084) or *S. grossum* (d.f. = 5, *F* = 1.83, *p* = 0.126), but *Z. vivipara* did show significant differences (d.f. = 5, *F* = 4.12, *p* = 0.003), where the earliest (HLS, 0.56 ± 0.04) restorations were lower than late stage restorations (Life 3, 0.80 ± 0.06; Life 2, 0.79 ± 0.04); scores are shown in Figure 5.8.

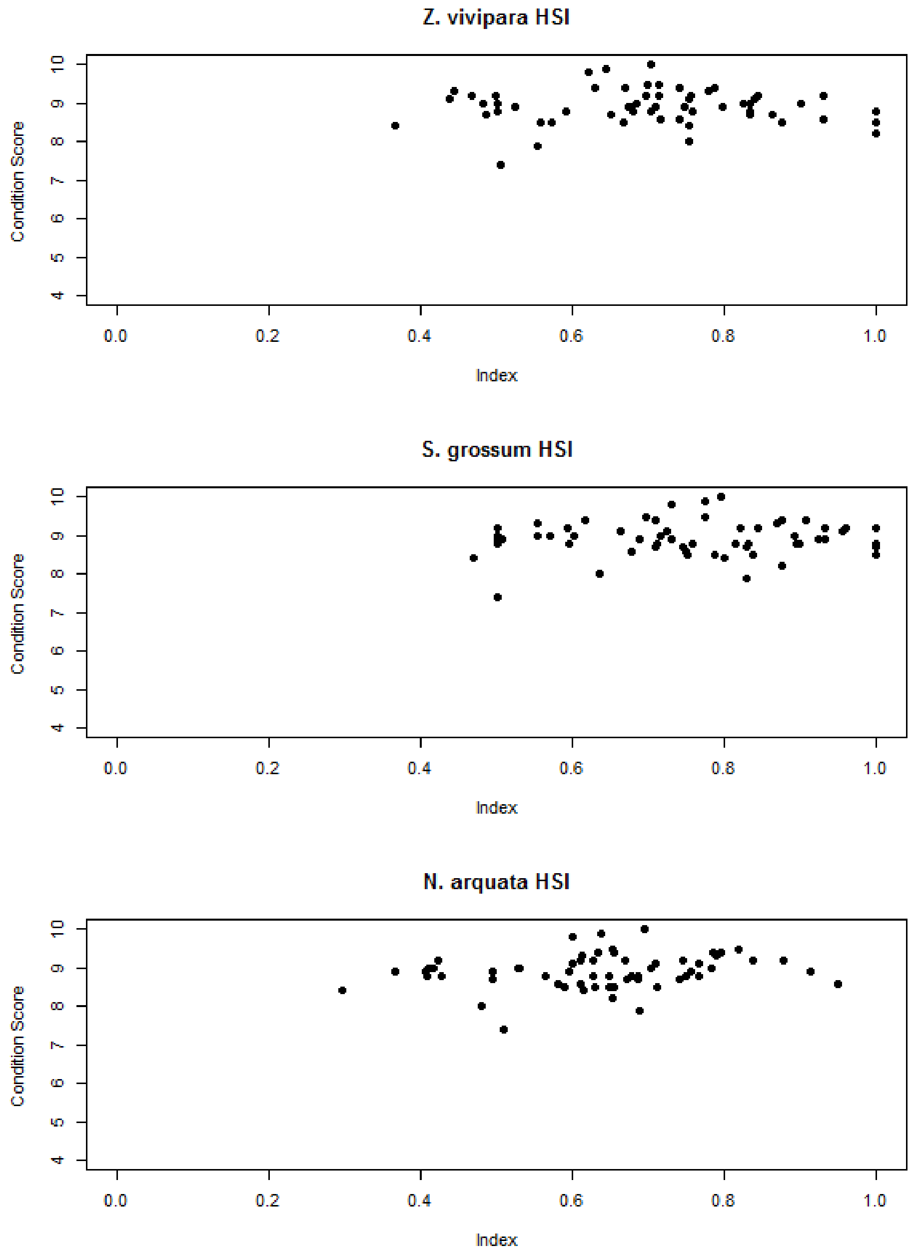

Figure 5.7 Scatterplots showing total condition score plotted against Habitat Suitability Indices for the three selected mire species. A wide range in habitat suitability for these species is shown, but no significant correlations were detected (Kendall's Tau, $p < 0.05$).

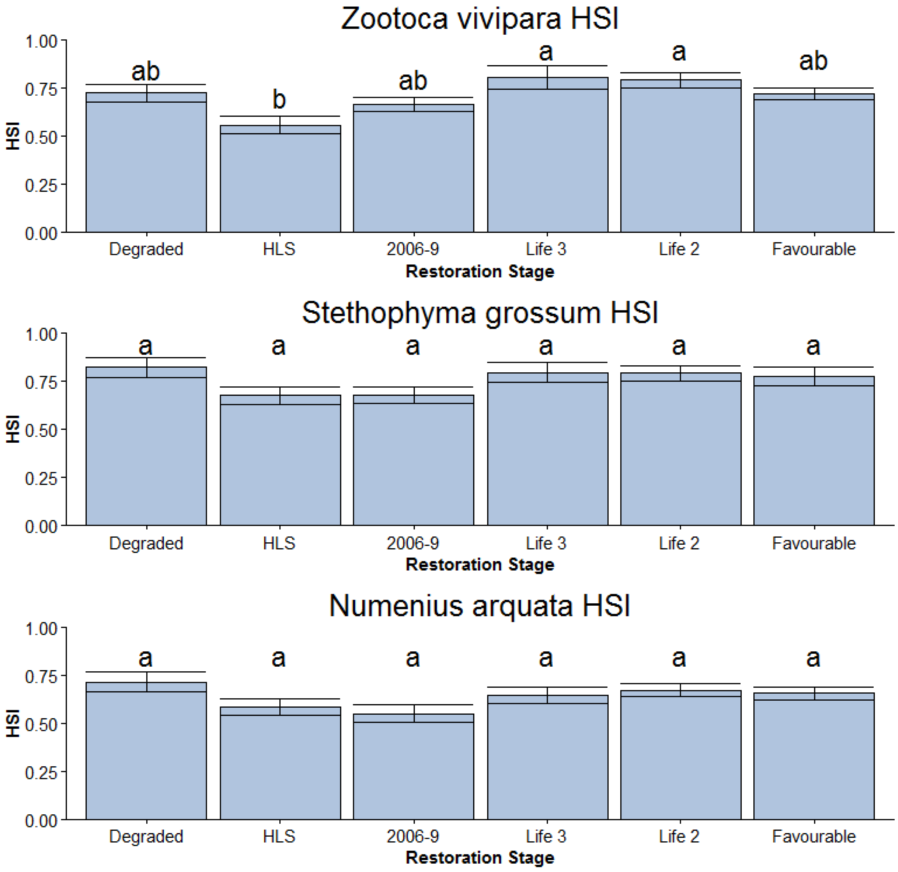

Figure 5.8 Graphs showing differences in HSI scores between different stages of restoration based on ANOVA analysis. HSIs for *N. arquata* and *S. grossum* showed no differences, while *Z. vivipara* showed differences between early and late stage restorations (Bars sharing the same letter were not significantly different, $p < 0.05$, Tukey HSD). Error bars are standard error.

Suitability Indices showed several correlations with environmental gradients recorded throughout the study. *Z. vivipara* index values were positively correlated with vegetation height ($\tau = 0.55$, $p = 0.001$) and negatively correlated with the amount of grazed vegetation ($\tau = -42$, $p = 0.001$). No correlations were present between index values and the cover of *Sphagnum* moss ($\tau = -0.14$, $p = 0.119$) or the surface cover of water ($\tau = 0.16$, $p = 0.073$), although the latter was close to significance. Scatterplots of these correlations can be seen in Figure 5.9. The index for *S. grossum* showed positive correlations with vegetation height and surface water ($\tau = 0.26$, $p = 0.003$ and $\tau = 0.46$, $p = 0.001$ respectively), but not with the cover of *Sphagnum* moss ($\tau = 0.13$, $p = 0.157$). A weakly negative correlation with grazed vegetation was close to significance ($\tau = -0.17$, $p = 0.055$, Figure 5.10). *N. arquata* HSI values were not correlated with vegetation height ($\tau = 0.14$, $p = 0.123$) or grazed vegetation ($\tau = -0.05$, $p = 0.562$, but showed positive correlations with the cover of water ($\tau = 0.31$, $p = 0.001$) and *Sphagnum* spp. (($\tau = 0.27$, $p = 0.003$, Figure 5.11).

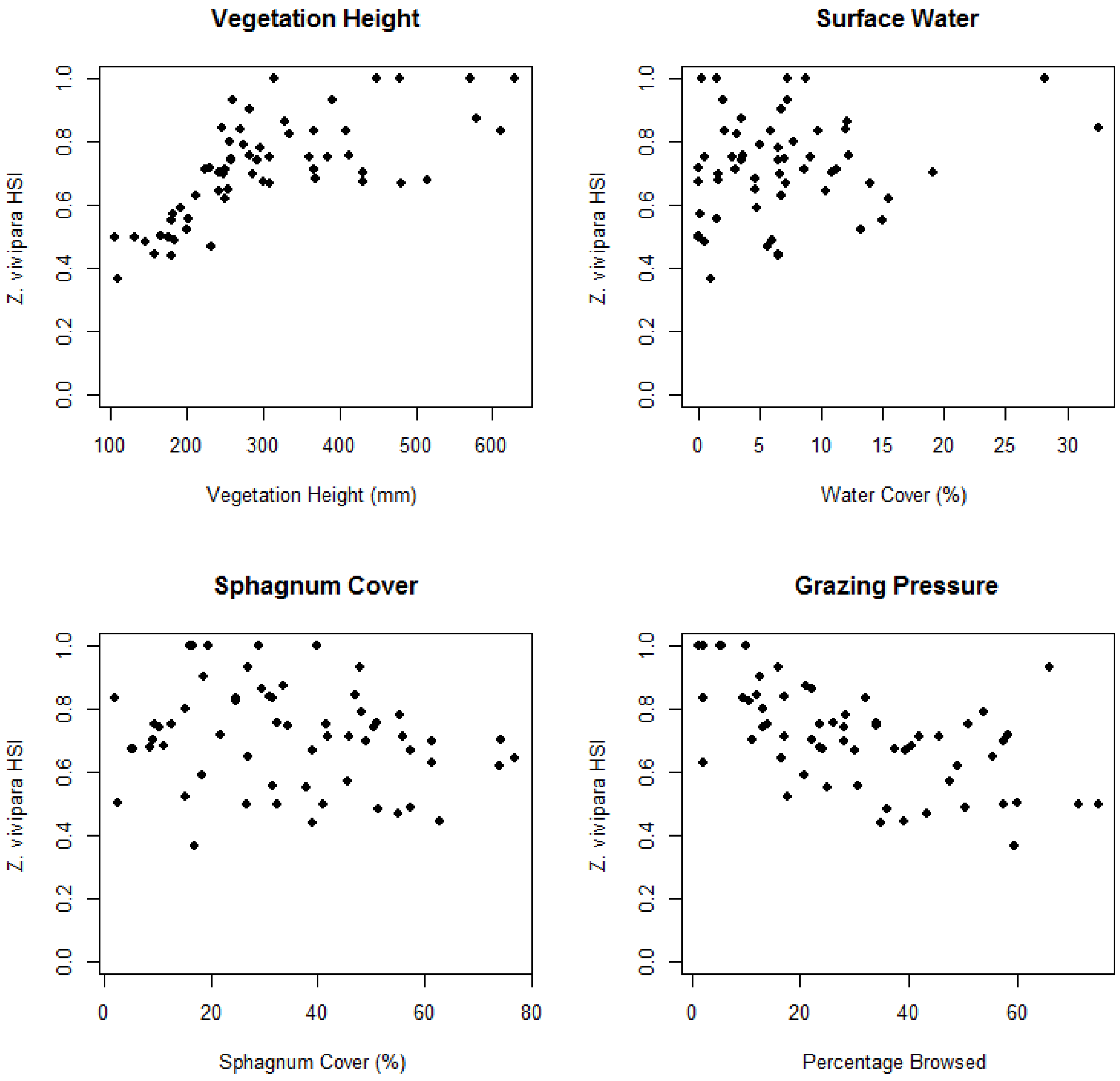

Figure 5.9. Scatterplots showing the HSI for *Z. vivipara* plotted against vegetation gradients recorded throughout the study. Positive correlations were found between the HSI score and vegetation height, negative correlations with grazed vegetation, but not with other variables (Kendall's Tau, $p < 0.05$).

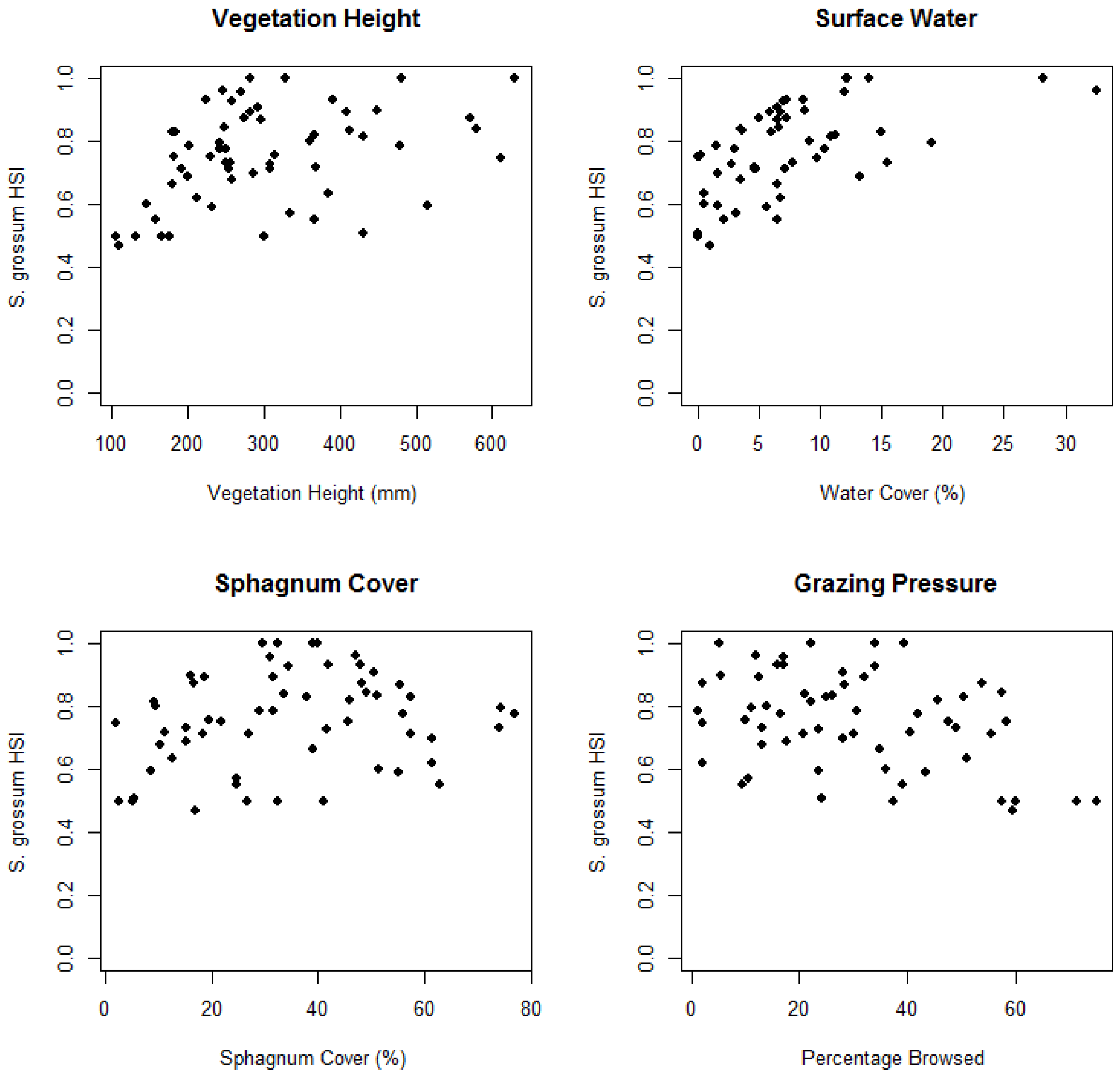

Figure 5.10. Scatterplots showing the HSI for *S. grossum* plotted against vegetation gradients recorded throughout the study. The index for S. grossum showed positive correlations with vegetation height and surface water, but not with the cover of Sphagnum moss or grazing pressure (Kendall's Tau, *p* < 0.05).

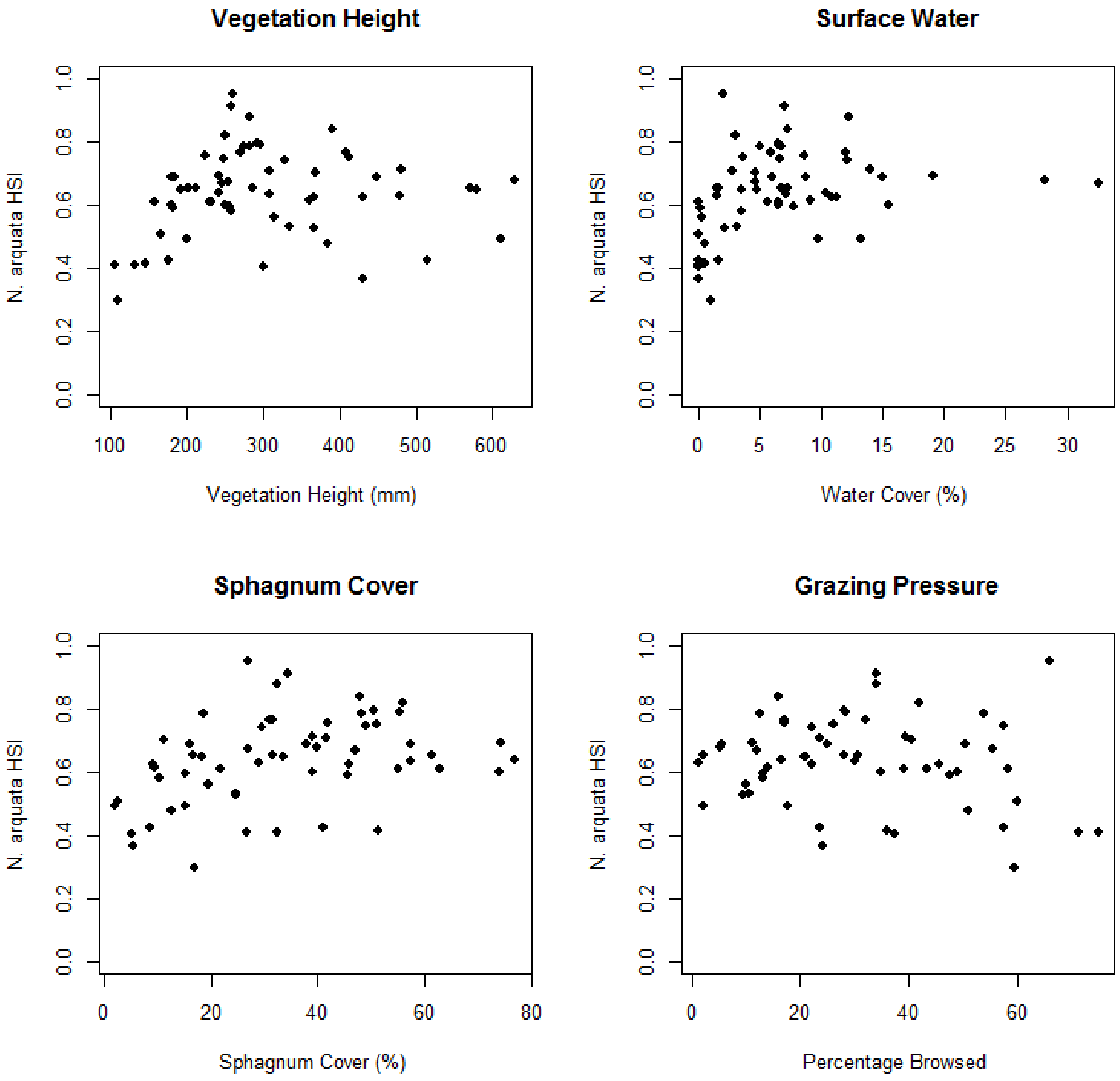

Figure 5.11. Scatterplots showing the HSI for *N. arquata* plotted against vegetation gradients recorded throughout the study. Positive correlations were found with water and *Sphagnum* cover (Kendall's Tau, $p < 0.05$) but not with other variables.

5.4 Discussion

The aim of this research was to examine whether CSM condition assessment was an effective monitoring system to evaluate wetland restorations in the New Forest. Critical evaluation of CSM has been very limited and it has not previously been tested in response to restoration projects such as the wetland work undertaken in the New Forest. Results here demonstrate that condition scores did not show any differences between restorations of different ages, or between restoration plots and areas where restoration had not taken place. More importantly, scores were also not associated with key measures of habitat characteristics, the exception being a correlation with *Sphagnum* cover. Finally, condition scores also showed no associations with Habitat Suitability Indices for selected mire taxa, indicating that habitat suitability for these species could not be predicted when using CSM measures.

5.4.1 Have New Forest wetland restorations resulted in changes to habitat condition, based on CSM techniques?

Despite improvements in condition being cited as the goals of mire restoration in the New Forest, no effect of these restorations was detected using the condition scoring process. This was not particularly surprising, because of the failures of condition scoring previously noted (Chapter 3) and in the poor performance of favourable plots and unfavourable plots as comparators to the restorations (as seen in Chapter 4). None of the plots met all the favourable condition requirements. These results contrast with the results of formal SSSI monitoring in the New Forest, employing CSM, where 97% of wet habitats including mires and heath were reported as ‘favourable’ or, contentiously, ‘unfavourable recovering’ (Cantarello et al. 2010). Poor condition in that study was linked to both the presence of drainage (in ‘unrestored’ areas) and scrub cover. The management goals and condition attributes are quite heavily focused towards scrub and tree encroachment on mires, which may have led to the lack of differences being detected. Because attributes such as ‘trees and scrub’ and ‘invasive species’ are very visible, they are relatively easily addressed by management. However, low scrub cover may then result in sites being considered favourable while other measures of condition, such as *Sphagnum* and associated species measures, receive less attention.

Previous validation tests of CSM for mire sites also focused on scrub clearance (Bealey and Cox, 2004), and reported that condition responded well to this feature. However, in a national study of fen SSSIs (Solly, 2000) no association between favourable condition and scrub clearance was found, despite the clearance being effective at removing tree and scrub encroachment. Bealey and Cox (2004) also highlighted concerns with water availability being insufficient for some species on the sites that they examined. The fact that there is also significant concern about New Forest mires being colonised by trees and scrub, such as Scots Pine *Pinus sylvestris,* already suggests that their surfaces are dry and vulnerable. However, colonisation by willows (*Salix* spp.) could occur in wetter areas.

Unlike previous work examining condition scores on heathland habitats, the individual attributes making up the CSM score also showed no differences over the stage of restoration. This result was somewhat surprising because the attribute scores are much more closely tied to ecological measurements than the total score, which is necessarily an amalgamation of the attributes. Two attribute scores showed consistently poor scores, M21 *Sphagnum* attribute A and M21 Negative Species B. These attributes appear to indicate genuine ecological concerns with the mire restorations: the first score, reporting poor cover of *Sphagnum*, is likely to be directly related to dry conditions on the mires that have not been resolved by restoration work. The M21 Negative species attribute relates to the high overall cover of *Molinia caerulea,* far in excess of the 10% cut-off for favourable condition. Condition guidance specifically refers to undesirable changes caused by this species, which can cause habitat changes from M21 communities to M25 communities while remaining in "favourable" status (JNCC, 2004). There has previously been debate about the cover of this species causing unfavourable condition in several habitats (Ross & Bealey, 2005), with suggestions that its presence should be recorded as a local characteristic. However, negative influences of *Molinia* could then potentially be overlooked by site assessors, with trends in species abundance unrecorded. This is a particular concern, as this species has the capacity to invade and largely modify wetland habitat (Marrs *et al.,* 2004; Gaertner *et al*., 2010).

Many locations examined were close to favourable condition, using the JNCC guidelines, which does not match widespread concern about drainage actions in these locations. Wetland habitats in the New Forest have previously been classed as 'unfavourable recovering' based on the fact that HLS management has taken place (documented by Cantarello *et al.,* 2010). However, the findings in this study have demonstrated that there is no evidence to support such a reclassification, as actual scores and attributes have shown no change following restoration work. This suggests that reclassifications have been based on whether management has taken place, rather than measurements recording the response of habitats to management. This has previously been identified as a potential problem with CSM (Williams, 2006; Gaston *et al.,* 2006). This could have resulted because the timescales for CSM assessment (once over a six-year period) are too infrequent to properly assess the mires, and so assessors assume that changes have occurred without measuring them. This is a serious concern for the use of CSM to assess management, and is thoroughly inappropriate for monitoring, as it propagates anecdotal information instead of gathering useful evidence about management effectiveness.

5.4.2 Is the CSM approach sufficient to identify whether restoration work has been successful, and is it a suitable tool to monitor the recovery of sites?

Tests of condition scores failed to detect changes that had been observed in several species using vegetation abundance data, suggesting that the CSM, in addition to the problems identified above, lacks enough precision to be effective for monitoring. These results support concerns by Bonnett *et al.* (2009) that CSM methods might fail to detect small changes in habitat condition. Additionally, measurements of important environmental variables which influenced the vegetation community (based on CCA analysis in Chapter 4), had no impact on condition scores. Water cover, ranging from 0-30% of plot area, showed no association with the total score, which is particularly concerning for restoration projects that are intended to reduce drainage of water from the habitat. A similar effect was reported for the percentage of browsed vegetation, which was likely the most important influence on the vegetation community. *Sphagnum* cover was a measure of vegetation structure rather than an influence on the vegetation community, and this measure was correlated with total condition score. However, this is not necessarily meaningful for measuring responses to restorations, as different species of *Sphagnum* can have very different habitat requirements (Daniels & Eddy, 1990). As a result, changes in the abundance of individual species could occur in response to environmental changes, while the overall cover stays relatively constant, and CSM measures do not appear to be able to detect this. Such changes could be particularly important where they impact upon ecosystem functioning (Malmer *et al.,* 2003; Robroek *et al.,* 2007). Although one of the attributes specifically examines the peat-building species *Sphagnum papillosum* and *S. magellanicum* (M21 Sphagnum attribute A), these species again have different requirements, with *S. magellanicum* considered relatively intolerant of drainage or burning of its habitat (Daniels & Eddy, 1990).

A study by Jackson and Gaston (2008) attempted to predict the likely condition status for England's SSSI sites based on several environmental and physical parameters, but found that these had very poor predictive power when compared to the actual condition categories, recorded by formal CSM. Jackson and Gaston (2008) could not determine whether this was because of poor relation of CSM output to the actual characteristics of sites, but the results here would support this view, and call into question the widespread use of CSM. A potential reason for the poor associations of overall CSM score to the environmental conditions discussed above could be because of the number of attributes (and for some of the attributes, a large number of individual characteristics) that must be calculated - the combination of which obscures any individual signal from these variables. In practice, as described by Gaston *et al.* (2006), there are also additional areas where subjectivity or different implementations of CSM could further affect the usefulness of these scores, such as how much the features are monitored in detail rather than subject to a general appraisal, how much negative influences affecting sites from outside their boundaries are taken into account, and whether species of concern are neglected where they are not priority features (Gaston *et al.*, 2006). Other concerns include the difficulty of maintaining consistent assessments across considerable space and time, to ensure that comparisons of these assessments are actually meaningful (Gaston *et al.,* 2006). Unfortunately, the results of this study suggest that CSM provides little information about individual species, so changes in the abundance of species could occur while measures of the community (i.e. “Associated Species” attributes) remain ‘favourable’, resulting in inconsistent assessments.

