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The role of livestock grazing in the conservation of lowland heath

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Abstract

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THE ROLE OF LIVESTOCK GRAZING IN THE CONSERVATION OF LOWLAND HEATH

by Sophie Lake

Domestic livestock grazing is increasingly used as a conservation tool on lowland heaths. However, the impacts of livestock on vegetation and the implications of their behaviour are not fully understood. This study considered habitat selection and behaviour in seven livestock groups on lowland heaths in Dorset, southern England. The impact of livestock on wet heath and valley mire vegetation was assessed experimentally. The potential role of livestock grazing and trampling in the germination of eight characteristic wet heath and valley mire plant species was established, and the potential for livestock-facilitated regeneration from buried seed banks was explored.

Livestock showed non-random use of habitats, and generally preferentially selected habitats with a high cover of grass species. Use of dry heath varied according to the age and structure of heather *Calluna vulgaris.* There was seasonal variation in habitat selection, notably the increased use of wet heath and valley mire in late summer. However, foraging was not the only activity dictating habitat selection, and selection for resting locations also influenced overall habitat use.

Livestock grazing and trampling had a significant impact on vegetation structure and the cover of bare ground within three years. The impact on substrate microtopography, compaction and standing water was also significant. Changes in species composition were more subtle, and may take longer than three years to become apparent.

The emergence of selected wet heath and valley mire species was shown to be increased by simulated livestock trampling. Higher water levels increased emergence, but reduced canopy shading was not important in most species studied. Habitat surveys suggest that canopy shading may be important in the subsequent establishment and survival of these species.

Livestock trampling also has the potential to increase species emergence through exposure of buried seed banks. This was shown to result in species regeneration at sites where they were considered extinct in the above ground vegetation.

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1. Introduction

Grazing is increasingly being considered as a suitable management tool for maintaining and enhancing the favourable conservation status of lowland heathland in Britain (e.g. UK Steering Group, 1995; Michael, 1997). The frequently-stated rationale for this has been that livestock grazing played a key role in the development and maintenance of lowland heathland until its decline in the 20th century (e.g. English Nature, 1998; Small *et al.,* 1999, Oates & Tolhurst, 2000). Comparison between the biodiversity of the New Forest (an area remarkable in having a continuous grazing history) and un-grazed heathlands elsewhere supports the view that grazing may provide an ecologically sustainable management tool for heathland conservation (Byfield & Pearman, 1994). However, little is known either about traditional lowland heathland grazing practices in the UK, or about the effect of livestock presence on lowland heathland communities. There is a need to establish how and to what extent livestock presence may affect heathlands before grazing is further promoted as an ideal management tool. This thesis aims to provide some of the information necessary to allow this assessment to be made by examining the impact of livestock grazing on the heaths of Purbeck in southern England, with particular reference to wet heath and valley mire plant communities.

1.1 The heathland habitat

Heath, a loosely defined word meaning 'uncultivated land' (Rubel, 1914), has been applied to a diversity of vegetation types across Europe, including dry grass heath, lichen and moss heath (Gimingham, 1972), dry, prairie and subalpine meadow (Specht, 1979), dry heath, damp heath, valley mire and shingle heath (Rodwell, 1991). In this thesis, the term lowland heath is used to describe vegetation dominated by ericaceous dwarf-shrubs growing on acidic nutrient-poor mineral soils, together with related wet heath and valley mire vegetation, at altitudes of less than 250m (e.g. Webb, 1986).

Lowland heathlands occur along the Atlantic fringe of north-west Europe, extending north from Portugal along the Atlantic coast to Norway, and inland east to Germany. Characteristic lowland heathland communities are considered to have arisen under humid oceanic climates about 4000 years ago as a result of forest clearance by humans followed by use for grazing livestock, together with burning, peat digging and mineral extraction (Webb, 1998). This use prevented the regeneration of forest (Gimingham, 1972). Before human activity, a thin forest cover on poor soils allowed a heathland ground flora to develop (Webb & Haskins, 1980; Tubbs, 1997), which may have been maintained in canopy gaps by large wild herbivores (Van Wieren, 1989). Heathland once extended over several million hectares in Western Europe and was maintained by traditional practices until the twentieth century (Webb, 1998). These traditional forms of land use have been mostly lost together with all but 350,000 ha of Atlantic heathland (Webb, 1998). The development, human use and subsequent loss of European lowland heathland has been extensively described elsewhere (e.g Gimingham, Chapman & Webb, 1979; Gimingham, 1972; Webb, 1986; Tubbs, 1986; Heil & Aerts, 1993; Traynor, 1995; Allchin, 1997).

European lowland heathland is not intrinsically species rich, and its conservation value is in providing habitat for a suite of species that are largely restricted to heathland in the UK. The more high profile of these species include Dorset heath *Erica ciliaris*, marsh gentian *Gentiana pneumonanthe,* Dartford warbler *Sylvia undata*, woodlark *Lullula arborea*, nightjar *Caprimulgus europaeus*, sand lizard *Lacerta agilis* and smooth snake *Coronella austriaca*, all of which are listed in the IUCN red data books. In particular, the heaths of southern England are one of the most important habitats for invertebrates in the UK, supporting more than 50% of the British species in some orders. Examples of species whose conservation status is of particular concern include the heath grasshopper, *Chorthippus vagans*, large marsh grasshopper *Stethophyma grossum,* dingy mocha moth *Cyclophora pendularia,* southern damselfly *Coebagrion mercuriale*, and mottled bee-fly *Thyridanthrax fenestratus.*

The survival of most these species in the UK is highly dependent on the persistence of lowland heathland. Lowland heathlands are also valued for their contribution to the overall diversity of habitats within the landscape (e.g. Moore, 1962). The conservation value of lowland heath is recognised by a number of statutory site designations, including that of "habitats of community value" under the European "Habitats" directive on the conservation of natural habitats (Directive 92/43/CEE1997) and as Sites of Species Scientific Interest under national legislation (Section 28 of the UK Wildlife and Countryside Act, 1981 and the Countryside and Rights of Way Act, 2000). A conservation action plan for British lowland heathlands was written in 1995 (UK Steering Group, 1995) as part of the UK's commitment to the international Convention on Biological Diversity (1992), and was implemented through the National Lowland Heathland Programme and currently through the Tomorrow's Heathland

Heritage programme.

Heathlands are also valued as cultural landscapes. In some places (such as Dorset in southern England) whole landscape units comprising open heath and mire, semi-improved riverine meadows, saltings, scrub and woodland have persisted and are greatly valued. The open panorama associated with heathlands has been unfavourably described in the past as 'sandy, wild and barren country' (Defoe, 1724) and as 'hungry, sandy, barren wastes' (White, 1798). However, these viewpoints were from an agricultural perspective and in a time when considerably more 'free nature' (Naess, 1971) survived in Britain. Today the beauty of their bleak wildness is appreciated in a society where little that is wild remains (e.g.Webb, 1986; Gimingham and de Schmidt, 1983; Heil and Aerts, 1993).

Plate 1.1 Grip Heath, Purbeck, Dorset, looking out over Middlebere Lake.

In this chapter, existing knowledge of historical grazing practices on lowland heathlands in Britain is reviewed, supplemented with information from other European countries. Current knowledge of the impact of livestock presence on heathland vegetation is summarised, together with information from similar systems such as upland heathlands and lowland heathlands elsewhere in Europe. The aims of this thesis in filling some of the gaps in this knowledge are then outlined.

1.2 Historical context

1.2.1 THE ROLE OF GRAZING IN TRADITIONAL HEATHLAND USE

Although little is known about traditional heathland use in the UK, more is known in continental Europe, where heathland management may have been similar to that practised in Britain. Webb (1998) described a range of traditional management practices that were responsible for the maintenance of open heathland until at least the 20th century, including grazing, burning vegetation to promote growth for stock, cutting vegetation for livestock fodder and bedding, and cutting turf, peat and gorse for fuel. These activities occurred in combination, although the relative importance varied from region to region. In typical heathland management from the north-western European heathlands of Flanders, the Netherlands and Germany, sheep were herded on the heath for about six hours a day and confined to barns for the remainder of the time. Small irregular areas of heathland were burnt from time to time to provide a continuous supply of nutritious forage. In the winter, fodder for animals in the byre was supplemented with heather, which was cut on a 3-5 year cycle. Dried, crumbled peat and cut turves were used in byres to absorb excrement and were then spread on arable plots as fertiliser. In Denmark, a similar system was used. Sheep remained out all year but cattle were periodically tethered on the heaths and taken to their stalls at night where their dung was collected.

This system of land-use shows the key role that heathlands played in the cultural landscapes of Atlantic Europe (Diemont & Jansen, 1998). The landscape comprised areas of heathland linked by traditional farming practices to adjacent or nearby arable land and meadows, and livestock grazing played a fundamental role in transferring nutrients from the heath onto cultivated land. For example, settlements on the west coast of Norway were surrounded by infields used for pasture, hay meadows and arable plots, then by extensive outfield areas of heathland (Webb, 1998). Heathland was used in conjunction with these other habitat types, and the farming system would not have been sustainable using only one element of the landscape.

In Britain, less is known as the traditional system has not been practised within living memory. Heathlands were frequently common land (Rackham, 1986), and played a similar role in the cultural landscape to that described above. It is known that heaths were grazed and burnt, and turves and gorse *Ulex* spp. were cut (e.g. Cunningham, 1974; Tubbs, 1986; Traynor, 1995; Allchin, 1997). There is evidence in Dorset to suggest arable plots were fertilised with the dung of animals grazed on the heaths

(Webb, 1998). The type of livestock used tended to depend on locality. In East Anglia, where sheep were predominantly used, animals were grazed on the heaths during the day and folded onto arable land at night. Free-ranging cattle and ponies were traditionally used in areas such as the New Forest and Ashdown Forest, where pigs were also grazed. The New Forest in particular still retains its ancient practices and grazing rights (Tubbs, 1986). In such free-ranging grazing systems, the social behaviour of the livestock is a key factor in determining the pattern and structure of the vegetation, as the animals are not herded.

1.1.2 DECLINE OF LOWLAND HEATHLAND GRAZING IN EUROPE

Grazing management of lowland heaths declined throughout the 18th and 19th centuries as agricultural improvement changed traditional farming practices. Heathland became increasingly redundant, and in many cases was ploughed up. The decrease in grazing on surviving heathland was accompanied by a reduction in associated activities such as burning (e.g. Moore, 1962, Allchin, 1997). In the New Forest the number of cattle and ponies grazing during the late 19th century declined from around 2200 and 3000 respectively to around 1000 and 750 in 1940, although numbers have subsequently risen (Tubbs, 1968). In the 1860s over 750,000 sheep were grazed on the Lüneburg heaths of Germany reducing to 250,000 by 1900 and 25,000 by 1950 (Henke, 1982). Similarly, on the heathlands of the Monts d'Arrée in Brittany, the extensive sheep grazing known to have occurred around 1900 had ceased entirely by the 1970s (Lefeuvre, 1980). Grazing on some of the Breckland heaths of East Anglia had ceased by 1956 (Sheail, 1971; Crompton & Sheail, 1975) and in Pembrokeshire the heathland commons had fallen into disuse by the 1960s (Evans, 1989). Sheep grazing, which was widespread on the Purbeck heaths of Dorset, had stopped by the 1920s (Arnold, 1999), while dairy herds were kept on the heaths until land clearance by the military in the 1940s. Agricultural intensification reduced the use of heaths for livestock grazing throughout the 1940s and 1950s and grazing by farm workhorses declined with the advent of the tractor in the 1940s. Use for grazing by gypsy horses also declined through the 1940s and 1950s. Similar unquantified declines have been recorded from other heathlands across Europe (e.g. Ejlerson, 1992; de Beaulieu & Fichaut, 1992; Påhlsson & Danielsson, 1995; Rösberg, 1995).

However, grazing is now increasingly being reintroduced to sites as a conservation management tool (e.g. Wallis De Vries *et al*, 1998). A third of heathland National Nature Reserves in England were being conservation grazed by 1997 (Michael, 1997), and the number continues to rise. For example, in Dorset, where only two sites were still intermittently grazed before 1990, grazing has now been reinstated on at least 20 sites (I. Alexander, pers. comm.). In general, grazing management is less intensive than previously - livestock remain on the heath throughout the night and are generally free-ranging, although in some cases confined, by temporary fencing rather than shepherding, to particular areas of heathland sites for more limited time periods. Grazing is generally used in conjunction with other management practices such as cutting and burning, and less frequently peat, turf or soil stripping. However, in many places heathlands no longer form an integral part of a functioning cultural landscape, and their management (and ownership) is generally detached from that of the surrounding land.

1.1.3 HISTORICAL STOCKING RATES

Historical references to livestock numbers are few, and generally contain insufficient information to estimate stocking rate. In the 12th century sheep densities on 930ha of heathland at Lakenheath Warren were nearly 2.3 ha^{-1} (Crompton & Sheail, 1975). On 660ha of unenclosed heath in the Suffolk Sandlings sheep densities were 1.3 ha^{-1} in 1770 (Chadwick, 1982). Stocking rates in the New Forest (including woodland) in the late 19th century averaged 0.11 cattle and 0.18 ponies ha^{-1}, although numbers fluctuated widely (Tubbs, 1997). It is worth remembering that these stocking densities were aimed at maximising agricultural returns from the heaths, rather than meeting conservation objectives, and were influenced by market prices and the state of the pastoral economy (Tubbs, 1997). Agricultural returns were likely to have been different from those expected today - for example, wool was more important than meat in medieval sheep farming (Small, 1994), while an important role of sheep on the light East Anglian soils was manuring (e.g. Crompton & Sheail, 1975). This would have required less forage per capita than meat orientated production (as practised today). In addition, different welfare standards from those maintained today would have permitted stocking at densities now considered unacceptable.

The manorial system under which much of rural lowland Britain functioned from the medieval period until the 19th century meant that most heathlands were grazed in common by tenants of the manorial estates. The earliest historical sources are the Anglo-Saxon charters (7th - 11th centuries) - the legal means of land conveyance, which in some cases contain information on management practices (Rackham, 1986). The Domesday Book from the 11th century frequently records livestock numbers and types but categorises heathland under pasture, and pasture was only recorded from the south west of England. Manor court books (generally held by County Records Offices) contain some references to grazing rights on common land heaths between the 10th and 18th centuries, and have the advantage that later books are written in English rather than Latin.

A superficial investigation of the manorial court records for estates in Dorset revealed only limited useful information. For example it is known that 'the usual cattle and horses' could be pastured on commons and heaths of a farm near Bere Regis (Cunningham, 1974), but the numbers of animals this entailed is not specified. There is some evidence to suppose that common grazing rights were not static (Gasden, 1988), but may have changed between years. Where rights were quantified, the upper limit to commonable livestock numbers was generally that which each commoner could support throughout the winter off the common (those *couchant* and *levant* on the tenanted land). The numbers established by this method did not therefore provide an absolute right, and may have changed according to the amount of forage available and the commoners' circumstances. An example of grazing rights is given by Brocklehurst (1968) for Affpuddle in Dorset in 1573. Commoners were entitled to graze five sheep on the heath per acre of tenanted (i.e. non-heathland) land, and one cow or horse per two acres of tenanted land (equivalent to 11 sheep ha^{-1} and c1 cow/horse ha^{-1}).

It is not known to what extent common rights of pasturage were taken up. Records of fines imposed for exceeding stocking rates in both Affpuddle and other parishes in Dorset indicate that rights were over- rather than under-exploited in this area. Young (1771) commented that the 'Dorset wastelands were held in common, and so suffered from overstocking and overgrazing'. Given the relative stability of the agricultural system from the 10th century until the agricultural revolution at the end of the 17th century, it may be assumed that heathland stocking rates must have been sustainable or this particular aspect of the agricultural system could not have persisted - there was no means of supplementary feeding from outside of the immediate agricultural unit and (at least in theory) the animals grazed on the heath were the same individuals as those supported by the tenanted land in winter. However, it seems possible that stocking rates on heaths increased throughout the late 18th and 19th centuries as the areas of commonable heathland diminished through enclosure relative to the areas of tenanted land. Gasden (1988) suggests that this may have led to overstocking on commons in general.

More recent sources of information on heathland grazing include various agricultural and land surveys and reports, e.g. Claridge (1793), Abraham & Driver (1794), and Tavener (1937). Few of these contain actual heathland stocking densities, although Tavener (1937) gives densities of 7-21 sheep and 2-6 cattle per 100 acres of *total* land area in parishes on the Bagshot sands of south-east Dorset (equivalent to 0.32 – 0.95 sheep $ha^{-1}$and 0.01-0.27 cattle ha^{-1}). These are predominantly heathy areas, but although Tavener estimates that 20% of the land area is permanent grassland, he does not give a figure for the percentage comprising heathland. The Commons Registration Act (1965) provides a more recent source of information on stocking rates, as it required all common rights holders to register and quantify their grazing rights. However, Evans (1989) considers that, in Pembrokeshire, the figures in fact bear little relation to the actual rights practiced, and this may be the case elsewhere.

References to grazing season are more easily found than those for stocking density. For example, in Dorset sheep could be pastured on the Weld Estate heaths throughout the winter from St. Thomas' Day (December 21) until the middle of March[1.] Ponies remained on the New Forest heaths all year (Tubbs, 1991) as did farm workhorses (when not in use) on the Pembrokeshire heaths (Evans, 1989), while in Cornwall ponies grazing on Bodmin Moor during the summer may have been brought to the coastal heaths for the winter (S. Ford, pers. comm.). Arnold (1999) speculated that sheep were only grazed on the Purbeck heaths in Dorset in summer, as farmers also had rights of common on the nearby downland. There is insufficient information to establish whether the timing and duration of grazing seasons were relatively constant for similar heathland types or within particular areas.

1.1.4 ANIMAL TYPES

There is very limited information available about the types of animal used. It is known that sheep were generally grazed on the drier grassier heaths of East Anglia (e.g. Crompton & Sheail, 1975; Chadwick, 1982). Cattle, ponies and pigs were pastured on the New Forest (Tubbs, 1991), and cattle, sheep, goats, geese and horses were grazed on the Pembrokeshire heaths (Evans, 1989). Gypsy horses apparently had a considerable impact on the Pembrokeshire commons until the early 20th century, together with goats and geese. Donkeys are known to have been used on the Purbeck heaths for transporting clay and agricultural produce, and were also grazed on the heaths (Plate 1.2) (Arnold, 1999). In some areas changes in the type of livestock used may have occurred over long time scales - for example it is suggested that ponies were formerly important on coastal heaths in Cornwall (J. Harvey, pers. comm.), although cattle were subsequently more common.

Plate 1.2 Donkeys grazing heathland (possibly Middlebere) in Purbeck, Dorset (from an archive held by the Dorset Natural History and Archaeological Society).

Prior to 1750, few farm livestock breeds were recognised, although there were regional types reflecting local needs and environmental influences (Small, 1994). Most breeds were created in the late 18th –19th centuries and subsequent use of these reflected changes in market demands. It seems likely that the hardy breeds most similar to the regionally adapted types were used on lowland heathlands. It is known that New Forest ponies were grazed in the New Forest, and in his literature Hardy refers to the ‘heathcroppers’ of Dorset (e.g. in ‘Return of the Native’). These were small, tough ponies, which may have been part Exmoor (Arnold, 1999). Chadwick (1982) refers to the Southdown and Norfolk sheep being used in the Suffolk Sandlings, and notes that they were simply called ‘heath sheep’. Stevenson (1812) commented that he was unsure what breed the “...half-starved sheep suffering from foot rot which could kill them...” were in Dorset, but thought they might be Portland sheep. He described cattle on the Dorset heaths as a “...long-horned kind rather short in the leg, with white backs and bellies and dark spotted or brindled sides. A mixture of various breeds from the surrounding counties and the Channel Islands. Some were Devons...”. Arnold (1999) suggests the cattle seen in photographs taken at the end of the C19th on the Purbeck heaths were Shorthorn x Hereford crosses

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[1] DRO D10/181 Accounts of Court Barons in the estates of Thomas Weld 1783-1800

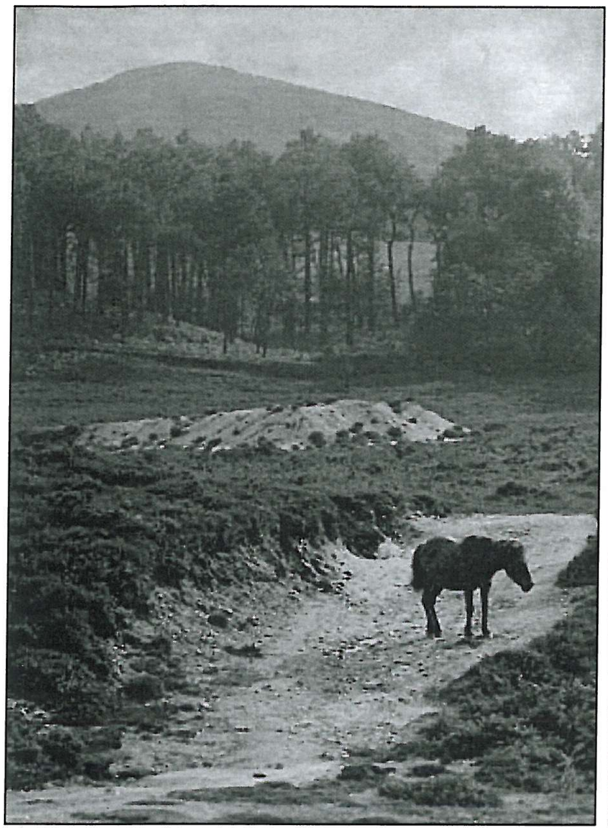


Plate 1.3 Heathland pony beneath Creech Hill in Purbeck, Dorset (from an archive held by the Dorset Natural History and Archaeological Society.

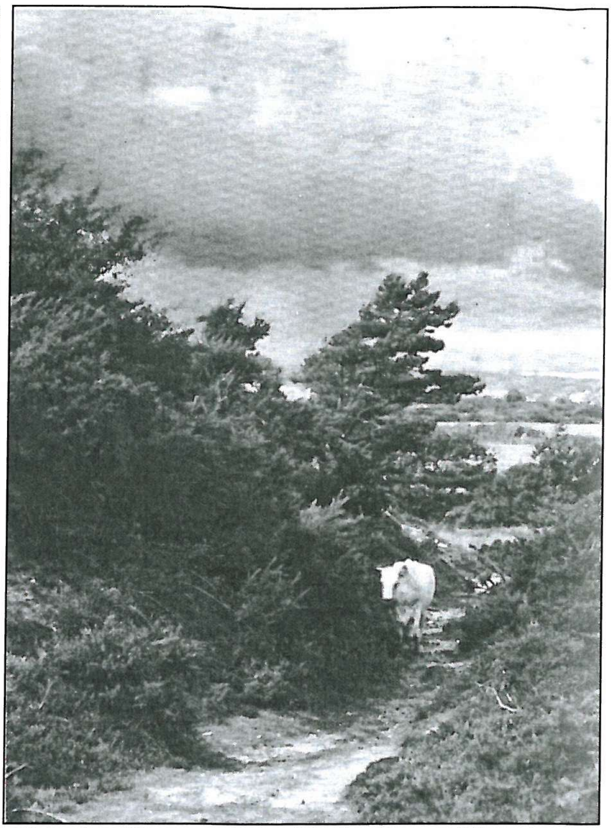


Plate 1.4 Cow on Arne Heath, Dorset (Dorset County Museum).

1.1.5 SHEPHERDING

Traditional practises in Europe included fairly intensive livestock management, with animals shepherded on the heaths, and often returned to the farms at night. It is probable that this also occurred in Britain. Evans (1989) found references to children shepherding stock across the unfenced Pembrokeshire commons in the late 16th and early 17th centuries. She also interviewed local farmers who described cattle and sheep being shepherded during the day and brought back to the farms at night, within living memory. Chadwick (1982) reports that sheep were shepherded on the Suffolk Sandlings, being walked out each morning from the farms and returned at night to be folded onto arable or improved grassland. Recent research (N.Webb, unpub. data) suggests that similar patterns of use may have occurred in Dorset. In interviews with local people who had memories of agricultural use of the Purbeck heaths in Dorset, Arnold (1999) established that cattlegrazed on the heaths were retained within enclosures made of earth banks topped with gorse. Manor court books in heathland areas in Dorset frequently refer to the employment of a shepherd within the parish. A census carried out in the parish of Corfe in 1795 lists four shepherds, who presumably worked on the adjacent heathland and downland (Legg, 1986). Legg also mentions records of labourers who kept single cows on the common and brought them back to the village daily for milking.

In summary, grazing was a fundamental part of traditional heathland use until its decline during the 20th century. While only a limited amount is known about how traditional management was practised in Britain, it is likely that this was similar to systems in Europe. In these, heathlands were an integral part of a larger cultural landscape, and were managed in conjunction with arable land and improved pasture. The limited information available about stocking rates and animal types used suggests considerable local variation.

1.3 Effects of livestock presence on heathland vegetation

Grazing management of lowland heathlands for nature conservation is generally aimed at maintaining open dwarf shrub vegetation with a high diversity of heath species while helping to control scrub and other unwanted species (Bullock & Pakeman, 1997). Although grazing has been reintroduced to large number of UK lowland heathland sites in the last few years (e.g. Davies, 1995; Bacon, 1998; Small *et al.*, 1999) there is surprisingly little research evaluating the success of grazing in achieving these aims (but see Byfield & Pearman, 1994; Bullock & Pakeman, 1997). While there is a significant body of work on the impacts of livestock activity on upland heathlands (e.g. Rawes & Welch, 1964; Grant, 1971; Grant, *et al* 1985; Armstrong & Milne, 1995; Welch, 1995; Hartley, 1997) much of this work assesses the impacts of grazing in an agricultural context.

Table 1.1 summarises the general trends observed in a number of studies of grazing impact on heathland throughout Europe. Livestock activity on heathlands is shown to reduce vegetation cover and re-establish earlier seral stages, creating an uneven aged mosaic in the dominant heathland species. Succession can be slowed, or the successional pathway changed, generally through a shift from ericaceous shrubs to graminoids (Welch, 1984, Grant *et al.*, 1985, Bullock & Pakeman, 1997; Hartley, 1997; Alonso *et al.*, 2001) and the inhibition of scrub growth (Marrs *et al.*, 1986, Tubbs, 1991 & 1997). The creation of bare ground can lead to an increase in leaching of soluble nutrients and help maintain low nutrient levels (Marrs, *et al.*, 1986), although dunging

Table 1.1 Summary of effects of introduction/exclusion of herbivores on heathland systems. 'Years' gives the time after grazing commenced/ceased. (Adapted from Bakker, 1998).

Introduction of herbivores	**Years**	**Habitat**	**Source**
1. Grasses decreased	?	Lowland grassland/heath	Bülow-Olsen, 1980
1. Locally increased species diversity, greater variation in vegetation type and structure, tree seedling prevented from developing	4	Lowland dry heath/grassland	Bakker *et al*, 1983
2. Grasses decreased, *Calluna* increased.	5	Lowland grassland/heath	Bokdam & Gleichman 1989
3. *Calluna* and grasses decreased.	10	Lowland dry heath	Van der Bilt, 1993
4. Increase in mire species, decrease in *Molinia*	3	Lowland wet heath/mire	Cox, 1998
5. Destabilisation of *Molinia*, (re)appearance of pioneer plants on wet heath	c8	Lowland wet heath	de Beaulieu, 1998
6. *Calluna* increased Invasive grasses and scrub did not decrease, initial increase in species diversity	10	Grass heath *Calluna* heath	Bokdam & Gleichman 2000
Exclusion of herbivores			
1. Increase in ericoids and tall grasses	25	Upland heath	Rawes, 1981
2. Increase in *Molinia*, *Juncus acutiflorus*		Lowland wet heath/mire	Clarke, 1988
3. Tree invasion	10	Lowland heath	Bokdam & Gleichman 1989
4. Decline in species associated with wet heath and related habitats	c60	Lowland wet heath & related habitats	Byfield & Pearman, 1994
4. Tall grasses become dominant	6	*Molinia* grassland	Grant *et al.*, 1996
Varied grazing pressure/season			
1. Increased grazing – increase in sedges, forbs, decrease in *Molinia*, dwarf shrubs	?	Lowland wet heath	Evans, 1989
2. Light grazing – ericoids, lichens & bryophytes increased; heavy – ericoids decreased, graminoids & forbs increased	20	Upland heath	Welch & Scott, 1995
3.. *Calluna*, *Eriophorum* decreased with increasing pressure, bare ground increased	11	Upland blanket bog	Grant *et al.*, 1996
4. Height of *Molinia caerulea* sward decreased, more open vegetation & increased species richness under higher grazing pressure.	5	*Molinia caerulea* dominated wet heath	Bullock & Pakeman, 1996
5. Decrease in gorse height, except in summer, when growth stimulated	2	Dry heath	Gallet & Roze, 2001

may lead to local enrichment (Bakker *et al.*, 1983). These processes may be expressed differently across the spectrum of heathland vegetation types from dry through humid and wet heath to valley mire. The impacts of livestock on dry and wet heath (including valley mire) are each reviewed here, before consideration is given to three areas of particular conservation interest: the control of invasive grasses and bracken, and scrub regeneration.

1.3.1 IMPACTS OF LIVESTOCK PRESENCE ON DRY HEATH COMMUNITIES

Much work has been done in the uplands concerning the effects of grazing on heather *Calluna vulgaris* moorland, and MacDonald (1990) gives a detailed review of these. In general, degenerate stands of heather are less tolerant of grazing than younger stands, and take longer to recover from overgrazing (Grant & Armstrong, 1993). A 40% removal of annual production over several years will damage even young vigorous heather sward (Grant *et al.*, 1978; Grant *et al.*, 1982). Removal of over 80% will lead to shoot death and a decline in stand density. Changes in *Calluna* cover and structure through grazing depend on grazing intensity. Welch (1984) showed that heather cover, height and biomass declined on upland sites with heavy livestock presence, but increased under lighter grazing. Light grazing may stimulate young growth while not adversely affecting mature or degenerate plants (e.g. Demopoulos, 1996), thus maximising structural diversity. Heavier grazing may damage both younger and older heather by repeated grazing of new shoots, uprooting of plants (Grant *et al.*, 1978) and trampling (Bayfield, 1979; Hester & Baillie, 1998). Absence of grazing may eventually lead to *Calluna* degeneration and scrub and tree encroachment (Hester *et al.,* 1991). The effects of grazing also depend on soil type and soil moisture content: in the uplands Welch (1986) found that *Calluna* was less damaged by grazing on dry soils than wetter ones, while in Brittany Gallet and Roze (2001) found that humid heaths were more sensitive to grazing than dry heaths. The effects of grazing on interactions between *Calluna* and the other characteristic dwarf shrub species, bell heather *Erica cinerea* and cross-leaved heath *E. tetralix,* have not been so well studied. *Erica* spp. tend to be grazed only lightly (Bannister, 1966; Rose *et al.*, 1996) or avoided altogether (Putman, 1987; Tubbs, 1991) and are therefore less vulnerable to grazing (although Gallet & Roze (2001) suggest that Dorset heath *Erica ciliaris*, found on wetter heaths, may be vulnerable to sheep grazing). However, changes in species cover may not reflect this. *Calluna* can respond to grazing by assuming a prostrate growth form, so increasing cover, whereas *Erica cinerea* continues vertical growth if cut (Gimingham, 1972), a response which may make *Erica* more susceptible to damage by trampling. In general, intermediate grazing is likely to favour *Erica* spp. in a mixed sward, while heavier grazing will decrease both *Erica* spp. and *Calluna*. However no examples of this process have been reported for lowland heathland.

Grazing also affects the relative proportions of dwarf shrubs and graminoids in heathland swards (Bakker *et al.*, 1983; Welch, 1984; Bullock & Pakeman, 1997). This is the result of a reduction in the cover of dominant species and increased opportunity for regeneration by other species, and the differential resilience of species to grazing and trampling. Generally, light grazing leads to an increase in dwarf shrub cover and heavy grazing leads to the replacement of *Calluna* with grassland species (Hartley, 1997; Alonso *et al.* 2001). The most detailed work is from the uplands where Welch (1984) found that light grazing favoured small increases in the cover of ericoids and lichens. Heavy grazing favoured graminoids and forbs, particularly sheep's fescue *Festuca ovina,* but also common bent *Agrostis capillaris*, sweet vernal-grass *Anthoxanthum odoratum*, smooth meadow-grass *Poa pratensis,* sheep's sorrel *Rumex acetosella* and white clover *Trifolium repens.* Mosses showed variable responses, with *Hypnum cupressiforme* favoured by light grazing and *Pohlia nutans* and *Rhytidiadelphus* spp. by heavy grazing. Once grassland areas become established within heathland, preferential grazing of *Calluna* may further encourage their expansion, as dwarf shrub species are more vulnerable to trampling and heavy grazing than grasses (Clarke, *et al.,* 1995; Hester & Baillie, 1998; Bokdam, 2001; Gallet & Roze, 2001. Recent work (e.g. Bokdam & Gleichman 2000; Alonso *et al.,* 2001) has shown that the role of grazing in the competitive balance between grasses and ericoids is mediated by resource availability.

The increase in graminoids and forbs in heath vegetation is also facilitated by seed dispersal in dung (e.g. Bülow-Olsen, 1980; Welch, 1984), and on animals' coats and in hooves (e.g. Kiviniemi, 1996; Fischer *et al.,* 1996). Welch (1984) found seed of 88 species in dung on upland heathland, although only seven of these germinated in any numbers. Vreugdenhill & Wieren (1979) also found viable *Calluna* seeds in dung on grassland. In extensive systems, more grassland species are likely to be transferred to heathland communities than *vice versa* due to differential patterns in habitat use by livestock for dunging and grazing (Putman *et al.,* 1987). No work has been carried out on how long species introduced in dung persist in heath communities. It is worth noting that most of the work on graminoids in dry heath communities has been carried out in the uplands, where invasion by grasses is considered a conservation problem. In contrast, on dry lowland heathlands these grassland communities are often of considerable nature conservation interest in their own right (Sanderson, 1998).

Finally, stimulation of regeneration from the buried seed pool is another way in which livestock may be expected to affect the composition of heath vegetation. Characteristic heathland species have been shown to form large and long-lived seed banks (e.g Gimingham, 1972; Pywell *et al.,* 1995; Thompson & Band, 1997). However, the potential for livestock trampling to expose these has not been studied directly. Most studies of trampling on heathlands concentrate on the effects of human trampling on above ground vegetation (e.g. Bayfield, 1979; Harris, 1981; Shaw *et al.,* 1995; Toullec *et al.,* 1999; Gallet & Roze, 2001, 2002).

The research reviewed here suggests that livestock activity on dry lowland heathland increases the number of species and structural diversity when at intermediate densities. However, dry heath communities are botanically intrinsically species poor (Rodwell, 1991). Where the grazing unit includes more mesotrophic communities, the increase in species diversity may be due to the introduction of species not generally characteristic of nutrient-poor heath communities (e.g. daisy *Bellis perennis,* dandelion *Taraxacum* agg, *Lolium perenne*, (Vreugdenhill & Wieren, 1979)) and not necessarily considered desirable by conservation managers. Therefore, the main beneficial impacts of intermediate grazing pressures are to increase structural diversity with benefits for animal communities (Lake, Bullock & Hartley, 2001). Little detailed information concerning livestock activity and structural diversity on lowland heathland is available.

1.3.2 IMPACTS OF LIVESTOCK PRESENCE ON WET HEATH COMMUNITIES

There has been comparatively little work on the effects of grazing on wet heath communities, which include humid heath, wet heath, valley mire, associated damp grassland and seasonal pools. However, various conservation agencies and other organisations have compiled observational reports that form a consensus regarding the effects of grazing on wet lowland heathland communities.

Ungrazed wet heaths and mires tend to become dominated by purple moor-grass *Molinia caerulea* (Clarke, 1988; Evans, 1989; Large, 2001), and a primary conservation aim is to reduce this dominance (e.g. Wright & Westerhoff, 2001). A number of studies have correlated a decrease in plant richness in valley mires with an increase in *M. caerulea* cover following cessation of grazing (and vice versa) over time scales varying between three and 40 years. Byfield & Pearman (1994) described the presence of desirable species in grazed wet heathland communities in the New Forest. The medium height turf characterising New Forest wet heaths is considered ideal for species such as heath lobelia *Lobelia urens*, lesser butterfly orchid *Platanthera bifolia* and pale dog violet *Viola lactea*. Grazing (or other forms of disturbance) is cited as essential to the long-term survival of species such as marsh clubmoss *Lycopodiella inundata,* which seems to colonise sites very slowly and is therefore reliant on the long-term continuity of favourable conditions. Poaching provides areas of bare ground needed for the establishment of new individuals in the population, especially of the more short-lived species such as slender centaury *Cicendia filiformis*, and coral necklace *Illecebrium verticillatum* (Chatters, 1996). *C. filiformis* and pale butterwort *Pinguicula lusitanica* are both known to grow within hoof prints (Neil Sanderson, pers. comm.; Evans, 1989). Grazing is also considered to be important in maintaining the open conditions of bog communities characterised by species such as bog orchid *Hammarbya paludosa,* great sundew *Drosera anglica* and slender cotton-grass *Eriophorum gracile*, although livestock use of these areas is limited (Pratt *et al.*, 1986).

Grazing has now been reintroduced as a conservation management tool to a number of sites supporting wet heath and valley mire, and this begs the question of whether there has been a resulting increase in species diversity. Data from a single 25m x 27m monitoring plot where grazing has been reintroduced on a site in Purbeck, Dorset (Cox, 1998) showed an increase in characteristic mire species such as bog asphodel *Narthecium ossifragum,* oblong-leaved sundew *Drosera intermedia,* round-leaved sundew *Drosera rotundifolia*, cotton grass *Eriophorum angustifolium* and some *Sphagnum* species, together with a decrease in *Molinia caerulea* and dwarf gorse *Ulex minor,* after three years. This is corroborated by two studies in Dorset and the New Forest (Clarke, 1988; Sanderson, 1994). In both, data were collected from a limited number of quadrats located on either side of a fence excluding (or at least reducing) grazing on mire habitat at three or four sites. Sanderson (1994) found a 240% increase in species number where grazing occurred, while Clarke (1988) found an average increase of 70% with increased grazing pressure. In both cases, the main contribution to increased species richness was the bryophyte and Cyperaceae groups, although herbs also increased. The main decrease was in *Molinia caerulea*, although it remained present. Clarke (1988) found that ericoids decreased, whereas Sanderson (1994) noted a slight increase. Sanderson (1994) also noted that while the ungrazed areas could be described by just one National Vegetation Classification (NVC) community type

(Rodwell, 1991), the grazed areas comprised a mosaic of two contrasting communities. There are, however, drawbacks to the methodologies used in this work, most notably a lack of replication between monitored plots.

Similar trends in wet heath vegetation dynamics have been observed in other UK heathland areas. Species abundance data from stands of grazed and ungrazed wet heath communities on a number of sites in Pembrokeshire (Evans, 1989) show increased cover of sedges and forbs and a decrease in *Molinia caerulea* tussocks after grazing. Evans (1989) also found that while changes in species composition were slight after the first season, vegetation structure continued to change for at least six years, mainly through a continuing decrease in the abundance of dwarf shrub species.

1.3.3 IMPACTS ON INVASIVE GRASS SPECIES

Grazing at both high and low densities can lead to a dominance of grasses and a reduction in species and structural diversity, and can be problematic for conservation managers. *M. caerulea* is a vigorously competitive grass species that becomes dominant in wet heathland if not inhibited by some form of disturbance (e.g. Clarke, 1988). Grazing is generally considered an appropriate tool to reduce *M. caerulea* cover (e.g. Diemont & de Smidt, 1987), although in the uplands *M. caerulea* replaces *Calluna* under heavy grazing on wet soils (Welch, 1984c). In contrast, in the lowlands Edwards (1985) and Tubbs (1986) suggested heavy grazing greatly reduces *M. caerulea* in damp heath communities. The studies on wet heath communities discussed above generally showed declines in the abundance of *M. caerulea* after grazing.

Deschampsia flexuosa plays a similar role in dry heathland, and communities can become dominated by this species (Rodwell, 1991). A number of studies show that grazing may decrease the cover of *D. flexuosa* (e.g. Bülow-Olsen, 1980; Hester *et al.*, 1991; Welch & Scott, 1995) although this is likely to be dependent on the initial abundance and the grazing pressure used (e.g. Alonso *et al.*, 2001). However, dominance of *D. flexuosa* is thought to be increasing as a result of increasing atmospheric nitrogen deposition (e.g. Vandereerden *et al.*, 1991; Berendse *et al.*, 1994; Bobbink *et al.*, 1998; Britton *et al.*, 2000), and this appears to reduce the efficacy of grazing in controlling *D flexuosa*. In the Netherlands, Piek (1998) noted that, in general, sheep grazing no longer controls *D. flexuosa* while Bokdam & Gleichman

(2000) found that cattle grazing did not decrease *D. flexuosa* cover nor prevent *D. flexuosa* invasion in heather.

Molinia caerulea stands have greater agricultural interest than *Deschampsia flexuosa*, and consequently more is known about utilization rates (e.g. Grant *et al.*, 1996; Common *et al.*, 1997; Wright *et al.*, 2000). Grant *et al.* (1996) suggest that *M. caerulea* will be retained as a prominent constituent of the sward if utilization rates are below 33% of annual growth. Leaf production was decreased by continuous offtake above this level after six years of cattle grazing. Newborn *et al.* (1993) initially found that Hebridean sheep controlled *M. caerulea* in an upland heathland sward through an offtake of over 39% of the vegetation. However, over a longer time span, no decrease in leaf density was found, despite an offtake of over 60% (Newborn, 2000). The heather component of the sward did however increase. The figure of 33% is sometimes quoted in conservation literature as a utilization threshold below which grazing will not affect *M. caerulea.* However, Grant *et al.* (1996) only compared two utilization rates (33% and 66%). Newborn's (2000) work suggests it may also depend on the management history of the site.

1.3.4 IMPACTS ON SCRUB AND TREE SPECIES

Scrub invasion is considered a key factor in the reduction of lowland heathland habitat quality and area (Webb & Haskins, 1980; Marrs *et al.,* 1986; Webb, 1990; UK Steering Group, 1995; Rose *et al.,* 2000) and affects both wet and dry heath communities. Studies in the uplands have shown that Scots pine *Pinus sylvestris*, birch *Betula* spp. and juniper *Juniperus communis* regeneration can be controlled by high grazing intensities (Miles, 1979), and that upland heath may succeed to woodland if grazing is excluded (French *et al.,* 1997). Similar studies in the lowlands are few. Circumstantial evidence suggests that heavy grazing may inhibit the growth of scrub and trees on lowland heaths (Marrs *et al.,* 1986, Tubbs, 1991; Tubbs, 1997). However, Bokdam & Gleichman (2000) found that medium-intensity cattle grazing was insufficient to prevent encroachment of pine and birch on a Dutch heath. Similarly, Bartolomé *et al.,* (2000) found that commercial sheep and goat breeds did not maintain an open landscape on heath vegetation in northern Spain.

Grazing livestock will have some effect in removing or reducing the size of tree seedlings (Bakker *et al.,* 1983; Pratt *et al.,* 1986) and have been reported to open up areas of scrub by pushing through it and eating growing tips (Read & Williams, 1997; J. Brooks, H. Garlik, R. Ekins, pers. comm.). Tubbs (1991) noted that pony browsing in the New Forest can kill gorse *Ulex europaeus* regeneration following burning. Livestock species and breeds that are predominantly browsers may have more of an impact (Oates & Bullock, 1997). Landes de Bretagne sheep grazing on the Breton heaths were shown to reduce gorse height, although summer grazing appeared to stimulate growth. In Denmark, Buttenschøn & Buttenschøn (1982) found that Icelandic sheep removed all above ground growth of bramble *Rubus fruticosus* agg. Studies carried out on lowland grassland suggest that both goats (Oliver, unpub. data) and Hebridean sheep (Wilkinson, 2000) can significantly reduce scrub and tree species. These species are also considered to have significantly reduced extant scrub and/or controlled regeneration on a number of lowland heathland sites (R. McGibbon, C. Fitzgerald, pers. comm.). Livestock ability to control or reduce scrub invasion appears strongly influenced by livestock type, in addition to grazing pressure and the vegetational composition of the site.

1.3.5 IMPACTS ON BRACKEN *PTERIDIUM AQUILINUM*

The problem of bracken *Pteridium aquilinum* invasion of heathland has generated a considerable amount of research (e.g. Lowday, 1984; Pakeman & Hay, 1996; Marrs & Britton, 2000) and conservation effort (e.g. Wright, 1993; Rutter, 2001). Livestock will consume small amounts of bracken (Putman *et al.*, 1987,) but generally not in sufficient quantity to reduce cover significantly. A spatial simulation model of vegetation dynamics applied to relationships between expansion of bracken patches and grazing found that small bracken patches could be controlled by livestock, but not the expansion of large patches (Birch *et al.*, 2000), and depended on the impact of trampling rather than grazing. Pakeman *et al.* (1997) found that where livestock activity was sufficiently high, bracken regeneration on upland heathland was slowed following herbicide (asulam) treatment. Again this was due to disturbance through trampling. Livestock grazing may in some cases allow the spread of bracken because it is avoided and competition from other species is reduced (Davies *et al.*, 1979). Bracken is toxic to livestock when consumed in quantity, and welfare concerns prevent the use of livestock grazing (as opposed to trampling or digging) as a tool for bracken management. No data are available on the impact of livestock presence on bracken on lowland heathland.

1.3.5 SUMMARY

In summary, the impacts of livestock presence on lowland heathland vegetation will vary according to stocking density, livestock type, grazing season, vegetation start point, site characteristics and climate. At intermediate stocking densities, grazing is likely to produce maximum species and structural diversity on all heathland vegetation types. High and low extremes can both lead to an increase in grass cover and a reduction in dwarf shrub cover; low density will generally allow an increase in scrub cover. Data are lacking on the degree of structural and species diversity and the scale of vegetation mosaics created by livestock presence, particularly for wet vegetation types. Livestock presence can control and reduce invasive grass species (provided the sward is not nutrient enriched) but will not necessarily control scrub and tree species.

1.4 The need for further research

1.4.1 THE DECLINE IN WET HEATH AND VALLEY MIRE PLANT SPECIES

In the early 1990s, Byfield and Pearman (1994) carried out research into the changes in Dorset's heathland flora. Comprehensive survey work carried out by Good (1948) provided base line data for the comparison of populations of selected heathland species of conservation note in the 1930s and the early 1990s. The work indicates a substantial decline in the floristic diversity of the Dorset heaths, and attributes this to the cessation of traditional management, in particular livestock grazing. Byfield & Pearman recorded the presence or absence of 41 indicator species of conservation interest that had been recorded by Good. No indicator species were chosen for dry heath, because it does not support any higher plants of conservation concern (www.ukbap.org.uk). Table 1.2 lists the proportion of populations on wet heath and associated habitats that had declined on extant heathland sites (i.e. those that were still regarded as supporting heathland vegetation).

Byfield and Pearman compared this decline with the situation in the nearby New Forest, considered to be one of the best remaining examples of extensively grazed systems in Europe (e.g. Wallis De Vries *et al.*, 1998), and concluded that the heathland flora had fared better in the grazed New Forest heaths. In Dorset, the biggest losses were in the three more fertile habitats, and Byfield & Pearman (1994) conjectured that these losses were due to faster rates of successional change on these habitats following the cessation of traditional management practices such as grazing.

Table 1.2 Proportion of extant stands where populations of indicator species have been lost in Dorset since the 1930s (after Byfield & Pearman, 1994). No indicator species were chosen for dry heath

Habitat type	**Populations lost since 1930s**
Humid grassland	75.0 %
Lawns and greens	92.9 %
Seasonal ponds	81.6 %
Wet heaths	40.5 %
Valley mires	50.0 %

There are some methodological drawbacks to Byfield and Pearman's work. For example, Good set out to survey characteristic stands of natural and semi-natural vegetation, which he then marked on 10 inch to the mile maps. Byfield and Pearman relocated these stands and searched them for their particular species of interest, rather than repeating Good's methodology in first identifying stands. This may have resulted in an over-estimate of the decline of species, as it did not take into account possible colonisation of new areas, which might counteract extinctions from the re-surveyed sites. Such 'regional dynamics' comprising extinctions and colonisations from habitat patches might be expected, especially for short-lived ruderal plants (Freckleton & Watkinson, 2002). In addition, a small number of stands do not appear to have been correctly relocated (although this may simply be a typographical error in the National Grid References provided by Byfield and Pearman). Nonetheless, the measured declines are large, and are corroborated by other observations (e.g. Bowen 2000), so it seems likely that there have been real declines in many species. By 2001, extensive grazing had been reinstated on 18 heathland sites in Dorset by (or with the help of) conservation organisations (I. Alexander, pers. comm.) largely in response to Byfield and Pearman's work (e.g. Byfield *et al.,* 1995; Nicholson, undated).

However, for both damp heath and valley mire, there is as yet little scientific study of the role of domestic livestock in heathland vegetation dynamics. Work from the New Forest (Pratt *et al.,* 1986, Putman *et al.,* 1987) suggests that habitat use by stock within heathland sites will be varied, and that use of wet heath and valley mires may be limited. The extent to which livestock use wetter communities, and the impact of both grazing and trampling on population dynamics and subsequent changes in community composition are not adequately understood. The work described in this thesis was therefore designed to fill some of the gaps in our knowledge about the use of livestock in managing lowland heathland, with particular references to the wet heath and valley mire communities of Purbeck, in Dorset.

The wet heath and valley mire communities found in Purbeck are described here in order to provide a context for the work presented in subsequent chapters, particularly chapters three to five, and the reader is referred back to this section where necessary. The work undertaken is then outlined.

1.4.1 THE DORSET HEATHS

The heaths of Dorset in southern England were chosen as a study area on the basis of the work carried out by Byfield and Pearman (1994), and because of the long history of heathland research in Dorset (e.g. Moore, 1962; Ripley, 1973; Cunningham, 1974; Haskins, 1978; Webb & Haskins, 1980; Chapman *et al.,* 1989; Webb, 1990; Pywell et al., 1994; Traynor, 1995; Allchin, 1997; Mitchell *et al.,* 1997; Rose *et al.,* 2000) which provides an excellent framework for further research. The specific heaths studied were all located within Purbeck (Figure 1.1) in the south east of Dorset.

The Dorset heaths have seen an 80% reduction in area in the last two centuries, mainly due to the improvement of nutrient poor podsols into less acidic and more nutrient rich agricultural land (Moore, 1962; Webb & Haskins, 1980). By 1990, although direct losses had virtually stopped, further reduction in the area of heathland was recorded (Webb, 1990). This was attributed to successional changes from heath to scrub and woodland, despite increasing conservation management effort (e.g. Nicholson, 1993; Woodrow *et al.,* 1996). More recent work (Rose *et al.,* 2000) has shown that the main subsequent losses have been in valley mire and wet heath, with a general trend of succession to drier communities dominated by scrub and trees. These changes have been attributed to the cessation of grazing and associated burning, plus a downward trend in rainfall in Dorset (Rose *et al.,* 2000).

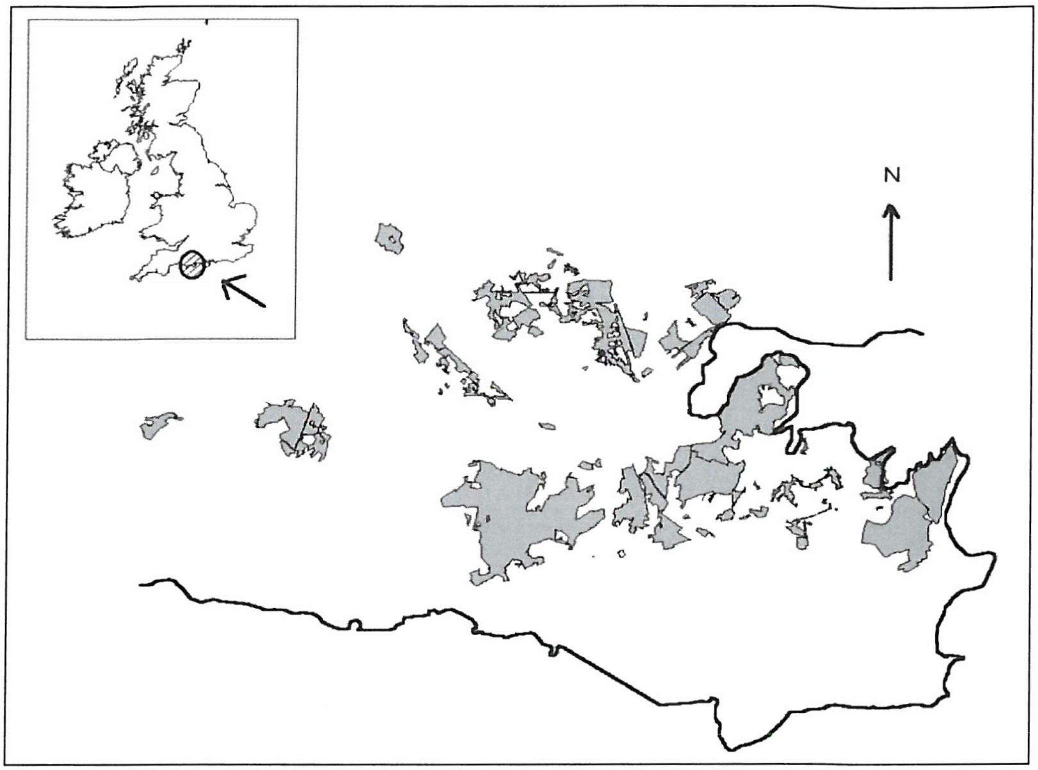


Figure 1.1 Main heathlands of Purbeck, Dorset, southern England.

Wet heath

Within the Dorset Heaths Natural Area (www.english-nature.gov.uk/science/natural/role. htm), of which Purbeck forms about one third, there are 2615 ha of wet heath (DERC, 1997). These comprise a suite of vegetation communities defined by soil moisture, nutrient and base status. The substrate is either nutrient poor mineral soils or very shallow peats that are at least seasonally waterlogged, but may be surface dry in summer. Such substrates are too dry to sustain large peat-building *Sphagnum* species associated with valley mires, but too wet to support bell heather *Erica cinerea*, a species characteristic of dry heaths. In general, the vegetation is referable to the M16 *Erica tetralix – Sphagnum compactum* wet heath community of the National Vegetation Classification (Rodwell, 1991). This includes two habitat types designated under Annex 1 of the Habitats Directive: Northern Atlantic wet heath with *Erica tetralix*, and depressions on peat substrates (*Rhynchosporion*).

The *Erica-tetralix – Sphagnum compactum* typical sub-community is frequent throughout the Dorset heaths, and is characterised by variable proportions of *Calluna vulgaris*, *Erica tetralix* and *Molinia caerulea*, with *Sphagnum compactum* occurring in drier situations and *Sphagnum tenellum* in wetter hollows. Lichens can be frequent, particularly in transitions to drier heathland communities. In such situations there can also be some enrichment of the sub-shrub cover by scattered bushes of *Erica cinerea,* western gorse *Ulex gallii and* dwarf gorse *U. minor.* Deer grass *Trichophorum cespitosum,* bog asphodel *Narthecium ossifragum* and cotton-grass *Eriophorum angustifolium* become more frequent towards the wetter transitions to valley mire. This is the most widespread M16 sub-community in Purbeck.

On more base rich substrates, the slightly richer M16b devil's-bit scabious *Succisa pratensis* – carnation sedge *Carex panicea* subcommunity is found, which includes species such as meadow thistle *Cirsium dissectum,* saw-wort *Serratula tinctoria* and milkwort *Polygala serpyllifolia* with occasional heath spotted orchid *Dactylorhiza maculata,* petty whin *Genista anglica,* red fescue *Festuca rubra,* jointed rush *Juncus articulatus,* and flea sedge *Carex pulicaris.* The nationally scarce marsh gentian *Gentiana pneumonanthe* occurs in this community. The more acidic species such as *Narthecium ossifragum, Trichophorum cespitosum, Eriophorum angustifolium* are less frequent, and *Sphagnum compactum* and *S. tenellum* tend to be replaced by *S. auriculatum.* This community is generally of a more tussocky nature due to the steady movement of surface water, which encourages *Molinia caerulea* growth.

The very local white beak-sedge *Rhynchospora alba* – oblong-leaved sundew *Drosera intermedia* sub-community (M16c) is more open, characterised by a reduced cover of *Calluna vulgaris, Erica tetralix* and *Molinia caerulea* and an extensive cover of *Sphagnum compactum* and *S. tenellum* with other mosses including *Campylopus brevipilus*, plus scattered tussocks of heath rush *Juncus squarrosus* and *Trichophorum cespitosus.* Wetter hollows and runnels support round-leaved sundew *Drosera rotundifolia* and the rarer great sundew *D. anglica,* together with *Rhynchospora alba.* Bare peat is colonised by *Pinguicula lusitanica* and the nationally scarce *Rhynchospora fusca* and *Lycopodiella inundata,* which provide a strong floristic link with the valley mire communities.

Mires

The Dorset Natural Area supports 618ha of ombrotrophic bog (DERC, 1997), which is further classified as valley mire (a habitat largely confined in western Europe to Dorset and the New Forest), moor-grass mire, and bog pools (after Sanderson, 1998). Valley mires develop along the floors of small, shallow valleys with low hydraulic gradients and impermeable subsoils, allowing peat accumulation. In the context of nature conservation, this term is also used in Dorset to encompass what are in fact closer to Schwingmoor type basin mires (Fojt, 1994), where a raft of vegetation colonises an open water surface before sinking to form peat, giving rise to a semi-floating structure. The majority of the Purbeck valley mires are referable to the M21 *Narthecium ossifragum-Sphagnum papillosum* mire community. However, where these have become degraded or overgrown, this community is replaced by the *Molinia caerulea* dominated M25 *Molinia caerulea-* tormentil *Potentilla erecta* mire community. Where mires receive base-rich ground water and support black bog-rush *Schoenus nigricans,* the very locally distributed M14 *Schoenus nigricans-Narthecium ossifragum* mire community is found. Lastly, in more open areas, the M1 *Sphagnum auriculatum* bog-pool community occurs.

The *Narthecium ossifragum-Sphagnum papillosum* mire community is dominated by *Sphagnum* species, including the nationally scarce *Sphagnum pulchrum*, which has the core of its British distribution in the Purbeck valley mires (Edwards, 1999). Other characteristic *Sphagnum* species include *S. papillosum, S. subnitens,* and *S. auriculatum*. Common associated species include *Drosera rotundifolia*, *Erica tetralix,* bogbean *Menyanthes trifoliata*, bog pondweed *Potamegaton polygonifolious* and bog myrtle *Myrica gale.* Rare and uncommon vascular plants associated with this community include slender sedge *C. lasiocarpa,* bog sedge *C. limosa,* bog hair-grass *Deschampsia setacea, Drosera anglica,* Dorset heath *Erica ciliaris,* bog orchid *Hammarbya paludosa* and intermediate bladderwort *Utricularia intermedia.* Water levels are constantly high, and the mires are soft and wet year-round, with natural shallow runnels and water tracks. Peat development is extremely slow compared to the uplands – similar mires in the New forest accumulate at a rate of 20cm per 1000 years (Clarke, 1988). Consequently, peat depths are generally shallow. The *Narthecium ossifragum-Sphagnum papillosum* mire community as a whole is not recognised in the Corine classification of European habitat types (Anon, 1991) or the Habitats Directive, but the *Rhynchospora alba - Sphagnum auriculatum* sub-community is encompassed within the Annex I habitat 'depression on peat substrates of the Rhynchosporion'.

Many of the valley mires in Dorset include bog pools, where water flow is very slow, or stops completely. Such pools are highly acidic and dystrophic. They are dominated by floating masses or soft wet carpets of *Sphagnum* species with scattered vascular plants growing through them, and fit into the M1 *Sphagnum auriculatum* bog pool community. Dominant species are generally *Sphagnum auriculatum* and *S. cuspidatum* with some submerged *S. pulchrum.* Vascular species are represented by *Menyanthes trifoliata, Rhynchopora alba* and *Eriophorum angustifolium.* Rare species such as *Drosera anglica,* brown beak-sedge *Rhynchospora fusca* and *Carex limosa* may also be present. It is likely to be one of the few communities which can sustain its botanical interest in the absence of grazing (Wright & Westerhoff, 2001).

Where valley mire systems drain into surrounding habitat, the M29 bog St. John's wort - bog pondweed *Hypericum elodes - Potamogeton polygonifolius* soakway community develops, sometimes as a marginal belt between the valley mire and surrounding wet heath. Grazing may play a part in preventing this community developing into carr woodland in some situations (Rodwell, 1991).

Conservation interest of wet heath and valley mire.

In addition to many notable plant species, damp heath and mire are of conservation interest due to the presence of a number of other rare and scarce species. Twenty two species of Odonata are found on wet heath and mires, including the small red damselfly *Ceriagrion tenellum* which is confined to pools in valley bogs in the New Forest and south east Dorset, and the southern damselfly *Coenagrion mercuriale* which is associated with livestock grazing (Brooks, 1997). These, together with more common, larger hawkers, such as the southern hawker *Aeshna cyanea* and migrant hawker *Aeshna mixta,* form prey for the hobby *Falco subbuteo,* which hunts over open heathland water. The raft spider *Dolomedes fimbriatus* can be seen on bog pools while the large marsh grasshopper *Stethophyma grossum* is only found in the wettest parts of bogs. Other characteristic heathland species such as the silver-studded blue butterfly *Plebejus argus* and the smooth snake *Coronella austriaca* (confined to the heathlands of southern England) are also associated with the presence of water or damper areas. *Genista anglica* hosts its own specific micromoth *Coleophora genistae* known from no other plant species, as does *Gentiana pneumonanthe (Stenoptilia pneumonanthe).* The natterjack toad *Bufo calamita,* although no longer found in the Dorset heaths, breeds in bog pools provided they are not too acid.

1.5 THESIS OUTLINE

The research presented here considers the role of domestic livestock in the conservation of lowland heathland today, with particular reference to the Purbeck heaths of Dorset.

Firstly, aspects of the behaviour of free-ranging livestock grazing heathland sites are explored – do livestock use heathland sites in ways that have the potential to fulfil conservationists' objectives for those sites? The hypotheses proposed in Chapter 2 are (i) that livestock use particular habitats more that would be expected through random choice, and (ii) that different heathland habitats are used for different activities.

The effect of livestock presence on vegetation communities is then considered experimentally in Chapter Three. The hypothesis that cattle activity alters the species composition and vegetation structure of damp heath and valley mire communities is examined. The physical effects of trampling, such as microtopographical variation and compaction in the substrate are established. (Effects of livestock on nutrient cycling within vegetation communities are not considered in this thesis).

Chapter Four looks at a suite of characteristic wet heath and valley mire plant species. Using a combination of experimental work on emergence requirements and survey work of general habitat requirements, it considers how livestock presence will affect these species. It is hypothesised that livestock presence can create the conditions required for emergence through canopy removal and compaction leading to raised water levels.

Chapter Five looks specifically at regeneration from the buried seed pool and examines the hypothesis that livestock trampling can facilitate the regeneration of wet heath and valley mire indicator species. The potential for regeneration at sites where species are considered extinct above ground is examined experimentally.

Chapter Six provides a synthesis of the results of the previous chapters in a conservation context. It reviews the impact of domestic livestock on heathland communities with particular emphasis on wet heath and valley mire plant communities, and then looks at how this can be manipulated to achieve conservation aims.

2. Livestock Behaviour

2.1 Introduction

Grazing is now recognised as a conservation tool for the management of lowland heathlands (e.g. Michael, 1993). However, there is a need for greater understanding of how free-ranging livestock use heathland systems to establish to what extent grazing has the potential to fulfil conservation aims, and how it may be refined as a management tool. The relationships between herbivores and vegetation are influenced both by the behaviour and ecology of the herbivore, and the ecological response of the vegetation to grazing and trampling damage (which is addressed in Chapter 3). This chapter considers livestock behaviour. Herbivores are selective in both their use of habitat and choice of plant species (e.g. Jarman & Sinclair, 1979; Duncan, 1983; Grant *et al.,* 1985, 1987; Welch, 1986; Gordon, 1989). As livestock selectivity influences which, and to what extent, particular vegetation communities and particular plant species within them are affected, it plays a key role in conservation grazing.

Work done to date exploring resource use by ungulates (including domestic livestock) on lowland heathland communities has been limited to the New Forest (e.g. Pratt *et al.*, 1986; Tyler, 1972; Pollock, 1980; Putman *et al.,* 1987; Ekins, 1989). There is, however, a substantial body of work from upland heathland, which gives some indication of the likely behavioural patterns on lowland heathland (e.g. Gates, 1979; Gordon, 1989b & c; Grant *et al.,* 1987; Duncan *et al.*, 1994; Clarke *et al.,* 1995b; Grant *et al.*, 1996; Hester *et al.*, 1996; Hester & Baillie 1998; Hester *et al.,* 1999). In addition, there are a number of studies of grazing on heathland and other semi-natural habitats from other countries (e.g. Duncan, 1983; Bakker *et al.*, 1983; van Wieren, 1991; Wallis de Vries, 1991; Prins, 1992; Bokdam & Wallis de Vries, 1992; Fedele *et al.,* 1993; Bartolome *et al.*, 1998), which provide a useful framework for considering livestock behaviour on heathland grazing in the UK. In this chapter, these studies are reviewed before observational work carried out on resource use and behaviour in seven livestock groups on heathland sites in Dorset is presented. The review is based on a report done for English Nature (Lake *et al.,* 2001).

The ideal free distribution (Fretwell & Lucas, 1970) predicts that, given individuals are able to move freely, they will go to the patch where their rewards (e.g. intake rate) are highest, and this theory has been widely applied in studies of habitat selection (see

Sutherland, 1996). For example, vegetation sward height and productivity was used to interpret habitat selection by cattle and ponies in the New Forest (Ekins, 1989). However, more complex versions of the model are generally needed to explain habitat selection (Sutherland, 1996). In the case of domestic herbivores grazing on lowland heathland sites, habitat selection is influenced by herbivores' foraging strategies, which in turn reflect herbivore physiology and social behaviour and therefore vary between species and breeds. Environmental factors such as topography, climate (e.g. Rawes & Welch, 1964) and human disturbance (e.g. Tyler, 1972; Duncan, 1983; Pratt *et al.*, 1986) also play a role, as well as the relative abundance and quality of plant material (e.g. Grant *et al.,* 1985). These factors are considered here.

2.2.2 HERBIVORE-BASED FACTORS DETERMINING SELECTION

Digestive morphology.

Domestic herbivores include both hind-gut fermenters (equids) and ruminants (bovids). In general, hind-gut fermenters have a higher rate of nutrient extraction from forage (including low-quality forage such as that found on lowland heathland) than ruminants (Duncan *et al.,* 1990). Non-ruminant hind-gut fermenters, such as ponies, are less efficient digesters than ruminants and have a faster throughput of food. However, as their intake is not limited by rumen capacity, they are able to process a much larger quantity of forage, which more than compensates for their digestive inefficiency. Their fast throughput means they tend to spend more time grazing than ruminants. For example, work in the New Forest suggests that ponies spend up 75% of their time grazing, while cattle graze for only 57% of the time (Pratt *et al.*, 1986). The digestive system of hind-gut fermenters also suggests they are likely to be less selective, as they can compensate for reduced quality by increasing quantity. Putman *et al.* (1987) show this to be the case in the New Forest, where ponies show weaker preferences for more habitat types than cattle. In addition, since hind-gut fermenters are more likely to be able to keep up their intake out of the growing season by consuming poor quality (including dead) material, they are less likely to experience winter nutrient stress. For example, van Wieren (1991) observed that Highland cattle lost a significantly greater proportion of their body weight over-wintering on a conservation area in the Netherlands than Shetland ponies. Such differential effects of a decline in winter forage quality are also illustrated in the New Forest, where the majority of cattle are removed from the Forest for the winter, whereas ponies are generally out-wintered (Ekins, 1989).

Hofmann (1989) categorised ruminants along a continuum from grazing animals, limited to consuming graminoids and forbs, at one extreme to browsing animals, which concentrate on lignified, woody vegetation, at the other. Differentiation between browsers and grazers is based upon gut morphology and consequent digestive ability, which lead to differential diet selection. Grazers, such as cattle and sheep, retain forage within the rumen for a longer period than browsers, enabling breakdown and exploitation of plant cell wall contents and so can cope with poor quality forage. Browsers such as roe deer have a short retention time and exploit rapidly digestible cell contents. This strategy requires forage with better quality cell contents, so browsers are likely to be more selective than grazers. Species such as goats are intermediate, grazing some graminoids species but switching to browsing when these become too fibrous. The difference between browsers and grazers is reflected in habitat choice in a number of studies carried out on heathlands (e.g. Bullock, 1985; Bartolome *et al.*, 1998; Gordon, 1989a). A summary review of habitat and diet selection on heathlands presented in Table 2.1.

Recent studies have shown, however, that whilst fibre digestibility is superior in grazers compared to browsers, there are few other digestive differences between grazers and browsers. In fact, several key elements of foraging strategy are better explained by body mass than by Hofmann's classification (Iason & van Wieren 1999).

Body size

Body size influences feeding strategy through its relationship with metabolic rate. Smaller animals have a greater metabolic rate per unit of body weight than larger ones, and so need relatively better quality forage to satisfy their metabolic requirements. Larger animals must intake a greater quantity of forage, but their relatively lower metabolic requirement plus the longer retention time of forage within the rumen means they can use forage of a lower quality (e.g. Jarman & Sinclair, 1979; Demment & Van Soest, 1985; Illius & Gordon, 1992). This was observed by Grant *et al.*, (1985) in the uplands, where cattle showed a greater readiness to graze more fibrous elements of the sward than sheep. In addition, larger animals have a smaller incisor breadth (determining bite size and so food intake) in relation to their metabolic requirements (Illius & Gordon, 1987). These two factors mean that larger animals cannot tolerate the short swards that can support smaller species, and so may be excluded by grazing pressure from mutually preferred swards (Clutton-Brock & Harvey, 1983). This has been described from grazing areas including heathland on the Isle of Rhum (Gordon, 1989d), where cattle were excluded by red deer from highly digestible species-rich grassland when the amount of available forage decreased in winter. The cattle are then forced to move onto less digestible oligotrophic grassland, where they can intake a greater amount of lesser quality forage. Similarly, Osborne (1984) suggests that the presence of sheep on upland heathland in the west Highlands reduces the degree to which red deer use their more strongly preferred swards.

The relationship between body size and metabolic requirements may also lead to sexual segregation and differential habitat choice in dimorphic species. This has been observed in goats on Rhum (Gordon, 1989d) and red deer in Scotland (Osborne, 1984), where larger males forage on more oligotrophic communities than smaller females when resources become scarcer in winter. Sexual segregation, facilitation and competitive exclusion have not been recorded from lowland heathland sites. However, these processes may occur on sites of sufficient size, where a wider choice is available to herbivores. They are less likely to be observed on smaller sites where the choice and extent of habitat types is limited.

Incisor morphology

The effects of relative incisor breadth on habitat selection have already been discussed. However, other morphological aspects of the mouth also affect ungulate foraging. Non-ruminants such as ponies have powerful opposed incisors that can easily cut through fibrous stems (van Wieren, 1991). Ruminants lack upper incisors (the lower incisors closing obliquely against a hard palate) and tear vegetation rather than cut it. They also use the tongue to wrap around vegetation and pull. This difference allows non-ruminants to graze closer to the ground than ruminants. In addition, species with smaller muzzles are more likely to be able both to graze shorter swards and to select plants from within a mixed sward. For example, sheep and deer have been shown to graze closer to the substrate surface than cattle, which are less able to be selective while feeding from fine-scale mixtures (Grant *et al.,* 1985; Grant *et al.*, 1987). Sheep also appear to be able to increase their search effort for preferred species when forage is scarce by taking fewer bites per step (Laca & Soriguer, 1993).

Table 2.1 Summary review of habitat and diet selection in domestic herbivores on heathlands.

Stock type	Time spent on habitat - habitat preference	Dietary preference/composition	Seasonal change in habitat/diet selection	Area	Study
Sheep	N/A	*Molinia caerulea*– preferred *Trichophorum cespitosum* – preferred *Carex* spp – preferred *Calluna* – preferred *Eriophorum* spp.– preferred	Use of *Calluna* & *Eriophorum* spp.increases as other spp. die off, *Eriophorum* spp. peaking Jan –Mar, *Calluna* Aug – Oct.	Scotland	Grant *et al.*, 1976
Sheep	Heathland – preferred during winter Pasture- preferred July-Oct	-	Switch from heath and woodland to pasture in summer	Netherlands	Bakker *et al.*, 1983
Sheep	Mesotrophic communities - Strong preference Oligotrophic communities – Strong avoidance	See seasonal change	*Agrostis –Festuca, Calluna— Trichophorum* and *Nardus stricta* swards used less, *Molinia* caerulea grassland used more in winter	West Highlands	Osborne, 1984
Sheep	N/A	*Calluna* utilisation in grass/heather mosaic increased with fragmentation	Autumn increase in *Calluna*	N.E. Scotland	Clarke, *et al.*, 1995
Sheep	N/A	*Erica* sp. – preferred *Calluna vulgaris* –preferred Other woody spp – avoided Graminoids – preferred Forbs – lightly preferred	Winter increase in *Calluna*	NE Spain	Bartolome *et al.*, 1998
Cattle Sheep	N/A	See seasonal change	Ericoids most heavily used Oct-Dec, less than graminoids in spring & summer. *Molinia caerulea & Trichophorum cespitosum* most heavily used in spring & summer	North east Scotland	Welch, 1984
Cattle	60-70 % grasslands –strong preference 10-20% wet and dry dwarf shrub heath -underexploited 10-20% woodland -avoided gorsebrake -no preference	0-1.2% *Molinia* 65-80% other grasses 9-27% *Calluna*	Little variation in use of habitat – artefact of supplementary feeding Most use made of *Calluna* in winter, *Molinia caerulea* in summer.	New Forest S. England	Pratt *et al.*, 1986; Putman *et al.*, 1987

Table 2.1 cont'd. Summary review of habitat and diet selection in domestic herbivores on heathland

Stock type	Time spent on habitat - habitat preference	Dietary preference/composition	Seasonal change in habitat/diet selection	Area	Study
Cattle	Grassland communities	N/A	Moved from mesotrophic to oligotrophic grasslands in winter	Isle of Rhum, Scotland	Gordon, 1989b
Cattle - Meuse-Rhine-Issel	63.8 % heathland - Under exploited	N/A	Less use of heathland in winter	Netherlands	Wallis de Vries, 1991
New Forest ponies	35-67% grasslands – strong preference <15% wet heathland - under exploited <10% valley mire - preferred in summer <37% woodland & gorse brake - preferred in winter	0.3-23% *Molinia caerulea* 36-75% other grasses 0.5-27% *Calluna* 6-10% *Pteridium aquilinum* 0-13% *Ulex* spp.	Greatest used of grassland in spring, wet and dry heathlands in late summer, gorse brake and woodland in winter Grass max. in summer, *Agrostis curtisii* replacing *Molinia caerulea* in autumn. Corresponding increase in *Ulex* spp. & *Ilex aquifolium.* *Pteridium aquilinum* max in autumn	New Forest, S. England	Pratt *et al.*, 1986; Putnam *et al.*, 1987
Ponies	Grassland communities	N/A	Increased use of *Molinia* grassland in summer	Isle of Rhum, Scotland	Gordon, 1989b
Exmoor ponies	Upland heath	*Calluna* – avoided *Juncus* spp.– preferred *Ulex* spp. preferred *Pteridium – aquilinum* no preference *Molinia caerulea* – preference unknown	Grasses spring to autumn; gorse increases in autumn, max. use in winter, *Juncus* spp. max. in spring, min. in winter, *Calluna* max in winter, min in summer, intermediate spring & autumn	Exmoor, SW England	Baker, 1993
Goats	Dwarf shrub heath	N/A	Females use mesotrophic grasslands and dry heath in summer, heath in winter. Males use dry heath, increasing use of mesotrophic grassland in spring & autumn	Isle of Rhum, Scotland	Gordon, 1989b
Goats	N/A	*Erica* sp –strongly preferred *Calluna vulgaris* – preferred Woody spp. – preferred Graminoids – avoided Forbs – avoided	Spring decrease, summer increase in *Calluna*	NE Spain	Bartolome *et al.*, 1998

Seasonal variation in foraging behaviour

Seasonal variation in foraging behaviour and therefore habitat selection is likely to occur on lowland heathlands as resource availability changes. It may differ between livestock species due to the constraints imposed by body weight, digestive ability and muzzle morphology outlined above. When forage availability declines in their preferred habitats, larger ruminants and hind-gut fermenters, such as ponies, are more likely to move onto areas with a greater abundance of poorer quality forage, while smaller ruminants stay on what remains of the better quality forage (as seen on Rhum). In the New Forest, ponies showed a significant shift from grasslands to gorse brake and woodland in the winter (Pratt *et al.,* 1986). However, this may be related to an increased requirement for shelter during winter. Cattle remaining in the Forest over winter were given supplementary feed, and spent most time in the area where the feed was supplied (Ekins, 1989). Gordon (1989b) noted seasonal trends on Rhum, where cattle, ponies, goats and red deer all showed a greater degree of habitat selectivity in the summer and winter, broadening their habitat use in spring and autumn. Similarly, Duncan (1983) found that habitat selectivity by Camargue horses was greatest in the growing season.