5.4.3 How do Habitat Suitability Indices compare with CSM Scores for New Forest mires?

Reflecting the lack of association with habitat characteristics, condition scores also showed no association with any of the habitat suitability indices calculated, indicating that CSM could not predict areas of suitable habitat for these species. This was despite many of the HSIs reaching high scores in the study, demonstrating the presence of appropriate habitat. In contrast to the condition scores, HSIs calculated here showed stronger associations with habitat gradients than did condition scores, which may point to them having greater ecological significance. This was especially the case for vegetation height, which was important for both *Z. vivipara* and *S. grossum*. Vegetation height was a measure that the CSM scoring system did not examine. Additionally, the variability of vegetation height (maximum recorded height – minimum recorded height at each plot) was an important contributor to HSI scores for all species. The failure to measure these characteristics supports previous criticism of CSM showing that its structural measures are poor (Davies *et al.*, 2007) and highlights concerns by Jofré and Reading (2012) that changes in structural conditions were going unobserved by conservation managers. Interestingly, the HSI for *Z. vivipara* showed differences over the stage of restoration, suggesting that later restorations were more effective at providing habitat for this species. This distribution can be linked to the findings of Chapter 4, where these later restorations appeared to be more typical of valley mire habitat according to JNCC guidelines and NVC classification. The use of Habitat Suitability Indices would require validation and testing before having a practical use

(Brooks, 1997), but the results obtained here are still informative. HSI calculations did suffer from some limitations because the selected species appeared to have relatively broad habitat requirements within the context of wetland communities, but this reflects the visible and protected fauna that utilise mire habitat. Species that have more specialist requirements may be better predictors of habitat quality, but many of these have poorly known biology. Valley mires are known to be important for several invertebrates, including an array of dragonflies, spiders and ants (Tubbs, 2001). Such species may also be likely to correlate poorly to CSM, and perhaps other habitat measures, because of specialist requirements that they may have (Davies *et al.,* 2007).

5.4.4 Limitations

Although a number of limitations apply to the research shown here, the overall conclusion that CSM measures are ineffective as a monitoring practice is well supported. As in Chapter 3, CSM scores were also recorded using a different method to that used in standard practice, and differences between point-based approaches and visual assessment have been recorded (Cantarello & Newton, 2008). However, if anything the steps taken here should have strengthened the ability of CSM to detect changes, because they remove subjective interpretations. The variety of potential implementations of CSM is acknowledged as a potential problem for the consistency of scores (Gaston *et al.,* 2006; Jackson & Gaston, 2007). Some changes that are failed to be detected using the method used here could potentially be found during field assessments, but this relies on the subjective observer, and could easily be missed or go unrecorded. Uncertainty about the scores goes beyond concerns previously raised (Everett, 2004; Williams, 2006; Gaston *et al.,* 2006), because it appears that attribute measurements themselves are imprecise, in addition to the subjective categories of condition.

The ‘control’ sites, in favourable condition and those outlined for future restoration, showed very few differences with other locations, and did not seem to be part of a continuum with plots recovering from restoration. This subsequently made it difficult to test changes in attributes (i.e. tree and shrub cover, bare ground, *Sphagnum* species composition) that may have affected condition score, because these attributes were generally very similar across the study. Favourable definitions were taken from the latest CSM assessment, and proposed sites for future restorations from Forestry Commission proposals. The fact that areas considered to be favourable and unfavourable by managers appeared to have few differences in condition scores and attribute scores suggests that CSM scoring was just as ineffective in practice as it has been in this study.

An additional limitation is that the assessment here only examines M21 communities in valley mires; there are several other communities often present in these habitats that have not been investigated. While these habitats are of interest for future research, in this study is was necessary to examine only one in order to reduce potential variability and biases in the data, a key concern when using a chronosequence approach (Fukami & Wardle, 2005). By keeping the assessment in one habitat, it is easier to make inferences from any changes in cover or scores that were detected. Such changes were subtle where they were found (Chapter 4), highlighting the importance of limiting biases.

5.4.5 Management Recommendations

There is some potential for condition attributes to be improved with the addition of more habitat structural measures (such as vegetation height, variability in height, and water cover) and the more widespread utilisation of attribute scores, which would increase the amount of ecological information available from the assessments. Some attributes (including bare ground and to some degree associated species) are potentially effective and could provide a useful source of information for habitat monitoring, despite the limitations with CSM. However, in practice these attributes are poorly recorded (i.e. they are not entered into any database or recorded as a variable themselves) or made available outside of the assessment process (Natural England, personal communication; Gaston *et al*., 2006), meaning that they cannot be assessed or examined beyond their contribution to the classification of sites into condition categories. Despite the usefulness of such attributes, they are also not a substitute for more widespread quantitative monitoring, which is clearly necessary for restoration work. The nature of CSM in recording against set criteria (i.e. a binary good or bad condition based on a given cover value) could mean that habitat changes outside these criteria go undetected, in addition to other concerns about subjectivity (Jackson & Glaston, 2008). Additionally, problems encountered in this study with the comparison sites suggest a high level of variability in New Forest mires, which hampers detection of restoration processes (Chapter 4). A strong implication from these issues is that monitoring *must* take place before and after restoration work, and preferably continually, to provide unequivocal documentation of ecological changes. Remote sensing is one area where large-scale changes in mires can be observed (Jauhiainen *et al.,* 2007; Langanke *et al.,* 2007), and this technique could alleviate some of the financial and time pressures for monitoring.

While CSM is similar to many other monitoring approaches that measure indicators (Stem *et al.,* 2005), it contrasts poorly with successful evaluating approaches that have stronger connections to the assessment of management goals. Improving monitoring here will require a move towards ‘effectiveness’ measurement (Stem *et al*., 2005), where there is explicit linking between the goals and objectives of management and the measurements of progress towards meeting them. In part, this is the fault of management goals in the New Forest, because favourable condition status cannot provide enough information to ensure that the actual causes of habitat degradation have been addressed. Therefore, restoration goals must be defined clearly and specifically, and clearly document, for example, where the goal is to increase water supply and where other actions are necessary, because concerns about scrub management could obscure other signals. According to the principles of evidence-based management, monitoring must then evaluate the core aim of the work, so in the case of the restoration work here, some form of hydrological monitoring is likely to be required, in nearly all circumstances. In order to form a more successful and adaptive management approach, detailed monitoring of environmental conditions will be necessary for such projects, echoing calls made elsewhere (Duncan and Wintle, 2008; Conroy *et al.,* 2011; Nichols and Williams, 2011) for increased monitoring. Although potentially expensive and time consuming for conservation practitioners, it is the only way to ensure that work is effective and directly explain the cause of any observed changes in the habitat community. One method would be to identify drain locations and then monitor the water table at a series of points around the area both before and after restoration activity. Under an adaptive approach, where continuous information about hydrology and community composition can be continually observed, gradual changes and tailoring of the restorations can then take place at each intervention site to ensure that specific objectives will be met.

5.4.6 Conclusion

This research has shown that Condition Assessment using CSM is an ineffective and inadequate method to monitor restoration projects in wetland habitats. For the first time the effectiveness of CSM in examining restoration work has been tested, and problems with this approach identified. CSM appears to have a poor relationship to variables measured at the plots, including surface water cover, *Sphagnum* cover and the amount of grazing pressure. The use of CSM to monitor restoration, and more widespread use to monitor management, must therefore be questioned, as it cannot provide enough information to properly evaluate the effectiveness of these practices. Together, these results indicate that CSM is not sufficient for monitoring and evaluating restoration projects, because many significant changes in habitat structure and diversity will go unrecorded, and overall scores appear to have limited ecological relevance. To move towards evidence-based and adaptive management in UK protected areas, better monitoring will be required. Such improvements will ensure that future management will be targeted to where it can be most effective and, ultimately, successful.

5.5 References

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Chapter 6: Conclusions and Recommendations

6.1 Conclusions

6.1.1 Original Contribution to Knowledge

Several original contributions to research knowledge were made during this research study. Restoration projects in the New Forest unexpectedly appeared to have had limited success in mires, and while heathland management was outwardly successful, there are indications of some negative effects, described in further detail below. Results from this longer-term study, from both heathland and mires, showed similarities with previous short-term experimental work, suggesting that these studies were effective at predicting long term changes. This study also provides information that contributes to our overall understanding of protected area management, the most important of which is the assessment of monitoring in this protected area. CSM has never been critically examined in practice before, other than some limited validation studies when it was being designed. It consistently failed to reflect environmental conditions in both heathland and mire habitat, and appeared to have especially poor ability to detect subtle patterns in mire habitats. The poor success of CSM will have important implications for the future of monitoring in the New Forest and other protected areas.

6.1.2 Summary of the Main Findings

This research has identified some serious problems with the management approach in the New Forest protected area both with some of the interventions and, critically, with the monitoring techniques used to assess them. In Chapter 2, a comparison of the effectiveness of different heathland management activities found that burning and cutting had significantly different outcomes where they were used. Controlled burning appeared to be the best technique for conservation, maintaining the appropriate heathland structure and species, but there was also evidence for negative impacts on rare species, limited structural variability, and a poor ability to control *Molinia caerulea*. The ability of burning practices to maintain heathland as a *Calluna*-dominated, plagio-climax community has previously been documented in previous studies (Mallik & Girmingham, 1983; Barker *et al.,* 2004; von Plettenburg *et al.,* 2004; Nilsen *et al.,* 2005), matching the findings here. Additionally, the impact on rare species and low diversity in burnt plots reflects some recent evidence for reductions in diversity following repeated burns elsewhere (Velle and Vandvik, 2014). This may be one area of interest for future research. In this study, cutting suffered from a high cover of grasses and an influx of non-heathland species, although these areas also supported some rare heathland species not found in the burnt plots, and it was also likely superior to burning in providing grazing for livestock. Several of the features of cut sites (i.e. high grass cover, low *Calluna*) had also been documented in prior work (Sedlákova & Chytrý, 1999; Marrs & De Luc, 2000; Britton *et al.,* 2000; Barker *et al.,* 2004). In these cases, the medium-term effects of management were similar to those reported from shorter-term studies. Results were, however, different for many of the abiotic conditions from prior studies which showed smaller nutrient reductions from burning (Marcos *et al.,* 2009; Green *et al.,* 2013) and larger ones from cutting (Härdtle *et al.,* 2009). The implementation of management interventions over many cycles might have resulted in some of the differences found compared to previous experimental studies

In Chapter 4, an examination of mire restorations found that there was evidence for only very partial success, with poor recovery of abiotic conditions and only limited evidence for community change towards more typical mire communities. There has been very little previous research on the restoration of lowland mires in the UK, with existing work focused on upland habitats (e.g. Large, 2001; Grand-Clement *et al.,* 2015) or in mainland Europe (Komulainen *et al.,* 1999; Tuittila *et al.,* 2000; Ruseckas *et al.,* 2008; Maanavilia *et al.,* 2015). Here, the results were similar to those studies demonstrating a poor community response to restoration (e.g. Haapalehto *et al.,* 2010) with highly variable conditions (Falk *et al.,* 2015) but contrast with others that showed rapid and dramatic community changes in a relatively short space of time (Maanavilja *et al.*, 2015; Poulin *et al.,* 2013). Successful mire restorations have, however, largely been those associated with less damaged mires with short-term repair rather than those addressing long-term problems (Gorham & Rochefort, 2003). A previous chronosequence study in Japan noted that original vegetation cover had not returned after three decades (Nishamura *et al.,* 2009), reflecting similar time-scales to this study. As with some recent studies (Maanavilja *et al.*, 2015; Jarasius *et al.,* 2015), changes in water conditions could not be confidently stated, and appeared to be lacking. Taking into account the many separate mires, the overall restoration project appears to have made little difference, and some interventions even appear to have failed because of erosive processes. Criticism of the technique used in these restorations have previously been raised (Green *et al.,* 2014; Grand-Clement *et al.,* 2015) and this study provides further evidence for these concerns.

The examination of monitoring in the New Forest was critically important, both for its own management and as a contribution to wider research into protected area management. In Chapter 3, an investigation in heathland habitat found that CSM condition assessment did not demonstrate several differences that were detected in the vegetation community. CSM also showed poor association with the habitat requirements of several important heathland species of conservation concern. The investigation in Chapter 5 showed that CSM assessment was furthermore an ineffective and inadequate method to monitor restoration projects in wetland habitats. CSM scores showed a poor relationship to the vegetation community present in the sites because many small changes in habitat structure and diversity were not recorded, and overall scores appeared to have limited biological significance. The results in both habitats were similar to many other studies suggesting that monitoring has been inadequate in protected areas (Pullin & Knight, 2005; Cook *et al*., 2009; Leverington *et al.,* 2010) and supported predictions that this technique was ineffective (Gaston *et al.,* 2006). Management objectives themselves were often non-specific, as documented in many other areas (Pullin & Knight, 2005; Margoluis *et al*., 2009). CSM did not meet many of the requirements for evidence-based and adaptive management, such as targeting specific outcomes (Lindenmayer *et al.,* 2012) and testing management-orientated hypotheses (Nichols & Williams, 2011).

6.2 Implications for Management

6.2.1 Management of New Forest Moorlands

Findings from this research have important implications for management practice in the New Forest and elsewhere for heathland and mire habitats. Although the assessed management interventions in heathland were superficially effective, particularly burning, evidence has also emerged that there are negative impacts of these practices. Declines or increased scarcity of rare species, as found by the heathland study, have also been found elsewhere, where rare species are negatively affected by management that is aimed at larger structural changes to the habitat (Doults & Doulst, 1995; Severns & Moldenke, 2010; Szinetar & Samu, 2012; Kotowski *et al.,* 2013). This suggests that efforts could be made to make more structurally heterogenous burns, including potentially leaving some areas unburnt, for example, to retain older structures within and alongside recent interventions. In some other areas, such as the Peak Distict, this practice is already widespread. Such recommendations have also been made by Pereoglou *et al.* (2016) to benefit mammals in Australian heathlands, and would improve habitat status for reptiles in UK heaths (Spellerberg & Phelps, 1977; Edgar *et al*., 2010). Outcomes from the heathland study have also supported previous claims that tailoring management to individual locations will enhance effectiveness (Britton *et al.* 2000b; Marrs & Britton, 2000; Vandvik *et al.* 2005; Newton *et al.,* 2009; Velle and Vandvik, 2014), and this is one area where the interventions could be improved. Potential methods could include postponing management if the vegetation community structure has not been colonised by scrub or lost ericaceous cover, or by selecting specific parts of sites to be cut or burnt and leaving others to age.

The most important area where mire restorations could be improved was with the water retention of the sites. Concerns over the effectiveness of individual drain blocking could be addressed by regular inspection or repair of these interventions, to ensure that they are working as intended, as failures in these techniques were observed during the study. Alternatively, results from the study support previous findings (e.g. Green *et al.,* 2014; Grand-Clement *et al.,* 2015) that heather bales could be replaced by alternative methods. One technique that could replace heather bales is clay blocking, which is currently used in river restorations in the New Forest, which is less susceptible to erosion. Another approach could be to give greater attention to processes affecting the whole watershed in valley mire locations. In upland peat habitats, burning has been found to reduce water tables (Worrall *et al.,* 2007) and have other deleterious effects (Brown *et al.,* 2014), so the burning of surrounding heathland could influence the flow of water into the valley mires. Controlled burns often occur very close to some mires (personal observation; Clarke 1988), and Clarke and Allen (1986) and Newbould and Gorham (1956) assigned importance to sub-surface flow from exterior habitats into mires. Land management to aid restoration would potentially involve the cessation of burning on heathland sites that surround the mires. Appraisal of surrounding land management is often missing from CSM assessments (Gaston *et al.,* 2006), a possible reason why this influence has not previously been examined by managers.

6.2.2 Monitoring Protected Areas and their Management

Findings from this research demonstrate that monitoring methods as currently used in the New Forest are inadequate, and clearly require significant revision in order to provide a suitable evidence basis for management. There are a number of ways in which the existing system of CSM could be improved or adapted to provide more information. One way is more extensive use of the attributes, perhaps making them the main focus of monitoring reports rather than using the overall condition categories of "favourable" and "unfavourable". Additional attributes, such as measures of habitat heterogeneity, could help tailor the applicability of CSM to fauna, which appeared to be lacking when compared to HSI scores. However, a new monitoring system, targeted at management, would be considerably more effective than retrospectively trying to make CSM work for a purpose it was not designed. This could then be based on the latest principles for scientific monitoring practice.

A new strategy for monitoring would be specifically adapted to the management approaches, and take place more frequently than CSM. The minimum time for monitoring intervals would likely be annually, because of immediate impacts following management, but it might be possible to increase the time period after the initial stages in order to maximise cost-effectiveness, a key area of concern for protected area governance (Nolte *et al.,* 2010). In these circumstances, care would have to be taken here to ensure that surveys were not biased by any such changes (Lovett *et al.,* 2007). Measures recorded by monitoring should cover the vegetation community, incorporate measures of vertical structure as well as horizontal structure, and, importantly, use quantitative techniques. Such an approach will be more effective and have a better scientific basis than qualitative techniques (Sutherland, 2000; Pullin & Knight, 2005; Margoluis *et al*., 2009; Lindenmayer & Likens, 2009). Monitoring of abiotic conditions could also be useful, especially for mire restorations, although these could potentially be expensive and time consuming additions. Although the measured variables should be specific for the management used, there could also be elements of overall habitat measurements, in order to insulate against unknown future changes (suggestions made by Legge, 2015 and Lindenmayer *et al.,* 2012a). One technique that might be effective for a future monitoring approach is the use of point-quadrats. These are good at reducing observer bias (Sutherland, 2006) and could also be effective for the monitoring of bryophytes, where estimating visual cover would be very difficult (personal observation). A transect or randomly distributed set of points could serve as the basis for these observations; it would be important to remove subjective selection of the points which could take place in CSM measures. Many of these recommendations are similar to those previously proposed under adaptive management principles and previous studies on monitoring effectiveness (e.g. Lovett *et al.,* 2007; Legge, 2015), because CSM appears to be poorly connected with those principles.

6.2.3 Protected Area Management

Previous understanding about the overall impacts of protected area management, and the effectiveness of these areas, has been hindered by a lack of data (Chape *et al.,* 2005; Gaston *et al.,* 2008). There is now growing literature and sources of data about protected area effectiveness to address this knowledge gap. Despite this, remarkably few studies have assessed the effectiveness of management and monitoring in combination with important conservation outcomes, particularly in the UK (Gaston *et al.,* 2006). While assessments of management effectiveness have grown, knowledge of the ecological impacts from management still remains poor (Coad *et al.,* 2015). Because this study has contributed an assessment of monitoring in combination with the ecological outcomes from management, it helps contribute to this area of research. Globally, protected areas are just meeting a ‘basic standard’ (Leverington *et al.,* 2010b), but detail on individual biodiversity species is often lacking (Gaston *et al.,* 2008). Similar results are present in this study, where the continuation of management in heathlands has been effective at maintaining overall structures but may have poor preservation of rare species. A typical weakness of protected areas, which has been found consistently, is effective monitoring (Ervin, 2003; Leverington *et al.,* 2010a; Leverington *et al.,* 2010b; Coad *et al.,* 2015), and the results from this study suggest that this is the area where protection in the New Forest is also at its weakest. Concerns found that CSM relies too much on subjective interpretation is also present for many other European monitoring assessments (Nolte *et al.,* 2010). Gaston *et al.* (2006) suggested that in the UK protected areas, species populations are poorly monitored and analysed, monitoring schemes area not systematically applied, and poorly validated from field data. Results from this study provide evidence strongly supporting those suggestions, reflecting the need to improve conservation practice in these areas.

Despite the differences observed in response to heathland management, and the lack of differences following mire restorations, management techniques have continued with little review for a long time. For both heathland and mires, there is a continuation of traditional and widely used techniques, rather than experimentation around hypothesis testing (Sutherland, 2000). In the New Forest, there are no official mechanisms for management to adapt to new trends or emerging evidence, which could help managers to adapt their approach to new knowledge. Similar concerns are present about protected area management internationally (Geldmann *et al.,* 2013; Pressey *et al.,* 2015) and particularly there are suggestions that there has been little uptake of the principles of adaptive management (Walters, 2007). Inadequate management planning, particularly for facilitating pro-active management, is also considered to be a major issue globally (Leverington *et al.,* 2010b) and in Europe (Nolte *et al.,* 2010). In the New Forest, because goals are set to 'improve condition' without specific pathways (e.g. Wright & Westerhoff, 2001) such concerns also apply here. There are a number of strategies that could be used to enhance conservation management. An important already established strategy is to build an adaptive strategy into the management approach, This would require a greater scientific focus from managers but be very useful in determining how to maximise the conservation value of the techniques used, particularly for heathland burning and cutting. An additional new strategy is for managers to adopt an experimental approach in some areas that would allow new types of management to be considered, such as alternating cutting and burning methods as suggested above, This might help to alleviate some of the negative impacts of both management types; without experimentation, the effects of such techniques will remain unknown.

6.3 Limitations

6.3.1 Methodology

The investigation into both mire and heathland management activities focused on the structure of the vegetation communities and their response to management. The biggest limitation in this study was, therefore, that the mechanisms behind differences in the vegetation communities were poorly understood. Explanatory variables in the heathland study identified 28% of the variation, and in the mire study a relatively low percentage of variance was also explained. This leaves the majority of variation in the communities unaddressed. One area that could be a very significant contributor to this variation is the effects of the management on plant physiology, which could cause changes in the vegetation community by altering competition between species or affecting survivorship for individual plants. Models of succession are often driven by interactions between species, both positive and negative (Davy, 2002), so it is difficult to explain dynamic changes without more detailed knowledge about these interactions. Information on the physiological responses of plants could help to link the observed differences in abiotic conditions with those in the vegetation community. Management could, for example, have had influences on oxygen availability and water stress for plants following wetland restoration work. Identifying these features could help to determine both whether restorations had been successful and also in quantifying the degradation that management aimed to repair.

Sphagnum plants, which have important influences on the function of mire habitats, can cause rapid successional changes in fens to bogs in response to physiological thresholds (Granath *et al.,* 2010), so these responses are particularly important. Additionally, different *Sphagnum* species have been shown in experiments to show different responses to N uptake through physiological differences, leading to altered competitive relationships with other plants (Granath *et al.,* 2012). Such alteration of competitive relationships may well have taken place in this study, but are unknown. In heathland environments, different effects of management on the mycorrhizal community could also have important influences on community composition, particularly for ericaceous plants, which has been documented for turf-cutting (Vergeer *et al.,* 2006). Structural measurements could have been improved in both habitats by measuring more components of vertical vegetation structure as well as horizontal structure. This could take the shape of measures of total vegetation cover in bands, eg 0-10cm, 10-20cm, and so on, which would give indications of the variation in structure that was present (Sutherland, 2006). Although in this study measures of vegetation height could be investigated for the same purpose, they are likely not as precise and more variable.

The statistical methods used here could also be subject to criticism, particularly where repeated measures (i.e. several ANOVA tests) have been used. Repeated measures are often subject to increased Type I error, so there is the potential for false detections to be made (Field, 2013). However, attempts to control this have been made in the study, by the use of effect sizes, which helped to determine whether significant detections were actually meaningful in the magnitude of their response (Levine & Hullett, 2002). Because of this, differences can be stated with reasonable confidence and are insulated against Type I error. Traditional techniques used to control this error, mainly corrections of *p* values using Bonferroni or other types of correction, are vulnerable to increases in Type II error (Field, 2013). Because many of the changes in this study have been quite subtle, an increase in Type II error would likely have a detrimental impact on the ability to detect genuine, ecologically significant results.

6.3.2 Research Design

A chronosequence approach was selected because most previous studies (in both mires and heaths) have examined only short term management changes, whereas chronosequences are effective when studying succession at decadal or longer time-scales (Walker *et al.,* 2010; Wardle *et al*., 2012; Sagarin & Pauchard 2010). Sites were also expected to follow similar trajectories following disturbance, another area where chronosequences are appropriate (Walker *et al.,* 2010). The large number of locations suitable for a space-for-time substitution made the New Forest an (ostensibly) ideal location as it enabled suitable replication of sites of similar ages. In addition, this technique allowed investigation of the practical outcomes of management operations over a wide area of the New Forest, thus providing information on how current management was performing in its goals. However, there are a number of limitations with this approach, many of which revolve around problems in controlling the sources of variability at the sites (Fukami & Wardle, 2005). In this study, variation was quite widespread and limited the number of findings, and confounding factors (i.e. grazing pressure) may have affected the response variables of interest. In order to address these concerns, attempts were made to measure and investigate these factors so that their influence could be determined. However, high variation in the study could itself be an important finding; much previous research focused on individual areas or small scale experiments does not feature such variation - which is potentially a significant influence on management success in practice. An additional limitation is with site history (Fukami & Wardle, 2005). This study relies on the information provided by managers for its records of site history, but there is the potential that unrecorded influences or management that took place prior to records have also influenced the outcome, highlighting the importance of such records and the role of monitoring in maintaining them.

6.3.3 Comparison of CSM and Quantitative Techniques

The comparison of CSM scores and the research study used here could be subject to criticism because of the difference between CSM measures in this study and in practice. CSM scores were also recorded using a different method to that used in standard practice, and differences between point-based approaches and visual assessment have been recorded (Cantarello & Newton, 2008). However, if anything the steps taken here should have strengthened the ability of CSM to detect changes, because they remove subjective interpretations. The variety of potential implementations of CSM is acknowledged as a potential problem for the consistency of scores (Gaston *et al.,* 2006; Jackson & Gaston, 2007). Some changes that failed to be detected using the method used here could potentially be found during field assessments, but this relies on the subjective observer, and could easily be missed or go unrecorded. This study necessarily relied on a more quantitative approach in order to test the effectiveness of the scoring and individual attributes used. An argument could be made to apply the measures over the whole site rather than individual quadrats, but this would also make statistical comparisons difficult and also would likely further reduce any relationship between CSM scores and the environmental conditions present. Future investigation of CSM or other monitoring techniques could assess their impact in practice rather than using the method here, although it would be difficult compare the approaches statistically. One method would be to undertake a CSM standard assessment during field studies, and compare results from that to more quantitative techniques.

6.4 Recommendations for Future Research

6.4.1 The Experimental Approach

The best way for future studies to replicate and build upon these findings is through experimental work. The experimental approach is effective at controlling causes of variance, while the factors of interest can be investigated (Sutherland, 2000). To be properly experimental, the study must have controls, be replicated, randomised and properly monitored (Sutherland, 2000). Before and after studies of management impacts in the New Forest (and in general) would be very useful, and provide greater information on the immediate impacts of these interventions, and reduce concerns that spatial variability was affecting the results. The downside to this approach, at least for the purposes of long-term research, is that it would potentially be expensive and time-consuming to carry out, especially in areas where existing monitoring has been poor and qualitative. Building a more quantitative and scientific monitoring regime would help to provide a future basis for this type of research and could help to address many of the concerns with its implementation, particularly those that require long-term research, in addition to the benefits of such monitoring itself. Field experiments and long-term data collecting are also highly complementary approaches to chronosequences (Fukami & Wardle, 2005), and the range of evidence collected from these techniques would help solid conclusions about management effectiveness to be reached.

How could an experimental approach be undertaken? In order to provide a fully experimental and controlled study, fenced plots would necessarily be required to prevent the confounding effect of grazing. Without this, measures of plant growth and physiology would be both extraordinarily difficult to collect and unlikely to be meaningful, and effects on the community could not be appropriately quantified. Critically, measures of control could be much more effectively built into an experimental approach when compared to chronosequence designs, for example by leaving some areas without management an observing if similar changes take place. For the example of heathland habitats, long-term and immediate impacts could be distinguished by setting up cutting and burning plots closely together, so that local effects are the same for each replicated plot.

6.4.2 Physiological Measures

A range of physiological measures could be recorded by future studies that would help to determine the effects of management on individual species and community interactions. Characteristics such as shoot elongation, leaf production and mean leaf area can give indications of how well individual plants are growing in response to management changes. Measures such as xylem potential pressure (XPP) and leaf water content can be used to assess water stress of vascular plants, one aspect of ecophysiology that could be particularly important for wetland restorations. All of the above techniques have previously been used on heathland habitat (Griffiths *et al.,* 2006), although not in response to management. The regenerative ability of heathers has previously been investigated (e.g. Pywell *et al.,* 1995) but is another aspect where studies might be able to show interesting trends. The techniques used for these species could also be used for other plants, such as in the mire community and elsewhere. As there are concerns about the impact of management activity on rare species, studying the establishment success of these species in response to different management activities could also be very useful.

The physiological response of *Sphagnum* plants could be an effective way to investigate the influence of abiotic conditions present within mire habitats, but requires slightly different techniques used to those for vascular plants. Biomass production, shoot formation and N and P concentrations in plant tissue could be useful measures, as used by Granath *et al.* (2012) to examine N inputs to mires, as well as examining the amount of *Sphagnum* affected by desiccation. With the exception of measuring desiccated *Sphagnum,* this approach would be very difficult outside an experimental context, and would require permanent plots to be set up in the field where individual plants could be assessed. Measuring some detailed physiological characteristics, such as maximum photosynthetic rate, is very difficult in field experiments because of the difficulty in ensuring the accuracy measurements and determining the response of individual plants.

6.4.3 Ecosystem Function

Future work could also investigate the ecosystem functioning of mire and heathland ecosystems in response to management interventions. For example, measuring biomass and aboveground net primary production (ANPP) in plants would give information about the overall system response following management interventions. A core feature of mire wetland sites is that productivity exceeds decomposition, which affects the diversity and type of plants present (Verhoeven *et al*.,1990). If the restorations have affected these rates, there will be impacts on the long-term success of the restorations, which may not be easily detected when only examining the vegetation community. Belowground measures, an area where many studies are weak, could also add useful information (Bardgett & Wardle, 2010). Belowground production and biomass could be measured using root cores and ingrowth tubes in heathland habitat, but more novel approaches would be required for mires. Measurement of mineralisation and decomposition rates could give indications of how management caused differences in the vegetation community. The litterbag method developed by Bocock and Gilbert (1960), is commonly used as a measure of plant litter decomposition due to biotic activity (Beyaert & Fox, 2008; Coleman & Crossley, 1996) and could be a suitable technique in heathland habitats. Soil respiration is another activity belowground that could provide informative evidence, that can be measured effectively with portable infrared (IR) gas analysers (Maher *et al.,* 2010); it is an important contributor to the carbon cycle (Schlesinger & Andrews 2000), and particularly relevant for wetlands because of their high productivity and the high carbon stocks in wetland soils (Raich and Schlesinger 1992; Raich and Tufekcioglu 2000). Therefore it would be interesting to investigate if restoration work has had any significant impact on this process.

For mire locations, further water measures, particularly dip-wells with data loggers (Sutherland, 2000), would have been more informative about the hydrological response to restoration, and the hydrology of mires and their response to management is an area where very extensive research could be conducted. While the measures used in this study (surface water and soil moisture) are informative, they may be subject to variation outside the design of the study. Continuous water table measurements would be superior to these measurements and also give an indication of seasonal variation (Dwire *et al.,* 2006; Faulkner *et al.,* 1989) and the depths at which different ecosystem functions take place.

6.5 Conclusion

During this PhD advances have been made into understanding the effectiveness of practical management techniques in the New Forest, and the effectiveness of this protected area itself in conserving biodiversity. It is hoped that outcomes from this research will influence future management activity in the New Forest and elsewhere, and stimulate additional research into conservation effectiveness in these areas. Both the findings and limitations of this study will be able to give direction to future research into heathland and mire habitat, helping to identify areas where new information would be most effective in enhancing conservation efforts. The most important outcome, and hopefully the area where this study will be most influential, is with the effectiveness of monitoring. Improved monitoring techniques are clearly needed to aid managers in determining the effectiveness of their conservation efforts in the New Forest, and results suggest that similar problems will be found elsewhere where this technique is used. Hopefully, future management cycles will include enhances and better collection of data, which will ensure that the effectiveness of this protected area can be maximised.

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Appendix I: The New Forest - Background

i.1 History

The New Forest has a long history of human settlement that has directly influenced its modern day appearance. Mesolithic sites are often found in association with heathland; it is thought that woodland clearance and shifting cultivation thousands of years ago resulted in large declines in soil fertility (Tubbs, 1968). However, not all of the Forest is situated on impoverished soils, and a continuation of human settlement is known, from the Iron Age to the present day (Tubbs, 1968). The most significant event in the Forest's history was its designation, in about 1079, as a 'Royal Forest' by William I ‘The Conqueror’. Within the Royal Forest, the Crown claimed for itself the rights of exploitation of natural resources, which were enforced by Forest Law and its associated courts of justice (Young, 1979). The primary function of a Royal Forest was the preservation of deer and the vegetation they depended on; fallow deer *Dama dama* were likely introduced to the forest at this time, and winter feeding boosted populations (Tubbs, 1968). People living within the Forest boundary suffered a considerable change in rights (Tubbs, 1968; 2001). Before afforestation (the act of creating a Royal Forest), common land would have been freely grazed (Tubbs, 1968). Except for some months of the year, cattle would be allowed to graze in the open forest; pigs would be sent out for pannage (browsing of acorns and beech mast) in the autumn, turves and peat cut for fuel, bracken cut for animal bedding and trees coppiced and pollarded for fuel or timber (Tubbs, 1968). Under Forest Law these activities were given strict limits, and felling for timber, cultivation and killing of game were expressly forbidden. However, over time, forest offences were increasingly settled through fines or license payments, and the forest became a source of income for the Crown (Tubbs, 2001). These licences, together with common practices that were always permitted, became formalised as the rights of common, which eventually became statute in the late 17th Century (Tubbs, 2001).

In later times the focus of the Crown moved towards timber exploitation. From the late 15th Century onwards, many trees (principally oak) were felled to provide timber for shipbuilding. Extensive broad-leaf plantations were also enclosed and planted at the same time. Repeated attempts were made to increase the land available for silviculture, with the commoners representing an opposing interest in wanting to keep land open for livestock. The New Forest Deer Removal Act 1851 marked a culmination in this change in focus from deer to timber (Tubbs, 1968). Masked as a plan to increase the quality of grazing for commoners, it created further timber Inclosures as 'compensation' to the Crown. Many Royal Forests and lowland commons across the country were broken apart at this time (Chatters, 2006); the Crown actively tried to sell areas of land in order to generate income

(Tubbs, 2001). In the New Forest, thanks to the commoners, other interest groups and some estate owners, such attempts were unsuccessful. The perambulation (the area of the extent of Forest Law) remained largely intact from the earliest records in 1217-18 up until 1964 (Tubbs, 2001). The New Forest Act of 1877, and subsequent Acts in 1949, 1964 and 1970 have provided a framework for discussion and debate between the interests of the commoners and the Crown (Chatters, 2006). However, a further significant impact came in the years following the establishment of the Forestry Commission (in 1923), established to manage Crown lands for forestry. Much land was converted to conifer plantations, with significant impacts on biodiversity. A secret plan to convert the majority of the New Forest's woodlands to plantations caused a Government intervention in the 1970s, resulting in a change to the FC's charter to include conservation (Tubbs, 2001). The Forestry Commission is now actively involved in, and responsible for, the management of conservation efforts in the Forest. In recent times, a number of designations have been added to the Forest, as awareness of the importance of biodiversity value increased in the UK. The most recent designation was given in 2005, when the New Forest was made the first National Park in almost 50 years (Chatters, 2006). The New Forest's recent history means that management currently balances a suite of interests, from conservation concerns to the maintenance of traditional practices such as common pasture, while supporting recreation and resisting outside pressure from development.

i.2 Biodiversity and ecology

The Forest has often been broadly considered as two different parts: The Open Forest (consisting of pasture woodlands, heathland, mires and the network of streams, ponds and rivers) and the Inclosures (areas of land enclosed by the Crown for timber production, containing some remnants of native woodland but largely coniferous or sometimes older deciduous plantations). Chatters (2006) described the Open Forest as permeable, and 'not a single unbroken block of land' but a patchy network, extending between fields, villages and fenced woodlands. Both Chatters (2006) and Tubbs (2001) agree that the landscape is shaped by both extensive grazing of livestock (and deer) and its complex and varied geology. The influence of different soil types is likely to be particularly important. Chatters (2006) also observed that habitats outside the forest boundaries play a key role in maintaining those within, such as the Avon valley.

The pasture forest and old growth forests are particularly important sources of biodiversity in the New Forest. Old growth trees supply spaces for nesting birds and bats in their structures – the woodlands have important populations of breeding birds such as wood warbler *Phylloscopus sibilatrix* (Tubbs, 1968), and the presence of the internationally rare Bechstein's bat *Myotis bechsteinii* and Barbastelle bats *Barbastellus barbastella* (Mainstone, 2010). Plentiful dead wood, both standing and fallen, supports a range of saproxylic insects – 55% of species found in the UK are present (Alexander, 2010). One invertebrate species is particularly noteworthy: the New Forest Cicada *Cicadetta montana*, the only species of Cicadidae in Britain. The New Forest is thought to be the only location in the country where this species is still thought to occur, although sightings are extremely rare (Pinchen & Ward, 2010). Since many of the woodlands have been present for so long, fungi and lichens are found in a diversity rarely matched in Europe (Tubbs, 2001). There are some 421 taxa of lichen present in the forest, many of which are dependent on the old growth woodland (Sanderson, 2010). Communities of fungi are highly diverse, owing to adaptations to the long continuity of forest cover and the presence of many ancient trees (Newton, 2010b). Browsing by livestock has contributed to the structure and composition of the forest; by opening the under-storey they provide additional light for lichens, and grazing tolerant plants survive in abundances rarely found elsewhere (Tubbs, 2001). Nationally rare vascular plants, such as slender cotton grass *Eriophorum gracile* and wild gladiolus *Gladiolus illyricus*, are present (Tubbs 2001). Where the woodlands meet extensive groundwater and river systems, particularly rare habitats of bog and riparian woodland occur (Wright & Westerhoff, 2001).

Inclosure woodlands also have a role to play in supporting biodiversity. Tubbs (2001) describes the creation of the Inclosures, and the associated reduction in grazing pressure, creating an abundance of butterflies. However, there have been significant declines in recent times. This is thought by Wright and Westerhoff (2001) to be caused by silvicultural operations and the incursion of livestock into the Inclosures; Tubbs (2001) attributes some long-term declines to replacement of broadleaved trees with coniferous species. Coniferous trees do, however, support a different bird fauna: Crossbills *Loxia curvirostra* in particular require them (Tubbs, 2001) and raptors frequently use them as nest sites (Page, 2010). Many small mammals are relatively uncommon across the forest; this is likely to be a response to the lack of ground cover in the Open Forest as a result of high grazing pressure (Tubbs, 2001). Inclosures could have a role to play in the persistence of these species, as they can provide shelter from grazing pressure in 'core' areas (Tubbs, 2001).

Heathland in the forest is present in a wide continuum of habitats, often encompassing a gradient from dry heath to wet heath to mire or bog. The undulating landscape and river valleys help to maintain this diversity. A particular feature of the forest is the wide band of humid heath, often absent in heathland communities, including Cross-leaved heath *Erica* *tetralix* and the moss *Leucobryum glaucum* (Wright & Westerhoff, 2001). The New Forest features the highest density of valley mires in the UK, a habitat rare across western Europe (Tubbs, 2001). Fens and mires supply habitat to a range of rare species, such as bog asphodel *Narthecium ossifragum* and white beak sedge *Rhynchospora alba*. Fens and mires are also distinguished along soil types and nutrient availability, with different species in these habitats. Tubbs (2001) suggests that the sequence and diversity of plant communities in the mires are 'more significant' than the presence of rare individual species. Grasslands in the forest are differentiated by soil fertility and moisture retention, as well as differences between acid soils and those that are more neutral. Many rare, grazing tolerant species are found within the grasslands, such as mossy stonecrop *Crassula tillaea* and smooth cat's ear *Hypochaeris glabra*. Acidic, moist grassland features a different community, with species such as heath lobelia *Lobelia urens* and pale dog violet *Viola lactea* (Wright & Westerhoff, 2001).

The heathlands support a rich bird community, with internationally important breeding populations of Dartford warbler *Sylvia undata*, Nightjar *Caprimulgus europaeus* and Woodlark *Lullula arborea* (Conway *et al*., 2010) as well as overwintering Hen harrier *Circus cyaneus* (Tubbs, 2001). The valley mires are particularly important habitats for breeding lapwings *Vanellus vanellus*, snipe *Gallinago gallinago*, curlews *Numenius arquata* and redshanks *Tringa totanus* (Conway *et al.,* 2010). All UK native reptiles are present on the heath (Noble, 2010), including smooth snake and sand lizards, which exclusively use the habitat (Tubbs, 2001). Invertebrate communities are also diverse; 27 damsel and dragonfly species occur, alongside 47 butterfly and moth species found on the heath (Wright & Westerhoff, 2001) 109 beetle species of conservation concern rely on the open wetlands for survival, with others specialising in the dung left by livestock (Wright & Westerhoff, 2001).

The network of ponds and streams supplies habitat to all but one native amphibian species, the Natterjack toad *Epidalea calamita*, which was present in the past (Noble, 2010). Despite the streams importance, they have been relatively little-studied (Landford *et al.,* 2010). They are, however, known to feature a diverse community of macro-invertebrates and fish. Once the target of drainage works in the forest, there has been considerable restoration work undertaken in recent years, although the impact on the wider forest environment has not been investigated. The New Forest contains over 570 ponds, 75% of which are temporary (Ewald et al. 2010). The temporary ponds are particularly heterogeneous, and in addition to the species that they directly support, increase the diversity of the forest's habitats as a whole (Ewald *et al.,* 2010). Two extremely rare invertebrates, tadpole shrimp *Triops cancriformis* and fairy shrimp *Chirocephalus diaphanus*, are also found in these ponds (Ewald *et al*. 2010). They both require a grazed heathland matrix with temporary ponds for survival, indicating the importance of connections between habitats (Wright & Westerhoff, 2001).

i.3 Socio-ecological system

People have played an important role in the evolution of the Forest, and are a valid and important component of a 'social-ecological' system that ensures the forests future. Former traditional practices revolved around five rights of common: those of pasture, mast, turbary, estovers and marl (Newton, 2011). Pasture is the grazing of livestock on common land, mast the use of pigs in autumn (pannage) to consume acorns and beech mast, turbary the cutting of peat for fuel, estovers the collection of timber for firewood and marl the removal of clay for building work. Modern practices are rather limited in comparison, and primarily involve the depasture of stock (principally New Forest ponies and cattle) on the Open Forest. Pannage still occurs, although in fluctuating numbers. Much of the traditional cutting of gorse and heather has been replaced by modern management, which uses different techniques. The *Verderer's Court* oversee the system of commons, together with the *Agisters*, who monitor animal welfare and commoning activities. The Verderers include five appointed and five elected members, so have a significant role to play in finding common ground across the forest. In addition, they have the power to restrict development. The perambulation, once considered the extent of the forest, is now the limit of the Verderer's authority. Commoners pay a fee for animals put out to common, but uniquely, there is no limit on the number of livestock allowed; instead the rights are linked to property. About 550 commoners depastured 6000-7400 livestock in recent years (Newton, 2010a).

In many ways, it is no longer economical to maintain the traditional commoner's lifestyle. Earnings are low, house prices have increased significantly, and small holding farming is now very rare. Commoning is now continued mainly as a family tradition (Newton, 2010c). For this reason, agri-environment schemes have been used (currently the HLS scheme) by means of subsidising the commoners for their livestock. It should also be noted that the Forest is surrounded by intense development pressure. In addition to the urban settlements east and west of the forest, two airports, direct motorway access, an extensive road and rail network, together with local ferries and marinas place significant demands on the Park (Chatters, 2006). Within the Park, house prices have risen astronomically, giving huge potential demand for development.

i.4 References

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Appendix II – Supplementary statistics for Chapter 2

Tukey HSD results and Levene's Test results (homogeneity of variance assumption) are provided in the digital supplement to this thesis (Chapter 2 supplementary materials)

ii.1 Correlation Matrix

Please see the digital supplement for detail on the bootstrapping results and individual correlation coefficients, in file "heath.correlation.xlsx"

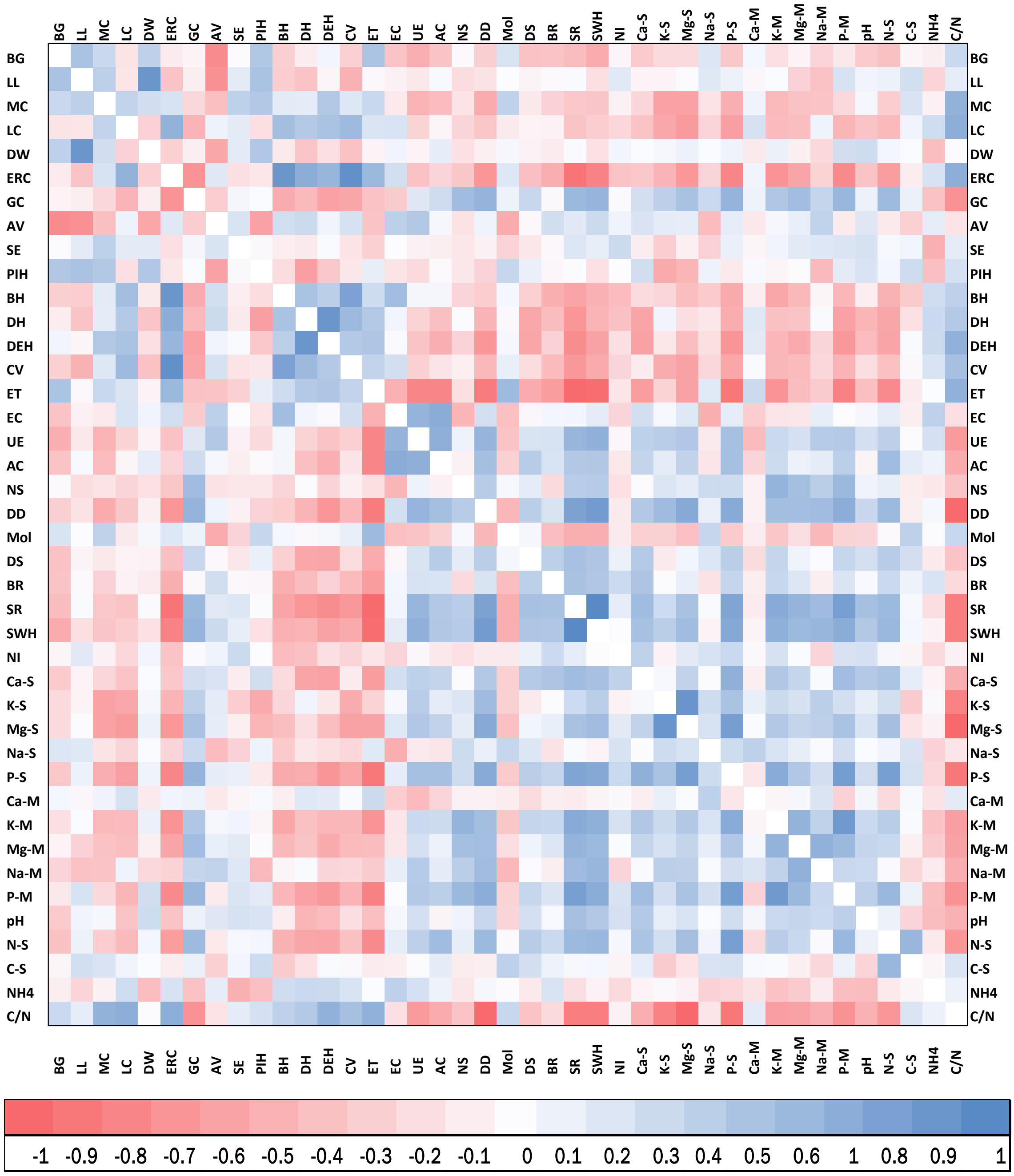

Figure II.1 Correlation matrix of selected heathland variables, based on bootstrapped Spearman's Correlation coefficients. Abbreviations are given in table II.1.

Table II.2 Key for abbreviations used in correlation matrix

BG	% Cover of Bare Ground	DS	% Deschampsia setacea cover
LL	% Leaf Litter	BR	% Bracken cover
MC	% Moss cover	SR	Total Species Richness
LC	% Lichen cover	SWH	Shannon Wiener Diversity
DW	% Dead Wood	NI	Nitrate concentration mg kg-1
ERC	% Cover of Ericaceae	Ca-S	soil Ca concentration mg kg-1
GC	% Grass cover	K-S	soil K concentration mg kg-1
AV	Average Vegetation height (mm)	Mg-S	soil Mg concentration mg kg-1
SE	No. of Seedlings ha-1	Na-S	soil Na concentration mg kg-1
PIH	Pioneer Heath	P-S	soil P concentration mg kg-1
BH	Building & Mature Heath	Ca-M	Ca concentration mg kg-1 in M.caerulea leaf tissue
DH	Degenerate Heath	K-M	K concentration mg kg-1 in M.caerulea leaf tissue
DEH	Dead Heath	Mg-M	Mg concentration mg kg-1 in M.caerulea leaf tissue
CV	% Calluna vulgaris cover	Na-M	Na concentration mg kg-1 in M.caerulea leaf tissue
ET	% Erica tetralix cover	P-M	P concentration mg kg-1 in M.caerulea leaf tissue
EC	% Erica cinerea cover	pH	Soil pH
UE	% Ulex europaeus cover	N-S	Total N Concentration
AC	% Agrostis curtisii cover	C-S	Total Organic C Concentration
NS	% Nardus stricta cover	NH4	Total Ammonium/Ammonia in soil mg kg-1
DD	% Danthonia decumbens cover	C/N	C/N Ratio
Mol	% Molinia caerulea cover		

ii.2 Additional ANOVA results

Table II.3 ANOVA results not reported in the main text for vegetation composition

Variable	Test	df	F	*p*	η^2
Moss %	Intervention Type	1	2. 635	0.120	0.092
	Age following Intervention	4	0.755	0.567	0.106
	Interaction	4	0.712	0.594	0.100
Lichen %	Intervention Type	1	0.967	0.337	0.032
	Age following Intervention	4	1.172	0.353	0.155
	Interaction	4	1.130	0.370	0.150
Average Veg. Height	Intervention Type	1	1.299	0.268	0.040
	Age following Intervention	4	2.247	0.100	0.274
	Interaction	4	0.629	0.648	0.077
Erica cinerea	Intervention Type	1	0.557	0.464	0.018
	Age following Intervention	4	0.309	0.869	0.039
	Interaction	4	2.437	0.081	0.309
Ulex europaeus	Intervention Type	1	1.776	0.546	0.060
	Age following Intervention	4	0.789	0.198	0.107
	Interaction	4	1.161	0.357	0.157

Table II.4 ANOVA results not reported in the main text for abiotic variables

Variable	Test	df	F	*p*	η^2
Total C	Intervention Type	1	0.027	0.871	0.001
	Age following Intervention	4	1.123	0.330	0.171
	Interaction	4	0.943	0.460	0.131
K	Intervention Type	1	3.149	0.091	0.101
	Age following Intervention	4	0.845	0.513	0.108
	Interaction	4	1.171	0.353	0.150
Na	Intervention Type	1	0.641	0.433	0.129
	Age following Intervention	4	1.071	0.397	0.019
	Interaction	4	2.040	0.127	0.247

Table II.5 ANOVA results not reported in the main text for nutrients in *Molinia* tissue

Variable	Test	df	F	*p*	η^2
Ca	Intervention Type	1	0.179	0.676	0.072
	Age following Intervention	4	0.457	0.766	0.007
	Interaction	4	0.822	0.527	0.130
Na	Intervention Type	1	3.570	0.073	0.124
	Age following Intervention	4	1.007	0.427	0.140
	Interaction	4	0.313	0.866	0.043

ii.3 Ordination Output and Validation (MDS and CCA)

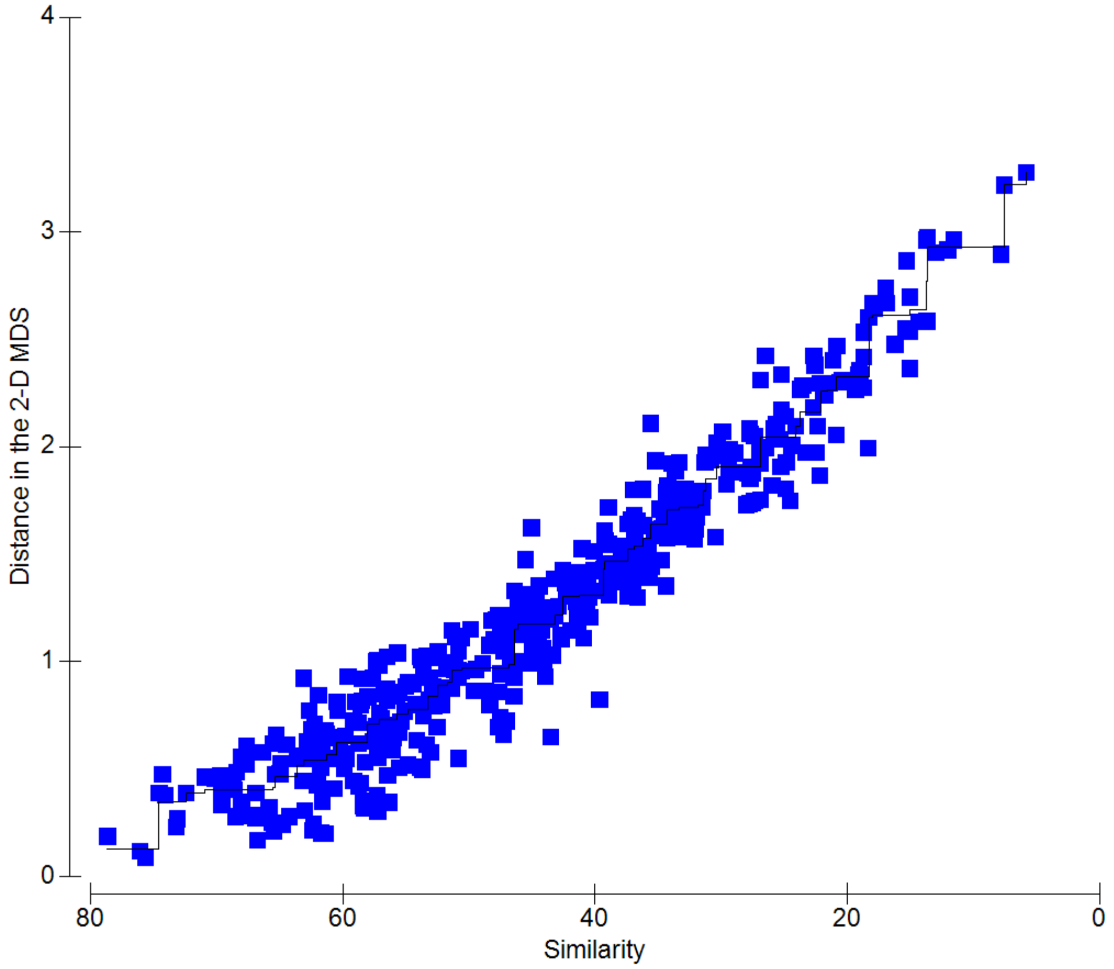

Figure II.2 Shepard Diagram for the MDS analysis from Chapter 2. Total stress was 0.1, and there is relatively little scatter on the diagram.