Herding behaviour

Cattle show strong herding behaviour (Arnold & Dudzinski, 1978). In the New Forest, cattle form groups of at least 10 individuals and are therefore less widely dispersed over the forest than ponies, which form smaller groups. Vegetation types occurring in small patches (e.g. <10ha) such as roadside verges and streamside lawns tend to be avoided - possibly because they are too small to accommodate the whole herd (Putman *et al.,* 1987; Ekins, 1989). The size of both the herd and the grazing unit may affect dispersal patterns – small numbers of cattle on small (<10ha) heathland sites in the London Basin are considered to roam as widely as ponies (R. McGibbon, pers. comm.). Group size may also influence behaviour. For example, smaller groups of cows have been shown to be more vigilant during grazing than larger groups (Rind & Phillips, 1999)

Wild horse populations with natural sex ratios generally have a harem structure (Wells & von Goldschmidt-Rothchild, 1979; Gates, 1981). However, on many sites grazed for conservation, stallions are not included within the herd, and consequently this structure is found infrequently. In the New Forest, ponies tend to form small family groups or associations of two to three individuals (Tyler, 1972). This results in a much wider dispersion of ponies than would be the case if large herds existed. Territorial behaviour is also generally absent, although groups tend to maintain home ranges and grazing pressure is greatest where these overlap. Simiarly, the sex ratio of sheep flocks is generally altered. Sheep on upland heathlands form matrilineal groups that use particular parts of the home range in a regular daily fashion (known as hefting), and do not herd as a whole unit (Hunter & Milner, 1963). This means that sheep tend to be widely dispersed over the grazing area. Social behaviour in sheep has not been reported from heathland habitats in the lowlands. Hefting is generally associated with upland breeds, and is less likely to occur on smaller sites. Feral goats also form matrilineal groups, with male groups that often range separately (D.Bullock, unpub. data). However, their home ranges have clearly defined boundaries with little overlap.

Learning behaviour

There is evidence to suggest that diet selection is learnt from an animal's mother and the other animals with which a young animal associates, and that this is more important than breed effects (Key & McIver, 1980; Provenza & Balph, 1987, 1988; Dwyer & Lawrence, 1997). For example, Provenza & Balph (1988) have shown that the diet selected by fostered lambs relates to their foster dam rather than to their breed. Feeding preferences may also reflect regionally determined feeding experience handed down by mothering (Biquand & Biquand-Guyot, 1992; Provenza, 1994). This is seen in goats from different regions of Italy, which, when grazing together, select different species from the sward (Fedele *et al.*, 1993). To what extent the observed differences in diet selection that occur between livestock breeds (Mercer *et al.*, 1997; Bartolome *et al.*, 1998; Wright *et al.*, 2000) are in fact due to learning behaviour is unclear. Learning behaviour has not been studied on heathlands.

Inter-specific interactions

On lowland heathland sites with more than one species grazing, interspecific interactions may also have an effect on habitat selection. Once resources become depleted, only smaller animals are able to exploit them, and larger animals are forced to move to areas where forage is still easily available to them (as discussed under *Body Size*). Such indirect competition has been observed on Rhum between cattle and red deer (see above). This process has also been observed in reverse (Clutton-Brock & Albon, 1992), where removal of sheep from upland heathland in Scotland has been followed by an increase in deer numbers. However, in manipulative experiments, the overall patterns of foraging behaviour by sheep and deer on upland heathland were little affected by the presence or absence of the other species (Hester *et al.*, 1999). Direct competition has rarely been reported between domestic livestock species grazing semi-natural communities. Ponies are considered dominant over cattle in the New Forest (Ekins, 1989), although the extent to which this may affect habitat choice by cattle is not clear.

2.1.2 PLANT-BASED FACTORS DETERMINING HABITAT SELECTION

Plant quality: within and between species differences

Like other herbivores, domestic livestock are constrained in their dietary selection by the variables associated with their body size and behaviour. However, a further factor driving diet selection is the variation in forage quality, both within a species (e.g. younger foliage is often more nutritious than older foliage) and between species (as plant species differ markedly in their nutrient and secondary compound content). Some heathland plant species are well known to be relatively unpalatable; for example, bracken *Pteridium aquilinum* has a variety of toxic constituents, whilst matt grass *Nardus stricta* is fibrous and has high levels of silica. In contrast, other species, such as many grasses (e.g. *Agrostis* and *Deschampsia* species) are very attractive to large herbivores. Generally plants or plant parts that are low in nitrogen or other nutrients and high in lignin, fibre and secondary compounds will be relatively unattractive to herbivores.

The majority of work on dietary selection by livestock concerns sheep in upland systems. Sheep have been shown to avoid *Nardus stricta,* purple moor-grass *Molinia caerulea* and rushes *Juncus* spp. in favour of other grasses and heather *Calluna vulgaris* (Welch, 1986; Hartley, 1997; Alonso *et al.*, 2001). In addition to showing marked between-species preferences, sheep are also very adept at detecting small variations in plant quality within a species. For example, they will graze the new shoots on *N. stricta* tussocks, which are higher in nitrogen and less tough than the older shoots. They have also been shown (Duncan *et al.,* 1994) to detect changes in the quality of *Calluna* produced by fertiliser, even though this experimental manipulation produced no visual cues (e.g. fertilised shoots were no longer, nor did they have more flowers). Cattle are believed to be rather less selective than sheep; for example, they will graze on *N. stricta* and can decrease its cover (Welch, 1986).

Plant quantity: the relative abundance of plant material

Despite the selection behaviour described above, studies on a range of herbivores have shown that plant quantity may be more important that plant quality in diet selection. Herbivores often prefer areas of high biomass when they can maximise their intake rate. For example, Arnold (1987) found that sheep concentrated their grazing in patches of the highest yield; only when this had been taken into account were effects of species palatability detected. Similarly, deer were found to select the trees they browsed purely on the basis of tree size (Hartley *et al.,* 1997). The chemical composition of the trees in terms of nitrogen and secondary compound content had no measurable effect on preference and the sole effect of fertiliser on preference was via the effects of nutrient addition on tree size.

Duncan (1983) showed a relationship between habitat use and forage availability for horses in the Camargue, where habitats with abundant green (as opposed to dead) forage were preferred. In contrast, Putman & Pratt (1987) found no clear correlation between forage availability and habitat use in the New Forest. In the uplands, diet selection has been shown to remain constant despite decreasing forage availability: both sheep and deer have been found to maintain their preference for grass over heather despite a decreasing availability of grass (Hester *et al.* 1999). Little direct work has been carried out on the implications of optimal foraging (i.e. maximising energy or nutrient intake per unit foraging time) by domestic livestock on semi-natural communities. Given that the need to meet food requirements is a primary determinant of differential habitat use by herbivores, clear relationships between the amount and suitability of forage available in habitat and habitat use may be expected on a broad scale (e.g. Ekins, 1989).

Plant distribution: spatial variation in vegetation.

Many heathland sites support relatively fine-grained mosaics of different vegetation community types. The distribution of vegetation types has an influence on selection. Again, most work in this area has been done in the uplands and has examined how sheep forage in grass heather mosaics (Clarke *et al.,* 1995 a & b; Hester & Baillie

1998, Hester *et al.,* 1999). Since sheep prefer palatable grasses (e.g. *Agrostis* and *Festuca* spp.) to woody species such as heather, they are attracted by grass patches within the heathland canopy. Grazing can then lead to fragmentation of heather cover and an increase in grass patches of a variety of sizes, shapes and distributions. Utilisation of heather may be greater at the edge of grass patches than further away in the canopy, regardless of the size of the grass patches (Hester & Baillie, 1998).

2.1.3 SITE-BASED FACTORS

Shelter

The importance of shelter varies with species and with site characteristics including climate. Pratt *et al.* (1986) found that foraging defined habitat use by New Forest ponies during the day, while the need for shelter was more important at night. Grassland communities were used during daylight hours, and woodland and gorse brakes were used more frequently after dusk, although the ponies still grazed for up to 67% of the night. Cattle also moved off grasslands at night, but tended to move onto dry heathland in clear conditions, only using woodland when visibility is reduced. They grazed little at night. Few other studies include diurnal observations (but see Hester *et al.,* 1996). New Forest ponies and cattle also show a clear selection for shelter in winter, increasing their use of gorse brake and woodland at the expense of grassland.

There are a number of examples of weather conditions influencing habitat selection. In the New Forest, ponies, and to a lesser extent cattle, seek out shaded areas conferring protection from sun and flies during the middle of the day in summer, and have been observed travelling up to 6 km to reach it (Tyler, 1972; Ekins, 1989). Shaded grasslands are used to a greater extent than more exposed ones during this time (Ekins, 1989). Sheep studied in the Basque region of Spain searched for shelter from the sun and rested for between 3 and 6.5 hours in the middle of the day during summer in N Spain (Marijuan-Angulo, 1996; I. Alonso pers. comm.). Goats require shelter in winter (Oates & Bullock, 1997), and make greater use of beaches for winter-feeding on Rhum due to the proximity of caves providing shelter (Gordon, 1989). Rawes & Welch (1964) showed that strong winds decreased the use of exposed Pennine slopes by sheep, and Oates *et al.*, (1998) noted that Welsh Mountain ponies and sheep caused localised vegetation damage by sheltering in certain locations in exposed weather on a Pembrokeshire coastal heathland. However, in contrast, shelter was not an important factor in habitat choice by Camargue horses (Duncan, 1983), although protection from flies and windy conditions was observed to have a minor affect on non-feeding activities.

Water and supplementary feed

Availability of water is assumed to have an affect on habitat choice (e.g. Tyler, 1972). This will only occur on lowland heathland where water availability is limited in the preferred grazing habitat. Water is generally readily available near most of the highly grazed areas of the New Forest, and habitat selection is attributed to forage rather than water availability (Ekins, 1989). Supplementary feeding also influences habitat selection. Cattle behaviour in the New Forest in winter was considered to be strongly influenced by the provision of supplementary feed (Putman *et al.*, 1986). Work in the uplands has shown that sheep congregate at feeding sites up to five hours before and after daily feeding (Hudson, 1986).

2.1.4 AIMS AND STUDY METHODS

Although there are significant amounts of data concerning livestock behaviour on lowland heathland in the New Forest, it is not clear to what extent these are transferable to other sites. The New Forest is unique in lowland England in remaining largely intact as an extensive uninterrupted tract of heathland and associated habitats - some 15 031 ha of heathland rough grazing plus 4545 ha of pasture woodland and carr (Westerhoff, 1992). There is also a continuity of land use in the New Forest not paralleled in other heathland areas. The traditional pastoral system revolving around the depasturing of stock, largely undertaken by those with commoners' rights, survives remarkably intact in the New Forest (Tubbs, 1986), and remains one of the best examples of such a grazing system in north-west Europe (Wallis de Vries *et al.*, 1998). In contrast, the Dorset heaths are fragmented (Chapman, Clarke & Webb, 1989), and the traditional pastoral economy disintegrated during the twentieth century (Cunningham, 1974, see Chapter 1). The lack of continuity in land-use means that on sites where grazing has been recently reintroduced vegetation communities may still be adapting to grazing pressure. The Dorset heaths are more typical of lowland heaths in Britain, especially England.

This chapter explores the behaviour of livestock reintroduced to four heathland sites in Dorset that had not been grazed for several decades. The specific hypotheses considered for six livestock groups were that:

(i) Use of heathland sites was non-random;
(ii) Habitat use varied seasonally;
(iii) The proportion of time spent on different behaviours (e.g. foraging, resting etc.) varied between habitats;
(iv) Habitat use for particular behaviours varied seasonally;
(v) Plant species were differentially selected in key heathland habitats.

The considerable differences between sites and livestock groups prohibited their use as replicates to provide a general model of the probability of particular habitats being selected, or being used for different behaviours, etc. The aim was therefore to use detailed information on livestock behaviour and habitat selection from a number of livestock groups to build a general picture of livestock grazing. The work was designed to provide information immediately relevant to heathland site managers. So, for example, habitat type, rather than the availability of forage, was considered in relation to non-random use of sites, as this has more immediate relevance to conservation management. Similarly, behaviours were grouped for analysis according to potential impact on vegetation, i.e. the time spent resting (leading to localised vegetation damage), foraging (leading to selective biomass reduction) and moving (leading to trampling of vegetation and poaching). Most studies of habitat selection tend to pool data between individuals and sampling times. However, variation among individuals (e.g Bowers, 1995), times of day (e.g. Beyer & Haufler, 1994), years (e.g. Schooley, 1994) and seasons may lead to incorrect results from combining data. Therefore, such differences have been taken into account wherever possible here to provide a realistic view of habitat selection and behaviour in the livestock groups studied.

Statistical analysis

Many statistical methods have been used to analyse resource selection (e.g. Friedman, 1937; Neu 1974; Quade, 1979; Johnson, 1980; Heisey, 1985; Aebischer *et al.*, 1993). Unfortunately, these methods do not necessarily provide a consensus, due to the differences in how they address variability in habitat use (Alldredge & Ratti, 1986, 1992; Manley *et al.*, 1991; McClean, *et al.*, 1998). The constraints imposed by data on habitat selectivity relevant to this study are: (i) the possibility of sequentially collected observations being serially correlated, (ii) the undesirability of pooling data between unidentified individuals if animals show significant individual variation, and (iii) the unit - sum constraint of compositional (proportional) data (Aebischer *et al.,* 1993).

The first two constraints inflate the apparent degrees of freedom and increase the possibility of rejecting the null hypothesis inappropriately (i.e Type 1 error, Sokal & Rohlf, 1981). Therefore, it is considered preferable to use individual animals as sample units, rather than observations (Alldredge & Ratti, 1986; Aebischer, et al., 1993). However, in herding animals, correlation between individuals at any one time is, by definition, likely. In these cases, provided that animals are congregating due to biological dependence rather than because of the presence of a resource (Millspaugh *et al,* 1998), the herd may be considered as the sample unit (e.g. Neu *et al*, 1974; Gionfriddo & Krausman, 1986; Schaeffer & Messier, 1995). However, it has been shown that in some cases methods that treat individuals (or herds as in this case) as sample units may be less able to detect habitat selection (McClean *et al.,* 1998). Attempts to increase independence between samples will in some cases decrease information unnecessarily (i.e. Type II error Sokal & Rohlf, 1981) (e.g. Solla, 1999).

The third problem, the unit–sum constraint, can result in a habitat apparently being avoided simply because of positive selection for another, i.e habitats are not independent of one another. Aebischer *et al.* (1993) advocate using a log-ratio transformation of proportional used and available values to overcome this constraint, and this method (compositional analysis) is used increasingly (e.g. Donázar, Negro & Hiraldo, 1993; MacDonald & Courteney, 1996; Linnell, *et al.*, 1999; Mysterud *et al*, 1999; Todd *et al*, 2000). Nonetheless, there is apparently currently no consensus on the unit-sum constraint in the statistical community and there is some controversy over the effectiveness of the transformation (McClean *et al.,* 1998). Compositional analysis performed poorly in McClean *et al.*'s (1998) evaluation of six techniques for assessing resource selection.

Similar problems apply to the analysis of behavioural data, although the technique of using log-ratio transformed data from individuals has been less widely applied (but see Wilkinson, 1986; Gauthier & Bedard, 1991; Putman *et al*, 1993; Elston *et al*, 1996). In addition, there is ongoing debate about the appropriateness of using individuals within a herd as sample units due to the unknown extent of social facilitation between members of a group (Phillips, 1998; Weary & Fraser, 1998; Rook, 1999; Iason & Elston, 2000; Phillips, 2002).

In some cases, the constraints are not biologically significant in the context of describing livestock grazing on heathland sites in relation to vegetation management needs. So, for example, it is not necessarily important if scrub is apparently avoided because acid grassland is positively selected, if the aim is simply to describe what occurred at that site. Given the plethora of analysis techniques available, those used were chosen to (i) best suit the idiosyncrasies of the data collected, and (ii) provide the most straightforward results that are relevant to the context of the aims. In some cases, a comparison was made between two techniques where they offered different advantages.

2.2 Sites and livestock

2.2.1 ARNE RSPB RESERVE

Arne RSPB Reserve is located on the Arne Peninsula in Purbeck, east Dorset (Grid Ref. SY 966880) (Figure 2.1). The grazing unit forms only part of the site, and covers 130 ha. It is characterized by two dry heath plateaux totalling 58ha. These largely support the National Vegetation Community (NVC) H2 *Calluna vulgaris – Ulex minor* (Rodwell, 1991), dominated by heather *Calluna vulgaris*, with bell heather *Erica cinerea*, bristle bent *Agrostis curtisii*, dwarf gorse *Ulex minor* and scattered patches of common gorse *Ulex europaeus.* In places, bracken *Pteridium aquilinum* dominates, and there are scattered stands of Scots' pine *Pinus sylvestris* and maritime pine *Pinus pinaster* and silver and downy birch *Betula pendula* and *B. pubescens.* Seven hectares in the north of the site have been restored to heathland from pine wood, and are characterised by very young *Calluna* and bryophytes (mainly *Campylopus introflexus* and *Bryum* spp.) and, in disturbed patches, sheeps' sorrel *Rumex acetosella*, heath groundsel *Senecio viscosus*, foxgloves *Digitalis purpurea* and nettles *Urtica dioica.* The two main areas of dry heath are bisected by a valley supporting wet woodland characterised by *Betula* spp. and grey willow *Salix cinerea* and a small areas of wet heath and valley mire. Two other areas of lower ground (including Coombe Bog, see Chapter 3) support larger areas of M16 *Erica tetralix – Sphagnum compactum* wet heath and M21 *Narthecium ossifragum – Sphagnum papillosum* valley mire (eight hectares in total), characterised by cross-leaved heath *Erica tetralix*, Dorset heath *Erica ciliaris* and *Molinia caerulea.* Sphagnum carpets consist largely of by *Sphagnum pulchrum*, *S. subnitens, S. papillosum*, with *S. cuspidatum* in wetter pool, and *S.*

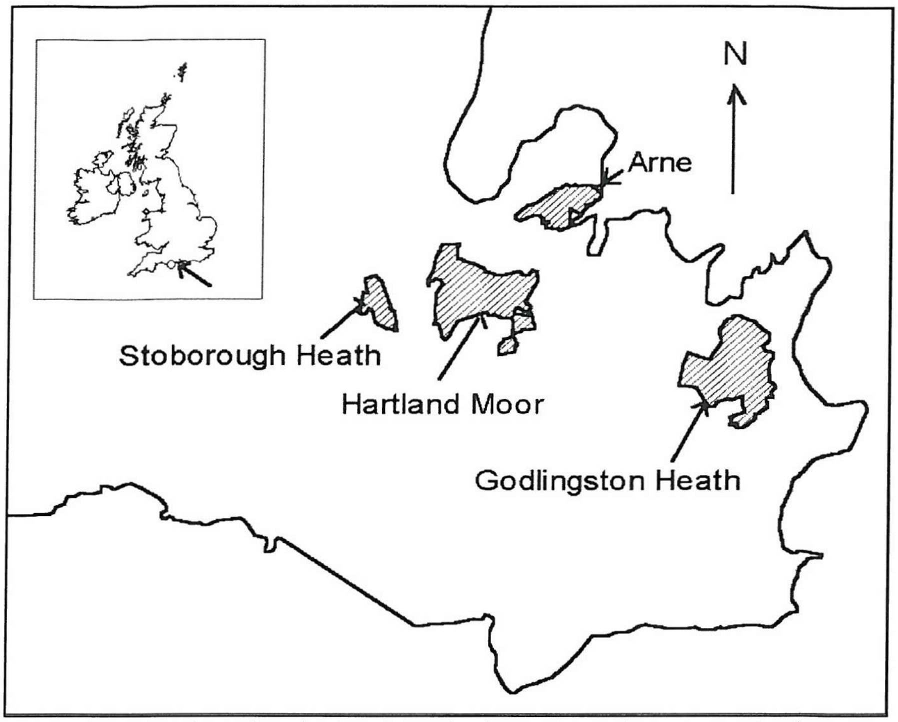


Figure 2.1 Location of study sites in Purbeck, south east Dorset, England.

compactum and *S. tenellum* on drier areas. The grazing unit is fringed to the south by saltings and reed swamp, and two small species-rich acid grassland fields are located on the northern boundary. Water sources for the livestock comprised four ponds located in wet heath, the valley mire pools, and a drinking trough in the acid grassland.

Plate 2.1 Hereford cross cattle herd on restored heath at Arne, Purbeck, Dorset

The livestock at Arne in 1998 comprised 13-14 mixed Hereford cows and a North Devon bull. The herd was new to the site, and had previously only been grazed on improved grassland. The herd was supplied by an independent grazier who sold it for slaughter over the following winter. In 1999, 12 mixed Hereford cross cattle, joined in August by an Aberdeen Angus bull, were brought to the site. Four of the same cows and the bull returned the following year, together with two other cows and six calves. These cattle were from the New Forest, Hampshire, and were therefore accustomed to grazing extensively on heathland vegetation. The cattle were out-wintered in the Forest. The grazing season was late May to November, (September in 1998).

2.2.2 HARTLAND MOOR NNR

Separated from Arne by a small area of ungrazed heathland and a forestry block, the Hartland Moor grazing unit (380ha) is located south east of Arne (Grid Ref. SY 948855) and includes Middlebere heath and Slepe Heath. It is a relatively complex site, comprising a central 'Y' shaped valley mire system surrounded by wet and dry heath. The two arms of the mire system show distinct vegetational differences, one acidic and characterised by *Sphagnum* species, and including rare species such as bog orchid *Hammarbya paludosa* (see Chapter 4) and bog sedge *Carex limosa* (M21 *Narthecium ossifragum – Sphagnum papillosum mire community)*, the other relatively base rich and dominated by tussocky black bog-rush *Schoenus nigricans* and *Molinia caerulea* (M25 *Molinia caerulea – Potentilla erecta* mire*)*. This area is separated by small roads to the east and west from areas of grassland reverting to heath vegetation, supporting a mosaic of relatively species rich acid grassland (U1 *Agrostis capillaris – Rumex acetosella – Festuca ovina*) and *Calluna*. Smaller mire systems are present adjacent to these, and there are further areas of dry and wet heath to the south, accessible to livestock from the road. Two main areas of acid grassland are included within the grazing unit, east and west of the central heath. The site also includes wet valley woods similar to those at Arne, reed swamp, and scattered deciduous copses. Water sources are four ponds in the wet heath, valley mire pools, and a drinking trough in the eastern area of reverting heath. Road puddles also provide drinking water in wet weather.

During the period of the study, Hartland Moor was grazed by a herd of 56 cattle, which included North Devon Reds and mixed Herefords crosses. The herd has been grazing the site since 1994, and had increased from 28 during this time. In 2000, it comprised mainly cows with yearling heifers and bullocks plus an impressive if elderly longhorn bull. A herd of seven Exmoor ponies has also grazed the site since 1994, and includes both mares and geldings. A group of 16 New Forest mares and geldings has grazed the site for a similar period. The cattle are wintered on an adjacent field complex between

Plate 2.2 Exmoor ponies on wet heath at Hartland Moor, Purbeck, Dorset.

late October and late May. Both pony herds were theoretically out-wintered on the site, although in December 2000 they found their way onto the adjacent fields with the cattle, and unfortunately the Exmoor ponies were not successfully lured back into the Hartland Moor grazing unit until the end of March. No supplementary feeding was carried out.

Plate 2.3 Longhorn bull with Red Devon and Hereford cross heifers, Hartland Moor

2.2.3 GODLINGSTON HEATH

South east of Hartland Moor and separated by 4.5km of forestry, Godlingston Heath (Grid Ref. SZ 030845) is another large heathland system (390ha). The distribution of habitats is not as complex as Hartland, although the site has greater topographical relief. It is characterised by four main M21 valley mires characterised by *Sphagnum* species, bog asphodel *Narthecium ossifragum*, marsh gentian *Gentiana pneumonanthe*, and common cotton-grass *Eriophorum angustifolium*. These are fringed by M16 wet heath supporting *Erica* species and *Molinia caerulea*, grading in turn to *Calluna*-dominated H2 dry heath on well-drained slopes. There are two areas of acid grassland fields separated by the bulk of the heath vegetation. There is only occasional scattered scrub, and tree-cover is limited to the boundaries. Drinking troughs are provided in both areas of acid grassland.

The site was grazed by up to 28 Exmoor ponies (year round) plus mixed cattle in the summer months (the cattle were not included in this study). Supplementary feeding was not carried out.

2.2.4 STOBOROUGH HEATH RSPB RESERVE

The Stoborough grazing unit covers a much smaller area, comprising 73 ha. Much of the site is seasonally damp and dominated by *Molina caerulea* humid heath (H3 *Ulex minor – Agrostis curtisii)*. There is a small valley mire and ponds of varying size are scattered throughout the site. The grazing unit slopes up to the south, where there is the most substantial area (2.5 ha) of H2 dry heath. There are extensive areas of dense scrub (9.7 ha), including three blocks of cleared *Ulex europaeus*, plus a central belt of wet woodland. Grassland is much more limited than at the other sites (under 0.5ha), with three small patches dispersed at the peripheries of the site. There are four small ponds accessible to livestock, and a stream running through the wet woodland, but no drinking trough.

Stoborough heath was grazed mainly by New Forest ponies, with occasional cobs. The herd composition and size varied, and ponies were introduced or removed from the site on several occasions during the study. Several foals were also born on site. The ponies were present year round, and received supplementary feed in the form of hay during the winter months. Supplementary feeding was carried out on a patch of acid grassland adjacent to the wet woodland and an area of dry heath and scrub.

Plate 2.4 New Forest mare and foal on Stoborough Heath, Purbeck, Dorset

2.3 Methods

2.3.1 DATA COLLECTION

Habitat survey

A GIS was constructed (Mapinfo, 1997) for data manipulation and spatial analysis. Site boundaries and livestock locations were superimposed over 1:10, 000 ordnance survey maps and aerial photographs. For each site, habitat polygons were digitised according to vegetation cover using a combination of aerial photographs and existing phase one habitat maps where available from English Nature. The resulting habitat maps were then refined in the field. Habitat patches under 10m x 10m were excluded. All habitat areas and length measurements were extracted from the GIS.

Dry heath types were defined according to Watt (1947) as building (including pioneer), mature or degenerate, and included scattered scrub. Building heath included areas that had been managed by burning or cutting within the last four years (and included the restored area at Arne). Restoration heath was used to describe areas of acid grassland reverting to heather heath characterised by a mosaic of grassland and *Calluna vulgaris* (present only on Hartland Moor). Small areas of slightly more mesotrophic or wetter grassland were included with the acid grassland category. ‘Saltings margins’ describes a distinct strip of mixed acid grassland and scrub that fringed the saltings (salt marsh) at Arne. A small area of reed swamp was grouped with the adjacent saltings at Arne. Areas of the Arne saltings unreachable by the livestock due to the width of creeks were excluded. ‘Tracks’ included mown firebreaks that were also used by vehicles. ‘Roads’ included the grassy margins and occasional scrub patches between the surfaced road adjacent heath vegetation. 'Wood' included small copses with heathland understories that occurred scattered on open heathland. Areas of recently cleared scrub and woodland were included as a separate category 'managed scrub/wood' where these occurred at Stoborough. Wet heath and valley mire are described in Chapter 1.

Table 2.2 Habitat types used in analysis of habitat selection by cattle and ponies on heathland sites in Purbeck, Dorset.

Habitat	Abbreviation	Habitat	Abbreviation
Acid grassland	AG	Roads	R
Restoration heath	AG/DH	Saltings	S
Building dry heath	bDH	Scrub	SC
Degenerate dry heath	dDH	Saltings margins	SM
Combined dry heath types	DH	Tracks	T
Wet heath + valley mire	M	Valley mire	VM
Mature dry heath	mDH	Wood	W
Managed scrub/wood	mSC/W	Wet heath (including humid heath).	WH

Daily location data

To test for non-random use of habitat by livestock throughout the grazing season, daily herd locations (during day light hours) were mapped and superimposed over habitat maps (see Table 2.3). Similar habitat types were in some cases combined where expected values were otherwise very low. Wet heath and valley mire were combined as mire (M) at Arne. All mature and degenerate dry heath were combined as dry heath (DH) on Hartland Moor and Stoborough Heath, and all dry heath types as DH on Godlingston Heath.

Table 2.3. Heathland sites for which daily location records of livestock herds were obtained in Purbeck, Dorset.

Site	Livestock	Season	Year	Data obtained by
Arne RSPB Reserve	Cattle	June- November	1998-2000	Self, RSPB
Hartland Moor NNR	Cattle	(May) June-October (November)	1994-1996	English Nature
Godlingston Heath NNR	Exmoor ponies	June-December	2000	The National Trust

At Arne and Godlingston, the livestock remained in a single herd, and therefore a single location was mapped at every observation time. At Hartland, the cattle formed between one and five herds, which were in theory all recorded. However, the total number of livestock recorded at each observation suggests that not all of the subsidiary groups were found on each occasion (it can take several hours to search the whole site thoroughly). Observations were approximately daily (weekly during one period at Hartland) and time of day varied. Observations were generally made by staff and volunteers or graziers. Due to time constraints, some observations were clearly opportunistic – i.e. locations were noted whenever the herd was encountered, and are likely to have resulted in a bias toward locations records closest to roads and work centres, and away from more distant areas and habitats where visibility is reduced (e.g. woodland).

Herd follows

To obtain more detailed information on relative habitat use and behaviour within each habitat, herd follows were carried out throughout the grazing season (see Table 2.4). Follows were tried out at two other sites, but were discontinued – at one because the herd of young heifers did not become habituated to the presence of an observer despite repeated visits, at the second (Godlingston Heath), because the Exmoor ponies left the acid grassland fields so rarely that insufficient data on their use of heath habitats could be collected using this method. Attempts to track individual animals fitted with radio-transmitters proved unsuccessful due to difficulties reliably fitting animals with the transmitters.

Table 2.4 Livestock types and sites used for livestock follows to establish habitat selection and behaviour on heathland sites in Purbeck, Dorset.

Livestock type	Site	Year	No. of follows	Period
Mixed cattle, mostly Hereford X	Arne RSPB Reserve	1999	6	June-November
		2000	6	June-November
North Devon and Hereford X cattle	Hartland Moor NNR	2000	5	June-October
New Forest ponies	Hartland Moor NNR	2000-01	11	June - November, January - May
Exmoor ponies	Hartland Moor NNR	2000-01	8	June - November, April - May
New Forest and mixed ponies	Stoborough Heath (RSPB)	2000-01	6	July - June

Follows were carried out once a month for each livestock group during the time the livestock were present on the site, except for the Stoborough ponies, which were followed once every two months. Over 24 hour periods the location of the herd was mapped and the habitat type and number of livestock carrying out each behaviour was noted.

Behaviour categories were: grazing, browsing, moving, standing, lying, standing ruminating and lying ruminating (cattle only), drinking, grooming, interacting with another animal, scratching, urinating, defecating plus an 'other' category including rolling, geophagy, and interacting with people. Suckling young was included at Arne in 2000 when calves were present. Note that grazing included eating ericaceous species as well as graminoids and herbs, while browsing including eating all other woody species. This differentiation was made due to the interest of conservationists in the role of livestock in the invasion of heathland and other semi-natural vegetation communities by non-ericaceous woody species (e.g. UK Steering Group, 1995).

For cattle, observation periods were from c13:00 hrs following a habituation period of about 30 minutes, longer if necessary, and continued until the herd settled for the night, usually about an hour after dusk. Observation continued from dawn the following day until 13:00 . Pony observations were for similar periods, although the ponies did not always remain in one place overnight, and on occasion had to be re-located the following morning. One or more 24-hour observations were also carried out for each livestock group. Note that all follows are referred to as 24-hour follows in the text, although observations were not necessarily carried out for the entire period. Observations were made from a variable distance using binoculars when necessary (magnification 8x, field of view 6.3°), and a night vision scope after darkness (magnification 3.6x, field of view 14°, infra-red illuminator range c30m). None of the livestock groups became habituated to strong torchlight. Care was taken at all times not to disturb the livestock. Trial observations were carried out in 1998 to establish appropriate techniques for observing livestock without disturbing them, including how close they would tolerate the observer, reactions to observer movement, etc. Once habituated, both cattle and ponies appeared to ignore the presence of an observer, provided care was taken.

Focal animals

During each follow, the behaviour and location of two randomly chosen and identifiable individuals was noted every five minutes to test for individual variation within the herd. Individuals were identifiable by numbered ear tags and a numbered list drawn up. Individuals were then selected at random from the list before each 24-hour follow. In addition, the number of steps and bites taken during one minute were recorded for one individual to establish a foraging rate, and the plant species eaten was noted (except on acid grassland and saltings, where species were highly intermingled).

Individual behaviour was only recorded during daylight hours, as the night vision equipment was not adequate for detailed observation. In 1999, the pair was changed between morning and evening sessions to increase replication, but for subsequent livestock groups, the same pair was followed throughout the entire session to maximise foraging data.

For habitats of key interest (e.g. dry heath, wet heath, valley mire, tracks, managed scrub/woodland, and scrub), availability of the main species present was estimated as percentage cover and compared to the number of observations of livestock eating each species. This is clearly only a two-dimensional estimate, and does not take into account the actual surface area or volume of available vegetation (for example, a greater biomass of *Ulex europaeus* occurs than grasses in the same sized area). However, this technique was considered adequate to establish basic selection patterns. Species cover was estimated from 20 randomly located 2m x 2m quadrats within each habitat type. Species were grouped if they occurred at less than 5% cover or less than 5% frequency (i.e. only occurred in one quadrat). Twenty quadrats was considered sufficient to estimate the cover of grouped species as the vegetation was relatively homogenous within patches. Only habitat patches where grazing was observed were surveyed, not the entire available habitat.

Dispersal of plant species in dung

To complement the data on dunging from 24-hour follows, a survey of plant species dispersed by dung was undertaken. The survey was made of the previous year's dung on Hartland Moor in May 2000 by walking five 100m transects in each of acid grassland, dry heath, wet heath and valley mire, and recording the plant species found germinating from dung within five metres either side of the transect.

2.3.2 ANALYSIS

Habitat use

Daily location records

Observed vs. expected habitat use each year or month was tested using a chi-square goodness-of-fit test (MINITAB, 1998) on daily location records (e.g. Neu *et al.*, 1974). Where differences were not significant between years, data were pooled across years to increase sample size. Where data sets were sparse and contained small values, a chi-square test using StatXact (Cytel, 1999) was used, which calculates an exact P value and is a more reliable method than traditional asymptotic calculation of P values in this case. Where data sets were large, but still contained small expected values, calculation of exact P values was too difficult within the current limitations of the software. In these cases P was calculated using Monte Carlo sampling with 100,000 permutations (option available within StatXact). Observed use was compared directly across months (as the habitat proportions did not change) using an R x C test for independence.

Jacobs' 'D' indices (Jacobs, 1974) were used to quantify selection for habitats on each site from the proportions of used (u_i) and available (a_i) habitat (e.g. MacDonald & Courtney, 1996; Walls & Kenward, 2001):

$$D_i = (u_i - a_i)/(u_i + a_i - 2.u_i.a_i)$$

This index was chosen because it is standardised (indices sum to 1) and allows straightforward comparisons between months and years. Values of +1 indicate a strong positive selection for the habitat and –1 indicates a strong negative selection.

At Hartland, the location records were weighted according to how many groups the total herd had split into at each observation. Missing herd records were ignored, since the analysis gave sensible results, but were kept in mind during interpretation of the results. In addition, location records on the road were removed if there were no other records for two days before or after. These were considered to be opportunistic observations by observers travelling on the road during one of the periods the herd were not searched for, and biased selection strongly toward roads (the smallest habitat category).

Monthly livestock follows

The departure of total habitat use from random was tested by a chi-square goodness-of-fit test (Neu *et al*, 1974). Jacobs' indices were used to quantify the selectivity. It is noted that the chi-square test treats sequential observations as independent samples, and therefore in this case artificially inflates the available degrees of freedom. However, as the results were very clear, this is not considered to have led to misinterpretation. The significance of the selection indices for each habitat over the season were tested using each 24 hour observation period as a sample. The departure of Jacob's scores from a median of zero (which indicates that a habitat is used in direct proportion to its availability) was tested for each habitat using a Wilcoxon signed rank test (e.g. Walls & Kenwood, 2001).

This analysis does not take into account the unit-sum constraint. To test whether this analysis technique provides qualitatively different results, habitat selection ranking was determined by compositional analysis, using Resource Selection for Windows (Leban, 1999) which constructs t-test confidence intervals around the log-transformed differences between used and available habitat data. This ranking was compared to a ranking constructed by testing pairwise differences between Jacobs' indices for each habitat using a Mann-Whitney test. The results of both techniques were qualitatively very similar for each livestock group, and therefore Jacobs' indices were used throughout for consistency with the daily location analysis.

To explore whether the sample differences showed seasonal trends, seasonal differences in proportional habitat use were tested for each habitat by ANOVA on arcsine square-root transformed proportional use data (necessary as the herd size changed between samples on some sites). In most cases, a division between summer (including any of May – September) and winter (including any of November – April) was made. Any variation in these seasonal categories is stated with the results.

The quality of the information was maximised by using observations of all (unidentified) animals to provide data on the proportion of the herd using each habitat, since individual differences in both habitat selection and behaviour were observed (see results). As each monthly observation was used as a sample unit, replication was over time, rather than between herds, meaning the results are clearly only relevant to the herd used in each study (Phillips, 2002). When herd size varied between habitats, analysis was also carried out by allocating a single habitat to the herd (i.e. the habitat that the majority of the herd occupied).

Behaviour

Behaviours were grouped for analysis (Table 2.5). Grazing and browsing were combined as foraging (F) where appropriate. Differences in time spent on each behaviour (measured as the number of observation of each behaviour in each habitat for each 24-hour observation period) were tested by General Linear Model (GLM) on arcsine square-root transformed data. Although this did not take into account the unit sum constraint, a log-ratio method analogous to the compositional analysis of habitat selection (e.g. Putman *et al*, 1993) was not used, as it uses $n+1$ behaviours to compare n behaviours, and in this case, a comparison was required between all behaviours recorded. Therefore the first method was used and the unit sum constraint taken into account in interpretation. The distance travelled during each 24-hour follow was extracted from the GIS, and seasonal differences tested (one-way ANOVA).

Table 2.5 Categories used in analysis of livestock behaviour.

Behaviour	Abbreviation	Notes
Resting	R	Including lying, standing, ruminating
Moving	M	
Grazing	G	Including animals moving with their head down while grazing
Browsing	Br	
Drinking	Dr	
Other	O	Including interacting with other animals, scratching, rolling, geophagy, urinating, defaecating.
Calves	C	Suckling/grooming calves
Out of site	OS	Use when an animal was away from the main herd and not visible

Overnight resting behaviour

Twenty-four hour observations plus information from the trial observations the preceding year showed that cattle stayed in one location after darkness until dawn, which was verified for 15 of the 16 follows. Overnight observations showed the herd rested for this period, although individuals occasionally grazed for a few minutes after suckling young in 2000 (see Figure 2.2 and Appendix I, Figure 1). Therefore, it was assumed that cattle remained resting in the same habitat during unobserved hours overnight.

In contrast, 24-hour observations of ponies showed that they tended to continue foraging throughout the night. Although they usually remained in the same habitat, they were occasionally found elsewhere the subsequent morning. Since observations were for varying periods, and changes in day length may affect behaviour, a comparison was made between analyses using data from a 12- hour period (7 a.m. to 7 p.m.) and analyses using all available data. The results were very similar, and therefore all available data were used to maximise the quality of the information.

Foraging

Log-transformed bite and step rate during foraging for identified individuals were compared between habitats (GLM). For key vegetation types selection of plant species was analysed using the methods used for habitat selection, with individuals as replicates. Use of plant species was measured as the number of bites taken of each plant species. Variation in habitat selection and behaviour between individuals was evaluated for each pair observed simultaneously with a chi-square test.

Resting locations

The proximity of resting locations to shelter was compared to that of random points. Locations where the livestock were known to have remained overnight or for over 2 hours during the day were used. Shorter, more frequent, resting periods were excluded, as it was less likely that they were independent of one another. 23 locations for combined livestock types at Arne and at Hartland were used. Shelter included wood, hedges and scrub patches over $10m^2$.

2.3 Results

The results are presented in three sections. Firstly, daily location data are used to assess daytime habitat selection and seasonal variation at each site. Twenty-four hour habitat selection and seasonal variation is then described using data from herd follows for cattle and pony herds respectively. In these sections, behaviour is described, together with habitat selection for key behaviours, and seasonal trends. Individual variation within each herd is assessed, and focal animal data used to show foraging rates in each habitat and selection of plant species.

2.3.1 DAILY LOCATION RECORDS

Daily location records showed a significant departure from random habitat use during daylight hours on all three sites studied (Table 2.6). Figures 2.2a-c show herd locations superimposed over habitat maps at Arne, Hartland and Godlingston respectively. At Arne, habitat use differed between years (Chi square = 51.77, df = 14, $P < 0.001$) but at Hartland differences between years were not significant.

Table 2.6 Departure of habitat use from random using daily location records by livestock on three heathland sites in Purbeck, Dorset. *** = $P < 0.001$

Site	Chi-square	No. observations
Arne 1998	1136.1***	143
Arne 1999	211.8***	72
Arne 2000	770.69***	212
Hartland 1995-7	315.6***	289
Godlingston, 2000	698.6***	199

Figure 2.2 shows habitat selection for cattle at Arne expressed as Jacobs' indices. Acid grassland (AG) was selected the most strongly at each site, followed by habitats with a significant fine grass component (saltings margin (SM), restoration heath (AG/DH), road (R)). Habitat differences remained significant when acid grassland was removed from the analysis, showing that differences between the remaining habitats were also significant. At Arne, cattle also positively selected habitats containing coarse grasses (saltings (S) in 1998 and 2000, and wet heath (WH), in 1999 and 2000). Dry heath (DH), scrub (SC), wood (W) and valley mire (VM) were never positively selected although their relative ranking changed.

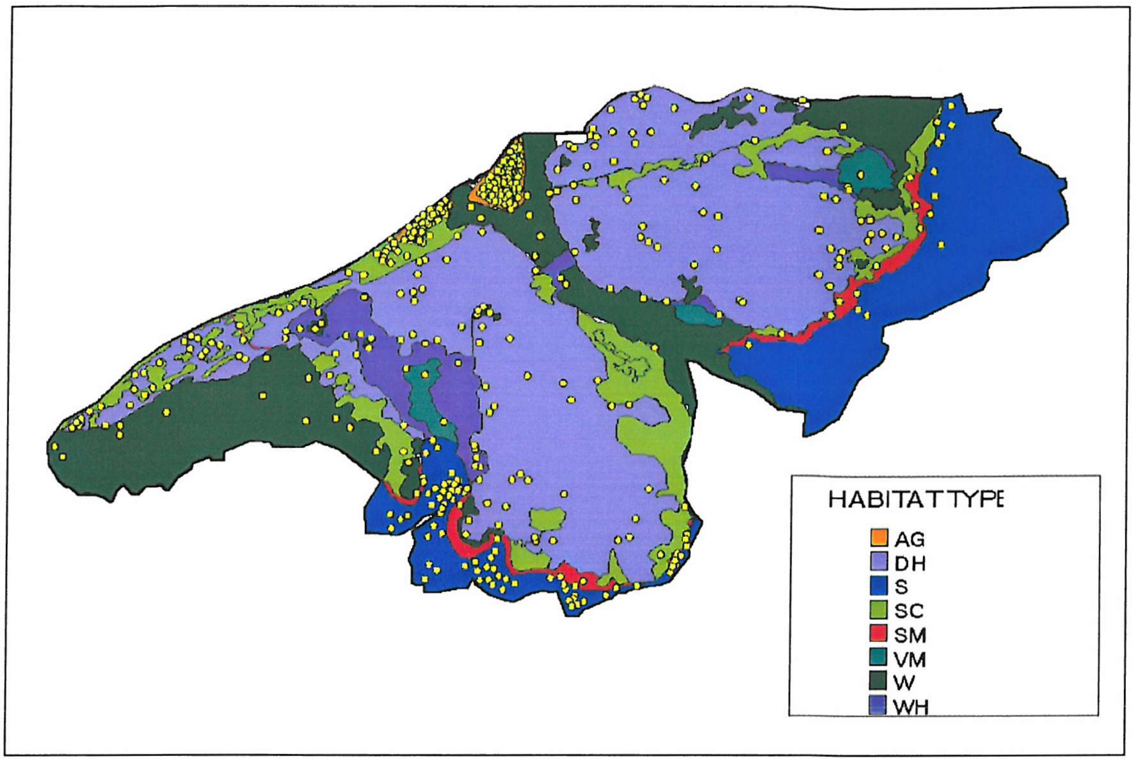


Figure 2.2a Daily cattle locations (yellow dots) and habitat type at Arne RSPB Reserve, 1998-2000

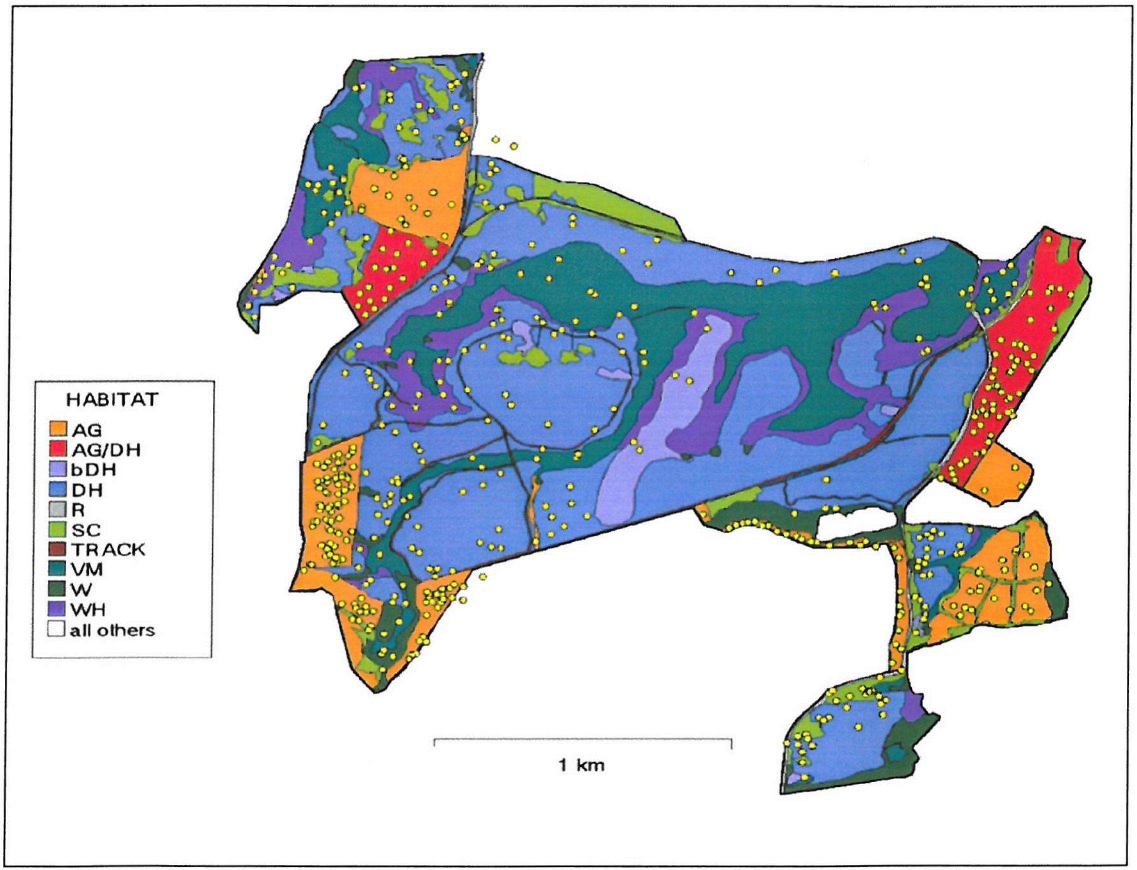


Figure 2.2b Daily cattle locations (yellow dots) and habitat type at Hartland Moor, 1995-1997

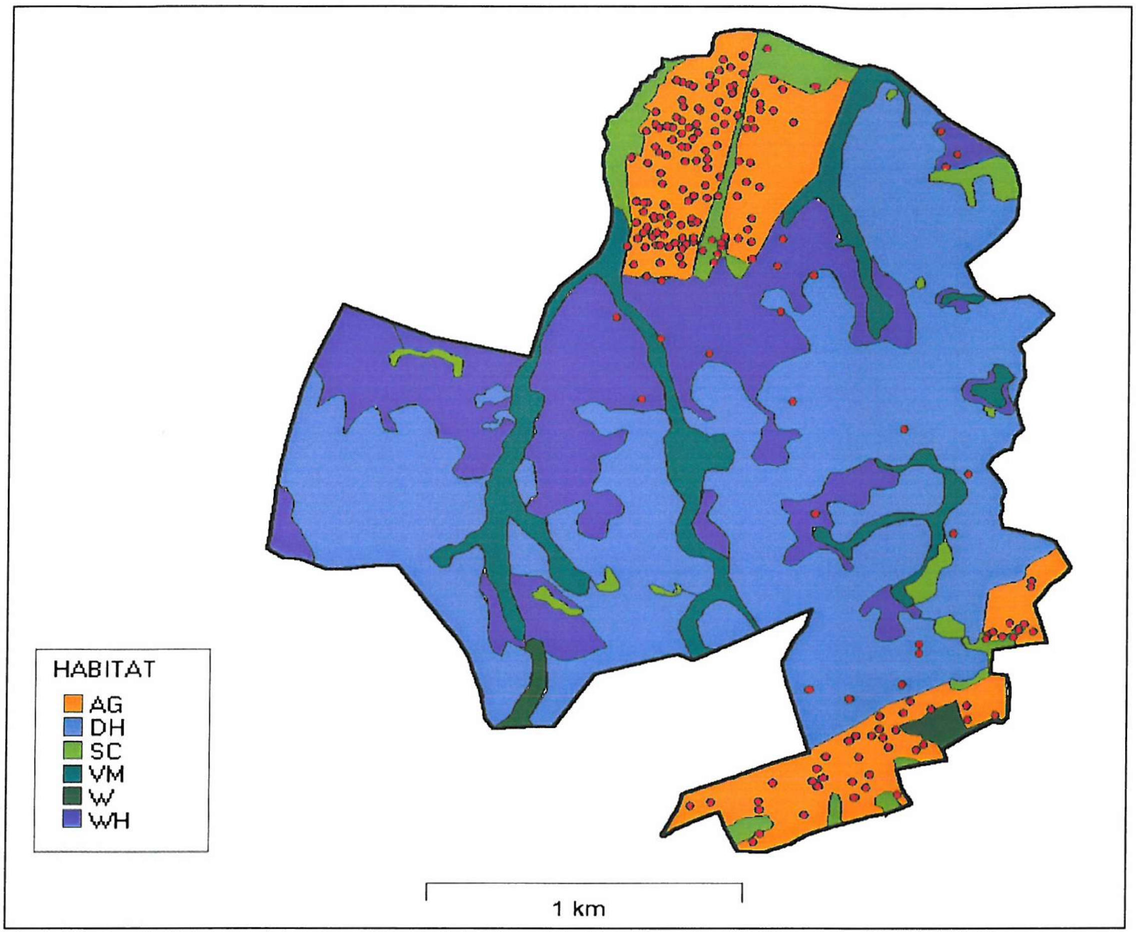


Figure2. 2c Daily pony locations (by red dots) and habitat type at Godlingston Heath, 2000

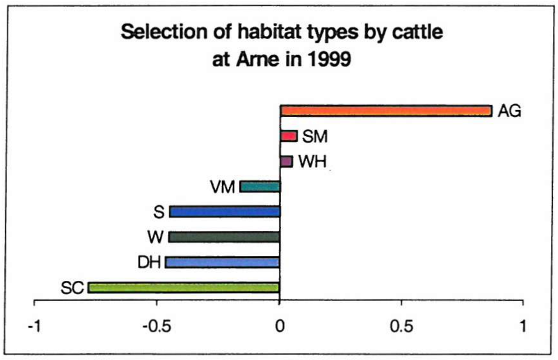


Figure 2.3 Habitat selection by cattle at Arne expressed as Jacobs' indices. See Table 2.2 for habitat abbreviations.

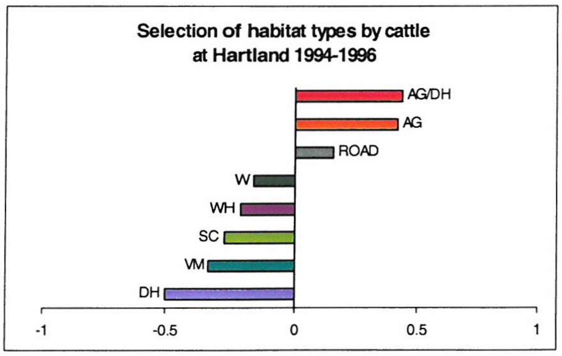


Figure 2.4 Overall habitat selection by cattle at Hartland Moor expressed as Jacobs' indices. See Table 2.2 for habitat abbreviations.

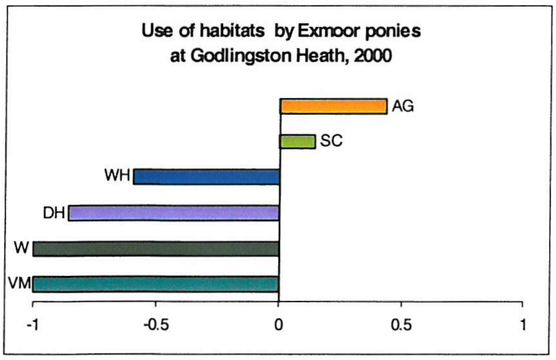


Figure 2.5 Overall habitat selection at by ponies at Godlingston Heath expressed as Jacobs' indices. See Table 2.2 for habitat abbreviations.

At Hartland, differences between habitats were less strong, but showed a similar pattern to Arne, except for the low selection of dry heath (DH) (Figure 2.4). On Godlingston (Figure 2.5), the ponies rarely left the acid grassland, and were never observed in valley mire (VM) or wood (W). Dry heath (DH) and wet heath (WH) only appeared to be used between the two grassland patches (see Figure 2.2c). However, in contrast to the cattle, the Exmoor ponies at Godlingston positively selected scrub. Most scrub observations were in a small area of scrub (SC) adjacent to the northern acid grassland (AG) patch.

Differences between months were significant in 1998 and 2000 at Arne (the 1999 data set was much smaller) and for both Hartland Moor and Godlingston Heath (Figures 2.6 – 2.7). Seasonal changes in habitat selection in 1998 and 2000 were broadly similar at Arne, with selection for wet heath and valley mire (M) increasing in September when selection for saltings (S) decreased. The use of dry heath (DH) increased markedly in November 2000, and was rising in September 1998, when the cattle were removed. The number of observations in scrub (SC) and wet heath and valley mire (M) were low, and as these habitats form a low proportion of the total area, small changes in observed numbers led to relatively large changes in apparent selection.

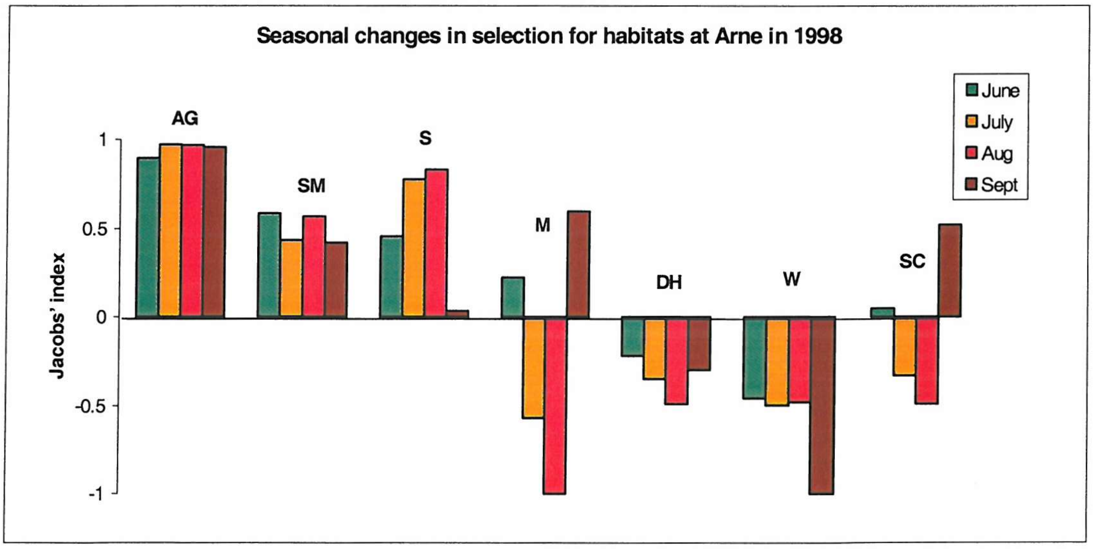


Figure 2.6a Seasonal habitat selection by cattle at Arne in 1998 expressed as Jacobs' indices. Wet heath and valley mire were combined as mire. R x C test for independence, chi square = 36.8, P(MC) = 0.005. For habitat abbreviations see Table 2.2.

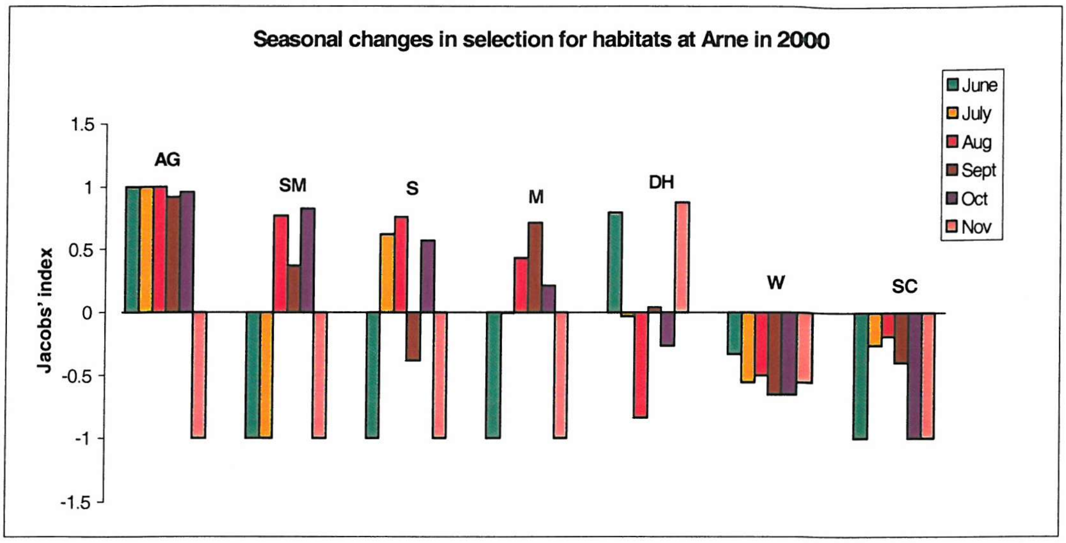


Figure 2.6b Seasonal habitat selection by cattle at Arne in 2000 expressed as Jacobs indices. Wet heath and valley mire were combined as mire. R x C test for independence, chi square = 70.2, P(MC) =0.0001. For habitat abbreviations see Table 2.2.

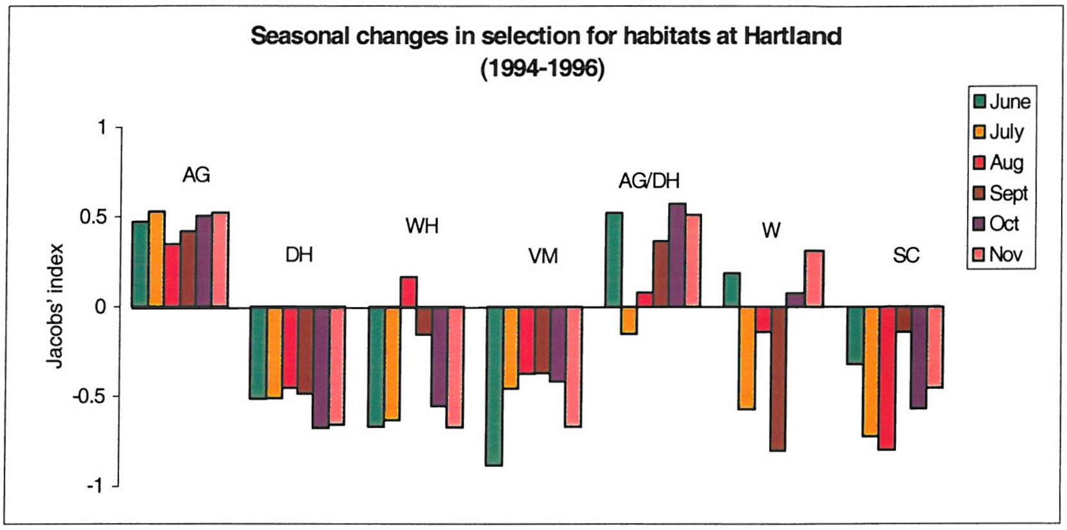


Figure 2.7 Seasonal habitat selection by cattle at Hartland expressed as Jacobs indices (1994-1996 combined). R x C test for independence, chi square = 61.7, P(MC) =0.0001. For habitat abbreviations see Table 2.2.

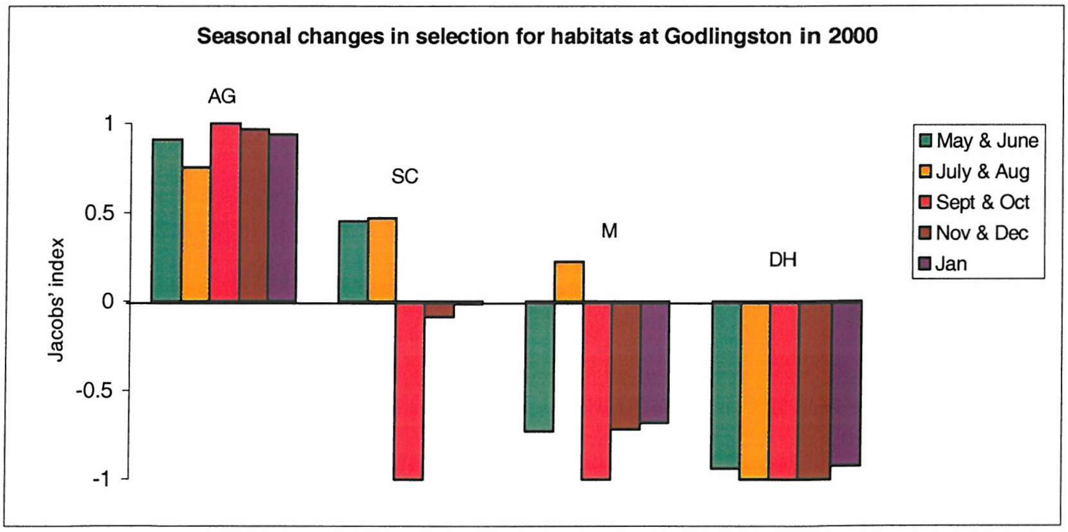


Figure 2.8 Seasonal habitat selection by Exmoor ponies at Godlingston Heath expressed as Jacobs indices. R x C test for independence, chi square = 32.2, P(MC) = 0.005. For habitat abbreviations see Table 2.2.

Changes at Hartland were not as marked, and selection for dry heath (DH), in particular, did not increase in autumn. Selection for wet heath (WH) and valley mire (VM) peaked in August and September, and was associated with a slightly lower selection for acid grassland (AG).

Similarly, selection for wet heath and valley mire (M) by Exmoor ponies at Godlingston peaked in July and August, although again, low numbers of observations in M and scrub (SC) mean small changes in observed numbers lead to large changes in the selection index. Selection for acid grassland (AG) remained high, reducing slightly in July and August.

Daily location data at Hartland showed a division of the cattle herd into smaller groups, and this varied seasonally. Total herd size differed between years (28-50), and therefore the seasonal change in the number of individuals within secondary groups, rather than the number of secondary groups, was tested. There was a clear reduction in herd size in September (Figure 2.9). Group size was not recorded for a sufficient number of the secondary herd in June and July 1994-5 and in 1996 to test for differences in size during these periods.

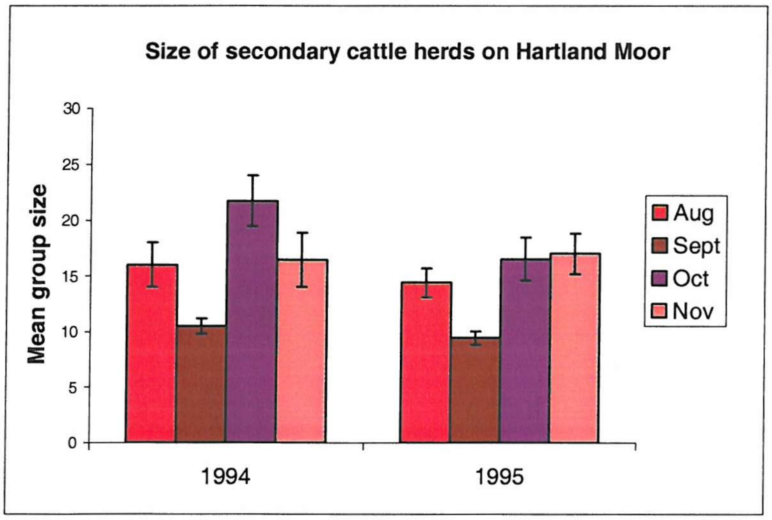


Figure 2.9 Difference in herd size for cattle at Hartland Moor between months. F = 7.80, P = 0.038 (one –way ANOVA)

2.3.2 CATTLE FOLLOWS

Routine

Results are presented for the Arne 1999 cattle follows and reference is made to data from Arne in 2000 and Hartland Moor where these are different. Complete results from Arne in 2000 and from Hartland can be found in Appendices I-II.

Figure 2.10 shows behaviour during each 24-hour observation period. In general, two to three main foraging bouts were interspersed with shorter periods of rest and movement during daylight hours. Cattle rested overnight, although continuing to graze for about two hours after sunset. Within each 24-hour period, the main foraging bouts were always carried out in different areas and often on different habitat types. Short periods of foraging associated with over-night resting occurred in the same habitat patch as that used for resting.

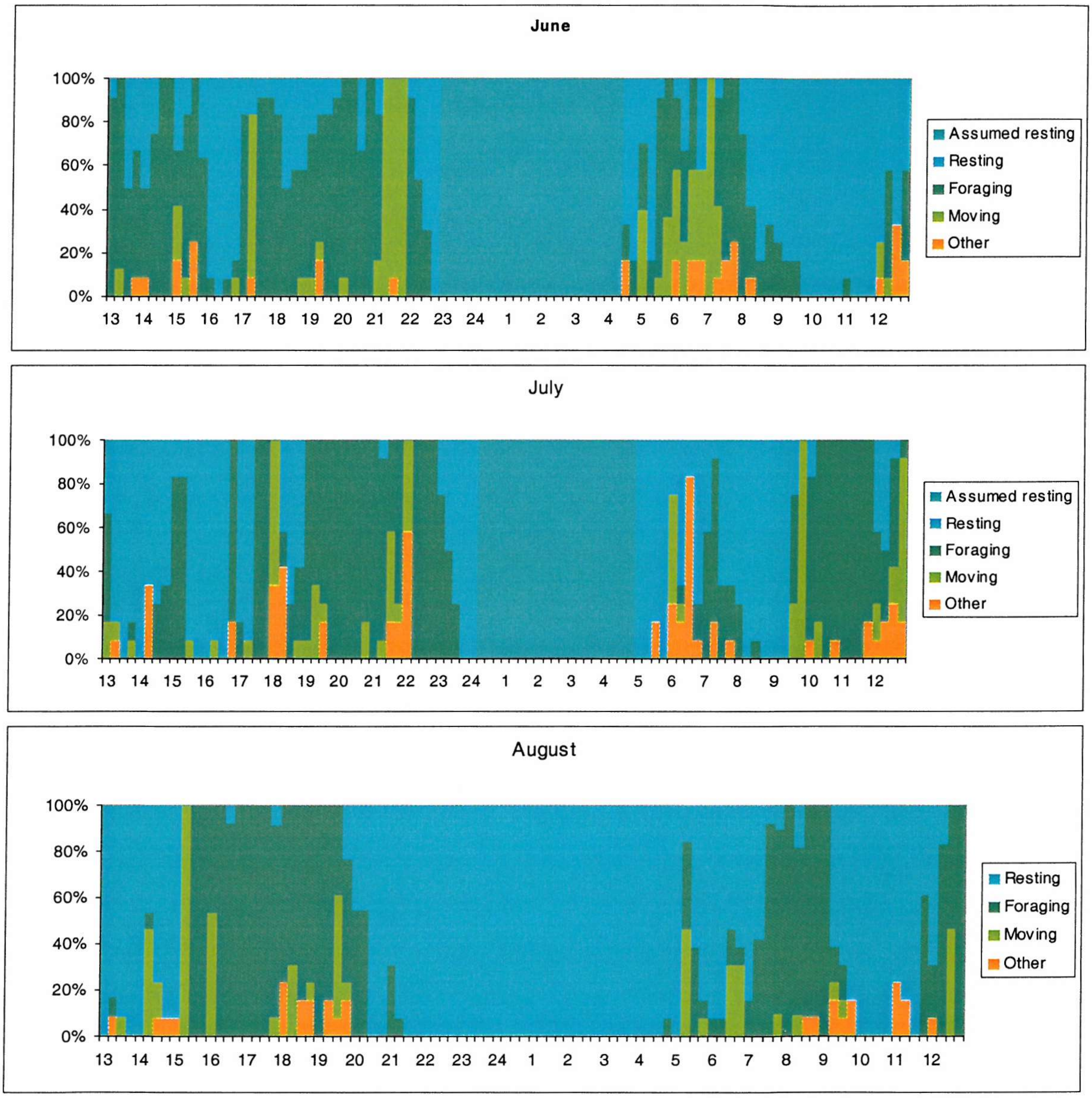


Figure 2.10. See next page for legend

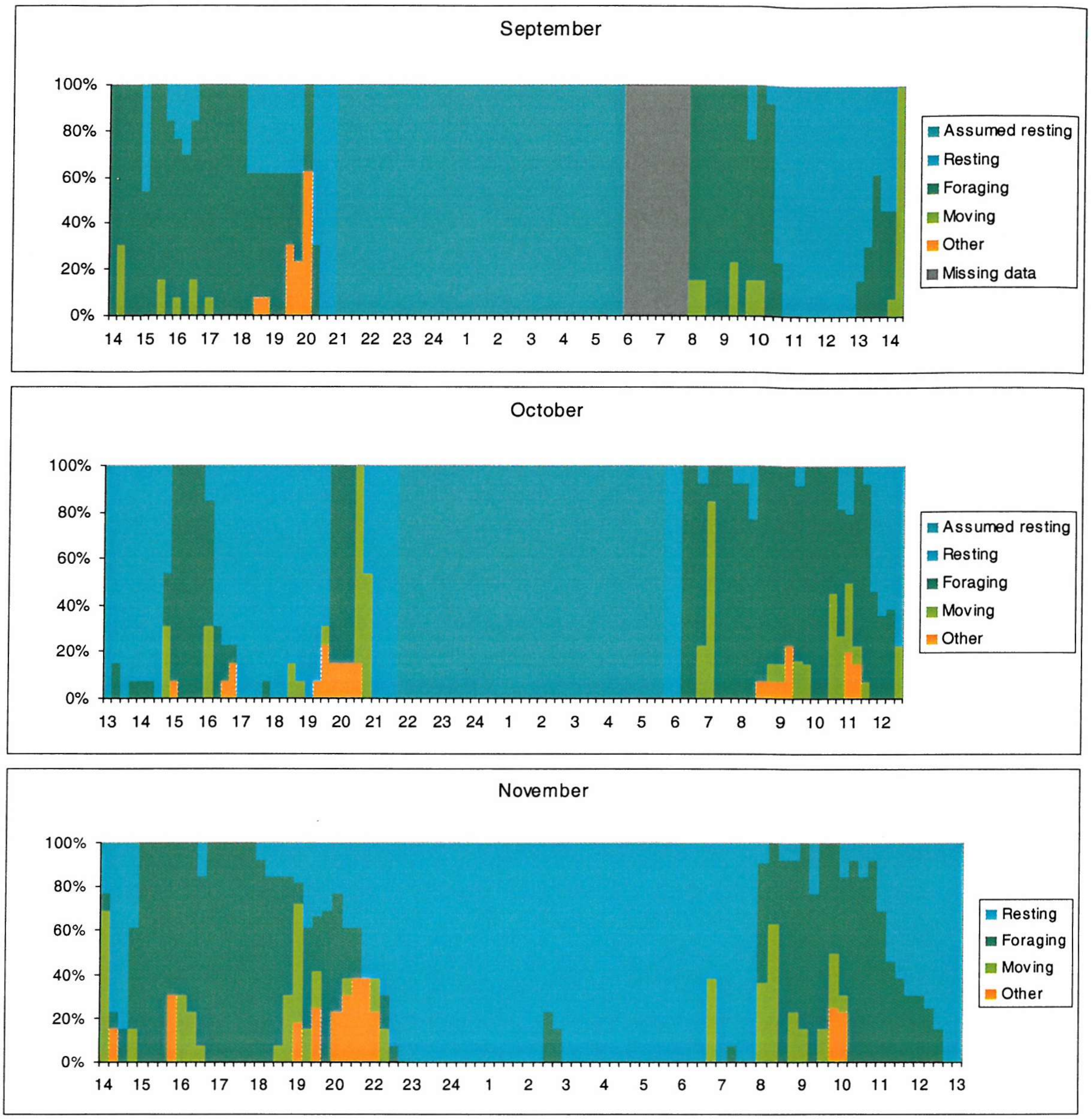


Figure 2.10 Behaviour of cattle at Arne during 24-hour observation periods. The proportion of the herd carrying out each behaviour is shown along the vertical axis and the time in hours along the horizontal axis. 'Assumed resting' represents night time periods.

Figure 2.11 shows the routes taken by cattle during each 24-hour observation period, and can be compared with Figure 2.2a for habitat types. Clusters of points indicate where resting or foraging occurred. The cattle were observed on all parts of the site during 24-hour follows, daily location searches or other casual observations. At Arne, the cattle herd rarely divided, although in 2000 one cow was sometimes observed remaining with all the calves while the remaining adults moved to another habitat to drink, and then returned. On Hartland, several smaller herds usually formed, although these frequently converged on the grassland (see Appendix II).

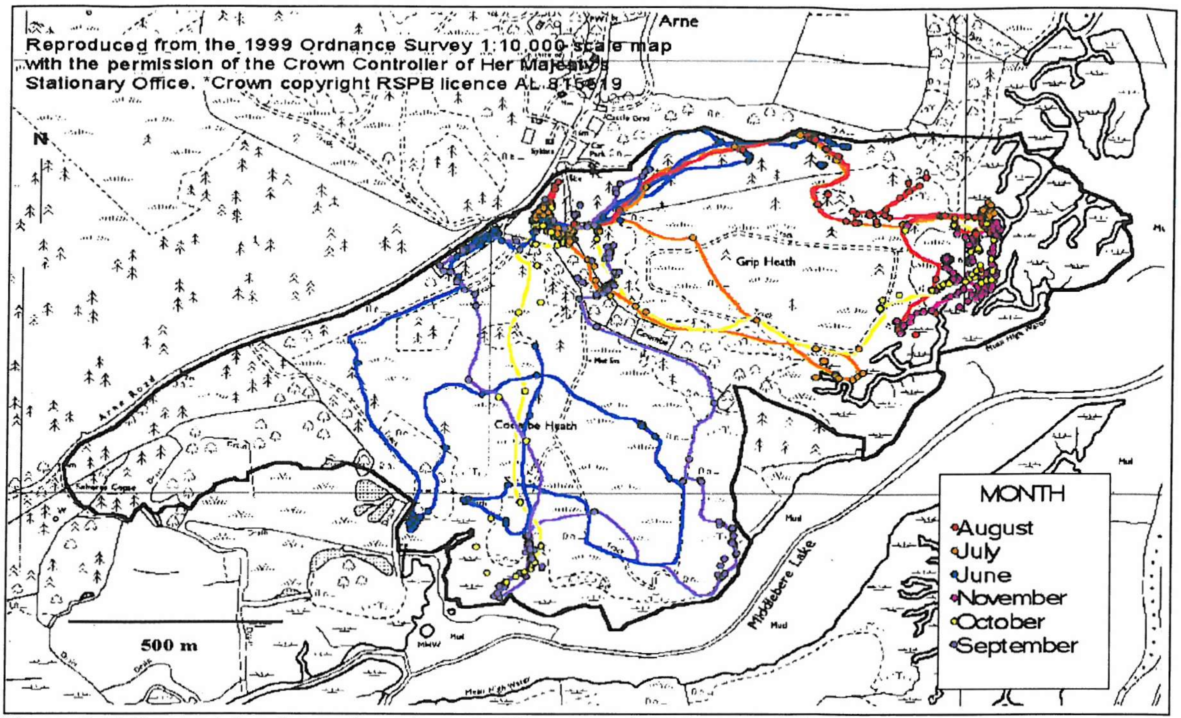

Reproduced from the 1999 Ordnance Survey 1:10,000 scale map with the permission of the Crown Controller of Her Majesty's Stationary Office. ©Crown copyright RSPB licence AL 815619

Figure 2.11 Routes taken by cattle at Arne, 1999, over each 24-hour period.

Individual variation

Although the herd usually occupied a single habitat type, on a number of occasions two or three habitats were occupied by the herd at a given observation time (Figure 2.12). There was no significant change in the number of habitats occupied between summer and autumn (ANOVA)

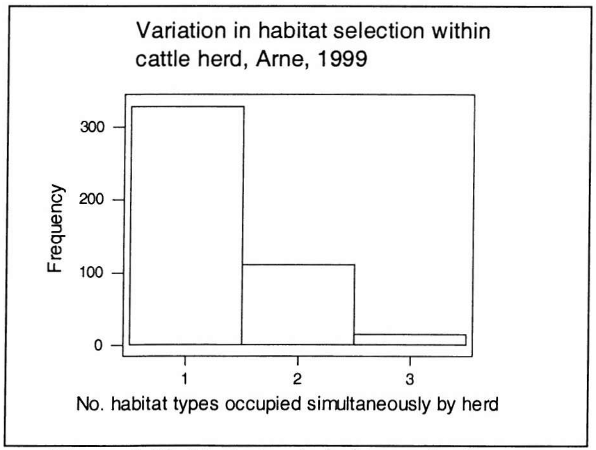


Figure 2.12 Variation in habitat selection within the Arne 1999 cattle herd measured as the number of habitats occupied simultaneously by the herd.

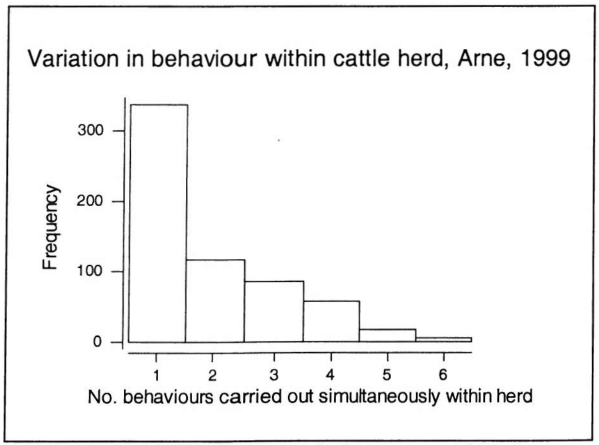


Figure 2.13 Variation in behaviour within the Arne 1999 cattle herd measured as number of behaviours carried out simultaneously..

The number of habitats used simultaneously was greater in 2000, with up to 6 habitats used at any one time (by the adults), although again only one habitat was most commonly used. Note that although this herd was the same size as in 1999, half the animals were calves.

Behaviour was more varied, with up to six behaviour types (total, not grouped) observed at any one time within the herd in 1999 (Figure 2.13). This suggests that there may have been significant individual variation within the cattle herd. The suckler herd in 2000 showed a similar pattern, although the occurrence of two behaviours was more frequent (largely due to cows suckling or grooming their calves).

Individual habitat use

Four out of 12 cow pairs showed significant differences in habitat use (note that one individual was in both pair 2 and pair 4), suggesting that total habitat use did vary, and variation within the herd was not necessarily a result of lack of synchrony between individuals (Table 2.7). This difference was greater in 2000, when three out of six pairs showed significant differences in habitat use (Table 2.8).

Table 2.7 Differences in habitat use between randomly paired cows observed concurrently at Arne in 1999. ** - P <0.01.

	Habitat difference	**Month**	**Chi -square**
Pair 1	Saltings/saltings margin	June	19.08**
Pair 2	Wet heath/dry heath	June	81.00**
Pair 3	Wood/saltings	September	57.28**
Pair 4	Saltings/saltings margin	November	34.08**

Table 2.8 Differences in habitat use between randomly paired cows observed concurrently at Arne in 2000. ** - P <0.01. Note that one particular animal was in Pair 1 and Pair 3.

	Habitat difference	**Month**	**Chi -square**
Pair 1	Wood/mire & building dry heath	July	27.63**
Pair 2	Wood/acid grassland Saltings margin/building dry heath Saltings & margin/degenerate dry heath Building dry heath/acid grassland, mire & wood	September	42.15**
Pair 3	Wood/acid grassland Degenerate dry heath/saltings	October	72.16**

Differences tended to occur when the herd occupied the acid grassland, when some individuals remained in the adjacent wood, or around the saltings margin, where a diversity of habitats were available in a small area. Differences on three of the seven occasions (Table 2.7) were noted when one cow stayed with all six calves, and the remaining cows moved onto a different habitat.

Individual behaviour

One pair (out of 12) showed a significant difference in behaviour in 1999 – this was Pair 3, and the difference occurred as one was grazing on the saltings, while the other was browsing in the adjacent woods. Both were, however, foraging. (Chi-square = 10.22, df = 6, P < 0.01). In contrast to habitat use, this suggests that apparent differences in behaviour within the herd are in fact due to lack of synchrony between individuals. No differences were found in 2000.

24-hour habitat use

Data from the follows were more detailed than the daily location records, as they included location records over 24 hours, used more detailed habitat definitions, and addressed variation between individuals within the herd. This resulted in differences in apparent habitat selection (Figure 2.14, see also Figure 2.3a)

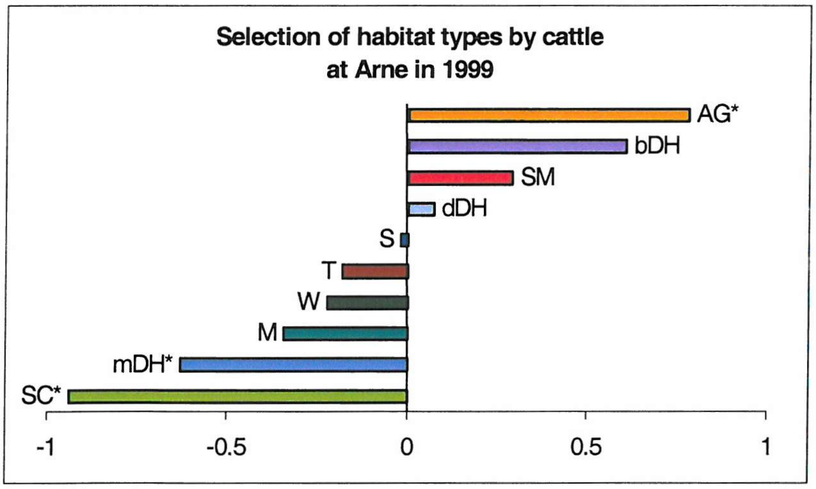


Figure 2.14 Habitat selection by cattle at Arne expressed as Jacobs' indices using pooled 24-hour data over the grazing season in 1999. Habitat use was non-random (chi square = 15107.9, df = 40, p<0.001). * denotes habitats for which selection was significantly different from zero(which indicates selection is in direct proportion to availability) (Wilcoxon signed rank test, P < 0.05). For abbreviations see Table 2.2.

The key changes were a large increase in selection for building dry heath (bDH) and degenerate dry heath (dDH), while mature dry heath (mDH) (the majority of the dry heath area) was avoided. Habitat selection was similar in 2000, when the herd also positively selected saltings (S), and all three grassy habitats were preferred over building dry heath (bDH) (Figure 2.15). Acid grassland (AG) was not significantly positively selected overall due to complete avoidance of this habitat in November.

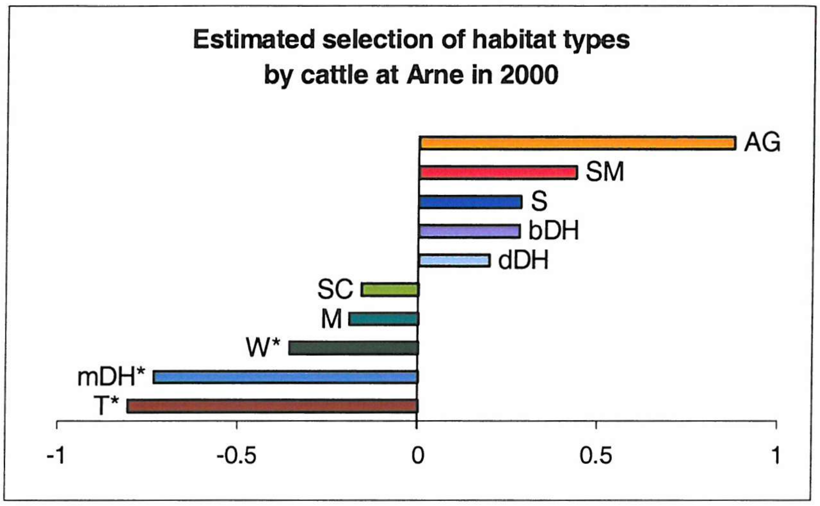


Figure 2.15 Habitat selection by cattle at Arne expressed as Jacobs' indices using pooled 24-hour data over the grazing season in 2000. Habitat use was non-random (chi square = 16556.4, df = 9, p<0.001). * denotes habitats for which overall selection was significantly different from zero (which indicates selection is in direct proportion to availability) (Wilcoxon signed rank test, P < 0.05). For abbreviations see Table 2.2. See Appendix II for explanation of estimation.

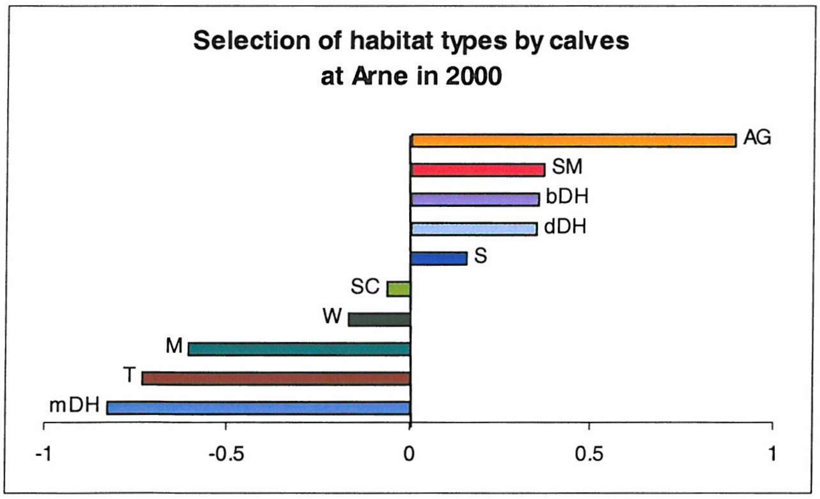


Figure 2.16 Habitat selection by calves at Arne expressed as Jacobs' indices using pooled 24-hour data over the grazing season in 2000. Habitat use was non-random (chi square = 51290.71, df = 9, p<0.001, chi square = 1617.59, df = 9, p<0.001). For abbreviations see Table 2.2.

Calves showed similar habitat preferences to the cows in 2000 (Figure 2.14), but with reduced selection for saltings (S) and increased selection for degenerate dry heath (dDH). This difference is mainly due to the November 24-hour observation, when the calves remained resting in nearby dDH and the cows grazed on S.

Habitat selection at Hartland was similar, although there was some variation due to the different habitats present (Figure 2.17). Building dry heath (bDH) was the most selected habitat, and roads (R) was also positively selected. Restoration dry heath (AG/DH) was negatively selected, as was wet heath (WH), valley mire (VM), dry heath (DH), scrub (SC), and wood (W). Only the use of tracks (T) was nearly in proportion to availability

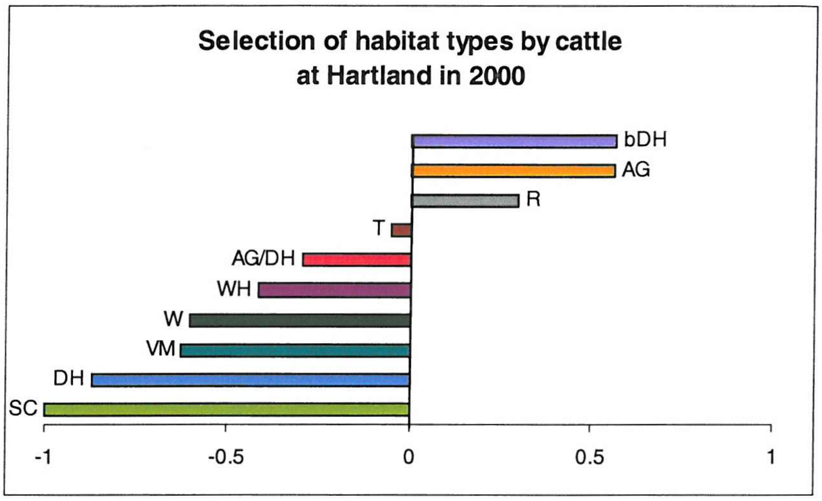


Figure 2.17 Habitat selection by cattle at Hartland expressed as Jacobs' indices using pooled 24-hour data over the grazing season in 2000. Habitat use was non-random (chi square = 1617.59, df = 9, p<0.001). For abbreviations see Table 2.2. Variation from zero was not be tested as n < 6.

The significance of the difference between the availability and use of habitats was inflated by the use of all quarter-hourly observations as samples (and this is seen in the very large chi-square test results). To address this, the medians of habitat indices for each 24-hour period were tested against a median of zero, thus using each observation period as an independent sample. This showed that at Arne, over the six-month period in 1999, acid grassland (AG) was significantly positively selected, and scrub (SC) and mature dry heath (mDH) significantly negatively selected (Wilcoxon signed rank test, P < 0.05) (Table 2.9). Compositional analysis of the data also showed a significant overall departure from random habitat use (chi-square = 161.58, df = 9 , P < 0.001). The ranking obtained from compositional analysis was the same as that obtained from a comparison of the Jacob's indices for habitats calculated for each 24-hour period, with the exception of the five habitats which showed selection closest to zero, degenerate dry heath (dDH), saltings (S), tracks (T), wood (W) and mire (M). As there was no significant difference between any of these five habitats using either method, the ranks for these habitats can be considered as interchangeable.

Table 2.9 Habitat selection rankings as calculated by compositional analysis and Mann-Whitney test between Jacobs' indices. There was no significant difference in rank between italicised habitats. See Table 2.2 for habitat abbreviations.

Compositional analysis	Jacobs' Index
AG	AG
bDH	bDH
SM	SM
S	*DDH*
M	*S*
DDH	*T*
MDH	*W*
T	*M*
W	*MDH*
SC	SC

The similarities between rankings also occurred at Arne in 2000 and Hartland (see Appendices I and II), and therefore only the Jacobs' indices were used for subsequent analysis.

The difference in the ranking using pooled data of the whole 1999 season (e.g. Figure 2.14) and the average ranking (Table 2.9) is due to the differences in habitat selection between months. Seasonal differences in this monthly variation in 1999 are presented in Figure 2.18a. Only acid grassland (AG), building dry heath (bDH) and wet heath and valley mire (M) were used every month. However, the only significant seasonal change was in the use of M (ANOVA, F = 33.66 P = 0.003), where summer use was greater than autumn use (T = 6.30, P = 0.003). There was also a trend for increased use of building dry heath (bDH) and degenerate dry heath (dDH) and a decreased use of wood (W) in the autumn.

Changes were similar at Arne in 2000 (Figure 2.18b). However, use of mature dry heath (mDH), rather than bDH increased in autumn, and the change in use of dDH was in this case significant (ANOVA, F = 21.28, P = 0.010), and was greater in autumn than summer. (T = 40631, P = 0.009). Use of acid grassland (AG) decreased in September and October and had ceased in November. Overall, these data are consistent with changes found using daylight location data.

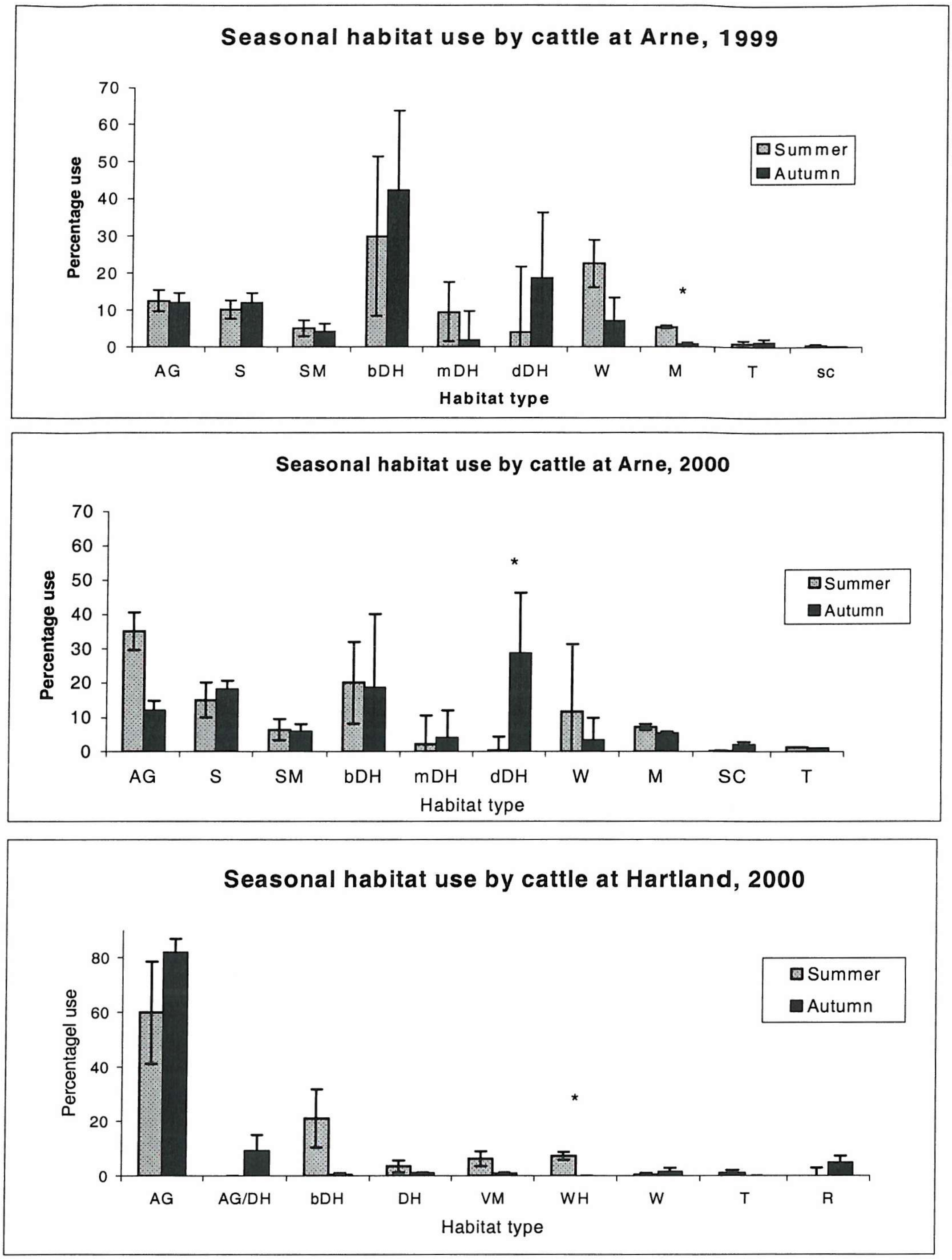


Figure 2.18 Seasonal variation in habitat use by.(a) cattle at Arne in 1999 (b) cattle at Arne in 2000, (c) cattle at Hartland in 2000 * denotes a significant differences at $P < 0.01$. See Table 2.2 for habitat abbreviations. Summer includes June, July August, and autumn includes September, October and November.

At Hartland, use of valley mire (VM) and wet heath (WH) also decreased in autumn (Figure 2.18c). However, in contrast to Arne, use of dry grassy habitats (acid grassland AG, restoration heath AG/DH, roads R) increased in autumn, and use of dry heath types decreased.

Behaviour

Figure 2.19 shows the average time spent on different behaviours by cattle at Arne in 1999. There was no significant difference between seasons.

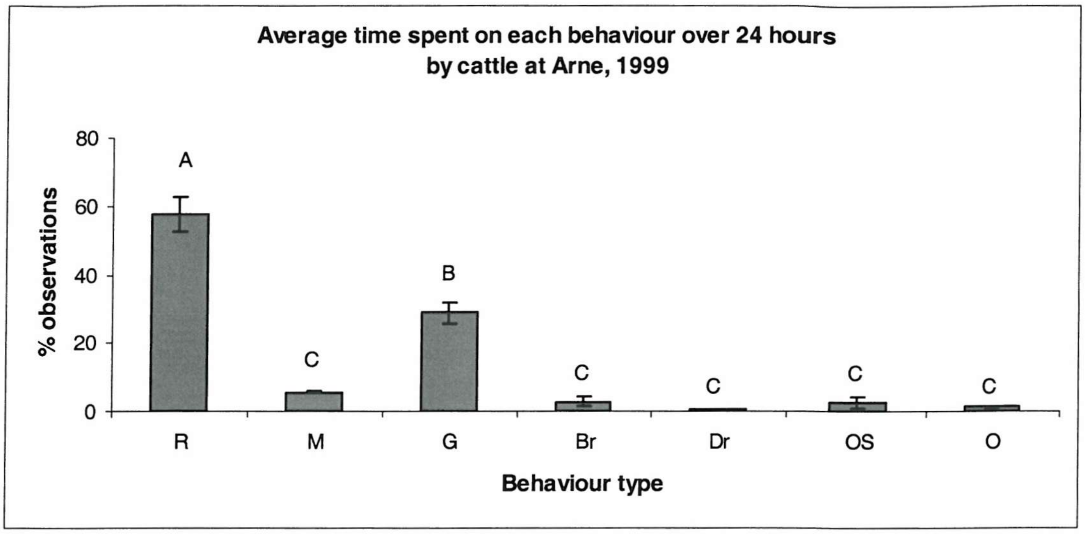


Figure 2.19 Mean time spent on each behaviour by cattle at Arne, 1999 (expressed as % of observations) . The difference in total time spent on each behaviour was significant (GLM, F = 10.39, $P < 0.001$). Significant differences are denoted by different letters, therefore behaviours with the same letter show no significant difference. R – resting, M – moving, G – grazing, Br – browsing, Dr – drinking, OS – out of sight, O - other

Cattle at Arne spent approximately the same amount of time on each behaviour in 1999 and 2000. The main difference was in resting (49.1% in 2000 compared to 57.8% in 1999). This can be accounted for by the increase in 'out of sight' (OS) records in 2000 (9.7% compared to 2.7%). Most of these occurred when the herd was resting in woodland (the missing individuals were generally also resting when found). There were more OS records in 2000 as the calves tended to be nervous which made moving around to find individuals without disturbing the herd more difficult. The percentage of drinking observations doubled between 1999 and 2000 (when the cows were lactating), although this remained a very small proportion of their time. Mean behaviour times were similar at Hartland, although slightly more time was spent grazing (at the expense of resting time).

Comparison of Figure 2.19 with 2.20 illustrates how habitats were used for different activities. In 1999, acid grassland (AG), saltings (S) saltings margins (SM) and scrub (SC) were used mainly for grazing, while dry heath (DH) was used mainly for resting, and tracks (T) for moving.

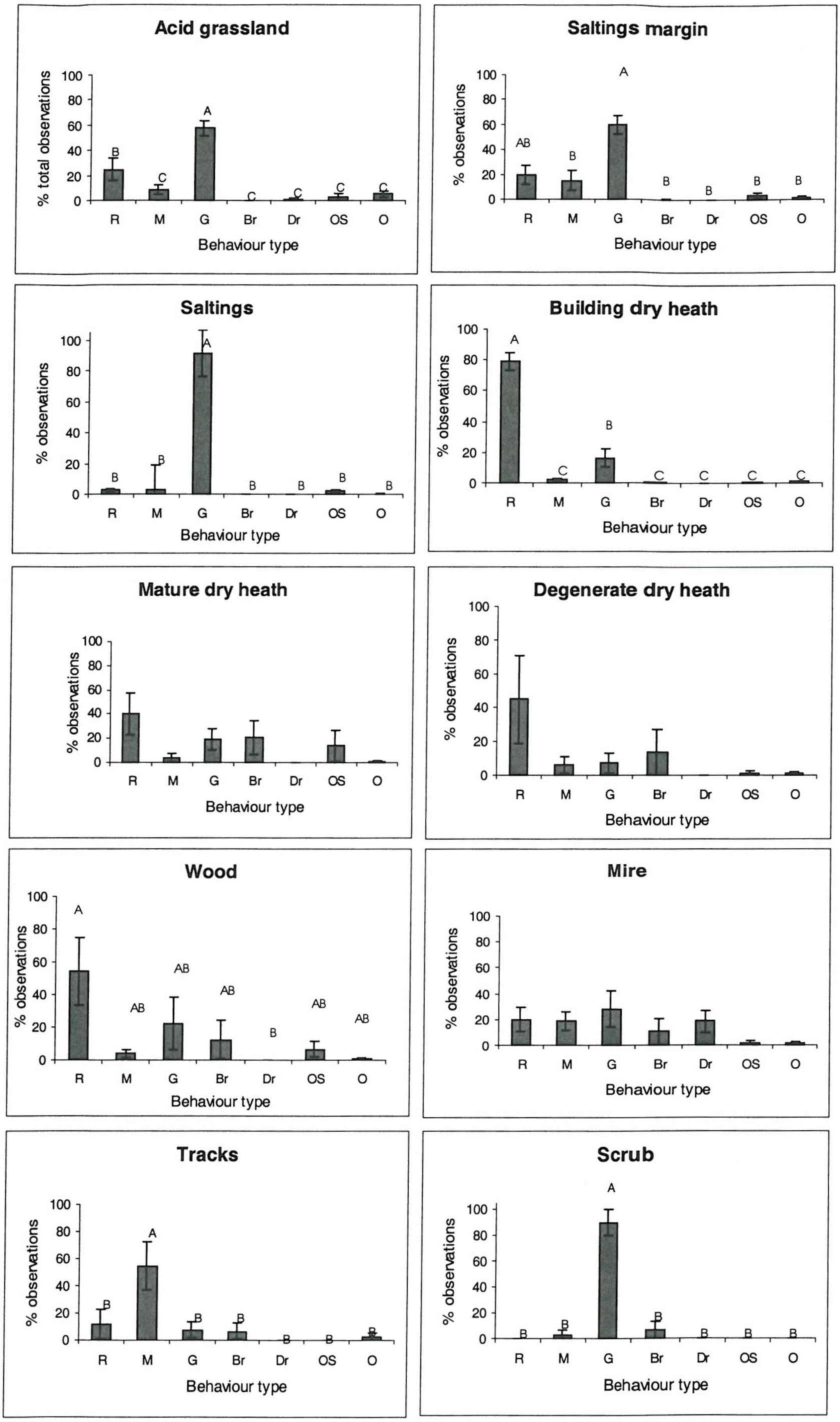


Figure 2.20 Mean time spent on each behaviour (expressed as % of observations) in different habitats by cattle at Arne in 1999. Significant differences are denoted by different letters. Where letter are omitted no significant difference between behaviour times was found.

GLM: - AG: F = 9.14***, S: F = 54.10***, SM: F = 6.92***; bDH: F = 5.63***; mDH: F = 11.21***; M: F = 11.56***; dDH, W, SC, T - differences not significant at P = 0.05.

Habitat use was similar at Arne in 2000, but there were some differences at Hartland (Figure 2.21).

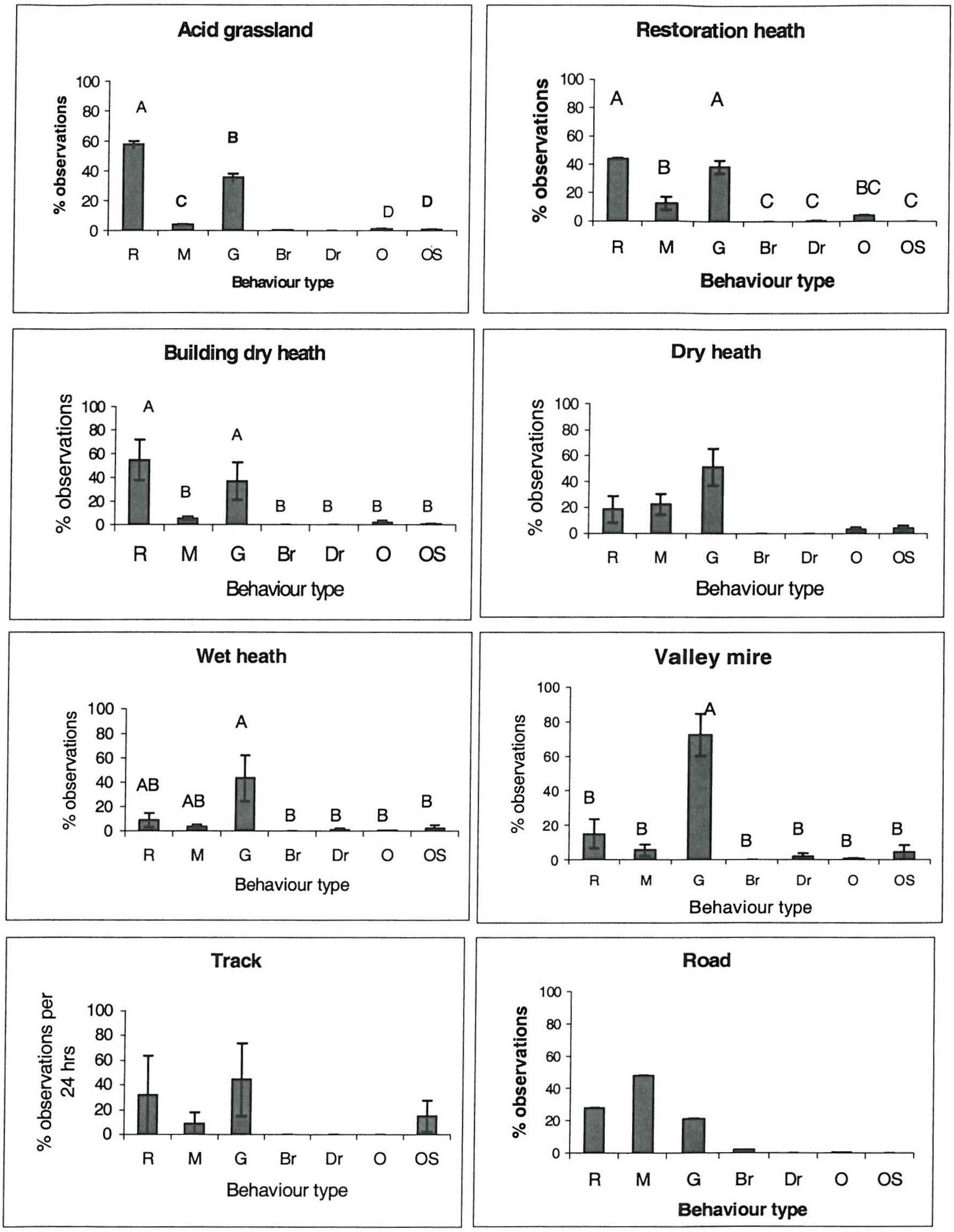


Figure 2.21 Mean time spent on each behaviour (% of observations) in habitats by cattle at Hartland. Significant differences are denoted by different letters, therefore behaviours with the same letter show no significant difference. Where letters are omitted no significant difference between behaviour times was found. GLM - AG: F = 227.31***; AG/DH: F = 64.67***; bDH: F = 8.89***, VM: F = 41.63***; WH: F = 4.73***; T, R – differences not significant at P = 0.05. *** = P < 0.001

Acid grassland (AG) was used more for resting than grazing, tracks (T) were used more for grazing than moving, and wet heath (WH) and (VM) were used predominantly for grazing. Restoration heath (AG/DH), not present at Arne was used similarly to AG. There were few observations on roads (R), which were used mainly for moving.

Figure 2.22 shows the Jacobs' indices quantifying the selection for each habitat for key behaviours. This gives a relative measure of what proportion of each behaviour is carried out in each habitat relative to its area, and may therefore be more useful information than Figure 2.20 in the context of managing livestock for heathland conservation. Comparison with Figure 2.14 (total selection) is given in Table 2.11.

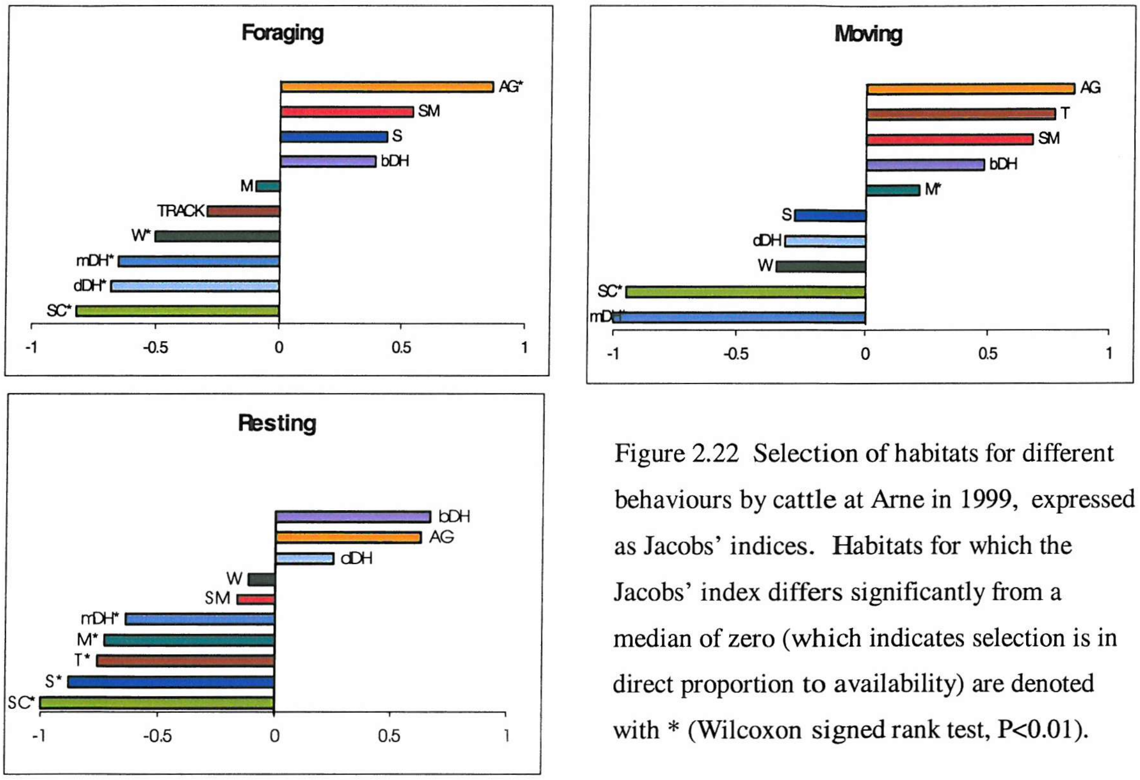


Figure 2.22 Selection of habitats for different behaviours by cattle at Arne in 1999, expressed as Jacobs' indices. Habitats for which the Jacobs' index differs significantly from a median of zero (which indicates selection is in direct proportion to availability) are denoted with * (Wilcoxon signed rank test, P<0.01).

Table 2.11 Differences in habitat selection for particular behaviours and total habitat selection (Fig. 2.14) by cattle at Arne in 1999. $^{+/-}$ indicates a change between negative and positive selection.

Behaviour	**Key change in habitat selection**
Resting	S decreases$^{+/-}$
	bDH increases
Foraging	SC decreases
	DDH decreases$^{+/-}$
Moving	T increases$^{+/-}$
	S decreases$^{+/-}$

Results were similar in 2000, although more habitats (saltings margins SM, wood W, saltings S, tracks T) were positively selected for resting. Wet heath and valley mire (M), but not building dry heath (bDH), was positively selected for foraging. Results from Hartland were also similar. Of the additional habitats, road (R) was positively selected for resting and moving, and negatively selected for foraging, restoration heath (AG/DH) was negatively selected for all behaviours.

Seasonal change in how habitats are used

The proportion of time within each habitat spent on key behaviours did not generally change significantly between seasons. The one exception was in wet heath and valley mire (M), where the proportion of time within the habitat spent grazing decreased to zero in (Figure 2.23). This is reflected in the overall reduction in use of M in autumn (Figure 2.18a), when use of this habitat was mainly for drinking.

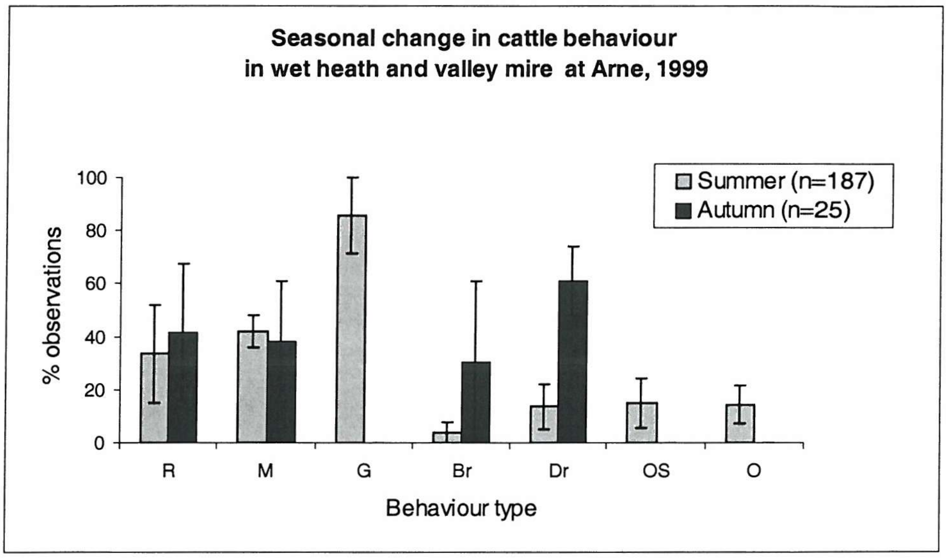


Figure 2.23 Seasonal use of mire by cattle at Arne in 1999. GLM, behaviour * season, F = 4.03, P = 0.005. R – resting, M – moving, G – grazing, Br – browsing, Dr – drinking, OS – out of sight, O – other. Summer includes June, July August, and autumn includes September, October and November.

Seasonal changes in the selection of habitats for particular behaviours are shown in Figure 2.24. In addition to the change in use of wet heath and valley mire (M) notable trends were the increased selection for mature and degenerate dry heath (mDH, dDH) for foraging in autumn (although these habitats were still negatively selected) and the change from negative to positive selection for dDH for resting.

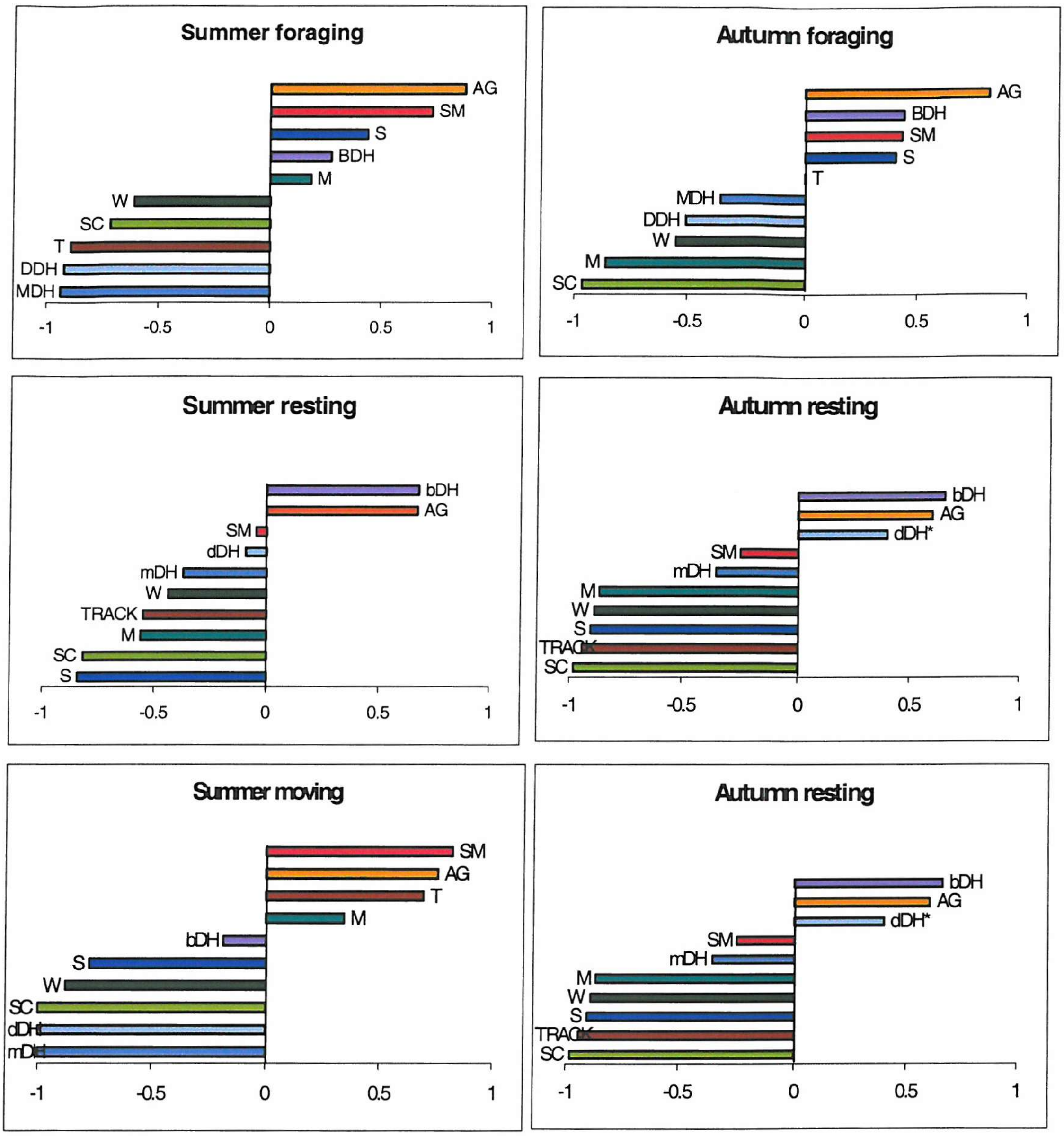


Figure 2.24 Seasonal habitat selection for key behaviours by cattle at Arne in 1999, expressed as Jacobs' indices. Summer includes June, July August, and autumn includes September, October and November. See Table 2.2 for habitat abbreviations

In 2000, there were no significant seasonal differences in use of habitats for particular behaviours. Seasonal changes in habitat selection for key behaviours were very similar to those in 1999. In addition, selection for saltings margins (SM) increased in autumn for all key behaviours. At Hartland, no significant seasonal changes in habitat use were found, and changes in selection for key behaviours tended to correspond to overall changes in habitat selection (Figure 2.25).

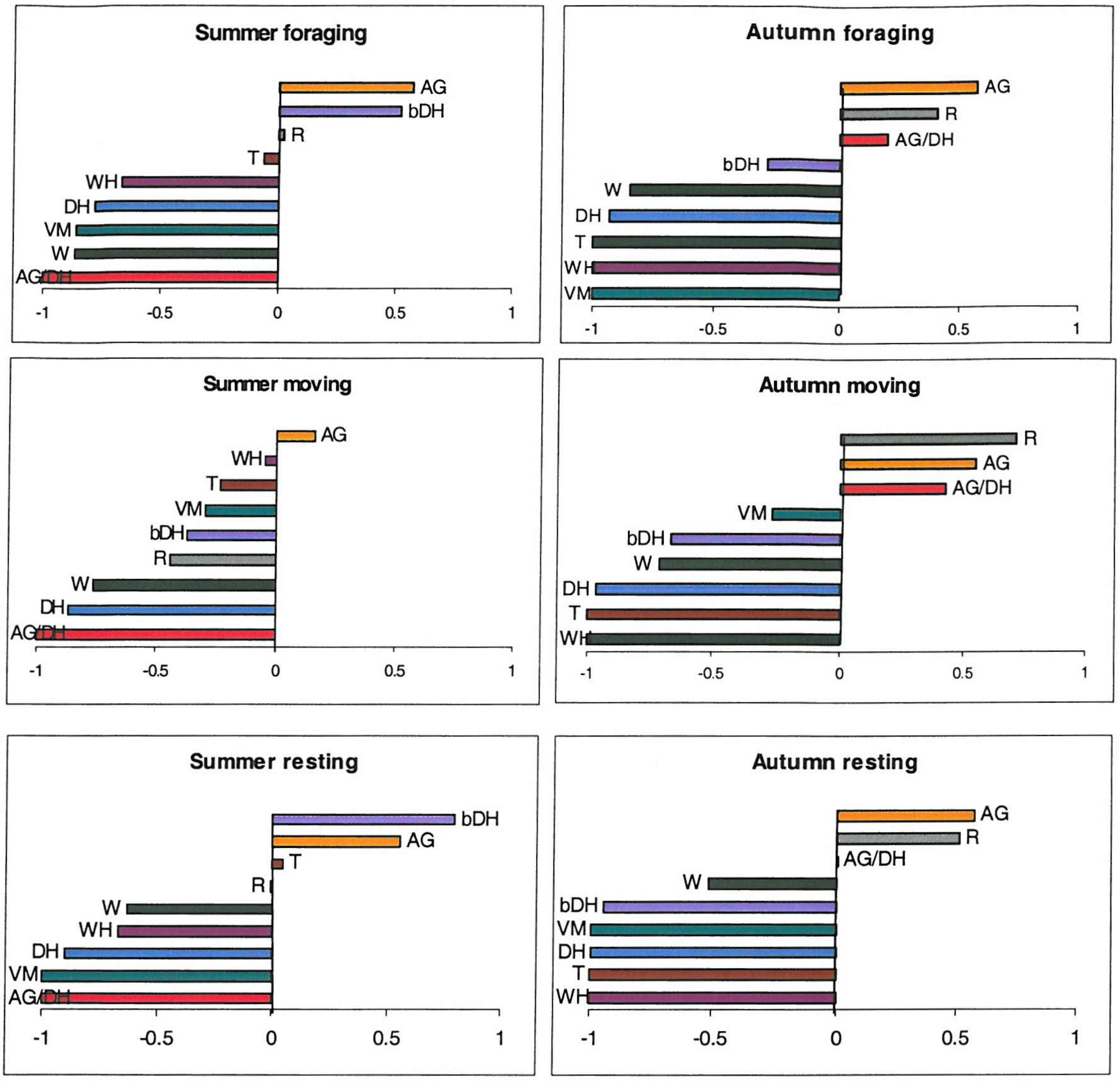


Figure 2.25 Seasonal habitat selection for key behaviours by cattle at Hartland in 2000, expressed as Jacobs' indices. Summer includes June, July August, and autumn includes September, October and November. See Table 2.2 for habitat abbreviations.

Distance travelled

The distance travelled over 24 hours by each cattle group is given in Table 2.12.

Table 2.12 Distance travelled in 24 hours by cattle at Arne and Hartland.

Livestock group	Range (km)	Mean (st.dev) (km)
Arne 1999	2.58-3.79	3.37 (0.84)
Arne 2000	0.92-5.66	2.85 (1.57)
Hartland 2000	4.12-7.05	6.18 (1.83)

The daily distance travelled reduced markedly in the autumn at each site (Figure 2.26) (differences are not significant at P = 0.05 (ANOVA) due to small sample sizes).

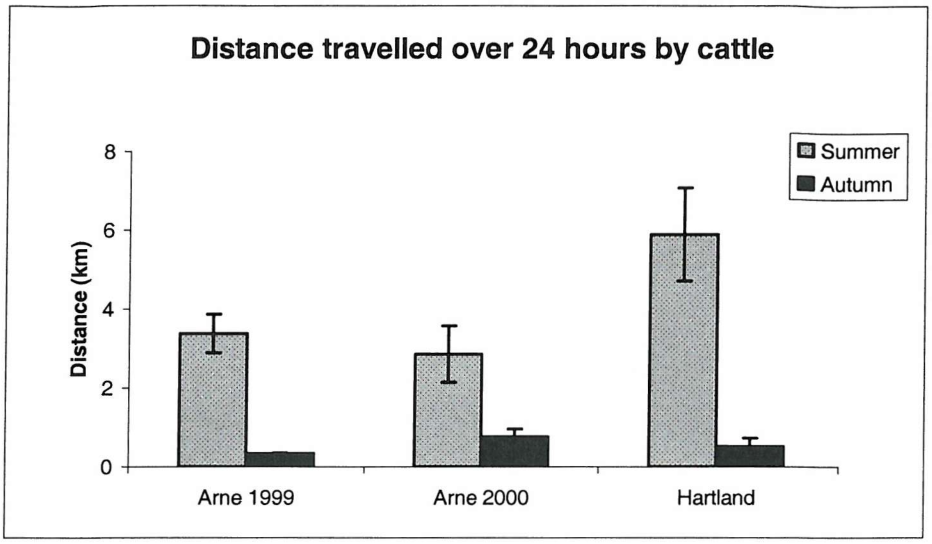


Figure 2.26 Seasonal difference in distance travelled by cattle herds on three Purbeck heathlands. Summer includes June, July August, and autumn includes September, October and November.

Dunging

Total dunging was greatest in the habitats where the most time was spent. However, dunging was not in proportion to the time spent in the habitat (chi square = 32.04, df = 10, P = 0.01) and was disproportionately high in woods (W) and scrub (SC). Dunging was also higher than expected in saltings margins (SM), wet heath and valley mire (M) a

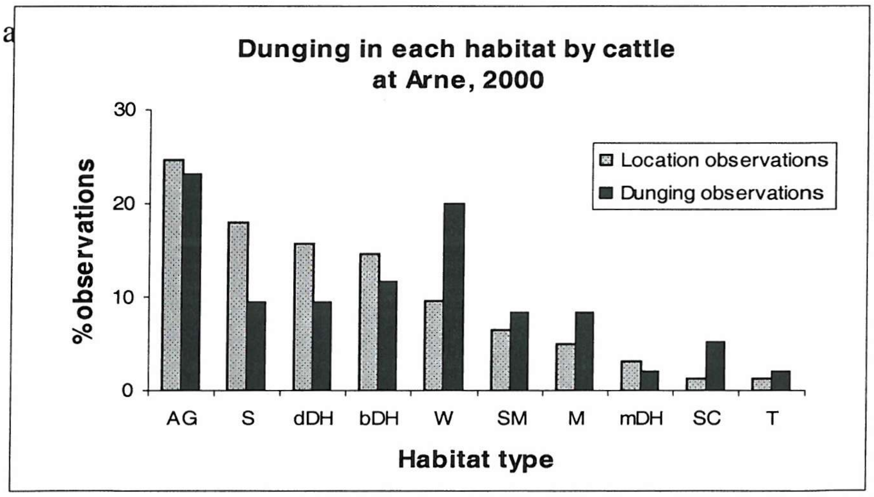


Figure 2.27 Total dunging observations compared to total location observations for cattle in each habitat at Arne in 2000. (Total dunging was not recorded at Arne in 1999). See Table 2.2 for habitat abbreviations.

At Hartland, dunging was in proportion to habitat use overall, although there were a disproportionately high number of dunging observations in valley mire (VM), restoration heath (AG/DH) and roads (R) (Figure 2.28)

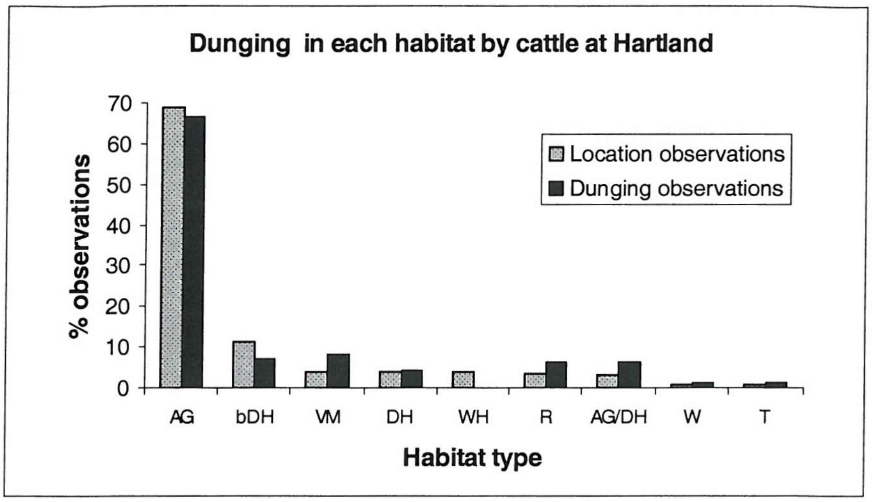


Figure 2.28 Total dunging observations compared to total location observations for cattle in each habitat at Hartland in 2000. See Table 2.2 for habitat abbreviations.

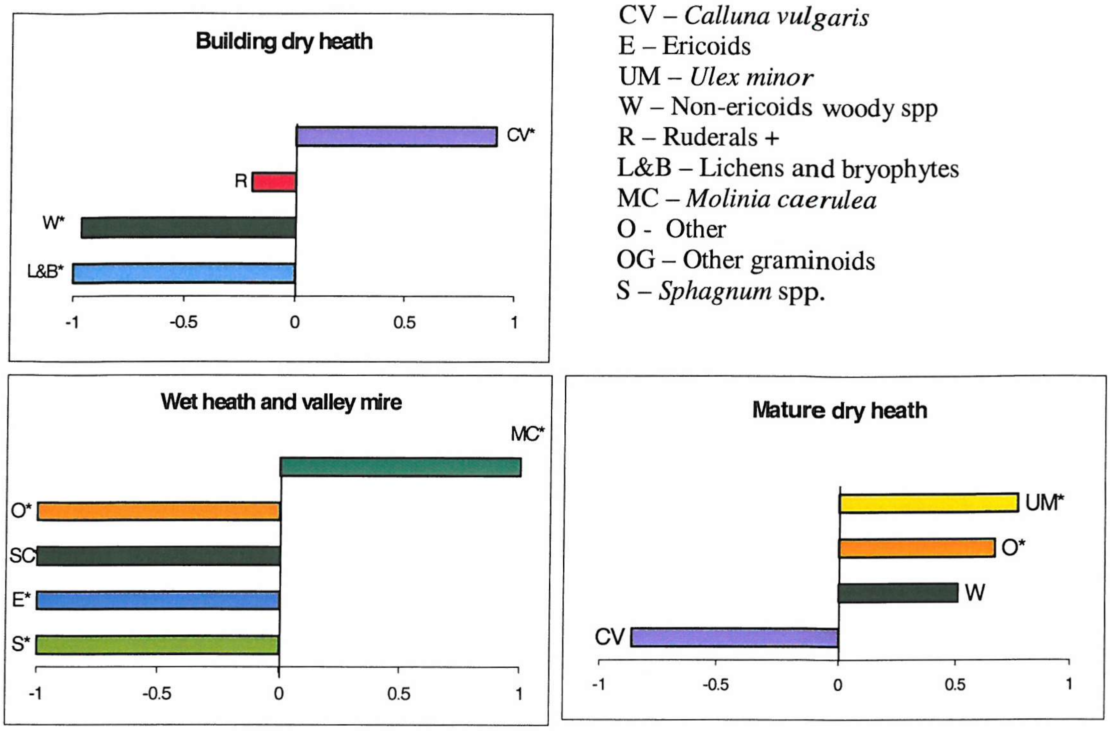
Foraging

Plant species were not selected in direct proportion to their availability (Table 2.13). On wet heath and valley mire, *Molinia caerulea* was positively selected by all cattle herds (e.g. Figure 2.29). In 1999, *Ulex minor* and *Pinus* sp were both positively selected on mature dry heath (mDH). On building dry heath (bDH), *Calluna vulgaris* was most selected for in 1999. However, in 2000, ruderal species (mostly *Rumex acetosella*) were selected in preference to *Calluna* on bDH, and *Ulex minor* and *Pinus* sp. were not eaten on mDH, where 'O' (of which only *Agrostis curtisii* was eaten) was most selected for (Figure 2.30).

At Hartland, grasses (*A. curtisii* and *Molinia caerulea)* were most selected for on both bDH and dry heath (DH). On restoration heath (AG/DH), acid grassland species were positively selected, although *Calluna* was slightly more selected for than on bDH and DH (see Appendix II for full data from Hartland). Figures 2.30 and 2.31 show selection for species in key habitats.

Table 2.13 Plant species availability (measured as % cover) and use by cattle (measured as the average % of total number of bites) at Arne in 1999. M - wet heath and valley mire, BDH – building dry heath, MDH – mature dry heath, SC - scrub.

Habitat	Species	% cover mean (SE)	% bites	Chi square
M	*Molinia caerulea* (MC)	75.7 (5.1)	100	34.02**
	Ericoids (E)	8.0 (2.5)	0	
	Other graminoids (OG)	9.1 (4.2)	0	
	Sphagnum spp. (S)	6.3 (2.8)	0	
	Pinus sp. & *Ulex europaeus* (SC)	0.9 (0.5)	0	
BDH	*Calluna vulgaris* (CV)	48.8 (6.2)	95.55	100**
	Lichens & bryophytes (L&B)	39.9 (5.9)	0	
	Ruderals and other species colonising disturbed ground (e.g. *Rumex acetosella Senecio viscosus, Urtica dioica, Digitalis purpurea, Juncus effusus*) (R)	6.3 (4.5)	4.45	
	Other (e.g. *Ulex minor, Pinus* sp, *Erica cinerea, Rhododendron* sp, *Pteridium aquilinum)* (SC)	5.0 (1.4)	0	
MDH	*Calluna vulgaris* (CV)	92.3 (1.6)	57.54	800.6**
	Ulex minor (UM)	1.5 (0.5)	12.78	
	Pinus sp (W)	2.7 (1.2)	9.72	
	Other (Agrostis curtisii, bryophytes, lichens) (O)	3.6 (0.8)	19.66	
SC	*Ulex europaeus*	81.2 (5.46)		Insufficient observations
	Molinia caerulea	4.0 (2.6)		
	Acid grassland	11.9 (4.4)		
	Other *(Pteridium aquilinum, Rubus fruticosus, Calluna vulgaris)*	2.9 (1.4)		



CV – *Calluna vulgaris*
E – Ericoids
UM – *Ulex minor*
W – Non-ericoids woody spp
R – Ruderals +
L&B – Lichens and bryophytes
MC – *Molinia caerulea*
O - Other
OG – Other graminoids
S – *Sphagnum* spp.

Figure 2.29 Selection of plant species/categories by cattle at Arne in 1999 expressed as Jacobs' indices. Species for which selection is significantly different from zero are denoted by * (Wilcoxon signed rank test, $P < 0.05$). See Table 2.13 for full species categories.

BDH: Selection for CV is higher than the remaining categories (Mann-Whitney, $P < 0.02$)
MDH: Selection for UM higher than CV (Mann-Whitney, $P < 0.005$)
M: Selection for MC higher than all other categories (Mann-Whitney, $P < 0.005$

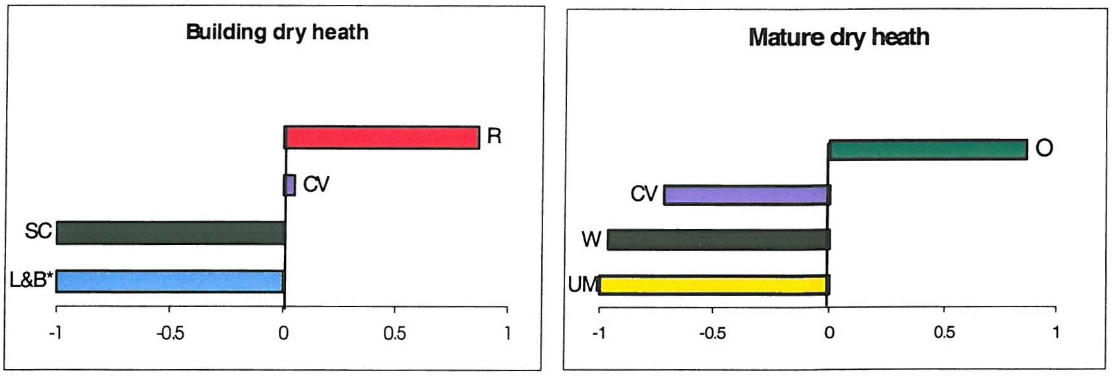


Figure 2.30. Selection for plant groups by cattle at Arne in 2000 expressed as Jacobs' indices, for habitats where selection differed from 1999. Species for which selection is significantly different from zero are denoted by * (Wilcoxon signed rank test, $P < 0.05$). Test not possible for mDH due to small sample size R – Ruderals, CV – *Calluna vulgaris*, SC – scrub, L&B – lichens and bryophytes, O – other, W – woody species, UM – *Ulex minor*. See Table 2.13 for species groups ,

BDH: Selection for L&B significantly smaller than for R and CV (Mann-Whitney, $P < 0.03$)

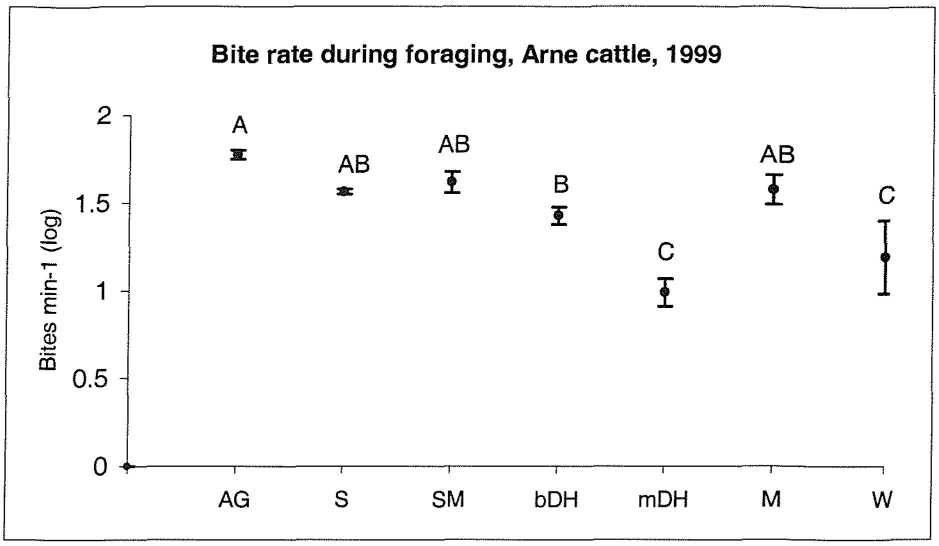


Figure 2.31 Bite rate in different habitats recorded for cattle at Arne in 1999. Different letters distinguish groups between which there were significant differences in foraging rate (GLM, F = 22.21, P < 0.001). There were no significant differences between individuals. See Table 2.2 for habitat abbreviations.

Bite rates were highest in grassier habitats, and lowest in habitats where woody species were eaten (Figure 2.31). No significant differences were found between individuals. There was no significant difference in the number of steps taken during foraging between habitats or individuals. Bite rates on grassy habitats were slightly higher in 2000, but the general pattern of differences between habitats remained similar. Rates at Hartland were again similar, with the additional habitat of restoration heath AG/DH showing a slightly lower rate than acid grassland AG (see Appendix II).

2.3.3. PONIES

Routine

Data from the Hartland New Forest pony follows are presented here, and reference is made to the Exmoor ponies and the Stoborough New Forest ponies where results differ (the full analysis for the Exmoor ponies and Stoborough New Forest ponies can be found in Appendices III and IV).

Figure 2.32 shows the behaviour of the New Forest ponies at Hartland over each 24-hour observation period. The pattern of behaviour was more uniform throughout 24 hours than that of the cattle (e.g. Figure 2.32, July and October), although the longest resting period (about two hours) was generally still during the night. (see also data for the Exmoor ponies in Figure 1, Appendix III). Figure 2.33 shows the routes taken by the Hartland New Forest ponies during each of the follows, and can be compared with Figure 2.2b for habitat types. The New Forest ponies were generally found on acid grassland (AG), wet heath (WH) or valley mire (VM), or moving between habitat patches on tracks (T) or dry heath (DH). Exmoor ponies were even more limited to AG. Both herds on Hartland were only observed venturing out into the main mire system in August and September. Neither herd was observed north of this mire. The New Forest ponies at Stoborough used a limited area of the site (despite its relatively small size) and were generally found on one or other of the two small patches of AG, or on WH between them. The Exmoor herd was never observed to split up, but the New Forest herds at both Hartland and Stoborough frequently divided into smaller groups. The composition of these groups did not remain constant, particularly on Hartland.

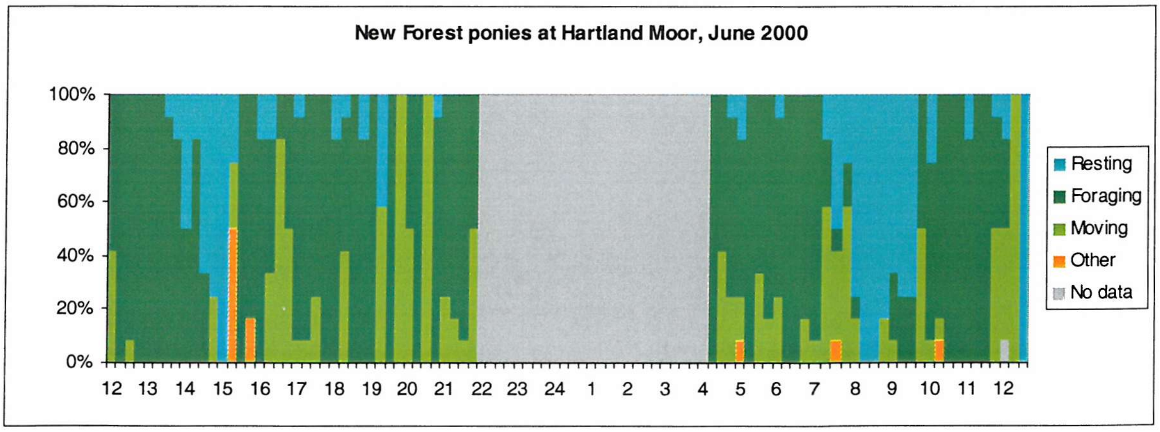


Figure 2.32. Behaviour of New Forest ponies at Hartland Moor during 24-hour observation periods. The proportion of the herd carrying out each behaviour is given on the vertical axis, and the time in hours along the horizontal axis. No data were collected in December, when the herd was in an adjacent field system fenced out of the heathland grazing unit.

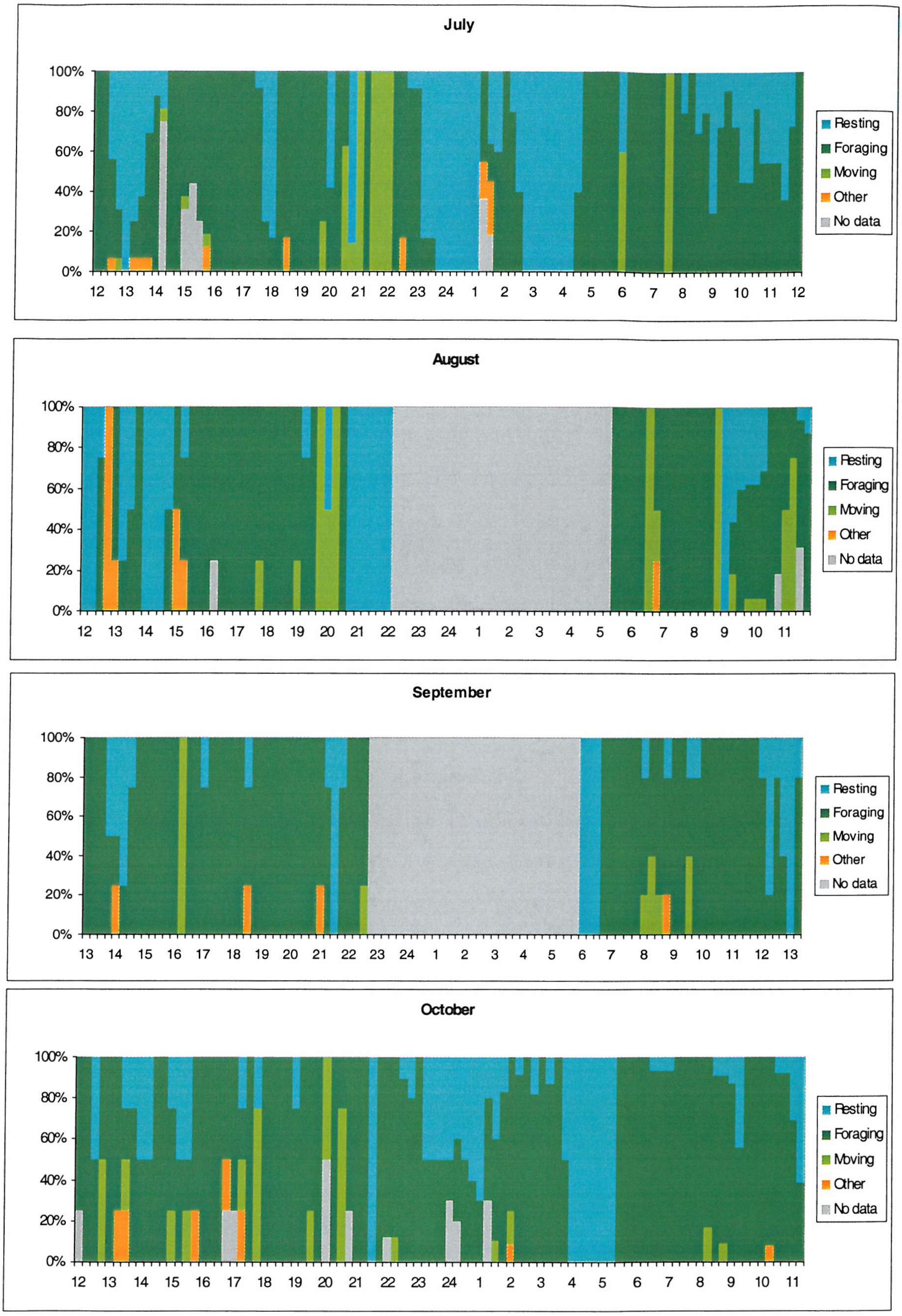
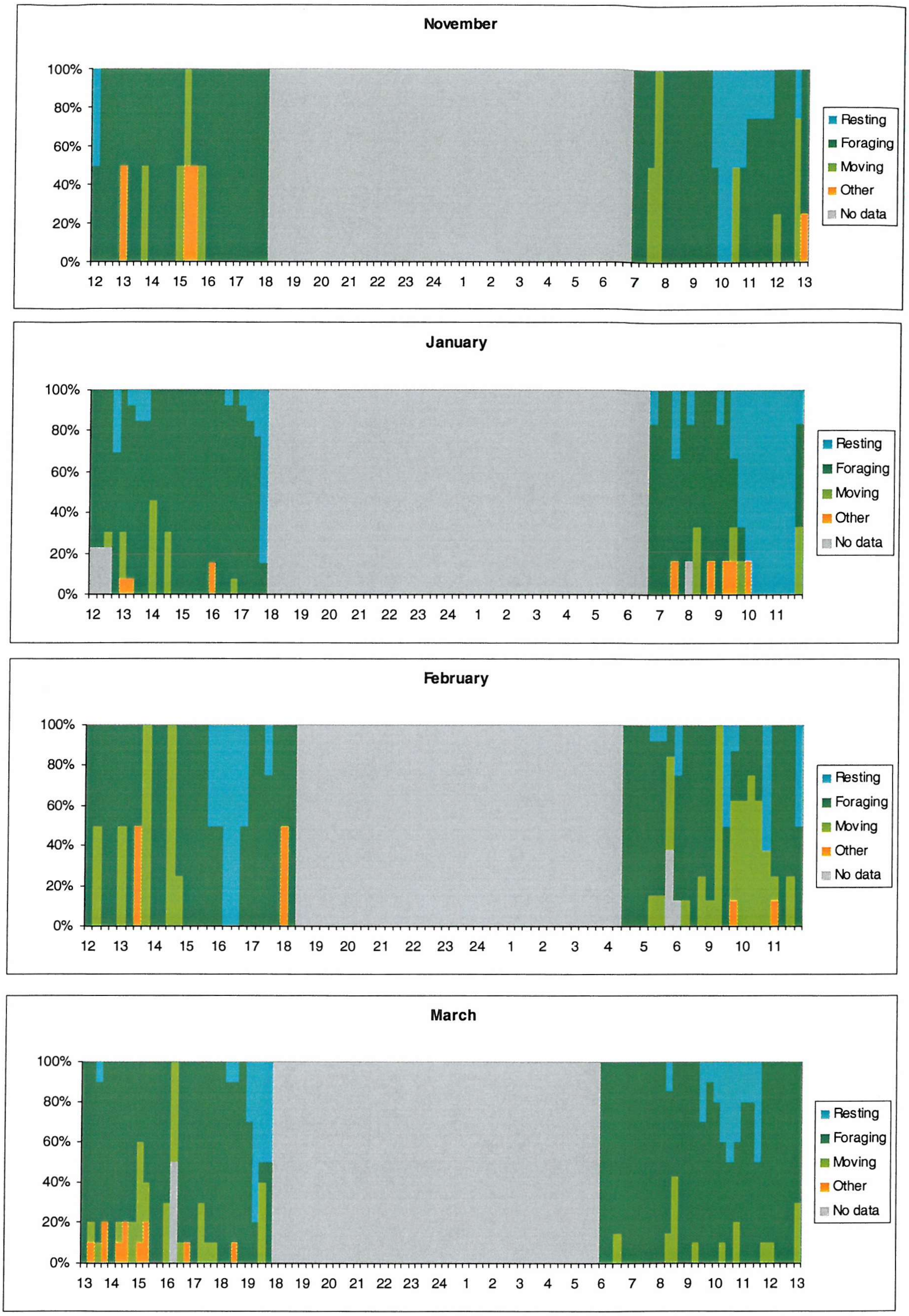


Figure 2.32 cont'd.



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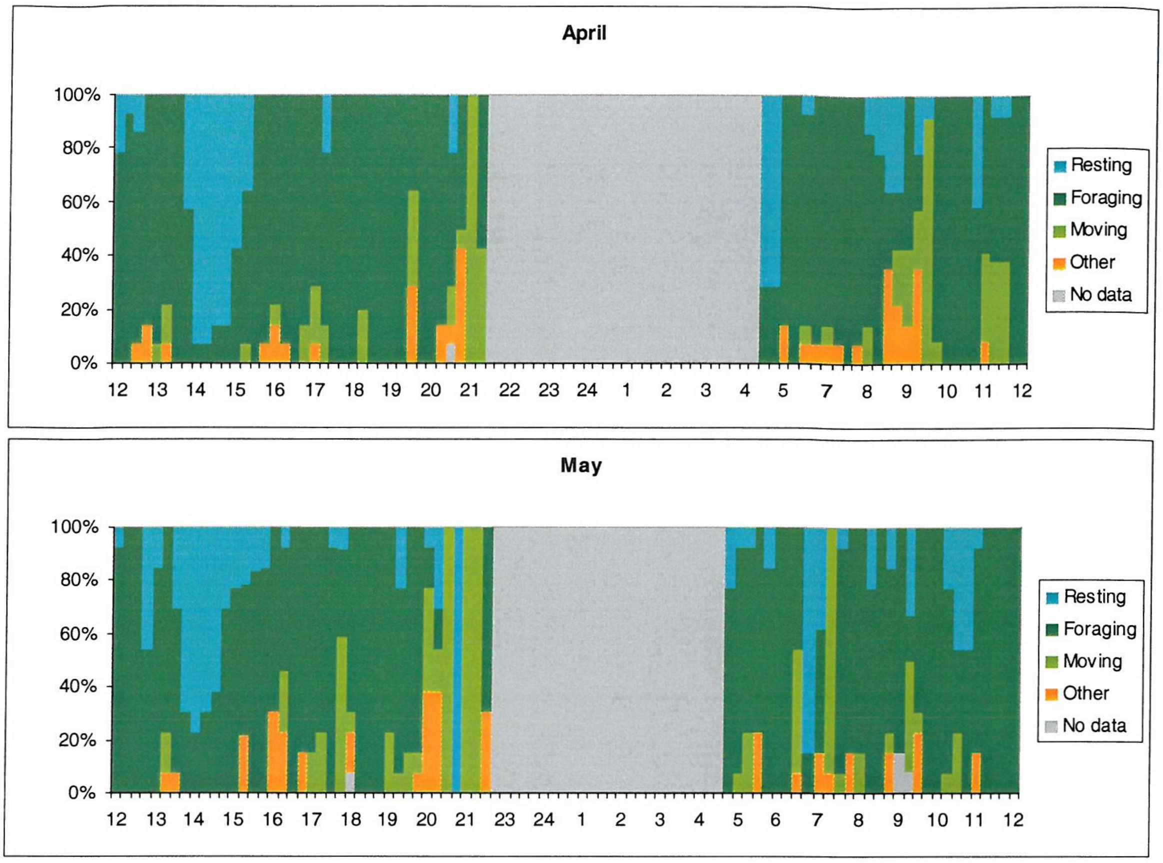

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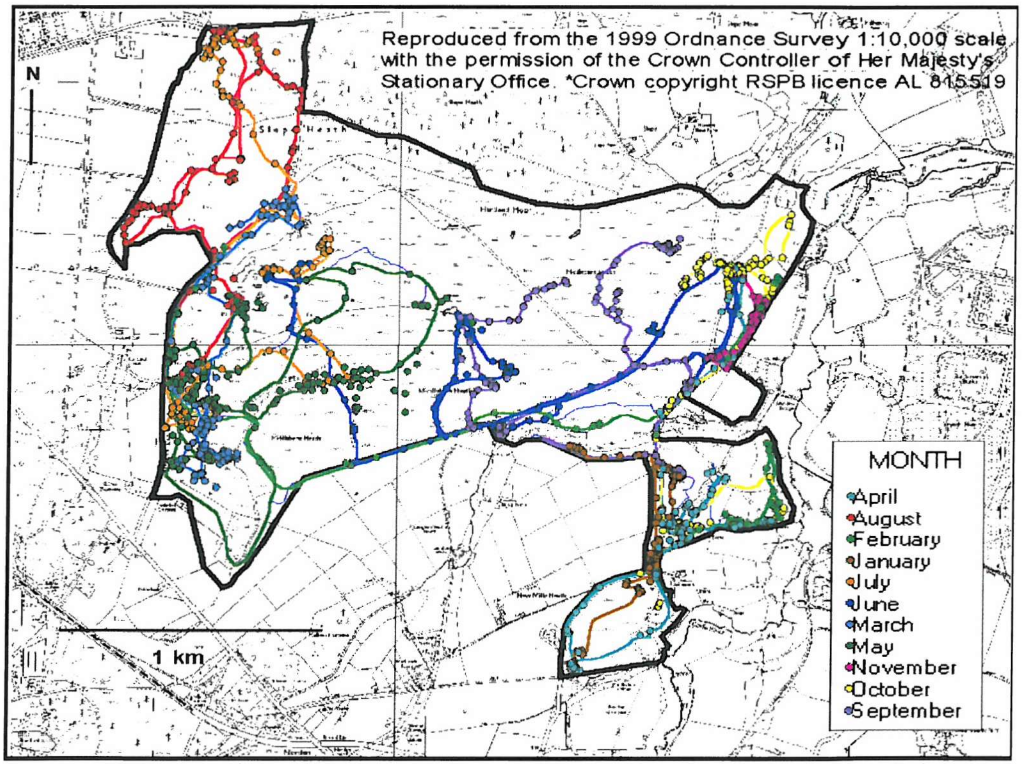


Figure 2.33 Routes taken by New Forest ponies at Hartland over each 24-hour period

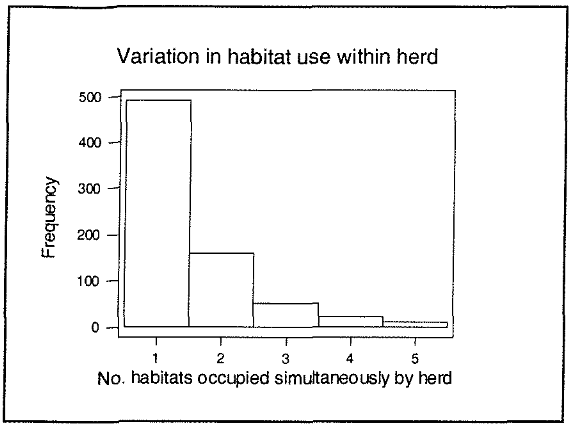


Figure 2.34 Variation in habitat selection by New Forest ponies at Hartland Moor, measured as the number of habitats occupied simultaneously by the herd. (N.B. changes in herd size occurred throughout each 24-hour follow, usually when the herd moved between habitats).