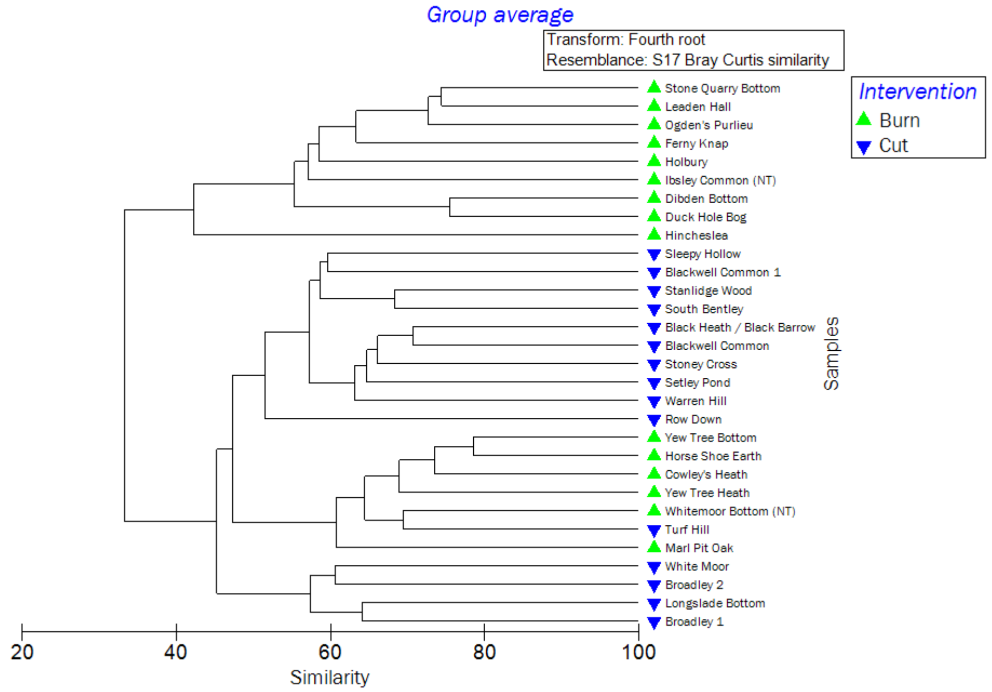

Figure II.3 Cluster Analysis output, based on Bray-Curtis similarity scores, for heathland sites. Names of the plots are displayed.

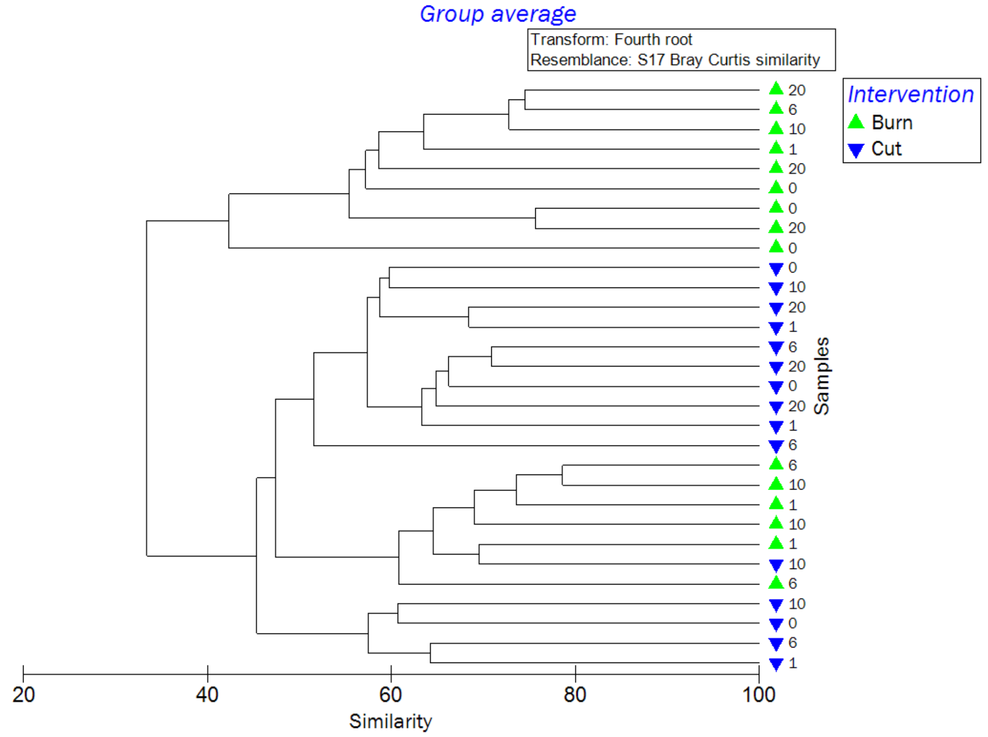

Figure II.4 Cluster Analysis output, based on Bray-Curtis similarity scores, for heathland sites. Similarity at 40% was displayed on the MDS diagram in chapter 2. Numbers refer to the age of the plot following management interventions.

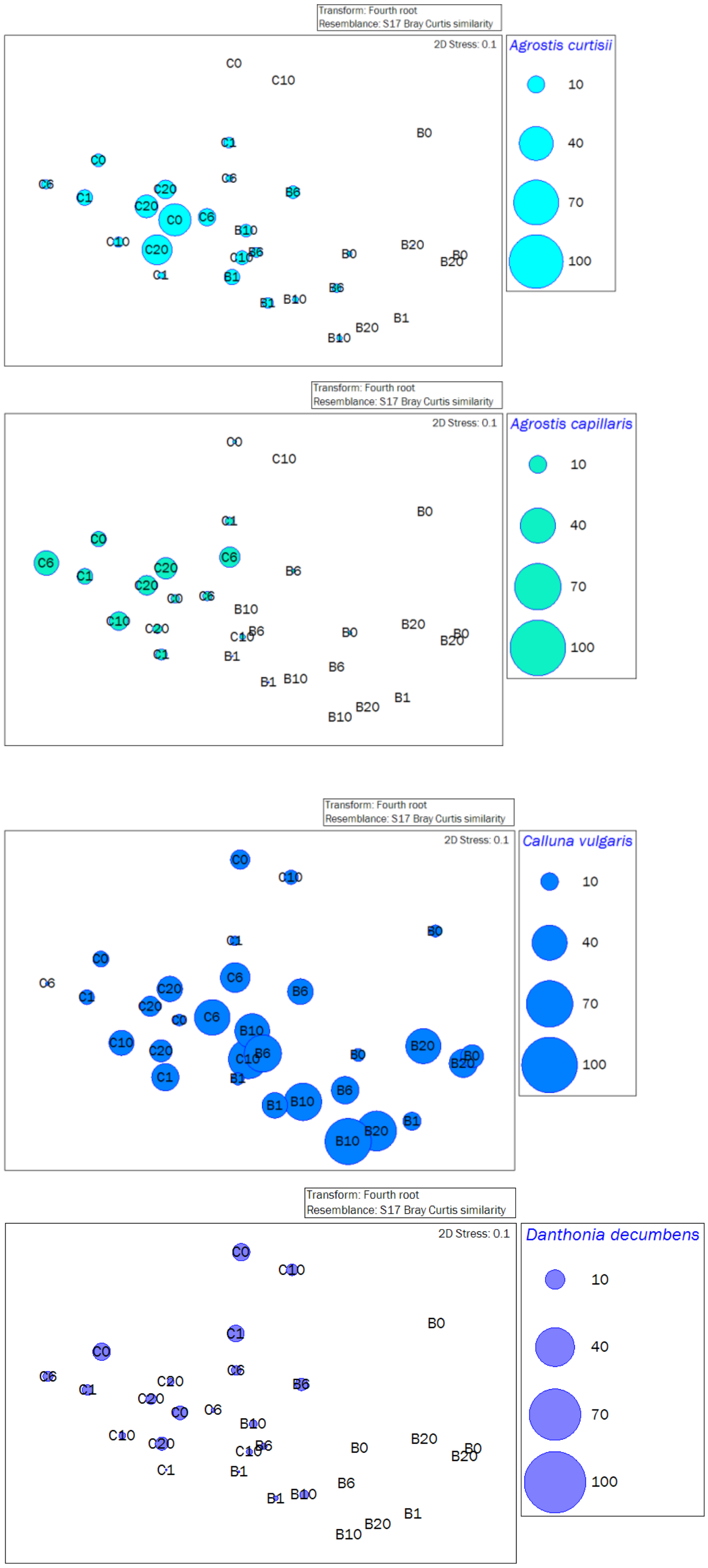

Figure II.5 Bubble plots from MDS analyses showing the abundance of heathland species *Agrostis curtisii, A. capillaris, Calluna vulgaris* and *Danthonia decumbens*. Letters refer to management (B = Burn, C = Cut) and numbers refer to the age following intervention.

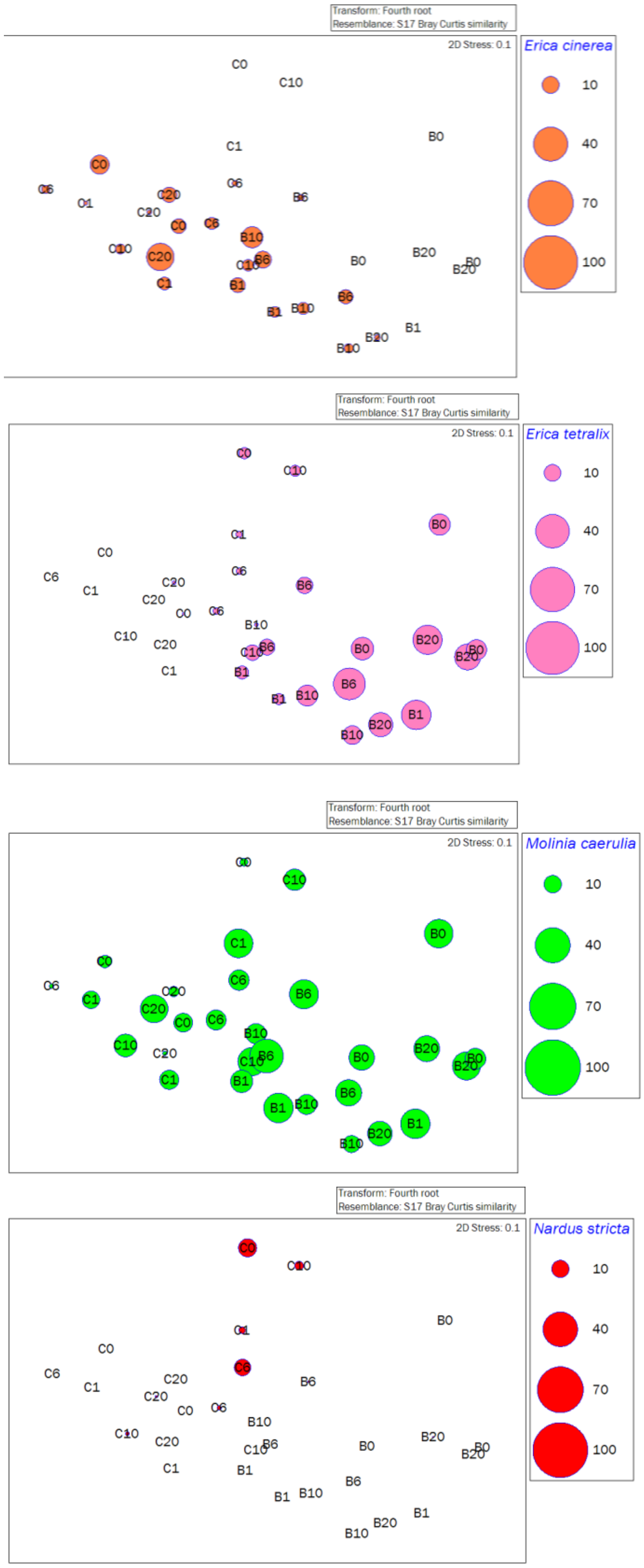

Figure II.6 Bubble plots from MDS analyses showing the abundance of heathland species *Erica cinerea, E. tetralix, Molinia caerulea* and *Nardus stricta*. Letters refer to management (B = Burn, C = Cut) and numbers refer to the age following intervention.

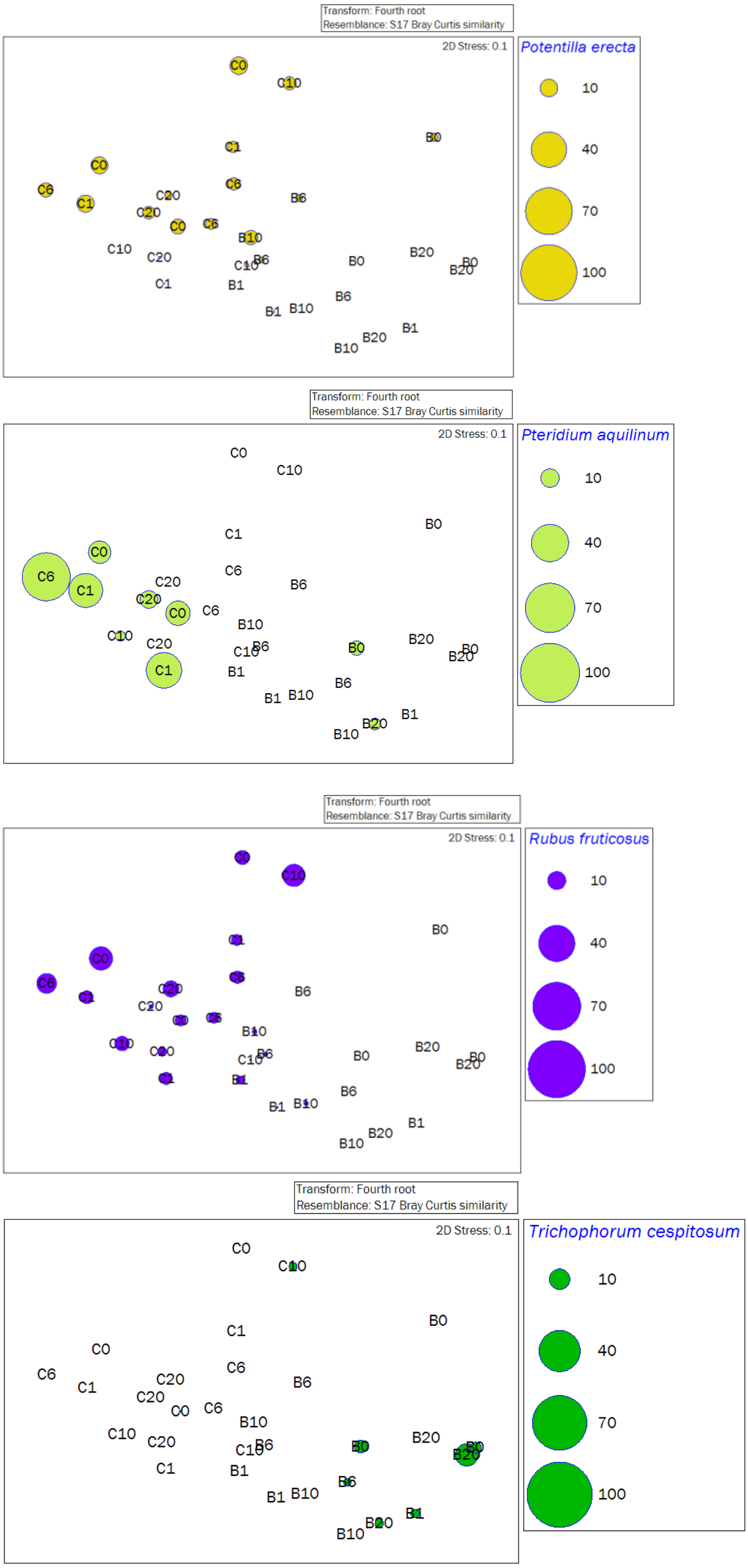

Figure II.7 Bubble plots from MDS analyses showing the abundance of heathland species *Potentilla erecta, Pteridium aquilinum, Rubus fruticosus* and *Tricophorum cespitosum* across the study. Letters refer to management (B = Burn, C = Cut) and numbers refer to the age following intervention.

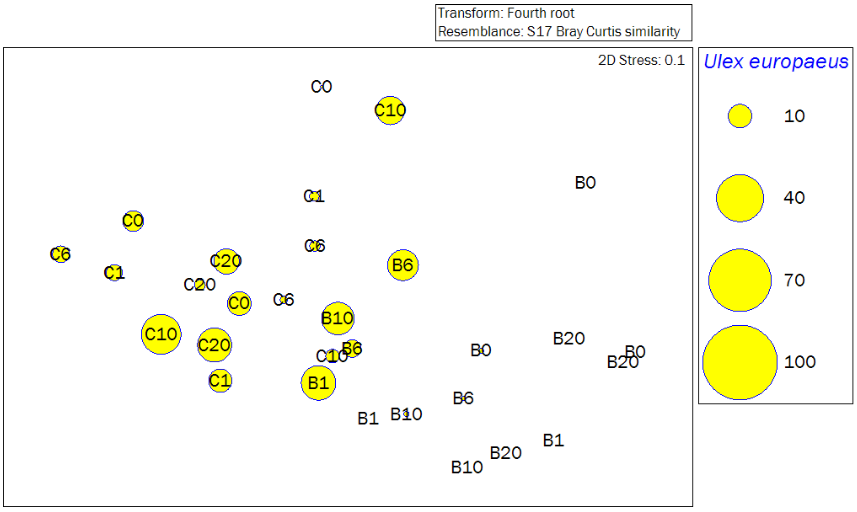

Figure II.8 Bubble plots from MDS analyses showing the abundance of *Ulex europaeus* across the study plots. Letters refer to management (B = Burn, C = Cut) and numbers refer to the age following intervention.

Table II.6 Dissimilarity scores for Burnt and cut plots from SIMPER Analysis

Average dissimilarity = 63.90

	Group Burn	Group Cut				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Erica tetralix	1.92	0.67	2.59	1.83	4.06	4.06
Agrostis capillaris	0.19	1.42	2.58	2.19	4.04	8.1
Rubus fruticosus	0.3	1.46	2.42	2.26	3.78	11.88
Ulex europaeus	0.68	1.53	2.35	1.65	3.68	15.56
Potentilla erecta	0.41	1.31	2.05	1.78	3.2	18.77
Agrostis curtisii	0.83	1.45	1.93	1.23	3.01	21.78
Danthonia decumbens	0.41	1.27	1.92	1.64	3.01	24.79
Carex demissa	0.27	1.08	1.92	1.61	3.01	27.8
Pteridium aquilinum	0.2	0.98	1.92	0.95	3	30.8
Erica cinerea	0.9	1.16	1.74	1.2	2.72	33.52
Rumex acetosella	0.2	0.92	1.72	1.41	2.69	36.2
Gallium saxatile	0.19	0.83	1.52	1.53	2.38	38.58
Carex binervis	0.06	0.72	1.47	1.77	2.31	40.89
Carex panicea	0.57	0.49	1.47	1.13	2.31	43.2
Agrostis stolonifera	0.13	0.58	1.3	0.76	2.03	45.23
Leontodon hispidus	0	0.66	1.28	1.49	2	47.23
Luzula multiflora	0.11	0.66	1.2	1.55	1.87	49.1
Trichophorum cespitosum	0.53	0.11	1.18	0.85	1.84	50.95
Nardus stricta	0.07	0.55	1.17	0.87	1.84	52.78
Calluna vulgaris	2.19	1.89	1.16	1.36	1.82	54.6
Molinia caerulea	2.13	1.73	0.95	1.16	1.49	56.09
Betula pendula	0.13	0.36	0.83	0.93	1.31	57.4
Polygala serpyllifolia	0.43	0.47	0.83	1.2	1.29	58.69
Ulex minor	0.34	0.15	0.81	0.77	1.26	59.95
Teucrium scorodonia	0	0.41	0.74	0.8	1.15	61.11
Salix repens	0.23	0.21	0.73	0.68	1.15	62.26
Viola riviniana	0	0.39	0.72	0.9	1.13	63.38
Unidentified	0.07	0.38	0.71	1.17	1.11	64.49
Juncus squarrosus	0.32	0.06	0.69	0.69	1.08	65.57
Cerastium fontanum	0.04	0.36	0.68	1.16	1.07	66.64
Festuca ovina	0	0.32	0.67	0.8	1.05	67.68
Poa pratensis	0.04	0.32	0.66	0.63	1.03	68.71
Pedicularis sylvatica	0.11	0.31	0.64	0.87	1	69.71
Prunella vulgaris	0	0.35	0.62	0.87	0.98	70.69
Poa annua	0.04	0.29	0.62	0.81	0.98	71.66
Deschampsia setacea	0	0.31	0.62	0.76	0.97	72.63
Lonicera periclymenum	0	0.31	0.57	0.66	0.9	73.53
Veronica officinalis	0	0.31	0.55	0.58	0.87	74.4
Carex flacca	0.23	0.07	0.51	0.72	0.8	75.2
Poa compressa	0.04	0.22	0.49	0.63	0.77	75.97

Juncus bufonius	0.06	0.18	0.44	0.53	0.69	76.66
Vaccinium myrtillus	0	0.21	0.43	0.58	0.67	77.33
Lotus corniculatus	0	0.24	0.43	0.59	0.67	78
Crataegus monogyna	0	0.24	0.42	0.56	0.65	78.65
Holcus mollis	0	0.2	0.41	0.44	0.65	79.3
Quercus robur	0	0.2	0.41	0.58	0.64	79.94
Rosa canina agg.	0	0.22	0.4	0.57	0.63	80.56
Rhynchospora alba	0.2	0	0.4	0.49	0.63	81.19
Juncus effusus	0.09	0.12	0.38	0.52	0.6	81.79
Agrostis canina	0	0.19	0.38	0.58	0.59	82.38
Drosera rotundifolia	0.15	0.05	0.36	0.55	0.57	82.95
Eriophorum angustifolium	0.17	0	0.36	0.37	0.57	83.51
Anthoxanthum odoratum	0	0.18	0.34	0.46	0.53	84.05
Hydrocotyle vulgaris	0	0.17	0.34	0.47	0.53	84.57
Pilosella officinarum	0	0.2	0.34	0.46	0.53	85.1
Pinus sylvestris	0.13	0.04	0.32	0.44	0.5	85.6
Lotus pedunculatus	0	0.17	0.3	0.47	0.48	86.08
Prunus spinosa	0.04	0.12	0.3	0.55	0.47	86.55
Taraxacum agg.	0	0.16	0.29	0.59	0.46	87.01
Ilex aquifolium	0	0.15	0.29	0.59	0.46	87.47
Cirsium palustre	0.06	0.1	0.28	0.47	0.44	87.9
Euphrasia nemorosa	0	0.16	0.27	0.48	0.42	88.32
Narthecium ossifragum	0.13	0	0.26	0.49	0.41	88.73
Centaurium pulchellum	0	0.14	0.25	0.49	0.38	89.11
Drosera intermedia	0.08	0.04	0.24	0.46	0.37	89.48
Ranunculus repens	0	0.13	0.23	0.49	0.36	89.84
Melissa officinalis	0	0.15	0.23	0.48	0.36	90.2

Table II.7 Similarity scores for burnt plots, from SIMPER Analysis

Average similarity: 51.91

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Molinia caerulea	2.13	12.99	5.78	25.03	25.03
Calluna vulgaris	2.19	12.6	3.16	24.28	49.31
Erica tetralix	1.92	11.1	2.87	21.38	70.69
Erica cinerea	0.9	2.82	0.69	5.43	76.13
Agrostis curtisii	0.83	2.72	0.82	5.24	81.36
Carex panicea	0.57	1.37	0.59	2.64	84
Polygala serpyllifolia	0.43	1.32	0.69	2.55	86.55
Trichophorum cespitosum	0.53	1.2	0.4	2.31	88.86
Ulex europaeus	0.68	1.14	0.46	2.19	91.05

Table II.8 Similarity scores for cut plots from SIMPER Analysis

Average similarity: 52.44

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Calluna vulgaris	1.89	4.98	2.78	9.5	9.5
Molinia caerulea	1.73	4.53	2.79	8.64	18.14
Ulex europaeus	1.53	3.79	3.02	7.23	25.38
Rubus fruticosus	1.46	3.71	3.78	7.08	32.46
Agrostis capillaris	1.42	3.37	2.17	6.43	38.9
Agrostis curtisii	1.45	3.26	1.49	6.21	45.1
Danthonia decumbens	1.27	3.25	3.63	6.2	51.3
Potentilla erecta	1.31	3.16	2.76	6.02	57.32
Carex demissa	1.08	2.46	1.49	4.68	62.01
Erica cinerea	1.16	2.36	1.14	4.5	66.5
Rumex acetosella	0.92	1.79	1.01	3.42	69.92
Gallium saxatile	0.83	1.7	1.4	3.24	73.17
Carex binervis	0.72	1.55	1.52	2.95	76.12
Luzula multifolia	0.66	1.3	1.22	2.48	78.6
Leontodon hispidus	0.66	1.08	0.99	2.06	80.66
Erica tetralix	0.67	1.01	0.66	1.93	82.59
Pteridium aquilinum	0.98	0.94	0.48	1.8	84.38
Polygala serpyllifolia	0.47	0.83	0.84	1.59	85.97
Nardus stricta	0.55	0.57	0.47	1.08	87.05
Unidentified	0.38	0.56	0.71	1.07	88.11
Cerastium fontanum	0.36	0.54	0.71	1.03	89.15
Agrostis stolonifera	0.58	0.45	0.32	0.86	90.01

Table II.9 Abbreviations used in CCA diagram, Chapter 2

Cal	*Calluna vulgaris*	Hyp	*Hypericum pulchrum*	Leo	*Leontodon hispidus*	Nar	*Nardus stricta*
Eri	*Erica tetralix*	Cer	*Cerastium fontanum*	Lea	*Leontodon autumnalis*	Dan	*Danthonia decumbens*
Erc	*Erica cinerea*	Cen	*Centaurium pulchellum*	Pil	*Pilosella officinarum*	Mol	*Molinia caerulea*
Poe	*Potentilla erecta*	Pic	*Picris hieracioides*	Hya	*Hyacinthoides non-scripta*	Hol	*Holcus lanatus*
Pon	*Potentilla anglica*	Rar	*Ranunculus repens*	Lot	*Lotus corniculatus*	Hom	*Holcus mollis*
Pol	*Polygala serpyfolia*	Ran	*Ranunculus flammula*	Lop	*Lotus pedunculatus*	Des	*Deschampsia setacea*
Rum	*Rumex acetosella*	Suc	*Succisa pratensis*	Orn	*Ornithopus perpusillus*	Poa	*Poa annua*
Ped	*Pedicularis sylvatica*	Hie	*Hieracium* agg.	Trr	*Trifolium repens*	Pot	*Poa trivialis*
Pru	*Prunella vulgaris*	Cir	*Cirsium palustre*	Trp	*Trifolium pratense*	Pop	*Poa pratensis*
Mel	*Melissa officinalis*	Myr	*Myrica gale*	Sag	*Sagina subulata*	Poc	*Poa compressa*
Sta	*Stachys officinalis*	Hed	*Hedera helix*	Dac	*Dactylorhiza maculata*	Ant	*Anthoxanthum odoratum*
Teu	*Teucrium scorodonia*	Nao	*Narthecium ossifragum*	Ule	*Ulex europaeus*	Tri	*Trichophorum cespitosum*
Pll	*Plantago lanceolata*	Ach	*Achillea millefolium*	Ulm	*Ulex minor*	Era	*Eriophorum angustifolium*
Plm	*Plantago major*	Scu	*Scutellaria minor*	Ulg	*Ulex gallii*	Rhy	*Rhynchospora alba*
Pla	*Plantago media*	Bet	*Betula pubescens*	Rub	*Rubus fruticosus*	Rhf	*Rhynchospora fusca*
Plc	*Plantago coronopus*	Bep	*Betula pendula*	Rui	*Rubus idaeus*	Ele	*Eleocharis multicaulis*
Cus	*Cuscuta epithymum*	Pin	*Pinus sylvestris*	Rus	*Rubus saxatilis*	Caf	*Carex flacca*
Eup	*Euphorbia amygdaloides*	Que	*Quercus robur*	Ros	*Rosa arvensis*	Cap	*Carex panicea*
Ver	*Veronica officinalis*	Ile	*Ilex aquifolium*	Roc	*Rosa canina* agg.	Cad	*Carex demissa*
Eun	*Euphrasia nemorosa*	Fra	*Frangula alnus*	Ror	*Rosa rubiginosa*	Cab	*Carex binervis*
Epn	*Campanula rotundifolia*	Hyd	*Hydrocotyle vulgaris*	Cra	*Crataegus monogyna*	Can	*Carex nigra*
Gal	*Gallium mollugo*	Epi	*Epilobium hirsutum*	Mal	*Malus sylvestris*	Cac	*Carex curta*
Gas	*Gallium saxatile*	Agr	*Agrostis capillaris*	Prp	*Prunus spinosa*	Jun	*Juncus effusus*
Lon	*Lonicera periclymenum*	Ags	*Agrostis stolonifera*	Sor	*Sorbus aucuparia*	Jua	*Juncus acutifolus*
Jac	*Jacobaea vulgaris*	Agc	*Agrostis curtisii*	Rho	*Rhododendron ponticum*	Jus	*Juncus squarrosus*
Jaa	*Jacobaea aquatica*	Aga	*Agrostis canina*	Vac	*Vaccinium myrtillus*	Jub	*Juncus bufonius*
Tar	*Taraxacum* agg.	Fes	*Festuca ovina*	Sal	*Salix repens*	Luz	*Luzula campestris*
Hyr	*Hypochaeris radicata*	Fer	*Festuca rubra*	Vio	*Viola riviniana*	Lum	*Luzula multiflora*
Hyg	*Hypochaeris glabra*	Feg	*Festuca gigantia*	Dro	*Drosera rotundifolia*		
Pte	*Pteridium aquilinum*	Bro	*Bromus hordeaceus*	Dri	*Drosera intermedia*		

Table II.10 Eigenvalues and canonical coefficients of explanatory variables for CCA axes.

Axis	1*	2*	3	4	5	6
Eigenvalue	0.462062	0.258703	0.111762	0.090367	0.079483	0.044997
Species-Environment Correlations	0.802299	0.655172	0.728736	0.577011	0.641379	0.535632
% variance explained	19.4216	10.8739	4.69761	3.79834	3.34087	1.89133
Cumulative % variance	19.4216	30.2955	34.9931	38.7915	42.1323	44.0237
Correlation Coefficient						
pH	-0.02367	0.37828	0.191497	0.051706	0.013481	0.099652
TN	0.506037	0.063114	0.172828	-0.24954	-0.1693	-0.20368
TOC	-0.36773	-0.0962	-0.05657	0.077437	0.15426	0.028776
Ca	0.131492	-0.32458	-0.06782	-0.05673	0.073325	0.21187
K	-0.0395	0.300257	-0.00358	-0.05936	0.197665	-0.12018
P	-0.19158	0.157154	-0.2341	0.323731	-0.0915	0.089091
Bare Ground	0.021448	0.254567	0.180543	0.012541	0.206573	-0.13575
Av. Vegetation Height	0.288457	-0.29802	0.30056	-0.04766	0.25548	-0.07891
Age	0.200375	-0.27696	0.047479	-0.31381	-0.02314	0.031784
Management	-0.46545	0.022369	-0.00306	-0.01808	-0.08007	0.089949
Elevation	-0.16174	0.170562	0.136553	-0.03036	-0.18413	-0.18576

*Significant (*p* > 0.05) on Monte-Carlo permutations, n = 1000

Table II.11 Multicollinearity test of selected explanatory variables for CCA

Dependent variable	R-squared	VIF*
pH	0.358986	1.56003
TN	0.881615	8.44701
TOC	0.755093	4.08318
Ca	0.702761	3.36429
K	0.55298	2.23704
P	0.861252	7.20732
Bare Ground	0.662449	2.96252
Avg. Vegetation Height	0.602584	2.51626
Young Site	0.486897	1.94893
Burn	0.642104	2.79411
Elevation	0.510118	2.04131

*VIF > 10 considered unsuitable

Appendix III – Supplementary Information and statistics for Chapter 3

iii.1 Description of NVC Community Types Assessed

H2 *Calluna vulgaris – Ulex minor* heath

This community, unique to the south of England, is dominated by *Calluna*, with *Erica cinerea* and *Ulex minor* providing distinctiveness. Vegetation structure is heavily affected by the *Calluna* growth strages, which are linked to burning and grazing in the New Forest (Rodwell, 1991), often found in a patchwork. Recovery after burning can lead to marked changes in species distribution, with more *Erica cinerea*. *Ulex minor* is reduced in the New Forest, likely due to grazing, whereas *U. europaeus* is more frequent. Molinia is present in damper areas. The presence of other plants is quite variable, although tree seedlings and saplings are likely without interventions (Rodwell, 1991). This community is characteristic of poor acid soils, and found with drier and more easily drained soils than other communities, with which it may be zoned (Tubbs, 1986; Rodwell, 1991). This boundary often not well defined. More fertile soils may have *Rubrus* sp. present. Two sub-communities are present in the Forest: H2a typical sub-community, and H2c *Molinia caerulea* sub-community. H2a features prominent lichens, particularly *Cladonia*, present in older or recently burnt stands, while trees and bracken are present but infrequent. *Molinia* is rare but *Deschampsia* locally common. H2c, a typically wetter community, features dominant *Calluna* with *E.cinerea* sometimes replaced by *E.tetralix*. *Molinia* is more common, but trees and bracken less so. There is a sparse ground layer (Rowell, 1991)

H3 *Ulex minor* - Agrostis curtisii heath ("humid heath")

There is a wide variety of heath vegetation present in this community, but common features are a low canopy and structure; *Calluna* dominates but degenerate heather is not very frequent (Rodwell, 1991). Uniquely, there is mixed occurance of *E. cinerea* and *E. tetralix* which are normally separate. *E. cinerea* is especially abundant on areas recently burnt. *Agrostis curtisii* and *Molina* maintain a constant presence, often in tussocks, and these species can also spread after burning. *Molinia* in particular spreads on wetter ground. Where the heather canopy is relatively open, there is a diverse sub-flora in the ground layer. This community is characteristic of poor acid soils which do not experience droughts, and occupies areas too dry for M16 and too wet for H2; it often transitions to both habitats. H3 is essentially confined to Hampshire and Dorset. There are three communities noted in the Forest: H3a Typical sub-community, where *Calluna* forms a dominant canopy, H3b *Cladonia* sub-community, which features a more open canopy, with greater abundance of grasses, and H3c *Agrostis curtisii* , where *A. curtisii* is much more abuntant, *E. cinerea* frequent, and Calluna and Molinia reduced in cover (Rodwell, 1991).

M16 *Ericetum tetralicis* wet heath

This habitat is dominated by mixtures of *Calluna*, *Erica tetralix* and *Molinia* in variable proportions, as a result of local conditions such as soil water content (Rodwell, 1991). *E. tetralix* is widespread, with *Calluna* less dominant than the other heath communities and featuring a low, prostrate canopy. *Molinia* can form dense tussocks in some areas. M16 is moderately species rich, and bryophytes should include the presence of *Sphagnum* (particularly *S. compactum* and *S. tenellum* – other species are rare). The community is found between the dry heath communities described above and valley bogs and mires (Tubbs, 1986). M16 occurs on acid and poor soils that are seasonally waterlogged, particularly on the edges of valley mires with high water tables. This boundary with mires and bogs is often associated with *S. papillosum* and *S. magellanicum*, starting from the point of transition. Burning has a marked effect on the community, which is somewhat insulated by the wetter soils but can be variable. Frequent burning however linked to the increase of *Calluna* and loss of distinctive bryophytes, and is considered especially destructive when linked to drainage (Rodwell, 1991). Two communities are described in the Forest. The M16a Typical sub-community, which is the most widespread type of vegetation, features a very diverse structure, often with *Sphagnum*. Sometimes very species poor communities are present in impoverished wet heath (Rodwell, 1991). M16b *Succisa pratensis* – *Carex panicea* sub-community features dominant *Molinia*, with *Succisa pratensis* and *Carex* sp. constant. Occasionally *Juncus effusus* and *J.acutiflorus* are present. Bryophytes and lichens less common in this sub-community.

iii.2 ANOVA Test Output for Chapter 3

Table III.1 ANOVA Tests for CSM Attributes and Scores (Dry Heath)

Variable	Test	df	F	*p*
Bare Ground	Age following Intervention	4	0.990	0.436
	Intervention Type	1	2.524	0.128
	Interaction	4	1.213	0.336
Dwarf Shrub Cover	Age following Intervention	4	1.538	0.229
	Intervention Type	1	16.216	0.001
	Interaction	4	0.146	0.962
Gorse Cover	Age following Intervention	4	0.327	0.856
	Intervention Type	1	4.082	0.057
	Interaction	4	0.316	0.864
Pioneer Heath	Age following Intervention	4	2.753	0.057
	Intervention Type	1	1.142	0.298
	Interaction	4	6.361	0.002
Building Heath	Age following Intervention	4	3.799	0.019
	Intervention Type	1	2.715	0.115
	Interaction	4	1.831	0.162
Degenerate Heath	Age following Intervention	4	3.969	0.016
	Intervention Type	1	7.200	0.014
	Interaction	4	4.981	0.006
Dead Heath	Age following Intervention	4		NA*
	Intervention Type	1		NA*
	Interaction	4		NA*
Dwarf Shrub Species	Age following Intervention	4	1.171	0.353
	Intervention Type	1	21.631	0.000
	Interaction	4	0.955	0.453
Graminoids	Age following Intervention	4	0.327	0.856
	Intervention Type	1	6.368	0.020
	Interaction	4	0.316	0.864
Desirbable Forbs	Age following Intervention	4	1.064	0.400
	Intervention Type	1	26.724	0.000
	Interaction	4	2.199	0.106
Exotic Species	Age following Intervention	4		NA*
	Intervention Type	1		NA*
	Interaction	4		NA*
Invasive Species	Age following Intervention	4	1.933	0.144
	Intervention Type	1	1.667	0.211
	Interaction	4	3.333	0.030
Trees and Shrubs	Age following Intervention	4	0.532	0.714
	Intervention Type	1	10.051	0.005
	Interaction	4	0.596	0.670
Bracken	Age following Intervention	4	0.925	0.469
	Intervention Type	1	6.811	0.017
	Interaction	4	0.844	0.514
Bryophytes	Age following Intervention	4	0.625	0.650
	Intervention Type	1	6.250	0.021
	Interaction	4	0.625	0.650
Erosion	Age following Intervention	4	1.454	0.253
	Intervention Type	1	4.082	0.057
	Interaction	4	1.454	0.253
Mosses	Age following Intervention	4	1.083	0.391
	Intervention Type	1	0.667	0.424
	Interaction	4	2.750	0.057
Total Score	Age following Intervention	4	1.579	0.219
	Intervention Type	1	0.090	0.767
	Interaction	4	1.338	0.291

* No difference in attribute score

iii.3 Scores Calculated for Wet Heath CSM Assessment.

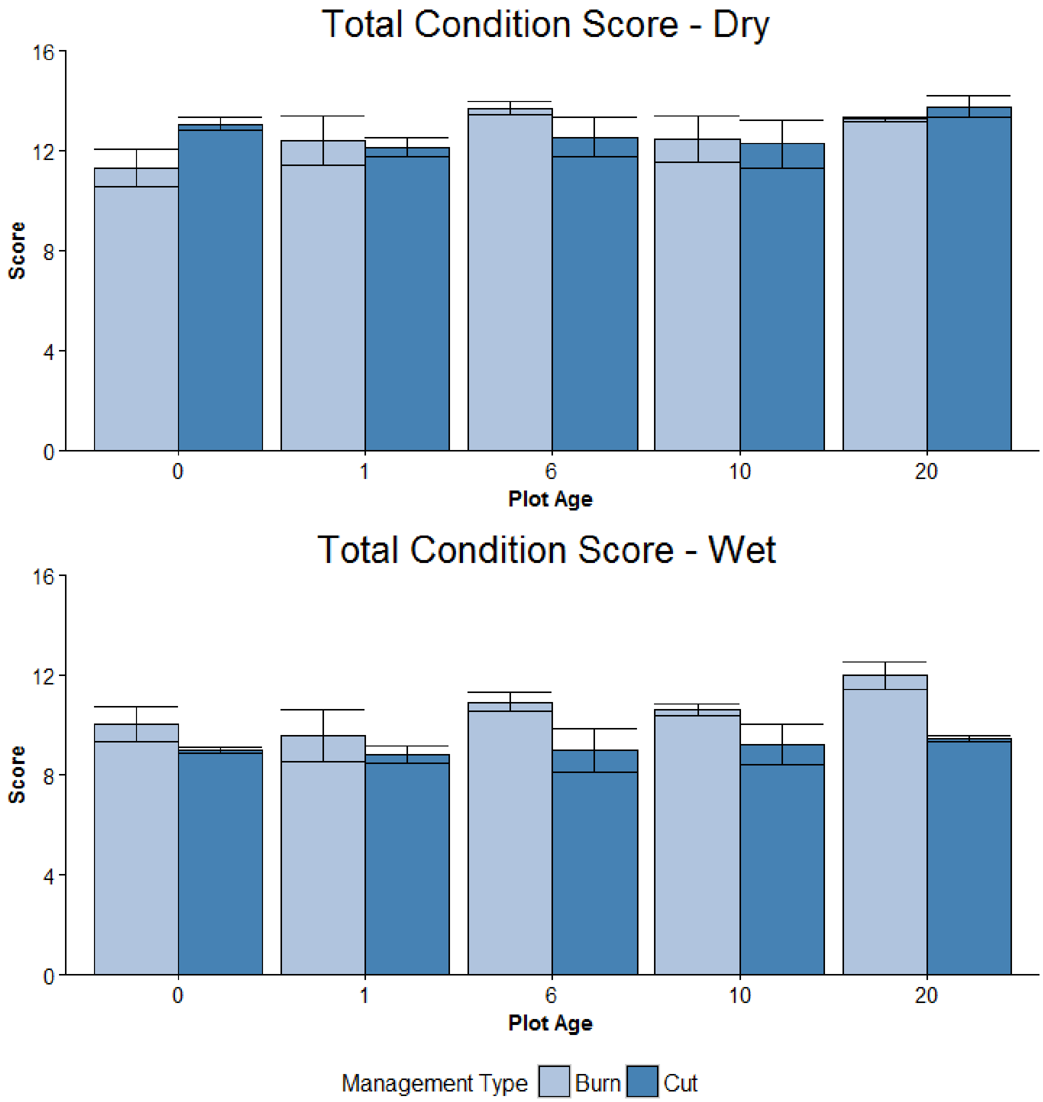

Figure III.1 Total condition scores for the study plots, based on dry heath condition (above) and wet heath condition (below) requirements. Error bars indicate standard error.

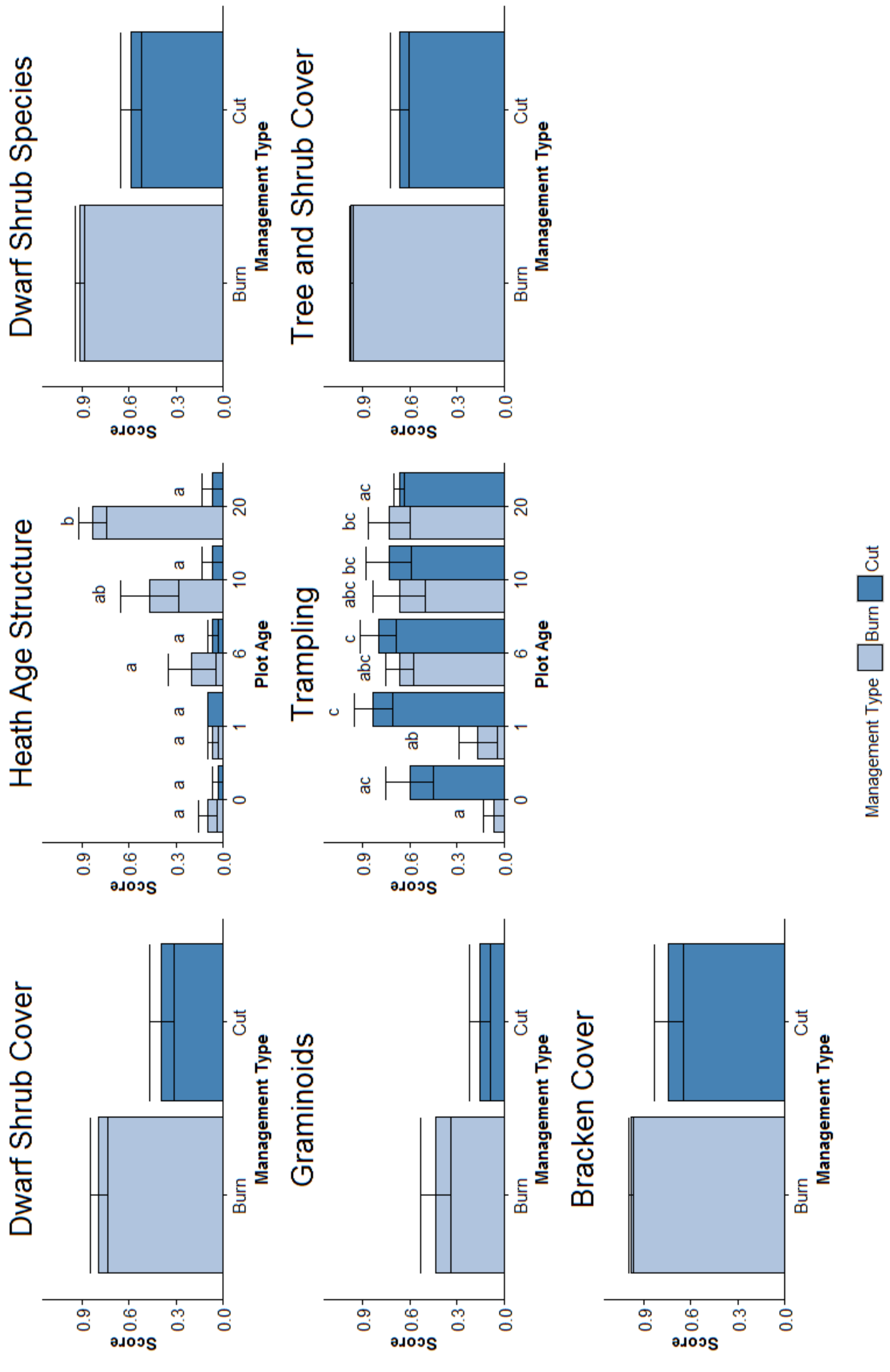

Figure III.2 Attributes that showed differences between plots, based on wet heath attributes. Error bars indicate standard error. Means grouped by the same letter are not significantly different ($p < 0.05$, Tukey test)

Table III.2 ANOVA Tests for CSM Attributes and Scores (Wet Heath)

Variable	Test	df	F	*p*
Bare Ground	Age following Intervention	4	0.990	0.436
	Intervention Type	1	2.524	0.128
	Interaction	4	1.213	0.336
Dwarf Shrub Cover	Age following Intervention	4	1.538	0.229
	Intervention Type	1	16.216	0.001
	Interaction	4	0.146	0.962
Dwarf Shrub Age Structure	Age following Intervention	4	5.640	0.003
	Intervention Type	1	19.512	0.000
	Interaction	4	5.701	0.003
Dwarf Shrub Species	Age following Intervention	4	1.171	0.353
	Intervention Type	1	21.631	0.000
	Interaction	4	0.955	0.453
Graminoids	Age following Intervention	4	0.400	0.806
	Intervention Type	1	5.582	0.028
	Interaction	4	1.650	0.201
Desirable Forbs	Age following Intervention	4	0.278	0.889
	Intervention Type	1	2.689	0.117
	Interaction	4	0.744	0.573
Sphagna	Age following Intervention	4	1.253	0.321
	Intervention Type	1	2.813	0.109
	Interaction	4	1.275	0.313
Lichens	Age following Intervention	4	0.724	0.586
	Intervention Type	1	0.862	0.364
	Interaction	4	1.552	0.226
Drains	Age following Intervention	4		NA
	Intervention Type	1		NA
	Interaction	4		NA
Trampling	Age following Intervention	4	4.076	0.014
	Intervention Type	1	12.214	0.002
	Interaction	4	3.435	0.027
Leaching	Age following Intervention	4		NA
	Intervention Type	1		NA
	Interaction	4		NA
Exotic Species	Age following Intervention	4		NA
	Intervention Type	1		NA
	Interaction	4		NA
Invasive Species	Age following Intervention	4	0.547	0.704
	Intervention Type	1	0.093	0.764
	Interaction	4	2.012	0.132
Trees and Scrub	Age following Intervention	4	0.709	0.595
	Intervention Type	1	24.605	0.001
	Interaction	4	0.767	0.559
Bracken	Age following Intervention	4	1.115	0.377
	Intervention Type	1	6.750	0.017
	Interaction	4	0.995	0.433
Gorse	Age following Intervention	4	0.664	0.625
	Intervention Type	1	1.641	0.215
	Interaction	4	1.323	0.296
Total Score	Age following Intervention	4	1.746	0.180
	Intervention Type	1	15.938	0.001
	Interaction	4	0.666	0.623

Appendix IV – Habitat Suitability Indices for Heathland Species

IV.1 Smooth Snake *Coronella austriaca*

Figure IV.1 Smooth Snake *Coronella austriaca.* Photograph by Mircea Nita, creative commons attribution.

General Characteristics

The smooth snake *Coronella austriaca* is a small Colubrid snake distributed from northern and central Europe stretching to northern Iran. It is cryptic and rarely encountered compared to the other snakes present in the UK, the grass snake *Natrix natrix* and the adder *Vipera berus* (Pernetta, 2009). It is the subject of conservation concern owing to declines observed in both its distribution and population, which are thought to be a result of habitat loss (Goddard, 1984) and in some cases persecution (Santos *et al.,* 2009). It has become extinct in Denmark and most European populations are thought to still be declining (Corbett, 1989; Santos *et al.,* 2009; Pernetta, 2009). In the United Kingdom, this species is exclusively found in heathland environments (Reading, 2012); it utilises a much wider range of habitats in continental Europe (Santos *et al.,* 2009). Smooth snakes rarely bask in the open and have lower temperature requirements than other British snakes (Edgar *et al.,* 2010). The main period of activity lasts from late March through to late October (Edgar *et al.,* 2010). It is classified as specially protected in the European Union (Council Directive 92/43/EEC on the Conservation of natural habitats and of wild fauna and flora, annex IV) and is protected under the Wildlife and Countryside Act, 1981.

Food and Foraging Habitat

In the UK, the smooth snake primarily preys on other reptiles and small mammals (Goddard, 1984; Reading & Jofré, 2013). A key prey species is the common lizard *Zootoca vivipara* (Brown *et al.,* 2014; Reading & Jofré, 2013) which is the preferred food of juveniles and gravid females; areas of heathland habitat which support this prey species are therefore also appropriate habitat for the smooth snake (Reading & Jofré, 2013). Grazing is often considered to damage such habitat by removing *Molinia caerulea* and *Agrostis curtisii* cover, leading to reduced food availability and affecting survival rates (Reading & Jofré, 2013). This is supported by recent study work that showed greatly reduced populations of *Z. vivipara* in response to grazing (Strijbosch, 2002; Wallis de Vries *et al*., 2013) and lower abundance of *C. austriaca* in grazed habitat compared to ungrazed areas (Reading & Jofré, 2015).

Reproduction and Nesting Habitat

Mating occurs in spring but has hardly ever been observed in the wild (Edgar *et al.,* 2010; Pernetta, 2009). The smooth snake gives birth to live young, with eggs hatched internally prior to giving birth (Edgar *et al.,* 2010). Typically, 4-15 young are born, and birth may require a very dense, thick ground layer with humid cover such as moss and lichen layers or large grass tussocks (Edgar *et al.,* 2010).

Interspersion

Smooth snakes have small ranges, with males ranging approximately twice as far as females (Reading, 2012). Male habitat sizes are approximately 2ha (Reading, 2012). Range size is variable depending on the availability of resources (such as cover and prey availability), which will result in smaller home ranges where resources are abundant (South, 1999; Reading, 2012). Daily movements are small, often as little as 20m and only rarely greater than 100m (Edgar *et al.,* 2010).

Special Considerations

Smooth snakes have specific requirements for hibernation, and shelter during cool conditions. The microenvironment of dense cover where snakes reproduce (see above) also helps to insulate individuals against extremes of cold or heat. Cover of lichen, moss and leaf litter is thought to provide suitable habitat for snakes to burrow into for shelter and hibernation (Reading & Jofré, 2015; Braithwaite et. al. 1989; Spellerberg & Phelps, 1977). When active, snakes are typically found in dense heather cover, with isolated patches of bare ground, but too much bare ground may leave the snakes exposed and vulnerable to predation (Spellerberg & Phelps, 1977). Large *Molinia* tussocks are considered to be good habitat indicators for shelter as well as feeding purposes (Edgar *et al.,* 2010).

Model Applicability

This model is geographically limited to the south of England, and is specific to heathland habitats in Dorset and Hampshire. The model should be applied to areas of 2-3ha to assess features in an appropriate scale for the range for individual smooth snakes. The model has not been verified in practice, but serves as a tool to compare differences in the habitat score with CSM scores.

Model Description

The model is based on several habitat requirements of *Coronella austriaca*, including feeding requirements, nesting requirements and hibernation and shelter. There is considerable overlap between the shelter and nesting requirements. The final output score of the model is simply the sum of these constituent habitat variable scores.

V1. Vegetation Height. A relatively high and deep cover of heath is required as shelter for the Smooth Snake. Heterogenous structure of heathland is considered best (Spellerberg & Phelps, 1977), so very low or very high cover is considered unsuitable. Cover heights related to high abundance of snakes has been recorded at 35-40cm (Reading & Jofré, 2015) and 40cm deep (Spellerberg & Phelps, 1977). Optimum suitability was considered to be a cover between 30-45 cm, with under 20cm and over 1m considered unsuitable because the canopy would either be too small to provide cover or too dense to provide basking and feeding opportunities.

V2. Ericaceous Plants Cover. As *C. austriaca* is exclusive to heathland habitats in the UK ericaceous cover is considered an essential component of its habitat requirements. Smooth snakes are also associated with high heather ground cover (Reading & Jofré, 2015). Cover below approximately 15% is considered unsuitable, because such low cover is likely not structurally suitable for the species. Over 30% cover is considered optimal, as this level of cover provides structure while also accounting for the presence of other plants that may contribute to the habitat.