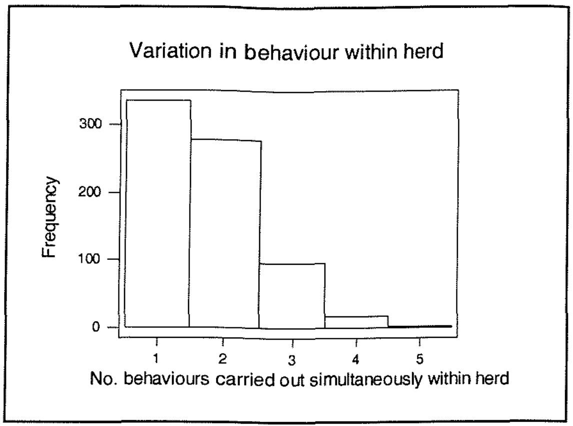


Figure 2.35 Variation in behaviour by New Forest ponies at Hartland Moor, measured as the total number of behaviours carried out simultaneously within the herd

Individual variation

Although pony herds generally occupied a single habitat type, herds were on occasion dispersed over up to five habitats simultaneously (Figure 2.34). Up to five behaviours types were recorded simultaneously, although one - two were most common (Figure 2.35)

Individual habitat use

There was one (out of 11) significant difference in habitat use by individuals at Hartland (i.e. the number of observations in each habitat type compared within simultaneously recorded pairs of ponies). This occurred during the May observation, when one pony grazed on wet heath (WH) for a period of 1.75 hours while its pair grazed on acid grassland (AG) (chi-square 210.2, $P < 0.01$). There were no significant differences in habitat selection between pairs of Exmoor ponies, or the New Forest ponies on Stoborough Heath.

Individual behaviour

There were no significant differences in the number of observations of behaviour types between individuals in each of the 11 pairs of Hartland New Forest ponies. There was one (out of eight) significant difference in behaviour between pairs of Exmoor ponies (chi-square = 34.86, $P < 0.01$). This occurred in March (when the herd remained on acid grassland AG throughout the 24-hour observation) and was a result of one pony browsing the *Ulex europaeus* hedges (included within the AG habitat classification) while its pair grazed on AG. Both were, however, foraging. In contrast, five out of the six pairs of New Forest ponies at Stoborough showed significant differences in behaviour (Table 2.14). Of these, three were differences between grouped behaviours (e.g. between resting and foraging), the remaining two were within-group (e.g. between grazing and eating hay).

Table 2.14 Behavioural differences in the Stoborough New Forest pony herd. * = P < 0.05, *** = P < 0.001

	Difference	**Chi-square**	**Grouped behaviour**	**Chi-square**
Pair 1	Standing/lying	27.2**	-	
Pair 2	Standing/grazing	17.96*	Resting/foraging	17.96***
Pair 4	Grazing/hay	86.79***	_	
Pair 5	Standing/hay	26.63*	Resting/foraging	9.64*
Pair 6	Standing/moving	22.58*	Resting/moving	10.87*

24-hour habitat selection

Habitat selection indices varied between breeds and sites, and are therefore presented for each herd. There was some variation in New Forest herd size on Hartland, and the full data set using observations from all ponies is used here (selection indices using pooled herd data can be found in Appendix V). New Forest ponies showed the greatest positive selection for tracks (T), followed by roads (R), restoration heath (AG/DH) and acid grassland (AG). However, only selection for AG was significantly positive throughout the grazing season, although dry heath (DH), scrub (SC) and valley mire (VM) were significantly negatively selected (Figure 2.36). Using the full data set to calculate selection indices is likely to result in a bias away from the habitats most used. However, this was not apparent (see Figure 1, Appendix V), possibly because selection for these habitats was so high. The total data set was used for subsequent analysis.

Due to the variation between monthly samples, the habitat ranking was not significant using either data set (Mann Whitney).

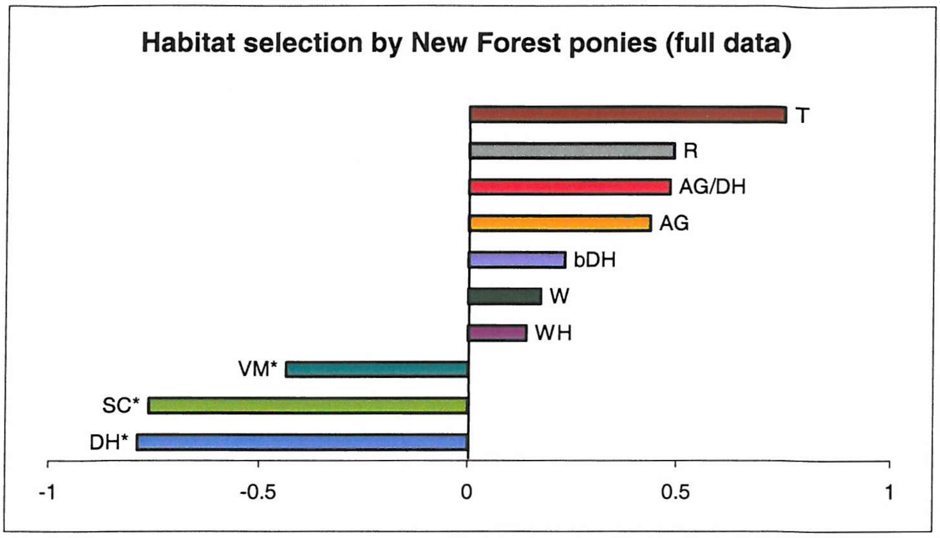


Figure 2.36 Habitat selection by New Forest ponies at Hartland expressed as Jacobs' indices (using the full data set). Habitat use was non-random (chi square = 1167.63, df = 9, p<0.001). * denotes habitats for which overall selection was significantly different from zero (i.e. not selected in proportion to availability) (Wilcoxon signed rank test, P<0.05) See Table 2.2 for abbreviations.

Tracks (T) was also the habitat Exmoor ponies selected most (Figure 2.37). However, in contrast to the New Forest ponies, roads (R) was the least selected, followed by scrub (SC) and dry heath (DH). There was variation between 24 hour observation periods, and, over the grazing season, the Jacob's indices for none of T, AG or SC were significantly different from zero, therefore overall habitat ranking were not significant (Mann-Whitney).

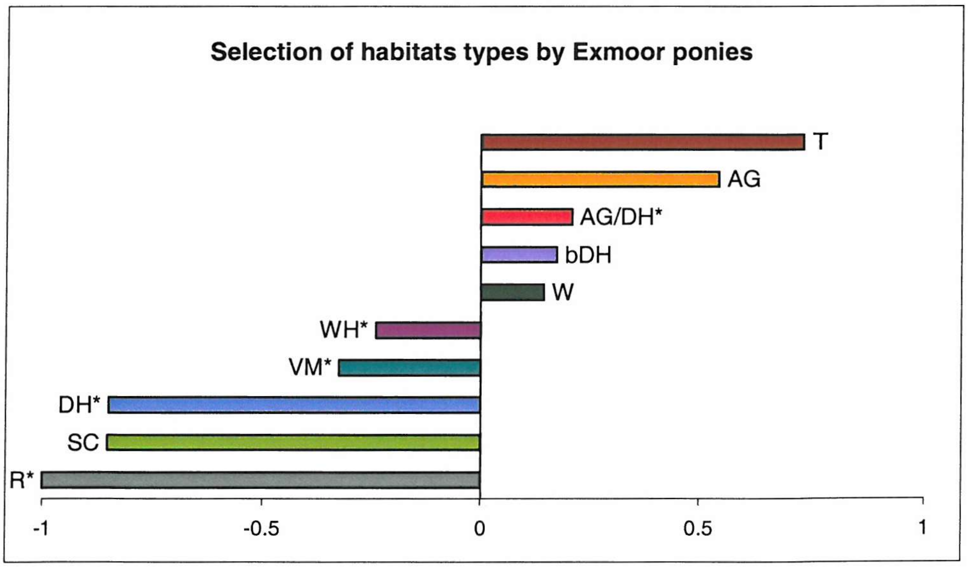
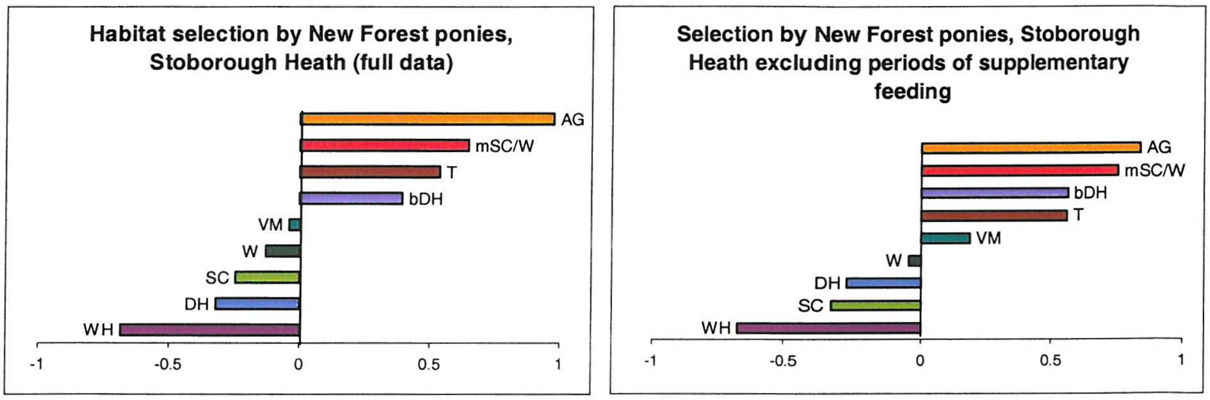


Figure 2.37 Habitat selection by Exmoor ponies at Hartland expressed as Jacobs' indices. Habitat use was non-random (chi square = 12818.71, df = 9, p<0.001). * denotes habitats for which overall selection was significantly different from zero (i.e. not selected in proportion to availability) (Wilcoxon signed rank test, P < 0.05). See Table 2.2 for abbreviations.

Habitat selection by New Forest ponies at Stoborough was basically similar, although selection for wet heath (WH) (the most abundant habitat type) was in this case strongly negative (Figure 2.38). Managed scrub and wood (mSC/W), which was not represented at Hartland Moor, was strongly selected for. Selection for acid grassland (AG) was significantly higher than for all other habitats except for mSC/W (Mann-Whitney, $P < 0.01$). Selection for tracks (T) was significantly higher than for dry heath (DH), scrub (SC) and WH (Mann-Whitney, $P < 0.05$).

Supplementary feeding was carried out in winter at Stoborough, and always took place at the same location on AG. The strong preference for supplementary feed shown by the ponies (see below) meant that AG was used almost exclusively when supplementary feeding was carried out. Habitat selection excluding the period when feeding occurred (January - March) is shown in Figure 2.35b. The pattern of selection was, however, largely the same, although valley mire (VM) was positively selected for when supplementary feeding was excluded.


Habitat selection by New Forest ponies, Stoborough Heath (full data)

Selection by New Forest ponies, Stoborough Heath excluding periods of supplementary feeding

Figure 2.38 Jacobs' indices quantifying habitat selection by New Forest ponies at Stoborough Heath (a) using the full data set (chi square = 58149.7, df = 8, $p<0.001$) (b) excluding periods of supplementary feeding* denotes habitats for which overall selection was significantly different from zero (Wilcoxon signed rank test, $P < 0.01$). Tests were not possible for (b) as there were fewer than six data points.

Seasonal habitat use

Habitat use by ponies varied seasonally (Figure 2.39). At Hartland, New Forest pony use of wet heath (WH) was significantly higher in summer (May-September) than winter (October, November, Jan - April) (ANOVA, $F = 24.22$, $P = 0.001$), with use peaking in July. Use of valley mire (VM) showed a similar trend (Figure 2.39a). The decrease in use of these habitats was associated with an increase in the use of restoration heath (AG/DH). Use of tracks (T) decreased, and roads (R) increased, in winter, while use of building dry heath (bDH) and dry heath (DH) both decreased in

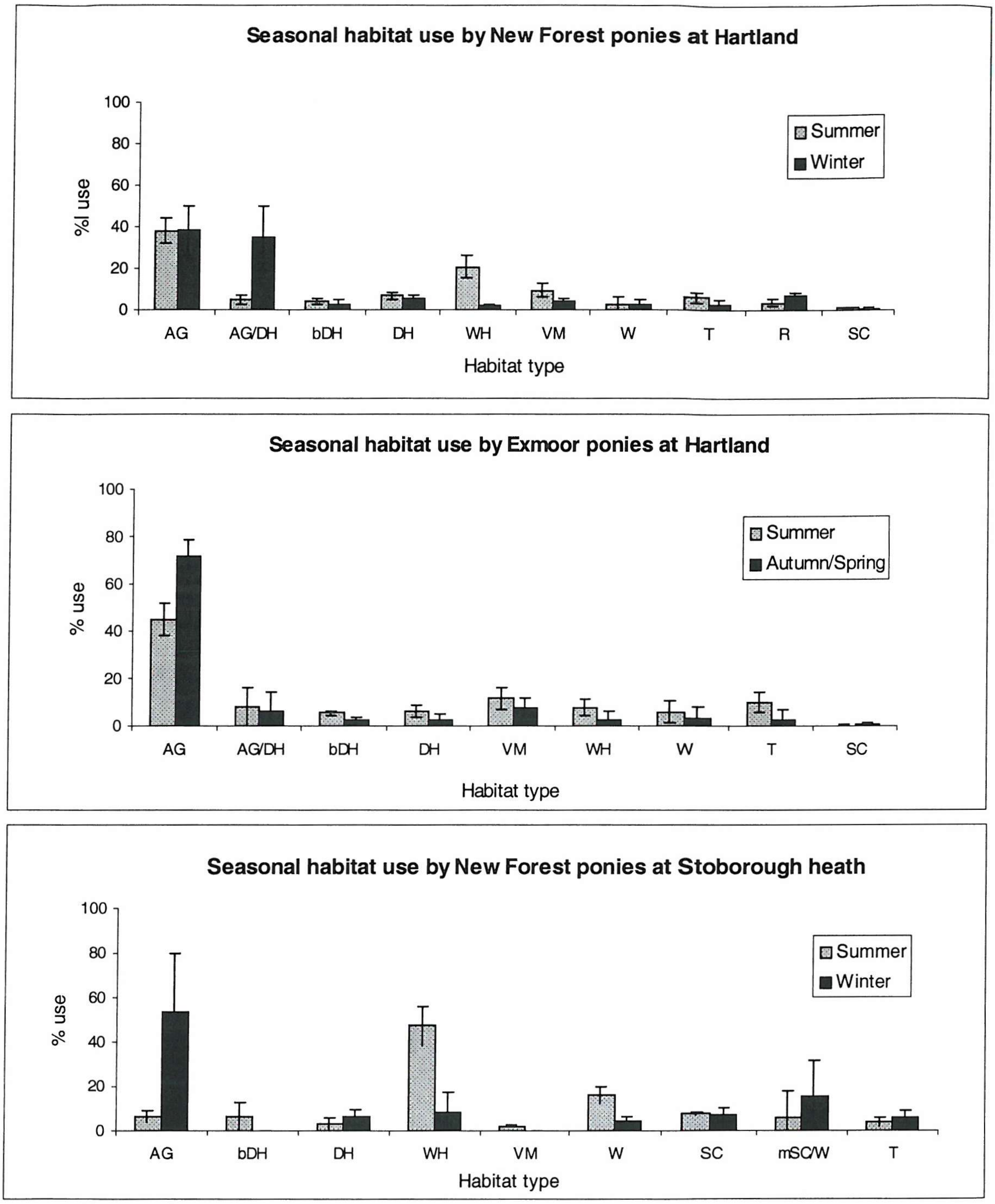


Figure 2.39 Seasonal habitat use by (a) New Forest ponies at Hartland Moor, (b). Exmoor ponies at Hartland Moor, (c) New Forest ponies at Stoborough Heath. * denotes individual significant difference at $P < 0.001$. Summer: May-September inclusive; Winter: October – April inclusive; Autumn/spring: October, November, March, April.

winter. Exmoor ponies showed significantly lower use of acid grassland (AG) in summer (May-August) than in autumn/spring (October, November, March, April) (one-way ANOVA, $F = 6.27$, $P = 0.046$) (Figure 2.39b). This decrease in use was associated with an increase of use of VM, WH and T. Similarly, at Stoborough Heath, there was significant seasonal difference in the use of AG (one-way ANOVA, 94.62 $P = 0.001$), and WH (one-way ANOVA, $F = 32.31$, $P = 0.005$) (Figure 2.39c). When the supplementary feeding period was excluded, the reduction in use of WH remained significant (one-way ANOVA, 8.67, P = 0.42) but differences in other habitats were not. The importance of supplementary feeding in influencing habitat selection can be seen in Figure 2.40, which compares habitat use under the two feeding regimes. The differences in use of AG and WH were again significant (one-way ANOVA, AG: F = 94.62, P< 0.001, WH: F = 32.31, P = 0.005) when feeding regime, rather than season, was taken into account. Other trends were the increase in use of scrub (SC) during the supplementary feeding period and reduction in use of all remaining habitat types.

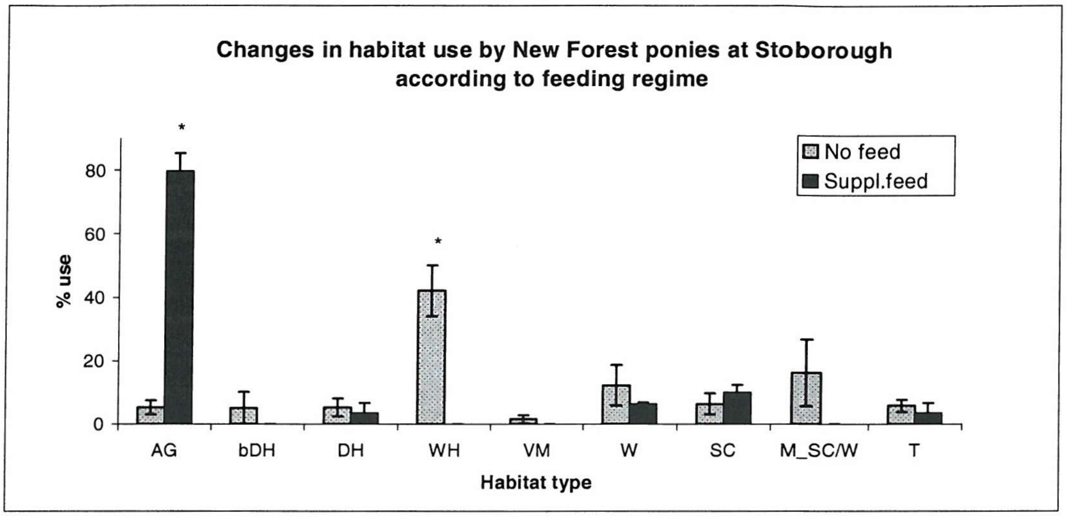


Figure 2.40 Habitat use by New Forest ponies at Stoborough Heath according to feeding regime. There was a significant interaction between season and habitat use GLM, F = 8.64, P < 0.001. * denotes significant differences. See Table 2.2 for abbreviations.

Behaviour

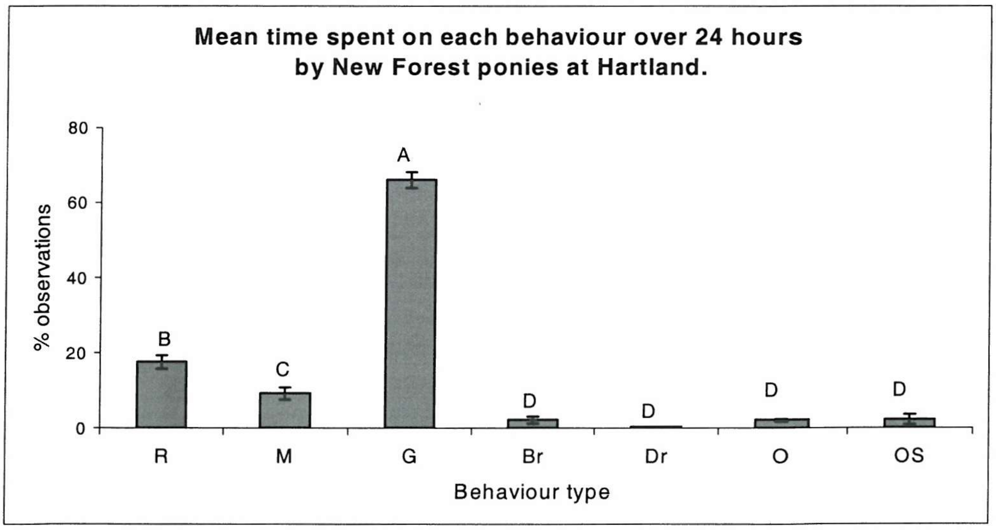


Figure 2.41 Mean time (percentage of observations) spent on each behaviour by New Forest ponies at Hartland. The difference in mean time spent on each behaviour was significant (GLM, F = 133.10, P < 0.001). Significant differences are denoted by different letters; therefore behaviours with the same letter show no significant difference. R – resting, M – moving, G – grazing, Br – browsing, Dr – drinking, O – other, OS –out of site

In contrast to the cattle, ponies on all sites spent most of their time foraging (e.g. Figure 2.41). (Note that this may be a slight overestimate of total time spent foraging, as ponies appeared to spend slightly more time resting after darkness - see Figure 2.32).

Behaviour in each habitat

Habitat-specific behaviour was very similar for all pony groups. Behaviours are shown for New Forest ponies at Hartland in Figure 2.42. Foraging was the most frequent behaviour in all habitats. The relatively high number of 'other' observations in woods (W) was due to ponies scratching on branches. Behaviour was slightly more habitat specific in Exmoor ponies, who spent a greater proportion of time on dry heath (DH) and scrub (SC) moving, and spent more time on tracks (T) and building dry heath (bDH) resting and less time in W resting than New Forest ponies (Figure 2.43). On Stoborough Heath, New Forest ponies spent more time on WH resting than ponies at Hartland. They spent more time on managed scrub/wood grazing than on unmanaged W or scrub (SC) (Figure 2.44).

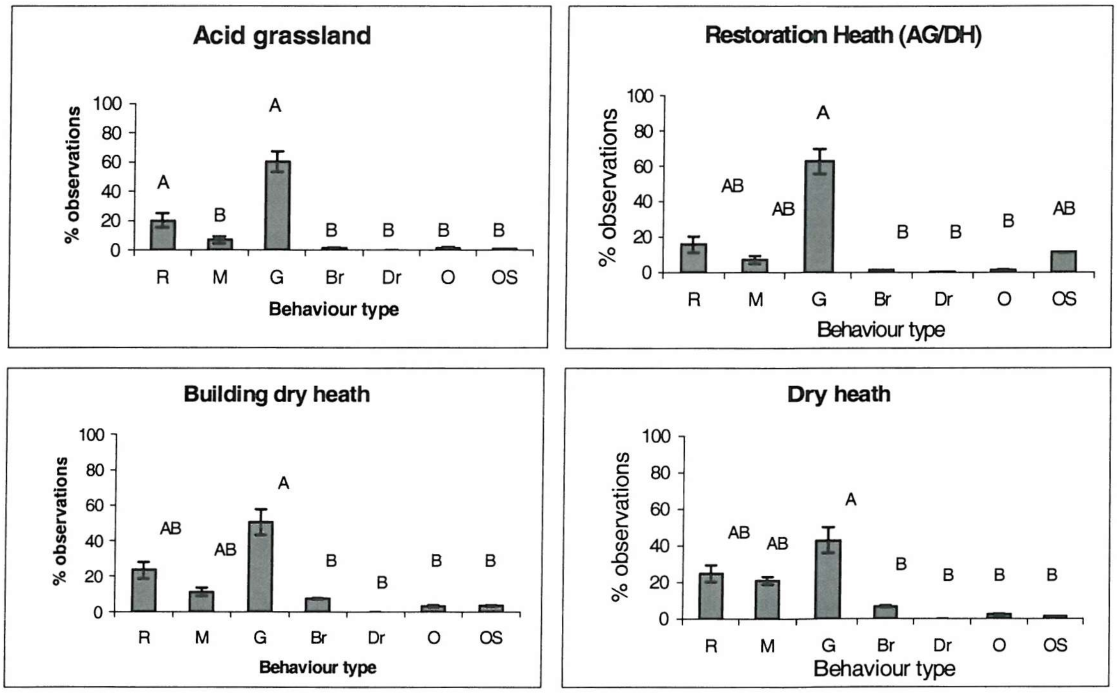


Figure 2.42 Mean time spent on each behaviour (% of observations) in different habitats by New Forest ponies, Hartland. Significant differences are denoted by different letters, therefore behaviours with the same letter show no significant difference. Where letters are omitted no significant difference between behaviour times was found. R – resting, M – moving, G – grazing, Br – browsing, Dr – drinking, O – other, OS –out of site. AG: F = 38.31***; AG/DH: F = 6.20***; bDH: F = 4.69***; DH: F = 9.07***; WH: F = 24.63***, VM: F = 24.68***; T:F = 7.15***; W: F = 10.11***, SC: F = 3.63**, R – differences not significant at $P = 0.05$. *** - $P < 0.001$, ** - $P < 0.01$

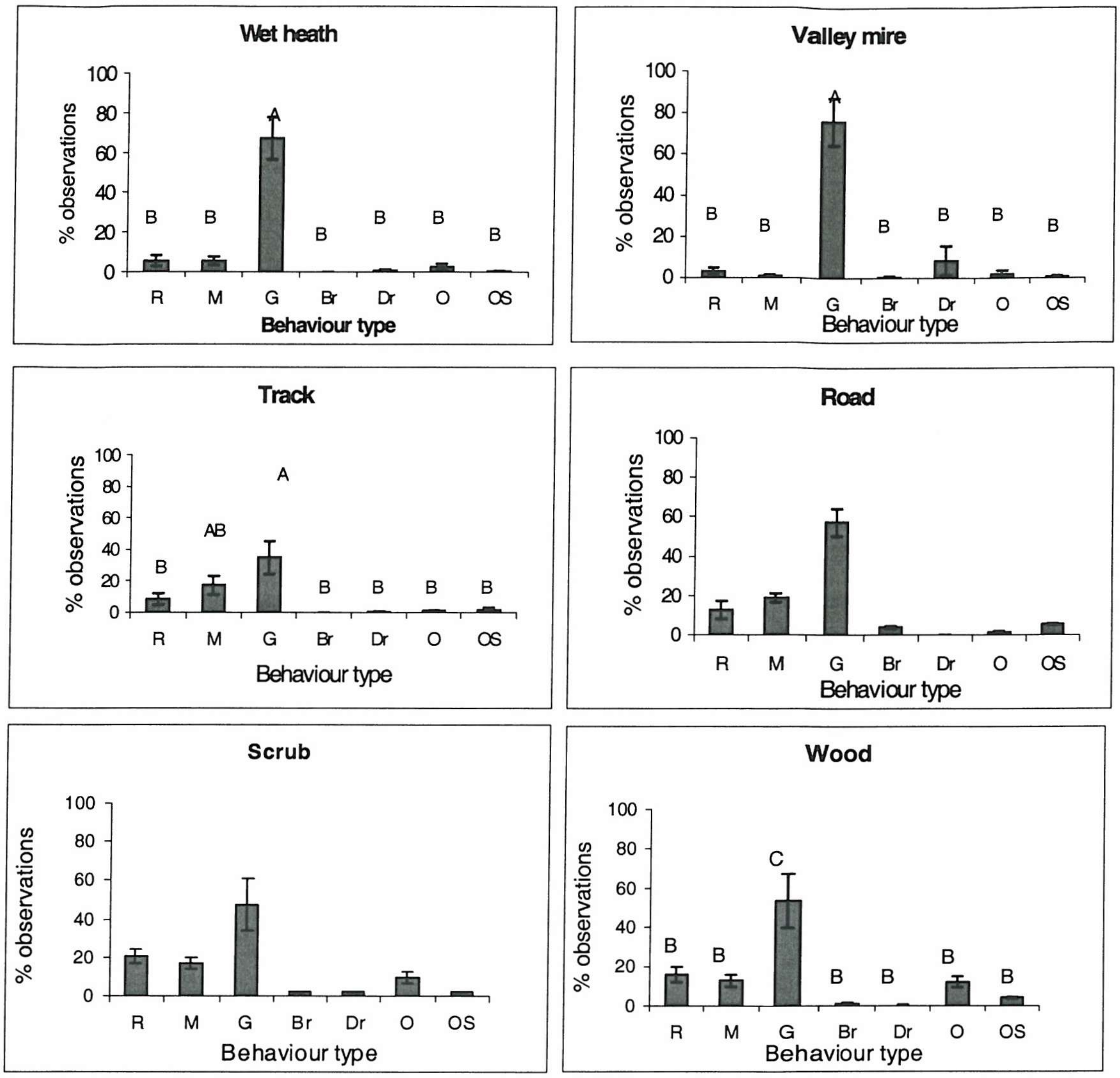


Figure 2.42 cont'd

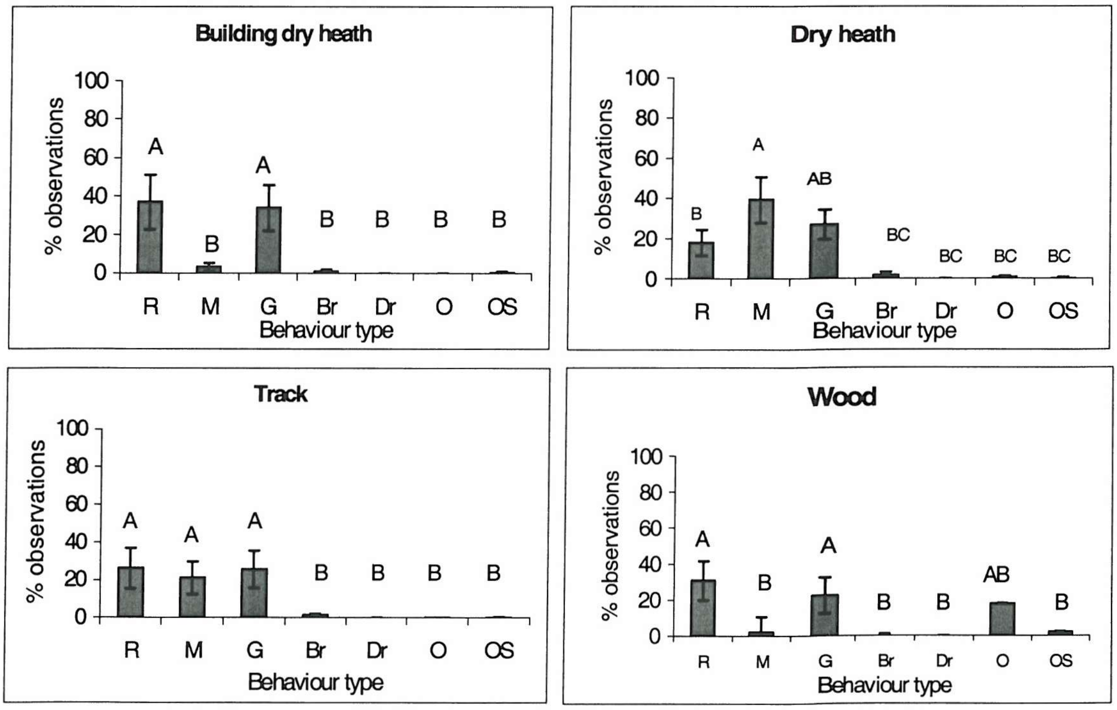


Figure 2.43. Mean time spent on each behaviour (% of observations) by Exmoor ponies at Hartland Moor for habitats where behaviour patterns differ from New Forest ponies at Hartland. Legend as above. bDH: F = 5.85***; DH: F = 8.98***; T:F = 5.28***; W: F = 5.05***. *** = $P < 0.001$.

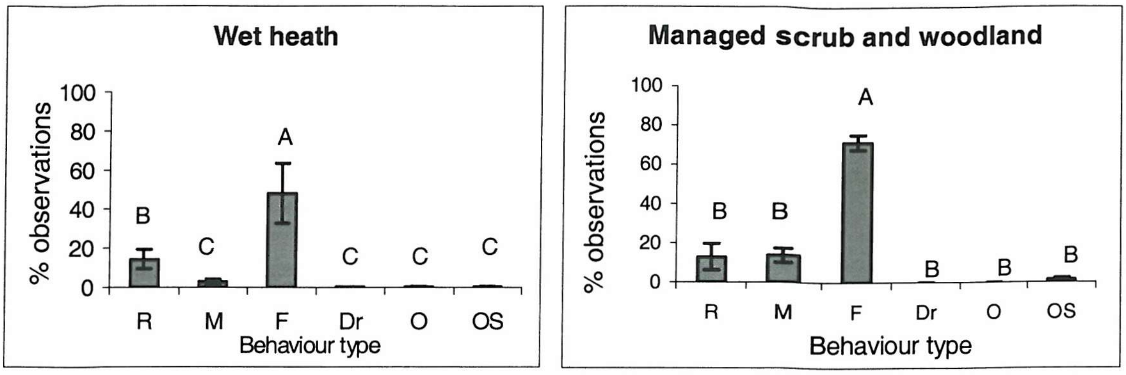


Figure 2.44 Mean time spent on each behaviour (% of observations) in different habitats on Stoborough Heath by New Forest ponies where these differ from New Forest ponies at Hartland. Significant differences are denoted by different letters, therefore behaviours with the same letter show no significant difference. . R – resting, M – moving, G – grazing, Br – browsing, Dr – drinking, O – other, OS –out of site. WH: F = 3.83*; mSC/W, F= 23.22***; *** - P < 0.001, * - P <0.05.

Selection of habitats for different behaviours

Figure 2.45 shows the Jacobs' indices quantifying the selection of each habitat by New Forest ponies for key behaviours. There is little difference in behaviour- specific selection and total selection. Selection for valley mire (VM) and wet heath (WH) was lower for resting and moving, selection for roads (R) and tracks (T) was higher for moving.

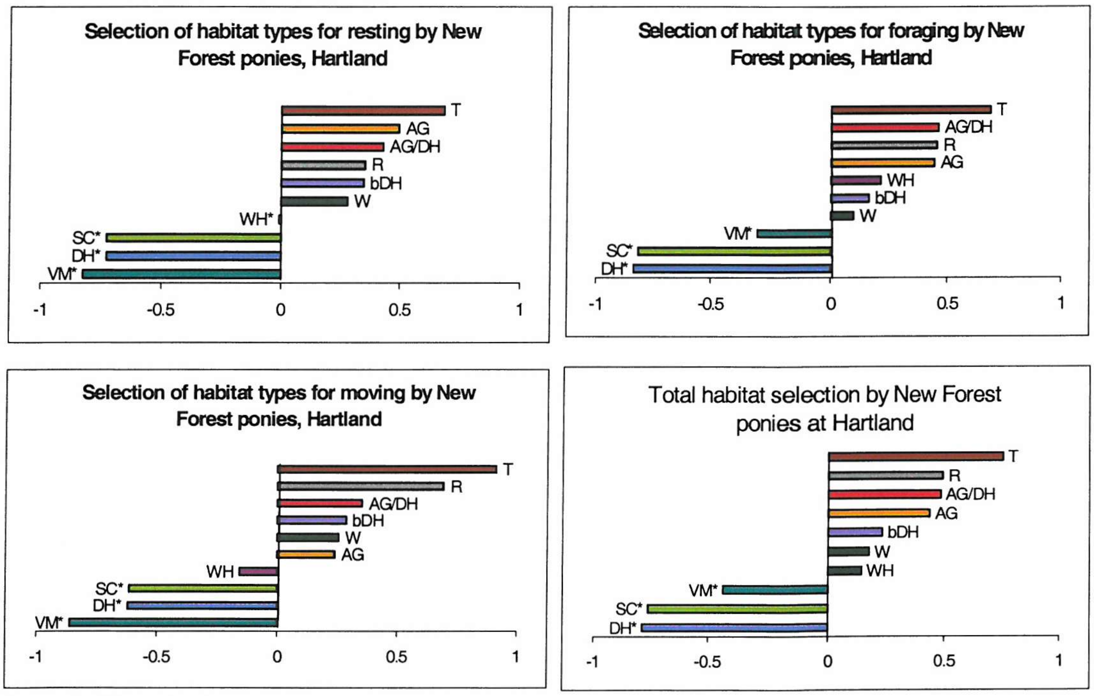


Figure 2.45 Selection of habitats for resting, foraging and moving, plus total selection, quantified by Jacobs' indices. Habitats for which the Jacobs' index differs significantly from a median of zero (which indicates selection is in direct proportion to availability) are denoted with * (Wilcoxon signed rank test, P<0.01). See Table 2.2 for abbreviations.

Exmoor ponies showed greater differential habitat selection for key behaviours (see Table 2.15, Figure 2.46). They showed greater selection for wood (W) for resting, building dry heath (bDH) for moving, and acid grassland (AG) and VM for foraging than New Forest ponies. Selection of habitats for particular behaviours by New Forest ponies at Stoborough largely reflected overall habitat selection (see Appendix IV).

Table 2.14 Differences in trends for habitat selection by Exmoor ponies for particular behaviours from total habitat selection. +/- indicates a change between negative and positive selection.

Behaviour	**Key differences in habitat selection**
Resting	W increases
Foraging	VM increases $^{+/-}$
	T decreases $^{+/-}$
Moving	bDH increases$^{+/-}$

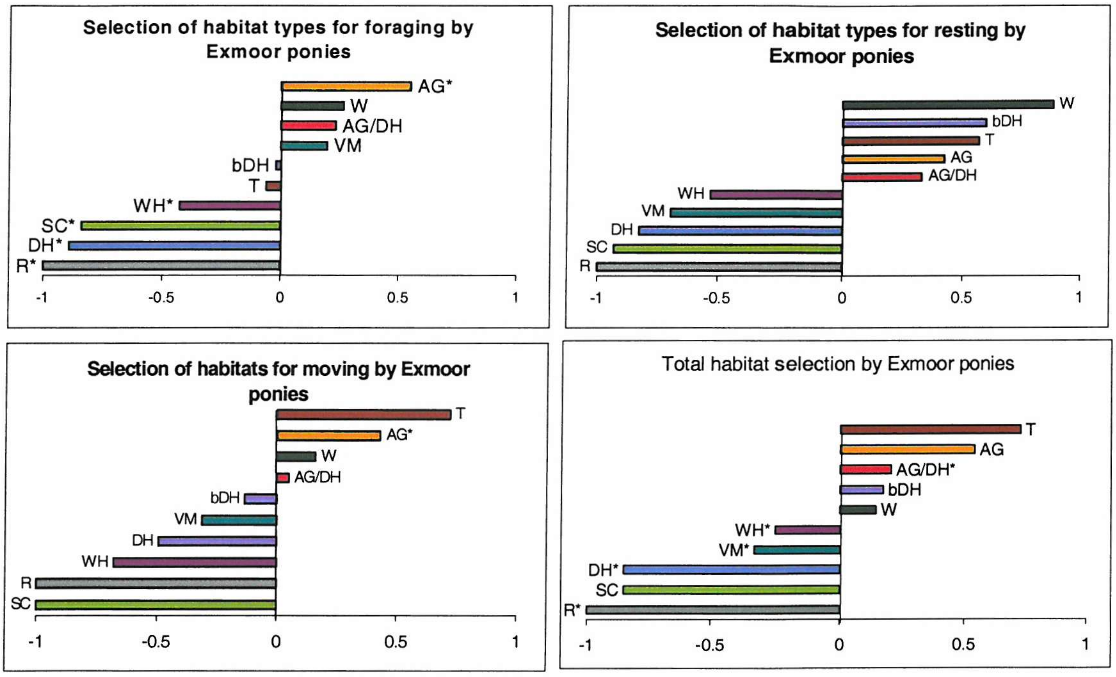


Figure 2.46 Selection of habitats for resting, moving and foraging, plus total selection, expressed as Jacobs' indices. Habitats for which the Jacobs' index differs significantly from a median of zero (which indicates selection is in direct proportion to availability) are denoted with * (Wilcoxon signed rank test, P<0.01). See Table 2.2 for abbreviations.

Seasonal changes in habitat use for particular behaviour

The proportion of time in each habitat that was spent on particular behaviours varied between summer and winter (autumn/spring in the case of the Exmoor ponies). Hartland New Forest ponies showed a significant switch from grazing to browsing on dry heath DH and building dry heath (bDH) in winter (GLM, DH: F = 2.27, P = 0.048; bDH: F = 2.97, P = 0.016) (Figure 2.47). Wet heath (WH) was not used for resting in winter, and consequently the proportion of time spent grazing increased slightly, although the actual number of observations decreased (GLM, F = 3.23, P = 0.009). Seasonal changes were not so clear for Exmoor ponies (possibly because of the smaller number of follows). Nonetheless, a significantly greater proportion of time in valley mire (VM) was spent foraging in summer than in winter (GLM, F = 2.69, P = 0.027).

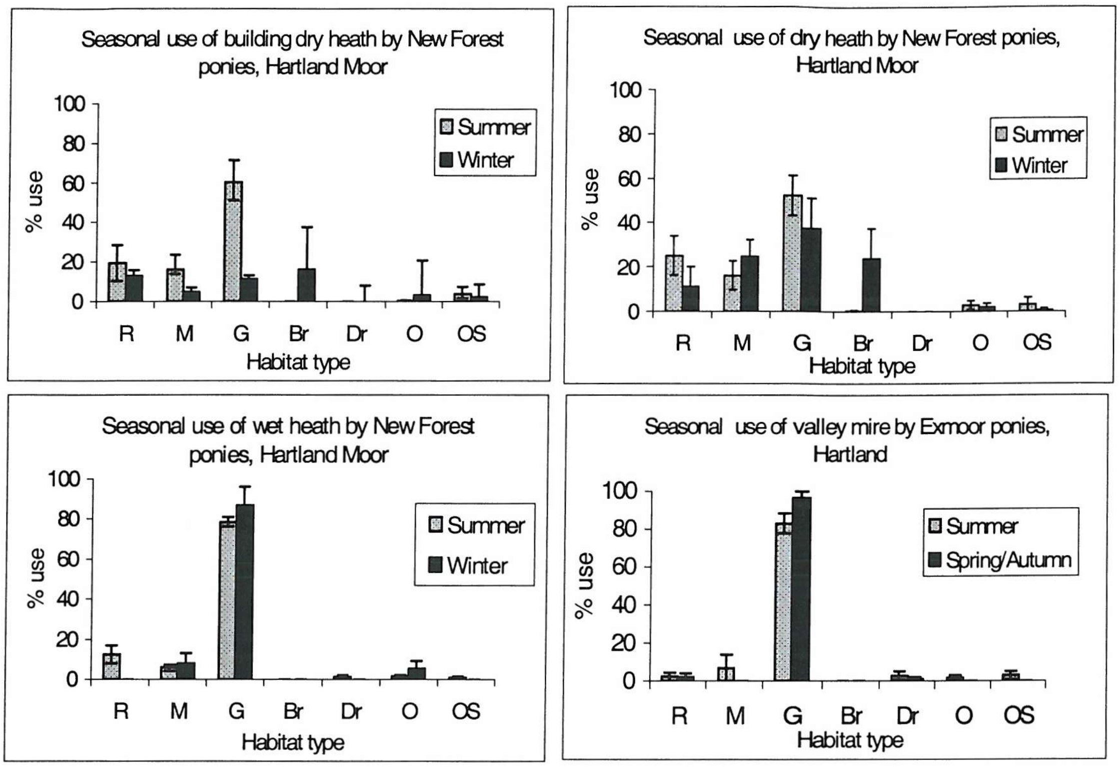


Figure 2.47 Seasonal changes in use of habitats by New Forest and Exmoor ponies on Hartland Moor. Summer includes May – September, winter includes October to April, Spring/autumn includes October, Novermber, March, April. R –resting, M – moving, G – grazing, Br – browsing, Dr – drinking, O – other, OS – out of sight.

On Stoborough Heath, differences in behaviour within each habitat between summer and winter were not significant. However, differences between supplementary feeding and non-feeding periods (July, September, November, May compared to January, March) were. The proportion of time within acid grassland (AG) spent foraging increased when supplementary feeding occurred (GLM, F = 17.01, P < 0.001) (Figure 2.48). There was also an increase in foraging in woodland during supplementary feeding (which was carried out directly adjacent to a patch of woodland).

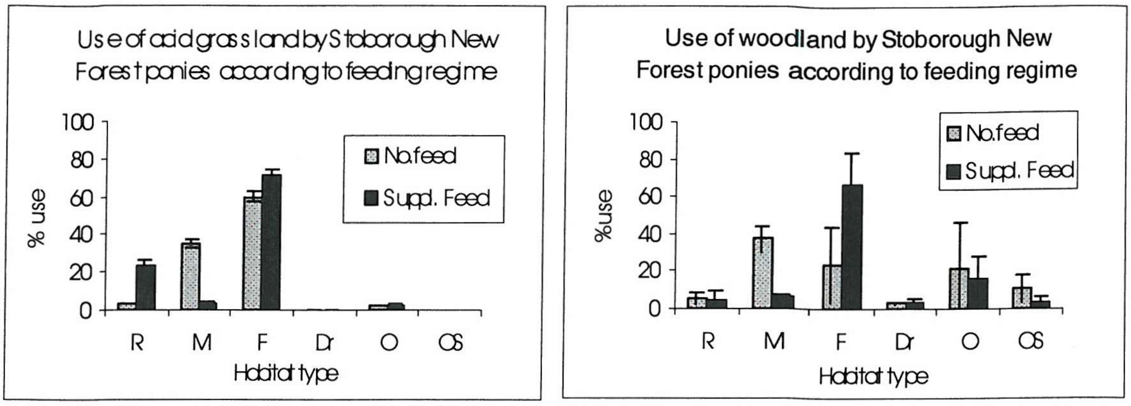


Figure 2.48 Changes in behaviour in acid grassland and woodland by New Forest ponies at Stoborough heath. R –resting, M – moving, G – grazing, Br – browsing, Dr – drinking, O – other, OS – out of sight.

Seasonal trends in selection of habitats for key behaviours

The analysis of seasonal change in the proportion of time spent on key behaviours within each habitat excluded times when the habitat was not used at all. Therefore, seasonal trends in the selection of habitats for key behaviours are given below.

The changes in the proportion of time spent on key behaviours within habitats led to changes in habitat selection by New Forest ponies at Hartland (Figure 2.49). Selection for roads (R) and restoration heath (AG/DH) increased in winter for foraging, while selection for wet heath (WH), valley mire (VM) and tracks (T) decreased. For resting, selection for WH and T decreased and AG/DH increased. Selection for R increased, and T and WH decreased, for moving.

Exmoor ponies also showed a lower selection for VM, WH and T, plus building dry heath (bDH), for foraging in winter. The corresponding increase was in wood (W) and in AG/DH. For both moving and resting, high selection for T was replaced with high selection for W, while selection decreased for bDH and increased for AG/DH (Figure 2.50)

At Stoborough, selection differences were more apparent between feeding regimes than between summer/winter. Supplementary feeding took place on AG, and when it was carried out this was the only habitat positively selected for foraging. Both T and managed scrub/wood (mSC/W) were also positively selected when feeding was not carried out. Similarly, AG became highly positively selected for resting and moving during feeding, but was highly negatively selected when feeding was not carried out (Figure 2.51).

New Forest Ponies, Hartland

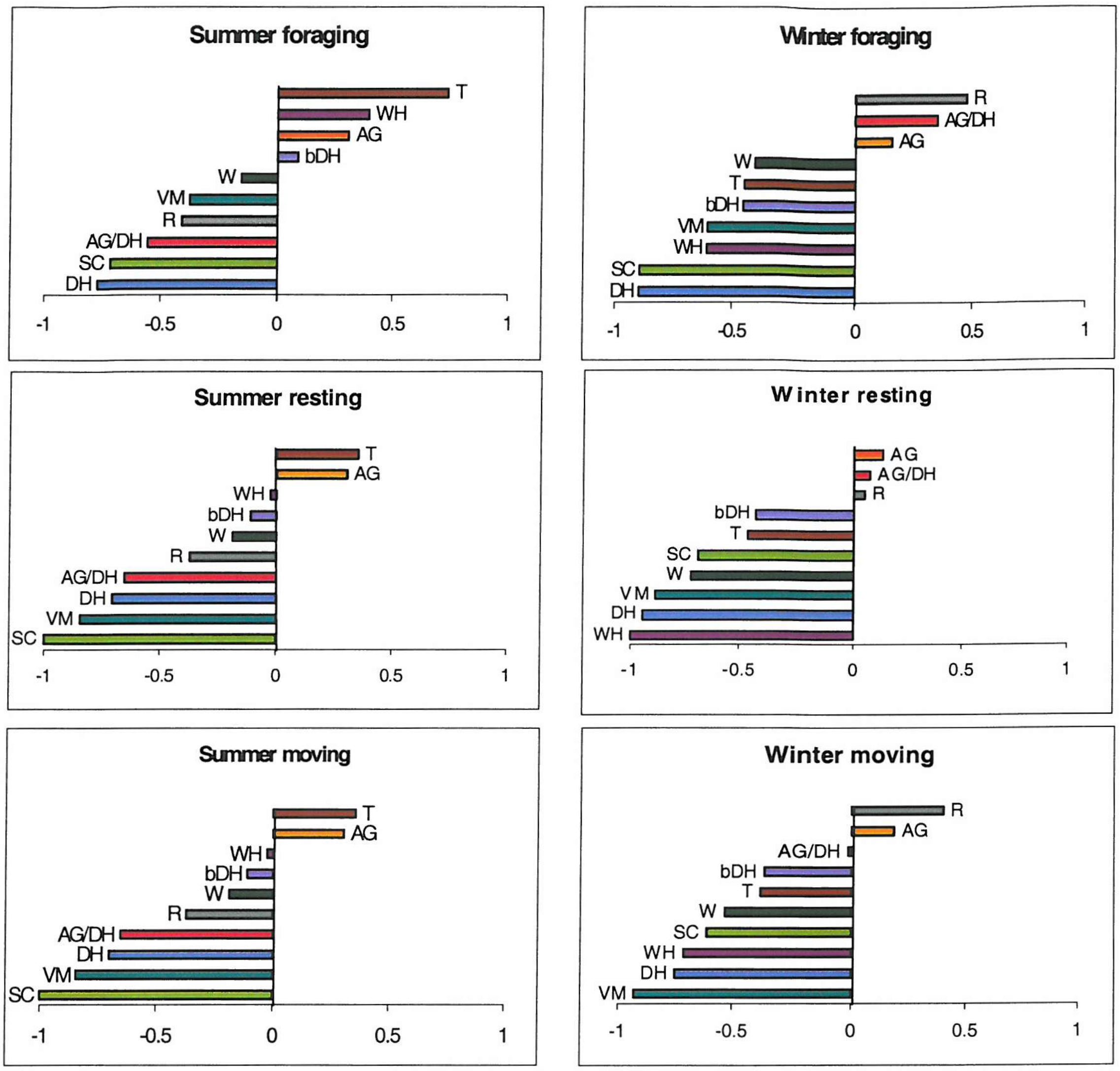


Figure 2.49 Seasonal selection of habitats for key behaviours by New Forest ponies at Hartland Moor, expressed as Jacobs' indices. See Table 2.2 for abbreviations. Summer includes May to September, winter includes October to April.

Exmoor ponies, Hartland Moor

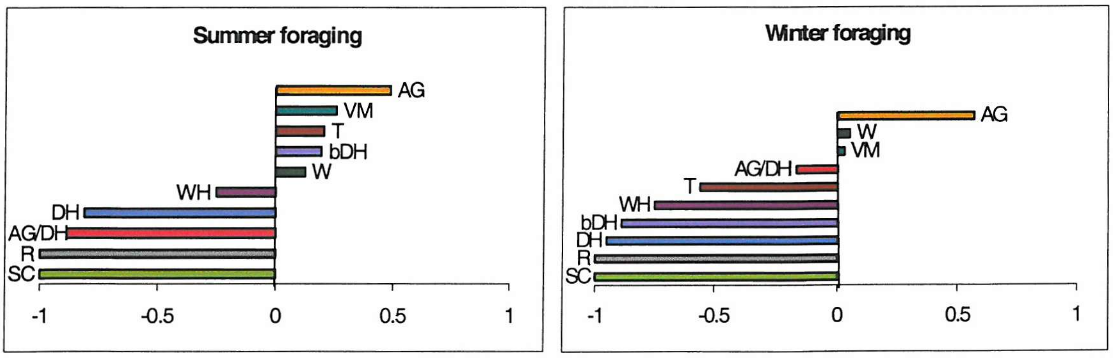


Figure 2.50 Seasonal selection of habitats for key behaviours by Exmoor ponies, Hartland Moor, expressed as Jacobs indices. See Table 2.2 for abbreviations. Summer includes May to September, winter includes October, March and April

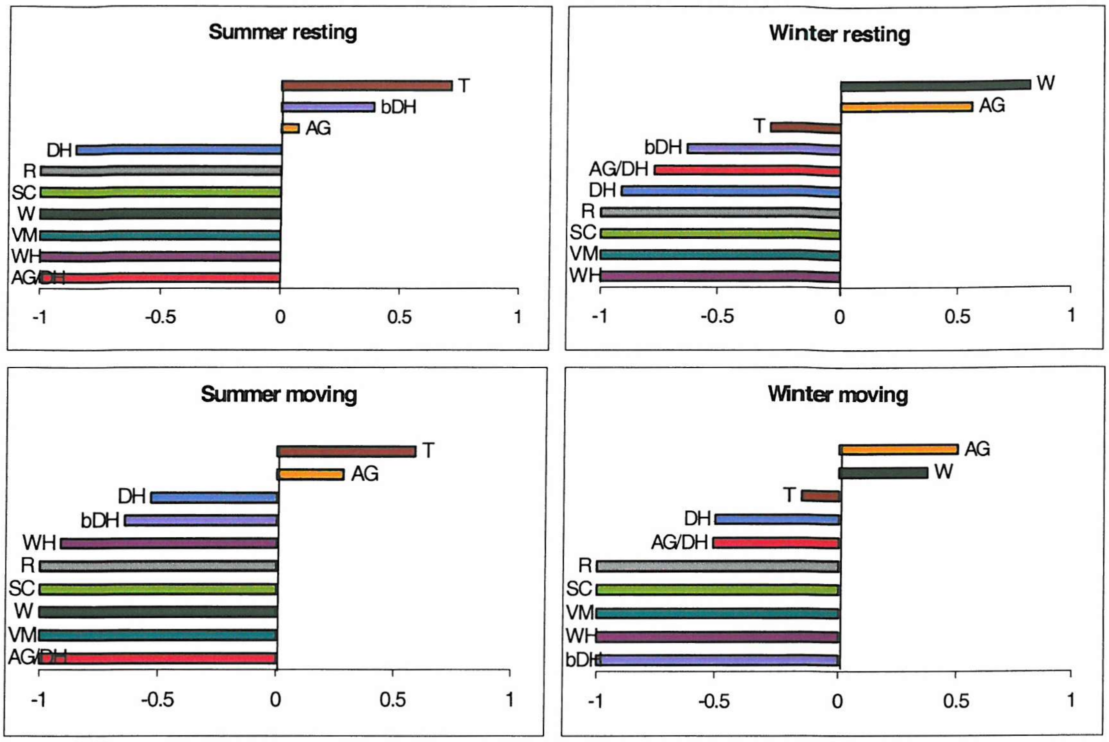


Figure 2.50 cont'd

New Forest ponies, Stoborough Heath

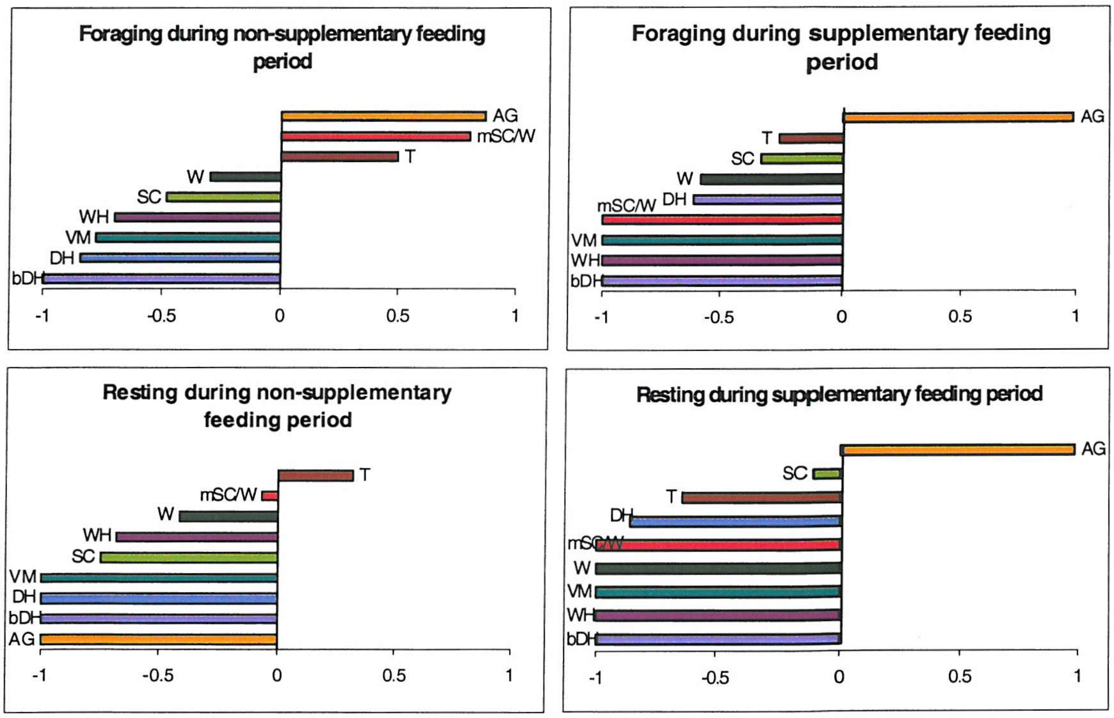


Figure 2.51 Selection of habitats for key behaviour by New Forest ponies, Stoborough Heath, according to feeding regime, expressed as Jacobs' indices. See Table 2.2 for abbreviations. Non-supplementary feeding periods includes July, September, November, April, non-supplementary feeding periods includes January and March.

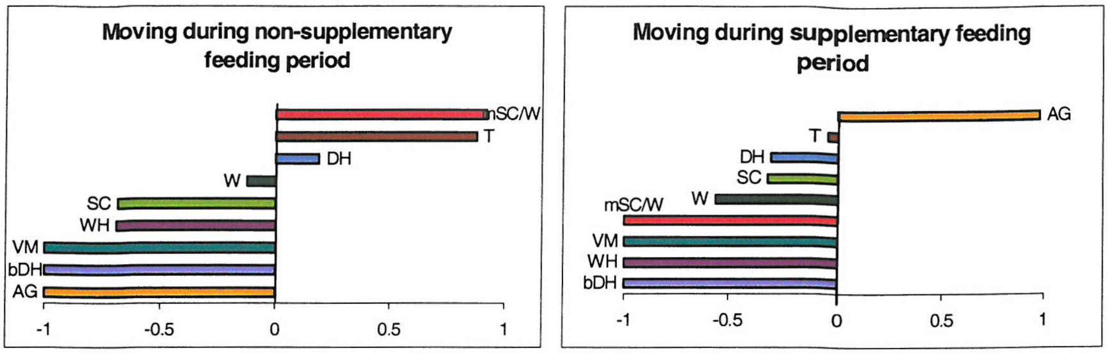


Figure 2.51 cont'd.

Distance travelled

The distance travelled over 24 hours by each pony group is given in Table 2.15. The distance travelled was significantly reduced in winter in all herds. Figure 2.53 shows the seasonal difference.

Table 2.15 Distances travelled in 24 hours by ponies at Hartland Moor and Stoborough heath.

Livestock group	**Range (km)**	**Mean (st.dev.) (km)**
New Forest, Hartland	1.56 - 8.32	5.67 (2.29)
Exmoor, Hartland	2.59 – 10.21	7.21 (1.80)
New Forest, Stoborough	2.87 – 4.10	2.41 (1.32)

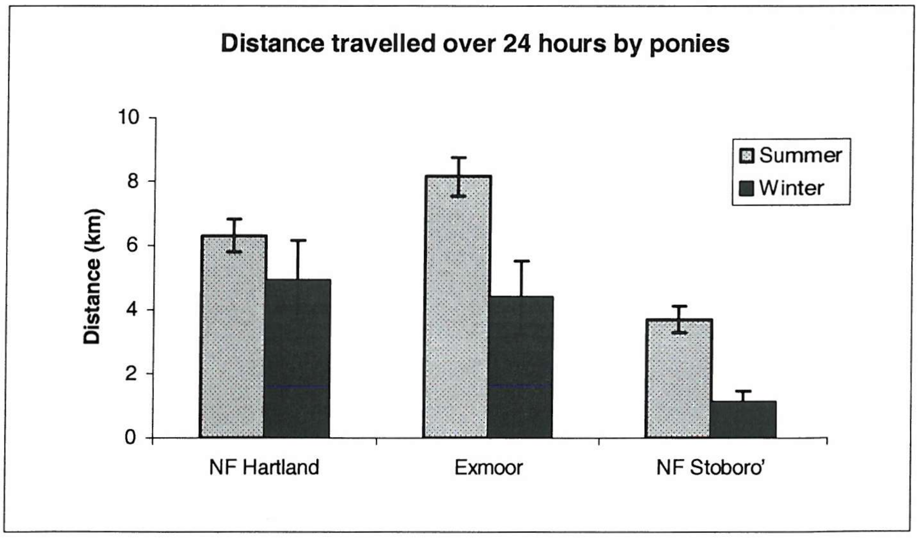


Figure 2.52 Seasonal difference in the distance travelled by ponies at Hartland Moor and Stoborough heath over 24 hours. For New Forest ponies, summer included May-September, winter included October-April. For Exmoor ponies winter included October, November, March , April. ANOVA, NF Hartland: F = 4.95, P = 0.050, Exmoor: F = 11.35, P = 0.008, NF Stoborough: F = 11.36, P = 0.028.

Dunging

Total dunging by New Forest ponies at Hartland was greatest in the habitats where the most time was spent. Dunging was slightly higher than expected in building dry heath (bDH), valley mire (VM), wood (W) and scrub (SC), but not significantly so (chi square). However, dunging by Exmoor ponies was disproportionately high in restoration heath AG/DH and dry heath DH (Chi-square = 17.51, df =8, P < 0.5) (Figure 2.49). At Stoborough, dunging was largely in proportion to the time spent in a habitat, although it was slightly higher in wet heath (WH), DH and acid grassland (AG). The disproportionately high number of dunging observations in AG was due to the two 24-hour observation periods when supplementary feeding was carried out.

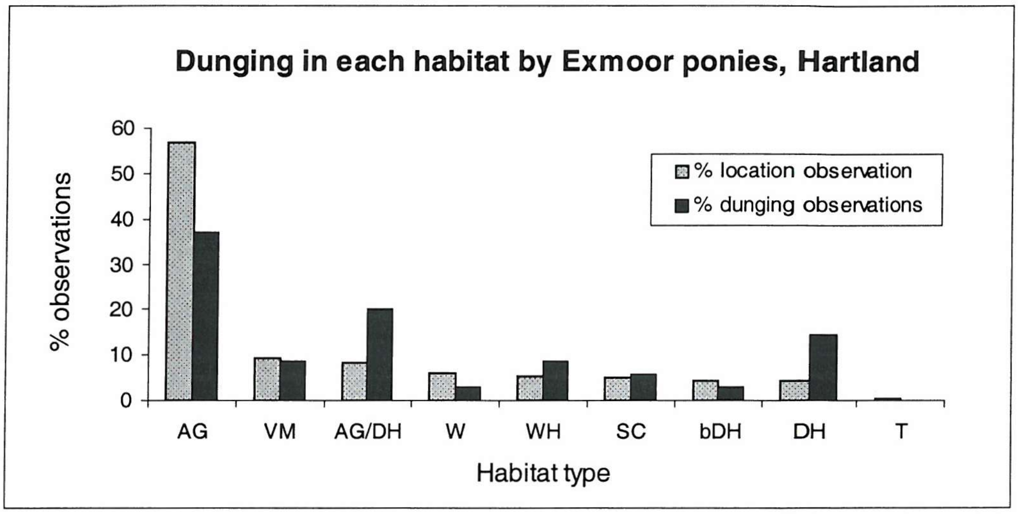


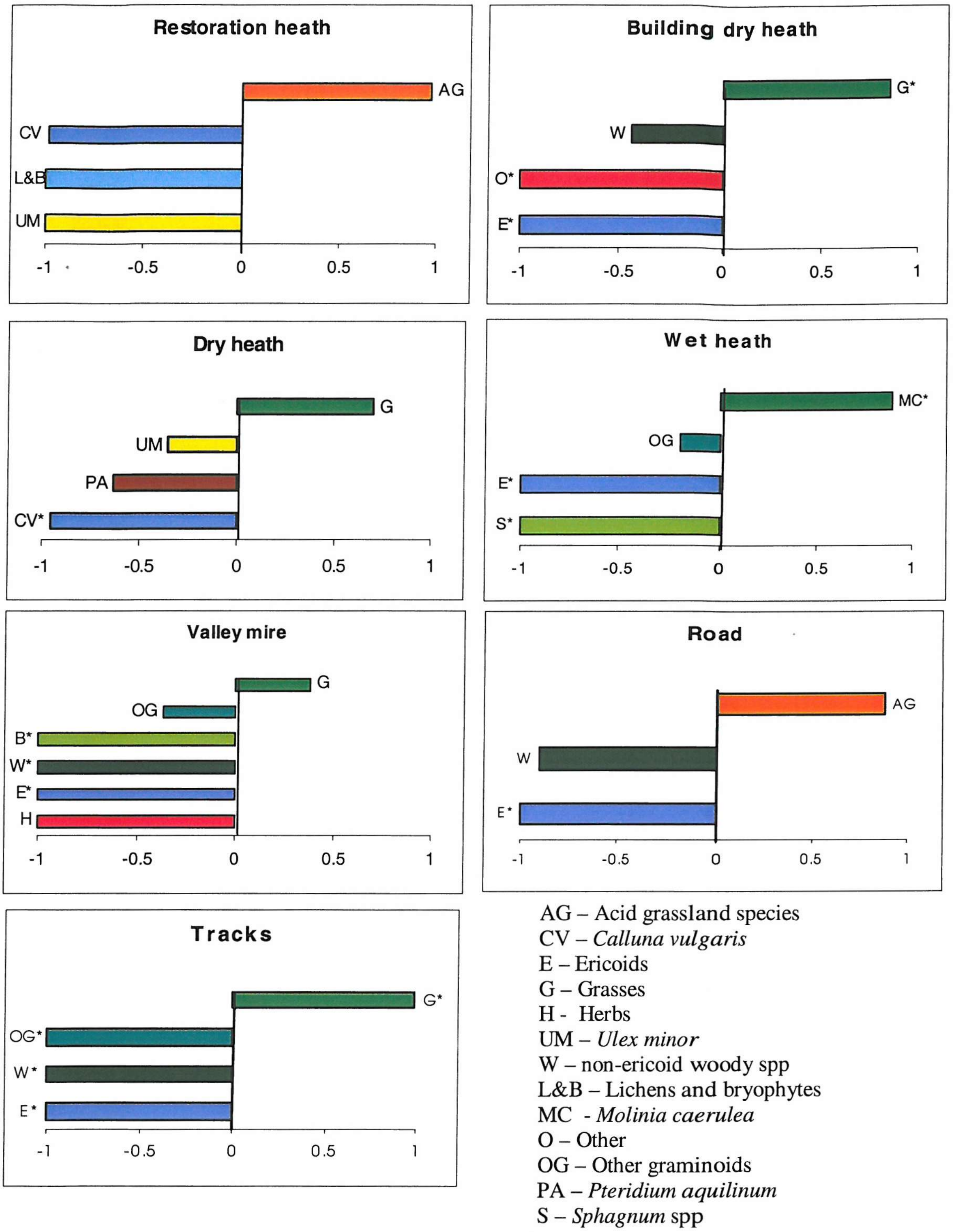
Figure 2.53 Dunging observations compared to total location observations in each habitat at Hartland by Exmoor ponies.

Foraging

Availability and use of plant species in each habitat by New Forest ponies at Hartland are presented in Table 2.16. Selection is shown in Figure 2.54. Results were very similar between herds, although a greater number of species were recorded as eaten by the New Forest ponies (possibly a result of the greater number of observation periods). In general, ponies preferentially selected grasses from every habitat. *Calluna vulgaris* (but no other ericaceous species) and *Ulex minor* were occasionally eaten on dry heath habitats, where Exmoor ponies also positively selected bracken *Pteridium aquilinum* (Figure 2.55). *Rubus fruticosus* was selected from roadside hedges. All graminoids were eaten in wet heath and valley mire, but no herbaceous species. There were insufficient foraging observations in scrub (SC) to test although selection appeared non-random, with. 89.8% of grasses, 3.9% of ericoids and 6.3% of woody species were eaten. On Stoborough Heath, New Forest ponies showed the same general selection pattern, positively selecting grasses in both scrub (SC) and managed scrub/wood (mSC/W) (Table 2.17, Figure 2.56).

Table 2.16 Plant species availability and use (the average number of bites of each plant) by New Forest ponies at Hartland Moor. * - P <0.05, ** P < 0.01, *** - P < 0.001

Habitat	**Species**	**% cover mean (SE)**	**% bites**	**Chi square**
AG/DH	Acid grassland	70.5 (5.3)	98.8	16.2**
	Ericoids	21.6 (2.2)	1.2	
	Ulex minor	4.1 (3.3)	0	
	Lichens & bryophytes	3.9 (2.7)	0	
BDH	Grasses (*Agrostis curtisii, Molinia caerulea*) (G)	27.7 (5.3)	94.8	65.1***
	Ericoids (E)	59.2 (5.8)	2.4	
	Other woody spp (W)	6.2 (2.7)	2.7	
	Other (*Pteridium aquilinum, Rumex acetosella, bryophytes*) (O)	6.2 (2.1)	0.1	
DH	*Calluna vulgaris* (CV)	92.1 (2.6)	8.0	1141.9***
	Ulex minor (UM)	5.5 (2.1)	5.2	
	Grasses *(Agrostis curtisii, Molinia caerulea)* (G)	1.2 (1.7)	86.2	
	Pteridium aquilinum	0.2 (0.7)	0.5	
WH	*Molinia caerulea* (MC)	45.9 (4.3)	83.8	26.6***
	Ericoids (E)	42.7 (3.1)	16.0	
	Other graminoids (*Eriophorum angustifolium, Trichophorum cespitosum, Rhynchospora, Juncus effusus, Narthecium ossifragum*) (O)	8.5 (1.6)	0.2 0	
	Sphagnum spp. (S)	2.9 (2.4)		
VM	Grasses (*Molinia caerulea, Phragmites communis, Agrostis stolonifera* (G)	50.0 (3.6)	75.2	11.25**
	Other graminoids (as for WH plus *Schoenus nigricans*) (O)	27.7 (3.5)	18.1 6.2	
	Herbs (various) (H)	8.4 (2.0)	0	
	Ericoids (E)	1.9 (0.8)	0	
	Other woody spp. (*Ulex* spp., *Betula* sp., *Salix* sp.)	3.2 (0.4)	0	
	Bryophytes (B)	9.6 (3.5)		
T	Grasses (*M. caerulea, A. curtisii, A. capillaris*) (G)	54.7 (5.9)	98.2	30.36***
	Ericoids *(C. vulgaris, E. cinerea)* (E)	39.2 (5.0)	1.3	
	Other woody spp (W)	4.7 (1.5)	0.4	
	Other graminoids (*Carex* spp., *Juncus* spp.) (OG)	1.4 (0.6)	0.1	
R	Acid grassland (AG)	67.1 (3.79)	95.1	33.25***
	Ericoids *(C. vulgaris, E. cinerea, E. tetralix)* (E)	3.2 (0.59)	0.4	
	Other woody species (*Ulex* spp., *Betula* sp, *Salix* sp. *Alnus* sp., *Rubus fruticosus* agg.) (W)	29.7 (1.41)	4.5	



AG – Acid grassland species
CV – *Calluna vulgaris*
E – Ericoids
G – Grasses
H - Herbs
UM – *Ulex minor*
W – non-ericoid woody spp
L&B – Lichens and bryophytes
MC - *Molinia caerulea*
O – Other
OG – Other graminoids
PA – *Pteridium aquilinum*
S – *Sphagnum* spp

Figure 2.54 Selection of plant species/categories by New Forest ponies at Hartland Moor, expressed as Jacobs' indices. Species for which selection is significantly different from zero are denoted by * (Wilcoxon signed rank test, $P < 0.05$. Tests were not possible for AG/DH, for which $n < 6$. See Table 2.16 for species categories

Ranking of species selected by New Forest ponies at Hartland Moor:

BDH: G > all other categories (Mann-Whitney, $P < 0.01$);

DH: G > all other categories (Mann-Whitney, $P < 0.005$);

WH: MC > all other categories (Mann-Whitney, $P < 0.001$), OG > E, S (Mann-Whitney, $P < 0.01$);

VM: > E, W, B (Mann-Whitney, $P < 0.001$), OG > E, W, B (Mann-Whitney, $P < 0.05$);

T: G > all other categories (Mann-Whitney, $P < 0.005$);

R: AG > all other categories (Mann-Whitney, $P < 0.05$).

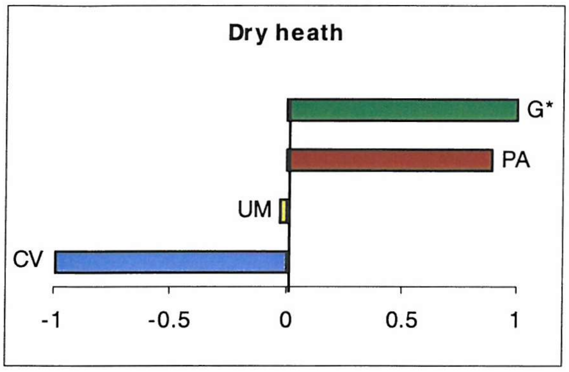


Figure 2.55 Species selection on dry heath by Exmoor ponies, expressed as Jacobs' indices.

<u>Ranking</u>: G > all other categories (Mann-Whitney, $P < 0.001$)

Table 2.17 Plant species availability and use (the average number of bites of each plant) from additional habitat types present on Stoborough Heath.

Habitat	**Species**	**% cover mean (SE)**	**% bites**	**Chi square**
SC	Grasses (*Molinia caerulea, Agrostis curtisii*) (G)	20.3 (5.1) 68.4 (4.4)	100	23.4**
	Ulex europaeus (UE)	11.3 (2.9)	0	
	Ericoids (*Calluna vulgaris, Erica cinerea*) (E)		0	
mSC/W	Grasses (*Molinia caerulea, Agrostis curtisii, Holcus lanatus*) (G)	25.1 (3.4) 3.8 (1.6)	90.6 0	84.2***
	Ericoids (*Calluna vulagris, Erica cinerea*) (E)	69.5 (7.2)	5.7	
	Ulex spp. (U)	0.1 (0.1)	0.1	
	Non-ericoid woody spp. (*Salix spp., Rubus frusticosus, Betula pendula*) (OW)	1.5 (0.7)	1.2	
	Other (*Cirsium arvense*) (O)			

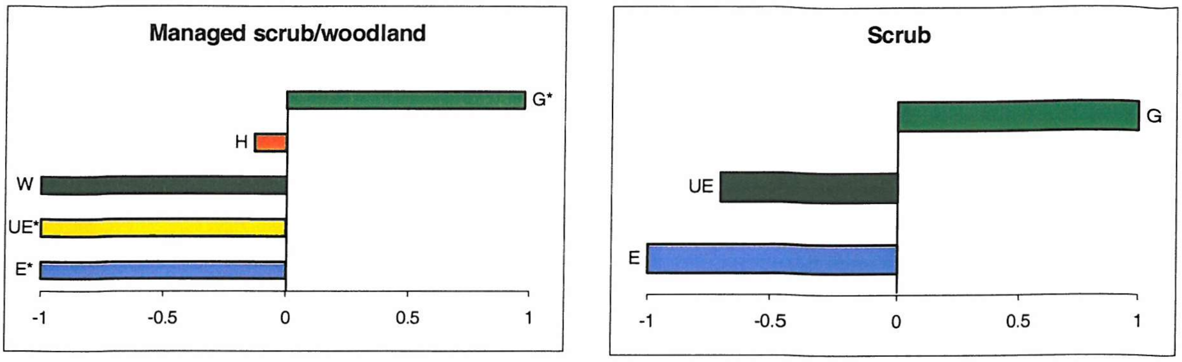


Figure 2.56 Selection of plant species/categories by New Forest ponies expressed as Jacobs' indices on additional habitat types present on Stoborough Heath. Species for which selection is significantly different from zero are denoted by * (Wilcoxon signed rank test, $P < 0.05$).

Ranking:

mSC/W: G > E, UE (Mann-Whitney, $P < 0.005$), G > W, H (Mann-Whitney, $P < 0.01$).

Bite rate (Figure 2.57a) was highest in grass-rich habitats (acid grassland AG, restoration heath AG/DH) and lowest in dry heath (DH) and wood (W). Step rate during foraging (Figure 2.57b) also varied between habitats, and was highest in DH and roads (R) and lowest in valley mire (VM).

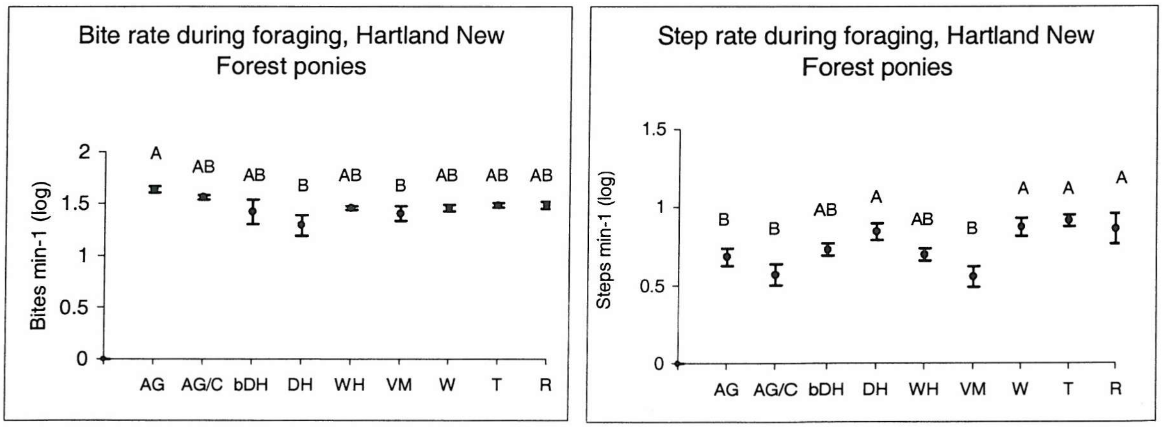


Figure 2.57(a) bite rate and (b) step rate during foraging rates of New Forest ponies at Hartland in different habitats. Letters distinguish groups between which there were significant differences in foraging rate (bites: GLM, $F = 3.06$, $P = 0.012$; steps: GLM, $F = 4.12$, $P = 0.002$). There were no significant differences between individuals. See Table 2.2 for habitat abbreviations.

Habitat use

A summary of habitat use by livestock groups at each site is presented in Figures 2.58a-c.

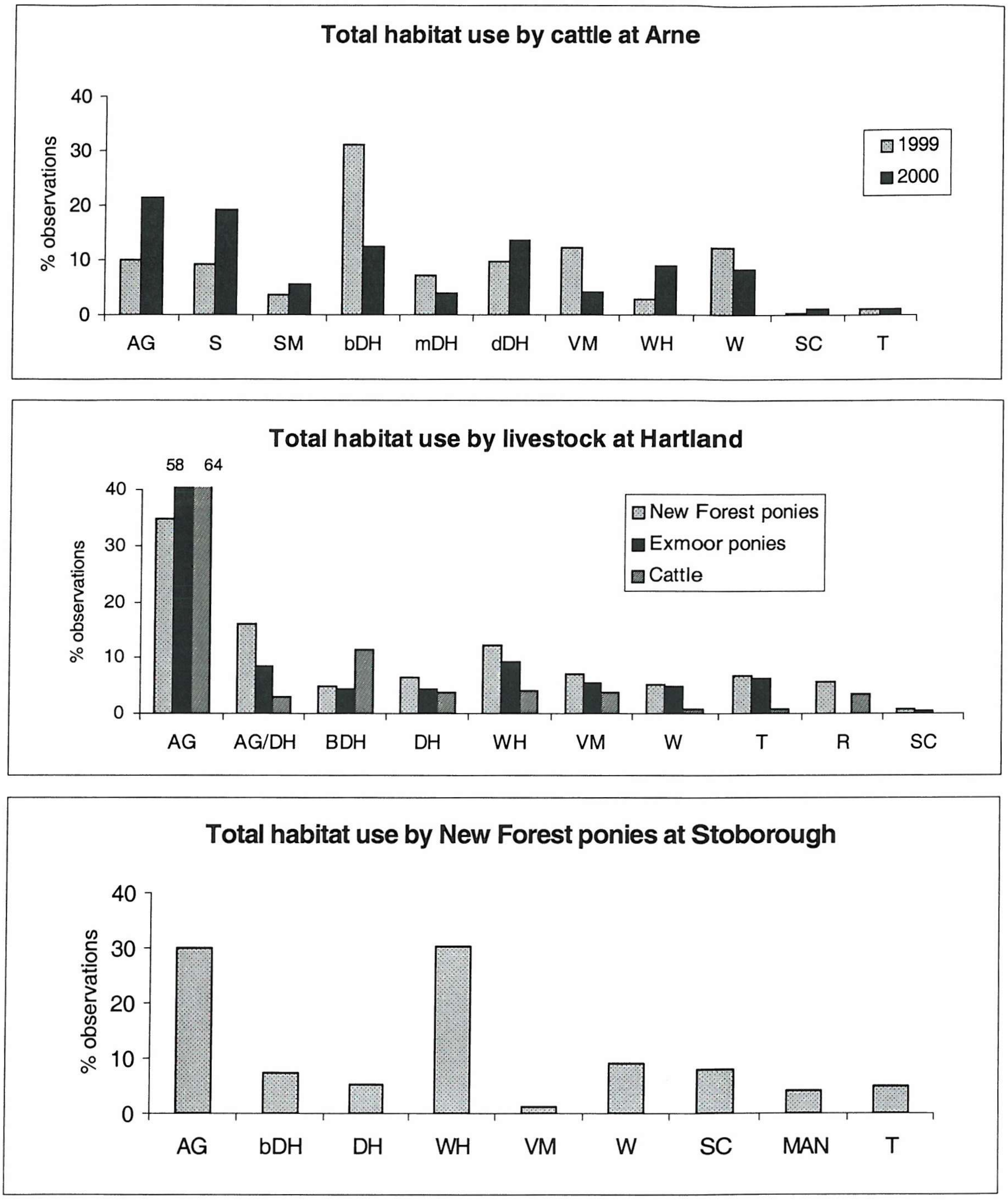


Figure 2.58 Total habitat use by (a) cattle at Arne in 1999 and 2000, (b) cattle, New Forest ponies and Exmoor ponies at Hartland and (c) New Forest ponies at Stoborough. See Table 2.2 for habitat abbreviations.

Resting locations

The position of 23 night resting locations for all livestock types combined were significantly closer to shelter that an equal number of randomly generated points on both Hartland and Arne were (one-way ANOVA, Hartland: F = 9.42, P = 0.004; Arne: F = 10.01, P = 0.003) (Figure 2.59). Particular locations were repeatedly used for resting - for example, Exmoor ponies used one place for overnight resting on four of the eight 24-hour observations. Casual observations also suggested that the New Forest ponies and the Arne cattle returned repeatedly to particular over-night resting sites.

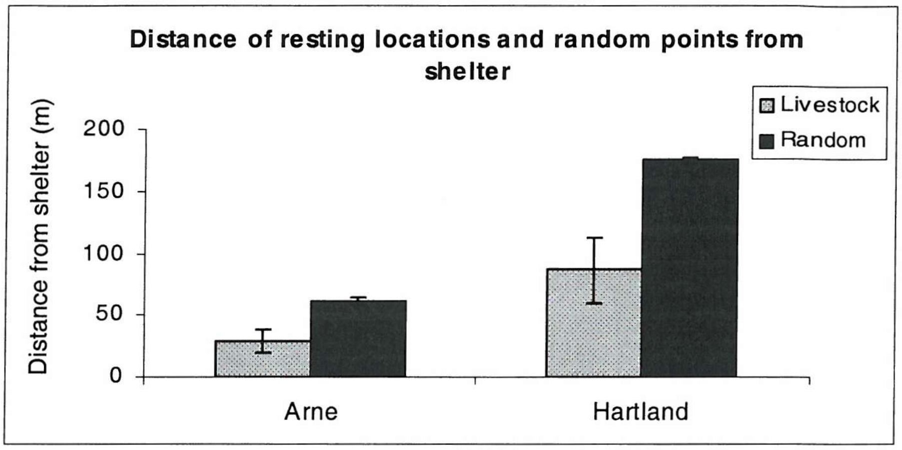


Figure 2.59 Distance between livestock resting locations and shelter compared to the distance between random locations and shelte.r

Dunging

Differential foraging/dunging patterns (see Figure 2.60) suggested that acid grassland species were likely to be introduced to heath vegetation. This was confirmed by the survey of dung seedlings (Table 2.18). There was no difference in the species found in each habitat (chi square).

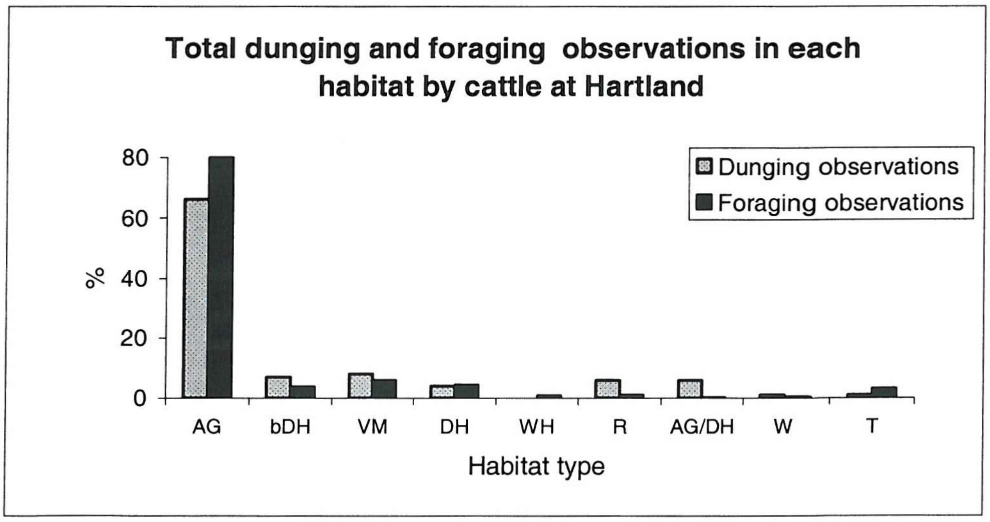


Figure 2.60 Percentage of cattle dunging and foraging observations in each habitat on Hartland Moor.

Fifty-one cowpats supporting seedlings were found. Seedlings were not found in (identifiable) pony dung, which appeared to disintegrate and become absorbed into the sward much faster than cowpats. Cowpats remained recognisable after at least seven months. Very little dung was found in wet heath (WH) and valley mire (VM) (partly because it disintegrated faster in wetter conditions). Only one cowpat with seedlings was found in VM. However, cowpats with seedlings were frequent in dry heath (bDH, DH), often clustered in resting areas. With the exception of *Polygala* sp. and *Agrostis curtisii*, only species found in the acid grassland sward were observed.

Table 2.18. Seedling species found in cowpats at Hartland Moor. All species were also recorded growing in the acid grassland sward, except *Agrostis curtisii* and *Polygala* sp.

Species	**English name**	**Frequency in cow pats (%)**
Agrostis capillaris	*Common bent*	11.1
Agrostis curtisii	Bristle bent	5.6
Bellis perennis	Daisy	2.7
Cerastium spp.	Mouse-ear	38.9
Geranium molle	Cranesbill	25.0
Holcus lanatus	Yorkshire fog	8.3
Lolium perenne	Perennial rye-grass	2.7
Luzula campetris	Field wood-rush	69.4
Medicago lupulina	*Black medick*	8.3
Poa annua	Annual meadow grass	11.1
Poa pratensis	Smooth meadow-grass	41.7
Polygala sp.	Milkwort	16.7
Rumex acetosella	Sheep's sorrel	36.1
Sagina sp.	Pearlwort	16.7
Senecio jacobea	Ragwort	5.6
Veronica spp.	Speedwell	44.4

2.4 Discussion

The results presented in this chapter show clearly that livestock did not use habitats present on heathland sites in Dorset according to their availability, but positively selected particular habitat types while avoiding others. In addition, some habitats were used more for particular behaviours than others, and use varied diurnally and seasonally. These apparent preferences differed somewhat between livestock species and breeds, and were not always in agreement with those found in previous work, reinforcing the idea that some aspects of resource selection will be specific to sites and/or livestock types and herds. Some clear overall trends did, however, emerge. In line with the theme of this thesis, these trends are considered in terms of livestock impacts on heathland rather than livestock performance.

2.4.3 HABITAT SELECTION.

Selection indices used in this context provide a relative measure of the livestock pressure (i.e. number of livestock per hectare) on each habitat type present when a mosaic of different habitat types is present. On the Dorset heathland sites studied, all livestock groups positively selected (i.e. pressure was greatest on) habitats supporting a high proportion of grasses and/or young heather. Habitats with a strong seasonal variation in the availability and/or quality of suitable forage (valley mire, wet heath) showed a corresponding variation in use by livestock. Non-grassland habitat types, in which recent deliberate management or use had led to a higher proportion of grasses or young heather (e.g. building dry heath, managed scrub/wood, tracks) than in the corresponding unmanaged habitat (mature dry heath, degenerate dry heath, wood, scrub), were also strongly preferred. In addition, foraging rate was greatest in these grassy habitats. This all suggests that suitability for foraging was a key factor influencing overall habitat selection, as suggested by previous studies on heathland and moorland (e.g. Gordon, 1989c, Pratt *et al.,* 1986; Wallis de Vries, 1994). Drier heathland types (mature dry heath on Arne, Hartland and Godlingston, also humid heath classified as wet heath on Stoborough), which were the most abundant habitat type, were generally negatively selected, as were habitats that had dense scrub or trees (e.g. wood, scrub), although there was some seasonal variation. However, the positive selection for degenerate dry heath at Arne, which supports a low cover of grasses and no young heather, suggests that factors other than suitability for foraging can also influence overall habitat selection.

A similar preference for grassland habitats over heathland habitats has been found in a number of other studies of resource selection on heathland sites (e.g. Bakker *et al.*, 1983; Osborne, 1984; Pratt *et al.*, 1986; Putman *et al.*, 1987; Gordon, 1989b; Wallis de Vries, 1991, see Table 1). However, in this study the fine-scale classification of dry heath into building, mature and degenerate heath, plus the separate classification of tracks, has shown that selection for different dry heathland types is not necessarily uniform, and that different heathland types may be selected for different behaviours.

2.4.3 SEASONAL CHANGES

Most studies of free-ranging livestock on heathland show clear seasonal changes in habitat use. These changes are likely to be a consequence of forage depletion and changes in forage quality within habitat types plus changing requirements for shelter, and therefore vary between sites and species. For example, Pratt *et al.* (1986) found that New Forest ponies used grassland most in the summer, wet and dry heath in late summer, and woodland and gorse-brake in the winter. Bakker *et al.* (1993) found that sheep preferred grassland in the summer, heathland and woodland in the winter, while Wallis de Vries found a decrease in use of heathland by cattle in winter. In this study, overall selection for particular habitats was so strong that seasonal changes in use were not necessarily reflected in significant changes in the relative ranking of selection indices for habitats. However, on all sites, selection of wet heath for foraging declined in autumn/winter as the water level rose and growth ceased (*Molinia caerulea*, the dominant grass species is deciduous and sheds dried, straw-like leaves in winter), and this agrees with previous studies (e.g. Pratt *et al.*, 1986; Gordon, 1989b). A greater number of 24-hour follows may have detected subtler trends.

At Hartland and Stoborough, use of acid grassland (and restoration heath) by ponies was highest during the winter months. Cattle were removed from the heath during the winter, but use was highest in autumn. At Arne, however, there was a decline in the use of acid grassland and a corresponding increase in the use of dry heath in autumn. Seasonal change will depend upon the alternative habitats available and the seasonal availability of forage, and may also vary between livestock types. At Arne, there were just under twice as many livestock units (1LU = one cow, or two ponies) per hectare of grassland as at Hartland, and probably a larger deer population. The availability of forage may therefore have declined more rapidly at Arne, forcing the livestock to use alternative habitats. Hartland also comprises more open, unsheltered habitat than Arne, and hedges and woodland adjacent to acid grassland may have provided necessary winter shelter, which, at Arne, could be found on other habitats (the building dry heath and degenerate dry heath used by cattle at Arne in winter were sheltered by small copses). Daily cattle records at Hartland also showed an increase in the use of woodland in autumn, which supports this. Pratt *et al.* (1986) found a clear selection for sheltered habitat (woodland and gorse brake) in the winter by ponies in the New Forest. (Note that the number of 24-hour follows that were possible within the constraints of this project was insufficient to analyse meaningfully the affects of daily weather conditions). At Stoborough, use of acid grassland was heavily influenced by the presence of supplementary feed in winter (although use of acid grassland was increasing in winter before supplementary feeding took place).

Table 2.19 Summary of significant seasonal habitat change from summer to autumn/winter by each livestock group studied in Dorset (non-significant trends are given in parentheses). AG – acid grassland, AG/DH – restoration heath, bDH – building dry heath, dDH – degenerate dry heath, WH – wet heath, VM – valley mire

	Increase	**Decrease**
Arne cattle 1999	(bDH, dDH)	WH & VM
Arne cattle 2000	dDH	(AG)
Hartland cattle	AG/DH	WH (VM)
New Forest ponies, Hartland	(AG/DH)	WH
Exmoor ponies, Hartland	AG	(all others)
New Forest ponies, Stoborough	AG	WH

There was also a seasonal difference in the distance travelled each day. At both Hartland and Stoborough, where livestock remained on site throughout the year, the distance travelled declined markedly in the winter months, when the animals restricted themselves mainly to acid grassland, with occasional forays to water sources. Distance travelled by cattle at Arne also decreased in the autumn. The reduction was probably a direct consequence of fewer daylight hours, as livestock tended not to move far after dark. Livestock may also have chosen to remain in more sheltered habitat patches – Pratt *et al.* (1986) found that shelter was a key factor in seasonal habitat selection by ponies in the New Forest.

A further seasonal change was in herd size. This did not occur in the Arne cattle or the Exmoor ponies on Hartland, but was observed in the cattle at Hartland. Daily cattle location records from Hartland showed that group size was at its lowest in September.

This was probably due to the depletion of resources on the preferred foraging habitat. The acid grasslands generally became parched during August, and growth did not recommence until rainfall increased in the autumn. During this period, the cattle made greatest use of wet heath and valley mire for foraging. The areas of wet heath and valley mire that the cattle could use formed smaller patches than the acid grassland. Therefore, the decrease in group size during this period may have been a consequence of the herd splitting into smaller groups to allow all individuals within the group to have access to the selected habitat.

2.4.4 BEHAVIOUR

Use of habitats for key behaviours are summarised in Table 2.20. Note that this is not the selection of habitats for key behaviours, but the habitats in which each behaviour was predominant. Due to the large proportion of time spent foraging, ponies used every habitat predominantly for foraging. Therefore, in their case, those habitats in which the time spent on other behaviours was also considered notable are listed.

Selection of habitats for key behaviours is summarised in Table 2.20. The small available area of tracks means that relatively few observations were needed to result in selection for this habitat being positive. However, time spent on tracks takes into account only a small proportion of the ponies' total time. In addition, the impact of livestock on this habitat type, which is already likely to have a short sward height and a large amount of bare ground (see Chapter 3) may be of limited importance compared to other habitats (but see Chapter 4 for discussion of damp track species of conservation concern). Therefore, for New Forest ponies on Hartland, habitats with Jacobs' indices just under 0.5 are also listed (in parentheses).

Table 2.20 Summarised main use of habitats by each livestock group for key behaviours. Habitats showing no significant differences between behaviours due to limited data were excluded. AG – acid grassland, AG/DH – restoration heath, bDH – building dry heath, DH – dry heath, R – roads, S – saltings, SC – scrub, SM – saltings margin, T – tracks, WH – wet heath, VM – valley mire.

	Foraging	**Resting**	**Moving**
Arne cattle	AG, SM, S	DH, W	T
Hartland cattle	VM, WH	AG, AG/DH,	-
New Forest ponies, Hartland	All habitats	T, DH, bDH, AG	T, R
Exmoor ponies, Hartland	AG, AG/DH, WH, VM, SC	bDH, T, W	T, DH
New Forest ponies, Stoborough	All habitats	DH, SC	T, DH

Table 2.21 Summary of habitats positively selected (Jacob's index > 0.5) by each livestock group for key behaviours. Habitats with indices just under 0.5 are included in parentheses for Hartland New Forest ponies (see text). AG – acid grassland, AG/DH – restoration heath, bDH – building dry heath, DH – dry heath, mSC/W – managed scrub/wood, R – roads, SM – saltings margin, T – tracks, WH – wet heath, VM – valley mire, W – woodland

	Foraging	**Resting**	**Moving**
Arne cattle	AG, SM	BDH, AG	AG, T, SM
Hartland cattle	AG	BDH, AG	R, AG
New Forest ponies, Hartland	T (AG, AG/DH, R)	T (AG, AG/DH)	T, R
Exmoor ponies, Hartland	AG	W, bDH, T	T
New Forest ponies, Stoborough	AG, mSC/W	AG, bDH, mSC/W	AG, T, mSC/W

Habitats positively selected for foraging were all relatively grass-rich habitats – acid grassland, saltings, saltings margin (also tracks). Other habitats used predominantly for foraging included wet heath and valley mire, also both comparatively grass-rich. Of the remaining habitat types, most foraging was observed in the managed patches. Dry heath managed through cutting, burning or undergoing other forms of disturbance (building dry heath, also tracks), and scrub and wood managed through cutting all supported a higher cover of grasses than unmanaged patches. This common characteristic, plus the positive selection for grasses in all habitats (see foraging below), suggests that it is the relative abundance of grasses that make them positively selected. However, burning and cutting also increase the nutritional value of *Calluna vulgaris* by stimulating young growth (e.g. Gimingham, 1972; Milne *et al.*, 1979) and this was reflected in the higher selection of *Calluna* in building dry heath and tracks than mature dry heath, where the plants are by definition older. The presence of young *Calluna* is likely to increase the selection for these habitats for foraging.

Habitats selected for resting were all dry (acid grassland, dry heath, track), and locations where resting continued for a significant period (the majority of which were during the night) were also relatively sheltered by wood, hedges, or scrub patches (within 50m at Arne, 100m at Hartland). Cattle at Arne tended to use wood or heathland for overnight resting, selecting degenerate dry heath in the autumn, presumably as the greater depth of heather provided more warmth. In contrast, cattle at Hartland used building dry heath in summer only, remaining on acid grassland in autumn, and ponies only used acid grassland for overnight resting, although they were observed resting on other habitats during the day. Pratt et *al.* (1986) also found that the presence of shelter influenced diurnal habitat selection by cattle (but not ponies) in the

New Forest, with more sheltered habitat used during the night. They also suggested that cattle were more likely to move to another habitat at dusk to seek a suitable resting site, and were frequently seen on 'purposeful route marches' whereas ponies tended to 'drift between habitats while feeding'. In Dorset, habitats were used for moving in three ways. Livestock either moved in an apparently directionless manner while foraging within a habitat patch; moved in a particular direction to another habitat patch, foraging en route; or moved quickly, sometimes running, and did not stop to forage. In general, tracks or small paths were used for the two more deliberate types of movement – livestock were rarely observed moving spread out through a habitat, but tended to move more or less in single file. All livestock groups generally moved purposefully to a different habitat patch (not necessarily habitat type) around dusk, and remained there until daybreak the following day. However, New Forest ponies tended to spend more time moving and foraging than the other livestock groups, which were more likely to remain foraging within a single habitat patch (and this is reflected in the New Forest ponies' positive selection for a greater number of habitat types).

Habitats are likely to be used predominantly for moving if they are not positively selected for another behaviour but are located between habitat patches that are, for example dry heath (and to a lesser extent saltings margin) on the sites studied. Consequently, the proportion of time in dry heath spent moving was greater than in other habitats. Tracks present the easiest route for moving between one habitat patch and another, and most livestock groups positively selected tracks for moving on – as can be seen clearly in the Hartland route maps (Figure 2.33). This was not the case at Arne, where, as the site was smaller, a smaller proportion of the total time was spent moving between habitat patches used for resting or foraging. The high selection of acid grassland on all sites for moving was a result of the high overall selection of this habitat, and it can be seen from Table 2.21. that moving was not a key behaviour in this habitat. Overall, tracks were most selected for moving by all livestock groups, together with habitats positively selected for other behaviours.

Only valley mire, wet heath and degenerate dry heath were used quite specifically for one behaviour: foraging in the case of valley mire and wet heath, and resting in the case of degenerate dry heath (Arne only). In some cases, habitats were positively selected for more than one behaviour – for example, building dry heath had a high selection index for resting, and was also significantly used for foraging (except at Stoborough). To some extent, if a habitat was used for foraging, it was also likely to be used for resting, particularly by ponies, who tended to rest frequently where they were for short periods throughout the day rather than in fewer, longer periods elsewhere. Conversely, habitats selected for resting were used for foraging either side of resting periods. Moving was also often carried out with another behaviour.

Another behaviour of key interest in the context of heathland management is dunging. Heathland vegetation can be affected by the nutrient enrichment and plant dispersal resulting from dunging (Chapter 1), which also changes conditions for invertebrates such as dung beetles and their predators. Non-random dunging behaviour leads to nutrient transfer and species dispersal within and between vegetation communities. Dung was, on the whole, dispersed in proportion to total habitat use. Cattle, and to a lesser extent ponies, were observed to defecate on standing up after resting and before moving off or starting to graze. Therefore dunging was expected to be disproportionately high in habitats used predominantly for resting. However, no clear pattern was apparent, although wood, dry heath and restoration heath were variously all used disproportionately for dunging. Dunging in acid grassland and saltings was slightly lower than expected, although possibly also under-recorded, since livestock tended to be most dispersed, and therefore less easy to see, in these habitats. Dunging on habitats used for resting over-night may have been underestimated when the livestock were already standing when relocated. Dung was noticed to be particularly concentrated around over-night resting areas, and particularly those used regularly (attempts to clarify this by dung counts were frustrated by variable decay rate and visibility in different habitats). Further patchiness in dung distribution was caused by the use of latrines by ponies. Work in the New Forest (Edwards & Hollis, 1982; Ekins, 1989) suggested that ponies segregate feeding and dunging areas on grassland communities. The ponies' reluctance to graze latrine areas leads to distinct areas of short and longer swards. This behaviour is well known from horses in captivity (Odberg & Francis-Smith, 1976), but has not been observed on extensive semi-natural habitat. Cattle are considered to dung at random (Marsh & Campling, 1970) although, as mentioned, there is a tendency for aggregations of dung to occur in areas used for lying up.

Dunging is considered to be an important mechanism by which plant species are dispersed between heathland habitats (e.g. Welch, 1984). Foraging behaviour on the Purbeck heaths suggested that dunging is likely to result in acid grassland species being introduced to heath communities, as livestock spent most time foraging on acid grassland. Even on heathy vegetation, cattle ate mainly grasses (see below), therefore grass seeds and seeds from herbs found in acid grassland were most likely to be dispersed by dung. Differential foraging/dunging patterns further increased the likelihood of acid grassland species being introduced to heath vegetation. This was confirmed by the seedling species recorded on dung.

2.4.4 SELECTION OF PLANT SPECIES

All livestock groups showed very strong selection between plant species/groups when foraging. The strongest positive selection was for grasses in every habitat studied, despite wide variation in availability (two exceptions to this are discussed below).

Considered together, bite and step rate give an indication of the search effort required in each habitat. However, this is blurred by the different handling times required by different vegetation types – for example, a prickly, woody species such as *Ulex minor* requires a greater handling time than grass species, and therefore fewer bites will be taken per step, irrespective of its distribution throughout the sward. Bite rates were relatively consistent between sites and livestock groups, and were highest in habitats with greatest grass cover (acid grassland, restoration heath, saltings margin). Wet heath and valley mire generally had a slightly lower rate, suggesting that slightly more time was spent searching. Bite rates were lowest in dry heath habitats, where step rates were generally highest. Foraging impact may therefore vary according to habitat type, as trampling and vegetation removal may occur at different rates in different habitat types.

No selection between species was observed in the intimate mix of grasses and small herbs comprising the short acid grassland sward. However, selection was apparent in wet heath and valley mire, where *Molinia caerulea* was the only positively selected species, although other graminoids were occasionally eaten (*Eriophorum angustifolium, Juncus articulatus, Juncus effusus, Phragmites communis, Rhynchospora alba, Schoenus nigricans, Trichophorum cespitosum*). These results are similar to those of Putman *et al.* (1987), who used faecal analysis. However, most foraging observations in Dorset were from wet heath and valley mire where *M. caerulea* was dominant, and selection may alter where this dominance is reduced. For example, the few records of ponies eating *T. cespitosum* and *R. alba* were from an area where *M. caerulea* was a very limited component of the sward. Using oesophageal fistulation on upland blanket bog Grant *et al.* (1987) found a greater proportion of *T. cespitosum* or *Eriophorum* spp. (depending on the season) in the diet than *M. caerulea*, which was less abundant in the sward. Preferences between graminoids within wet heath and valley mire will depend on their relative abundance, and this will be influenced by season, as *M. caerulea* is deciduous.

Foraging rates were generally lowest in building dry heath, tracks, scrub, managed scrub/wood and particularly (mature) dry heath (which also consistently had the widest range). Grasses were positively selected in each of these habitats, but were much scarcer than in the habitats discussed above. Therefore longer was spent searching in these habitats. Ericoids (predominantly *Calluna vulgaris*, but very occasionally both *E. tetralix* and *E. cinerea*) and *Ulex* spp. were eaten, but rarely positively selected. The exception was at Arne in 1999, where cattle preferentially selected *Calluna vulgaris* and *Ulex minor* in building dry heath and mature dry heath respectively. In the remaining livestock groups, *Calluna* was preferred to woody species (including *U. minor*) by cattle, whereas, *Ulex* spp. appeared to be preferred to *Calluna* by ponies, irrespective of heather phase. This difference is supported by work in the New Forest, where cattle ate more *Calluna* than ponies (Putman *et al.*, 1997), but only ponies ate *Ulex* spp. Woody species other than ericoids and *Ulex* spp. were generally little eaten in Dorset, although *Pinus* sp was positively selected by cattle at Arne in 1999. Occasional observations of cattle eating *Betula pendula* were also made, and ponies on Stoborough were seen to browse *Salix* sp. All livestock types were observed eating *Rubus fruticosus* at Hartland, and ponies occasionally ate *Pteridium aquilinum*. This was positively selected by Exmoor ponies. No positive preference or avoidance of this species has been found in previous studies (e.g. Bakker, 1993), and the positive selection for bracken seen in this study may have been a consequence of the distribution of the bracken patches where grazing was observed (e.g. between grass-rich patches regularly used for grazing). As discussed in Chapter 1, the toxicity of bracken to grazing livestock makes livestock grazing an inappropriate tool for bracken control. Tree leaves were eaten by cattle and particularly ponies in the New Forest (Putman *et al.*, 1997), where woodland cover is greater, but did not contribute much to the diet.

2.4.5 VARIATION IN RESOURCE SELECTION

Differences in resource selection were apparent between individuals, livestock groups and sites. There is much interest in the differences between livestock types used for conservation grazing (e.g. Wallis De Vries, 1993; Read, 1994; Oates, 1994; Hearn, 1995; Bullock & Armstrong, 2000; Oates & Tolhurst, 2000). In this study, some notable differences were apparent between the livestock groups, but the observations were not designed to provide a comparison of resource selection and behaviour between species or breeds.

There are a number of reasons why results may not be transferable between livestock groups, years and sites. Firstly, vegetation may change between years as consequence of the weather conditions, other management practices, and livestock presence. Secondly, the needs of livestock will vary according to physiological differences between groups (e.g. lactating and non-lactating). Thirdly, the mix of livestock may affect habitat selection through direct or indirect competition. In addition, site characteristics can affect apparent resource selection, preventing indices from any particular site from being more widely applicable. For example, the use of habitats for different behaviours may lead to differences in selection indices between sites. If the time spent on a particular behaviour (for example, resting) is constant irrespective of habitat availability, changes in the availability of a habitat selected for another behaviour (for example, foraging) will not necessarily alter the amount of time spent in the resting habitat. However, as relative availability has changed, so will the selection indices (i.e. relative livestock pressure). Similarly, if livestock need to leave a habitat that is positively selected for foraging to obtain water, an increase in the availability of that habitat will not necessarily reduce the time spent in the habitat containing water or the habitats travelled through to reach the water.

Habitat accessibility is another consideration. In this study, availability of each habitat was expressed as its area. However, the accessibility of habitat patches varied. Areas considered inaccessible to livestock were excluded (e.g. parts of the saltings at Arne). However, there were other areas, particularly at Hartland, where accessibility was reduced, but access was theoretically still possible. Figure 2.62 shows livestock location records for all livestock groups at Hartland (including daily location records). There are very few records in the central northern section of the grazing unit. This area is separated by a deep valley mire from the south of the site, and the only access is via a narrow strip of dry heath from the western road, and the very limited use of this central northern section of dry heath further decreased the apparent selection for this habitat. Again, while differential accessibility to habitat patches is not a problem in describing habitat selection on a specific site, it limits the use of those selection indices to predict habitat use on another site.

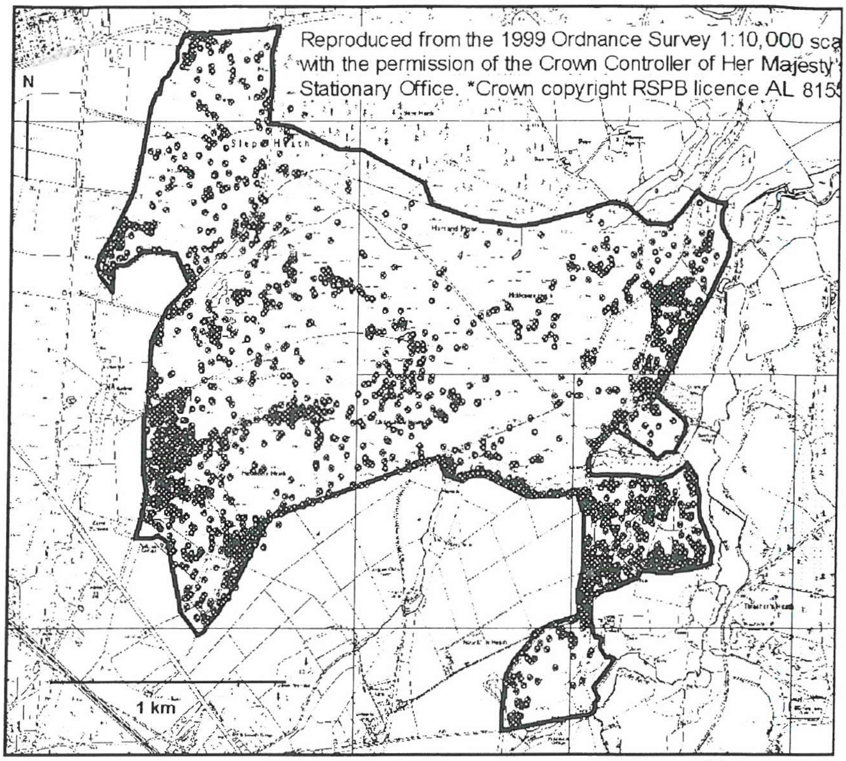


Figure 2.62 All recorded livestock locations on Hartland Moor, 1996-2001.

However, there were some differences between livestock types in Dorset that are considered worth highlighting, either because they are likely to have some relevance elsewhere, or because further work is necessary to establish whether they are more widely applicable. Most notably, total time spent on behaviours varied markedly between cattle and ponies, with cattle spending about half of the amount of time foraging spent by ponies: 27.5-32.1% of total time compared to 57.1% - 60.8% (considering complete 24 hour observations only). This is less than shown by data from the New Forest, where Pratt *et al.*, (1987) described cattle feeding for 57% of their time and ponies for more than 75%. This may be due to the classification of animals moving with their head raised during a grazing bout as 'moving', not 'foraging' in this study. However, the difference between species is shown by both studies, and suggests that (i) suitability for foraging may play a greater role in the overall selection of habitats by ponies than by cattle; (ii) ponies may remove more biomass per capita than cattle and therefore have a greater impact on vegetation. In addition to spending less total time resting, the ponies studied also spent a greater proportion of it standing as opposed to lying (e.g. at Hartland, Exmoor ponies spent 76.5% of resting time standing, New Forest ponies 55.2%, and cattle 9.6-14.0%). Although short swards (including building dry heath) does not appear to suffer, mature and degenerate *Calluna* tends to be crushed where cattle lie on it.

Other behavioural differences were slight. On average, Exmoor ponies at Hartland travelled further within 24 hours than cattle, who moved further than New Forest ponies. However, New Forest ponies spent longer moving. Exmoor ponies spent slightly less time foraging and more time resting than New Forest ponies. In contrast to the New Forest, where tree leaves formed a greater part of the diet in ponies than cattle (Putman *et al,* 1987), there was little difference in the proportion of foraging observations made by browsing, although in this study actual observations were greater in ponies. An exception to this was at Arne in 1999, where cattle browsed on felled *Betula pendula* and *Pinus sylvestris* in a woodland clearing.

Variation in habitat use was greater than that in behaviours. The 2000 cattle herd at Arne made greater use of grassy habitats such as acid grassland, saltings margin and saltings, and less use of dry heath habitats building dry heath and mature dry heath (although use of degenerate dry heath was greater – mainly due to use for overnight resting) than the 1999 herd. The herd in 2000 was lactating and therefore is likely to have had higher nutritive requirements. At Hartland, cattle made most use of acid grassland, followed by Exmoor ponies, and then New Forest ponies, who in turn made greatest use of restoration heath. Exmoor ponies appeared dominant in aggressive encounters, both with cattle and more particularly New Forest ponies, and were observed to drive them off acid grassland on a few occasions. Cattle made less use of valley mire or wet heath but greater use of building dry heath. Greater use of dry heath and less use of wet heath by cattle compared to ponies was also noted in the New Forest by Pratt *et al.* (1986) and Putman *et al.* (1987). A striking difference between Exmoor and New Forest ponies at Hartland was in use of roads – Exmoor ponies rarely made use of this habitat type, which was positively selected by New Forest ponies. This may have been a genuine breed difference, or perhaps due to learnt behaviour – the New Forest ponies had been brought from the New Forest, where ponies are frequently fed on the roadside by visitors. In general, New Forest ponies made greater use of different habitat types than both Exmoor ponies and cattle, which were more selective.

Individuals within each livestock group showed some differences in both behaviour and habitat use. Cattle within the same herd frequently used adjacent habitats to a different extent. Apparent differential habitat use by ponies was generally due to a lack of synchrony within the herd. Conversely, behavioural differences were more common in ponies, particularly at Stoborough (where the bonding between herd members appeared looser).

2.4.6 IMPLICATIONS OF DIFFERENTIAL RESOURCE SELECTION FOR CONSERVATION MANAGEMENT OF HEATHLAND SITES

The non-random use of habitat types by free-ranging livestock on the Dorset heaths clearly has consequences for conservation management of the heaths, as it will lead to a differential impact on vegetation. Results from this study and others suggest that pressure is likely to be greater on grass-rich habitats, with habitats such as acid grassland used most, followed by wet heath and valley mire and lastly dry heathland. However, not all dry heathland will be used in the same way, and this study suggests that recently managed areas and tracks are likely to be highly positively selected for foraging and resting, while degenerate dry heath may be positively selected for resting. The actual pressure will depend on the relative availability and location of each habitat type and the number and type of livestock used.

Differential impact may be increased by the use of different habitats for different behaviours. The main impact of foraging will be selective biomass removal, plus some trampling and poaching. Use of a habitat for resting will lead to localised vegetation damage through trampling (including lying) and limited biomass removal through associated foraging. It may also lead to disproportionately high nutrient input through dunging. Use of a habitat for moving will lead to trampling and poaching, and limited biomass removal through associated foraging. However, effects may be disproportionate according to habitat characteristics, for example, trampling may have a greater impact on wetter habitats and on older woody plants that are more brittle.

Actual impact on vegetation will depend on a number of site and livestock specific factors, so that each site needs to be assessed individually. However, if behaviour by free-ranging livestock is not resulting in the required livestock pressure on particular habitats or habitat patches, from the work presented here there are a number of ways in which it may be manipulated.

Firstly, seasonal grazing may be used to focus grazing pressure – for example, wet heath and valley mire are likely to be grazed in late summer. Seasonal use of dry heath may be more site-specific, but livestock are only likely to spread out over mature heath when browsing in autumn (this behaviour has also been observed on the Devon

Pebblebed heaths (L. Kerry pers. comm.). Degenerate heath may be used for resting in autumn. Therefore, the grazing season may be designed to either include or exclude these periods, therefore maximising or minimising the likelihood that livestock have an impact on these habitat types.

Secondly, the juxtaposition of habitat types within a site may also be used to change livestock behaviour. For example, at Godlingston, heathland habitat was only used when livestock crossed it to reach acid grassland on either side of it. Ponies at Stoborough used humid heath while crossing the site from one acid grassland patch to another. Opening or fencing off areas of acid grassland is likely to influence livestock movements. Similarly, the proximity of shelter may increase the extent to which habitats are used at night and in winter, and the provision of shelter (for example, by leaving small copses when carrying out heathland clearance) may encourage use of adjacent habitat. There was some indication in this study that copses on dry heath were also used for shading in summer (ponies were also observed 'shading' in the New Forest by Pratt *et al.,* 1986). The provision of water and supplementary feed can also be used to influence livestock behaviour (although supplementary feeding is often discouraged on heathland sites -see Chapter 6). On both Arne and Hartland, water supplies were limited to a single water trough or the many pools in wet heath and valley mire. Livestock visited one of these water sources at least once a day, and generally lingered in the vicinity, often also foraging while moving to and from it. Similarly, careful positioning of supplementary feed, particularly in the form of mineral licks or protein blocks (which will have less impact on the surrounding vegetation than bulky feed such as hay), can be used to attract livestock into an area they might not otherwise use.

A third area which has not been explored in the published literature is that of habit – the tendency for livestock to 'timetable' round a site, using the same areas at the same time of day, returning to particular habitats. However, graziers often refer to and use this tendency to facilitate livestock management. For example, when livestock new to a site are first released, they are often initially supplied with supplementary feed in a particular place, with the intent that they will then regard this particular area as 'home' and regularly return to it even when supplementary feeding is ceased (this facilitates inspection and the removal of livestock should the need arise). On the sites studied, this probably reinforced habitat selection patterns, as livestock were generally initially released into acid grassland fields, since these were most accessible by road and provided a better situation for handling livestock, and this will often be the case for reasons of livestock management. However, 'timetabling' may result in livestock regularly returning to habitat patches where foraging is not optimal, for example acid grassland in high summer when parched.

Finally, the results from this study suggest that habitat management can be used to attract livestock to a particular area. Managed dry heath, scrub and woodland were consistently more selected than their unmanaged counterparts. While physically managing an area of habitat may circumvent the need to manage it through livestock grazing, this technique can again be used to increase livestock use of surrounding areas. It is also possible that, given positive selection for tracks shown by some livestock, and the tendency for all livestock types to use existing tracks or small paths for moving, tracks or paths created through cutting vegetation could be used to manipulate livestock movements. For example, providing a path around an area where livestock are not required may discourage them from using it. Equally, providing a path to a particular area may increase use.

2.4.7 FURTHER WORK

Arising from the results presented here, there is a need for further work into two particular aspects of livestock behaviour on heathland sites if the potential conservation benefit of livestock grazing is to be maximised (the results from such work will also be applicable to other semi-natural systems):

(i) Adequately replicated research into differences between livestock type (particularly breed, physiology and background) in habitat selection, diet, intake rate and ranging behaviour;

(ii) Experimental manipulation of tracks, water, shelter and supplementary feed locations to establish to what extent these provide viable tools for influencing livestock resource selection.

2.4.8 CONCLUSIONS

- Livestock will exert differential pressure on habitats within heathland sites;
- This is largely determined by suitability for foraging, although resting will also influence selection;
- Foraging is highly selective – both cattle and ponies show a strong selection for grass species, although young *Calluna vulgaris* and *Ulex minor* are preferentially selected in particular circumstances;
- Habitat use by domestic livestock can be focussed on target habitats by seasonal grazing, the provision of water and supplementary feed, manipulation of the availability and juxtaposition of habitat types, and through habitat management;
- Habitat use will vary according to site characteristics and livestock type.

3. Livestock effects on wet heathland vegetation

3.1 Introduction

Grazing management of lowland heathlands for nature conservation is generally aimed at maintaining open dwarf shrub vegetation with a high diversity of heath plant species while helping to control scrub and other unwanted species (Bullock & Pakeman, 1997). However, little is known about the effects of grazing livestock on wet heath vegetation communities (see Chapter 1), and it is worth considering the mechanisms by which these aims could be met.

Herbivore presence will alter plant communities by affecting individual plants directly and by changing vegetation structure and aspects of the abiotic environment. This may lead to an increase in species diversity (e.g. Hill *et al.*, 1992; Humphrey & Patterson, 2000; Bullock *et al.*, 2001), a decrease (e.g. Welch, 1986; Bullock *et al.*, 2001), or have no effect (e.g. Gibson & Brown, 1991; Bullock *et al.*, 2001), depending on both the initial habitat characteristics and the grazing regime used.

Plant species richness has never been shown to be a simple, increasing function of productivity or nutrient supply (Tilman, 1993), but rather can be plotted as a bell-shaped curve along a gradient of standing crop, with the number of plant species a unimodal function of nutrient supply rate (Bakker, 1998). Low maximum standing crop and low species diversity may be correlated with high environmental stress and /or high disturbance where stress is defined as the sum of external constraints limiting the rate of dry matter production of the vegetation, such as dry, shaded, saline or nutrient poor conditions and disturbance the sum of mechanisms which limit plant biomass by causing partial or total destruction, such as grazing, burning, cutting (Grime, 1979). A large standing crop is likely to mean high productivity of some species, and this can limit species diversity by competitive exclusion.

Huston (1979) suggested that moderate levels of either disturbance or environmental stress are prerequisites for high species diversity. In mesotrophic environments high species diversity can be expected where the above ground biomass is removed frequently, and in more oligotrophic environments where the above-ground biomass is removed occasionally (Bakker, 1998). The latter is evidently applicable to heathland systems, and suggests that moderate disturbance by livestock presence may allow maximal biodiversity in wet heath and valley mire vegetation.

Four mechanisms have been identified by which livestock presence may cause changes in species composition (from Briske, 1996; Augustine & McNaughton, 1998; Olff & Ritchie, 1998; Bullock & Marriott, 2000; Sternberg *et al.*, 2000; Bullock *et al.*, 2001):

- **Selective feeding** by livestock - biochemical (e.g. secondary metabolites, high fibre content) and/or morphological deterrents (e.g. a rosette growth form) in plants lead to particular species undergoing lower biomass removal than others;
- **Plant tolerance to grazing and trampling** – the degree to which individual plant growth is affected by grazing is influenced by growth form (e.g. number and position of meristems), intrinsic growth rate and ability to regrow after tissue loss.
- **Gap colonisation** – grazing, trampling, dunging and urination create gaps in vegetation cover. The ability of species to colonise these gaps through clonal spread or seedling establishment will influence subsequent species composition;
- **Dispersal** - differential seed dispersal of plant species by livestock, including species from outside the community will affect species composition.

These four mechanisms may increase species diversity in the following ways (see Bullock, 1996; Olff & Ritchie, 1998):

- **Reduced competition** as tall, dominant species such as *Molinia caerulea* are grazed down, thus releasing resources (particularly light) which can be used by different species.

- **Increased large-scale structural heterogeneity** caused by spatially patchy grazing, allowing different vegetation types to co-exist;

- **Gap dynamics** – During succession (e.g. following fire) rapidly growing, short-lived early gap colonisers are replaced gradually by slower-growing dominants, for example the replacement of *Calluna vulgaris* by birch *Betula* sp. on upland heath (Hester *et al.*, 1991a). The constant creation of gaps in the vegetation cover and the presence of gaps of different ages allow a range of plant types to co-exist.

- **A diversity of gap types** is created by the different activities of livestock classified above, which results in gaps of different sizes and environments (light infiltration, nutrient concentrations, amount of litter, and soil disturbance, compaction and moisture) and may favour different species of coloniser.

However, livestock presence may also decrease species diversity, and this may occur when rarer species are preferentially grazed or when a few tolerant species persist. This is seen in many upland heaths, where matt grass *Nardus stricta* persists in heavily grazed swards when heather *Calluna vulgaris* cover is reduced through grazing (Armstrong & Milne, 1995).

It is clear that there is insufficient information available concerning the effects of livestock presence on wet heathland communities to establish how changes occurring through the mechanisms discussed above are expressed in wet heath communities. In addition, the methods used to gain such information as is available are frequently flawed (see Chapter 1). This chapter investigates the effects of livestock presence on the species composition and vegetation structure of wet heath and valley mire communities experimentally. It also considers the effect of livestock presence on environmental factors: substrate microtopography and compaction, and the cover of surface water. These factors have not previously been published for wet heathland communities. Data collected on the impact of cattle on regenerating pine *Pinus* sp scrub at Arne, is also presented in this chapter (although this was on humid and dry rather than wet heath). The results of this chapter are then used to provide the context of potential changes to germination and regeneration from the buried seed pool explored experimentally in subsequent chapters.

3.2 Methods

3.2.1. SITE DESCRIPTION

The experimental area, Coombe Bog, is located on the Arne Peninsula in Purbeck, south-east Dorset, National Grid reference SY970872 (Figure 3.1). It forms part of a larger heathland area extending over much of the peninsula, owned and managed by the RSPB. Its importance for wildlife conservation is reflected in its designations as a Site of Special Scientific Interest, Special Protection Area, and Ramsar site. The Peninsula is underlain by the Bagshot Beds of the Southampton Basin Tertiary deposits, on which freely draining humus-iron podsols, characteristic of dry heathland, have formed. In addition, clay lenses covering areas of up to several hectares and between 5-15m thick occur within the Bagshot beds. Where the clay is close to the surface, drainage is impeded and gleyed soils have developed. In valley areas such as Coombe Bog this has encouraged the establishment of peat and the development of wet heath and valley mire vegetation communities. Coombe Bog lies less than five metres above sea level.

The central area of Coombe Bog comprises 1.33 ha of valley mire over permanently waterlogged peat. This largely conforms to the National Vegetation Classification *Narthecium ossifragum – Sphagnum papillosum* (M21), subcommunity (a) *Rhynchospora alba- Sphagnum auriculatum* (Rodwell, 1991), characterised by carpets of *Sphagnum* species with scattered *Rhynchospora alba, Narthecium ossifragum, Erica tetralix* and sparse *Calluna vulgaris.* The valley mire is surrounded on three sides by 5.32ha of shallow peat which is only seasonally waterlogged. This supports wet heath vegetation best described by the *Erica tetralix – Sphagnum compactum* wet heath (M16) community. Both the typical subcommunity, characterised by *Molinia caerulea, Erica tetralix* and *Calluna vulgaris,* and the *Rhynchospora alba – Drosera intermedia* subcommunity are represented. The latter is found patchily where the cover of ericoids and *Molinia caerulea* is reduced, and there is more extensive cover of *Sphagnum compactum* and *S. tenellum with R. alba* and *D. intermedia* occurring around wetter hollows and runnels.

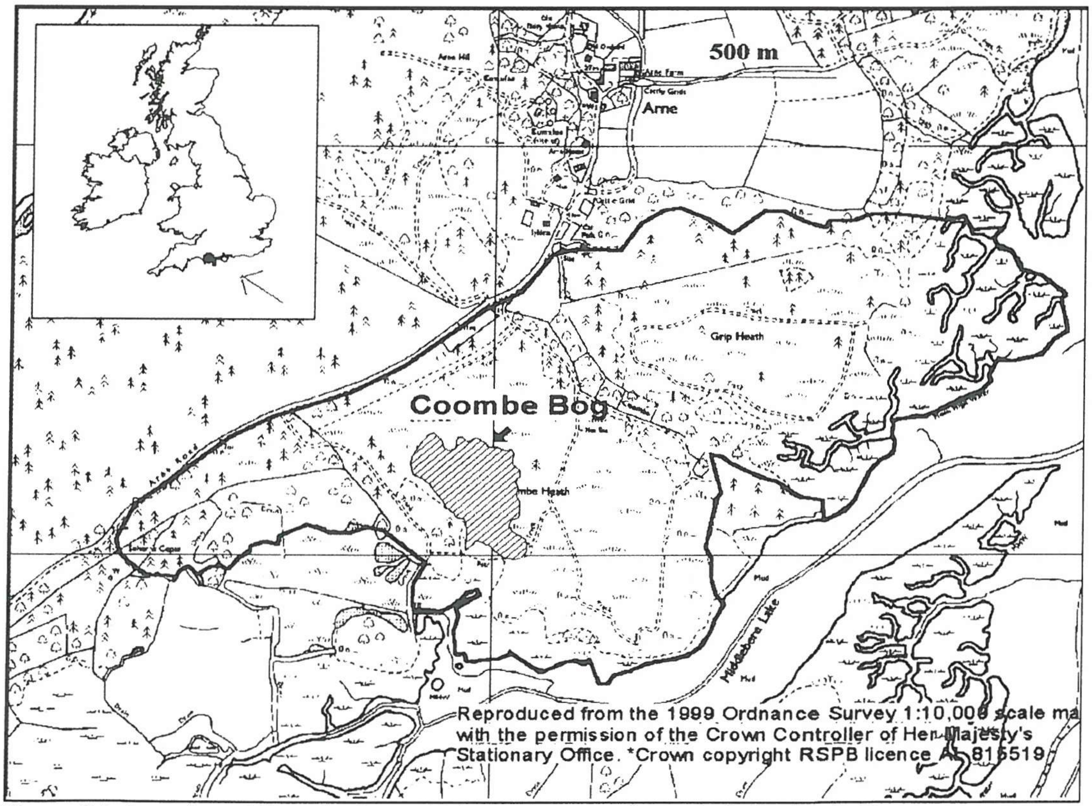


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Figure 3.1 Location of Coombe Bog within the Arne grazing unit, in Purbeck, Dorset.

Domestic livestock grazing ceased on the Arne peninsula heaths in the 1960s (B. Pickess, pers. comm.). In 1998 cattle were reintroduced as a conservation management tool to a 130ha grazing unit, which includes Coombe Bog (Figure 3.1). The herd comprised 12 mixed Hereford cross cattle with calves at foot and a Red Devon Bull, and were present on site from May to late September. The following year a similar herd of 12 cows (without calves) were grazed from May until November, 1999. Four of these individuals, together with two similar cows, plus six calves and an Aberdeen Angus bull, returned the following May and again remained on site until November, 2000.

The site is also grazed by a wild population of Sika deer *Sika nippon*. The population for the Arne peninsular as a whole increased during the study period and was estimated to have doubled from 200 in 1998 to about 400 individuals in 2001 (N. Gartshore, pers. com). Observation suggests that Coombe Bog is regularly used by groups of between 2 and 50 individuals. Deer grazing pressure is difficult to estimate, as the deer are not confined to the grazing unit. A daytime count carried out in May 2001 suggested that 127 individuals were present within the grazing unit simultaneously, a density of 0.84 deer ha^{-1}. These deer are considered to use an additional area of 170ha outside the grazing unit, in effect halving grazing pressure to 0.42 deer ha^{-1}. Ten counts carried out within the Coombe basin at dusk in autumn 1999 found a median number of 7 deer within 20ha, providing a grazing pressure of 0.35 deer ha^{-1}, roughly equivalent to that calculated above.

3.2.2 EXPERIMENTAL DESIGN

Ten randomly located replicates of three grazing treatments were set up in a randomised block design within Coombe Bog in April, 1998. Plots measured 10m x 10m and were either unfenced, cattle fenced (with three strands of barbed wire), or cattle and deer fenced (1.8 m high with a 15 cm mesh), allowing three grazing treatments to be imposed: (i) ungrazed, (ii) deer-grazed, and (iii) cattle plus deer-grazed in the marked, unfenced plots (see Table 3.1). Plots were adjacent to one another. Five blocks were located on wet heath vegetation and five on the adjacent valley mire vegetation. Average cattle grazing was at a stocking rate of 0.04 LU $^{-1}$ ha^{-1} yr^{-1}. Deer grazing was estimated at approximately 0.04 LU $^{-1}$ ha^{-1} yr^{-1} (assuming a livestock unit equivalence of 0.1). Overall, stocking rate was therefore 0.08 LU $^{-1}$ ha^{-1} yr^{-1}.

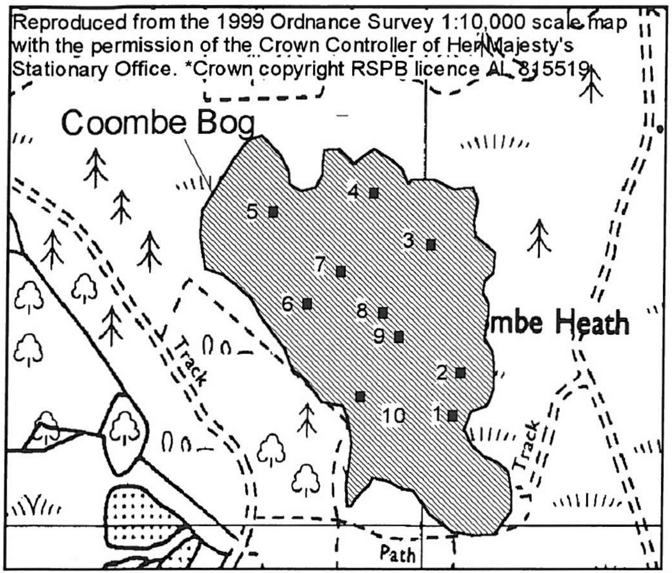
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Figure 3.2 Experimental block (exclosure) locations within Coombe Bog.

Plate 3.1 Exclosure 4, Coombe Bog, Arne, in 2000.

Table 3.1 Grazing treatments carried out on Coombe Bog from 1998 to 2000. Each treatment was replicated 5 times on the wet heath and 5 times on valley mire.

Treatment	Abbreviation
Ungrazed	UG
Deer-grazed	DG
Cattle and deer-grazed	CDG
Habitats	
Wet heath	WH
Valley mire	VM

3.2.3 ASSESSMENTS

Nomenclature follows Stace (1995) for vascular plants, Purvis *et al.* (1992) for lichens and Watson (1981) for bryophytes. The percentage cover of species present was recorded from four 2m x 2m randomly located fixed quadrats within each plot in September 1998, 1999, and 2000. Numbers of rooted plants of *Drosera intermedia, D. rotundifolia* and *Pinus sylvestris* were counted as these species had particularly low percentage cover. Within valley mire samples, damaged *Sphagnum* material was locally frequent – this could not be identified and was recorded as *Sphagnum* spp. In addition to the fixed quadrats, in 2000, species frequency (presence within quadrats) was recorded within 10 randomly located sets of nested quadrats (5cm x 5cm, 25cm x 25cm, and 100cm x 100cm) within each treatment plot. This method allows the most appropriate quadrat size to be chosen to identify differences between treatments for species occurring at different densities, and was also intended to reduce the subjectivity of the assessment.

Vegetation height was recorded each year from 20 random points within each plot using a drop disc (as discussed in Stewart *et al.,* 2001) (polystyrene disc diameter 30cm, weight 120g, dropped from one metre above ground level). Sward structure at a given point was measured by noting all the heights at which vegetation touched a pin (diameter 3mm) dropped vertically. The height of the tallest vegetation 'hit' was also recorded. Sward structure was measured 20 times per treatment plot at randomly located points in 2000.

Soil compaction and microtopography were measured in 2000. Compaction was measured using the BJPS method (Jones & Reynolds, 1996) - a steel rod (diameter 1cm, length 100cm, weight 300g) sharpened to a point and dropped vertically from a height of one metre through a piece of plastic tubing (diameter 2cm, height 100cm) ensuring it remained vertical while falling. The depth to which the rod penetrated the substrate was recorded. This was repeated 20 times in each treatment plot.

Microtopography was recorded by dropping a pin (diameter 2mm, length 300mm) vertically through a horizontal bar every 1cm over a distance of 25cm. The pins were allowed to rest on the substrate surface and the relative height of each pin top marked on graph paper fixed vertically above the horizontal bar. This allowed the difference in height between consecutive pin tops and the lowest pin top to be measured, giving a measure of relative microtopographical variation. The effect of microtopographical variation on water level was measured directly by noting the proportion of pins that fell on water. These measurements were repeated at ten random locations for each treatment plot. The bar contained an in-built level and was positioned at 90° to the line of any overall gradient observed, ensuring microtopography rather than overall gradient was measured.

There was very limited scrub within the Coombe Bog experimental area. Therefore, the impact of cattle on scrub was assessed from an adjacent area, although this comprised humid and dry heath rather than wet heath. Regenerating saplings (under 1.5 m) were examined for browse damage (evidence of recent browse damage, or multiple shoots growing following previous browse damage), and compared with trees in a similarly aged stand outside the grazing unit (both areas were used by deer – deer pressure was estimated to be 0.35 deer ha^{-1} within cattle unit and 0.28 deer ha^{-1} outside using the methods described in section 3.2.1). Age was estimated by counting the number of nodes on the main trunk. The first sapling encountered every five metres along transect line through the stands were examined. This was carried out in autumn, 1998 after the first seasons' grazing. After two further grazing seasons, the stands were re-examined, and the sample size increased from 100 to 300 for each set.

3.2.3 DATA ANALYSIS

Analysis of changes in species cover (%).

Species were grouped as ericoids (*Calluna vulgaris, Erica tetralix, Erica ciliaris*), forbs (*Drosera rotundifolia, Drosera intermedia, Polygala serpyllifolia, Potentilla erecta*), graminoids (*Molinia caerulea, Eriophorum angustifolium, Trichophorum cespitosum, Rhynchospora alba, Juncus acutiflorus)* lichens (*Cladonia ciliata, C. portentosa, C. uncialis, C. arbuscula, C. subulata, Coelecaulon aculeate, Pycnothelia papillaria*) and Sphagnum (*Sphagnum compactum, S. tenellum, S. papillosum, S. pulchrum, S. cuspidatum, S. rubellum, S. auriculatum*) other bryophytes (*Campylopus brevipilus, Leucobryum glaucum, Hypnum jutlandicum*) and tree seedlings (*Pinus sylvestris, Quercus robur, Betula pubescens, Sorbus aucuparia*). All percentage data were arsine square root transformed before analysis.

A comparison between the observed plant frequencies in the treatment plots and those recorded in the diagnostic communities and subcommunities described in the NVC was conducted using MATCH (1999). Comparison of the coefficients of similarity for observed and diagnostic data between treatment plots allowed an examination of the differences in NVC profile.

Data from quadrats taken in 2000 were averaged for each treatment plot, grouped, and compared using a general linear model (GLM) (MINITAB, 1998). Response to grazing treatment, vegetation type, and blocks (nested within vegetation type), plus the interaction between grazing and vegetation, were tested. Each species response was then analysed individually. Ten species with fewer than 6 occurrences within the 120 quadrats were omitted from the analyses. Since 30 analyses were done, the null hypothesis that cover was the same under each grazing treatment was rejected when $P < 0.01$ to ensure that the probability of a difference occurring by chance was less than one in 30. The number of occurrences of each species within each of the three nested quadrat sizes was summed for each plot to provide a measure of frequency, and then analysed as above.

Differences in percentage cover over the three years were analysed by repeated measures ANOVA using PROC GLM in SAS (1990). Results from the univariate test were used (with the Greenhouse Geisser adjustment) due to the relatively small sample sizes (Maxwell & Delaney, 1990). Results from the multivariate analyses run as part of the SAS repeated measure procedure gave qualitatively the same results. Again a P value of <0.01 was used.

Ordinations

Differences in the vegetation composition of the experimental plots in 2000 were analysed by principal component analysis (PCA). PCA is a direct ordination technique for linear data, in which a theoretical variable is constructed to account for the variation in the species data by minimizing the total residual sum of squares after fitting straight lines to the data. The differences between sample scores reflect the differences in vegetation composition. Samples may be plotted in an ordination diagram with species represented by arrows. The direction of each arrow shows the direction in which the abundance of that species increases most, the length of the arrow shows the rate of change in that direction (Jongman *et al.*, 1987).

To directly relate vegetation composition to grazing treatment, redundancy analysis (RDA) was used. This selects the linear combination of environmental variables (i.e. grazing treatments) that best accounts for the variation in the species data i.e. the variable that gives the smallest total residual sum of squares for the species data. Species may be plotted in an ordination diagram together with the environmental variables, showing how each species is related to each grazing treatment. The statistical validity of the ordination was tested using an unrestricted Monte Carlo permutation test (ter Braak & Šmilauer, 1998).

To explore whether excluding the variation due to initial differences in the vegetation increases the amount of variation accounted for by the grazing treatments, the proportional difference in species cover in the fixed quadrats between 2000 and 1998 was investigated by RDA . Values of zero were replaced by 0.1 (the lowest % cover value recorded). Values of zero which were paired with values of 0.1 in the original data were replaced by 0.01.

For both direct and indirect ordinations percentage cover vegetation data were transformed by a log(y+1) transformation. Species data were further transformed by dividing by the standard deviation (after extraction of the axes) to reduce the effect of species with large variances on the ordination diagram. Ten rare species (occurring in six or fewer quadrats) were excluded from the analyses to prevent undue influence in the ordination. PCA and RDA were carried out using the programme CANOCO version 4 for windows (ter Braak & Šmilauer, 1998).

Sward height and structure

Sward height measurements were averaged for each treatment plot and year and compared within years using a GLM with the same factors as above. Changes in sward height over the three years were analysed using the repeated measures procedure described above. Sward structure was analysed by GLM as above, and the relationship between sward structure and height was explored using regression.

Environmental variables

Substrate compaction data from wet heath and valley mire were analysed separately by two-way ANOVA. To examine microtopography variation the variance of the absolute differences between adjacent points were calculated for each treatment plots. The proportion of points lying below water level was arcsine square root transformed. Both were then analysed by GLM as above.

Scrub

The difference in height, the number of shoots browsed and age between pine saplings within and without the grazing unit were compared using a Kruskall-Wallis test.

3.4 Results

A total of 35 species was found, 29 in wet heath samples (of which seven were restricted to this vegetation type) and 28 in valley mire samples (of which six were restricted to this vegetation type). Five species were noted within Coombe Bog that did not occur in quadrats: these were *Potamegaton polygonifolius, Juncus squarrosus*, Carex *panicea*, *Sphagnum subnitens*, *Cladonia squamosa, Cladonia floerkeana* and *Cladopodiella fluitans.* All species occurring in more than six quadrats (5% of all quadrats) were represented in each of the three grazing treatments i.e. species richness was little affected by grazing treatment after three years. However, of the remaining rare species *Polygala serpyllifolia*, *Cladonia arbuscula*, *C. subulata* and *Coelecaulon aculeata* occurred only in grazed treatment plots. Although changes in the percentage cover of species occurred, these were too subtle to be picked up by comparison of diagnostic profiles in the NVC. Treatments within each vegetation type fall within the same subcommunities - MATCH coefficients are similar and relatively high, showing a good match to NVC types (Table 3.2).

Table 3.2 Fit with NVC subcommunities for vegetation after 3 years grazing treatments at Coombe Bog. Species numbers are compared to the test community by MATCH.

Grazing treatment	**NVC communitiy**	**MATCH coefficient**	**Mean spp. no.**	**Min. spp. no.**	**Max spp. no**
test	*M16a*		*13*	*8*	*24*
Wet heath UG	M16a	58.6	11	6	15
Wet heath DG	M16a	65.2	10.4	4	13
Wet heath CDG	M16a	61.8	11.5	8	15
test	*M21(typical)*		*14*	*8*	*24*
Valley mire UG	M21	68.3	8.9	5	13
Valley mire DG	M21	69.4	9.1	5	13
Valley mire CDG	M21	67.2	8.6	5	15

3.4.1 SPECIES COVER AND FREQUENCY

Figure 3.3 shows the percentage cover of species groups in 2000 following three years of different grazing treatments). Significant differences (see Table 3.3) were found between grazing treatments for graminoids, in which CDG samples had lower cover than UG samples (Tukey simultaneous comparison, CDG – UG , T = 2.580, P<0.05) and for ericoids, in which CDG samples had a lower cover than DG (but not UG) samples (Tukey simultaneous comparison, CDG – DG , T = 2.785, P<0.05).

Comparison of individual species (including individual counts), bare ground and litter between grazing treatments found significant differences only for bare ground. Eight of the 22 species showed significant differences between treatment blocks, and 14 between vegetation type (F values are presented in Appendix VI). Bare ground cover was significantly greater in CDG samples than both DG and UG samples (Tukey simultaneous comparison, CDG – UG , T = -7.044, P<0.0001, CDG – DG, T = -6.784, P<0.0001) (Figure 3.4).

Table 3.3 Percentage cover and F-ratios generated by nested ANOVA for species groups and bare ground following three years of grazing treatments at Coombe Bog. CDG – cattle and deer grazed, DG – deer grazed, UG – ungrazed, *** P <0.001, ** P<0.01, NS – not significant (P>0.01).

	CDG		**DG**		**UG**		**VEG**	**BLOCK**	**TREAT**	**TREAT* VEG**
Species groups	mean	st.dev	mean	st.dev	mean	st.dev	F-ratio	F-ratio	F-ratio	F-ratio
Graminoids	27.24	13.22	31.74	18.26	40.21	22.29	39.32***	1.85^{NS}	4.29**	1.38^{NS}
Ericoids	30.50	20.35	38.56	21.53	35.25	20.34	57.21***	2.53^{NS}	4.09**	0.66^{NS}
Lichens	4.36	4.57	4.99	6.37	5.48	7.08	35.09***	2.29^{NS}	0.18^{NS}	0.13^{NS}
Sphagnum	18.14	16.46	20.32	12.63	18.40	13.54	8.49^{NS}	1.24^{NS}	0.16^{NS}	0.34^{NS}
Forbs	0.18	0.20	0.46	0.56	0.25	0.27	0.01^{NS}	1.16^{NS}	0.84^{NS}	1.53^{NS}
Bare ground	19.91	2.57	1.19	0.75	1.48	0.89	1.48^{NS}	0.99^{NS}	31.91***	3.69^{NS}

The relatively high number of species showing significant differences between vegetation type and block could be due to variation in the initial composition of the fixed quadrats. Therefore, the change in species cover in the same fixed quadrats over time was considered (Table 3.4a and b). On wet heath, there was a significantly greater increase in bare ground in CDG plots than in DG or UG plots over three years. This pattern was repeated in the valley mire where, in addition, graminoids decreased significantly more in CDG plots than in DG or UG plots (Figure 3.5, Table 3.5).

Table 3.4a Percentage cover and F-values generated by repeated measures ANOVA for wet heath species at Coombe Bog, Arne CDG – cattle and deer grazed, DG – deer grazed, UG – ungrazed, *** P <0.001, ** P<0.01, NS – not significant (P>0.01).