V3. Cover of degenerate Heath. Smooth snakes prefer older, mature heath and the presence of degenerate growth forms of heather is an indicator of limited disturbance to the site. Even-aged stands of heath are unsuitable habitat as they indicate a lack of structural diversity. Above 15% cover of degenerate heath is considered optimal habitat. Areas with no degenerate heath at all are likely sub-optimal for the smooth snake but may still be used, so low cover values are considered partly suitable.

V4. Cover of *Molinia caerulea. Molinia* cover is a preferred cover of *Zootoca vivipara* (Stumpel and Ven der Werf, 2012), an important prey species for juvenile and gravid snakes (Brown *et al.,* 2014; Reading & Jofré, 2013) and *Molinia* also provides appropriate cover for the snakes themselves (Edgar *et al.,* 2010). Snakes are often associated with tall grass and high cover of grass litter (Reading & Jofré, 2015). However, cover values in excess of 60% are likely to lead to poor structure and unsuitable habitat. A medium cover of 20-40% is estimated to be optimal.

Table IV.1 Attributes making up the Habitat Suitability Index for Smooth Snake, *Coronella austriaca*

Variable	Source	Assumption
Vegetation Height	Reading and Jofré, 2015; Spellerberg & Phelps, 1977	Suitability of habitat is related to the vertical structure of the vegetation. Optimum suitability occurs at a vegetation height of 350-450mm. Below 20mm and above 1000mm is unsuitable because of reduced structural diversity.
Cover of Ericaceous plants	Reading and Jofré, 2015; Spellerberg & Phelps, 1977	*C. austriaca* is exclusive to heathlands in the UK and has been found in strong associations with heather cover. Below 15% cover is likely unsuitable because of the lack of structural diversity, optimal cover is over 30%
Cover of Degenerate Heath	Spellerberg & Phelps, 1977; Edgar *et al*., 2010	Degenerate, older heath with open patches is suitable habitat. Above 15% cover is optimal, below 5-10% is unsuitable. Even-aged stands are also unsuitable.
Cover of *Molinia caerulea*	Reading & Jofré, 2015; Stumpel and Ven der Werf, 2012; Reading, 2012	Cover of *Molinia,* especially large tussocks, is preferred by the important prey species, *Zootoca vivipara*. Optimal cover is estimated at 20-40%. Below 20 is not sufficient but above 60 may be poor habitat because of *C. austriaca*'s thermal and movement requirements.
Bare Ground	Pernetta, 2009; Spellerberg & Phelps, 1977	Some patches required for basking of prey species, but above 10% on a heath patch is unsuitable, and patches are probably not necessary for *C. austriaca* itself.
Ground Layer	Reading & Jofré, 2015; Edgar *et al*., 2010; Braithwaite *et. al.* 1989; Spellerberg & Phelps, 1977	Combined cover of Moss, Lichen and Leaf Litter is important for nesting and sheltering during cold periods. >20% optimal, but not at the expense of shrub or grass canopy cover. Below 10% is unsuitable.
Grazing Pressure	Reading & Jofré, 2015; Stumpel and Ven der Werf, 2012	Disturbance is likely to affect habitat suitability in addition to the direct effects of grazing on habitat structure. Trampling above 10% is unsuitable, but the optimum is 0 %.

V5. Bare Ground. The presence of bare ground patches is typical of areas frequented by smooth snakes (Spellerberg & Phelps, 1977) and may provide basking opportunities for its prey species. Small patches are also likely to be indicative of mature, late-stage heath. However, the snake does not require patches for its own basking, and large patches caused by disturbance are poor habitat. Bare patches above 10% are considered unsuitable habitat, with 0-5% considered optimal.

V6. Ground Layer. A deep and humid ground layer is required for nesting and shelter by the snake (Reading & Jofré, 2015; Edgar et al., 2010; Braithwaite et. al. 1989; Spellerberg & Phelps, 1977). This attribute is measured by the combined cover of bryophytes, lichens and leaf litter. A high cover above 20% is considered optimal, but not where this results in a loss of ericaceous or graminoid canopy cover. Below 10% cover is considered unsuitable with not enough habitat to shelter in.

V7. Grazing Pressure. Snakes are considered to respond poorly to disturbance (Reading & Jofré, 2015), and grazing is also associated with declines in prey species (Stumpel and Ven der Werf, 2012) and a loss of structural diversity (Newton *et al.,* 2010). The optimum is 0%, but recorded pressure above 10% cover of quadrats is considered unsuitable habitat for the snake because of the high levels of disturbance.

Model Relationships

Suitability Index (SI) graphs for habitat variables are presented here in figure A-1. The suitability index (SI) is read directly from the graph, with 1.0 equalling perfectly suitable habitat and 0.0 indicating unsuitable habitat.

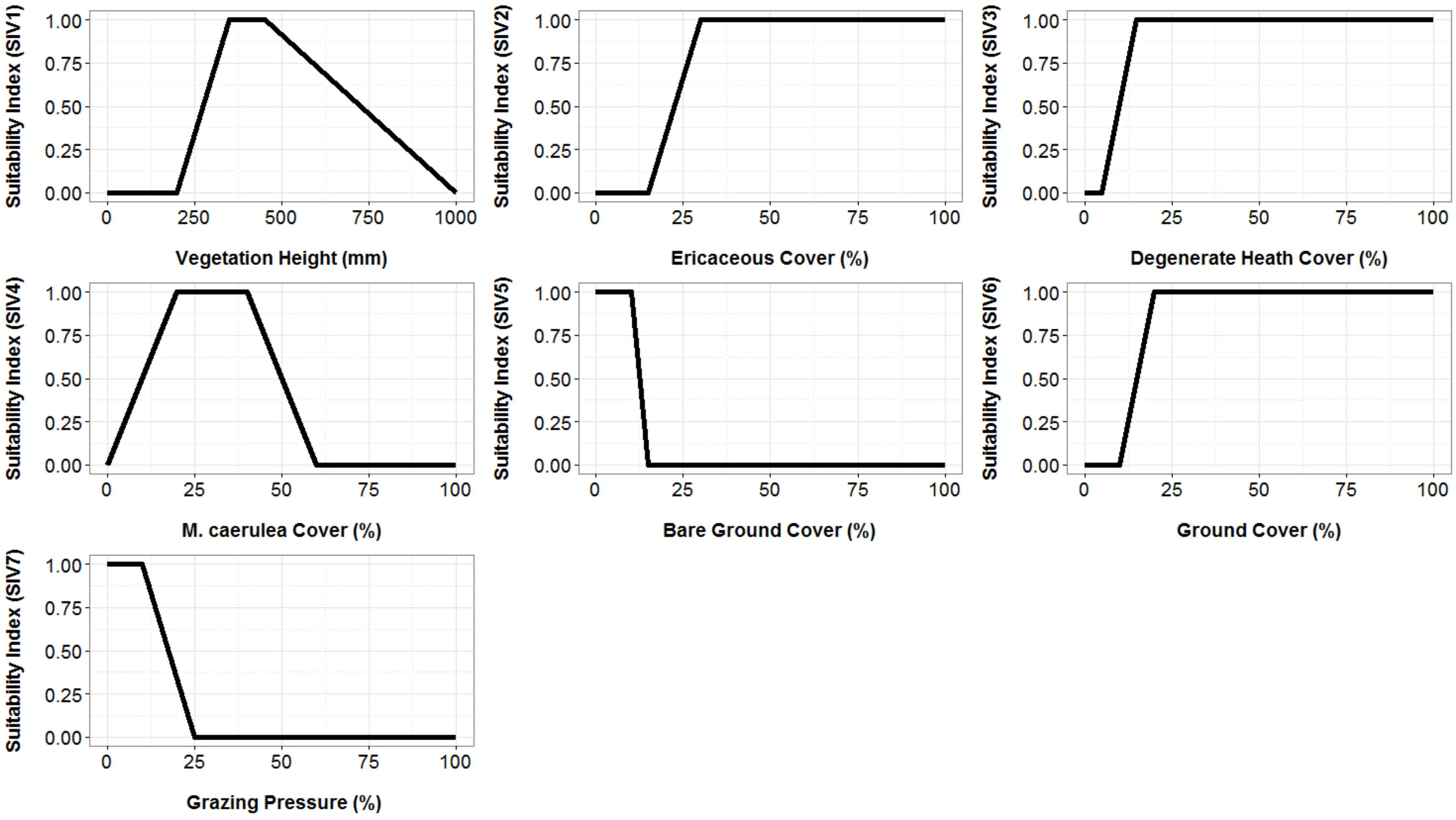

Figure IV.2 Habitat Suitability Graphs for the Smooth Snake

IV.2 Silver-studded blue butterfly *Plebejus argus*

Figure IV.3 Silver-studded blue butterfly *Plebejus argus*. Photograph by the author.

General Characteristics

The Silver-studded blue butterfly *Plebejus argus* is a distinctive butterfly found in temperate habitats of Europe and Asia. It tolerates a wide range of habitats in Europe, but in Britain is found in heathland and calcareous grassland environments, where it is considered an indicator of active management (Ravenscroft & Warren, 1996). In Britain it has experienced a severe decline and considerable losses in range and is now confined to southern England, the Welsh coast and parts of East Anglia (Ravenscroft & Warren, 1996). However, it is locally abundant in appropriate habitat in Hampshire and Dorset. Declines are linked to habitat loss and fragmentation (Ravenscroft & Warren, 1996). In heathland habitats, *P. argus* is often found on short lived pioneer habitat, and is typically absent from later stages of succession (Thomas, 1985a; Schimel & Fartmann, 2014). Adults fly from mid-June or July until August, with males emerging before females (Ravenscroft & Warren, 1996). It is a UK Biodiversity Action Plan species and is protected under the Wildlife and Countryside Act 1981 (Schedule 5).

Food Requirements

The larvae of *P. argus* utilise a wide range of plants commonly found in heathland environments, particularly ericaceous and leguminous plants, including *Calluna vulgaris, Erica* spp., *Ulex* spp. and *Lotus corniculatus* ((Thomas, 1985a). Despite the wide range of food sources, there is some evidence that the butterfly larvae specialise on young, early growth (Thomas, 1985b). The species generally avoids densely grass-covered heathland sites (Schimel & Fartmann, 2014), where food sources may be scarce. Adult butterflies take nectar from several plants; at most sites these will be from the flowers of heather (Ravenscroft & Warren, 1996).

Reproduction

Mating butterflies utilise shrubs and tall herbs a short distance from hostplants (Dennis, 2004). Eggs are typically laid at the margin of vegetation cover and bare ground on or under short vegetation (Thomas, 1985b). Thomas (1985a) found a majority of eggs in vegetation less than 7cm tall. *P. argus* is often absent from consistently tall or thick (10–40cm) vegetation without bare ground, despite an abundance of larval foodplants (Thomas 1985a). The woody stems of heather or gorse are a favoured location for eggs (Ravenscroft, 1990). In southern England, slightly taller vegetation is tolerated in humid environments (Ravenscroft & Warren, 1996).

Vegetation Cover

Butterfly habitat is often considered based on the presence of larval hostplants and breeding resources, but other features, such as cover for roosting and mate location are also important (Dennis, 2004). Most colonies occur on shallow slopes or flat ground (Thomas, 1985b), although steep slopes are sometimes used for shelter (Dennis, 2004). Roosting is often on heather bushes or grass tussocks (Ravenscroft & Warren, 1996). Most heathland colonies exist on areas that are recently disturbed or actively managed (Thomas, 1985a; Ravenscroft, 1990).

Interspersion

Typically, *P. argus* territories are at least 0.1 ha in size, ranging up to 0.5 ha (Thomas, 1985b), and only rarely disperse more than 100m (Lewis *et al.,* 1997). They may range further if nectar resources are not present (Mair *et al.,* 2015).

Special Considerations

As with other Lycaenid butterflies, *P. argus* shows a mutualistic relationship with ants, and specifically with *Lasius* spp. This relationship may have evolved to avoid ant predation or to use the ants to protect larvae against parasites and other predators (Pierce & Mead, 1981). The larvae produce a honey-like substance rich in sugars that the ants consume (Pierce, 1983). The presence of ants has been found to be is often highly correlated with the presence of butterflies (Guitierrez *et al.,* 2005; Jordano *et al.,* 1992; Ravenscroft, 1990), and females select sites with the presence of ants to lay their eggs (Jordano *et al.,* 1992). *Lasius niger,* one of the most frequently recorded mutualists, often shows a preference for habitats with relatively high soil humidity (Guitierrez *et al.,* 2005). In dry areas without *L. niger, L. alienus* is utilised instead (Ravenscroft & Warren, 1996). Ants are largely absent from dense *C. vulgaris* heathlands (>80% cover) and butterflies also do not occur in these areas (Ravenscroft, 1992). Presence of *Erica* (both *E. cinerea* and *E. tetralix*) in humid heath is usually well associated with the butterfly in heathlands in the South of England.

Model Applicability

The habitat suitability model described below is appropriate for the south of England, and is specific to heathland environments. The minimum habitat area is 0.25 ha. The model has not been verified but is based on scientific literature about the habitat requirements of this species. The model's purpose is to compare measures of habitat suitability with measures of the condition of heathland habitat with CSM, and it should not be used in practice without additional verification work.

Model Description

The model is based on several habitat requirements of *P. argus*, including the presence of host plants, feeding opportunities for adult butterflies, three aspects of successional age, two measures of heathland structure and two measures of habitat structure. The final output score of the model is simply the sum of these constituent habitat variable scores.

V1. Presence of Hostplants. Host food plants for butterfly larvae are required to be present in abundance. The species has a wide diet (Thomas 1985b), so the combined cover can be used for this measure. Species are as follows: *Calluna vulgaris, Erica cinerea, Erica tetralix, Ulex* spp*., Genista anglica, Lotus corniculatus* (Thomas, 1985a; Thomas 1985b; Ravenscroft, 1990; Douglas, 2009). Optimum is considered to be >50%, with no maximum value, as this indicates where these species dominate the vegetation community.

V2. Adult Food plants. Adult butterflies require nectar as an energy source, and can use a variety of flowers, but *Erica* spp. in particular is strongly associated with nectar feeding (Mair et al., 2015), and is often correlated with butterfly density (Ravenscroft, 1990). Cover of these species can therefore indicate ideal habitat conditions for nectar feeding (Ravenscroft & Warren, 1996). A cover of greater than 20% is optimum, but 0% is still calculated as (partly) suitable habitat because of the ability to feed from other species.

V3. Pioneer heath. Butterflies nest and reproduce on short heather growth in heathland habitats, and pioneer heath is ideal (Thomas, 1985a; 1985b; Ravenscroft, 1990; Douglas, 2009). Additionally, larvae may specialise on young plants, particularly young ericaceous plants (Thomas, 1985a), Pioneer heath should therefore be the dominant growth age, with >20% optimum habitat. 0% is unsuitable because of lack of suitable breeding locations and feeding plants.

V4. Building and mature heath. This habitat structure is unsuitable for the butterfly, because it indicates aged areas of heath and likely poor feeding for larvae (Ravenscroft, 1990), and has been associated with low density of butterflies (Thomas, 1985b). However, some scattered areas are not likely to be very detrimental (Ravenscroft, 1990). Based on recorded associations, combined building and degenerate heath cover is estimated to be optimum at <20% cover, with <50% completely unsuitable if the vegetation height above 10cm.

Table IV.2 Attributes making up the Habitat Suitability Index for the Silver Studded Blue butterfly, *Plebejus argus*

Variable	Source	Assumption
Presence of Host Plants	Thomas, 1985a; Thomas 1985b; Ravenscroft, 1990; Douglas, 2009	Host food plants are present in abundance. Combined frequency of occurrence of *Calluna vulgaris, Erica cinerea, Erica tetralix, Ulex* spp., *Genista anglica*, *Lotus corniculatus* optimum >50%, no maximum value.
Adult food plants	Ravenscroft, 1990; Ravenscroft & Warren, 1996; Mair *et al.,* 2015.	*Erica* spp. strongly associated with nectar feeding for adult butterflies. Frequency >20% optimum, but 0% is still suitable habitat
Successional Age 1	Thomas, 1985a; Thomas 1985b; Ravenscroft, 1990; Douglas, 2009	Pioneer heath is dominant growth age, >20% optimum. 0% is unsuitable.
Successional Age 2	Ravenscroft, 1990; Thomas, 1985b	Combined building and degenerate heath <20% cover is optimum, <50% unsuitable if vegetation height above 10cm.
Successional Age 3	Thomas, 1985a; Thomas 1985b; Ravenscroft, 1990; Thomas & Harrison, 1992.	Vegetation height <10 cm optimum, <25cm is unsuitable. No minimum values.
Heath Structure	Schimel & Fartmann, 2014	Graminoid cover unsuitable above 50%. Optimum is below 30% (Heathland specific)
Heath Structure	Ravenscroft, 1990	*C. vulgaris* cover poor above 70% because of loss of heterogeneity, optimum 20-50%.
Structural Diversity - Bare Ground	Ravenscroft, 1990; Thomas, 1985a; Thomas 1985b; Dennis & Sparks, 2006	Bare ground patches are optimum from 10-15%, but are unsuitable above 25%. Areas with no bare ground are below optimum, but still suitable habitat.
Structural Diversity - Sheltering	Dennis, 2004; Dennis & Sparks, 2006; Schimel & Fartmann, 2014	Combined frequency of occurence of *Ulex* spp., *Pteridium aquilinum* and *Rubus* spp. is unsuitable below <5%, optimum at 10-20%, unsuitable above 30%. Tree cover 0% is optimum, above 10% is unsuitable.

V5. Vegetation height. Vegetation height is a key indicator of suitable nesting locations and appropriate structure of the habitat, with strong associations with the presence of butterflies (Thomas, 1985a; Thomas 1985b; Ravenscroft, 1990; Thomas & Harrison, 1992). Less than 10 cm is considered optimum. Areas above 15cm are usually considered unsuitable (Dennis, 2004), but some higher values might still be utilised for sheltering, so here a value <25cm is considered unsuitable. There are no minimum values for optimum condition, as vegetation as low as 3cm is frequently used (Thomas, 1985b).

V6. Graminoid Cover. This attribute is highly specific to heathlands, as butterflies also nest in grassland habitat. Grass encroachment and habitat modification is likely to degrade suitable habitat for this species (Schimel & Fartmann, 2014), so combined graminoid cover is considered unsuitable above 50%. The optimum is below 30% so that suitable habitat cover can be attributable to ericaceous or leguminous species.

V7. Calluna vulgaris *cover.* Despite being an important species for the butterfly, high cover of *C. vulgaris* is likely to result in a loss of structural and community diversity, leading to poor habitat suitability (Ravenscroft, 1990). Cover is therefore unsuitable above 70% with an optimum of 20-50%.

V8. Bare Ground Cover. Bare ground patches are often utilised by the butterfly (Thomas, 1985a; Dennis & Sparks, 2006), which favours frequently disturbed habitats (Thomas, 1985b; Ravenscroft, 1990). Therefore, cover is optimum from 10-15%, but also unsuitable above 25% because this likely indicates excessive amounts of disturbance to the habitat to the detriment of other attributes. Areas with no bare ground at all are below optimum, but still partly suitable habitat for the butterfly.

V9. Sheltering Cover. Adult butterflies are known to utilise shrub and scrub areas for breeding and sheltering during adverse weather conditions (Dennis, 2004; Dennis & Sparks, 2006). For optimum habitat, such areas should be available close to more typical habitat. Here combined cover of *Ulex* spp., *Pteridium aquilinum* and *Rubus* spp. (associated species, Dennis, 2004) is unsuitable below <5%, optimum at 10-20%, unsuitable above 30% where scrub may detrimentally affect other habitat requirements. Additionally, tree cover is detrimental (Schimel & Fartmann, 2014), so 0% is optimum, with above 10% cover unsuitable.

Model Relationships

Suitability Index (SI) graphs for habitat variables are presented here in figure IV.4. The suitability index (SI) is read directly from the graph, with 1.0 equalling perfectly suitable habitat and 0.0 indicating unsuitable habitat.

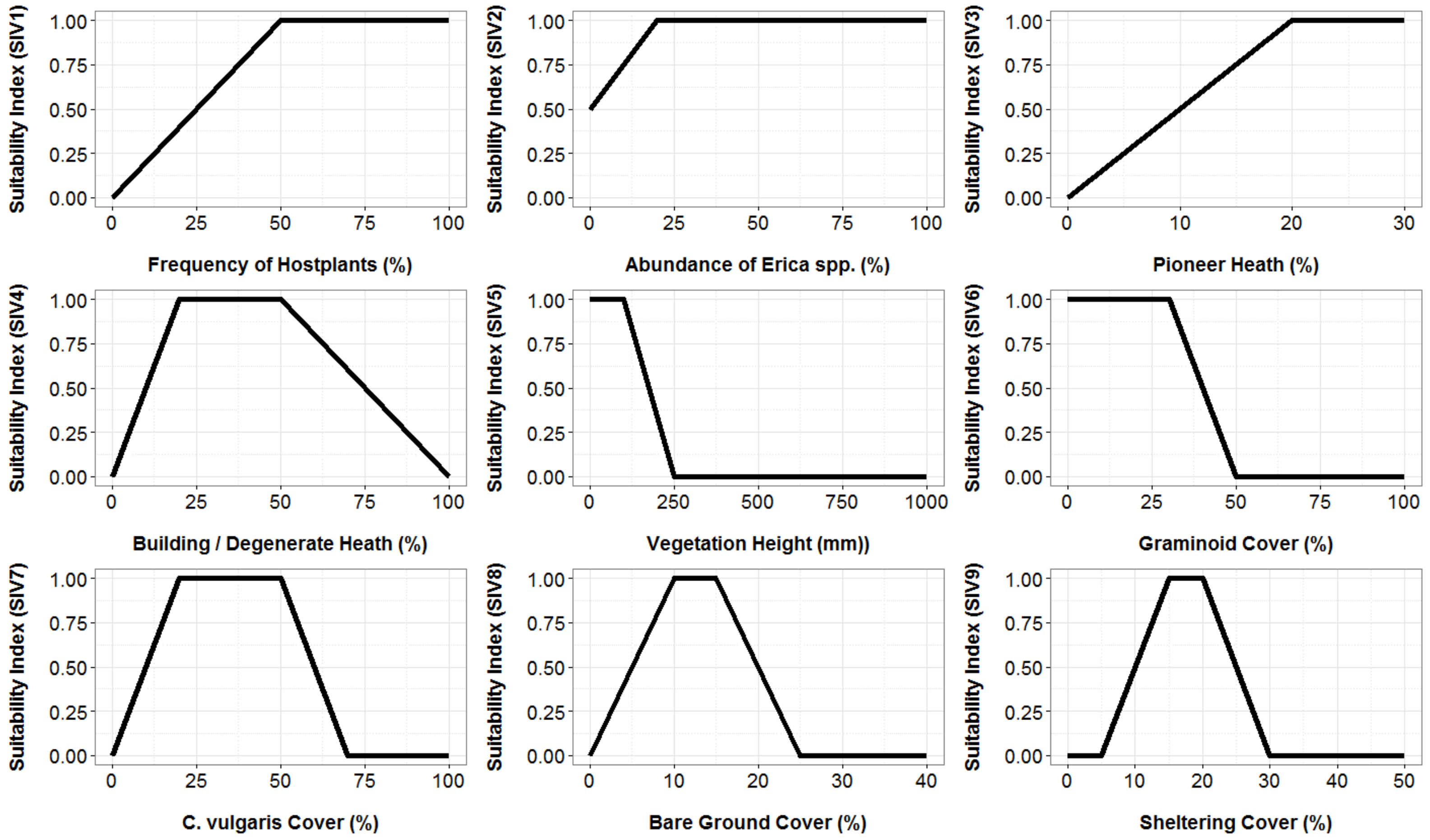

Fig IV.4. Habitat Suitability Graphs for the Silver-Studded Blue Butterfly.

IV.3 Dartford Warbler *Sylvia undata*.

Figure IV.5 Dartford Warbler *Sylvia undata*. Photograph by Paul Roberts, creative commons attribution

General Characteristics

An insectivorous bird restricted to southern and western Europe, and north-west Africa, the Dartford Warbler is a colourful and charismatic species. The European population has experienced continuous declines, and the species is classified as Near Threatened by the IUCN (Birdlife International, 2015). However, in the United Kingdom, at the northern edge of its distribution, it has recently increased and expanded its range (Wotton *et al.,* 2009). Dense scrub is the preferred habitat of *S. undata*; in the UK the species is restricted almost entirely to lowland heathland habitat (Bibby, 1979a). Dartford Warblers have typically been limited by severe winters in the UK (del Hoyo *et al.,* 2006). However, as a result of climate change, the suitability of habitat may become a more important limitation (Bradbury *et al.,* 2011; Catchpole & Phillips, 1992). The New Forest population is an important component of its UK distribution (Westerhoff & Tubbs, 1991). It is protected under the EC Birds Directive 1979 and the Wildlife and Countryside Act 1981.

Food Requirements

Dartford warblers are exclusively insectivorous and are the only resident terrestrial birds in Britain with such an ecology (Bibby, 1979a). The year-round requirement to feed on insects is believed to cause the species' reliance on heathland habitat. Although territories of the warbler include large areas of heathland (Bibby & Tubbs, 1975), studies have shown that the majority of feeding takes place on gorse stands, with ericaceous shrubs only a secondary feeding area (Bibby, 1979a). Gorse stands also usually feature a higher density of invertebrate prey, and are used year-round (Bibby, 1979a). Cover of snow in winter is a significant limitation to the warbler, as food cannot be found in these conditions. Birds frequently take any insect prey available them, and most frequently feed on spiders and caterpillars (Bibby, 1979a). Bare ground may also be an important feature for feeding (van den Berg *et al.,* 2001), because of both its importance for invertebrates and the ease of foraging for warblers, which have been observed using these areas (Bibby, 1977; Zamora et al., 1992). The species also avoids areas of dense woodland cover (van den Berg *et al.,* 2001; Jiguet & Williamson 2013), although they make use of isolated trees (Bibby & Tubbs, 1975).

Reproduction and Nesting

S. undata favours nest sites in tall ericaceous shrubs or small gorse stands surrounded by heather, with surrounding territories typically featuring a high cover of gorse (*Ulex europaeus* or *U. gallii*), *Calluna vulgaris* and bell heather *Erica cinerea* (Bibby and Tubbs, 1975; Moore 1975; Bibby, 1979a, b; Catchpole and Phillips, 1992; van den Berg *et al.,* 2001; Murison *et al.,* 2007). Nests are generally placed below 1.5m in height (Bibby & Tubbs, 1975; Bibby, 1979b). Brood size and success has often been linked to the presence of gorse in the surrounding habitat, as a result of increased food availability (Bibby, 1979b). The most significant factor affecting reproductive success is thought to be adult mortality (Catchpole and Phillips, 1992).

Interspersion

S. undata is largely sedentary, although relatively little is known about its dispersal ability (van den Berg *et al.,* 2001). Territory size is thought to be about 2-3 ha (Bibby & Tubbs, 1975; Catchpole & Phillips, 1992). It is currently expanding in the UK (Wotton *et al.,* 2009), and may expand into less suitable habitat, at the cost of reduced reproductive success (Murison *et al.,* 2007).

Model Applicability

This model is limited to the south of England, and is specific to heathland habitats in Dorset and Hampshire. The model should be applied to areas of 2-3ha to assess features in an appropriate territory size. The model has not been verified in practice, but serves as a tool to compare differences with CSM procedure.

Model Description

The model is based on several habitat requirements of *S. undata*, primarily those that are required for feeding. There is considerable overlap between the feeding and nesting requirements. The final output score of the model is simply the sum of these constituent habitat variable scores.