	2000						**1999**						**1998**								
	UG		**DG**		**CDG**		**UG**		**DG**		**CDG**		**UG**		**DG**		**CDG**		**YEAR**	**YEAR* TREAT**	**YEAR * BLOCK**
	mean	st.dev	mean	st.dev	mean	st.dev	mean	st.dev	mean	st.dev	mean	st.dev	mean	st.dev	mean	st.dev	mean	st.dev			
Bare ground	**3.78**	**2.18**	**2.93**	**1.99**	**16.20**	**11.89**	**4.38**	**2.64**	**3.63**	**2.38**	**12.95**	**11.50**	**5.55**	**3.09**	**4.73**	**2.17**	**8.98**	**8.44**	**0.35^{NS}**	**22.44****	**0.71^{NS}**
Litter	2.25	1.70	1.95	0.42	2.80	1.56	1.95	0.54	1.78	0.45	3.20	1.05	1.90	0.82	1.78	0.38	2.90	1.26	0.17^{NS}	0.37^{NS}	0.49^{NS}
Betula pubescens	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.20	0.45	0.00	0.00	-	-	-
Calluna vulgaris	15.65	1.97	15.75	4.97	16.10	6.22	15.35	5.13	17.10	4.49	15.45	6.52	16.20	4.17	16.90	6.79	16.35	5.64	0.46^{NS}	0.66^{NS}	0.67^{NS}
Campylopus brevipilus	0.24	0.33	0.01	0.01	0.07	0.11	0.25	0.32	0.01	0.01	0.18	0.21	0.19	0.34	0.06	0.11	0.07	0.13	1.05^{NS}	1.31^{NS}	0.46^{NS}
Cladonia arbuscular	0.00	0.00	0.00	0.00	0.08	0.11	0.00	0.00	0.00	0.00	0.08	0.11	0.00	0.00	0.00	0.00	0.08	0.11	-	-	-
Cladonia ciliata	0.93	0.48	0.61	0.80	1.18	1.33	0.44	0.38	1.03	0.89	1.38	1.63	1.14	0.88	0.73	1.01	2.38	2.00	0.63^{NS}	1.31^{NS}	0.08^{NS}
Cladonia portentosa	9.40	6.84	8.23	6.36	6.85	2.22	6.20	5.97	7.88	5.23	7.10	2.00	6.30	5.61	5.65	4.97	5.85	3.69	4.70^{NS}	0.97^{NS}	0.76^{NS}
Cladonia subulata	0.00	0.00	0.00	0.00	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	-	-	-
Cladonia uncialis	0.09	0.09	0.02	0.02	0.13	0.11	0.08	0.10	0.04	0.05	0.09	0.06	0.18	0.21	0.13	0.23	0.14	0.09	1.58^{NS}	1.49^{NS}	0.40^{NS}
Coelecaulon aculeata	0.02	0.02	0.05	0.11	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	-	-	-
Drosera intermedia	0.21	0.31	0.17	0.23	0.06	0.03	0.21	0.32	0.12	0.17	0.07	0.04	0.61	1.07	0.33	0.44	0.07	0.04	12.09**	3.19^{NS}	3.69^{NS}
Drosera rotundifolia	0.01	0.02	0.04	0.05	0.20	0.21	0.17	0.33	0.07	0.11	0.13	0.12	0.63	1.06	0.32	0.44	0.08	0.08	6.76^{NS}	4.36^{NS}	1.65^{NS}
Erica ciliaris	2.00	2.73	4.10	6.91	1.75	2.17	3.55	5.78	2.10	2.44	1.80	2.27	2.85	4.48	3.55	5.19	1.25	1.75	0.05^{NS}	1.33^{NS}	0.27^{NS}
Erica tetralix	35.15	8.64	37.25	12.71	30.90	10.91	34.35	8.83	39.25	12.77	29.60	10.52	33.45	9.03	37.55	13.17	29.75	11.78	0.76^{NS}	0.87^{NS}	0.17^{NS}
Eriophorum angustifolium	1.81	1.56	1.76	1.68	0.99	1.38	1.48	1.27	1.21	1.17	0.88	1.03	1.21	1.22	0.96	0.65	0.71	0.96	10.09**	0.93^{NS}	1.54^{NS}
Juncus articulatus	0.20	0.45	0.28	0.26	0.30	0.31	0.28	0.41	0.15	0.22	0.09	0.16	0.16	0.35	0.05	0.11	0.20	0.29	1.37^{NS}	1.03^{NS}	0.35^{NS}
Molinia caerulea	12.90	6.74	10.15	6.93	13.80	9.55	13.35	7.58	10.10	7.04	18.95	15.26	13.40	6.11	11.45	8.35	21.20	16.04	11.89^{NS}	5.48^{NS}	1.76^{NS}
Narthecium ossifragum	0.00	0.00	0.15	0.34	0.23	0.44	0.20	0.27	0.00	0.00	0.21	0.33	0.00	0.00	0.40	0.89	0.30	0.45	0.20^{NS}	1.76^{NS}	0.43^{NS}

Figure 3.4a continued

	2000						1999						1998						YEAR	YEAR* TREAT	YEAR * BLOCK
	UG		DG		CDG		UG		DG		CDG		UG		DG		CDG				
	mean	st.dev	mean	st.dev	mean	st.dev	mean	st.dev	mean	st.dev	mean	st.dev	mean	st.dev	mean	st.dev	mean	st.dev			
Pinus sylvestris	1.15	1.62	0.88	1.12	0.45	0.75	0.97	1.71	0.69	0.89	0.44	0.76	1.03	1.81	0.64	0.92	0.46	0.89	0.77^{NS}	0.41^{NS}	0.39^{NS}
Polygala serpyllifolia	0.00	0.00	0.06	0.13	0.02	0.03	0.01	0.01	0.00	0.00	0.03	0.04	0.00	0.00	0.00	0.00	0.00	0.00	2.08^{NS}	1.18^{NS}	1.27^{NS}
Pycnothelia papillaria	0.01	0.01	0.01	0.01	0.04	0.05	0.03	0.07	0.00	0.00	0.03	0.05	0.00	0.00	0.00	0.00	0.13	0.21	0.16^{NS}	0.33^{NS}	0.91^{NS}
Quercus robur	0.01	0.01	0.00	0.00	0.00	0.00	0.01	0.01	0.00	0.00	0.00	0.00	0.01	0.01	0.00	0.00	0.00	0.00	-	-	-
Rhynchospora alba	0.91	2.01	1.21	2.06	0.06	0.12	0.85	1.76	0.21	0.46	0.00	0.00	0.40	0.89	0.23	0.31	0.00	0.00	1.87^{NS}	0.88^{NS}	0.94^{NS}
Sorbus aucuparia	0.01	0.01	0.00	0.00	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	-	-	-
Sphagnum compactum	0.60	0.76	0.65	0.99	0.55	0.58	1.45	1.77	0.25	0.43	0.73	0.60	0.30	0.67	0.20	0.45	0.00	0.00	14.64***	2.51^{NS}	2.77^{NS}
Sphagnum tenellum	12.00	7.38	12.90	8.67	6.80	3.04	11.35	6.55	13.25	8.88	6.70	4.41	11.73	7.13	13.80	8.73	10.30	10.96	1.78^{NS}	0.60^{NS}	0.84^{NS}
Trichophorum cespitosum	4.98	5.80	4.35	2.24	2.40	1.46	5.13	4.71	3.65	2.18	2.41	1.58	4.61	4.63	3.93	1.81	1.25	0.77	1.96^{NS}	1.49^{NS}	0.75^{NS}
ericoids	52.80	8.41	57.10	10.88	48.75	8.03	53.25	7.70	58.45	11.14	46.85	8.06	52.50	7.97	58.00	10.38	47.35	9.43	0.07^{NS}	0.69^{NS}	0.86^{NS}
graminoids	20.79	6.50	17.89	9.43	17.78	10.96	21.29	6.21	15.32	6.26	22.53	16.88	19.77	4.80	17.01	10.12	23.66	17.20	0.76^{NS}	3.29^{NS}	1.00^{NS}
forbs	0.22	0.33	0.27	0.26	0.27	0.25	0.38	0.64	0.19	0.27	0.23	0.17	1.24	2.13	0.65	0.65	0.15	0.09	7.29**	4.31^{NS}	2.24^{NS}
lichens	10.44	7.14	8.90	6.89	8.28	2.78	6.75	5.87	8.94	5.80	8.67	2.84	7.62	6.02	6.50	5.62	8.57	2.52	1.65^{NS}	1.71^{NS}	0.22^{NS}
Sphagnum	12.75	7.91	13.65	8.77	7.35	2.95	12.85	7.37	13.98	8.82	7.48	4.14	12.31	7.28	14.83	8.84	11.10	11.04	1.63^{NS}	0.81^{NS}	0.96^{NS}
other bryophytes	0.24	0.33	0.01	0.01	0.07	0.11	0.25	0.32	0.01	0.01	0.18	0.21	0.19	0.34	0.06	0.11	0.07	0.13	1.05^{NS}	1.31^{NS}	0.46^{NS}
tree seedlings	1.16	1.63	0.88	1.12	0.45	0.75	0.98	1.71	0.69	0.89	0.44	0.76	1.04	1.81	0.84	1.36	0.46	0.89	0.66^{NS}	0.34^{NS}	0.56^{NS}

Table 3.4b cont'd

	2000						**1999**						**1998**						**F VALUES**		
	UG		**DG**		**CDG**		**UG**		**DG**		**CDG**		**UG**		**DG**		**CDG**		**YEAR**	**YEAR* TREAT**	**YEAR * BLOCK**
	mean	st.dev	mean	st.dev	mean	st.dev	mean	st.dev	mean	st.dev	mean	st.dev	mean	st.dev	mean	st.dev	mean	st.dev			
Quercus robur	0.01	0.02	0.01	0.03	0.00	0.00	0.01	0.02	0.01	0.03	0.00	0.00	0.01	0.02	0.02	0.03	0.00	0.00	-	-	-
Rhynchospora alba	8.65	11.53	11.25	6.87	6.55	7.29	6.85	9.10	9.25	6.18	5.41	7.43	5.90	8.48	8.05	6.15	6.50	9.44	4.70^{NS}	0.73^{NS}	1.14^{NS}
Sphagnum spp.	2.85	5.45	5.70	4.88	4.65	6.16	4.25	5.37	5.45	5.44	4.40	4.84	2.95	6.05	5.90	5.80	1.15	2.18	1.42^{NS}	0.89^{NS}	0.95^{NS}
Sphagnum auriculatum	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.11	-	-	-
Sphagnum compactum	0.05	0.11	0.00	0.00	2.80	6.12	0.00	0.00	0.00	0.00	0.20	0.45	0.00	0.00	0.00	0.00	3.80	7.56	0.89^{NS}	0.82^{NS}	0.92^{NS}
Sphagnum cuspidatum	10.55	11.78	6.08	6.84	14.58	19.08	10.80	12.87	8.28	11.62	16.30	20.48	11.60	12.95	8.35	12.49	16.15	21.28	0.05^{NS}	0.02^{NS}	1.20^{NS}
Sphagnum papillosum	0.15	0.22	5.10	9.57	1.56	3.34	0.15	0.22	5.30	10.26	1.75	3.77	0.10	0.22	5.10	9.81	1.30	2.37	0.05^{NS}	0.35^{NS}	1.87^{NS}
Sphagnum pulchrum	8.85	16.28	8.45	18.07	3.85	6.47	8.50	15.22	8.30	18.14	4.55	5.73	11.50	21.20	8.80	19.40	7.50	9.95	2.05^{NS}	0.77^{NS}	1.23^{NS}
Sphagnum rubellum	0.10	0.22	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	-	-	-
Sphagnum tenellum	1.50	1.73	1.65	1.68	1.50	2.95	1.50	1.73	0.65	0.80	2.95	3.77	0.00	0.00	0.00	0.00	3.25	4.47	3.57^{NS}	2.39^{NS}	4.10**
Trichophorum cespitosum	5.01	6.22	0.30	0.41	7.15	10.39	4.41	5.63	0.23	0.34	6.40	11.44	4.20	5.74	0.25	0.35	4.85	9.14	0.58^{NS}	0.83^{NS}	0.38^{NS}
ericoids	17.70	9.50	20.03	8.11	12.25	5.92	17.31	8.29	19.53	7.59	15.36	10.93	18.49	8.51	19.85	7.98	18.50	12.40	0.57^{NS}	0.35^{NS}	0.87^{NS}
graminoids	**59.64**	**11.51**	**45.60**	**13.46**	**36.71**	**7.02**	**52.42**	**10.85**	**41.27**	**14.48**	**34.15**	**6.82**	**46.98**	**11.09**	**40.77**	**13.77**	**38.21**	**4.76**	**11.78***	**6.12**	**0.85^{NS}**
forbs	0.28	0.25	0.66	0.73	0.09	0.07	0.49	0.62	0.82	0.93	0.06	0.05	0.58	0.66	0.83	0.70	0.66	1.31	1.97^{NS}	0.39^{NS}	0.79^{NS}
lichens	0.53	0.77	1.07	2.35	0.44	0.87	0.38	0.49	1.00	2.24	0.52	0.97	0.33	0.45	1.05	1.84	0.37	0.68	0.03^{NS}	0.32^{NS}	1.20^{NS}
Sphagnum	24.05	16.45	26.98	13.07	28.93	17.60	25.20	17.98	27.98	15.65	30.15	17.57	26.15	18.97	28.15	16.39	33.20	20.71	0.17^{NS}	0.05^{NS}	1.02^{NS}
tree seedlings	0.09	0.17	0.40	0.83	0.17	0.24	0.08	0.17	0.24	0.49	0.11	0.23	0.02	0.05	0.25	0.49	0.06	0.12	1.01^{NS}	0.24^{NS}	1.24^{NS}

Table 3.4b Percentage cover and F-values generated by repeated measures ANOVA for valley mire species at Coombe Bog, Arne. CDG – cattle and deer grazed, DG – deer grazed, UG – ungrazed. *** P <0.001, ** P<0.01, NS – not significant (P>0.01).

	2000						1999						1998						F VALUES		
	UG		DG		CDG		UG		DG		CDG		UG		DG		CDG		YEAR	YEAR* TREAT	YEAR * BLOCK
	mean	st.dev	mean	st.dev	mean	st.dev	mean	st.dev	mean	st.dev	mean	st.dev	mean	st.dev	mean	st.dev	mean	st.dev			
Bare ground	**0.73**	**0.99**	**0.75**	**1.12**	**24.15**	**12.42**	**1.15**	**1.15**	**1.75**	**2.44**	**18.65**	**9.05**	**1.95**	**2.73**	**2.50**	**3.85**	**11.25**	**7.30**	**0.83**^{NS}	**5.65****	**2.96**^{NS}
Litter	3.70	3.19	3.45	2.27	1.10	0.58	4.15	3.38	4.05	2.13	2.00	1.21	4.45	3.79	4.30	1.92	3.05	2.22	15.83***	2.37^{NS}	1.36^{NS}
Betula pubescens	0.01	0.03	0.00	0.00	0.00	0.00	0.01	0.03	0.00	0.00	0.00	0.00	0.01	0.03	0.00	0.00	0.01	0.01	-	-	-
Calluna vulgaris	4.60	4.63	3.53	3.19	3.65	4.91	4.86	4.35	3.73	3.22	4.65	5.81	5.59	4.46	3.55	3.01	5.73	7.46	0.50^{NS}	0.11^{NS}	1.09^{NS}
Cladonia arbuscular	0.00	0.00	0.00	0.00	0.01	0.03	0.00	0.00	0.00	0.00	0.01	0.03	0.00	0.00	0.00	0.00	0.01	0.03	-	-	-
Cladonia ciliata	0.00	0.00	0.00	0.00	0.05	0.11	0.00	0.00	0.00	0.00	0.05	0.11	0.00	0.00	0.00	0.00	0.05	0.11	-	-	-
Cladonia portentosa	0.53	0.77	1.07	2.35	0.38	0.77	0.38	0.49	1.00	2.24	0.46	0.87	0.33	0.45	1.05	1.84	0.31	0.54	0.03^{NS}	0.32^{NS}	1.19^{NS}
Drosera intermedia	0.02	0.03	0.10	0.14	0.02	0.03	0.02	0.03	0.10	0.14	0.01	0.01	0.02	0.03	0.11	0.13	0.01	0.02	0.50^{NS}	0.11^{NS}	1.09^{NS}
Drosera rotundifolia	0.26	0.27	0.56	0.72	0.06	0.05	0.47	0.63	0.72	0.89	0.05	0.04	0.56	0.68	0.73	0.70	0.65	1.32	2.03^{NS}	0.41^{NS}	0.73^{NS}
Erica ciliaris	3.15	5.04	5.70	7.85	2.15	4.81	2.70	3.61	5.36	7.27	4.40	9.84	2.65	3.50	4.70	5.85	4.85	10.57	0.79^{NS}	0.96^{NS}	1.29^{NS}
Erica tetralix	9.95	3.80	10.80	6.89	6.45	3.66	9.75	3.46	10.45	7.26	6.31	4.12	10.25	4.00	11.60	8.22	7.93	3.39	1.00^{NS}	0.21^{NS}	0.73^{NS}
Eriophorum angustifolium	22.55	14.57	13.80	7.78	11.65	4.22	16.55	9.35	11.65	6.21	10.90	4.57	11.93	8.15	10.30	5.22	13.70	5.16	4.23^{NS}	3.43^{NS}	1.37^{NS}
Hypnum jutlandicum	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.18	0.39	0.00	0.00	0.00	0.00	-	-	-
Juncus articulatus	0.33	0.60	0.30	0.41	0.81	1.14	0.25	0.56	0.12	0.12	0.66	1.24	0.30	0.67	0.09	0.09	0.63	1.19	1.19^{NS}	0.02^{NS}	0.18^{NS}
Leucobrium glaucum	0.35	0.78	0.00	0.00	0.00	0.00	0.35	0.78	0.00	0.00	0.00	0.00	0.50	1.12	0.00	0.00	0.00	0.00	-	-	-
Molinia caerulea	21.15	28.38	18.70	24.95	7.38	4.54	21.70	28.03	18.60	24.95	7.63	4.48	21.90	26.79	20.80	23.43	9.43	6.03	1.21^{NS}	0.07^{NS}	0.21^{NS}
Myrica gale	1.60	2.71	4.95	5.30	1.45	3.24	1.30	1.99	4.91	5.22	1.50	3.35	2.00	3.01	4.96	5.21	1.75	1.96	2.00^{NS}	0.85^{NS}	0.85^{NS}
Narthecium ossifragum	1.96	2.58	1.25	1.29	3.18	5.10	2.66	3.34	1.43	1.72	3.16	5.24	2.76	3.67	1.28	1.64	3.11	5.16	0.28^{NS}	0.13^{NS}	1.44^{NS}

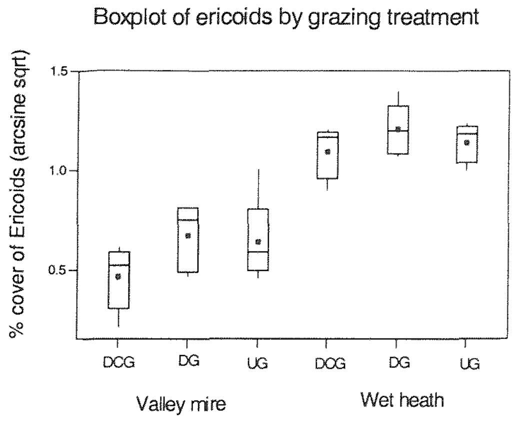


Figure3.5a

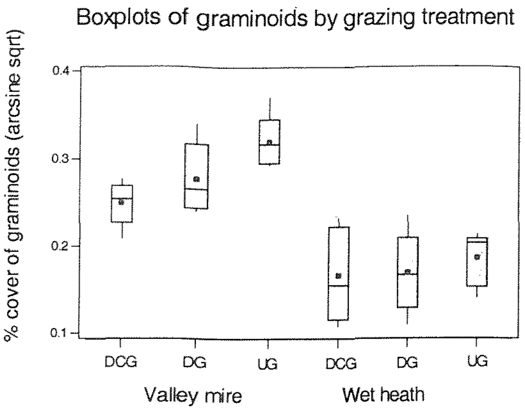


Figure 3.5b

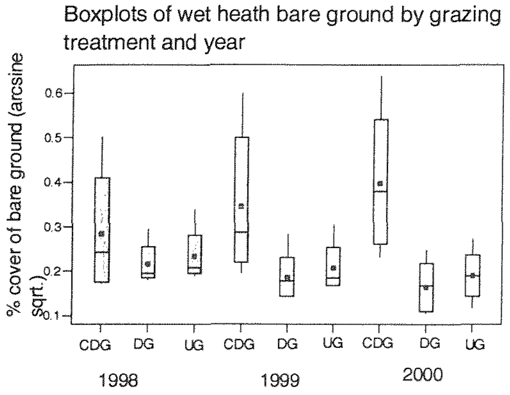


Figure 3.5c

Figure 3.5 Boxplots of (a) valley mire graminoids, (b) valley mire bare ground, (c) wet heath bare ground by treatment and year. Solid dots show means, range boxes the interquartile range, whiskers show confidence limits. CDG – cattle and deer grazed, DG – deer grazed, UG – ungrazed.

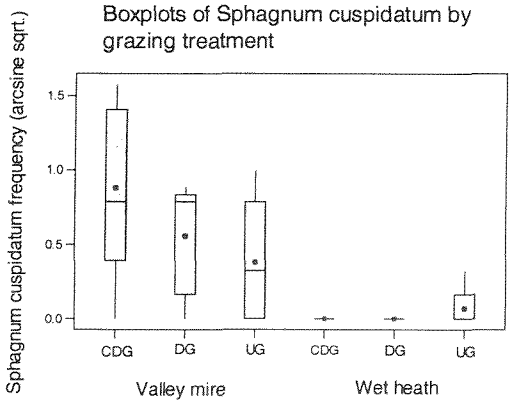


Figure 3.6 Boxplot of *Sphagnum cuspidatum frequency* against grazing treatment. Solid dots show means, range boxes the interquartile range, with the box bottom at the 25th percentile and box top at the 75th percentile, whiskers show confidence limits. CDG – cattle and deer grazed, DG – deer grazed, UG – ungrazed.

Frequency data from nested quadrats showed a similar pattern to percentage cover data. Bare ground was again significantly greater in CDG than DG and UG plots. In addition, an interaction between grazing treatment and vegetation type was found for *Sphagnum cuspidatum* - there was a clear increase in *S. cuspidatum* in CDG plots in valley mire, while the species was only found in UG plots in wet heath (Figure 3.6). F-ratios are presented in Table 3.6, and the results from Tukey comparison tests in Table 3.7

Table 3.6 Species, bare ground and water frequency (percentage presence in quadrats) and F-ratios generated by nested ANOVA following three years of grazing treatments. CDG – cattle and deer grazed, DG – deer grazed, UG – ungrazed, *** P <0.001, ** P<0.01, NS – not significant (P>0.01). Only those with significant results are presented here.

	CDG		**DG**		**UG**		**VEG**	**BLOCK**	**TREAT**	**TREAT* VEG**
Species	mean	st.dev	mean	st.dev	mean	st.dev	F-ratio	F-ratio	F-ratio	F-ratio
S cuspidatum	19.16	0.37	7.72	0.15	4.90	11.47	8.86^{NS}	9.71***	4.31^{NS}	9.71**
Bare ground	57.03	8.81	18.61	4.04	2.04	6.86	7.11*	2.26^{NS}	23.92***	0.706^{NS}
Water	77.89	25.32	21.65	11.79	12.89	14.20	26.25***	2.98	25.92***	3.02^{NS}

Table 3.7. Tukey comparison test for nested ANOVA of species frequency after three years of different grazing treatments. *** P <0.001, * P<0.05, NS – not significant. Only those with significant results are presented. Quadrat size gives the size for which the greatest difference between treatments was found.

Species	Quadrat size	CDG - UG T value	CDG - DG T value	DG - UG T value
S.cuspidatum	100	-2.82*	-2.09^{NS}	0.73^{NS}
water	25	-6.76***	-5.52***	-1.24^{NS}
bare ground	25	-6.90***	-3.76**	-3.14*

3.4.2 DIRECT GRADIENT ANALYSIS

Changes resulting from grazing treatment over the three-year study period were subtle, and as a consequence few were found to be statistically significant. Multivariate techniques were therefore used to detect trends in the pooled species data. Data collected in 2000, which showed the strongest effect of grazing treatment, were first analysed by an ordination unconstrained by the grazing treatment. Figure 3.8 shows the location of samples from each vegetation type in relation to the first two PCA axes: between them these axes explain 42.1% of the variance of the species data (λ_1= 0.306,

$\lambda_2 = 0.115$). Figure 3.7 shows a clear separation between wet heath and valley mire plots along the first axis. There is also a separation between CDG plot and UG or DG plots along the second axis, which accounts for 12.0% of the variation (Figure 3.8).

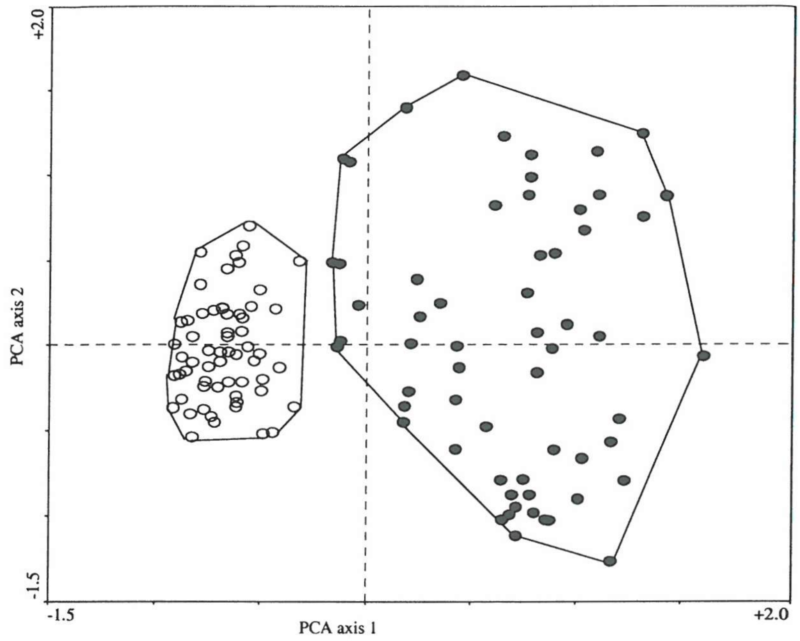


Figure 3.7 Ordination diagram from CANOCO for principal components analysis of vegetation data from 2000 showing the first two axes. Samples are plotted and differentiated by vegetation type. ∗ wet heath samples) valley mire samples

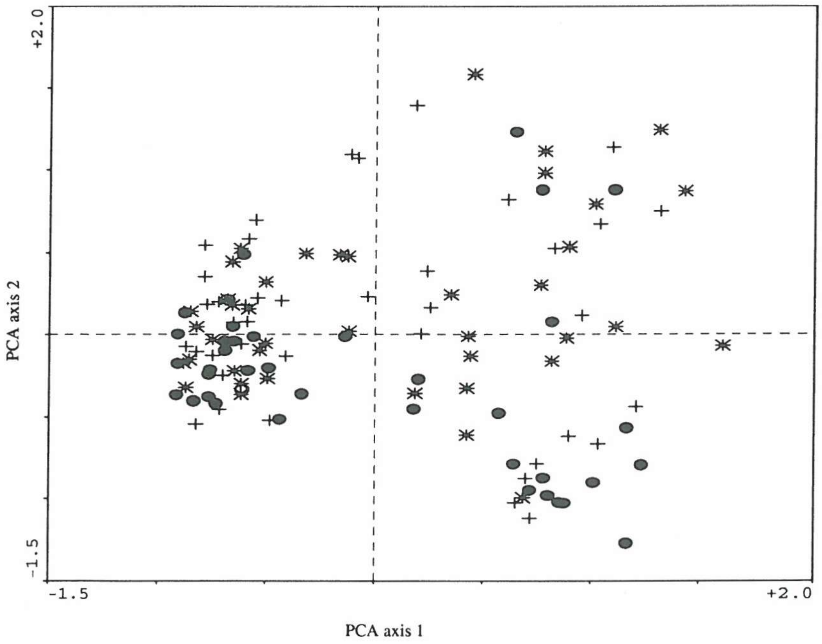


Figure 3.8 Ordination diagram from CANOCO for principal components analysis of vegetation data from 2000 showing the first two axes. Samples are plotted and differentiated by grazing treatment.● Cattle and deer grazed samples ∗ Deer grazed samples + Ungrazed samples

Figure 3.9 is a biplot diagram showing how the species contribute to each individual sample. From this it can be inferred that valley mire samples have a higher abundance of *Myrica gale, Sphagnum pulchrum, Eriophorum angustifolium, Rhynchospora alba, S. cuspidatum* and to a lesser extent *Sphagnum spp., S. papillosum, Narthecium ossifragum, Drosera rotundifolia* and *Juncus acutiflorus*, while wet heath samples have a higher than average abundance of the remaining species. Vegetation type was subsequently related indirectly to the sample scores for the axes: axis scores of wet heath samples are significantly different from valley mire sample scores (ANOVA $p<0.001$, F-ratio = 455.46). Grazing treatment was similarly related: CDG plots have significantly lower samples scores that both UG and DG (ANOVA p = 0.003, F-ratio = 6.13, Tukey's pairwise comparison).

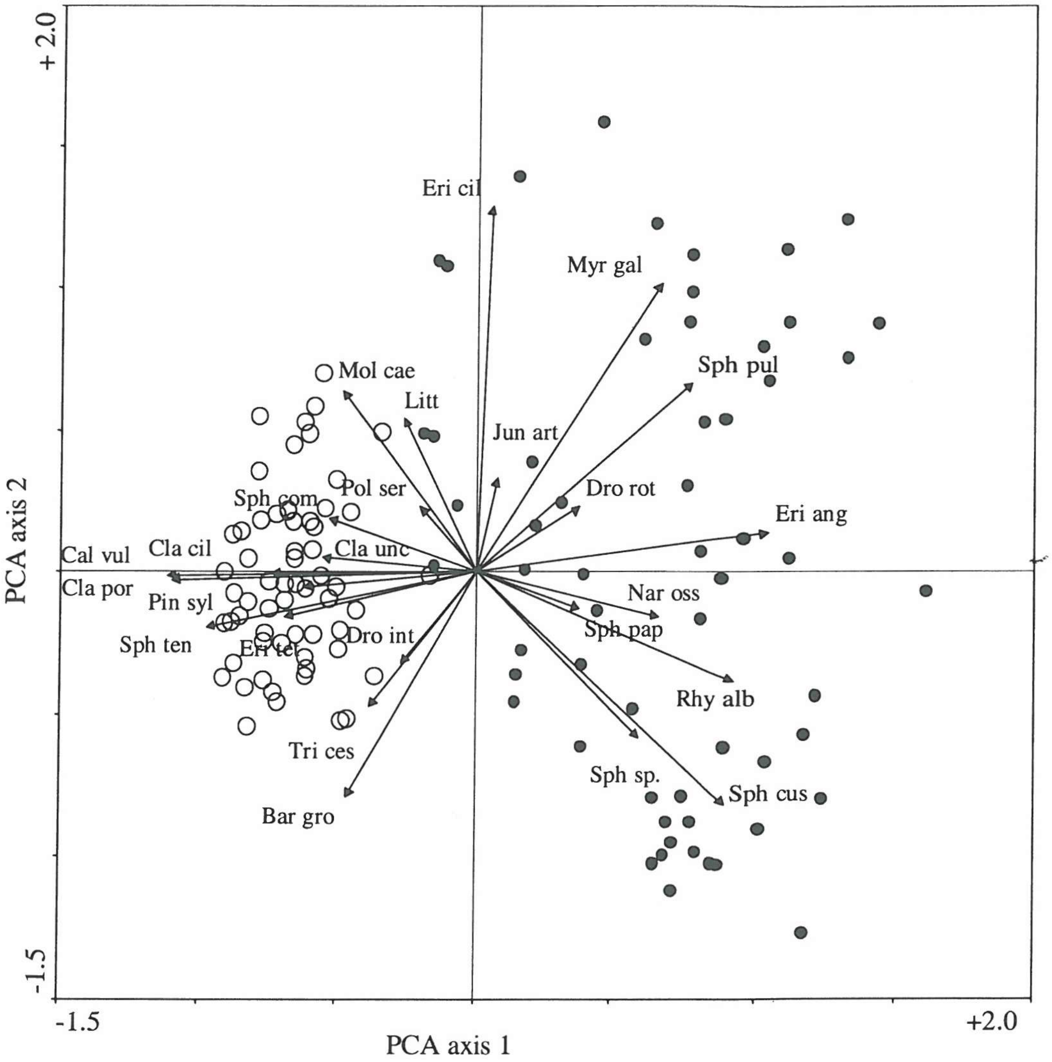


Figure 3.9 Ordination diagram from CANOCO for principal components analysis of vegetation data in 2000 giving the first two axes showing the relationship between species and samples. The direction of the species arrow shows the direction in which the abundance of the species increases most, the length of the arrow shows the rate of change in that direction.* wet heath samples) valley mire samples. Species codes are given in Appendix VII.

Axis 3 scores were significantly related to vegetation type (ANOVA $p = 0.017$, F-ratio = 5.91) while the CDG treatment is again differentiated from the remaining two treatments along axis 4 (ANOVA, $p < 0.001$, F-ratio = 23.86). However, axes 3 and 4 only accounted for a further 10% and 8.1 % of the variation respectively, and are not discussed further.

The two vegetation types were separated to remove variation due to the overall differences between the two communities. Species data were then directly related to grazing treatment by RDA. Figure 3.10 shows the relationship between species and grazing treatment for wet heath. Axes 1 and 2 account for 7.1 % of the species variation ($\lambda_1 = 0.61$, $\lambda_2 = 0.01$), and the relationship between the species and grazing treatment is significant (Monte Carlo test $p = 0.005$, F-ratio 2.163). From Figure 3.10 it is possible to infer correlations between species abundances and grazing treatment. Bare ground is highly correlated with CDG treatments, *Cladonia uncialis, C. ciliata* and *Drosera rotundifolia* moderately so, while *Trichophorum cespitosum, Sphagnum tenellum* and *Sphagnum* spp. are negatively correlated with CDG treatments and positively correlated with UG treatments. *Rhynchospora alba* and both *Erica* species are positively correlated with DG treatments, *Molinia caerulea* and *Sphagnum compactum* negatively so. However, if bare ground is removed from the analysis the percentage variation in the species data accounted for reduces to 4.5% ($\lambda_1 = 0.036$, $\lambda_2 = 0.009$) and the relationship between the axes and grazing treatment is no longer significant (Monte Carlo test $p = 0.13$, F-ratio 1.34).

Similarly, for valley mire vegetation the relationship between species and grazing treatment is significant when bare ground is included (Monte Carlo test $p = 0.005$, F-ratio 3.50) and the axes account for 10.9% of the species variation ($\lambda_1 = 0.91$, $\lambda_2 = 0.18$). From the ordination biplot (Figure 3.11) it can be inferred that a high abundance of bare ground, plus *Juncus articulatus, Cladonia ciliata* and *Sphagnum cuspidatum,* is highly correlated with CDG treatments, and *Erica tetralix, Cladonia portentosa* and *Calluna vulgaris* negatively so while being positively correlated with UG treatments. *Myrica gale, Rhynchospora alba* and *Drosera rotundifolia* are positively correlated with DG treatments, *Trichophorum cespitosum* and *Sphagnum compactum* negatively so. Again, when bare ground is excluded, the relationship is no longer significant (Monte Carlo test $p = 0.318$, F-ratio 0.17), and the amount of variation accounted for drops to 4.4% ($\lambda_1 = 0.033$, $\lambda_2 = 0.01$).

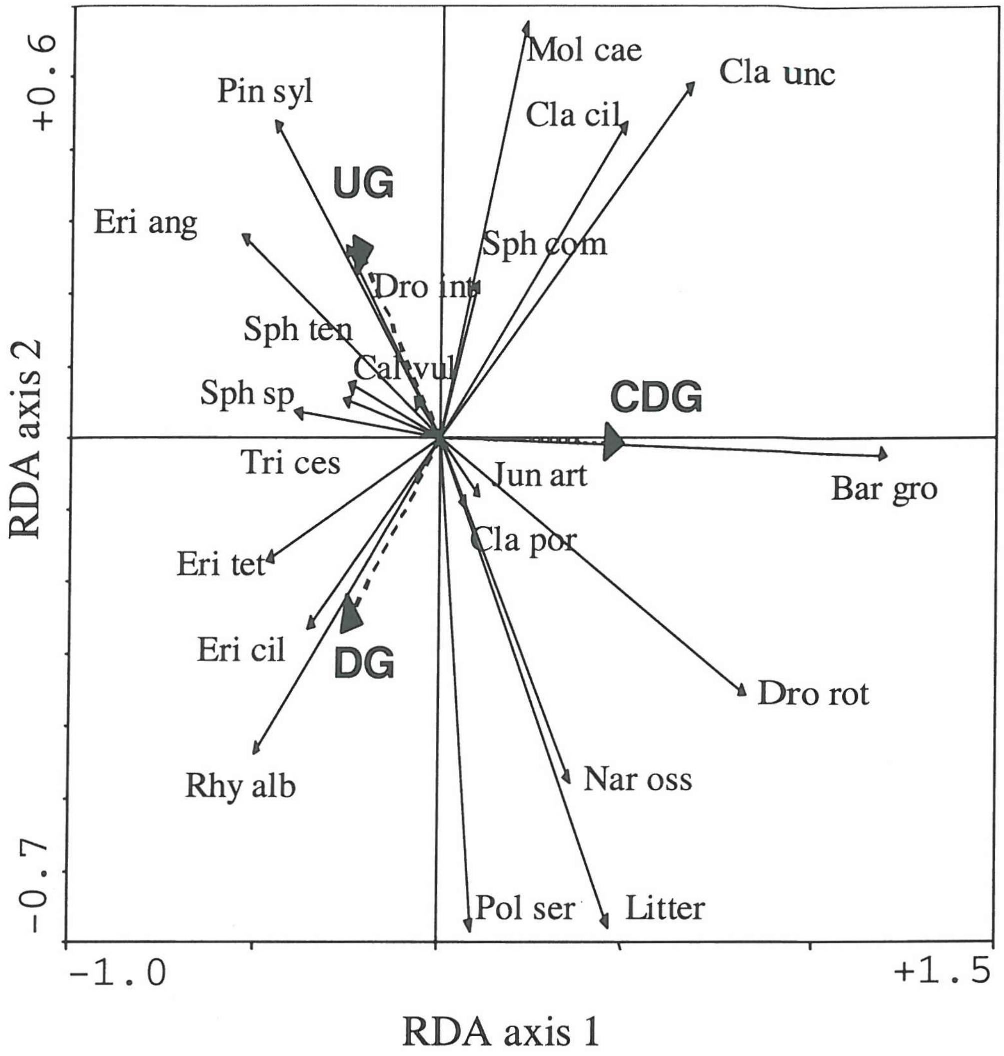


Figure 3.10 Ordination diagram from CANOCO for redundancy analysis of wet heath vegetation data from 2000 giving the first two axes, showing the relationship between plant species and grazing treatment. The cosine of the angle between species and grazing treatment arrows is an approximation of the correlation coefficient between them. Arrows pointing in the same direction therefore indicate a high positive correlation, in the opposite direction a high negative correlation, and at right angles a near-zero correlation. UG – ungrazed, DG deer grazed, CDG cattle and deer grazed. Species codes are listed in Appendix VII.

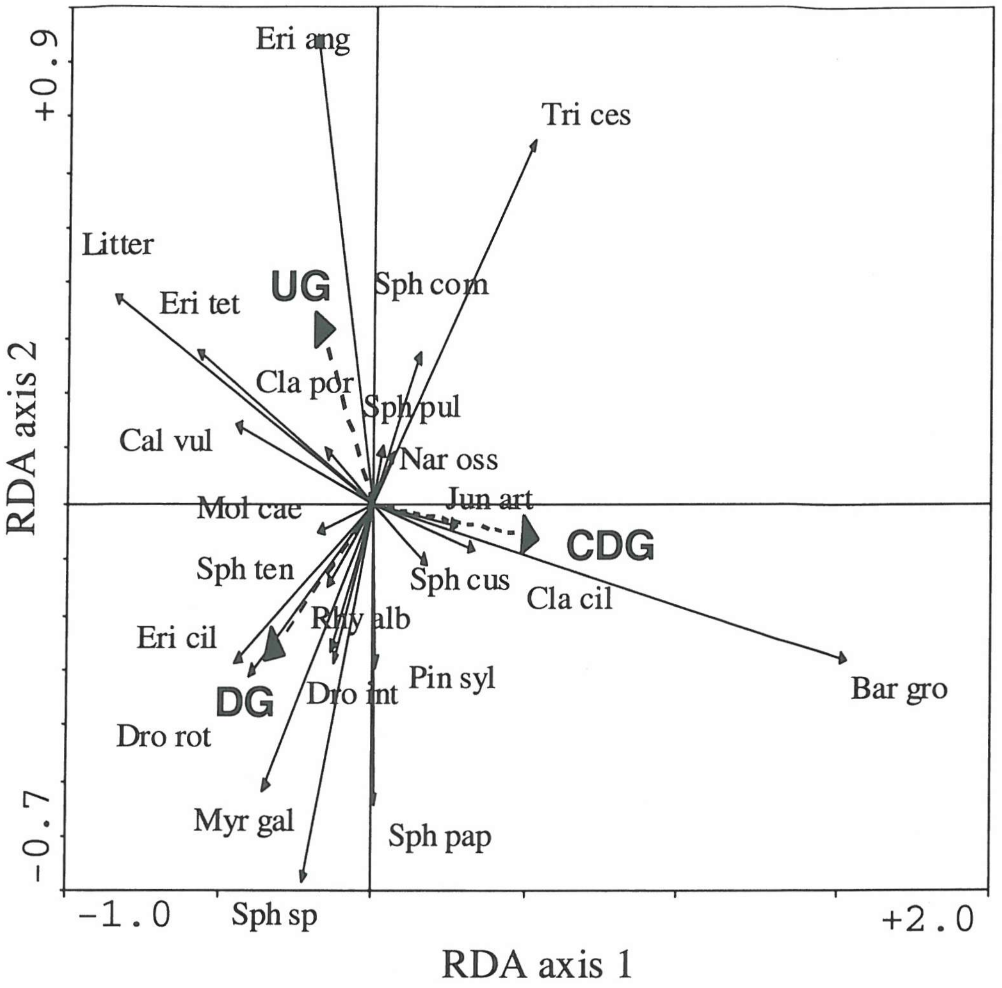


Figure 3.11 Ordination diagram from CANOCO for redundancy analysis of valley mire vegetation data from 2000 giving the first two axes, showing the relationship between plant species and grazing treatment. UG – ungrazed, DG deer grazed, CDG cattle and deer grazed. Species codes are listed in Appendix VII.

The amount of variation explained by the RDA is low. Previous analysis of the data shows that between-block variation was high (see species cover and frequency above). Consequently, the proportional difference in percentage species cover between 1998 and 2000 was also considered in order to reduce the variation due to initial differences between blocks. Wet heath axes 1 and 2 (the remainder showed zero correlation with grazing treatment in the RDA correlation matrix) accounted for 5.7% of the species variation (λ_1= 0.042, λ_2 = 0.15) with bare ground excluded, and all canonical axes are significant (Monte Carlo test p = 0.01, F-ratio 1.72). The biplot diagram (Figure 3.12) shows the changes associated with each grazing treatment. Thus, increases in *Drosera rotundifolia,* and to a lesser extent *Cladonia uncialis, Drosera intermedia and Trichophorum cespitosum* are positively correlated with the introduction of cattle grazing. Increases in *Molinia caerulea* are negatively correlated with cattle and deer grazing together with *Eriophorum angustifolium, Cladonia ciliata* and *Sphagnum* *compactum.* The exclusion of grazing over three years is positively correlated with increases in *Molinia caerulea, Sphagnum tenellum and Pinus sylvestris* seedlings, and negatively correlated with a large increase in *Polygala serpyllifolia.* Continued deer grazing is positively correlated with smaller increases in Juncus *acutiflorus* and *Calluna vulgaris.*

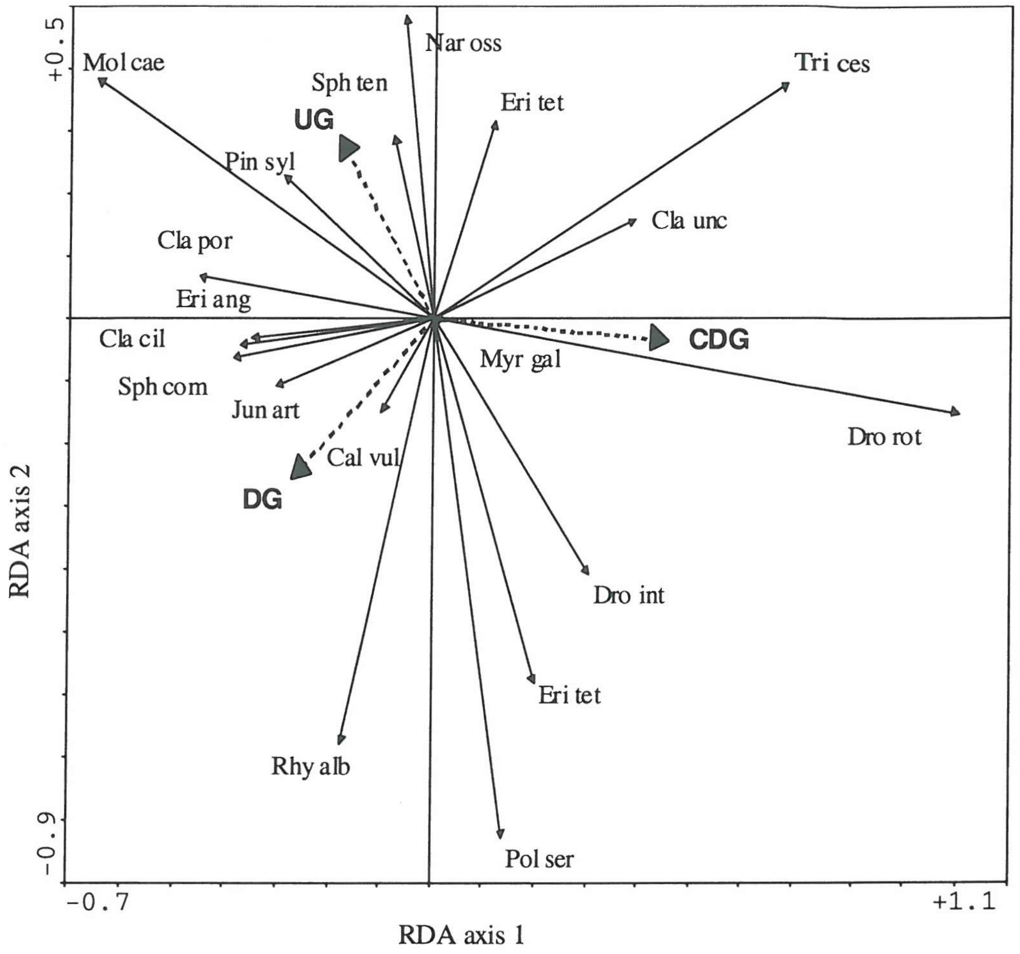


Figure 3.12 Ordination diagram from CANOCO for redundancy analysis of the proportional difference between species % cover in 1998 and 2000 for wet heath vegetation, giving the first two axes showing the relationship between species change and grazing treatment. UG – ungrazed, DG deer grazed, CDG cattle and deer grazed. Species codes are listed in Appendix VII.

Valley mire axes 1 and 2 accounted for 8.6% of the species variation ($\lambda_1 = 0.074$, $\lambda_2 = 0.012$) excluding bare ground (again, the remainder of the axes showed no correlation with grazing treatment). The canonical axes are significant (Monte Carlo test $p = 0.005$, F-ratio = 2.69). The biplot diagram (Figure 3.13) shows that cattle and deer grazing is again positively correlated with increases in *Drosera rotundifolia* and *Trichophorum cespitosum*, and also *Sphagnum* spp., which is absent from wet heath. It is negatively correlated with increases in *Sphagnum tenellum* and small increases in *Erica tetralix*. Increases in *Sphagnum pulchrum* and *Calluna vulgaris* are positively correlated with the maintenance of deer grazing, opposing changes in *Drosera intermedia* and *Pinus sylvestris* seedlings. As with wet heath vegetation, the cessation of deer grazing (UG plots) is positively correlated with large increases in *Eriophorum angustifolium, Sphagnum tenellum* and here, to a lesser extent, *Molinia caerulea*. Increases in *Myrica gale* and *Rhynchospora alba* are negatively correlated with the cessation of grazing.

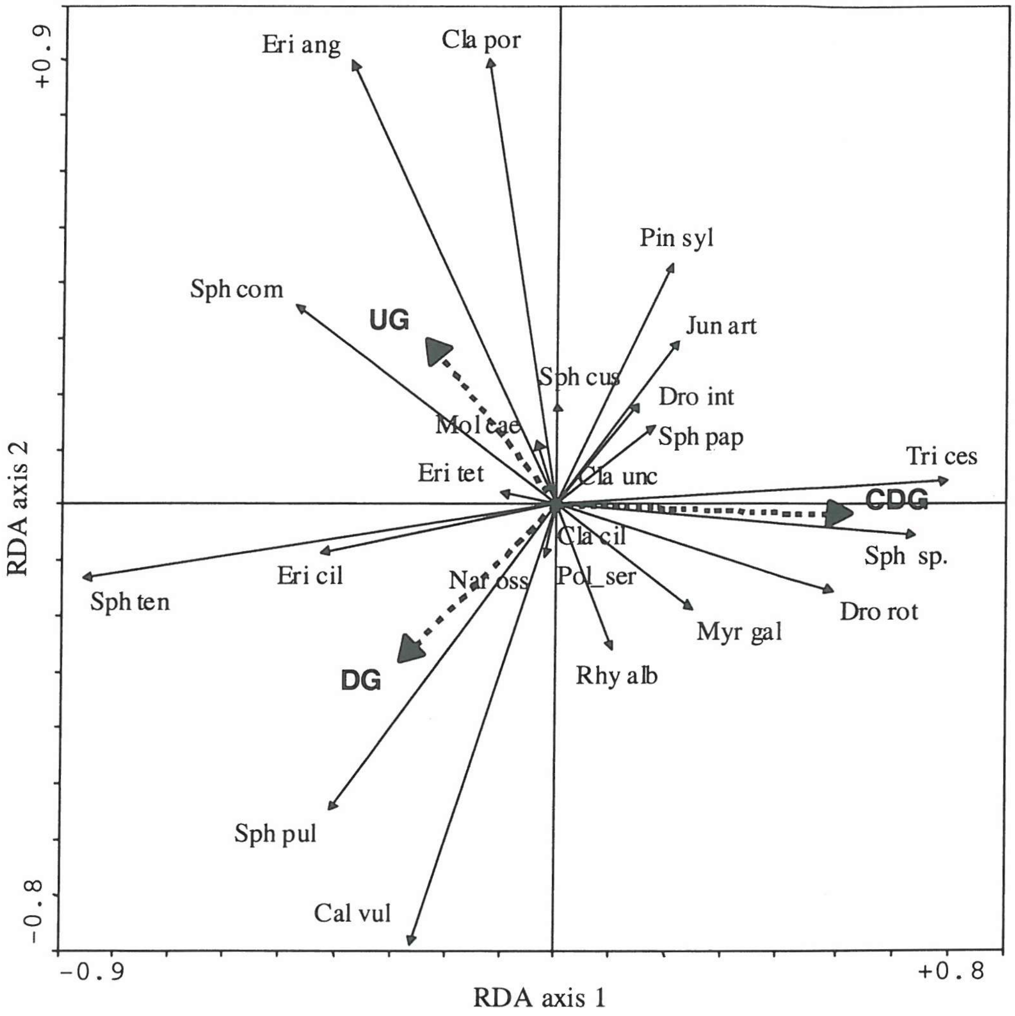


Figure 3.13 Ordination diagram from CANOCO for redundancy analysis of the proportional difference between species % cover in 1998 and 2000 for valley mire vegetation, giving the first two axes showing the relationship between species change and grazing treatment. UG – ungrazed, DG deer grazed, CDG cattle and deer grazed. Species codes are listed in Appendix VII.

3.4.3 SWARD HEIGHT AND STRUCTURE

Sward height in CDG plots was consistently lower than DG and UG plots. The sward height in DG and UG plots was not found to be significantly different (Table 3.8, 3.9). A repeated measures test shows that there is an interaction between year and treatment (Table 3.10), with treatment effects becoming stronger by 2000 (Figure 3.14a). In 2000, an interaction between vegetation type and grazing treatment is apparent, with the difference in sward height between CDG and UG plots larger in valley mire samples than wet heath samples (Figure 3.14b).

Table 3.8 Sward height (mm) and F-ratios generated by nested ANOVA for over three years of grazing treatments at Coombe Bog. CDG – cattle and deer grazed, DG – deer grazed, UG – ungrazed, *** P <01, ** P<0.01 * P<0.05, NS – not significant

Species groups	**CDG**	**DG**	**UG**	**VEG**	**BLOCK**	**TREAT**	**TREAT *VEG**
	mean (st.dev)	mean (st.dev)	mean (st.dev)	F-ratio	F-ratio	F-ratio	F-ratio
1998	135.06 (48.07)	172.47 (101.87)	163.06 (76.02)	1.79^{NS}	7.29***	3.26*	0.13^{NS}
1999	115.12 (25.81)	156.82 (60.75)	159.05 (62.65)	14.05**	2.31*	14.73***	0.88^{NS}
2000	82.70 (18.48)	161.25 (83.50)	175.93 (75.70	6.35*	4.34***	74.96***	3.54*

Table 3.9. Tukey comparison test for nested ANOVA of sward height following three years of grazing treatments at Coombe Bog. **** P <0.0001, * P<0.05, NS – not significant

	CDG - UG	CDG - DG	DG - UG
YEAR	T value	T value	T value
1998	2.310^{NS}	2.100*	-0.209^{NS}
1999	4.519****	4.862****	0.3437^{NS}
2000	9.684****	11.331****	1.647^{NS}

Table 3.10. F-ratios and adjusted P values generated by univariate repeated measures ANOVA tests on changes in sward height 1998-2000 at Coombe Bog. ** P<0.01, * P <0.05, NS – not significant (P>0.05).

	YEAR	**YEAR*BLOCK**	**YEAR*TREATMENT**
Vegetation type	F-ratio	F-ratio	F-ratio
Wet heath	9.83**	5.28*	7.18**
Valley mire	1.87^{NS}	3.03*	6.82**

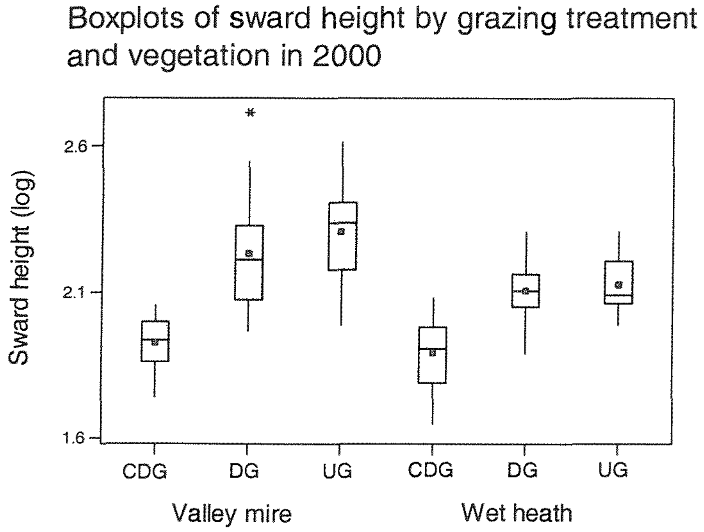


Figure 3.14 Boxplots of sward height (mm) against grazing treatment and vegetation type in 2000. Solid dots show means, range boxes the interquartile range, with the box bottom at the 25th percentile and box top at the 75th percentile, whiskers show confidence limits and * outliers. CDG – cattle and deer grazed, DG – deer grazed, UG – ungrazed.

Sward structure shows a similar response to grazing treatment as sward height (Figure 3.15). Vertical vegetation cover is less in CDG plots than in UG or DG plots (Tukey test CDG-UG: T = 8.514, P <0.0001, CDG – DG: T = 6.542, P<0.0001). Again, there is a significant interaction between treatment and vegetation type (Table 3.11) with a larger difference between treatments in valley mire vegetation (Figure3.15). Sward structure as measured by vertical vegetation cover can be predicted by sward height (Figure 3.16). No differences in the number of vegetation hits expressed per unit of sward height was found between treatments, and the decrease in sward structure in CDG plots can be assumed to be a direct consequence of the reduction in sward height.

Table 3.11 Vertical sward cover (measured as the number of vertical layers of vegetation) and F-ratios generated by nested ANOVA after three years of grazing treatments at Coombe Bog. CDG – cattle and deer grazed, DG – deer grazed, UG – ungrazed, *** P <0.001, ** P<0.01 * P<0.05.

	CDG		**DG**		**UG**		**VEG**	**BLOCK**	**TREAT**	**TREAT* VEG**
	mean	st.dev	mean	st.dev	mean	st.dev	F-ratio	F-ratio	F-ratio	F-ratio
Sward structure	2.17	0.63	3.85	0.97	4.35	1.28	9.87*	2.92*	39.73***	6.80**

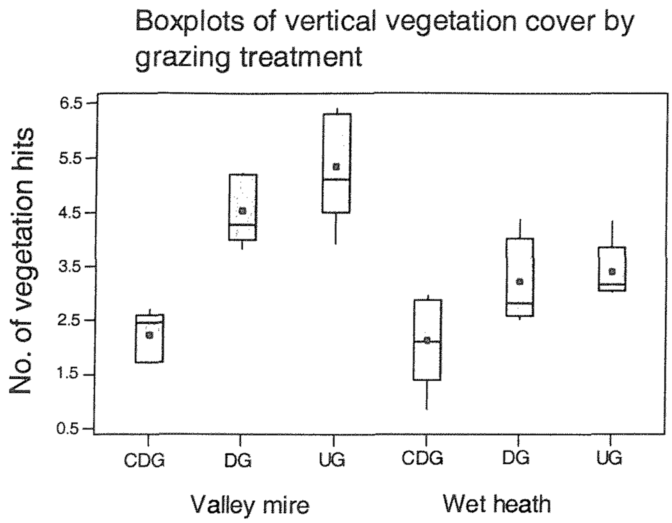
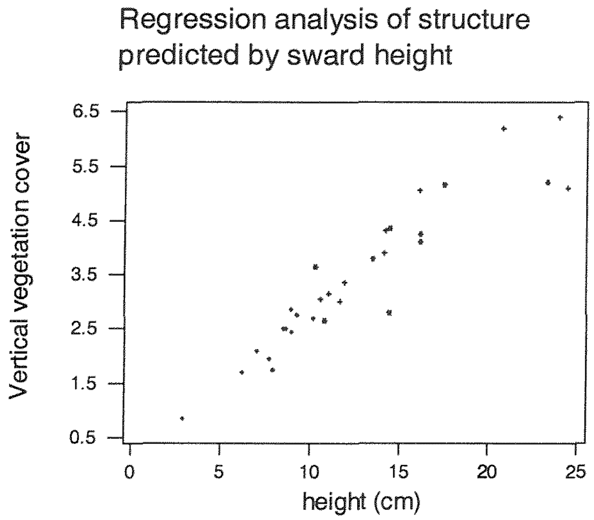


Figure 3.15 Boxplots of vertical vegetation cover against grazing treatment in 2000. Solid dots show means, range boxes the interquartile range, with the box bottom at the 25th percentile and box top at the 75th percentile. CDG – cattle and deer grazed, DG – deer grazed, UG – ungrazed.



R-Sq = 87.3%, F ratio = 191.91, P = <0 .001

Figure 3.16 Regression of vertical vegetation cover against sward height. Regression equation: structure = 0.463y + 0.236 height, ● Cattle and deer grazed samples * Deer grazed samples + Ungrazed samples.

The effect of grazing treatment on substrate compaction differed according to vegetation type (see Figure 3.17). In wet heath samples, compaction, considered as the inverse of substrate penetrability, is higher in CDG samples than in UG or DG samples. F-ratios generated by two-way ANOVA are presented in Table 3.12. This relationship is significant when Block 5 is removed from the analysis (CDG-UG T = 3.666, P =

0.0258, CDG – DG, T = 3.368, P = 0.039). Block 5 was removed as it is the only block located at the drier edge of the wet heath vegetation. It is positioned at right angles to the transition zone between wet heath and dry heath vegetation, with the CDG plot at the wetter end, and the UG plot at the drier end – i.e. the treatment (grazing) likely to decrease penetrability was carried out in the plot with assumed greater initial penetrability due to substrate composition and moisture. It is considered that the initial differences in the substrate unduly influenced the compaction measurements for this plot. In contrast to the wet heath, in the loose wet peat of the valley mire cattle and deer grazing led to a higher degree of penetrability than was found on UG or DG plots (CDG-UG T = -5.071, P = 0.002, CDG-DG T = -3.778, P = 0.013).

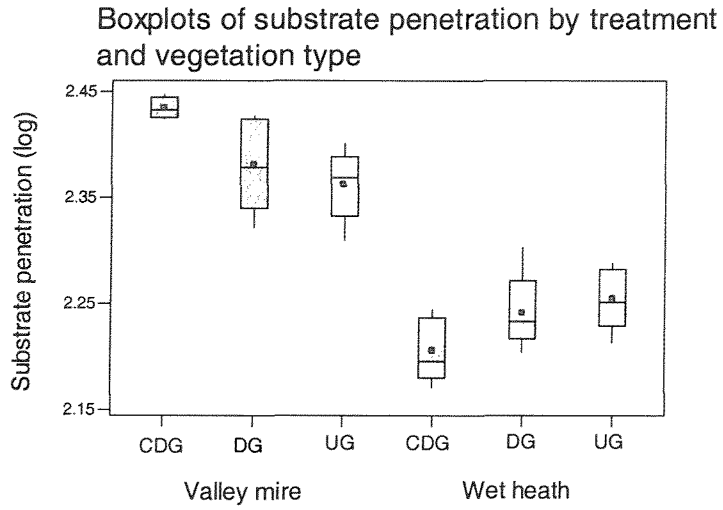


Figure 3.17 Boxplots of substrate penetrability (log) against grazing treatment in 2000. Solid dots show means, range boxes the interquartile range, with the box bottom at the 25th percentile and box top at the 75th percentile, whiskers show confidence limits. CDG – cattle and deer grazed, DG – deer grazed, UG – ungrazed

Table 3. 12. F-ratios generated by two-way ANOVA for soil penetrability (mm) following three years of grazing treatments at Coombe Bog. CDG – CDG, DG –DG, UG – UG, *** P <0.001, * P<0.05, NS – not significant.

	CDG		**DG**		**UG**		**BLOCK**	**TREAT**
Vegetation type	mean	st.dev	mean	st.dev	mean	st.dev	F-ratio	F-ratio
Wet heath	157.27	1.056	178.44	1.083	180.93	1.079	3.02^{NS}	7.97*
Valley mire	271.76	1.023	240.32	1.107	230.30	1.099	4.31*	13.89**

3.4.4 MICROTOPOGRAPHY

Grazing treatment has a clear effect on microtopographical variation, which is greater in CDG plots than either DG or UG plots (CDG – DG T = -5.945, P = 0.0001, CDG – UG, T = -5.076, P = 0.0003). Figure 3.18 shows that the pattern of response was the same in each vegetation type, although the variation between treatments was greater in wet heath.

The increased microtopographical range in CDG plots meant that a significantly greater proportion of each plot lay below the standing water level than in the remaining treatment plots (CDG - DG T = -6.329, P <0.0001, CDG – UG T = -7.545, P <0.0001). Again the difference between vegetation types is significant and there is no interaction between grazing treatment and vegetation (Figure 3.19, Table 3.13).

Table 3.13 Microtopographical variation (expressed as the variance of the vertical differences between adjacent points) and water cover (expressed as proportion of points below water level) with F-ratios generated by nested ANOVA for after three years of grazing treatments at Coombe Bog. CDG – cattle and deer grazed, DG – deer grazed, UG – ungrazed, *** P <01, ** P<0.01 * P<0.05.

	CDG		**DG**		**UG**		**VEG**	**BLOCK**	**TREAT**	**TREAT* VEG**
	mean	st.dev	mean	st.dev	mean	st.dev	F-ratio	F-ratio	F-ratio	F-ratio
Microtopographical variation	58.61	1.80	23.29	1.81	26.61	1.07	6.95*	4.49**	20.62***	1.17^{NS}
Water cover	0.40	0.30	0.12	0.15	0.06	0.09	178.91***	0.42^{NS}	32.82***	1.26^{NS}

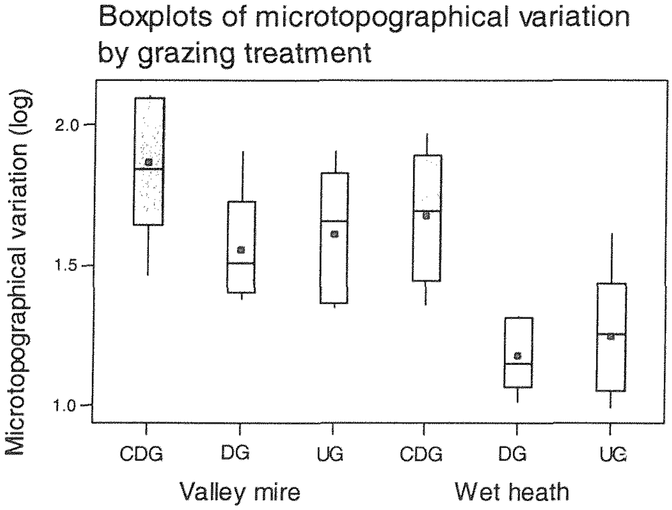


Figure 3.18 Boxplots of microtopographical variation against grazing treatment in 2000. Solid dots show means, range boxes the interquartile range, with the box bottom at the 25th percentile and box top at the 75th percentile, whiskers show confidence limits. CDG – cattle and deer grazed, DG – deer grazed, UG – ungrazed.

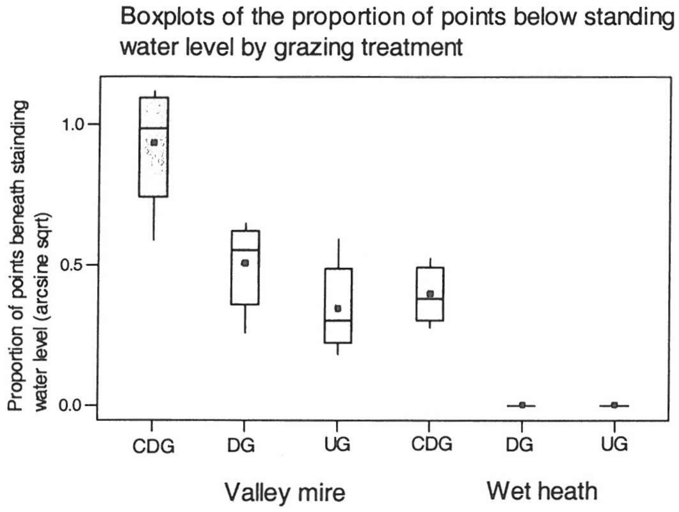


Figure 3.19 Boxplots of frequency of water cover against grazing treatment in 2000. Solid dots show means, range boxes the interquartile range, with the box bottom at the 25th percentile and box top at the 75th percentile, whiskers show confidence limits. CDG – cattle and deer grazed, DG – deer grazed, UG – ungrazed

3.4.5 SCRUB

After one grazing season, pine saplings within the grazing unit showed more browsing damage than those outside it, but not significantly so. After two further grazing seasons, saplings within the grazing unit had a significantly greater number of browsed side branches and lead shoots (Kruskall-Wallis, H = 155.35, P < 0.001) and were slightly shorter (Kruskall-Wallis, H = 8.04, P < 0.005), than those outside the grazing unit (Figure 3.20). There was no difference in estimated age.

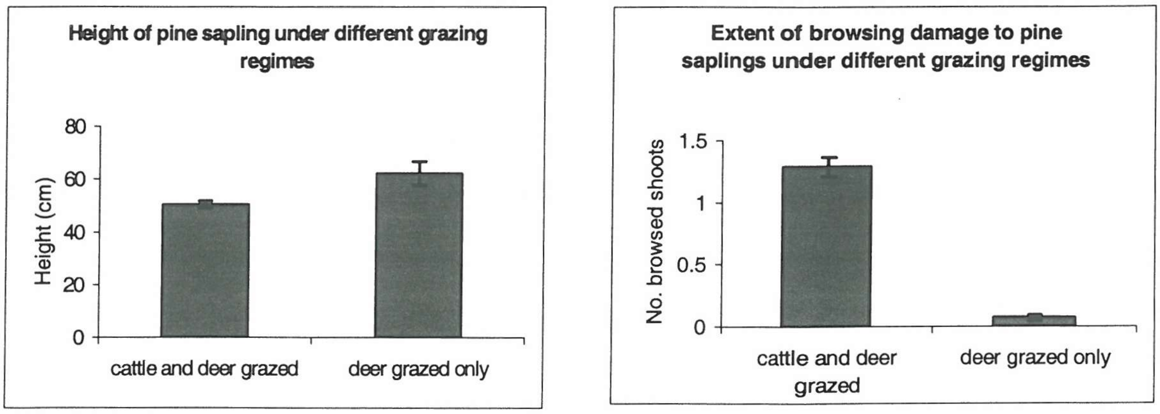


Figure 3.20 Effects of different grazing regimes on regenerating pine at Arne.

3.4 Discussion

The introduction of cattle to Coombe Bog had a marked effect on both wet heath and valley mire. Despite the short time-scale of the experiment, changes in vegetation structure and environmental variable were apparent, and suggest that further species changes will occur over time. Changes in vegetation structure were apparent after just one season's grazing, and differences were reinforced over the subsequent two seasons. Environmental variables also showed a strong treatment effect, with both compaction and microtopography clearly responding to livestock presence after three grazing seasons. However, it is clear that plant species responses to grazing at Coombe Bog were slower to occur. Differences in cover of two species groups according to treatment were apparent, with cover of graminoids and ericoids lower in CDG plots, where bare ground cover was highest. The contributions of individual species to these differences were less clear - no treatment effects on individual species were statistically significant using either percentage cover or frequency data from 2000, after three grazing seasons. However, when variation due to vegetation type and initial differences between blocks were accommodated and the species data pooled, clear trends emerged (Figures 3.12 and 3.13). Changes in species composition over time under grazing treatment are discussed below.

Taking the initial grazing status of the site into account, the grazing treatments used in this experiment were (i) the introduction of cattle grazing and maintenance of deer grazing, (ii) the maintenance of deer grazing, and (iii) the cessation of deer grazing and prevention of cattle grazing (i.e. UG plots). The deer-only grazed plots were accessed through a three-strand wire fence to exclude cattle. This is likely to have discouraged deer, although they could easily move through it, and there was no indication of deer use in 1998. However, in 1999 and 2000, by which time the number of deer using the site is thought to have increased (N. Gartshore, pers.com.), deer were occasionally observed grazing in these plots and deer droppings were found within them. The lack of difference between UG and DG plots in most analyses generally suggests, however, that the deer pressure remained low. That the deer, when allowed uninhibited access to wet heath vegetation, did have an impact is evident from the species responses to the cessation of deer grazing in the ordination diagrams.

3.4.1 VEGETATION CHANGES

The most striking contribution to changes in both valley mire and wet heath vegetation was that of bare ground. High cover of bare ground was characteristic of CDG vegetation, and although this is most easily attributed to the significant decline in graminoids, species in this group showed a variable response to grazing. Of the three graminoids showing greatest responses in valley mire plots, increases in *Molinia caerulea* and *Eriophorum angustifolium* were both negatively correlated with the CDG treatment while *Trichophorum cespitosum* showed a positive correlation with cattle and deer grazing in valley mire. In wet heath vegetation, increases in *Molinia caerulea* were positively correlated with UG treatments, but the remaining graminoids species were not strongly correlated with any treatment.

Previous studies suggest that the nature of graminoids response to grazing is dictated by grazing intensity. For example, *Molinia caerulea* is preferentially selected by herbivores from heathland swards (Grant *et al*, 1976; Putman *et al.*, 1987; see also Chapter 2), but response to grazing varies. While both a reduction in cover due to grazing (e.g. de Beaulieu, 1998) and an increase following the cessation of grazing (e.g. Grant *et al.*, 1996, Clarke, 1988) have been recorded, an increase under heavy grazing has also been noted (Welch, 1984) presumably due to the concurrent grazing of competitors. Grant *et al.* (1996) showed that offtake of over 33% lead to a reduction in biomass. Below this level biomass may not be reduced; and therefore a light grazing pressure may not result in a significant reduction. This may be why *Molinia* reduction was not as strong as expected although preferentially grazed at Coombe Bog.

In upland blanket bogs the removal of sheep grazing can result in a decrease in *E. angustifolium* (Rawes, 1983), although heavy sheep grazing can also lead to a reduction (Grant *et al.,* 1985). Suggested community changes in upland blanket bogs resulting from grazing and burning generally involve a shift towards *Eriophorum vaginatum* or *Trichophorum cespitosum* dominated communities with a reduction in ericoids and *Sphagnum* species (Thompson & Miles, 1995). However, in Coombe bog increases in *Eriophorum angustifolium* contributed to the ordination description of UG plots – this was visually very apparent when in flower (Plate 3.2). This is likely to be a consequence of its ability to rapidly expand vegetatively (Philips, 1954). Within the three-year time scale of this project, this increase was not checked and reversed by the expansion of other species, which may have been the case over the longer time spans considered by Rawes (1983) and Grant *et al.* (1985).

Trichophorum cespitosum is characteristic of sheep grazed and trampled swards in the uplands, and cessation of grazing is reported to have resulted in the loss of distinctive

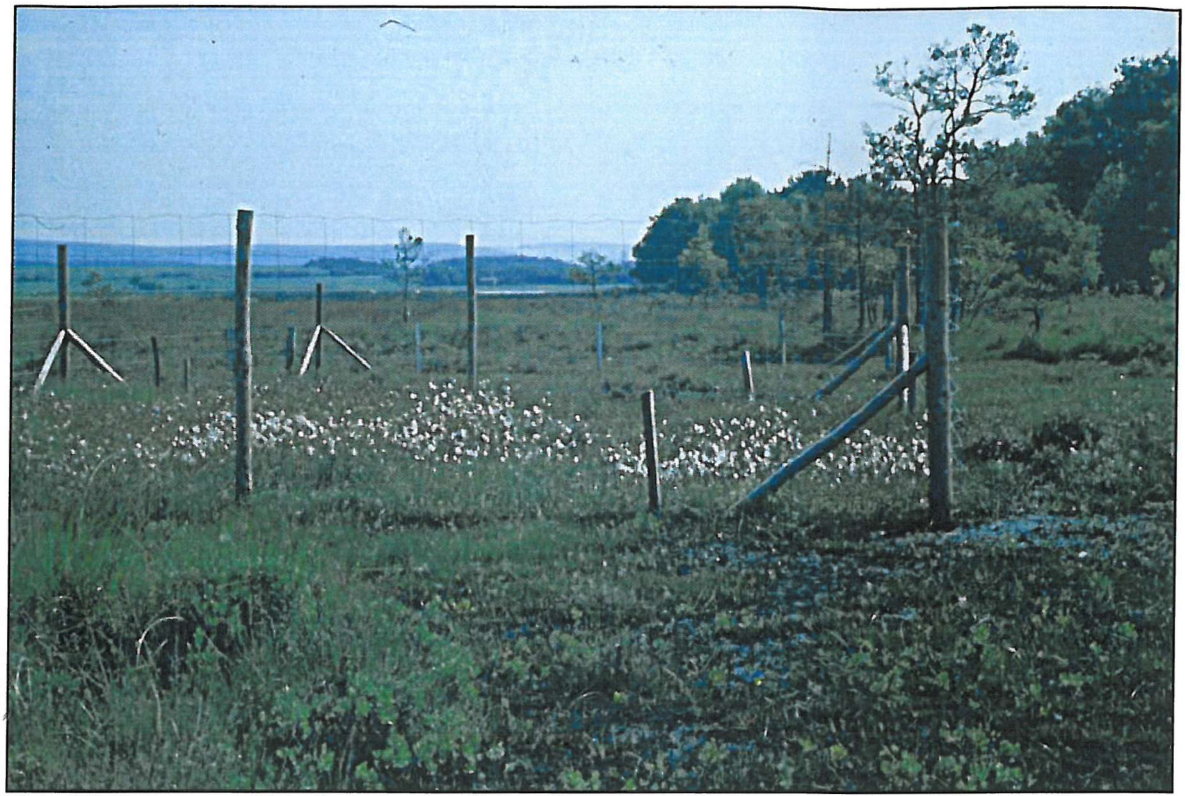


Plate 3.2 Cotton grass *Eriophorum angustifolium* flowering inside a cattle and deer exclosure at Coombe Bog, Arne, in 2000.

stands of *T. cespitosum*, which became intermingled with other species (Rawes, 1983), although the species is palatable to both sheep and deer (Hudson & Newborn, 1995). Guile (1965 cit. Shaw *et al.*, 1996) suggests that surface consolidation (together with repeated burning) favours *T. cespitosum*, while Shaw *et al.* (1996) consider that reductionof *M. caerulea* through burning favours *T. cespitosum* (1996). Rodwell (1991) suggests that it may spread along pathways in bogs where trampling decreases aeration. However at Coombe Bog the increase in *T. cespitosum* is neither closely correlated with a decrease in *M. caerulea* not attributable to increased compaction in valley mire plots, where soil penetrability was increased under cattle and deer grazing (see below). The mechanisms by which this species responds to grazing are not clear. However there is some evidence that this species regenerates from the buried seed bank given appropriate conditions (see Chapter 5). A similar response might be expected from *Rhynchospora alba*, which also regenerates from the buried seed bank. This may occur in wet heath, where a small increase is negatively correlated with lack of grazing, but the relationship is more ambiguous in valley mire.

A key characteristic of deer and cattle grazed treatments is the increase in *Drosera rotundifolia.* In addition, on wet heath *Polygala serpyllifolia* colonised grazed treatments, and *Potentilla erecta* was recorded twice from frequency quadrats in CDG plots. All three species are diminutive, and colonisation and regeneration opportunities are presumably improved by an increase in resources, particularly light, typified by a decrease in more competitive species and increases in bare ground. An increase in compaction, leading to increased water cover, may also increase germination in *Drosera* spp. (see Chapter 4). In addition, if these species are represented in the buried seed bank, churning of the substrate through trampling may expose seeds and enable germination (see Chapter 5).

Ericoids played a more limited role in the characterisation of vegetation responses to grazing treatment at Coombe. As a group, ericoids were found to have significantly lower cover in CDG plots than in DG plots. This was largely due to an increase in *Calluna* in DG plots in both wet heath and valley mire. *Calluna* is generally palatable to livestock, whereas *Erica* spp. are considered less so (Putman, 1987; Tubbs, 1991). The increase in *Calluna* under deer grazing may have been due to the stimulation of new growth by grazing (Grant & Hunter, 1993). Young vigorous heather is only likely to be damaged if >40% of annual production is removed per year (Grant *et al.*, 1982), although this figure may be as low as 20% in wetter communities (Hudson & Newborn, 1995) where growth of ericoids is generally weaker (Rodwell, 1991). This is unlikely to be the case in Coombe Bog where grazing pressure within the deer exclosures was under 0.04 $LU^{-1}\ ha^{-1}\ yr^{-1}$.

Several changes in *Sphagnum* characterised different grazing treatments. In valley mire, CDG plots showed increases in the cover of *Sphagnum* spp. (damaged, discoloured shoots which could not be identified) and the frequency of *Sphagnum cuspidatum* but decreases in *Sphagnum compactum*. An increase in *Sphagnum pulchrum* was associated with deer grazing. On the wet heath, *Sphagnum tenellum* and *compactum* both appeared negatively affected by livestock presence.

Bryophytes are likely to benefit from grazing when it reduces competition from vascular plants and provides patches of bare substrate, but may be damaged by heavy trampling. *Sphagnum* species are mostly likely to be affected by light availability and water table level (Hayward & Clymo, 1983). Livestock presence may both increase light availability through reducing the cover of graminoids, and raise the relative water level through trampling (discussed below), and this may be reflected in the greater frequency of *Sphagnum cuspidatum,* a pool species, in CDG plots. However, the cover of this species did not increase, which may be due to direct trampling damage.

Similarly, *Sphagnum pulchrum*, which increased in DG plots, is generally a species of wet, open mire and may benefit from light grazing of the surrounding vegetation, but may be susceptible to trampling. However, studies from the uplands suggest that trampling can lead to an increase in *Sphagnum* species (Rawes, 1983; Marrs *et al.*, 1988). Lindsay & Ross (1994 cit. Shaw *et al.*, 1996) found that *Sphagnum tenellum* is able to colonise bare areas and is common in trampled sheep tracks. In Coombe Bog this species appeared to be colonising old deer tracks within UG plots, but not recently disturbed bare areas. *Sphagnum* species have impressive powers of regeneration (Clymo & Duckett, 1986) so may potentially recover well from trampling damage. However, trampling can also disrupt the *Sphagnum* carpet, damaging the typical microtopography associated with hummock forming species such as *Sphagnum papillosum* (Shaw *et al.*, 1996). In Coombe Bog, the increase in damaged *Sphagnum* suggests that in general *Sphagnum* species were vulnerable to trampling, although it is not clear which, if any, species were disproportionately affected, or whether regeneration will occur in the disturbed areas. The intensity and duration of tramping will dictate whether or not it is beneficial for bryophyte species over a longer time span.

Lichens play a limited role in the characterisation of grazing treatments. Lichens are not common within valley mire vegetation, although more frequent in wet heath. *Cladonia portentosa*, plus *Cladonia ciliata* in wet heath, appears vulnerable to livestock presence - increases in these species are negatively correlated with CDG in wet heath and positively correlated with UG treatments in valley mire. Increases in *Cladonia uncialis* are negatively correlated with DG in wet heath. Species such as *Coelecaulon aculeata* and *Pycnothelia papillaria,* which might be expected to respond to an increase in bare ground, occurred at too low a frequency to analyse meaningfully.

Evidence for the impact of livestock on lichen species is conflicting, and no clear trends emerged from this study. Rawes & Hobbs (1979) showed that excluding sheep from blanket bog resulted in an increase in lichen cover after 21 years, while intense grazing caused a decline. Rodwell (1991) suggests that trampling in mire communities can destroy larger *Cladonia* species and favour an increase in crustose lichens, a view which is to some extent corroborated by the changes in lichen cover at Coombe Bog. Grazing appears to be important in maintaining lichen cover on lowland dry heath (Sanderson, 1994; Bullock & Pakeman, 1997; Saunders 1997; Wright & Westerhoff, 2001). Sanderson (1994) considers these lichens to be dependent on constant low-level disturbance such as grazing, or periodic heavy disturbance such as fire, to prevent dominance by dwarf shrubs. In general, wet heath and valley mire communities are not characterised by as great a diversity of lichens as their drier counterparts. Levels of disturbance that create suitable conditions for these species on dry heath may have a more profound effect on wetter communities, which are more susceptible to poaching.

3.4.2 SWARD STRUCTURE

The relatively limited changes in species composition under cattle and deer grazing outlined above suggest that this treatment does not have a substantial effect on wet heath and valley mire vegetation in the short term. However, the increase in bare ground combined with the pronounced decrease in sward height in CDG plots refutes this. Livestock presence significantly impacted on the vegetation, even if this impact was not clearly expressed in terms of species composition changes within the time span of this study. The increase in bare ground may be largely due to vegetation loss through trampling rather than grazing. This interpretation is to some extent supported by the general lack of any significant difference between species responses to grazing, since livestock grazing on heathland habitats is likely to be selective (Grant *et al.*, 1976; Baker, 1993; Putman *et al.*, 1987, Bartolome *et al.*, 1998, see also Chapter 2). However, a closer examination of the sward height data suggests this is not the case. Sward height data for CDG plots in 2000 have a normal distribution. If the decline in sward height were due to a total loss of vegetation in some patches, with the remaining sward having a similar height to previously, this would not be the case. The maximum sward height recorded in 1998 in cattle and deer gazed plots was 473mm, compared with 186mm in 2000, again suggesting an overall reduction in sward height, which can be seen in Figure 3.14. Differences in sward height between treatments were greatest in valley mire plots, which had a greater overall cover of graminoids than wet heath (see Table 3.4b). Grazing of graminoids evidently contributed to sward height reduction.