V1. Ericaceous cover. *S. undata* is almost entirely exclusive to heathland habitats in the UK., and use ericaceous cover for nesting in addition to occasional feeding (Bibby; 1979a; 1979b; Catchpole & Phillips, 1992; Van den Berg *et al*., 2001; Murison *et al*., 2007). Therefore >40% cover is optimal to show where these conditions predominate. Areas without Ericaceous cover are considered unsuitable.

V2. Ulex spp. cover. This cover type is the primary feeding area for Dartford Warblers (Bibby; 1979a; 1979b; Van den Berg *et al*., 2001; Murison *et al*., 2007), as it provides better year-round supply of invertebrates compared to ericaceous cover (Bibby, 1979a). 20-80% cover is considered optimal, but large gorse stands without heather are poor habitat. Territories are sometimes documented without gorse (Bibby & Tubbs, 1975), so areas without gorse are sub-optimal habitat rather than completely unsuitable.

V3. Average Vegetation Height. Generally, vegetation of medium depth (approx. 350mm) is preferred in territories of the warbler, but a range of depths occur (Bibby & Tubbs, 1975). Habitat selection in New Forest heaths appears to select deeper vegetation cover than recording elsewhere (Bibby & Tubbs, 1975), often up to 600mm. Areas of short heather <200mm are subsequently considered unsuitable, with 350mm - 1m cover optimal. Higher gorse heights are often utilised (Catchpole & Phillips, 1992; Van den Berg *et al*., 2001). Above 2m the habitat is unsuitable for foraging, and seems to be selected against in territory selection (Bibby, 1979b).

V4. Bare ground Cover. Small patches of bare ground are useful for foraging based on observations of the species (Bibby, 1977; Zamora et al., 1992). 5-15% is considered optimum, with no cover at all sub-optimal, as it may not be an essential habitat requirement, but it frequently associated with density of the birds (Van den Berg *et al*., 2001). Above 25% is unsuitable because this amount indicates excessive exposure and disturbance to the habitat.

V5. Tree Cover. Isolated trees are used as perches but any increased density is detrimental to feeding and nesting requirements, and is indicative of potential habitat change (Van den Berg *et al.,* 2001). Isolated, <5% cover of mature trees in all areas is considered optimal, with immature trees or saplings not recorded as detrimental. Above 10% cover is recorded as unsuitable habitat.

Model Relationships

Suitability Index (SI) graphs for habitat variables are presented here in figure A-3. The suitability index (SI) is read directly from the graph, with 1.0 equalling perfectly suitable habitat and 0.0 indicating unsuitable habitat.

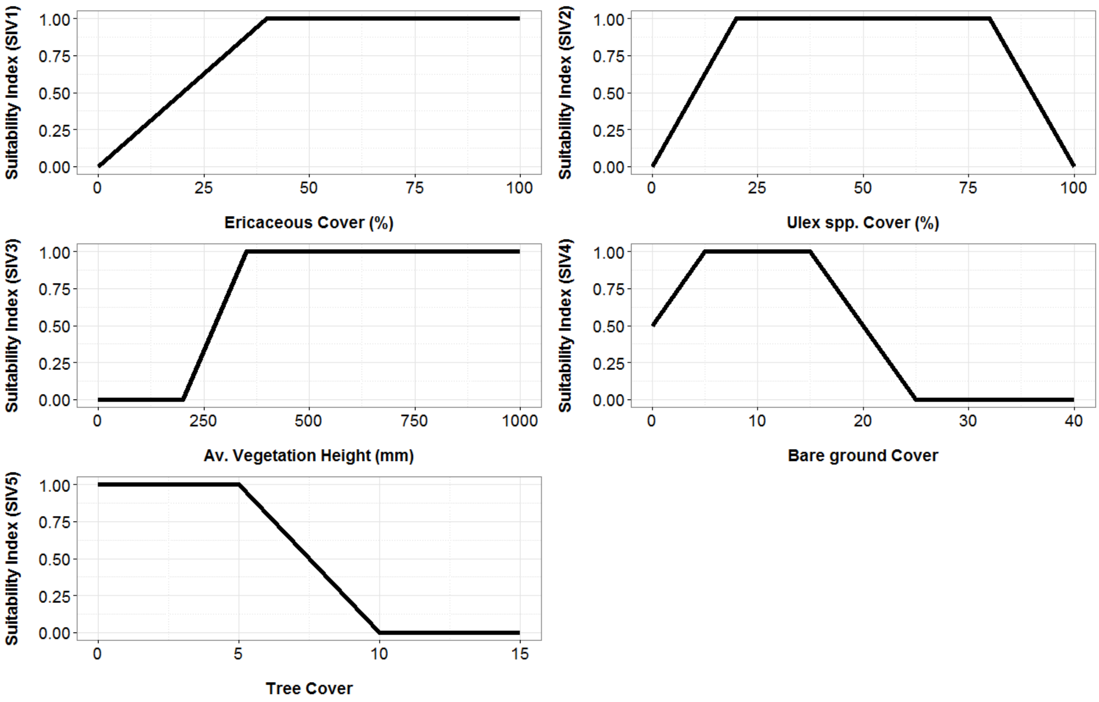

Figure IV.6 Habitat Suitability Graphs for the Dartford warbler *Sylvia undata*.

Table IV.3 Attributes making up the Habitat Suitability Index for the Dartford Warbler *Sylvia undata*

Variable	Source	Assumption
Ericaceous Cover	Bibby; 1979a; 1979b; Catchpole & Phillips, 1992; Van den Berg *et al*., 2001; Murison *et al*., 2007	Species is almost exclusive to heathland habitats in the UK. >40% cover Optimal as heather cover is an important nesting and feeding source. Areas without Ericaceous cover are unsuitable.
Ulex spp. Cover	Bibby; 1979a; 1979b; Van den Berg *et al*., 2001; Murison *et al*., 2007	The primary feeding area for Dartford Warblers. 20-80% cover is optimal, but large stands without heather are poor habitat. Territories are sometimes documented without gorse, so areas without gorse are sub-optimal habitat rather than unsuitable.
Average Vegetation Height	Bibby; 1979a; 1979b; Westerhoff & Tubbs, 1991; Catchpole & Phillips, 1992; Murison *et al*., 2007	Areas of short heather <200mm are unsuitable. 350mm - 1m cover is optimal. Above 2m the habitat is unsuitable for foraging.
Bare Ground	Van den Berg *et al.,* 2001	Small patches of bare ground are useful for foraging. 5-15% optimum, no cover is sub-optimal but above 25% is unsuitable because of excessive exposure.
Tree Cover	Van den Berg *et al.,* 2001	Isolated, <5% cover of mature trees in all areas is optimal. Immature trees or saplings are not detrimental

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Appendix V – Supplementary Information and statistics for Chapter 4

Spearman's Correlation, Tukey HSD results and Levene's Test results (homogeneity of variance assumption) are provided in the digital supplement to this thesis (Chapter 4 supplementary materials)

V. 1 MDS Output

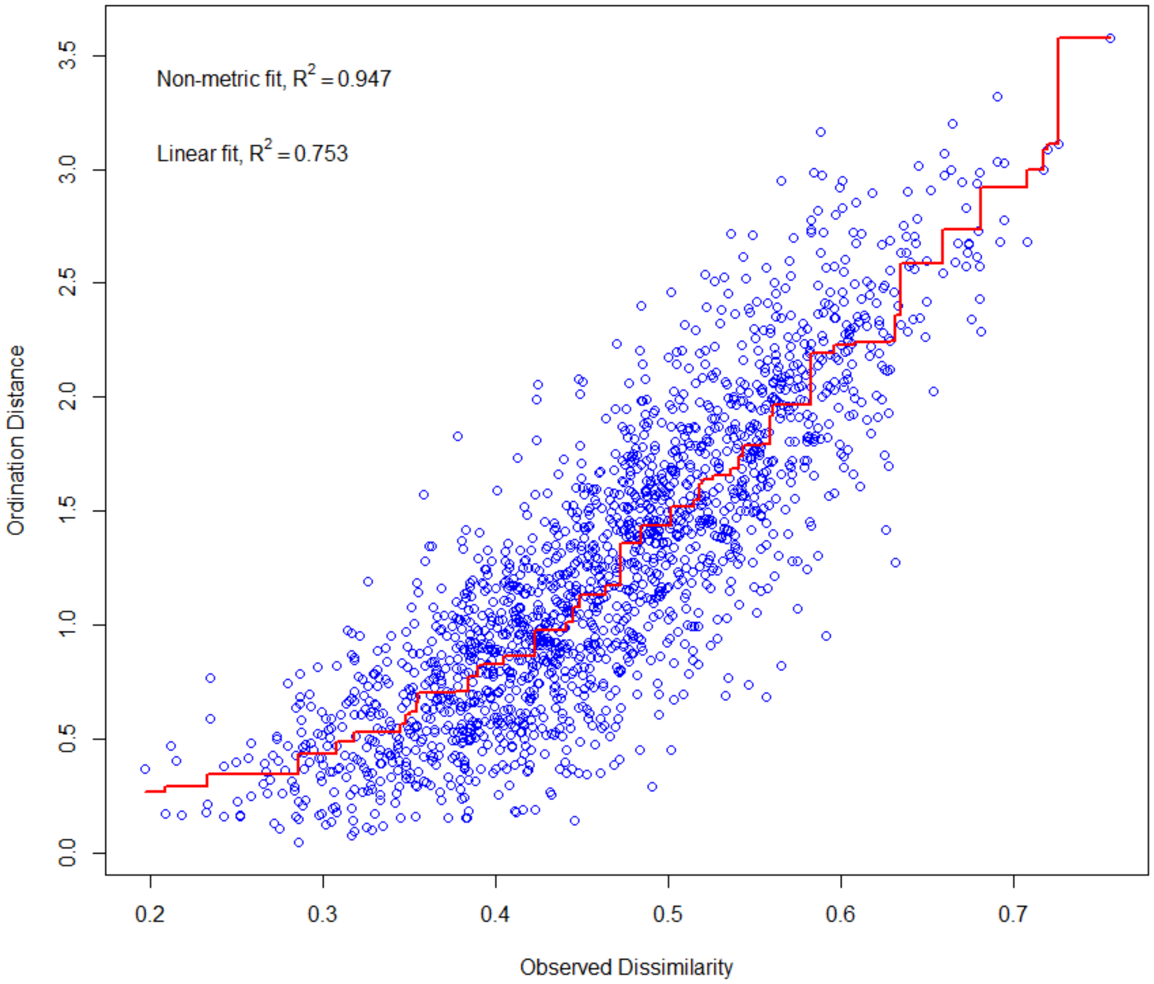

Figure V.1 Shepard diagram showing the output from MDS analysis. Stress = 0.23

Table V.1 Adonis output for MDS analysis

Test	Df	F	R^2	*p*
Restoration Stage	5	1.6506	0.13257	0.004995
Residuals	55		0.86743	

V.2 CCA Output

Table V.2 Partitioning of mean squared contingency coefficient:

	Inertia	Proportion
Total	1.6535	1.0000
Constrained	0.2473	0.1496
Unconstrained	1.4062	0.8504

Table V.3. Eigenvalues and canonical coefficients of explanatory variables for CCA axes.

Axis	CCA 1	CCA 2	CCA 3	CCA 4	CCA 5
Eigenvalue	0.1148	0.05812	0.02909	0.02556	0.01972
Proportion explained	0.4642	0.23503	0.11764	0.10335	0.07976
Cumulative proportion	0.4642	0.69925	0.81689	0.92024	1
Correlation Coefficient					
Surface Water	-0.72942	-0.73094	-0.64649	-2.2888	5.60756
Trampling	0.906627	-4.29151	1.22579	-2.30944	-0.0961
% Browsed	2.788257	2.28774	-2.69912	-0.31961	1.96285
Soil Moisture	-1.112154	-1.05159	-3.7279	-3.3458	-1.5638
Total N	0.726004	-1.07666	-0.07307	5.18804	3.17206

Table V.4 Multicollinearity test for CCA explanatory variables using Variance Inflation Factors (VIF)

Variable	VIF
Surface Water	1.38176
Trampling	1.549047
% Browsed	1.59636
Soil Moisture	1.937635
Total N	1.913708

VIF > 10 is unsuitable

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Appendix VI – Supplementary Information and Statistics for Chapter V.

VI. 1 M21 Vegetation Community

As described in Rodwell (1991), typical valley mires consist of a zonation, with a M6 central community progressing to M29 and M21a, with surrounding land often M16 or other heathland types. The M21 community are typical vegetation for lowland valley mires. This habitat type represents the most frequently found vegetation in New Forest valley mires, and as one of the reasons for European protection under the Habitats Directive, is the focus of the condition work here. M21 habitats consist of carpets of *Sphagna* with some herbaceous and shrubby plants, forming lawns, and in wetter areas, hummocks and hollows. Usually *S. papillosum* is dominant, but other species are frequent, with *S. magellanicum* often found in the New Forest. Relatively few other species of moss are present. *Eriophorum angustifolium* is almost constant, as with *Narthecium ossifragum*. *Molinia caerulea* is common but sparse in wetter areas. In some communities of this vegetation type, *Rhynchospora alba* is very frequent. The ericaceous species *Calluna vulgaris* and *Erica tetralix* are often present but usually small and patchy. *Drosera*, particularly *D. rotundifolia*, is very common, with *Polygala* and *Potentilla* occassional. *Myrica gale* has been noted to be frequent in some sub-communities that are present in the New Forest.

Soils found under M21 should be permanently waterlogged, acidic and peaty. The habitat is typical for valley mires with a high water table. The peat is usually not very deep (20-150cm). The high water table provides protection from the grazing and burning practices found in heathland that surrounds this community, as both of these management activities cause severe damage and do not play any role in maintaining the habitat. Drainage of land supporting this community is particularly damaging, as it removes protection from these land practices provided by the high water table and opens the habitat to further colonisation by *Molinia* and in turn other woody and shrubby species (this may be the natural succession of these communities but is greatly accelerated by drainage). Wetness of soil and poor nutrients are considered to limit the growth and abundance of *Molinia*, which is nethertheless found with high frequency. In addition, modern stands of this community are often highly fragmented.

JNCC guidelines note that shift from one habitat type to another occurs, and is often considered unfavourable, in particular the shift from M21 to M25. This community includes a wide variation of vegetation and habitat, which features an overwhelmingly abundant extent of *Molinia*. This habitat is found frequently in southern British lowlands. It is also relatively common among other community types which it is known to colonise. Increase in this community is often related to particular changes or circumstances in local areas, and is almost always deleterious for species richness. *Juncus* sp., especially *J. acutiflorus* and *J. effusus*, are frequent, except for especially acidic soils where *J. effusus* is absent (Rodwell, 1991).

VI. 2 References

Rodwell, J.S., 1991. Editor. *British Plant Communities, Volume 2: Mires and Heaths*. Cambridge University Press, Cambridge.

JNCC, 2004. Common Standards Monitoring Guidance for Lowland Wetland Habitats. Issue date February 2004. Joint Nature Conservation Committee, Peterborough.

VI. 3 Statistics

Table VI.1 Tukey HSD output for *Z. vivipara* HSI

	diff	lwr	upr	p.adj
HLS-Degraded	-0.16583	-0.35185	0.020179	0.106557
2006-9-Degraded	-0.05833	-0.24435	0.127679	0.93792
Life 3-Degraded	0.079833	-0.10618	0.265846	0.800794
Life 2-Degraded	0.0675	-0.11851	0.253512	0.89025
Favourable-Degraded	-0.0015	-0.18751	0.184512	1
2006-9-HLS	0.1075	-0.07851	0.293512	0.533017
Life 3-HLS	0.245667	0.059654	0.431679	0.00346
Life 2-HLS	0.233333	0.047321	0.419346	0.006282
Favourable-HLS	0.164333	-0.02168	0.350346	0.112316
Life 3-2006-9	0.138167	-0.04785	0.324179	0.257218
Life 2-2006-9	0.125833	-0.06018	0.311846	0.356583
Favourable-2006-9	0.056833	-0.12918	0.242846	0.944158
Life 2-Life 3	-0.01233	-0.19835	0.173679	0.999958
Favourable-Life 3	-0.08133	-0.26735	0.104679	0.788143
Favourable-Life 2	-0.069	-0.25501	0.117012	0.880875

Table VI.2 Levene's Test output for Mire CSM attributes

variable	Df	F	*p*
Bare Ground	5	2.8973	0.02176
Trees / Shrubs	5	0.5538	0.7347
M21 Sphagnum A	5	2.0795	0.08213
M21 Sphagnum B	5	0.8	0.5546
M21 Associated Species	5	1.0009	0.4261
M21 Negative Species A	5	1.8956	0.1103
M21 Negative Species B	5	1.7509	0.1388
Total Score	5	0.413	0.8377

Table VI.3 Levene's Test for HSI scores

variable	Df	F	*p*
Z. vivipara	5	0.92	0.4752
S. grossum	5	0.6183	0.6864
N. arquata	5	0.5153	0.7635

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Appendix VII – Habitat Suitability Indices for Mire Habitats

VII.1 Viviparous lizard *Zootoca vivipara*

Figure VII.1 Viviparous Lizard *Zootoca vivipara*. Photograph by Ben Mitchell, creative commons attribution.

General Characteristics

The viviparous lizard *Zootoca vivipara* (formerly *Lacerta vivipara*) is a widespread, ground-dwelling species, found across much of northern Europe and Asia (Agasyan *et al.,* 2010). Scattered populations in Britain and Ireland are found in a variety of habitats, including wet and dry heathland, moorland and most types of grassland (Edgar *et al*., 2010). A favoured habitat of this species is peat bogs and humid heathlands (Peñalver-Alcázar *et al*., 2016; Lorenzon *et al.,* 1999; Pilorge, 1987) and they are frequently encountered in New Forest mires (pers. obs.). In the northern part of its range the species is viviparous, giving birth to between three to 11 young, whereas in the southern areas, including Spain, southern France and Italy, it is oviparous, laying one to 13 eggs (Agasyan *et al.,* 2010). Usually one clutch is produced annually in British populations (Avery, 1975). It is considered to be an indicator for other reptiles (JNCC, 2010a), for which it is a frequent prey species (Reading & Jofré, 2013), and it is also a food source for raptors (Steen *et al.,* 2011). Although listed as Least Concern by the IUCN, its global population is declining and some regional extinctions have occurred (Agasyan *et al.,* 2010). The UK sub-population is also declining, especially in southern and south-eastern England (JNCC, 2010a). It is protected here under the Wildlife and Countryside Act, 1981 and as a Biodiversity Action Plan species (JNCC, 2010a).

General Habitat Requirements

Pilorge (1987) found that high global vegetation cover, vegetation height up to 30cm, high proportions of dwarf shrubs, south facing exposure, high soil humidity, and possibly increased depth of the earth layer were associated with increased densities of lizard populations and these findings are supported by recent studies (Reading and Jofré, 2016). Other reports indicate that the highest densities tend to be found in damp or wet areas, especially where abundant grass tussocks are present to provide food, shelter, basking and hibernation sites (Edgar *et al.,* 2010). Exposure to sunlight, structural diversity and adequate cover, are considered essential requirements (Edgar *et al*., 2010). Studies examining the association of *Z. vivipara* with vegetation species found correlations with *Molinia caerulea*, *Erica tetralix*, *Ulex minor* and a variety of grass species (Dent & Spellerberg, 1987; Reading and Jofré, 2016).

Food and Foraging Habitat

Zootoca vivipara is almost entirely insectivorous, with an unspecialised diet (Avery, 1966), typically following a searching foraging strategy (Avery, 1962). Heathland and bog habitats are believed to be preferentially chosen by this species because of the high density of invertebrates for feeding (Farren *et al.,* 2010). Vegetation structure is also believed to be an important contributor to prey availability (Brady & Phillips, 2006), with heterogeneous height and structure of the local habitat considered to indicate good condition (Edgar *et al.,* 2010). Abundant grass tussocks are also thought to provide good habitat for feeding, in addition to other requirements (Edgar *et al.,* 2010). Lizards will not feed when their body temperature is low, so temperature requirements are also especially important in relatively cold areas such as Britain (Avery, 1971; these requirements are discussed further in the special considerations section).

Reproduction and Nesting Habitat

Female lizards require a sheltered, humid microhabitat to give birth (Edgar *et al.,* 2010), and new-born lizards have been recorded in mosaics of *Molinia caerulea, Calluna vulgaris* and scattered *Ulex* spp bushes (Jofré & Reading, 2012). Breeding has also been documented in afforested areas where the ground cover consisted almost completely of *M. caerulea* (Jofré & Reading, 2012). Gravid females have reduced mobility, because of the increased gestation time owing to the viviparous lifestyle, so increased shelter requirements are important to avoid predation (Bauwens & Thoen, 1981). These requirements likely include a high vegetation height, such as that recorded by Reading and Jofré (2016), Stumpel and Ven der Werf (2012) and Pilorge (1987), in addition to an extensive and humid ground cover layer (Pilorge, 1987; Reading and Jofré, 2016).

Interspersion

Z vivipara dispersal distances are small at less than 100 m (Beebee and Griffiths, 2000) and unsuitable habitat is considered to provide an effective barrier, making the species particularly vulnerable to population fragmentation (Boudjemadi, Lecomte and Clobert, 1999; Farren *et al.,* 2010). Therefore, habitat structure is important on relatively small scales. Humid habitats with ample cover are optimal for dispersal (Zajitschek *et al*., 2012).

Special Considerations

Special considerations include basking areas, sheltering and hibernation requirements, low levels of disturbance and high humidity. Basking areas are generally considered to be necessary (Brady & Phillips, 2006), but areas of bare ground are not required because the lizards utilise vertical vegetation structure, such as grass tussocks (Edgar *et al*., 2010) and have lower temperature requirements than other lacertids (Dent & Spellerberg, 1987). In some areas, lizards avoid bare ground patches, but this is not a consistent finding (Peñalver-Alcázar *et al*., 2016). *Z. vivipara* avoids structurally uniform vegetation, whether closed or open (Edgar *et al.,* 2010). Heterogeneous structure in heath and bog habitats provides lizards with suitable shade and shelter, and raised areas of peat, litter and moss provide burrow-rich matrix for shelter and hibernation, while also providing suitable areas for basking (Farren *et al*., 2010). Lizards are particularly reliant on shelter and burrows, which offer protection against cold wet weather, because of the climatic conditions of its range (Pilorge, 1987; Farren et al., 2010). Sheltering habitat is also important to avoid predation (Bauwens & Thoen, 1981). Strong associations have been found with leaf litter cover and moss cover (Reading and Jofré, 2016).

Hibernation can take place in a wide range of places but is not always underground, so lizards are vulnerable to winter disturbance to this habitat (Brady & Phillips, 2006). Studies of grazed areas show that populations were 3-5 times higher in un-grazed areas compared to grazed areas (Strijbosch, 2002; Wallis de Vries et al., 2013; Reading & Jofré, 2016), and this may be attributable to direct disturbance in addition to poor structural diversity in heavily grazed habitats. Reptiles are often restricted to small patches of vegetation types that may be preferentially grazed, making them particularly vulnerable (Edgar *et al*., 2010). *Z. vivipara* is often associated with high soil humidity and water requirements are known to affect growth rates even when adequate prey and heat are present (Lorenzon *et al.,* 1999), and some European populations have also been recorded in association with humid or wet habitats (Covaciu-Markov et al., 2008). Therefore, some measurement of habitat humidity is likely necessary to indicate preferred habitat. In bog habitats, this requirement is likely to be met in most conditions, unless drying has been severe. *Z. vivipara* was associated with sites which had a predominance of *E. tetralix*, one species with high humidity requirements, but also including *M. caerulea* (Dent & Spellerberg, 1987; Reading & Jofré, 2016).

Model Applicability

This model is geographically limited to the south of England, and is specific to heathland and mire habitats in Dorset and Hampshire. The model is most effective when applied to areas of 2-3ha to assess features in an appropriate scale for the range for individual lizards. The model has not been verified in practice, but serves as a tool to compare differences in the habitat score with CSM scores.

Model Description

This model is based on several habitat requirements of *Zootoca vivipara*, including feeding requirements, nesting requirements and those for hibernation and shelter. Important contributions to habitat suitability include conditions suitable for behavioural thermoregulation, reproduction, support of prey species and for shelter or protection from predators and unfavourable weather conditions (Dent & Spellerberg, 1987). The final output score of the model is the sum of the constituent habitat variable scores, giving an overall account of suitability.

V1. Vegetation Height. Together with structural variability this measurement forms a measure of heterogeneity in the habitat, which is required for feeding, nesting and temperature requirements. High densities of the species have been found in heather that was 25-35cm tall in studies by Reading & Jofré (2016), and with vegetation heights of 30-35cm by Pilorge (1987). Vegetation heights greater than this have also been found to be suitable (Stumpel and Ven der Werf, 2012). Very low vegetation typically results in lower densities of lizards and reduces dispersal ability in the habitat (Zajitschek *et al*., 2012). Vegetation height above 35cm is therefore considered to be optimum habitat, while measurements below 20cm indicative of grazing or other disturbances and considered unsuitable.

V2. Structural Variability. An additional measure of variability, which is very important for this species. Measurements are calculated by subtracting the minimum vegetation height from the maximum vegetation height recorded at each plot. Plot scales are used because this is the scale most likely to affect the species. Measurements below 20cm are considered unsuitable with limited variation in vegetation height, but above 30 cm is considered suitable. This is a relatively narrow band because measurements below 20cm are a relatively high variation to consider homogenous, but reflects appropriate differences when compared to total vegetation height.

V3. Molinia cover. Cover of *Molinia caerulea* has been shown to be associated with population density in several studies of *Zootoca vivipara* (Dent & Spellerberg, 1987; Edgar *et al*., 2010; Stumpel and Ven der Werf, 2012; Reading and Jofré, 2016). It is included as a measure of suitability because of this association and because it affects feeding, shelter and dispersal requirements for the species. Cover greater than 20% is considered to be suitable, as at this level of abundance tussocks of vegetation can form. There is no maximum cover value because this habitat is well utilised by the species.

V4. Presence of E. tetralix. *E. tetralix* is considered to be a good indicator of humid conditions as it is associated with wet habitats (Bannister, 1966), and could be used as a proxy for humidity measurements. An alternative is to use *Sphagnum,* but this species will not necessarily be found in all suitable habitat, and *E. tetralix* has also been recorded in association with high densities of lizard populations (Dent & Spellerberg, 1987). As the only purpose of this measurement is as an indicator, frequency of occurrence is used as the measurement, with >10% frequency considered to indicate appropriate humidity.

V5. Ground Layer. Combined cover of Moss, Lichen and Leaf litter is important for nesting and sheltering during cold periods. High densities have been associated with litter cover (Reading & Jofré, 2016) and moss and lichen provide humid microclimates for reptiles (Edgar *et al.,* 2010; Pilorge, 1987). Areas above 15% cover are considered suitable, with cover above 15% optimum, but not at the expense of shrub or grass canopy cover.

V6. Grazing Indications. Disturbance is likely to affect habitat structure in addition to the direct effects of disturbance on lizard populations. As several studies have found deleterious effects of grazing on lizard populations (Reading & Jofré, 2012; Stumpel and Ven der Werf, 2012; Reading & Jofré, 2016), the optimum measure is 0% of the habitat recorded as grazed. Light grazing may be less detrimental than heavy grazing, so up to 10% cover is considered partly suitable, with measures in excess of 10% unsuitable for lizards.

Table VII.1 Attributes making up the Habitat Suitability Index for the viviparous lizard *Zootoca vivipara*

Variable	Source	Assumption
Vegetation Height	Pilorge, 1987; Zajitschek *et al.*, 2012; Jofré & Reading, 2012; Reading and Jofré, 2016	Cover must be at least of medium height, with above 350 mm optimal. Below 200mm is too short to provide adequate cover and may be indicative of disturbance.
Vegetation Height Variability	Brady & Phillips, 2006; Edgar et al., 2010; Jofré & Reading, 2012	Variability below 200mm is considered unsuitable with limited variation in vegetation height, but above 300mm is considered optimum, with significant difference in vegetation height.
Molinia cover	Dent & Spellerberg, 1987; Edgar et al., 2010; Stumpel and Ven der Werf, 2012; Reading and Jofré, 2016	Greater than >20% cover and tussock formation are suitable habitat types. Absence of *Molinia* is considered unsuitable because of strong associations between this grass type and *Z. vivipara*.
Presence of *E. tetralix* (indicator of humidity)	Pilorge, 1987; Dent & Spellerberg, 1987; Lorenzon *et al.,* 1999; Covaciu-Markov et al., 2008; Peñalver-Alcázar *et al.*, 2016	>10% frequency considered to indicate appropriate humidity.
Ground Layer	Pilorge, 1987; Reading and Jofré, 2016	Combined cover of Moss, Lichen and Leaf litter is important for nesting and sheltering during cold periods. >15% optimal, but not at the expense of shrub or grass canopy cover
Grazing Indication	Reading & Jofré, 2012; Stumpel and Ven der Werf, 2012	Disturbance likely to affect habitat in addition to direct effects of grazing on habitat structure. Above 10% of plots cover grazed is unsuitable with an optimum of 0.

Model Relationships

Suitability Index (SI) graphs for habitat variables are presented in Fig X, below. The suitability index (SI) is read directly from the graph, with 1.0 equalling perfectly suitable habitat and 0.0 indicating unsuitable habitat.

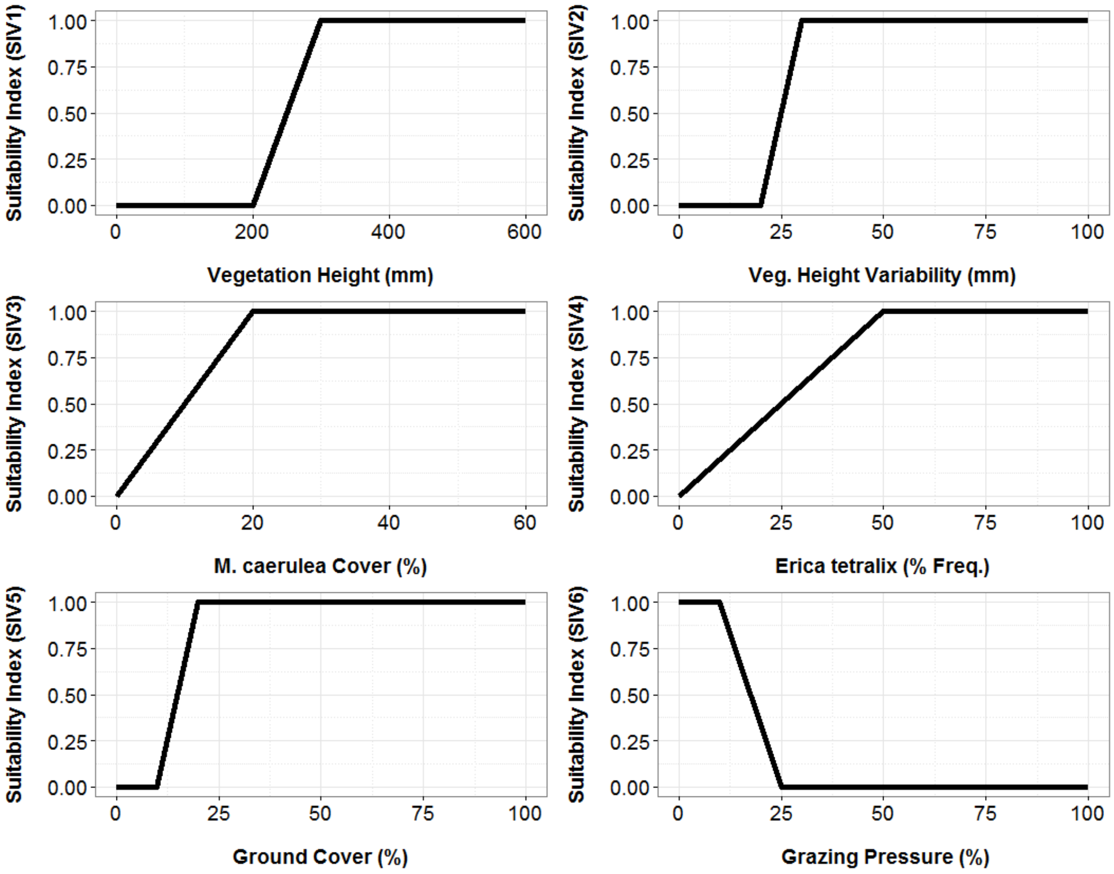

Figure VII.2 Habitat Suitability Graphs for the Viviparous lizard *Zootoca vivipara*.

VII.2 Large Marsh Grasshopper *Stethophyma grossum*

Figure VII.3 Large Marsh Grasshopper *Stethophyma grossum*, used under creative commons attribution.

General Characteristics

The Large marsh grasshopper *Stethophyma grossum* is Britain's largest native grasshopper (Haes & Harding, 1997), and globally is distributed throughout Europe and Siberia (Keller *et al.,* 2013; Keller *et al.,* 2012). It is believed to be entirely restricted to restricted to wetland habitat, and is often used as an indicator species for extensively managed wetlands in Europe (Keller *et al.,* 2012). In Germany, *S. grossum* is ranked as an endangered species and is considered susceptible to extinction because of large-scale drainage and fragmentation of wetland habitats, with similar situations described in other European countries (Bönsel & Sonneck, 2011). Although the UK population is unlikely to represent a substantial proportion of the global population, edge of range characteristics may enhance their value in the wider conservation of the species and it may serve as a charismatic species for wetland habitats (BAP). *S. grossum* shows a long-term trend of decline in the UK, with an 85% decline in abundance observed from 1979-2004 (JNCC, 2010b; Beckmann *et al.,* 2015). The species is now locally extinct in the Thames Valley, East Anglia, Surrey and Somerset, the UK population now confined to the New Forest and Dorset (JNCC, 2010b). It is protected under the Wildlife and Countryside Act, 1981 and is a UK Biodiversity Action Plan species.

Currently the species is restricted to acid bogs with grass tussocks on lowland heath in Britain, though it formerly occurred more widely (JNCC, 2010b). The New Forest is a stronghold for this species which typically occurs in undisturbed bogs (Haes & Harding, 1997; Sutton, 2015). Populations appear to fluctuate from year to year, but it is not known whether this is due to recording inconsistencies or to the natural fluctuations (Sutton, 2015). Decisive habitat characteristics are considered to be a continuously high atmospheric humidity near the ground, seasonally wet soil and heterogeneous vertical structure of the vegetation (Sonneck et al. 2008). Habitats in Europe are often wet grassland over peat substrates (Bönsel & Sonneck, 2011), but the grasshopper is most abundant in bogs (Sergeev, 2011). Although more widespread in Europe, it is still considered to be a moderate habitat specialist (Schouten *et al.,* 2007).

Food and Foraging Habitat

In Western Europe *S. grossum* has a cosmopolitan diet, feeding on grasses of the genera *Alopecurus, Agrostis, Avena, Bromus, Brachypodium, Calamagrostis, Cynosurus, Holcus, Poa, Lolium, Molinia, Phleum, Festuca, Dactylis, Elytrigia, Deschampsia, Phragmites, Triticum, and Trisetum*, and also on sedges and rushes (Savitysky, 2010). In Estonia and Russia's Volga region the species appears to specialise on sedges, especially *C. panicea, C. nigra*, and *C. lasiocarpa*, but captive individuals also consumed grasses and herbaceous plants (Savitysky, 2010).

Reproduction and Nesting Habitat

The reproduction of *S. grossum* determines much of its habitat requirements. Soil moisture changes between autumn and spring are particularly important (Marzelli, 1997). Larvae hatch where the soil has a high moisture content during the egg development stage (often accompanied by temporary flooding or water saturated soil), but after hatching wet soil conditions have negative consequences for the development of nymphs (Marzelli, 1997). This means that heterogeneous habitat conditions are advantageous for the grasshopper, and temporary flooded sites are ideal habitat in contrast to permanent flooding (Marzelli, 1997). The nymphs usually occupy the same habitat as the adults once they have hatched (Bönsel & Sonneck, 2011)

Interspersion

The few investigations of movement of *S. grossum* have suggested a low dispersal behaviour with covered distances of 250 m on average and of 1,500 m at most (Bönsel &

Sonneck, 2011). Knowledge on its dispersal habitat and potential is scarce, but compared with other grasshoppers, *S. grossum* is a good flier with observed flight distances of up to 41 m (Keller *et al*. 2013). *S. grossum* mainly uses its reproductive habitat as its preferred dispersal habitat, so movement is enhanced by maintaining and restoring the species' reproductive habitat, to preserve existing populations and to enhance or re-establish dispersal (Keller *et al*., 2013)

Special Considerations

Forest habitat often causes fragmentation of *S. grossum*'s habitats (Bönsel & Sonneck, 2011), but at low abundance tree and scrub species are unlikely to have much effect on habitat suitability.

Model Applicability

This model is geographically limited to the south of England, and is specific to wetland habitats in Dorset and Hampshire. The model should be applied to areas of 2-3ha to assess features in an appropriate scale for the range for individual grasshoppers. The model has not been verified in practice, but serves as a tool to compare differences in the habitat score with CSM scores.

Model Description

This model is based on several habitat requirements of *Stethophyma grossum*. The final output score of the model is the sum of the constituent habitat variable scores, giving an overall account of suitability. Scoring for individual habitat variables is described below, and Table X shows the assumptions and literature sources for these variables.

V1. Graminoid Cover. This measure records the availability of food sources for the grasshopper. Because of its cosmopolitan diet (Savitysky, 2010), total graminoid cover is used. Below 20% cover is considered unsuitable because grasses may have patchy distribution at these cover scores, but over 30% is considered optimum because at this cover graminoids become a significant part of the vegetation community.

V2. Vegetation Height Variability. Heterogenous habitat structure is ideal for the grasshopper. At least 300mm of variation is considered necessary in order to have adequate structural diversity, as vegetation height can appear somewhat homogenous at lower measures.

V3. % Cover of Surface Water. The cover of surface water is used to indicate appropriate humidity for the grasshopper. 15-30% cover is considered optimum across the plots, but at more than 50% plots are too wet (and therefore unsuitable), affecting survivability of nymphs.

V4. Sphagnum cover. Sphagnum is used as a further indicator for soil humidity here, in addition to its own association with *S. grossum* in UK habitats. More than 50% occurance is considered optimum, indicating bog habitat is widespread.

Table VII.2 Attributes making up the Habitat Suitability Index for the Large Marsh Grasshopper *Stethophyma grossum*

Variable	Source	Assumption
Graminoid Cover	Sutton, 2015; Bönsel & Sonneck, 2011	Food source and habitat indicator. Below 20% cover is unsuitable, but over 30% optimum
Vegetation Height Variability	Sonneck et al., 2008; Bönsel & Sonneck, 2011	At least 30mm in variation in order to have adequate structural diversity for the survival of grasshoppers.
% Cover of Surface Water	Sonneck et al., 2008; Bönsel & Sonneck, 2011	15-30% optimum for nesting, >50% unsuitable for larval growth. Measured at the plot scale.
Sphagnum Cover	Marzelli, 1997; Sonneck *et al.,* 2008; Sutton, 2015	Sphagnum is used both as an indicator for soil humidity and as a positive habitat association by itself. More than 50% occurance is optimum.

Model Relationships

Suitability Index (SI) graphs for habitat variables are presented below in Fig X. The suitability index (SI) is read directly from the graph, with 1.0 equalling perfectly suitable habitat and 0.0 indicating unsuitable habitat.

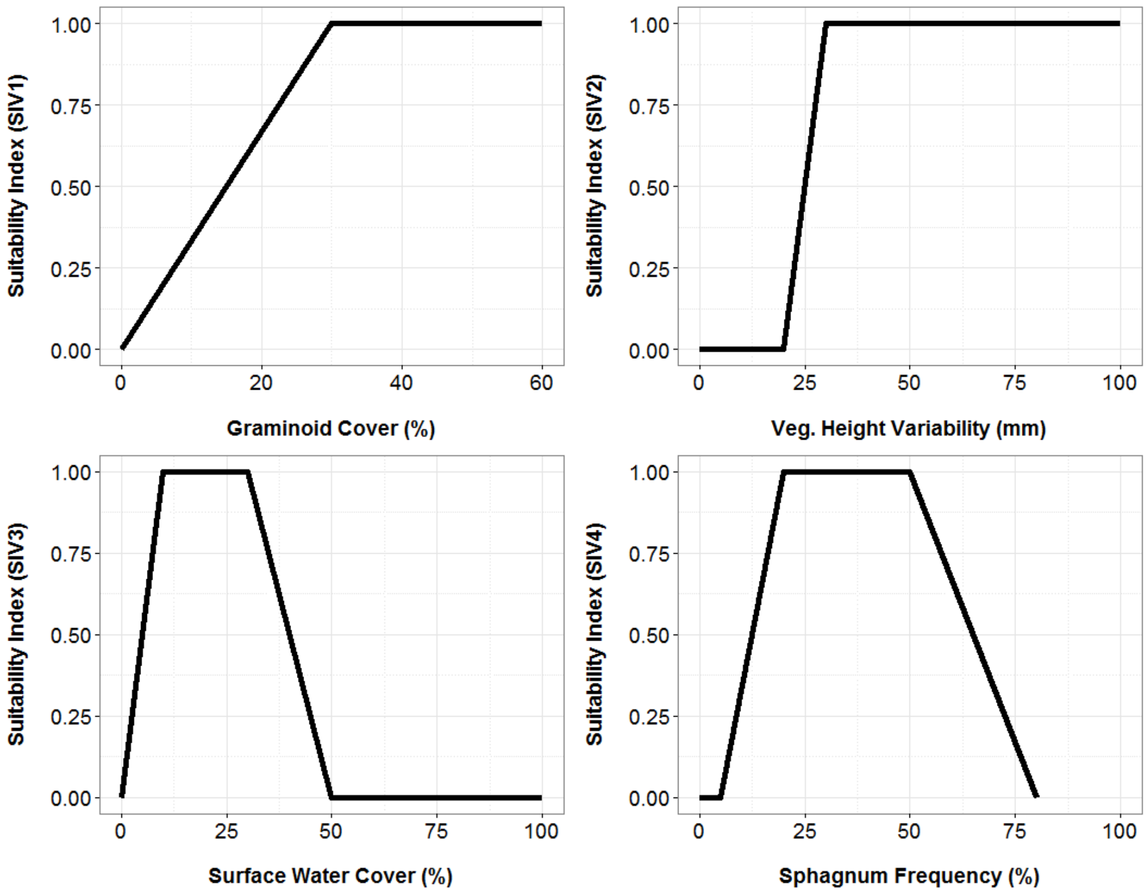

Fig VII.4 Habitat Suitability Graphs for the Large Marsh Grasshopper *Stethophyma grossum.*

VII.3 Eurasian Curlew *Numenius arquarta*

Figure VII.5 Eurasian Curlew *Numenius arquarta.* Photograph by Tony Sutton, used under creative commons attribution.

General Characteristics

Eurasian Curlews *Numenius arquarta* (subsequently referred to as 'Curlews') are large, distinctive wading birds, and are widespread across temperate areas of Europe and Asia. The species was previously common but is now severely declining in several populations, including Ireland (O'Connell, 2011) and the United Kingdom (Eaton *et al.,* 2012). Because of these declines, it is listed as Near Threatened under IUCN criteria (BirdLife International 2013). The breeding curlew population in the UK is of international importance, possibly representing as much as 39% of the estimated European population (Grant *et al*., 1999), and the species is protected under the EC Birds Directive 1979 and the Wildlife and Countryside Act, 1981; it is also a UK Biodiversity Action Plan species. Threats to the species are because of agricultural intensification and drainage (del Hoyo *et al.,* 1996), although Asian populations have experienced declines where farmland has been abandoned because of subsequent increases in vegetation height (Johnsgard 1981).

Curlews are migratory (del Hoyo *et al.,* 1996), and in the UK move from feeding in coastal areas in winter to inland breeding areas during summer months. Therefore, specific habitat requirements are distinct at different times. Curlews require a mosaic of nesting and wet feeding habitat in preferred territories, and high breeding densities occur on moorland-wetland complexes (Henderson *et al.,* 1999). The New Forest is an important breeding area (Tubbs & Tubbs, 1996), and high densities have been recorded in bogs and bog-wet heath mosaics, with only low numbers found on dry heath (Tubbs & Tubbs, 1996). Summer abundances in Ireland have been strongly associated with peatlands with a high density of vegetation cover, with a preference for wet conditions and areas with low and sparse grass–dwarf shrub vegetation (Bracken *et al.*, 2008). Dallimer *et al.* (2012) found that intensive grass, vegetated boundaries and wet features affected Curlew density on farmland and moorland habitat.

Food and Foraging Habitat

During the breeding season, curlews principally consume annelid worms and terrestrial insects (del Hoyo et al. 1996) although spiders (Johnsgard 1981), berries and seeds, as well as occasionally small fish, amphibians, lizards, young birds and small rodents are also taken (del Hoyo et al. 1996). The long curved bill is typically used to probe for food in soft soils and sediments (O'Connell, 2011). The provision of damp ground within their territory, where invertebrate prey are both abundant and accessible, is particularly important (Berg, 1992a), with stronger habitat preferences shown early in the breeding season compared to later (Berg, 1992b). Henderson *et al.* (2002) found that most occupied areas contained areas of standing water, but unoccupied areas did not.

Reproduction and Nesting Habitat

Breeding Curlew are typically associated with bogs and heather moorland (Henderson *et al*., 2002), but on a more local scale birds nest in a shallow depression on the ground or on a mound, either in the open or in the cover of grass or sedge (del Hoyo et al. 1996; Birdlife International, 2013). In bog habitats, grass tussocks are often selected (O'Connell, 2011). Bog habitat is particularly useful in comparison to dry grasslands, with higher densities of young in these areas (Berg 1992a). For young birds, studies sometimes find low or sparse vegetation structure to be important requirements, since chicks are often found were they can easily walk and find prey (Berg, 1992b). Associations with wet conditions were also found by Stillman and Brown (1993), who also note a preference for tall ericaceous species. Although this contrasts with Berg's findings, other studies also show associations between curlews and tall vegetation within their territories, which may provide shelter for young chicks (Valkama *et al.,* 1998). Such differences reflect the need for structural diversity in the habitat at scales within territory sizes, and studies which include measurements of this characteristic have shown that curlews prefer heterogeneous areas (Pearce-Higgins & Grant, 2006).

Interspersion

Because curlews are a migratory species, they are highly capable of dispersal. Few characteristics will negatively affect interspersion on territory-sized indices. Curlew territories are relatively large, although Tubbs and Tubbs (1996) recorded five (successfully) breeding pairs in a small three-hectare bog. Berg (1992a) suggests that territory size is dependent on habitat quality.

Model Applicability

This model is geographically limited to the south of England, and is specific to heathland and mire habitats in Dorset and Hampshire. The model should be applied to a relatively large scale (at a minimum, plot scale) to assess features in an appropriate scale for the range for individual birds. The model has not been verified in practice, but serves as a tool to compare differences in the habitat score with CSM scores.

Model Description

V1. Graminoid Cover. An important measure, as high grass cover preferred (Robson, 1998; Pearce-Higgens and Grant, 2006; O'Connell, 2011; Dallimer *et al.,* 2012). Grass–dwarf shrub vegetation is also associated (Bracken *et al.*, 2008). 0-30% considered suitable, with cover >50% considered detrimental because of the loss of other associated species.

V2. Tall Heather Cover. Associated with heather cover and heathland-mire matrices (Tillman & Brown, 1994; Henderson *et al*. 2002; Pearce-Higgens and Grant, 2006). Greater than 30% cover considered suitable.

V3. Presence of Water. This measure shows the provision of damp ground within their territory, where invertebrate prey are both abundant and accessible. 5-10 % optimum, as this indicate enough wet areas to feed. Areas greater than 50% lack suitable nesting sites and are too wet.

V4. Vegetation Height Variability. Strong associations with heterogenous vegetation are an important feature of nesting habitat (Calladine *et al*., 2014; Pearce-Higgens and Grant, 2006*)*, with a combination of dense vegetation to hide and open vegetation to feed. <200 mm height is unsuitable, >300mm suitable, covering a range of vegetation heights.

V5. Average Vegetation Height. Curlew are strongly associated with peatlands with high horizontal density of vegetation cover (Bracke*n et al.,* 2008). Relatively flat areas are needed to nest while denser areas are necessary to feed and raise juveniles. Therefore moderate height measurements are necessary (>250-400mm optimum). >1000mm too high and unsuitable.

V6. Sphagnum Cover. Strong association with bogs, particularly in the New Forest (Tubbs & Tubbs, 1996). Sphagnum is also indicative of wet conditions for feeding. 20-50% optimum cover, >75% suboptimal because other nesting conditions are affected.

Table VII.3 Attributes making up the Habitat Suitability Index for the Curlew *Numenius arquata*

Variable	Source	Assumption
Graminoid Cover	Robson, 1998; Pearce-Higgens and Grant, 2006; O'Connell, 2011	Often associated (but not always) with high grass cover; also nests in farmland. 0-30% considered suitable, with cover >50% detrimental because of loss of other species
Tall heather cover	Stillman & Brown, 1994; Henderson *et al*. 2002; Pearce-Higgens and Grant, 2006	Associated with heath cover and heathland-mire environments. Greater than 30% cover suitable, indicating good cover.
Presence of water	Henderson et al. 2002	Shows provision of damp ground within their territory, where invertebrate prey are both abundant and accessible. 5-10 % optimum, areas greater than 50% lack suitable nesting sites and are too wet.
Vegetation Height Variability	Bracken *et al*., 2008; Pearce-Higgens and Grant, 2006; O'Connell, 2011	Heterogenous vegetation and important feature of nesting habitat, with a combination of dense vegetation to hide and open vegetation to feed. <200 mm height is unsuitable, >300mm suitable.
Average vegetation height	Valkama *et al.,* 1998	Moderate requirements necessary (>250-400mm optimum) as relatively flat areas required to nest and denser areas to feed and raise juveniles. >1000mm too high and unsuitable.
Sphagnum Cover	Tubbs & Tubbs, 1996	Strong association with bogs in the New Forest. Sphagnum indicative of wet conditions for feeding. 20-50% optimum cover, >75% suboptimal because other nesting conditions are affected.

Model Relationships

Suitability Index (SI) graphs for habitat variables are presented here. The suitability index (SI) is read directly from the graph, with 1.0 equalling perfectly suitable habitat and 0.0 indicating unsuitable habitat.

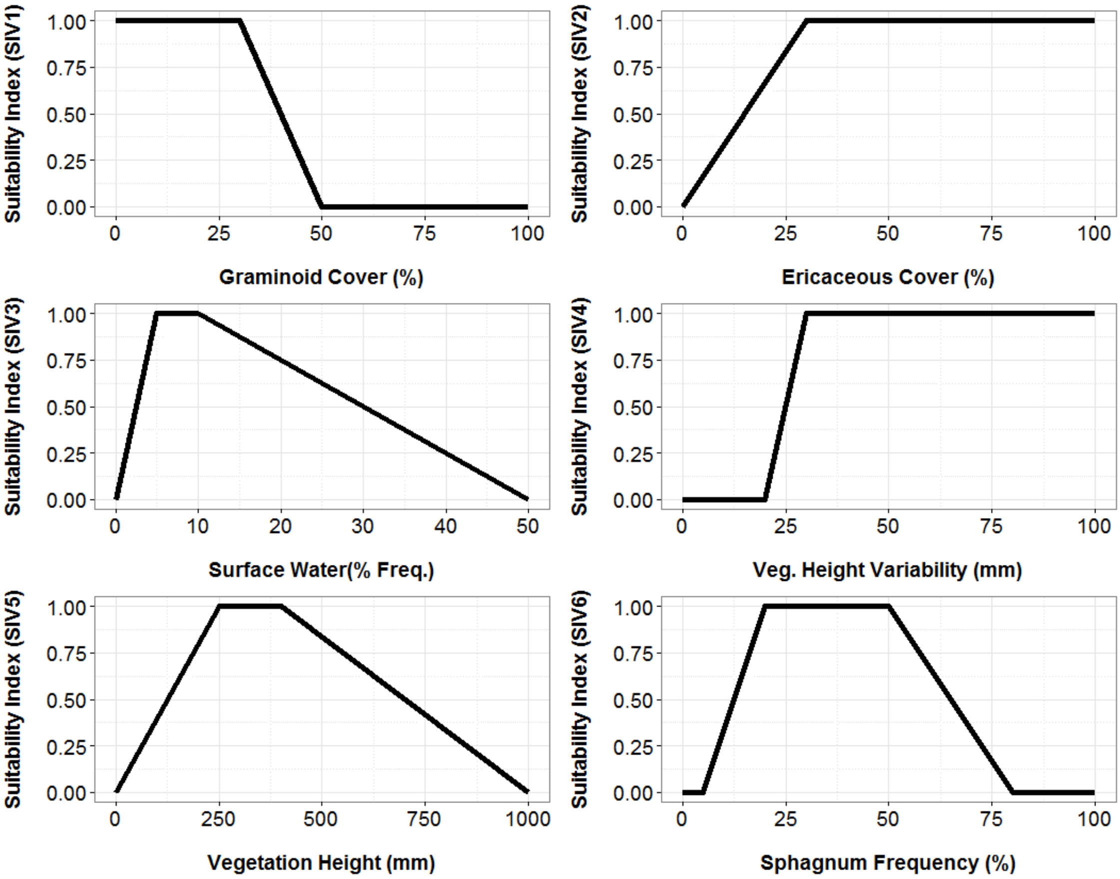

Figure VII.6 Suitability graphs for habitat requirements of the Curlew *Numenius arquata.*

VII.4 References

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Appendix VIII – Selected Site Photographs

VIII.1 Heathland Plots

VIII.1.1 Burnt Plots

Figure VIII.1 Heathland plot “Ibsley Common”, a 0 year Burn.

Figure VIII.2 “Ferny Knapp”, 1 year Burn.

Figure VIII.3 “Yew Tree Bottom”, 6 year Burn.

Figure VIII. 4 “Horse Shoe Earth”, 10 year Burn.

Figure VIII.5 “Holbury”, 20 year Burn.

VIII.1.2 Cut Plots

Figure VIII.6 “Broadley”, 0 year Cut.

Figure VIII.7 “Warren Hill”, 1 year Cut.

Figure VIII.8 “Black Heath”, 6 year Cut.

Figure VIII.9 “Turf Hill”, 10 year Cut.

Figure VII.10 “Blackwell Common”, 20 year Cut.

VIII.2 Mire Plots

Figure VIII.11 “Rans Hill”, a mire proposed for Future HLS restoration (“degraded” mire).

Figure VIII.12 “Site 93”, mire following HLS Restoration.

Figure VIII.13 “Site 96”, 2006-2009 Restoration.

Figure VIII. 14 “Markway Mire”, Life III Restoration.

Figure VIII.15 Two Life II Restoration locations. Top, “Redhill Bog”, is a very grassy mire. Bottom, “Milking Pound Bottom” still has high grass cover but shows more open conditions.

Figure VIII. 16 “Site 37”, a mire in “Favourable” condition. This was one of the smaller mires surveyed.

Figure VIII.17 *Drosera rotundifolia.* Photograph by author.

Figure VIII.18 A variety of *Sphagnum* species, including *S. papillosum, S. cuspidatum, S.magellanicum* and others, growing in a mixed species cushion. Photograph by author.