Sward structure corresponded closely to sward height, and the structure value was significantly less in grazed plots. This suggests that no discernable morphological response to grazing occurred, for example increased branching as a consequence of grazing (although this is only likely to occur within the ericoids, which were only lightly affected by grazing). As a consequence, the reduction in sward height presumably resulted directly in an increase in available light for lower growing species such as the forbs.

3.4.3 ENVIRONMENTAL VARIABLES

Trampling nevertheless had a significant effect on these vegetation communities. The effect of cattle trampling on soil depends on its water content (Mulholland & Fullen, 1991). Trampling generally results in compression of soils with low to medium water content (Scholefield *et al.*, 1985), and on wetter soils may cause plastic flow resulting in deep hoof prints (Taboada & Lavado, 1993). On wet heath, both occurred - compression was greater and microtopography more varied in CDG plots. However, in the valley mire, where the waterlogged peat was in a layer no more than 30cm thick overlying firm sand, cattle trampling had the reverse effect, apparently further loosening the substrate. On wet heath the microtopographical variation frequently took the shape of clearly definable hoof prints. In valley mire, the softer substrate did not hold the impression, but rather trampling appeared to result in a generally more hummocky surface. These changes appeared to be due to the immediate effects of trampling on the substrate, rather than as a consequence of changes in the topography of the *Sphagnum* carpet (see below) or in reduction of tussocky plants.

Increase in microtopographical variation was correlated with an increase in the amount of standing water. This may be due to the compaction of the substrate, lowering it relative to the water table and making flooding more likely, or to the decreased porosity of compacted substrate and the presence of small hummocks retaining surface water. This may have greater consequence for wet heath species, since only CDG wet heath plots held surface water during the survey period (late summer).

Apart from affecting the persistence of vulnerable species such as *Sphagnum*, the consequences of trampling are important for the regeneration and colonisation of species. In addition to providing bare substrate for colonisation, trampling may both expose buried seed banks and affect seed germination though changes in light and water availability (Thompson & Booth, 1992). It seems probable that changes in cover of forbs are a direct result of trampling, but the effects were not easily defined in other groups, and may take longer than the three years of this study to evolve. The effects of livestock presence on regeneration from buried seed bank and on germination are explored experimentally in Chapters 4 and 5.

3.4.5 HABITAT DIFFERENCES

Livestock presence affects wet heath and valley mire communities to different extents. Valley mire plots show a stronger response to grazing treatment than wet heath plots in terms of species cover and sward structure. Microtopographical differences were similar in both communities, although the increased surface water in CDG plots may have greater consequences on wet heath. Changes in substrate compaction were of similar magnitude but in opposing directions.

Habitats use by herbivores is highly selective (Jarman & Sinclair, 1979; Duncan, 1983, Putman, 1997; Gordon, 1989). Cattle are likely to use valley mire less than wet heath (Pratt *et al.*, 1986). This was the case in Coombe Bog, where observed use of valley mire was rare. Use of wet heath was seasonal, increasing throughout the summer until it became a preferred habitat in September (see Chapter 2). Cattle location data (see Chapter 2) suggest that livestock pressure on valley mire was four times lower than that on wet heath. However, valley mire appears considerably more sensitive to livestock presence than wet heath with regard to both vegetation and environmental variables. This can be explained in terms of the higher cover of species vulnerable to grazing (graminoids and *Sphagnum*) in valley mire, and the susceptibility of wet substrates to poaching.

3.4.6 CHANGES IN SPECIES DIVERSITY

In the introduction, a number of mechanisms were discussed by which species diversity may be changed. A number of these can be tentatively identified at Coombe Bog. The most significant change in species composition is clearly the reduction of graminoids due to selective feeding, thus reducing competition for resources. Changes due to gap colonisation can be seen in the increase in forbs through seedling establishment, and possibly changes in the abundance of *Trichophorum cespitosum*. Differences in diversity over the site as a whole were not studied directly, although significant differences found between exclosures suggest that patchy habitat use may be increasing large-scale structural heterogeneity. For example, the increase in bare ground was greatest in exclosure 4. This exclosure is located close to a pool used by the cattle (see Figure 2.9, Chapter 2), and a cow path runs along the edge of the exclosure. Changes in bare ground, sward height, compaction and microtopography generally showed relatively high standard deviations from the treatment means, suggesting that a diversity of gap types were created. However, species changes within the time span of the project were not sufficient for the effect of this on species composition to be detected. Dispersal of species by cattle is likely to be largely through dunging (e.g. Welch, 1984). While this was noted on other heathland habitat types (see Chapter 6) there was no indication of this occurring at Coombe. Dunging on wet heath and valley mire communities is infrequent (see Chapter 2). Traces of dung were found in 12.5% of CDG quadrats, but were not observed to have resulted in species colonisation.

Species richness is slightly higher in CDG plots than DG and UG plots (Table 3.2). Although NVC descriptions are by no means prescriptive, it is useful to have an indication of what other species might be expected in these communities. Comparison of the experimental plots with their closest NVC descriptions shows that species richness remained lower even after grazing than might be expected. However, examination of botanical records from the 1930s suggests that the key vascular species absent from the experimental plots, such *as Pinguicula lusitanica, Gentiana pneumonanthe, Hammarbya paludosa*, *Dactylorhiza maculata, Succisa pratensis* were not historically present on the Arne peninsula, perhaps as a consequence of particularly low pH and nutrient status, or of maritime influence. The majority of the remaining expected species are bryophytes, in particular liverworts, of which only one species was noted from Coombe. Three species previously recorded (Edwards, 1997) - *Kurzia pauciflora, Riccardia latifrons, Odontoschisma sphagni* - were either absent or overlooked in the experimental plots. A further ten liverwort species listed within the *Erica tetralix – Sphagnum compactum* typical subcommunity and the *Narthecium ossifragum – Sphagnum papillosum* valley mire community have not been recorded. In contrast, two lichens not listed within the NVC communities have been noted at Coombe Bog but were not recorded during the grazing experiment - *Cladonia strepsilis,* and *Cladonia verticillata* (B. Edwards, pers. com.).

Species richness at Coombe is inflated by the presence of tree seedlings not generally characteristic of these communities. With the exception of *Pinus sylvestris*, which increased in UG plots, frequency was too low to make any generalization about response to grazing treatment. Rodwell (1991) suggests that the wet heath and valley mire communities are likely to progress to woodland without grazing – evidently not an immediate problem at Coombe, although the increase in *P. sylvestris* cover suggests this could be a long-term problem if grazing does not prevent further growth in tree seedlings.

After three grazing seasons, cattle did appear to have had a limited impact on regenerating pine on humid and dry heath, and a similar response could be expected on wet heath. However, the differences were small, and unlikely to control population growth, although a longer tern study is needed to verify this.

Rose (1953) suggests that overgrazing on valley mires can result in changes towards poor fen communities. While there is currently no suggestion of this occurring at Coombe, if under ongoing grazing management the substantial increases in bare ground continue without a corresponding increase in colonization or regeneration, species richness may decline.

Considering the changes at Coombe Bog in terms of overall species richness is not currently helpful in guiding conservation management of the site. Species richness is intrinsically relatively low in these communities, and there is some reason to believe that even within this context the communities at Arne are naturally impoverished. However, responses to grazing management may evolve over many years (Hill *et al.*, 1992; Bullock *et al.*, 1994). Increases in colonisation of rare species may take a long time, while species' responses may differ between seasons (Bullock *et al.*, 2001). Rawes (1983) considers recovery following removal of sheep on heavily grazed upland heathland as 'fast', but this was within the context of a fifteen-year study. During this time no new species appeared, and colonisation of bare peat was slow. Clearly then, changes in response to grazing treatment at Coombe Bog may take substantially more than three years to manifest. A longer time span is needed to understand how the initial changes in overall species cover, sward structure and substrate characteristics found during the three-year experimental period may affect species composition. The conservation implications of the results of this study are discussed in Chapter 6.

Conclusions

- The introduction of cattle to Coombe Bog had a marked affect on both wet heath and valley mire;
- Changes in vegetation structure were apparent after the first grazing season, and differences were reinforced over the subsequent two seasons.
- Environmental variables measured showed a strong treatment effect, with both compaction and microtopography clearly responding to livestock presence after three grazing seasons;
- Plant species response to grazing at Coombe Bog appears slow. Cover of graminoids and ericoids was lower in cattle and deer-grazed plots, where bare ground cover was highest. There was some indication that this led to an increase in forbs. Valley mire *Sphagnum* species appeared to be susceptible to livestock presence;

- Deer grazing by itself had a limited impact and responses were rarely different to the ungrazed treatment. However, the plot design meant that deer grazing was lower in the deer-grazed plots than in the cattle and deer-grazed plots;
- Changes in species richness were slight, and are not considered to be currently useful in assessing the suitability of grazing as a management tool at Coombe Bog;
- However, the marked grazing effect on environmental variables and sward structure suggest that species composition will change over time.

4 Livestock effects on plant species of conservation interest

4.1 Introduction

Conservation action is generally targeted at habitats rather than at individual plant species (e.g. Sutherland & Hill, 1995). The exception is those species considered as being of particular conservation concern (e.g. www.ukbap.org.uk/plans). The potential effects of using livestock as a conservation management tool for wet heath and valley mire habitats is investigated experimentally in Chapter 3. This chapter, together with Chapter 5, investigates the impact of livestock presence on selected plant species. One of the selection criteria was that the species should be characteristic of wet heath or valley mire vegetation of high conservation interest. Therefore, information concerning these species can both provide useful information for the conservation of the individual plant species, and offer further guidance into the potential effects of livestock presence at the plant community level. This chapter therefore:

(i) Explores how seedling emergence by the selected species may be affected by livestock presence,

(ii) Describes the habitat of existing populations and links this to emergence requirements and the potential effects of livestock, and

(iii) Provides a review of current knowledge of the study species relevant to their conservation.

Regeneration from the buried seed pool is considered separately in Chapter 5.

The process of regeneration is a critical factor in plant community dynamics, and seedling establishment has assumed a central role in the debate concerning declining plant diversity (Grime & Hillier, 2000). Seedling emergence is important in determining population size in both perennials (e.g. Silvertown *et al.* 1993; Bullock, Clear Hill & Silvertown, 1994) and annuals (e.g. Silvertown *et al*, 1992). The importance of grazing in colonisation and gap dynamics has been shown in perennial grasslands (Silvertown & Smith, 1989; Bullock *et al*, 1994; Smith & Rushton, 1994; Bullock *et al*, 2001). It seems likely that grazing also plays an important role in heathland plant community dynamics through the creation of gaps providing suitable regeneration opportunities in dense perennial vegetation.

While the importance of regeneration in wet lowland heathland communities has not been studied, its relevance can be inferred from Byfield and Pearman's (1994) work on declining heath species (see Chapter 1). Byfield and Pearman (1994) used a suite of indicator species to show changes in composition over time in wet heath and valley mire habitat. One of their aims was to identify possible causes of changes in species abundance, where this occurred. Overall, they concluded that lack of disturbance (including livestock disturbance) appeared to be a key factor in the loss of species from many stands where they had previously been recorded. This suggests that the lack of gap creation and subsequent regeneration may have played an important role in the loss of diversity from heathland stands.

Livestock will also affect other plant life-cycle components, and effects on all components are briefly reviewed here before going on to consider the selected wet heath and valley mire species and their regeneration. Livestock may affect a plant directly, e.g. by eating it or trampling on it, or indirectly through habitat changes. While indirect effects are likely to be most important for species regeneration, both direct and indirect effects will impact later in plant life-cycles. The mechanisms by which such effects will occur have been extensively reviewed (e.g. Crawley, 1983; Hodgson & Illius, 1996; Crawley, 1997; Olff & Ritchie, 1998; Bullock *et al.*, 2001), and are summarised here (after Lake, Bullock & Hartley, 2001):

(i) **Feeding**: Livestock remove and eat leaves, stems, flowers and other plant parts. This can lead to injury or death of the plant, and also change plant structure, reduce height, reduce photosynthetic area, change plant chemistry, change growth patterns, change phenology, or increase susceptibility to disease or invertebrate herbivory (by weakening plant defences). Removal of plant parts will also create canopy, stem and root gaps (Bullock, 2000), releasing resources such as light and nutrients which may be then be exploited by other individuals and emerging seedlings.

(ii) **Trampling and poaching**: As with feeding, trampling by livestock causes damage to and loss of plant parts, with the same potential direct and indirect effects as listed above. Trampling returns litter and dead material to the soil surface, increasing litter depth, soil organic matter and nutrient status and the rate of nutrient cycling. It may also cause poaching by disturbing the soil surface, breaking up and destabilising soil structure, integrating litter into the soil, changing soil microtopography (e.g. creating small depressions), and compressing the underlying soil.

(iii) **Dung and urine deposition:** Livestock return some of the plant material they ingest in the form of dung and urine. This results in the redistribution of nutrients and organic matter gathered over wide areas into small discrete patches. Urine deposition is analogous to addition of inorganic nitrogen fertiliser and can stimulate plant growth (Jaramillo & Detling, 1992). This can increase plant nitrogen concentrations, leading to increased utilisation by livestock (e.g. Day & Detling, 1990). The acidity of urine can also damage plants (Steinauer & Collins, 1995), but dung is more damaging initially as it smothers plants, leading to death in some cases. Dung also contains toxins that can damage and kill plants (Malo & Suarez, 1995). However, dung has high phosphorus and nitrogen concentrations, which can stimulate plant growth as it breaks down.

(iv) **Dispersal:** Livestock can move plant seeds and other propagules (e.g. rhizome fragments) on their coats and hooves (Fischer *et al.*, 1996). Dung can also contain seeds that have been eaten and passed through the gut (Welch, 1984); thus seeds are deposited in a nutrient-rich, competitor-free substrate. In both cases, livestock can transport seeds distances and into areas not attainable by other dispersal modes.

4.2 Indicator species

Eight species were chosen from the list of 41 indicator species used by Byfield and Pearman (1994) to show the overall decline of wet heath and valley mire species in Dorset in extant heathland stands. These 41 species were all described as 'locally distributed due to exacting requirements' and were considered to be of particular nature conservation value. They were either (i) nationally rare species occurring in 15 or fewer 10km National Grid squares (Wigginton, 1999); (ii) nationally scarce species occurring in 16-100 10 km National Grid squares (Stewart *et al.*, 1994); or (iii) species rare in Dorset, occurring at three or fewer sites (Mahon & Pearman, 1993); or (iv) additional species of note that do not fall into any of the above categories but were nevertheless considered by the authors to be of particular conservation value. The species in this study were chosen from the above list in consultation with English Nature, Dorset Environmental Records Centre (DERC) and local field botanists according to the following criteria:

- Value as an indicator of good quality habitat and representative of other wet heath and valley mire species - a subjective assessment made by representatives of the above organisations;
- Intrinsic interest - conservation status as defined above combined with a subjective assessment of the conservation interest of the species in this context;
- Ease and feasibility of study - species should not be too rare, too obscure or with too complex a life cycle, current knowledge of the species should be sufficient to give a starting point for further work, but the species has not previously been intensively studied;
- Availability of study sites for each species;
- Anecdotal evidence of response to livestock grazing and or trampling.

The species chosen were allseed *Radiola linoides,* slender centaury *Cicendia filiformis,* pale butterwort *Pinguicula lusitanica,* great sundew *Drosera anglica,* petty whin *Genista anglica,* bog orchid *Hammarbya paludosa,* brown beak-sedge *Rhynchospora fusca* and marsh clubmoss *Lycopodiella inundata.* Rarer species can be difficult to work with, as populations are harder to find, and potentially damaging seed collection or experimental work have more serious consequences for populations, in some cases prohibitively so. However, it is inevitable that some species in habitats of conservation concern will be rare, and it was felt that the benefits of using the species selected by the above criteria outweighed the disadvantages connected with their scarcity.

Table 4.1 presents the nature conservation status of each species, together with its recorded decline in Dorset. Figures for the 'decline within surviving stands' (Table 4.1) are based on the presence or absence of each species within those stands surveyed by Good in the 1930s that were still considered to comprise heathland vegetation by Byfield and Pearman in the 1990s (Byfield & Pearman, 1994; also see Chapter 1).

Table 4.1 Conservation status and decline in Dorset of study species. NS – Nationally scarce (Wigginton, 1999); BAP – biodiversity action plan prepared for species (UK Steering Group, 1995); SoCI species of local conservation interest in Dorset (Byfield & Pearman, 1994) (see methods). ' % decline in surviving stands' - % decline (and sample size) for each species within stands surveyed in the 1930s still extant in the 1990s (excludes decline due to habitat destruction); 'Overall % decline in Dorset' - decline based on presence/absence within 1 km squares according to pre- and post-1980 data held by DERC (includes decline due to habitat destruction).

Species	**UK Status**	**% decline in Dorset in surviving stands (no.)**	**Overall % decline in Dorset (no.)**
Cicendia filiformis	NS	100 (9)	75.8 (29)
Drosera anglica	SoCI	12.5 (8)	25.8 (27)
Genista anglica	SoCI	70.6 (17)	43.4 (83)
Hammarbya paludosa	NS	50 (2)	52.9 (17)
Lycopodiella inundata	NS, BAP	84.6 (39)	58.8 (89)
Pinguicula lusitanica	SoCI	73.9 (23)	26.7 (86)
Radiola linoides	SoCI	94.7 (38)	35 (40)
Rhynchospora fusca	NS	68.4 (19)	24.6 (73)

This excluded the loss of species through gross habitat change, and reflects the reduction of suitability of the surviving stands for each species. A comparison of changes in occurrence at different scales in Dorset shows that, for six of the eight selected species, presence within surviving stands has reduced to a greater degree than the presence within all one km squares in the county. Recording at the level of the km square is, in this case, disguising the severity of the decline. Good's vegetation stands were generally sufficiently small and clustered for several to be included with a single one km square. For example, the 18 one km grid squares covering the Arne - Hartland - Stoborough heathland complex in Purbeck contain 67 of Good's (surviving) stands. Therefore, in theory, a species could be lost from 49 of these stands (73%) before the decline was noticeable at the one km level. In addition, the one km records include loss through gross habitat change. Byfield and Pearman's data suggest a sharp reduction in the suitability of the stands surveyed for most species (although see the discussion of their methodology in Chapter 1). However, *Hammarbya paludosa* shows a similar level of decline within stands as within one km squares, and *Drosera anglica* actually shows a higher degree of survival within stands than within the one km squares.

Current knowledge of each of the selected species is reviewed below. Information presented is extracted from ECOFLORA (http://beta.bids.ac.uk/ecoflora.html) and Stace (1991) unless otherwise stated, and is supplemented with observational data from this study in Dorset (1998-2000). Nomenclature is according to Stace (1991). Dorset distribution maps for each of the species can be found in Appendix VIII, and UK distribution maps in Appendix IX

Cicendia filiformis (L.) Delabre (Gentianaceae).
A small summer annual found on damp acidic substrates, largely restricted to south and west England and south-west Wales, with strongholds in the New Forest and the Lizard peninsula. The plant reaches between 10 and 120 mm in height, and produces tiny yellow flowers, which only open in direct sunlight. It may be extremely abundant in favourable years, but absent in others (Byfield, 1994). It may form mycorrhizal associations (Brouwer *et al.*, 2001).

Plate 4.1 Yellow centaury *Cicendia filiformis* growing on a track in Purbeck, Dorset.

In Dorset, it is found along tracks in or on the rims of seasonally inundated wheel ruts in very open vegetation, and has been observed in similar habitat in Co. Cork, Ireland (pers.obs.). In the New Forest, it is characteristic of the damp lawns that occur where high levels of grazing result in very short swards, and has been noted growing in livestock hoof prints (N. Sanderson, pers. com.). On the Lizard Peninsula in Cornwall, it has been reported from livestock 'pinch points' in gateways on suitable habitat (R. Fitzgerald, pers. com.). *C. filiformis* is adapted to low nitrogen and relatively high phosphorus availabilities (Brouwer *et al*., 2001). Brouwer *et al.* (2001) suggest that flooding and/or compaction reduce soil aeration and so reduce nitrogen mineralization and promote denitrification while promoting phosphorus availability, providing favourable conditions for *C. filiformis* .

In his *Flora of Dorsetshire,* Mansel-Pleydell (1875) recorded it as 'frequent in suitable places'. However, it has now been lost from all of the surviving sites where it was recorded by Good in the 1930s in Dorset, and is currently known from only six sites in the county. Together with *Radiola linoides* (see below) it forms part of a suite of species characterising lawns and greens that has declined more severely than those characterising other habitats associated with wet heathland (Byfield & Pearman, 1994). This suite of species is considered to be highly dependent on livestock grazing in the New Forest (Sanderson, 1994).

Drosera anglica Huds. (*Drosera longifolia*, L.) (Droseraceae).
A small carnivorous hemicryptophyte, growing to a height of between 10 and 18cm. It forms a basal leaf rosette covered in sticky hairs which trap insects. It is characteristic of areas with a water level permanently at the soil surface and a pH range of 3.6-7.6. Plants live for two to three years (Crowder *et al.,* 1990).

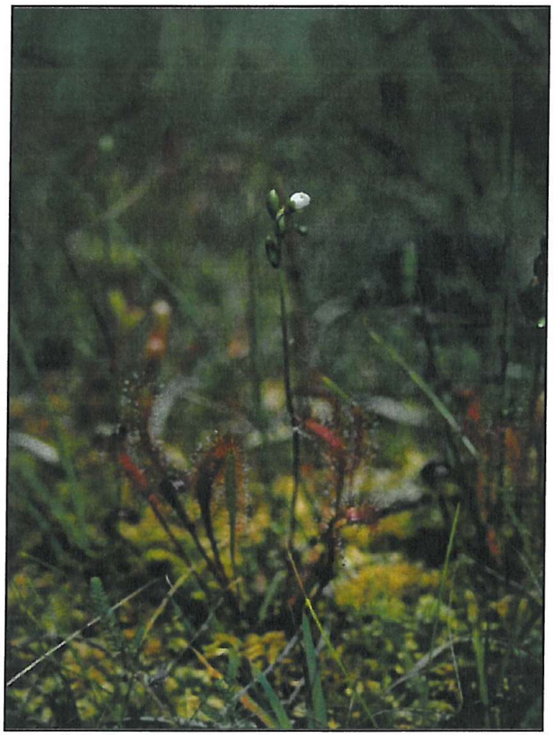


Figure 4.2 Great sundew *Drosera anglica*, Purbeck, Dorset.

On the Dorset heaths, it is confined to wetter, stable, and open mires where it grows on *Sphagnum* spp. in small patches of tens of individuals. This is in contrast to the more common *Drosera* species round-leaved sundew, *D. rotundifolia* and oblong-leaved sundew *D. intermedia,* which occur on bare peat and can be locally very abundant. Plants produce abundant seed, which is dispersed by wind, and they can also reproduce vegetatively from axillary and leaf buds, although most reproduction occurs by seed (Crowder *et al.,* 1990). Seeds may be dormant by maturity in late summer, and germinate after cold stratification (Baskin et al., 2001). Mature plants over-winter as buds.

This species has not declined to the same extent as most of the other indicator species. It has probably never been common in Dorset - Mansell-Pleydell described it as 'rare' in 1875. It was recorded from eight stands by Good, and was relocated in seven of these by Byfield and Pearman (1994). DERC holds current records for 20 1- km squares, compared with 27 from before 1980. This relatively small decline may be because it is only found in the wettest, most open mires, which *Molinia caerulea* is slower to dominate.

Genista anglica L. (Fabaceae).

A low straggling shrub, protected with sparse, spreading spines which are leafy when young. Stems are erect to spreading, from 10cm to 1m in height. It occurs in sandy places and peaty heaths and moors, in suitable places throughout most of the British Isles. It can form large shrubs with thick woody stems, such as in the New Forest, or be virtually prostrate, for example on coastal sites on the Lizard Peninsula in Cornwall (pers. obs.). Plants in Dorset are generally rather straggly, often growing in thick *Molinia caerulea* swards.

Genista anglica forms seed pods containing between one and 10 seeds, but may also propagate vegetatively through layering of stems (B. Edwards, pers. com.) and through shooting from the rootstock. It is considered to form a transient seed bank (Thompson *et al.,* 1997) The plant is generally scattered where present, although prostrate plants may form thick patches. It is probably frequently overlooked when not in flower. The species is reported to be avoided by cattle (Buttenschøn & Buttenschøn, 1982). Browsing was observed only once during this study, and the pony involved soon desisted.

Plate 4.3 Petty whin *Genista anglica* growing in *Molinia caerulea* dominated wet heath, Purbeck, Dorset

Byfield and Pearman recorded a decline of 70.6% in surviving stands for this species. However, their survey was carried out during the summer, when the plant is not in flower and is easily missed. Between 1998 and 2000, four of the 12 surviving stands where the species was reported extinct by Byfield and Pearman were relocated and searched by me, and three were found to support several woody, well-established plants. New records were also made at two further sites that were not surveyed by Good. Although Byfield and Pearman's figure may be an overestimate, *G. anglica* has clearly declined substantially since being described as 'locally frequent' by Mansel-Pleydell (1875). DERC holds current records from 47 one km squares, compared with 83 from before 1980.

Hammarbya paludosa (L.) Kuntze (Orchidaceae).

A diminutive orchid (3-12cm in height) found growing scattered in *Sphagnum* spp. or on peaty mud around pools in valley mires where there is some water flow (Porter, 1994). It is normally mycorrhizal, and found on moderately acidic substrates (pH 4.4 - 6.12) and has two or three pseudobulbs growing on a vertical rhizome (Delfage, 1994). It forms seed pods, although these were not found in Dorset during the period of this study.

Formerly scattered throughout much of the British Isles, it is now rare except in Dorset, Hampshire, Cumbria and central and north-west Scotland, and is considered Nationally

Scarce (Wigginton, 1999). However, it is likely to be under-recorded due to its small size and unobtrusive colouring. Byfield and Pearman (1994) recorded a 50% decline in surviving stands since the 1930s. Although this estimate was based on only two stands, the extent of the decline was corroborated by Bowen (2000) who recorded seven current and eight extinct sites for the species in Dorset. Like *D. anglica*, it has probably never been common, and was described as 'rare' by Mansel-Pleydell in 1875.

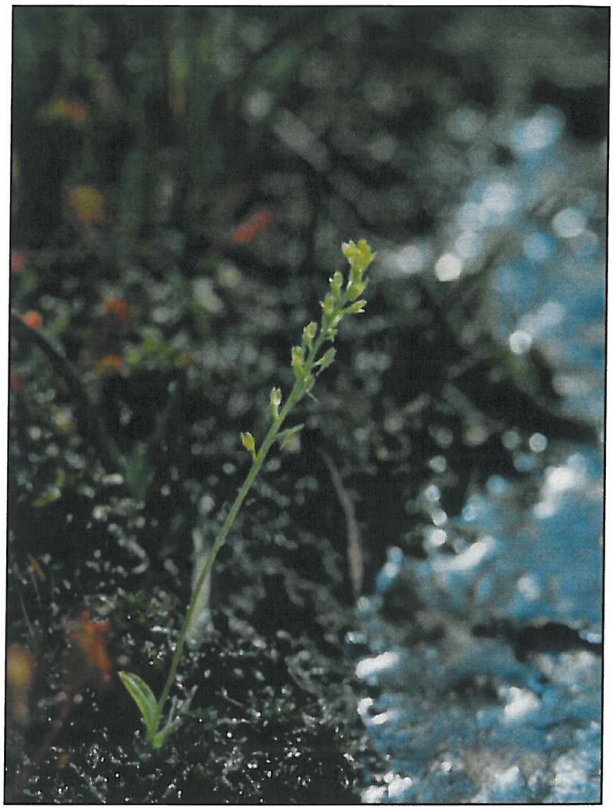


Plate 4.4 Bog orchid *Hammarbya paludosa* in a valley mire in Purbeck, Dorset

Lycopodiella inundata (L.) Holub (Lycopodiaceae).
A perennial clubmoss of bare peaty soil, usually associated with wet heath and mires and, in Scotland, with lake margins. It often grows in places where human activity provides disturbance and maintains areas of bare, seasonally flooded peat (UK Steering Group, 1995) and is considered most common where high grazing pressure results in the poaching of wet heath and mire surfaces (Headley, 1994). It has procumbent stems to 20 cm long, plus erect stems reaching to 10cm.

Headley (1994) describes elements of the plant's life cycle: branches remain evergreen for two years, then the clone fragments by the disintegration of older sections of the branches. It spreads relatively slowly at two to 10 cm each year.

Strobili produced in the summer mature in autumn, and spores may be dispersed by air, or by water within intact sporangia when the plant becomes submerged. It is considered to have a poor long-range dispersal ability (Stewart *et al.*, 2000).

Plate 4.5 Marsh clubmoss *Lycopodiella inundata*

Lycopodiella inundata is declining in most of Europe and is a Priority Species under the UK government's Biodiversity Action Plan (UK Steering Group, 1995) and is considered Nationally Scarce (Wigginton, 1999). The main threats to the species in the UK are thought to be the encroachment of rank vegetation through lack of heathland management, and from heath fires (UK Steering Group, 1995; Edwards, 1998). Formerly scattered almost throughout Britain and Ireland, remaining strongholds are Dorset, Hampshire and Scotland. Mansell-Pleydel (1875) described it as 'frequent' on the Dorset heaths, and Good recorded it from 48 sites. Of the 39 surviving sites, it was only relocated in six, representing a decline of 84.6 % (Byfield & Pearman, 1994).

Pinguicula lusitanica L. (Lentibulariaceae).

A small, carnivorous hemicryptophyte found on bare ground on organic or flushed mineral soils, occasionally on *Sphagnum* carpets, generally within a pH range of 5.1-7.2. It grows between 3 and 15 cm in height, on runnels in mires, wet heath and ditch edges.

Plate 4.6 Pale butterwort *Pinguicula lusitanica.*

P. lusitanica is reported to become locally very abundant following disturbance (C. Chatters, pers. comm.). Although perennial, it can behave as an annual, and produces a large amount of seed (D'Amato, 1998). Seeds germinate in summer or autumn and plants over-winter as a small rosette. In common with other carnivorous plants (Aldenius *et al.*, 1983), it has a weakly developed root system, and is easily dislodged and therefore may be vulnerable to livestock trampling. Populations can be scattered, or in small patches of 100s of plants where it has recently become established. The species has declined substantially in Dorset. Byfield and Pearman's 1994 survey shows a decline of 73% since the 1930s within extant stands. However, it is less restricted in its Dorset distribution than most of the species discussed here.

Radiola linoides Roth (Linaceae).

A tiny, much branched summer annual found on seasonally damp, bare peaty or sandy acid soils. It can form very large populations, which may shift from year to year according to conditions (C. Chatter, pers. com.). In the New Forest, the largest colonies are on very short lawns, ditch edges or in open trackways across heaths on open ground, which is damp in winter and parched in summer. In Dorset, it is mainly found on tracks through short acid grassland or wet heath, often in seasonally wet ruts.

It is known to form a buried seed bank where it occurs in dune slacks (Bekker *et al.*, 1999), remaining at low but detectable numbers even after the vegetation has developed beyond the early successional phase preferred by this species.

Plate 4.7 Allseed in *Radiola linoides* growing on a track side in Purbeck, Dorset

Its distribution is centred on southern England, with scattered localities north to Scotland where it is strictly coastal. In Dorset, it has declined steeply, and by 1994 was only found in two of the 38 surviving stands where Good had recorded it in the 1930s, a decline of 94.7%. This is considered to be due to habitat loss and to changes in land use, particularly the cessation of grazing (Byfield & Pearman, 1994; Edwards, 1999).

Rhynchospora fusca (L.) Aiton f. (Cyperaceae)

A perennial sedge found on wet heath and the margins of valley mires, but generally avoiding the wettest areas. On bare peat, it is often the only species present and gradually gives way to other plants, particularly *Molinia caerulea* (Pearman, 1994; Sansen & Koedam, 1996). It spreads actively by rhizomes and readily colonises bare peat. It has successfully re-established itself at a heathland site in Dorset following the removal of the invading vegetation and top two cm of peat (R. Brunt, pers. com.), suggesting propagation by seed. Similarly, at a site in Wales, a new population appeared following disturbance by heavy machinery and a rise in the level of the water table (Hale & Bailey, 1995). Piek (1998) reported an increase in *R. fusca* following the introduction of cattle grazing on heathland sites in the Netherlands. Raimund and

Urban (1999) found plants established by seed and often spread rapidly through clonal growth following experimental sod-cutting. It grows to between 10 and 30 cm in height. Seeds may remain on the plant until the following spring (Pearman, 1994). The species forms a mycorrhizal association that enables it to uptake nutrients in nutrient-poor soils (Sansen & Koedam, 1996).

Plate 4.8 Brown beak-sedge *Rhynchospora fusca*, Purbeck, Dorset

In Dorset, one *Rhynchospora fusca* population is reported to have been adversely affected by pony grazing (Pickess, 1998). However, closer examination of this stand suggested that the plants may not have flowered, possibly due to a reduction in habitat quality, rather than the flower heads having been grazed off. There was no indication of grazing damage at other populations on grazed sites. Ponies have, however, been observed grazing *R. alba* (Chapter 2), and may graze *R. fusca* if it is encountered.

The species is declining in southern Britain, and Dorset is now its national stronghold. However, in Dorset only six of the surviving 19 stands surveyed in the 1930s still support the species (Byfield & Pearman, 1994). Cessation of grazing is now thought to be its greatest threat (Pearman, 1994).

4.3 Methods

4.3.1 SELECTION OF STUDY SITES

Locations for each of the selected species were found using data from DERC, plus records from local naturalists and site managers. Additional locations were identified by searching putative suitable habitat.

Table 4.2 Study sites for selected indicator species of wet heath and valley mire in Purbeck, Dorset. Some species occurred at sites surveyed for other species – these sites are included in italics for secondary species, as habitat measurements were based around locations for the primary species.

Species	**Location**	**Grid ref.**
Cicendia filiformis	Arne Triangle	SY 950866
	Great Ovens Hill	SY 921904
	Thrashers Heath track	SY 971852
	(plus Stoborough Heath see *RL)*	
Drosera anglica	Slepe Heath	SY 941857
	Spur Bog	SZ 027853
	Turford Bog	SY 967888
Genista anglica	Bank Gate Common	SY 957865
	Jubilee Bridge, Middlebere Heath	SY 947848
	Orchard Mire, Povington	SY 947848
Hammarbya paludosa	Slepe Heath	SY 941859
	Three Barrows Mire	SY 941859
	Winfrith Heath	SY 803866
Lycopodiella inundata	Slepe Heath	SY 941859
	Spur Bog	SZ 027853
	Stoborough Heath	SY 934834
Pinguicula lusitania	Arne Triangle	SY 856865
	Spur Bog	SZ 027853
	Stoborough Heath	SY 927847
	(plus Slepe Heath, Three Barrows see *CF)*	
Radiola linoides	Grange Barn Field	SY 922853
	Slepe Heath	SY 944863
	Three Barrows Heath	SY 936852
	(plus all locations for CF)	
Rhynchospora fusca	Stoborough Heath	SY 922855
	Tadnoll Heath	SY 792874
	Three Barrows Mire	SY 941859

4.3.2 HABITAT AND LIFE HISTORY

For each species, the approximate dates of emergence, flowering and seed dispersal were noted between 1998 and 2000. Habitat characteristics for each location were recorded as follows in July-August 2000:

- Vegetation composition within ten $20cm^2$ quadrats, each centred on an individual plant ($1m^2$ quadrats in the case of *G. anglica*, because it is a bigger plant);
- The percentage cover of bare ground and surface water within each vegetation quadrat. Where surface water was not present, a subjective assessment of the
- humidity of the site was made based on the presence or absence of standing water throughout the year. Dip wells were installed to measure ground water levels, but they provided no data because they silted up rapidly;
- The surface microtopography, measured using the technique described in Chapter 3.
- Soil compaction was measured as penetration resistance at 20 random points within the population using the BJPS method (see Chapter 3);
- Sward height measured by drop disk (see Chapter 3);
- The degree of shading of 20 individuals at each site was assessed by recording the number of times a vertical rod (diameter 3mm) held directly above the plant was touched by vegetation (with the exception of *G. anglica*, as all plants of this species were growing through dense *Molinia caerulea* tussocks). This was directly comparable to the method used for assessing sward structure in Chapter 3.

4.3.3 SEEDLING EMERGENCE EXPERIMENTS

Seedling emergence experiments were designed to provide information on how the establishment of each species might be influenced by livestock presence, in particular through effects on canopy cover and water level. Seeds of *Cicendia filiformis, Drosera anglica, Genista anglica, Pinguicula lusitanica* and *Radiola linoides* were collected each year between 1998 and 2000 and stored at 4°C in dark conditions until used. Seeds of *Radiola linoides* and *Cicendia filiformis* were not collected in 2000. *C. filiformis* populations appeared to be declining and therefore further seed collection was not permitted. *R. linoides* plants in the main two populations died from drought in July before seed could be collected. *Hammarbya paludosa, Rhynchospora fusca* and *Lycopodiella inundata* proved unsuitable for emergence experiments due to the difficulty in obtaining seeds/spores. These problems illustrate the constraints on studying rare and uncommon species.

Experiment 1

Preliminary germination trials were carried out in 1998 using seed of *Cicendia filiformis*, *Genista anglica*, *Pinguicula lusitanica* and *Radiola linoides*, kindly supplied by the Millennium Seedbank (location of source populations unknown) to establish basic requirements and a suitable methodology for further work. Incubator germination of five replicates of 10 seeds (half of which had been stored at 0°C for two months) with varying light and temperature (24 hour cycle, 12 hours at 20°C in light conditions alternating with 10°C in dark conditions) resulted in zero germination. Therefore, more natural conditions were used for subsequent experiments, which also included *Drosera anglica.*

Treatments included pre-treatments to the seeds, and subsequent growing treatments (Table 4.3). Seed treatments were:

(i) Freezing (at 0°C) for one month to break possible dormancy mechanisms;

(ii) Soaking for one month to mimic natural conditions, as the sites for these species are generally water-logged over the winter months. This was carried out for the species of wetter ground *Pinguicula lusitanica, Drosera anglica* and *Cicendia filiformis* only. Soaking water was collected from a valley mire;

(iii) Both soaking and freezing;

(iv) An untreated control without soaking or freezing;

In addition, all *Genista anglica* peas were scarified with fine grain sandpaper for three minutes. This is a standard method to increase water-permeability and thus facilitate germination in seeds with thick coats (Baskin & Baskin 2001, p104).

Growing treatments were:

(i) Shading - commercial greenhouse shading mesh was used to mimic the presence of canopy shade;

(ii) Water level - seed-trays were placed in un-perforated trays in which tap water was maintained constantly at a level just below the peat surface to mimic the increase in surface water brought about by trampling;

(iii) An untreated control without shading or raised water level.

Treated seed was spread in seed-trays (160mm x 204mm x 50mm) prepared with a 2cm peat[1] layer and arranged randomly on capillary matting on glasshouse benches. Peat for *Cicendia filiformis* and *Radiola linoides* trays was mixed with an equal amount of sand to approximate more closely to the substrate on which the species naturally occur. Each combination of seed and growing treatment was replicated three times, with the exception of *Genista anglica* and *Radiola linoides*, for which only one replicate of 25 seeds per treatment was possible. One hundred *Cicendia filiformis* and *Drosera anglica* seeds and 50 *Pinguicula lusitanica* seeds were used for each replicate. Water trays were topped up every other day and the capillary matting kept constantly moist using rainwater, as tap water had a relatively high pH (7.3) compared to that found in valley mires and wet heath (pH range 3-5, Rodwell, 1991). As the seeds of the majority of the species are very small, and germination is difficult to observe, emergence needed to take place before they could be counted. Planting was carried out in mid-April 1999, and counts of emerged seedlings were carried out weekly until July 1999.

Table 4.3 Seed and growing treatments used in Experiment 1 and 2.

Seed treatment		**Growing treatment**	
Frozen	F	Shaded	SH
Soaked	S	Wet	W
Frozen + soaked	FS	No treatment	O
No treatment	O		

Experiment 2

Based on the results from Experiment 1, further trials were carried out in 2001 for those species for which additional seed could be obtained. The aim was to differentiate between the effects of neutral shading, and the increase in humidity caused by the shading mesh, take into account the lowered red:far red ratio caused by lead shading also to clarify the interaction between seed and growing treatments

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[1] In accordance with RSPB policy a peat substitute would have been used . However, split peat bales found in a skip at ITE Furzebrook were saved and used.

where possible. The species were *Genista anglica* (using seed obtained from the Lizard Peninsular, Cornwall), *Drosera anglica* and *Pinguicula lusitanica.* All *P. lusitanica* and *D. anglica* trays were placed in water as before, and *G. anglica* trays were placed directly on capillary matting which was kept moist at all times. All *G. anglica* seeds and half each of *P. lusitanica* and *D. anglica* seeds were frozen. Green-shade (as opposed to the neutral shade created in Experiment 1) was created using dark green coloured cellophane (to absorb red light) stretched over frames supported five cm above the top of the trays to facilitate air circulation. Un-shaded trays were covered similarly with clear cellophane. Trays were placed on the levelled floor of a polytunnel. For this experiment, seeds were planted in August, approximately the time when seed would be dispersed in natural populations. Each combination of treatments was replicated ten times. One hundred *D. anglica*, 60 *G. anglica* and 50 *P. lusitanica* seeds were used per replicate.

Experiment 3

To validate results from the glasshouse experiments in the context of the effects of livestock presence, experiments were repeated in the field in 2000 for *Drosera anglica, Pinguicula lusitanica, Cicendia filiformis, and Genista anglica* (the species for which sufficient seed was available). Suitable habitat for each species was selected using information collected from sites where the species were present. Experiments were replicated five times per site at three sites for *D. anglica* and *P. lusitanica*, and six times at two sites for *C. filiformis.* Each experiment included three treatments:

(i) 'Grazing' - canopy removal carried out by clipping all vegetation back to the substrate surface using garden shears

(ii) 'Trampling' carried out using an artificial cow hoof. Each plot received 30 'trample s' from a cow hoof moulded in concrete and set on a spade handle swung from shoulder height (weight 5kg);

(iii) An untreated control.

Treatments were laid out in a randomised block design, and each treatment covered one square metre. 100 seeds were scattered over the central 20 cm^2 of each treatment plot. Seeds of *D. anglica* and *P. lusitanica* were not pre-soaked as the sites were considered sufficiently wet for this to occur naturally when the seeds were planted in mid-April. Since the surface water level on mires responds rapidly to rainfall, plots were surrounded by fine mesh to prevent seeds from washing out of the plots, while allowing water through. These were sunk approximately six centimetres below the surface and allowed to protrude by the same amount above. *C. filiformis* was not sown until mid-May, as the sites were flooded until this time. Consequently, *C. filiformis* seeds were soaked in glass pots using flood-water from the sites, and left in a shaded position outside until planting. All plots were checked approximately weekly until emergence appeared to have ceased, and the number of seedlings counted. The last census was carried out in October, 2000.

4.3.4 DATA ANALYSIS

Germination Experiments

Percentage emergence under different treatments was arcsine square-root transformed and compared using general linear model (GLM) and Tukey pairwise comparison tests (Minitab, 1998) for glasshouse data in Experiments 1 and 2. Two-way ANOVA was used for *Radiola linoides* and *Genista anglica* data from Experiment 1, as the lack of replication prohibited tests of interactions between the factors. Large numbers of zero values in the field data for Experiment 3 meant normality of residuals could not be achieved even by transformation. Therefore, the untransformed data was subjected to a non-parametric Friedman test (Minitab, 1998).

Habitat

Quadrat data were compared to diagnostic data for National Vegetation Classification (NVC) communities (Rodwell, 1991) using MATCH (1999) (see Chapter 3, Methods). For each species, data from different sites were pooled unless sites were clearly referable to different NVC communities, in which case data were analysed separately, or pooled only with similar sites. Microtopography, compaction, bare ground, surface water, sward height and shading data were averaged for each species.

4.4 Results

4.4.1 EMERGENCE EXPERIMENTS

Experiment 1

Seed treatments had a significant effect on *Cicendia filiformis*, with a smaller proportion of untreated seeds emerging than either soaked, frozen, or frozen plus soaked (Table 4.4, Figure 4.1). There was no clear pattern of response to seed treatment in *Pinguicula lusitanica*, but freezing appeared to increase emergence in *Genista anglica*, while freezing appeared to decrease, and soaking increase, emergence in *Drosera anglica,* although these results were not statistically significant (Table 4.4, Figure 4.2)

Emergence was significantly greater in wet treatments for *Drosera anglica, Pinguicula lusitanica* and *Cicendia filiformis.* Emergence appeared lower in wet treatments for *Radiola linoides* and *Genista anglica*, although these results were not statistically significant. Emergence was significantly higher in shaded than un-shaded treatments in *C. filiformis*, a trend that was also seen in *D. anglica*, and more noticeably in *G. anglica*, although again this was not statistically significant (see Table 4.4).

A significant interaction was found between seed and growing treatments for *Cicendia filiformis* (see Figure 4.1). Under wet growing conditions, freezing seeds clearly increased emergence, and soaking seeds decreased it relative to shaded conditions. However, under the other treatments freezing resulted in a lower (zero) emergence than soaking.

Table 4.4 F-values and P-values for emergence of wet heath and valley mire species under different seed and growing treatments (*P<0.05, **P<0.01, *** P<0.001, NS – not significant). Results from Tukey multiple comparison test are given where differences were significant. Total df – total degrees of freedom .*CF – Cicendia filiformis, DA – Drosera anglica, GA – Genista anglica, PL – Pinguicula lusitanica, RL – Radiola linoides.* F – frozen, S - soaked, SH – shaded, W – wet, O - control

Species	**Total df**	**Seed treatments**			**Emergence treatments**			**Seed*Grow interaction**
		F value	Tukey test		F value	Tukey test		
CF	35	18.17***	FS > O F > O S > O	6.11*** 5.93*** 6.03**	36.73***	W > O SH > O W > SH	8.54*** 6.60** 4.87***	17.59***
DA	35	2.93 ^{NS}	-		3.66*	W > O	2.46*	0.32 ^{NS}
GA	5	12.65 ^{NS}	-		12.96 ^{NS}	-		-
PL	35	0.03 ^{NS}	-		5.06*	W > O W > SH	2.76* 2.76*	0.03 ^{NS}
RL	5	0.00 ^{NS}	-		3.21 ^{NS}	-		-

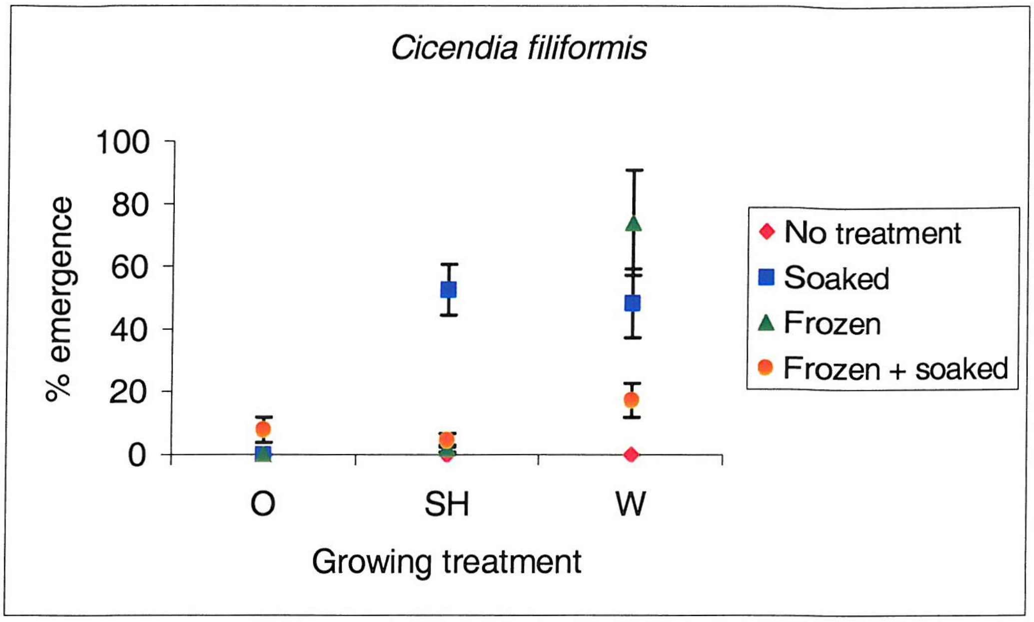

Figure 4.1 Mean (+/- standard error) % emergence of *Cicendia filiformis* seed under different growing treatments in Experiment 1. O – no growing treatment, SH – shaded, W – wet.

Experiment 2

The effect of green-shading was not found to be significant for any of the three species Seed soaking increased emergence for *Drosera anglica* and *Pinguicula lusitanica* (Table 4.5).

Table 4.5 F-values and P-values for emergence of wet heath and valley mire species under different seed and green-shade treatments in Experiment 2 (*P<0.05, **P<0.01, *** P<0.001, NS – not significant). Results from Tukey multiple comparison test are given where differences were significant. Df. – total degrees of freedom. *DA – Drosera anglica, GA – Genista anglica, PL – Pinguicula lusitanica..*

Species	**Total df**	**Seed treatments**			**Shade**	**Seed*Shade interaction**
		F value	Tukey test		F value	
DA	39	77.06***	S > O S > F FS > O FS > F	10.08*** 9.87*** 11.53*** 11.32***	0.55^{NS}	1.05^{NS}
PL	39	33.04***	O > F O > FS O > S S > FS F >FS	5.99*** 9.86*** 4.89*** 5.02*** 3.86***	0.05^{NS}	4.90**
GA	19	0.00^{NS}			1.51^{NS}	0.02^{NS}

Seed-soaking significantly increased the emergence of *Drosera anglica* (Figure 4.2). Freezing seeds also very slightly reduced emergence. Green-shading decreased emergence, but not significantly so, and there was no apparent interaction between the effects of seed treatments and that of shading.

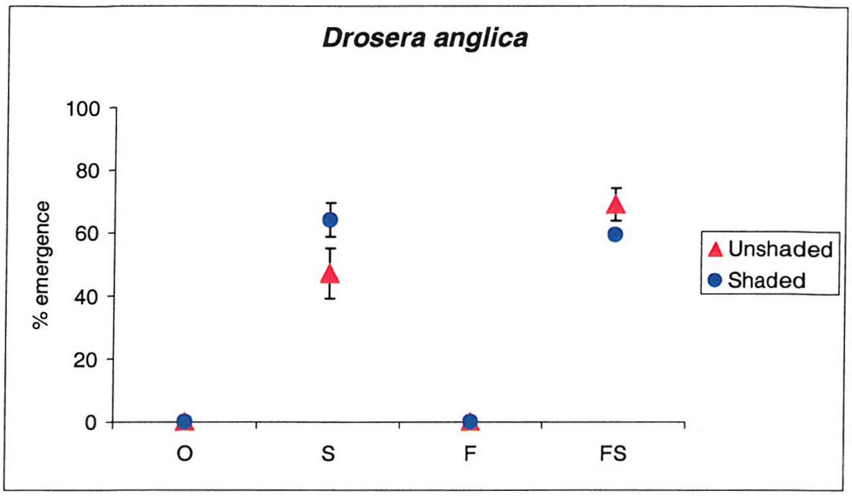

Figure 4.2 Mean (+/- standard error) % emergence of *Drosera anglica* seed under different growing treatments in Experiment 2. O - no seed treatment, S - soaked, F - frozen, FS - Frozen + soaked.

In contrast, for *Pinguicula lusitanica,* untreated seeds showed significantly higher levels of emergence, and both soaking and freezing alone resulted in a greater percentage emergence than the combined freezing and soaking treatment. The slightly higher emergence in un-shaded trays was not significant, but a significant interaction between green-shading and seed treatment was apparent (Figure 4.3). Green-shading clearly decreased emergence for untreated seeds.

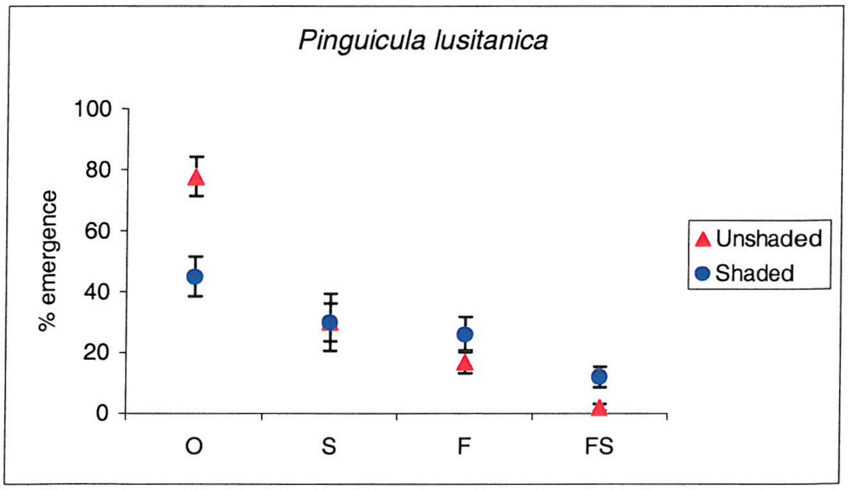

Figure 4.3. Mean (+/- standard error) % emergence of *Pinguicula lusitanica* seed under different growing treatments in Experiment 2. O - no seed treatment, S - soaked, F - frozen, FS - Frozen + soaked.

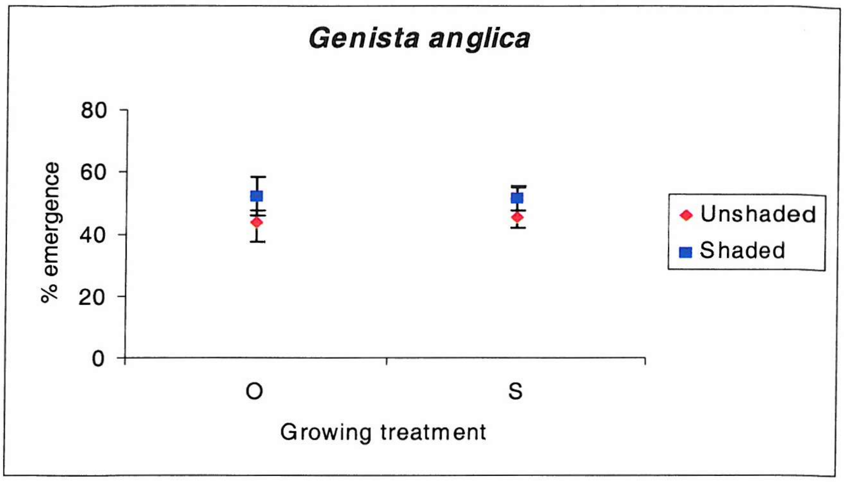

Figure 4.4 Mean (+/- standard error) % emergence of *Genista anglica* seed under different growing treatments in Experiment 2. O – no growing treatment, S - soaked

Seed-soaking did not influence final percentage emergence in *Genista anglica* (Figure 4.4). Soaked seeds did, however, emerge more quickly than un-soaked seeds, and seeds in shaded conditions emerged more quickly than those in un-shaded conditions This was tested by comparing the emergence rate under each treatment. Germination rate can be defined as the reciprocal of the time until 50% of seeds have germinated ($1/T^{50}$) (Probert, 2000). Due to the low numbers of seedlings of some species germinating, the rate used here was the reciprocal of the time until 50% of the total number of seedlings that emerged had germinated. The emergence rate defined in this way was higher for soaked seeds (represented by solid lines in Figure 4.5) than un-soaked seeds (GLM, F = 8.21*) and greater under shaded treatments (points represented by triangles in Figure 4.5) than shaded treatments (GLM, F = 5.28*) The rate of emergence of soaked seeds slowed after day 29, after which emergence rates were more comparable between treatments.

Drosera anglica showed a similar pattern, with soaked seed emerging first (Figure 4.6). Freezing did not appear to have an effect on the rate of *Drosera anglica* emergence - frozen plus soaked seed emerged at the same rate as soaked seed. Shading had no effect on emergence rate.

Although soaked *Pinguicula lusitanica* seed emerged before other seed treatments, this was not significant using T^{50}. The shape of the emergence curves in Figure 4.7 suggest that a lower percentage may be a more appropriate measure in this case. Soaked seeds were found to emerge at a faster rate than frozen and frozen plus soaked, but not untreated seed using T^{25}. After day 22 untreated seed emerged at a faster rate than all other treatments.

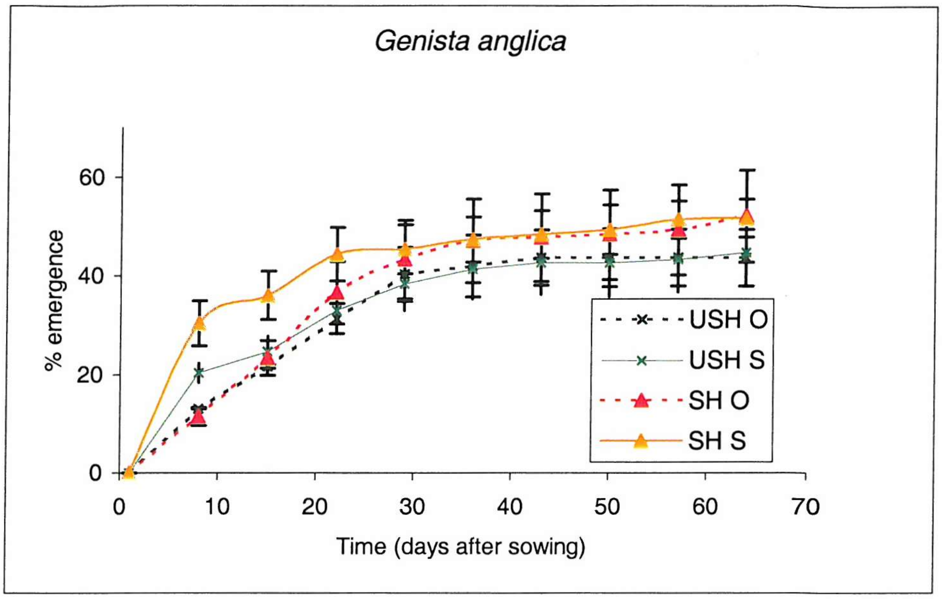

Figure 4.5. Emergence rate of *Genista anglica* under different treatments over time. USH – un-shaded, SH – shaded, O – no seed treatment, S – soaked.

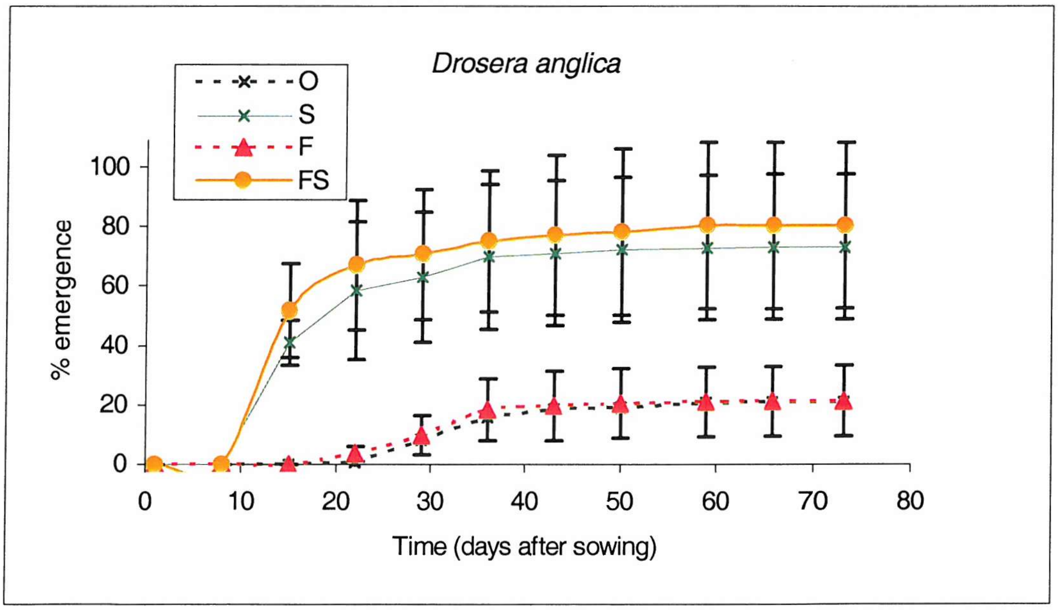

Figure 4.6 Emergence of *Drosera anglica* under different treatments over time. O – no seed treatment, S – soaked, F – frozen, FS – frozen plus soaked. Dotted lines represent soaked treatments. GLM F=44.96; FS >F, T = 7.81***; S>F Y= T = 6.35***; FS>O, T = 9.73***; S>O, T = 8.25***.

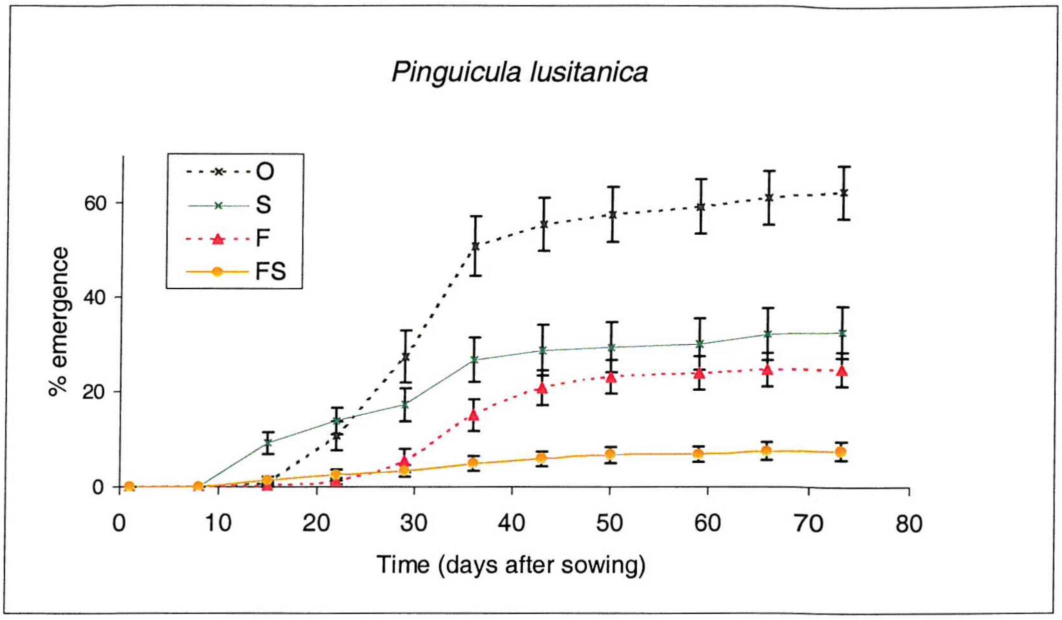

Figure 4.7. Emergence of *Pinguicula lusitanica* under different treatments over time. O – no seed treatment, S – soaked, F – frozen, FS – frozen plus soaked. Dotted lines represent soaked treatments. GLM, F = 10.04***; S-F, T = 3.42**; S-FS, T=5.29***

Experiment 3

Treatment had a highly significant effect on seed emergence in the field (see Table 4.6, Figure 4.8). *Genista anglica* and *Pinguicula lusitanica* only emerged in 'trampled' plots. *Drosera anglica* emergence was significantly higher in 'trampled' plots than 'grazed' plots, and did not occur in control plots. The difference in *D. anglica* emergence between grazed and control plots was significant once trampling was removed from the test. Blocks were taken account in a Friedman test (adjusted for ties), but could not be nested within site in this test. No *P. lusitanica* seedlings emerged under any treatment at one site.

Table 4.6 Medians, sum of ranks and S-values for emergence of wet heath and valley mire species under different field treatments in Experiment 3 (*P<0.05, **P<0.01, *** P<0.001, NS – not significant). *CF – Cicendia filiformis, DA – Drosera anglica, GA – Genista anglica, PL – Pinguicula lusitanica.*

Species	**Trampled**		**Grazed**		**Control**		**S value**
	Median	Sum of ranks	Median	Sum of ranks	Median	Sum of ranks	
CF	0	-	0	-	0	-	-
DA	21	45.0	0	24.5	0	20.5	28.20***
	-	-	0	24.5	0	20.5	4.00*
GA	3	45	0	22.5	0	22.5	30.00***
PL	1	38.0	0	26.0	0	26.0	16.00***

A number of *Cicendia filiformis* seeds germinated under water within the soaking pots before water levels in the field had subsided sufficiently for seeds to be sown (Figure

4.8). None of the remaining seed, which was subsequently sown, was observed to emerge. Significantly higher numbers of unfrozen seed germinated within the soaking pots than frozen seed (Mann Whitney, W = 233.0, P <0.0001). This supports the trend seen in Figure 4.2, where a higher number of soaked seed germinated than did frozen plus soaked seed.

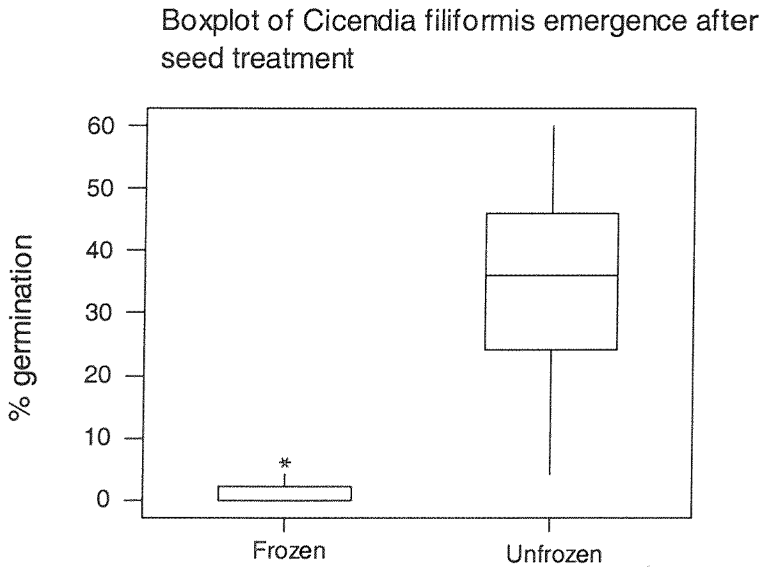

Figure 4.8 % Emergence of soaking *Cicendia filiformis* seeds under different freeze treatments. Boxes show interquartile range, whiskers show confidence limits and horizontal line shows median.

4.4.2 HABITAT DESCRIPTIONS

Data collected for each species are summarised in Table 4.7a & b. Four NVC communities were found: M16 *Erica tetralix – Sphagnum compactum* wet heath, M21 *Narthecium ossifragum – Sphagnum papillosum* mire, M25 *Molinia caerulea – Potentilla erecta* mire and M1 *Sphagnum auriculatum* bog pool. Three M16 *Erica tetralix – Sphagnum compactum* sub-communities were identified: the M16a typical sub-community, the M16b *Succisa Pratensis – Carex panicea* sub-community and the M61d *Juncus squarrosus - Dicranum scoparium* sub-community. Further subcommunities identified were the M21a *Rhynchospora alba – Sphagnum auriculatum* subcommunity and the M25a *Erica tetralix* subcommunitiy. NVC community alone was not sufficient to describe the communities each species was found in, and MATCH scores varied between 27.4 – 63.3. Therefore, three habitat categories were identified, based on an assessment of the level and type of disturbance together with the NVC descriptions.

These were:

(A) Disturbed tracks, paths or ditch edges within a wet heath context (supporting *Cicendia filiformis, Lycopodiella inundata, Pinguicula lusitanica, Radiola linoides*);

(B) Grazed valley mire (supporting *Drosera anglica, Hammarbya paludosa, Lycopodiella inundata, Pinguicula lusitanica, Rhynchospora fusca*);

(C) Grazed wet heath (supporting *Genista anglica*)

The NVC and environmental descriptions for each species presented Table 4.7a and b are used in Section 4.5 to describe characteristic sites where the species were found during this study and integrated with knowledge of how factors related to livestock presence may affect their emergence. The aim is not to make comparisons between species sites or categories, or to compare sites with those where the species have become extinct, but to present descriptive data for sites that support the target species.

Table 4.8 summarises the phenological observations made between 1998 and 2000 for each species. *Cicendia filiformis* and *Radiola linoides* germinated in late spring, and were observed in flower through to October, with occasional individuals of C.. *filiformis* in flower in early November. *R. linoides* emerged a week or two earlier at its drier sites, with both species emerging at wetter sites once the surface water had dried out in mid to late May. A second, much smaller, flush of emergence was observed for both species throughout September. Growth of both *Drosera anglica* and *Pinguicula lusitanica* recommenced in early April. Seedlings of *D. anglica* appeared to germinate in the spring, while seedlings of *P. lusitanica* were observed in September. Although all located bushes of *Genista anglica* were in flower between by mid-April, and early June, very few seed pods were found. However, vegetative spread was apparent at all sites. *Hammarbya paludosa* was extremely difficult to find when not in flower, and reliable emergence dates were not recorded, however, it was consistently found in flower by mid-August. No seed pods or apparent juveniles were found, although bulbils were observed each year. Non-flowering tubers were occasionally observed.

Table 4.7a Vegetation and environmental characteristics for each species (see text for explanations of measurements). For each NVC community type identified, the number of sites within each is listed under 'species'. Data presented are Min-max, mean (st. dev.). The median replaces mean for shade and cover of bare ground and water. *CF – Cicendia filiformis, DA – Drosera anglica, GA – Genista anglica, HP – Hammarbya paludosa, LI – Lycopodiella inundata, PL – Pinguicula lusitanica, RL – Radiola linoides , RF – Rhynchospora fusca.*

Species (No. of sites)	**NVC Community** and category	**MATCH score**	**Species richness** range, mean	**Bare ground** % cover range, median	**Water** % cover range, median	**Sward height** (mm) range, mean (st.dev)	**Shade** range, median
CF (3)	M25a (A)	31.7	5-13, 8	4-7, 35.5	0 (winter flooded)	10-171, 33.8 (28.7)	0-3 0
DA (3)	M21a (B)	63.3	8-24, 14	0	0-75, 0	20-192, 84 (30.4)	0-2, 0
GA (1)	M16b (C)	55.0	4-10, 5.8	0-10, 0	0 (humid thru'out yr)	126-573, 289 (97)	-
(2)	M25a (C)		7-22, 15	0-3.0, 0	0 (humid thru'out yr)	42-612, 241.1 (138.5)	-
HP (3)	M21a (B)	61.6	6-11, 8.2	0-5, 0	0-70, 18	19-212, 80.85 (47.5)	0
LI (1)	M16d (A)	51.4	4-8, 5.9	35-80, 60	0 (winter flooded)	16-6, 40.2 (15.0)	0-3, 1
(2)	M21a (B)	55.6	8-10, 8.8	0-60, 0	0-35, 7.5	21-6, 37.7 (11.7)	0-2, 0
PL (1)	M16a (A)	43.3	4-9, 6.3	0-80, 40	0 (water-logged thru'out yr)	-	0
(2)	M21a (B)	56.1	4-11, 7.3	0-47, 0	0 (humid thru'out yr.)	25-115, 62.9 (20)	0-4, 1
RL (4)	M25	(see CF)					-
(2)	M25a (A)	39.7	5-13, 8	0-20, 0	0	92-38, 56.5 (14.2)	0-4, 2
RF (1)	M1 (B)	36.7	5-9, 7	0-10, 0	0-90, 65)	50-181, 119.0 (32.2)	0
(2)	M16b (A)	27.4	2-7, 4.3	3 -65, 47.5	0 (drying out)	11-51, 28.2 (8.1)	0

Table 4.7b Environmental characteristics for each species (see text for explanations of measurements).

Species	**Substrate**	**Disturbance**	**Compaction** range, median	**Microtopographical variation** range, median
CF	clay-sand	Tracks	81-109, 95	5.0-3301, 168.4
DA	peat	Cattle & pony/deer grazed	121-442, 230	5.0-110.5, 44.5
GA	Peat	Deer grazed	102-247, 143	29.0-1422.0, 276.3 (all sites)
		Cattle Grazed	106-195, 135	
HP	Peat	Cattle & pony grazed	103-232, 156	10-253.3, 20.4
LI	Compacted peat	Path, deer grazed	96-169, 122	12.1-160.0, 28.7
		Cattle & pony /deer grazed	120-226, 166.5	12.1-186, 54.3
PL	Peat, sandy peat	Ditch	-	-
		Cattle/pony grazed	128-284, 176	6.9-364.3, 27.3
RL	Compacted sand	Tracks, stock paths cattle & pony or horse grazed	108-180, 124	4.07-227.1, 105.2
RF	Peat	Grazed mire	109-517, 190	4.5-31.28, 13.2
	Compacted peat	Fire-break, grazed	87-145, 112.5	1.49-165, 80.1

In contrast, *Rhynchospora fusca* over-wintered with seeds still attached. Again, very few seedlings were found, although vegetative spread by rhizomes was common. Some *Lycopodiella inundata* stems persisted over-winter, but most died back by October, buds re-growing in April. Cones were observed between June and early-September. Spore release was not observed, but clonal growth was apparent.

Table 4.8 Phenological data collected from Dorset in 1999 for the selected wet heath and valley mire species. *CF – Cicendia filiformis, DA – Drosera anglica, GA – Genista anglica, HP – Hammarbya paludosa, LI – Lycopodiella inundata, PL – Pinguicula lusitanica, RL – Radiola linoides , RF – Rhynchospora fusca.*

Species	**Seedling emergence/ adult growth**	**Flowering**	**Seed production and dispersal**
CF	Mid-May onwards	June - October (November)	late May to November
DA	Adult growth from late March, April Seedling emergence from March	June - August	July to October
GA	Adult growth from late-March Seedlings not found	Mid April – June	June
HP	Adult growth between July-September Seedlings not found	August	Did not occur
LI	Adult growth from April Young gametophytes not observed	NA	Spore production not observed
PL	Adult growth in late March Seedlings observed by September	June until October	Late-July to mid-November
RL	Early May onwards	Late June-August (September)	July to September (occasionally October)
RF	Stems over-winter	July -August	August onwards

4.5 Discussion

Livestock may play a key role in the regeneration of the eight wet heath and valley mire species studied here. Experiment 3 showed that trampling can have a beneficial effect on the emergence of the three species tested - *Drosera anglica, Genista anglica* and *Pinguicula lusitanica* (see Table 4.9). The glasshouse experiments suggest that such effects may also enhance *Cicendia filiformis* emergence.

The potentially beneficial effects of livestock presence on seven of the species can be inferred from the habitat surveys. These species were growing in places showing similar habitat characteristics to the experimentally grazed wet heath and valley mire at Coombe Bog (see Chapter 3), and most of these sites were in fact grazed. The one exception was *Genista anglica,* which was found growing in habitats that have greater affinities with the characteristics of ungrazed wet heath. On sites supporting the reaming species, sward heights were low and well within the range (20-186mm) found at Coombe Bog. There was a very low degree of vertical shading over all individual plants, which was lower than the average figure for sward structure in grazed plots at Coombe Bog (2.17). The area of bare ground was very variable, but bare ground was apparent on most sites. Unsurprisingly, most sites were characterised by a high area of surface water, or were moist throughout the year. Compaction varied according to the habitat type, with tracks showing a relatively high degree of penetration resistance (higher than grazed wet heath plots at Coombe Bog) and mire a much lower penetration resistance (similar to grazed valley mire plots at Coombe Bog). Microtopography was very variable, depending on the presence of track ruts or large tussocks. Although the habitat sampled for each species may not be optimum (and this was clearly the case for two *Rhynchospora fusca* and one *Cicendia filiformis* site, where populations were observed to decline between 1998 and 2000), in general, the populations were well-established and their requirements seem to have been adequately met. Each category identified is discussed below.

Table 4.9 Summary of the effects of shading, water level, grazing and trampling on the listed species. Only significant results are included

Species	**Seed treatment**	**Growth treatment**	**Mimicked livestock presence in the field**
Cicendia filiformis	• Soaking and freezing increased emergence.	• Wet and shaded conditions increased emergence	• No effects shown
Drosera anglica	• Soaking and freezing increased emergence. • Soaking increased emergence rate	• No effects shown	• Trampling and to a lesser extent grazing increased emergence
Genista anglica	• Soaking increased emergence rate	• Shading increased emergence rate	• Trampling increased emergence
Pinguicula lusitanica	• Soaking and freezing reduced emergence • Soaking increased initial emergence rate	• No effects shown	• Trampling increased emergence
Radiola linoides	• No effects shown	• No effects shown	NA

4.5.1 HABITAT DESCRIPTIONS

(A) Disturbed tracks, paths, fire-breaks or ditch edges within a wet heath context.

Cicendia filiformis, plus some populations of *Lycopodiella inundata*, *Pinguicula lusitanica*, *Radiola linoides and Rhynchospora fusca* occurred within M16 *Erica tetralix – Sphagnum compactum* wet heath and M25 *Molinia caerulea – Potentilla* *erecta* mire vegetation in highly disturbed locations such as tracks, paths, fire-breaks or, in one case, a recently cleared ditch edge. With the exception of one *Radiola linoides* stand (discussed separately), these sites were characterised by a high average percentage cover of bare ground (28.8-42.0), low average sward height (28-56 mm) and zero shading. Substrate penetration resistance was higher than in other stands, presumably due to compaction from people, livestock and vehicles. Microtopographical variation was varied, depending on the presence of wheel ruts. All sites were flooded in winter. Livestock presence at appropriate densities is likely to create, or contribute to, similar disturbed conditions.

Cicendia filiformis populations were all located on tracks on heathland. They were generally characterised by *Juncus* spp. and grasses, with forbs mostly limited to annuals such as *Anagallis minima* and *Centaurium erythraea* (growing here as an annual) or more robust rosette species such as *Plantago major* and *Leontodon autumnalis*. This community seems to be derived from an impoverished version of M25 *Molinia caerulea – Potentilla erecta* and is characteristic of tracks adjacent to, or running across, wet heath on clayey-sandy substrates that result in the retention of surface water from October through to April/May. All sites sampled were flooded until early May, and this was reflected in the enhanced emergence of soaked seed in wet conditions described above. *C. filiformis* was also shown to be able to germinate under water. Disturbance clearly plays a key role for this species, and this is reflected in the particularly large areas of bare ground and the low vegetation cover and species present. The large microtopographical variation reflects the presence of track ruts, and the largest population was found along ruts at a site where the track had been heavily used for timber extraction machinery the previous winter. The population subsequently declined. All Dorset *C. filiformis* populations are located on tracks similar to those described here. This may be linked to the low levels of phosphorus found in heath substrates in Dorset (Chapman *et al.*, 1989), as compaction can increase available P, and *C. filiformis* is known to be P limited (Brouwer *et al.*, 2001).

Radiola linoides was found in the same M25 *Molinia caerulea - Potentilla erecta* - derived, impoverished vegetation characterised by *Cicendia filiformis*. It was also found in a grassier, more species-rich version of the same NVC community in less heavily disturbed situations, although some disturbance appears important to maintain the community. All three sites of this community type were heavily rabbit grazed and disturbed by burrowing, and were also grazed by ponies and/or cattle. MATCH gave a poor identification of this community, and further interpretation of the species data

(Malloch, 1999) suggests that it is intermediate between M25 *Molinia caerulea – Potentilla erecta* mire and a humid version of U1 *Festuca ovina – Agrostis capillaris – Rumex acetosella* acid grassland (Rodwell, 1992). The sites were damp (but not flooded) in winter and parched in summer. At each site, *Radiola linoides* populations were located on tracks or livestock paths where the sward was particularly low. The germination experiments were inconclusive for this species but suggested that emergence was higher under un-shaded conditions. This species does not appear to have the same dependence on very wet conditions, and this is supported by the fact that it grows in slightly drier habitats than *C. filiformis.*

Lycopodiella inundata occurred within *M16 Erica tetralix – Sphagnum compactum* vegetation on compacted peaty ground that was seasonally flooded and remained humid throughout the summer. The NVC description provided a reasonable fit, although the quadrats probably occurred at the wetter end of the spectrum for this community, and include species such as *Sphagnum cuspidatum* and *S. papillosum.* One site was along a footpath, another located in an area around a seasonal pool used by livestock, both with substantial areas of bare ground. Microtopographical variation was limited, but both sites are flooded in winter.

Pinguicula lusitanica appears to regenerate readily following disturbance (see Chapter 5). This was observed on a near-vertical ditch side of compacted, sandy peat, which had been cleared two winters previously. The vegetation was referable to M16a *Erica tetralix – Sphagnum compactum*, although MATCH gave a poor fit, as the cover of vascular plants was low and included plants such as *Prunella vulgaris* from the adjacent area of more mesotrophic roadside vegetation. The ditch was flooded throughout the winter, although not to the height of the main *Pinguicula lusitanica* population; nevertheless, this remained humid throughout the year. All plants were un-shaded. The emergence experiments showed a preference for wetter conditions: although seed-soaking did not increase emergence, soaked seed emerged first. Light appeared to be important for untreated seeds, and may be needed to break dormancy mechanisms otherwise broken by freezing or soaking (discussed below).

Rhynchospora fusca occurred in similar habitat to *Lycopodiella inundata* within this category of disturbed situations within a wet heath context. Although the NVC fit was poor, as the site comprised mainly bare ground with only occasional plants of *Molinia caerulea, Erica tetralix, Juncus* spp., *Narthecium ossifragum* and *Trichophorum* *cespitosum*, it was clearly derived from the adjacent M16 *Erica tetralix – Sphagnum compactum* wet heath. Both sites remained only slightly humid over the summer, but both are currently becoming drier, one as a consequence of long-term changes in adjacent land use (B. Pickess, pers. comm.), and one because of succession following the conservation management that led to the population becoming established. Both populations appear to be declining.

(B) Grazed valley mire

All populations of *Hammarbya paludosa,* plus some *Lycopodiella inundata, Pinguicula lusitanica* and *Rhynchospora fusca* populations occurred within grazed valley mire. The stands were all referable to the M21a *Narthecium ossifragum – Sphagnum papillosum* mire community, and in one case the M1 *Sphagnum auriculatum* bog pool community that occurs within it. The habitat was similar for all species, which are discussed together.

The potential role of livestock within valley mires, while not as striking as for disturbed tracks, may still be significant. M21 *Narthecium ossifragum – Sphagnum papillosum* mire can become overgrown with *Molinia caerulea* (Rodwell, 1991) and a reduction in *Molinia Caerulea* and other graminoids is associated with grazing (Chapter 3). *Molinia caerulea* formed no more than 45% of the vegetation cover at these sites, and generally considerably less. Plants of all four selected species were on the whole completely un-shaded. Very light shading occasionally occurred over *Pinguicula lusitanica* and *Drosera anglica* plants. However, emergence of these species was not shown to be affected by shading in the glasshouse experiments. *Hammarbya paludosa* was un-shaded here, but has been observed in slightly shaded locations in taller stands in the New Forest. The average sward height was between 34 and 84mm, again similar to the range within grazed valley mire plots at Coombe Bog. The area of bare ground was generally limited, but the area of surface water was high. At sites where surface water was absent (where there was a continuous cover of *Sphagnum* species), the water table was just below the surface throughout the year. A need for humidity was shown in the higher emergence under wet conditions of *Drosera anglica* and *Pinguicula lusitanica* in the germination experiments. While microtopographical variation was limited, the gentle relief provided by *Sphagnum* tussocks may nonetheless be important. For example, *Hammarbya paludosa* was generally found adjacent to bog pools growing on *Sphagnum* hummocks. The increase in surface humidity associated with livestock presence may therefore be important for these species.

(C) Undisturbed or grazed wet heath

Genista anglica was the only species occurring in this habitat category. It comprised two NVC communities, M16 *Erica tetralix – Sphagnum compactum* and M25 *Molinia caerulea – Potentilla erecta,* although both were substantially different here, than where they occurred in Category A. Two stands comprised relatively species-rich M25 *Molinia caerulea – Potentilla erecta.* Both appeared transitional to M24 *Molinia caerulea - Cirsium dissectum* fen meadow *Juncus acutiflorus - Erica tetralix* sub-community, and were significantly different from the impoverished M25 – derived track vegetation described above. Both stands were cattle-grazed and supported large numbers of mature bushes. The remaining stand, which was grazed only by deer and dominated by *Molinia caerulea,* was referable to species-poor M16b *Erica tetralix – Sphagnum compactum* wet heath.

Genista anglica is the only species of those studied that was found in relatively rank vegetation. The area of bare ground was minimal, and the sward height comparatively high. Vertical shading was not measured, as all *Genista anglica* plants located were growing through *Molinia caerulea* tussocks, therefore numerous branches were shaded within the tussocks. However, the bushes themselves were never heavily shaded by taller vegetation, although several were growing adjacent to a birch (*Betula pendula*) copse at one site. Shading increased emergence in the glass house. There was no surface water apparent at any of the stands when the quadrats were recorded, but the substrate remained humid throughout the year, and was partially flooded throughout the winter at one site. Although soaking seed increased the emergence rate, it did not increase overall emergence. Soil penetrability was varied, and was influenced by the presence of thick *Molinia caerulea* tussocks, as was the microtopographical variation.

At one stand, several plants apparently appeared four years after grazing was re-introduced to the site. Closer inspection revealed that most of the plants had thick woody stems, and cannot have regenerated from seed in the time since grazing was re-introduced. It seems likely that removal by grazing of the thick *Molinia caerulea* which previously characterised the site facilitated growth and flowering, and possibility vegetative reproduction. Clearly, this species is able to persist in sub-optimal conditions for a significant period. However, no seedlings were found at any of the sites. Results from the field experiments showed clearly that livestock trampling can enhance *Genista anglica* seedling emergence.

4.5.2 MECHANISMS OF EFFECTS ON EMERGENCE

The creation of gaps with the required humidity appears to be important for the wet heath and valley mire species studied. Trampling and grazing may affect emergence through creating gaps in vegetation, providing release from competition for light, nutrients and other resources (Bullock *et al*, 2001), increasing compaction and microtopographical variation (see Chapter 3), and bringing buried seed to the soil surface.

The creation of stem gaps, i.e. the absence of rooted stems, is clearly important in providing the physical space for establishment - all species studied were either found growing on bare ground or the *Sphagnum* carpet of quaking mires. Compaction has been shown to reduce vegetation regeneration in peat cuttings (Cooper, McCann & Hamill, 2001), and may therefore play a role in the persistence of bare ground following disturbance. Compaction may also have a rather precise role increasing phosphorus availability for species such as *Cicendia filiformis*. It may also have an adverse affect on seedling emergence by reducing the amount of oxygen available within the soil. The very small size of the seeds of all species studied (with the exception of *Genista anglica* whose seeds are nonetheless no more than one mm in diameter) suggests that these seeds are likely to germinate on or very close to the substrate surface. The role of compaction in reducing available oxygen is not considered further for these species.

More importantly, compaction is likely to lead to greater water retention, particularly on sandy and clayey soils. Together with increased microtopography (see Chapter 3), this is likely to create a diversity of humid microsites that can provide the right degree of moisture to suit different species. Results from the emergence experiments confirmed most species' need for wet conditions (see Table 4.8), and this is likely to be the case for other wet heath and valley mire species, for example, seed soaking is important for marsh gentian *Gentiana pneumonanthe* germination (R. Rose, pers. com.). However, all species may not have precisely the same requirements, for example, *Drosera anglica* emergence was enhanced by both seed soaking and wet growing conditions, while *Pinguicula lusitanica* emergence was reduced by seed soaking, although wet growing conditions enhanced it. Where germination requirements are precise, the varied microtopography and compaction and therefore varied humidity caused by trampling may ensure that a diversity of conditions are available.

However, trampling may also disturb existing microtopographical variation. For example, the gentle hummocks and hollows characteristic of *Sphagnum* carpets, which are exploited by species such as *Hammarbya paludosa,* may be destroyed by trampling. Trampling also may dislodge seedlings (and also adults of species such as *Drosera anglica, Pinguicula lusitanica* and *Hammarbya paludosa,* which have poorly developed root systems).

In bringing buried seed to the surface (discussed in Chapter 5), trampling may increase exposure to frost. The results from seed treatments were varied (see Table 4.8), with freezing decreasing, increasing or having no effect, depending on the species. When chilling over a sufficient time-span is required to break dormancy mechanisms, germination will occur in the spring rather than immediately after dispersal in the autumn, ensuring that seedlings germinate in more suitable conditions. Therefore freezing may be expected to break dormancy mechanisms in species that germinate in the spring. This was seen clearly in *Cicendia filiformis* and *Drosera anglica,* and a similar trend was seen for *Genista anglica*, but not for *Radiola linoides*, which might also be expected to share this trait (although sample sizes and replication were too small to make this conclusive). However, as *C. filiformis* and *R. linoides* both appear to produce seed that germinates in the same season in addition to seed that over-winters, a diversity of responses may be expected depending on when the seed is produced. Freezing decreased emergence in *Pinguicula lusitanica*, and *Pinguicula lusitanica* seedlings were observed to emerge in the autumn immediately after dispersal. Dormancy release has been reported over a wide range of temperatures from 1.5 to 15°C (Totterdell & Roberts, 1979), therefore (i) actual freezing may not be necessary to break dormancy; (ii) sufficiently low temperatures may be experienced by buried seed. It seems unlikely that livestock presence plays much of a role in chill-induced dormancy release.

Grazing could be expected to increase emergence through reduced canopy shading. Light responses are very common in small-seeded species capable of emerging after disturbance (Pons, 2000), such as *Cicendia filiformis, Radiola linoides* and *Pinguicula lusitanica.* Although light is not a prerequisite for germination, it can serve as a signal for germination, and can have a different effect depending on the wavelength, the plant species and the season. Canopy shade has a low red:far-red light ratio due to the absorption of photosynthetically active wavelengths by chlorophyll. This was imitated by the use of green shading to absorb red wavelengths (although the precise effect of this material on the red:far-red ratio was not known). While exposure to red light can break dormancy mechanism, exposure to far-red light can impose dormancy (Baskin & Baskin, 1998). This means that (i) seeds are likely to germinate in canopy gaps rather than where plants are already established, and (ii) seeds may be prevented from germinating shortly after dispersal, facilitating their inclusion within the buried seed bank. A number of studies have tested the effects of light on seed germination, for example, Fenner (1980), Silvertown (1980) and Pons (1991). Sixty percent of the 58 species tested in these studies showed enhanced germination under full light, while 24% were unaffected by light. Similarly, there is a general trend for increased emergence in canopy gaps compared to intact vegetation (e.g. Johnson & Thomas, 1995; Keizer, Van Tooren & During, 1985; Bullock *et al*, 1994; Morgan, 1997).

Drosera anglica showed increased emergence in grazed treatments compared with ungrazed controls in the field experiments, while *Radiola linoides* showed a trend towards increased emergence in un-shaded compared to shaded treatments. However, emergence in the remaining species did not appear to be enhanced by light. *Cicendia filiformis* and *Genista anglica* showed increased emergence under shaded (not green-shaded) treatments, and this may be due to the increased humidity under the shading mesh, suggesting that in drier conditions, slight canopy shading increasing moisture retention may be beneficial for seedling emergence (e.g. O'Connor, 1996). It would appear that, in general, light is not the main factor limiting emergence for the species studied, and therefore canopy gaps may not significantly increase emergence.

However, germination response to light is complex, and may be affected through interactions with alternating temperatures, nitrate and desiccation (Probert, 2000). There are a number of ways in which the experiments may not have fully identified the response to light. For example, for those species which showed limited germination overall, a longer exposure to light may have increased emergence. Both *Radiola linoides* and *Cicendia filiformis* are relatively late germinating in the spring, and a longer exposure requirement would delay germination until sites have dried out sufficiently. Alternatively, the required exposure time may be very short (Pons, 2000), and the exposure to full light that occurred while the seeds were being sown may have been sufficient to break dormancy.

In contrast to the experimental results, the habitat characteristics for all species except *Genista anglica* suggest a preference for virtually totally un-shaded conditions. Shading and other competitive effects can greatly reduce seedling growth and survival (e.g. Fenner, 1978; Van der Toorn & Pons, 1988; Hanley, Fenner & Edwards, 1996), and it could be this, rather than the effects on germination and emergence, that influence whether or not plants are found in shaded positions. All seedlings under shade in both glasshouse experiments became etiolated as they developed, suggesting that lack of light could have a deleterious effect on seedling survival.

Different components of plants' life-cycles may require different habitat conditions. For example, while a species may require humid, bare ground for regeneration, the degree of disturbance required to provide this may negatively affect the survival of the adult plants, which require reduced canopy cover rather than bare ground. So, where this disparity occurs, which life-cycle component should conservation management be focussed on?

4.5.3 LIFE CYCLE AND REPRODUCTIVE STRATEGY

Assuming that conservation management is aimed at maximising population growth (since most of these species are declining), it needs to be targeted at the life-cycle phase that will contribute most to population growth. Silvertown *et al.* (1993) explored the correlations between three major demographic parameters (fecundity, growth and survival) and the intrinsic population growth rate of a variety of species. They showed a positive correlation between fecundity (i.e. recruitment to the seed pool and seedling emergence) and population increase for herbaceous species, and suggested that short-lived species (e.g. annuals) are likely to be more strongly controlled by fecundity than long-lived perennials. Bullock *et al.* (1994) showed that the importance of fecundity in determining population growth can further increase with grazing. In contrast, woody species were more controlled by longevity than fecundity (Silvertown *et al.,* 1993), therefore the relationship between grazing and fecundity is not of such importance in conserving populations of woody species.

Consideration of the contribution of demographic parameters to population growth can provide a framework for designing conservation management. However, iteroparous herbs of open habitats, such as some of those studied in this chapter, show the greatest variation in the importance of each demographic parameter (Silvertown *et al.,* 1993, 1996). A closer consideration of reproductive strategies can help refine this framework. For example, no seedlings of *Genista anglica, Hammarbya paludosa,* or young sporophytes of *Lycopodiella inundata* and very few seedlings of *Drosera anglica* and *Rhynchospora fusca* were found in the field, suggesting that regeneration by seed/spore is infrequent for these perennials (shrub, herb, graminoid and bryophyte alike). Silvertown *et al.* (1993) found a negative correlation between the elasticity (i.e. changes in population growth in relation to small changes in the component) of seedling recruitment and the elasticity of the clonal component of growth. This suggests that species showing clonal growth are unlikely to be dependent on seedling recruitment for population growth. Vegetative regeneration was observed in *G. anglica, H. paludosa, L. inundata* and *R. fusca.* Other *Drosera* spp. are known to reproduce clonally (Crowder *et al.,* 1990), although this was not observed in *Drosera anglica.* In nutrient-poor situations, vegetative regeneration will reduce the need for resource expenditure on seed production, and it is therefore likely that many wet heath and valley mire species will reproduce vegetatively. This corresponds with the first of Grime's (Grime *et al.,* 1988) regenerative strategies - vegetative expansion in productive or unproductive habitats subject to low intensities of disturbance. Perennial species, including long-lived woody species such as *Genista anglica*, do not need to reproduce successfully each year, and regeneration requirements may not be as demanding as for seedling emergence while the ramets remain dependent on the parent plant. For these species, conservation management should therefore be mainly focussed on maintaining suitable conditions for adult survival, with seedling regeneration requirements being met more occasionally. Sporadic or light disturbance, such as that created by low intensity livestock presence, may be sufficient to achieve this.

Regeneration by seed is more likely where adult survival chances are limited compared to juveniles (e.g. more heavily disturbed areas such as paths). This has been shown for long-leaved sundew *Drosera intermedia* (Deridder & Dhondt, 1992) and was seen in Dorset for *Pinguicula lusitanica*, which reproduced prolifically by seed in heavily disturbed conditions. This strategy is obviously also essential for annuals such as *Cicendia filiformis* and *Radiola linoides.* Such species can be considered within the context of species which form persistent seed or spore banks in habitats subjected to temporally unpredictable disturbance (Grime *et al.,* 1988). For these species, regeneration is clearly the most important component to target for conservation, and this can be achieved using more intense stocking rates to create a greater degree of disturbance. The ability to form seed banks will dictate whether continuity of suitable emergence conditions is required to maintain populations in the long term.

4.5.4 SUGGESTED GRAZING REGIMES FOR THE EIGHT STUDY SPECIES AND SIMILAR WET HEATH AND VALLEY MIRE SPECIES

Taking into account life-cycle and reproductive strategies, together with habitat and phenological data, the potential of livestock in the conservation management of each of the study species is summarised in Table 4.10.

Table 4.10 Livestock recommendations for each of the eight species. CF – *Cicendia filiformis*, DA – *Drosera anglica*, GA – *Genista anglica*, HP – *Hammarbya paludosa*, LI – *Lycopodiella inundata*, PL – *Pinguicula lusitanica*, RL – *Radiola linoides* , RF – *Rhynchospora fusca*. Grazing intensity as used here: low < 0.01 LU ha^{-1} yr^{-1}, medium $= 0.05$ LU ha^{-1} yr^{-1} , HIGH > 0.1 LU ha^{-1} yr^{-1} (but see text).

Species	**Life-cycle stages grazing management should be targeted at**	**Suggested grazing intensity**	**Potential methods**
CF	Seed bank and seedling recruitment	High intensity	Focus livestock activity using water troughs, gateways etc. Winter stocking to provide bare ground in late spring, early summer stocking if reduction in dominants required.
DA	Clonal regeneration and adult survival	Low-moderate intensity	Summer only, to prevent undue poaching in wetter seasons
GA	Clonal regeneration and adult survival	Medium intensity	Late spring - early summer to control *Molinia caerulea* dominance and provide light poaching.
HP	Clonal regeneration and adult survival	Low intensity	Summer only. Small population may be susceptible to trampling
LI	Clonal regeneration and adult survival, (possibly regeneration from spores)	Low intensity, sporadically high	Summer only. Location of drinking pools will influence trampling pressure where occurs on pool edges.
PL	Seed bank and seedling recruitment, adult survival	Moderate intensity, sporadically high	Sporadic higher disturbance for regeneration from seeds required. May occur along paths.
RL	Seed bank and seedling recruitment	High intensity	As for CF
RF	Clonal regeneration and adult survival	Low intensity	Summer only, to prevent undue poaching in wetter seasons. Possibly increase season if habitat drying out.

These recommendations provide general guidance only, and should be tailored to the particular site and year in question. Both grazing season and intensity need to be considered in the light of likely livestock behaviour on a site. A given grazing intensity on a site overall will not necessarily result in that grazing pressure on individual habitat patches. For example, on sites with a diversity of vegetation communities present, valley mire is little used by livestock, and generally only in mid – late summer (see Chapter 2). Therefore, year round livestock presence on such a site is unlikely to compromise the requirements of species such as *Hammarbya paludosa* and *Drosera anglica*. However, in a particularly dry year, use of this valley mire may be heavier, particularly if it provides the only water source, and stocking rates may need to be adjusted accordingly. Wet heath is used similarly to valley mire, but may be used more for moving between habitats. This will provide the slightly heavier degree of disturbance that is likely to be more favourable for *Pinguicula lusitanica.* Tracks tend to be used more intensively than wet heath and valley mire, and are used all year. This may provide adequate disturbance for *Cicendia filiformis* and *Radiola linoides*, or

Table 4.11 Some additional wet heath and valley mires species that may benefit from livestock presence.

Category	**Indicator species**	**Other wet heath or valley mires species potentially showing a similar response to livestock presence**
• Trampled tracks, reliant on disturbance. • Very short, damp grassland in heathland context (e.g. lawns and greens)	*Cicendia filiformis,* *Radiola linoides*	*Anagallis minima,* *Chamaemelum nobile*
• Herbaceous perennials of wet heath benefiting from gap creation and suppression of dominants	*Pinguicula lusitanica*	*Gentiana pneumonanthe,* *Lobelia urens,* *Viola lactea* *Drosera rotundifolia*
• Woody perennials of wet heath, persistent in absence of grazing but regeneration and growth enhanced by livestock	*Genista anglica*	*Erica ciliaris*
• Open valley mire species benefiting from control of *Molinia caerulea*, may be sensitive to overgrazing	*Drosera anglica,* *Hammarbya paludosa,* *Lycopodiella inundata*	*Narthecium ossifragum,* *Drosera intermedia* *Sphagnum pulchrum* *Sphagnum magellanicum*
• Bog pools – not directly affected by livestock, but benefiting from control of succession in surrounding valley mire to maintain water levels	*Rhynchospora fusca*	*Utricularia intermedia,* *Utricularia minor,* *Potamegeton polygonifolius* *Meyanathes trifoliate* *Carex limosa*

livestock activity may need to be focused by the provision of water troughs and pools and the positioning of fencing and gates. In practice, livestock at appropriate densities over sufficiently large areas may provide a suitable diversity of conditions to meet the varied requirements of different species and life cycle components. Where this is not possible, more intensive management may be needed, and the above suggestions will need to be integrated with other heathland management practices such as mechanical disturbance or burning (Michael, 1993). Practicalities of heathland management through livestock are discussed further in Chapter 6.

Finally, Table 4.11 lists additional wet heath and valley mire species that are likely to be influenced by livestock. In practice, species may fit in more than one category. Common dominants have been excluded. The purpose of this table is to suggest other species of interest that may be affected in a similar way to the indicator species chosen, rather than to account for all wet heath and valley mire species.

Conclusions

- Conservation management for population growth should be targeted at seed bank and seedling recruitment in annuals and more transient herbaceous perennials characteristic of disturbed habitat, and at adult survival in woody perennials and longer-lived herbaceous perennials characteristic of more stable habitat;
- Livestock trampling can play a key role in providing suitable conditions for regeneration in characteristic wet heath and valley mire species. This occurs through the creation of gaps, providing establishment opportunities, and an increase in a diversity of humid microsites;
- Canopy removal through livestock grazing was not shown to be important in seedling emergence for these species. However, it is likely to play a key role in subsequent survival;
- Livestock at appropriate stocking rates have the potential to provide a diversity of conditions that may promote population growth in a range of characteristic valley mire and wet heath species.

5. Effects of livestock activity on species regeneration from the buried seed bank

5.1 Introduction

The conservation value of lowland heaths is threatened by successional change to other communities (Marrs *et al.,* 1986; Webb, 1990). On wet heath and valley mire, successional change leads to dominance by *Molinia caerulea* and eventually to the development of woodland dominated by such species as grey willow *Salix cinerea.* and downy birch *Betula pubescens* (Rodwell, 1991, see also Chapter 1). Suitable techniques are therefore required to restore vegetation where this has occurred. Although there has been increasing interest in restoration techniques for dry heath (e.g. Putwain & Gillham, 1990; Aerts *et al.*, 1995; Pakeman & Hay, 1996; Pywell *et al.,* 1997; De Graaf *et al,* 1998; Dunsford *et al.,* 1998; Mitchell *et al.,* 1998; Britton *et al,* 2000; Owen & Marrs, 2000; Bossuyt, 2001; Degn, 2001), wet heath and heathland valley mires have so far received scant attention (but see Sansen & Koedam, 1996; Kesel & Urban, 1999).

Habitat restoration may be aimed at achieving overall changes in vegetation type (e.g. heathland reversion following succession to woodland or agricultural improvement), or at the enhancement of species diversity within the existing vegetation types (for example in purple moor-grass *Molinia caerulea* dominated wet heath vegetation). In either case, conservation managers need to know what, and how many, plant propagules are available on a site, as this will influence both the initial floristic composition of the site after restoration management (Hobbs & Gimingham, 1984) and future floristic developments (Egler, 1954). Natural dispersal may be slow and unreliable (e.g. Bakker *et al,* 1996; Bullock *et al.,* 2002). This is particularly the case in the modern landscape, where the potential for long-range dispersal by water, large animals or other vectors has become increasingly limited (Poschlod & Bonn, 1998). As many conservation restoration sites are isolated from source populations (Verkaar, 1990), seed banks can play an important role in natural and facilitated regeneration (e.g. Putwain & Gillham, 1990; Beatty, 1991; Bakker *et al.,* 1996; McDonald et al., 1996; Strykstra *et al.,* 1998) . Alternative means of vegetation restoration, such as sowing and transplanting, may be prohibitively expensive (e.g. Putwain & Gillham, 1990) or be complicated by practical and ethical considerations (e.g. Pywell & Putwain, 1996). Recruiting from the seed bank can be a convenient and efficient means of re-establishing or enhancing target vegetation types. However, the management of vegetation by deliberately manipulating environmental conditions to recruit preferred species from the seed bank is still relatively uncommon (Van der Valk & Pederson, 1989).

When the viable seed bank and vegetation at a site are compared, species may be present in one and not the other (see Warr *et al.,* 1993; Thompson, 2000), as the composition of a seed bank is a function of the seed production of present and previous vegetation, and the longevity of seeds of each species under local conditions (Parker *et al.,* 1989). Seed bank composition is only likely to coincide with standing vegetation in highly disturbed habitats, for example arable fields (Pickett & McDonnell, 1989). This difference is what makes the exploitation of seed banks so attractive to conservation managers – the desired species at a site may be present in the seed bank even if extinct in the above ground vegetation. Restoration from seed banks is probably best suited to semi-natural habitats historically characterised by relatively high frequencies of disturbance, in which regeneration from the seed banks is a key part of the defining ecological processes. Van der Valk and Pederson (1989) outline three basic requirements for successful vegetation management based on seed bank exploitation:

(i) seeds of the required species should be present in the seed bank;
(ii) seeds of non-target species should be uncommon or absent;
(iii) conditions suitable for germination of the preferred species should be established and maintained.

Seed rain from nearby populations of both required and non-target species may also be important. In addition, ongoing management will be necessary if competitive, non-target species become established.

To some extent, it is possible to predict whether a suitable seed bank is likely to be present, by considering the characteristics of both the community and the individual species of interest (e.g. Strykstra *et al.,* 1998). The formation of seed banks is a strategy that allows plant populations to persist even when seed production does not take place. Therefore, it is frequently found in annuals or short-lived ruderal perennials that colonise vegetation gaps in disturbed communities (Rice, 1989). More stable and/or late successional communities are less likely to have extensive seed banks. Seed characteristics can also be used to establish the likelihood of a species forming a seed bank - species that form persistent seed banks tend to have small, compact seeds, which are stimulated by changes in temperature and light quality to germinate (Thompson & Grime, 1979; Thompson *et al.,* 1993). Thompson, *et al.* (1997) have categorised over 1000 taxa according to whether they have persistent or transient seed banks. These are defined as (Thompson, 1993):

(i) *transient seed banks*, consisting of seeds which persist in the soil for a year at most and are able to germinate immediately;

(ii) *short-term persistent seed banks*, comprising seeds which are on a state of dormancy and can remain viable for between one and five years; and

(iii) *long-term persistent seed banks*, species with seeds which can persist in the soil for at least five years.

Clearly, persistent seed banks are required if vegetation restoration is to be successful. However, the presence of a persistent seed bank will not guarantee success - subsequent seedling and adult mortality (through drought, flooding, herbivory, disease, and competition) play a key role in determining the composition and structure of the restored vegetation (e.g. Pywell *et al.,* 2002). On the other hand, species with transient seed banks often compensate with alternative mechanisms for persistence, e.g. long-lived adults, persistent bulbs or rhizomes (Grime, 1989; Leck, 1989), and this has been shown to be the case for some characteristic valley mire species (see Chapter 4).

There is increasing recognition of the potential of seed banks in the restoration of heathland (e.g. Putwain & Gillham, 1990; Pywell *et al.,* 1995; Pakeman & Hay, 1996; Pakeman & Marshall, 1997; Mitchell *et al.,*1998). As successional communities, heathlands are dependent on some degree of disturbance to persist, and have long been known to have large seed banks of the dominant species (Chippindale & Milton, 1934, Champness & Morris, 1948; Gimingham, 1972). Seeds of species such as heather *Calluna vulgaris*, bell heather *Erica cinerea* and cross-leaved heath *Erica tetralix* are known to remain viable for several decades and heathland seed banks can be large (up to 33 700 seeds m^{-1}) (e.g. Pywell *et al.,* 1995; Thompson & Band, 1997). Such large and persistent seed populations may allow heathland species to disperse through time and space, re-colonising sites after considerable disturbance.

However, in practice it has been found that a relatively large number of species of semi-natural communities cannot re-establish from long-term persistent seed banks (Thompson *et al.,* 1997; Bekker *et al.,* 1998). There is little work directly addressing wet heath or valley mire seed banks (but see McGraw, 1987; Poschlod, 1995; Sansen & Koedam, 1996; Jauhiainen, 1998; Kesel & Urban, 1999; Sundberg & Rydin, 2000), and this is probably a reflection of the practical difficulties associated with sampling these habitats. Re-establishing vegetation from seed banks can be problematic in many habitats (see Bekker *et al.,* 1998), and even re-establishment of dry heath, which has large persistent seed banks, is not necessarily straightforward (e.g. Pakeman & Marshall, 1997; Pywell *et al.,* 1997; Mitchell *et al.,* 1998). Pywell *et al.* (1997) showed that on damper dry heathland sites there was a relatively poor correlation between seed bank and vegetation composition. Pakeman and Hay (1996) found that the dry heath seed bank under bracken *Pteridium aquilinum* decreased after 50 years. The deciduous species *Molinia caerulea,* often dominant on degraded damp heath and mire (see Chapter 3), can, like *P. aquilinum*, lead to a large build up of litter, and may also lead to an impoverished seed bank in precisely the situation where a large seed bank would be useful for restoration. However, Sansen & Koedam (1996) and Kesel & Urban (1999) showed that round-leaved sundew *Drosera intermedia*, cross-leaved heath *Erica tetralix*, brown beak sedge *Rhynchospora fusca*, marsh clubmoss *Lycopodiella inundata* and marsh gentian *Gentiana pneumonthathe* all re-established following sod-cutting in degraded wet heath communities, presumably from the buried seed bank.

In Dorset, the cessation of livestock grazing and associated activities has been identified as a possible cause of the decline of characteristic wet heath and valley mire species in existing sites (Byfield & Pearman, 1994, and see Chapter 1). Livestock trampling, in particular is considered to have played a key role in both creating and maintaining suitable conditions for a number of characteristic species. This may partly have been through stimulation of regeneration by exposure of the buried seed bank. Anecdotal evidence from the Dorset and New Forest heaths in the UK suggests that a number of wet heath and valley mire species which were considered extinct at particular sites apparently regenerated from the buried seed bank following disturbance (Table 5.1).

Table 5.1 Cases where the (re)appearance of damp heath and valley mire species has been attributed to livestock presence or other disturbance on the Purbeck and New Forest heaths.

Species	Occurence	Site	Source
Pinguicula lusitanica	Appeared in site disturbed by machinery Reappeared in cleared ditch Appeared in livestock trampled area	Three Barrows Heath Arne Triangle Three Barrows Heath	T. Brodie-James (1998) Personal observation (1999) Personal observation (2000)
Rhynchosphora fusca	(Re?) appeared in soil stripped area	Tadnoll Heath	R. Blunt (pers. comm. 1998)
Lycopodiella inundata	(Re?) appeared in soil stripped area	Tadnoll Heath	Stewart *et al.*, 2000
Genista anglica	Re-discovered in newly cattle grazed area	Hartland Moor	J. Cox (pers. comm. 2000)
Cicendia filiformis	Frequently located within hoof prints	New Forest	N. Sanderson (1994)
	Increase in population after disturbance	Thrashers Heath	Personal observation (1999)
Radiola linoides	Increase in population after disturbance	Thrashers Heath	Personal observation (1999)
	(Re?)appearance after reinstatement of grazing	Grange Barn Field	J. Day, B. Pickess (pers. comm., 1999)
Drosera spp.	Populations concentrated in livestock disturbed areas	Purbeck Heaths	Personal observation (1998-2000)

Livestock may be a particularly valuable tool for creating appropriate disturbance. In addition to exposing buried seed, livestock trampling on wet heath and valley mire increases microtopographical variation (see Chapter 3), creating a variety of microsites which may allow the germination requirements of characteristic species to be met (see Chapter 4, also Dinsdale *et al.,* 2000). Grazing and trampling also reduce canopy cover and simplify the vegetation structure (see Chapter 3), thus increasing the amount of light reaching seeds and reducing competition from dominant species. In addition to exposing the buried seed bank and providing suitable conditions for germination and subsequent survival, livestock may at the same time act as effective dispersal vectors (e.g. Willson, 1993), connecting seed source areas (including seed banks elsewhere) and target areas (Fischer *et al,* 1996; Bekker *et al*, 1998).

This chapter considers the persistence in the seed bank of a number of characteristic wet heath and valley mire species (described fully in Chapter 4). The ability of these species to regenerate from the seed banks at sites where they are extinct in the above-ground vegetation is evaluated. The effectiveness of livestock trampling in providing suitable conditions for regeneration of these and other wet heath and valley mire species from the seed bank on degraded sites is considered.

5.2 Methods

5.2.1 GLASSHOUSE SEED BANK EXPERIMENTS

Sampling

The soil seed bank for six wet heath and valley mire species was sampled to establish whether regeneration of these species from a buried seed bank was possible under suitable conditions. The species were *Cicendia filiformis, Genista anglica, Hammarbya paludosa, Lycopodiella inundata, Pinguicula lusitanica,* and *Radiola linoides*.

The seed bank was sampled in March 1999 from sites where the species had been located the previous season. For each species, a total of 40 cores 10 cm diameter and four cm in depth (heathland seed banks have been shown to be concentrated in the top four cm (Pywell *et al.*, 1997)) were taken from between one and four sites at random positions within the known population area. The optimum number of 60 cores (Benoit *et al.*, 1989) was reduced due to concern by relevant conservation agencies about possible detrimental effects on the populations concerned. *Radiola linoides* was later found at one of the sites (Thrashers Heath) sampled for *Cicendia filiformis* in 1999, and therefore these samples were also searched for regenerating *R. linoides*.

For species growing on firmer substrates (*R. linoides*, *C. filiformis* and *G. anglica)* the core was divided into two sections (0-2 cm and 2-4 cm depth) to assess the persistence of the species within the seed bank (Thompson *et al.*, 1997).

Samples were prepared differently according to the humidity of the substrate. Samples for species growing on sandy substrates (*Radiola linoides* and *Cicendia filiformis*) were air dried for two weeks at 20°C and then sieved. The fraction of each core section smaller than two mm was spread in a layer approximately one cm thick in seed trays (160mm x 204mm x 50mm) prepared with two cm of peat substitute (Bower's 'New Horizon' peat-free multicompost) and placed on capillary matting on glasshouse benches. For the wetter valley mire species (*Drosera anglica* and *Hammarbya paludosa*), there was little or no mineral or humus substrate present in the cores, which comprised only *Sphagnum* species and water. This was spread out in unperforated trays, half of which also contained a two cm layer of peat substitute. This was intended to provide a more persistent substrate for seedlings to root in if the *Sphagnum* failed to survive. Samples for species found in slightly drier peaty conditions (*Genista anglica* and *Pinguicula lusitanica)* were crumbled by hand while wet and as much root material as possible extracted. These samples were then all spread over a two cm layer of peat substitute. For all species, trays containing only the peat substitute were included to check for the establishment of species through seed rain. All trays were watered once a week in the winter and three times a week in the summer, using tap water. Emerging seedlings of all species were noted and removed after two, six and twelve months. After six and twelve months, the samples were stirred to expose any further seeds. The names and grid references for the sites sampled are given in Table 5.2.

Table 5.2 Wet heath and valley mire seed bank sampling locations in Purbeck, Dorset.

Species	**Site**	**Grid reference**
Radiola linoides	Thrashers Heath track	SY 971852
	Great Ovens Hill	SY 923904
	Stoborough Heath	SY 925855
	Three Barrows Heath	SY 937852
Cicendia filiformis	Thrashers Heath track	SY 971852
	Great Ovens Hill	SY 923904
Drosera anglica	Turford Bog	SY 967888
Genista anglica	Orchard Mire	SY 897832
	Bank Gate Common	SY 957865
Hammarbya paludosa	Tramway mire, Three Barrows Heath	SY 938885
	Three Barrows Mire	SY 941859
Pinguicula lusitanica	Spur Bog	SZ 027853
	Winfrith Heath	SY 811869

Analysis

For each species, the number of seedlings germinating from each site and within each depth range was analysed using a Kruskal-Wallis test (the large numbers of zero results precluded the use of a blocked analysis to consider the effects of both depth and site).

5.2.2 FIELD SEED BANK EXPERIMENTS

Sampling

The glasshouse experiments were limited by the difficulties faced in meeting the germination requirements within a glasshouse environment (see Chapter 4 and results below), and difficulties maintaining the substrate in valley mire samples (i.e. *Sphagnum* spp.) in a favourable condition. Therefore, experimental seed bank exposure was also carried out in the field. Field seed bank experiments were carried out for the six species above, plus brown beak sedge *Rhynchospora fusca* and marsh clubmoss *Lycopodiella inundata,* as some of the sites chosen for the other species also supported these.

Two types of site were identified for each species:

(i) Extant sites, where the species were observed in 1999. Regeneration from these sites would show that the disturbance techniques used (described below) successfully allowed regeneration from exposed buried seed banks;

(ii) Extinct sites where the species are now thought extinct (i.e. absent in the above-ground vegetation) but had been recorded within the last 70 years. Any regeneration from these sites would provide evidence that the species can regenerate from a persistent seed bank at sites where they are extinct in the above-ground vegetation.

In total, 36 suitable sites were identified and used (Table 5.3). Extinct sites were identified from records held by the Dorset Environmental Records Centre plus records made by Prof. Ronald Good in the 1930s and 1940s and since held at the Institute of Terrestrial Ecology's Furzebrook station (now NERC Centre for Ecology and Hydrology, Dorset). A number of Good's stands were subsequently revisited in 1994 by Byfield and Pearman (1994) who searched for a number of key species, including those species considered here (see Chapter 1). Byfield and Pearman provided six figure grid references for the stands they revisited, and these were used to relocate stands. The locations of the remaining stands were taken from Good's original six-inch to the mile maps.

At each site, thirty pairs of 30 cm x 30 cm plots were located on a grid system covering the potential habitat within the marked site on Good's maps, or the 100m^2 area of the available six figure grid references from Byfield and Pearman (1994) or, at extant sites, the area of the population (which was mapped earlier in the season). Potential habitat was identified as that where the species concerned was considered likely to be able to germinate (based on information presented in Chapter 4). So, for example, if the grid reference for a valley mire species included an area of dry heath, this was excluded. Thirty plots were treated at each site, and for each an adjacent control plot was left

Experiments were carried out during November 1999, when seed production had ended. However a number of sites were found to be too wet, and so these sites were treated in early May 2000, when water levels dropped (see Table 5.3). The number of treatments was also doubled at three of the larger sites at this timeuntreated. Thirty small plots were used rather than fewer larger ones because: (i) results from the glasshouse experiments suggested that seedling emergence could be highly clumped, therefore dispersion of plots was maximised to address the consensus that a larger number of smaller samples will reduce variance (Benoit *et al.*, 1989; Bigwood & Inouye, 1988); and (ii) some site managers had expressed concern over larger areas being disturbed, particularly at extant sites.

Alternating plots were experimentally disturbed using one of two methods: either they were artificially trampled, or a turf of no more than four cm thickness was inverted and the surface loosened with a spade. Artificial livestock trampling was carried out using a concrete cast of a cow's hoof (weight 5kg) attached to a spade handle and swung from shoulder height 30 times. The turf inversion technique was included to establish whether general disturbance was likely to allow regeneration from the seed bank where a seed bank was present, and compare this with disturbance broadly similar to that created by livestock, which was considered likely to result in a varied microtopography and degree of surface water cover, and greater mixing of the substrate.

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Plate 5.1 Simulating cattle trampling in seed bank experiments

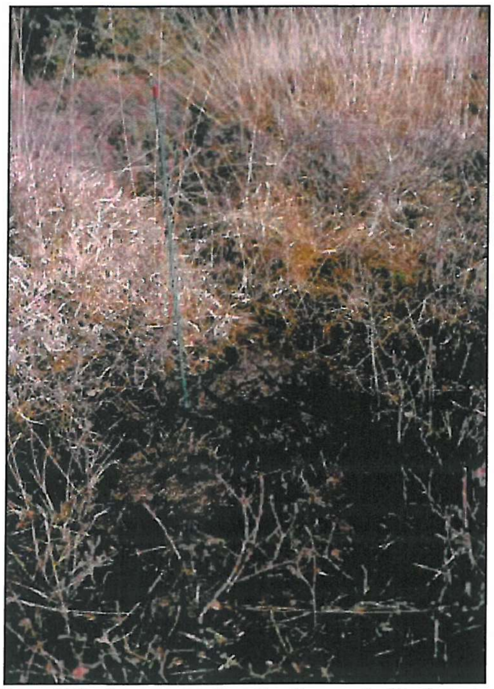


Plate 5.2 An example of a 'trampled' plot at a *Molinia caerulea*-dominated wet heath site

The plots (see Table 5.3) were searched for the species in May 2000 and then checked at monthly intervals throughout the growing season until November 2000 (as the glasshouse experiments suggested that seedlings may take up to 12 months to emerge). Seedlings of *Cicendia filiformis, Radiola linoides,* and *Pinguicula lusitanica* were not counted after August, as any such seedlings could have been a result of dispersal of that year's seeds. Plots treated in May 2000 were also checked in May 2001, as were all *Lycopodiella inundata* and *Rhynchospora fusca* plots, as potential germination periods for these species are unknown. At extant sites, the number of seedlings in adjacent undisturbed plots of the same area were also recorded. The vegetation composition (% cover) within all plots and adjacent undisturbed plots was recorded in June 2000 and the number of seedlings of all species were also recorded at both extant and extinct sites. Example seedlings of monocots were grown on to aid identification where necessary. Seedlings of *Carex panicea* were readily identified by their colour. However, other *Carex* species were less easily distinguished (although a large proportion were thought to be *Carex pilulifera*), and were therefore grouped as *Carex* spp. *Juncus* species, which included *J. articulatus*, *J. bulbosus* and *J. bufonius* were grouped as *Juncus* spp, and *Ulex europaeus*, *U. gallii* and *U. minor* were grouped as *Ulex* spp..

Table 5.3 Heathland sites in Dorset where field seed bank experiments were carried out. Dates for the last record are given in each case. Species recorded by Good but not relocated since are marked 'X', as it is not known how soon after the 1930s the species became extinct at a site. ✓ indicates that the species was present in 1999, (✓) indicates that the site was selected as an extinct site because the species was recorded by Good and not relocated by Pearman and Byfield in 1994, but was subsequently rediscovered during the experiment in untreated controls or other parts of the site. 'M' indicates the plots were treated in May 2000. All other plots were treated in November 1999. *CF Cicendia filiformis, DA Drosera anglica, GA Genista anglica , HP Hammarbya paludosa, LI Lycopodiella inundata, PL Pinguicula lusitanica, RL Radiola linoides, RF Rhynchospora fusca*

Site	**Grid Reference**	*CF*	*DA*	*GA*	*HP*	*LI*	*PL*	*RL*	*RF*
Arne Heath mire	SY 974891								X
Bank Gate Common	SY 957865			✓					
Creech Heath	SY 925837						1997		
Creech Heath Footbridge	SY 923841					X			
Drove Hill, Winfrith Heath	SY 802860							X	
Grange Barn Field	SY 923843							✓	
Grange Mire (M)	SY 911839		✓			✓			
Half way House	SY 935845			✓					
Hartland Moor	SY 962856		1994	X	1970	X	X	X	
Horse track (east), Stoborough Heath	SY 925855	1998						1998	
Horse track (north), Stoborough Heath	SY 922855	✓						✓	
Jubilee Bridge, Hartland Moor	SY 947848			(✓)					
Nr. tramway drain, Three Barrows Heath	SY 936853		1985						
Nr. tramway, Stoborough Heath (north)	SY 936846			(✓)			X		
Slepe Roadside	SY 958869	X					X	X	
Soldiers Road Bog, Three Barrows Heath	SY 944864				1988		1990		

Table 5.3 cont'd

Site	**Grid Reference**	*CF*	*DA*	*GA*	*HP*	*LI*	*PL*	*RL*	*RF*
Soldiers Road Track, Three Barrows Heath	SY 944863							✓	
Soldiers Roadside	SY 945864	X					X	X	
Spur Bog	SZ 027853		✓			✓	✓		1994
Stoborough Heath (central)	SY 926848					X			X
Stoborough Heath (north)	SY 935859				1978				
Stoborough Heath (west)	SY 922855					1995			✓
Stoborough Heath mire	SY 927847						✓		
The Moors bog (M)	SY 951866		✓				X		
The Moors mire (M)	SY 950865		1991						
Thrasher's Heath track	SY 971852	✓						✓	
Three Barrows Mire (M)	SY 941859				✓		✓	✓	
Track (north), Arne triangle	SY 955867							1993	
Track, Arne triangle	SY 950866	✓						✓	
Track, Stoborough Heath (east)	SY 934834					✓			
Tramway mire, Three Barrows Heath	SY 935854		1994		✓		(✓)		
Turford Bog	SY 967888		✓						
Upper Fen Arm, Hartland Moor	SY 943845			(✓)			(✓)		
Whitcombe Hill, Winfrith Heath	SY 805876					X			
Whitcombe Vale, Winfrith Heath	SY 803866				✓				
Winfrith Heath (south)	SY 803860			✓		X	X		X

Analysis

Seedling emergence of the study species under disturbed (trampled and dug) and undisturbed treatments was compared by General Linear Model (GLM) using the percentage of samples in which the species of interest emerged. Data were arcsine transformed and blocked by site. Trampled and dug treatments were compared in the same way.

Vegetation composition was compared between treatments by GLM using arcsine square-root transformed mean percentage cover data for each treatment blocked by site. Vegetational differences between the sites resulted in a large number of low values for percentage cover of species present only at some sites. 19 species with an average percentage cover below 0.1 were excluded. Differences were accepted as significant when $P < 0.025$, as the analysis was repeated for 39 species. The percentage cover of seedlings was generally very low, therefore differences in the number of identified seedlings in each treatment were compared by GLM using untransformed mean data for each site.

5.3 Results

5.3.1 GLASSHOUSE SEED BANK EXPERIMENTS

Seedling emergence from samples taken from extant sites are summarised in Table 5.4 (species derived from seed rain are excluded). The existence of seed banks was shown for *Cicendia filiformis, Genista anglica, Pinguicula lusitanica* and *Radiola linoides,* but not *Drosera anglica* and *Hammarbya paludosa.* The sampling method may have been inappropriate for *D. anglica* (see below), as virtually no seedlings of any species emerged from the samples, but may have been adequate to show the presence of a propagule bank for *H. paludosa* if one existed, as seedlings of other species emerged. Two samples for *Pinguicula lusitanica* and *Genista anglica* did not produce seedlings of any species (including glass house weeds) and were discarded.

Table 5.4 Summary table of seedlings emerging from seed bank samples taken from extant sites for key wet heath and valley mire species in Purbeck, Dorset. CF - *Cicendia filiformis*, DA - *Drosera anglica*, GA - *Genista anglica*, HP - *Hammarbya paludosa*, PL - *Pinguicula lusitanica*, RL - *Radiola linoides*.

Species	**CF**	**DA**	**GA**	**HP**	**PL**	**RL**
Total no. seedlings emerged	59	0	4	0	12	87
No. samples	39	40	38	40	38	60
% seedlings from 0-2cm depth	96.7	-	100	-	-	100
% seedlings from 2-4cm depth	3.3	-	0	-	-	0
No. other species recorded (Mean +/- st.dev.)	34 (4.81+/- 2.01)	1 (one sample only)	39 (5.30+/- 1.97)	14 (1.97+/- 1.43)	15 (2.45+/- 1.35)	45 (4.48+/- 2.09)

Radiola linoides

A total of 87 seedlings emerged from the 60 *Radiola linoides* seed bank samples after six months (no further seedlings emerged after this period). None of these germinated from the 2-4 cm fraction, suggesting that the species did not form a persistent seed bank at the four sites sampled (Kruskall-Wallis, H = 117.00, P < 0.001). The number of emerged seedlings per sample varied between sites (Kruskall-Wallis, H = 15.66, P < 0.001), with 84% of the total number of *R. linoides* seedlings recorded from one site, Three Barrows track.

At this site, the fifteen core holes were still visible in July 1999 (when *R. linoides* was flowering, but not yet seeding). 200 seedlings were recorded from the side walls and base of the core holes. No significant difference was found between the density of seedlings on the walls between 0-2cm and 2-4cm depth and on the base. This suggests that a persistent seed bank may be present (although seeds may have fallen to the base of the core hole after the cores were extracted).

A total of 144 *R. linoides* seedlings (mean 9.6+/-6.4 per quadrat) was found in 15 20 cm^2 quadrats adjacent to the core holes. A comparison (corrected for area) between the number of seedlings recorded from the core holes, the glasshouse samples and the adjacent areas showed the number in the glasshouse samples to be significantly lower than in both the core holes and adjacent areas (Kruskal-Wallis H = 11.56, P = 0.003). The difference remained significant when seedlings from the base of the core holes were excluded in case seed had fallen from the sides. This suggests germination from the seed bank was reduced in the glasshouse (Figure 5.1).

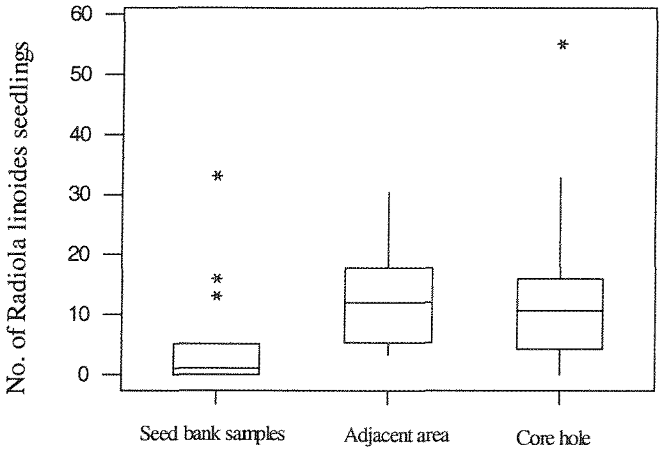


Figure 5.1 Number of *Radiola linoides* seedlings germinating at Three Barrows track (adjusted for area) within seed bank samples, adjacent areas, and seed bank sample core holes.

Cicendia filiformis

59 *Cicendia filiformis* seedlings germinated from six of the 39 samples within six months. No further seedlings emerged. No significant difference was found in the number of seedlings between sites or depths, although 57 of the seedlings (from 5 samples) were in the 0-2 cm fraction, while the remaining two seedlings germinated in one sample in the 2-4 cm fraction.

Genista anglica

Four *Genista anglica* seedlings were recorded from four of the 38 samples, three in the 0-2cm depth category and one in the 2-4cm category. No seedlings were recorded in the eight samples from Bank Gate common. None of the differences were statistically significant. All four seedlings germinated between January and March 2000, 10-12 months after the samples were prepared.

Pinguicula lusitanica

A total of 12 *Pinguicula lusitanica* seedlings germinated from five out of the 38 0-4 cm samples, again showing considerable clumping. All seedlings germinated between September 1999 and March 2000, 6-12 months after the samples were prepared. All seedlings were in samples from Spur Bog, with none germinating from the Winfrith Heath samples. However, the Winfrith core holes were relocated in September 1999, and seedlings were noted in several. The seedlings were larger that those likely to have germinated from the 1999 season's seed.

Drosera anglica

The *Drosera anglica* seed bank trials were unsuccessful. The *Sphagnum* deteriorated within two months, even when spread over a layer of peat substitute, and appeared unsuited to glasshouse conditions. A single plant of *Drosera rotundifolia* emerged from one of the 40 samples.

Hammarbya paludosa

Although *Hammarbya paludosa* seedlings were not recorded from the seed bank samples, several other bog species did emerge. This suggests that the technique used did not preclude the possibility of *H. paludosa* emerging should there be a viable buried seed/propagule bank present. The samples contained more humic matter than those for *D. anglica*, and the samples laid over peat substitute did not deteriorate.

5.3.2 FIELD SEED BANK EXPERIMENTS.

Regeneration of study species from the seed bank

Pinguicula lusitanica, Radiola linoides and *Cicendia filiformis* seedlings were all found in experimentally disturbed plots at both extinct and extant sites. Seedlings of the remaining species were not found at either. Emergence of key species in the field seed bank is summarised in Table 5.5.

There were no significant differences between trampled or dug plots, and therefore these were combined as 'disturbed' treatments. No significant differences were shown in the frequency of emergence in disturbed plots between extinct and extant sites (see Figure 5.2, Table 5.6). There was considerable variation in emergence between sites and between samples. The number of seedlings present in samples where they occurred fluctuated widely (e.g. 1-330 for *Pinguicula lusitanica*, 1 - 81 for *Radiola linoides*, and 1 - 39 for *Cicendia filiformis*).

fluctuated widely (e.g. 1-330 for *Pinguicula lusitanica*, 1 - 81 for *Radiola linoides*, and 1 - 39 for *Cicendia filiformis*).

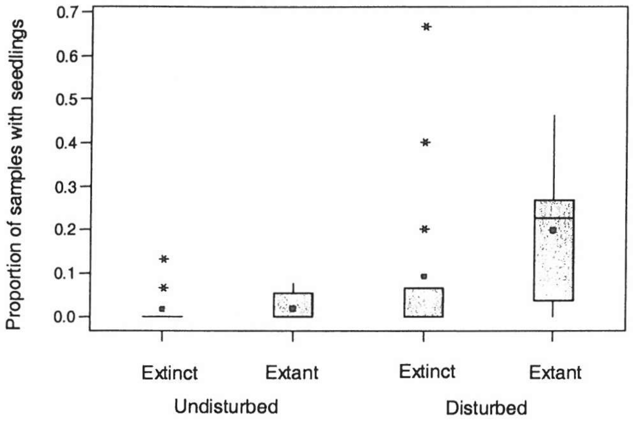
Table 5.5 Summary of seedling emergence in field seed bank experiments at sites supporting current populations and sites where above ground populations were not found

Species	**Extant sites**		**Extinct sites**	
	No. sites sampled	***No. sites where species found***	***No. sites sampled***	***No. sites where species found***
Cicendia filiformis	3	2 (1 vandalised)	3	1
Drosera anglica	4	0	4	0
Genista anglica	3 (3)	0	1	0
Hammarbya paludosa	3 (1)	0	2	0
Lycopodiella inundata	6	0	3	0
Pinguicula lusitanica	3 (2)	2 (2)	8	5
Radiola linoides	5	3 (1 vandalised)	5	1
Rhynchospora fusca	4	0	3	0

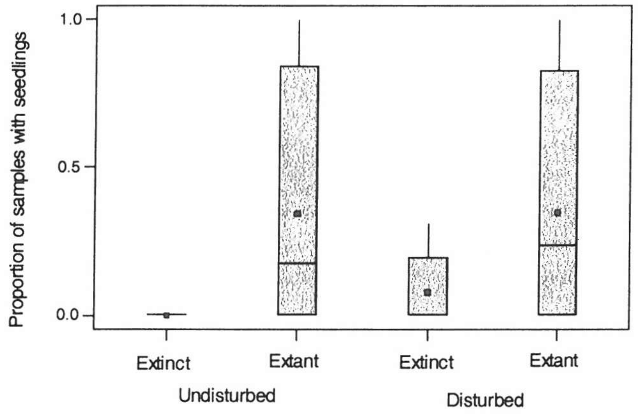
Table 5.6 Means and F-values generated by GLM for seedling emergence under disturbance treatment of the seed bank for *Pinguicula lusitanica* (PL), *Radiola linoides* (RL) and *Cicendia filiformis* (CF).

Species	**Population**	**Disturbed Mean (st.dev.)**	**Undisturbed Mean (st.dev.)**	**Treatment F- value**	**Site F- value**
PL	Extant	19.9 (15.2)	18.5 (3.4)	16.91**	4.19*
PL	Extinct	9.6 (18.6)	0	6.95*	3.45**
RL	Extant	46.4 (43.0)	45.4 (43.8)	0.17^{NS}	97.96***
RL	Extinct	7.6 (12.8)	0	3.62^{NS}	3.62^{NS}
CF	Extant	83.3 (99.9)	33.3 (3.8)	12.30^{NS}	30.16*
CF	Extinct	18.15 (25.4)	0	3.93^{NS}	3.93^{NS}

Pinguicula lusitanica emergence from the buried seed bank in experimental plots



Radiola linoides emergence from the buried seed bank in experimental plots



Emergence of Cicendia filiformis from the seed bank in experimental plots

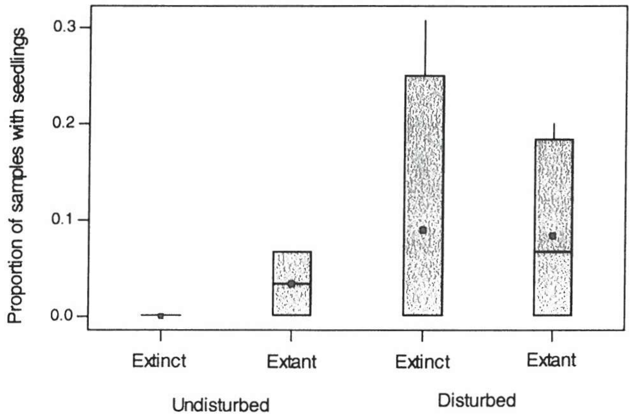


Figure 5.2 Boxplots of seedling emergence from the buried seed bank under experimental treatments. Boxes indicate interquartile range, * denotes outliers, and solid circles indicate the mean.

Small *Genista anglica* shoots were also found in disturbed plots at two extant sites. Careful excavation showed that these were growing from the roots of nearby mature plants (within 50 cm).

Regeneration of other wet heath and valley mire species from the seed bank.

Both disturbance treatments resulted in a significant increase in the cover of bare ground and standing water. This resulted in a significant decrease in the cover of most (21) species (Table 5.7). However there was also a significant increase in the percentage cover of *Drosera intermedia*, *Salix spp.,* and *Pinguicula lusitanica.* There was also an increase in *Gentiana pneumonanthe*, which was not found in any of the control plots (three seedlings were found in trampled plots at two sites). All plants which increased in disturbed plots were seedlings which probably regenerated from the buried seed bank.

A comparison of dug and trampled plots showed significantly more *Eriophorum angustifolium* (GLM, F = 4.62*), *Erica tetralix* (GLM, F = 22.71***), *Molinia caerulea* (GLM, F = 13.95**), and *Narthecium ossifragum* (GLM, F = 11.49**), in trampled plots than in dug plots.

Seedlings

There were significantly more seedlings of *Calluna vulgaris, Carex panicea, Drosera intermedia, D. rotundifolia, Potentilla erecta,* and *Ulex* spp. in disturbed plots compared to undisturbed plots (Table 5.8). There were also more seedlings of *Erica tetralix, Molinia caerulea, Narthecium ossifragum* and *Rhynchospora alba,* all species which were shown to have reduced cover following disturbance.

Table 5.7. Percentage cover (st. dev) of species under disturbed and undisturbed treatments plus F-values generated by GLM. * - $P < 0.25$, ** - 0.01, *** - $P < 0.001$. ↑ - increase in cover, ↓ decrease in cover.

Species	**Undisturbed**	**Disturbed**	**Effect**	**F value**
Bare ground	6.7 (9.3)	70.2 (17.9)	↑	465.9***
Water	10.1 (5.1)	19.2 (23.1)	↑	46.8***
Agrostis curtisii	0.9 (3.1)	0.2 (0.6)	↓	4.9*
Auloconium palustre	1.1 (4.5)	0.2 (0.7)	↓	2.2^{NS}
Calluna vulgaris	5.2 (6.3)	0.4 (0.8)	↓	56.9***
Campylopus introflexus	0.2 (0.8)	0.1 (0.1)	↓	4.0*
Carex panicea	0.5 (0.9)	0.4 (0.6)	↓	0.1^{NS}
Carex spp.	0.1 (0.5)	0.1 (0.5)	↓	0.1^{NS}
Cladonia. portentosa	0.3(0.7)	0.1 (0.2)	↓	6.7**
Danthonia decumbens	1.0 (4.0)	0.1 (0.3)	↓	2.5^{NS}
Drosera anglica	0.1 (0.6)	0.1 (0.1)	↓	0.6^{NS}
D. intermedia	0.2 (0.3)	0.5 (0.8)	↑	10.6**
D. rotundifolia	0.5 (1.1)	0.4 (0.6)	↓	1.6^{NS}
Erica ciliaris	4.3 (7.7)	1.0 (2.7)	↓	14.9***
Erica cinerea	0.3 (0.7)	0.1 (0.1)	↓	9.9**
Erica tetralix	16.1 (13.16)	1.6 (2.4)	↓	105.5***
Eriophorum angustifolium	1.2 (2.0)	0.3 (0.5)	↓	13.16***
Gentiana pneumonanthe	0	0.1 (0.1)	↑	1.6^{NS}
Juncus spp.	0.4 (1.4)	0.1 (0.3)	↓	2.3^{NS}
Molinia caerulea	24.4 (17.7)	5.3 (3.8)	↓	122.9***
Myrica gale	0.9 (2.0)	0.3 (0.7)	↓	8.4**
Narthecium ossifragum	30.9 (5.1)	1.5 (2.3)	↓	30.9***
Potentilla erecta	0.3 (0.5)	0.3 (0.5)	↓	0.6^{NS}
Pinus sp.	0.1 (0.4)	0.1 (0.1)	↓	0.1^{NS}
Pinguicula lusitanica	0.1 (0.1)	0.2 (0.5)	↑	6.4*
Potamegaton polygonifolius	0.1 (0.3)	0.3 (1.2)	↓	1.0^{NS}
Rhynchospora alba	3.9 (6.6)	0.9 (1.6)	↓	11.8**
R. fusca	0.4 (1.8)	0.1 (0.2)	↓	2.9^{NS}
Salix spp.	0	0.1 (0.2)	↑	15.8***
S. repens	0.1 (0.4)	0	↓	0.5^{NS}
Schoenus nigricans	1.8 (5.4)	0.4 (1.6)	↓	6.4*
S. compactum	1.2 (2.8)	0.2 (0.48)	↓	4.0*
S cuspidatum	2.1 (5.6)	0.6 (1.3)	↓	6.2*
S. papillosum	1.6 (3.0)	0.7 (1.2)	↓	9.5**
S. pulchrum	4.1 (8.8)	0.8 (1.9)	↓	15.2***
S. subnitens	0.6 (1.6)	0	↓	4.8*
S. tenellum	0.4 (0.8)	0.2 (0.6)	↓	2.7^{NS}
Trichophorum cespitosum	1.9 (3.8)	0.4 (0.5)	↓	11.4***
Ulex sp.	0.1 (0.2)	0.1 (0.1)	↓	2.4

Table 5.8 Mean number of seedlings (st dev) of each species (where mean no. of seedlings < 0.01) under disturbed and undisturbed treatments plus F-values generated by GLM. * = P < 0.25, ** = 0.01, *** = P < 0.001.

Species	Disturbed	Undisturbed	F value
Calluna vulgaris	0.98 (2.02)	0.02 (0.08)	4.4**
Carex panicea	0.08 (0.20)	<0.01 (0.01)	3.1*
Carex spp.	0.39 (1.38)	0.01 (0.07)	1.6
Cicendia filiformis	0.23 (0.91)	0.03 (0.13)	0.9
Drosera anglica	0.12 (0.76)	0.15 (0.95)	1.0
D. intermedia	2.12 (3.84)	0.41 (1.01)	4.0**
D. rotundifolia	1.25 (2.58)	0.26 (0.62)	3.3*
Erica tetralix	10.03 (12.08)	0.83 (1.98)	9.2***
Gnaphalium uliginosum	0.14 (0.63)	0.02 (0.13)	1.0
Juncus articulatus	0.40 (0.15)	0.11 (0.47)	1.1
Molinia caerulea	0.14 (0.25)	<0.1 (<0.1)	4.0**
Narthecium ossifragum	0.49 (1.28)	0.04 (0.19)	2.8*
Pinguicula lusitanica	0.53 (1.33)	0.14 (0.56)	2.7
Plantago major	0.36 (1.84)	0.11 (0.68)	1.02
Polygala serpyfolia	0.12 (0.37)	0.01 (0.04)	0.87
Potentilla erecta	0.32 (0.58)	0.04 (0.12)	4.9**
Potamegaton polygonifolius	0.15 (0.52)	0.01 (0.08)	1.19
Radiola linoides	0.22 (0.89)	0.16 (0.67)	0.45
Rhynchospora alba	1.24 (1.94)	0.11 (0.36)	6.6**
Salix spp.	1.07 (2.81)	0.23 (0.62)	2.0
Trichophorum cespitosum	0.23 (0.56)	0.15 (0.72)	0.7
Ulex spp.	0.26 (0.45)	0.03 (0.08)	5.6**

Other species for which seedlings were found were *Agrostis capillaris, Agrostis curtisii, Crepis capillaris, Danthonia decumbens, Erica ciliaris, Gentiana pneumonanthe, Hypochaeris radicata, Juncus bulbosus, Myrica gale, Pedicularia minor*, *Salix repens* and *Scutularia minor.* There were no significant differences in the number of seedlings of each species between trampled and dug plots. There was a trend for a greater total number of seedlings in trampled plots (23.39 +/-16.29) than dug plots (18.91 +/- 13.86), although this difference was not significant.

5.4 Discussion

5.4.1 REGENERATION OF KEY SPECIES

Radiola linoides, Cicendia filiformis, and *Pinguicula lusitanica* regeneration from buried seed banks was stimulated by simulated livestock trampling at sites where the above ground populations were extinct. *Genista anglica* also formed a buried seed pool, but livestock presence seems more likely to stimulate vegetative regeneration.

In addition to the experimental plots, *Cicendia filiformis*, *Pinguicula lusitanica* and *Genista anglica* were also found following disturbance at sites where they had been considered extinct, or had not been recorded recently. A second *C. filiformis* population was found south of the main population at Great Ovens in May 2000 following forage harvesting the previous winter. Two small populations of *P. lusitanica* were found in 2000 in separate valley mires on Hartland that had been trampled by cattle and ponies, and two further populations of *G. anglica* were also found on Hartland. However, there were no previous records of these species at these sites, which did not fall inside any of Good's stands, and colonisation may have occurred through dispersal.

It is not clear to what extent the two annual species, *Radiola linoides* and *Cicendia filiformis,* formed persistent seed banks at the sites sampled. Both species are summer annuals, germinating in mid to late May in Purbeck (see Chapter 4). The seeds of summer annuals maturing at the end of the summer have at least conditional dormancy, preventing them from germinating until the following spring (Baskin & Baskin, 1998). Cores for glasshouse trials were taken in March, therefore it is possible that the seeds that germinated were solely from a transient seed bank formed by the previous season's seeds. Virtually no seedlings emerged from the 2-4 cm fraction of the cores sampled, which could have indicated the presence of a persistent seed bank (Thompson *et al.*, 1997). However, seedlings were observed in the base of the core holes at one site sampled for *R. linoides*, although these may have fallen into the base once the core was removed In addition, germination occurred in disturbed plots at an extinct site in the field experiments. As both species had been observed at this site two years previously (Pickess, 1998), this persistent seed bank may nonetheless be short-term. However, *Radiola linoides* is known to form a buried seed bank where it occurs in dune slacks (Bekker *et al.,* 1999), remaining at low but detectable numbers even after the vegetation has developed beyond the early successional phase preferred by this species.

At the extant site where neither *Cicendia filiformis* nor *Radiola linoides* were recorded from disturbed plots, the populations of both species appeared to have become all but extinct, with only two *C. filiformis* plants and no *R. linoides* plants found in 2000, and none in the undisturbed control plots. Two of the extinct sites sampled for both species were roadsides. The roads had been metalled subsequent to Good's observations, and the roadside verges appeared relatively eutrophied, both conditions which may have greatly reduced the likelihood of the species surviving.

The number of seedlings of both species found within disturbed plots within extant populations was usually greater than the number within the adjacent undisturbed plots (Figure 5.3), suggesting that livestock trampling can provide a suitable mechanism for the regeneration of this species from its buried seed bank. In the case of *Radiola linoides*, this is probably through seeds being brought to the surface, and a reduction in canopy cover. *Cicendia filiformis* germination may be further stimulated by the increase in water level (see Table 5.1 and Chapter 3) associated with livestock trampling.

Fewer seedlings established from samples for the two perennial species for which seed banks were found (*Pinguicula lusitanica* and *Genista anglica*), and *G. anglica* was not found in the field experiments. However, up to 350 *P. lusitanica* seedlings were found in 30 cm x 30 cm treatment plots, suggesting that this species can form a significant buried seed bank. *P. lusitanica* germinates in the autumn after dispersal, therefore the seedlings germinating the following summer were likely to be from a persistent seed-bank. This was verified by the field experiments, in which *P. lusitanica* germinated from sites where it had become extinct between 60 and 6 years previously. Livestock presence may have a positive effect on the regeneration of *Pinguicula lusitanica* from the seed bank through exposing seeds and also increasing the cover of standing water (Chapter 4) (see below).

Genista anglica seedlings can also germinate in the same season in which they disperse (Chapter 4), therefore seedlings germinating in the glasshouse experiments may have been from a persistent seed bank. Legume seeds have hard coats leading to physical dormancy, and can therefore form large seed banks with high longevity (Rice, 1989). However, no seedlings emerged following experimental disturbance in the field, and only one seedling emerged from the 2-4 cm depth cores. Plants in Purbeck, although flowering profusely, produced few seed pods. Hester *et al.* (1991a & b) did not find this species in the seed bank when sampling a birch wood in which it was present in the standing vegetation. The lack of a large persistent seed bank suggests this long-lived species does not rely on a large seed bank. This is supported by the fact that *G. anglica* regenerates vegetatively very readily. A small fragment of root found in one of the cores produced a sizeable flowering plant a year later, and vegetative shoots were found in trampled plots after 18 months. In Purbeck and the New Forest, larger plants are generally surrounded by smaller plants comprising single or few shoots growing from the roots of the original plant, and individuals often cannot be readily distinguished. In the New Forest, *G. anglica* is considered to persist well in a vegetative state in adverse conditions (C. Chatters, pers. comm.). The plants that appeared following grazing at Hartland Moor (Table 5.1) were found to have thick woody stems, and were clearly not young plants. The species is notoriously difficult to find except when in flower, and evidently had been obscured by *Molinia caerulea* tussocks. While livestock presence may not enhance regeneration from the seed bank it may encourage growth in, and vegetative regeneration from mature plants.

Two additional perennials, *Drosera anglica* and *Hammarbya paludosa,* were not shown to form seed banks in either the glasshouse or field experiments, and again, both can reproduce vegetatively (see Chapter 4). However, livestock trampling was not shown to directly promote regeneration. Recent work by Baskin *et al.* (2001) implies that *D. anglica* may form a transient seed bank - seeds were dormant by late summer, and germinated after cold stratification, suggesting germination after winter in the field. However, trampling was shown to fulfil emergence requirements in the field (Chapter 4), therefore any seeds exposed by trampling would be expected to germinate. *D. anglica* can form abundant seed – Crowder *et al.* (1990) reports an average of 6.3 capsules per plant and 47-252 seeds per capsule, and the plants from which seed was collected in Dorset had a similar quantity. However, comparison between the number of seedlings that emerged in field emergence experiments and those that emerged in glasshouse emergence experiments suggest that a large proportion of seeds may have been lost in the field. Seeds (and seedlings) can float (Crowder *et al.,* 1990) and may have been washed out of trampled plots. *D. anglica* plants are found within the wettest, relatively stable, valley mires and populations are small, therefore it seems unlikely that their persistence is reliant on a large persistent seed bank.

No seed banks have been recorded for any members of the *Orchidaceae* family (Thompson *et al.*1997). It is possible that, if *Hammarbya paludosa* seeds were present in the glasshouse samples, conditions were not suitable for the emergence of seedlings of this species. Germinating orchid seeds lack a radicle and usually a cotyledon, so a greater degree of development and growth is needed before a seedling is produced. In addition, *Hammarbya paludosa* forms mycorrhizal associations (Nilsson, 1977) which may not have persisted in the glasshouse environment. However, no seedlings were found in the field experiments either. *Hammarbya paludosa* appears to set seed rarely (seeds were not found at any of the three population in Purbeck between 1998 and 2000). It has two small pseudobulbs, the lower one of which is from the previous years growth, and may be able to persist as a tuber even if conditions for growth are unfavourable. It also reproduces vegetatively, forming small bulbils on the ends of leaves. Therefore, this species is not reliant on a seed banks to maintain populations in adverse conditions.

The perennial sedge, *Rhynchospora fusca* did not show regeneration from seed bank in the field experiments, although studies of lakeside populations in Canada have shown the presence of a long-term persistent seed bank (Nicholson & Keddy, 1982), as have sod-cutting experiments in Germany (Kesel & Urban, 1999). *R. fusca* is rhizomatous, and can rapidly spread vegetatively (Pearman, 1994), and its reliance on seeds may be limited to initial colonisation of a site (Kesel & Urban, 1999). In some of the plots within extant populations re-visited 18 months after disturbance in this study, a small amount of vegetative regeneration was apparent. The site at which *Rhynchospora fusca* apparently regenerated from the seed bank (see Table 5.1) was wet heath rather than valley mire, with a firm substrate possibly subject to seasonal inundation. The experimental sites for *R. fusca* were all valley mire, with one exception. However, this site is drying out and the population appears to be declining. Germination requirements may not have been fulfilled at this site. Two further populations were found in 2002 on wet heath tracks (one in an area which had been searched unsuccessfully on three separate occasions in 1999 and 2000, as the species had been previously recorded there). Both tracks had been used by heavy vehicles over the winter, and in spring 2002 a large number of seedlings emerged on the disturbed edges and ruts, suggesting the presence of a seed bank (Plate 5.2). Seed banks may play a greater role in wet heath populations of this species, where seed is less likely to be washed away and the substrate forms a more suitable medium for germination when disturbed (see also section 5.5.3).

Plate 5.3 *Rhynchospora fusca* growing in vehicle ruts

In the populations studied, *Rhynchospora fusca* was serotinous, i.e. the seed crop was retained after seed maturity, forming an aerial seed bank, and thus changing the germination phenology from autumn to the following spring. Given that many *R. fusca* sites are on seasonally inundated areas (see Chapter 4), this may be a means of ensuring the seeds are dropped when the water level has reduced and bare peat is exposed. However, any seedlings found the subsequent spring in trampled plots would therefore not necessarily have been from a seed bank No seedlings were found in either May 2000 or May 2001, six months and 18 months after disturbance.

Similarly, a spore bank was not found for *Lycopodiella inundata.* There has been recent speculation in England (Stewart *et al.,* 2000) about the possibility of this species regenerating from a buried spore bank, largely due to its reappearance at Tadnoll Heath in Purbeck, one kilometre from the nearest current population, although such dispersal differences are possible for tiny spores, e.g. Okubo & Levin (1989). Sansen & Koedam (1996) imply that a *L. inundata* spore bank was found following sod-cutting at a site in Sweden, although its emergence was considered to be limited by very low pH values. At Mordon Bog in Purbeck, *Lycopodiella* spp. spores were buried under an experimental earthwork, and were relocated 9 years later when the earthwork was excavated (Evans & Limbrey, 1974). Clearly, they had not deteriorated, but their viability was not established. While there is some evidence for the existence of a viable spore bank for this species, this study has not shown that regeneration from the spore bank can occur following livestock trampling. Possibly insufficiently large areas were exposed – Sansen & Koedam (1996) do not give the dimensions of their sod-cut plots, but the scrape at Tadnoll was several metres square, and the majority of *Lycopodiella inundata* sites in Dorset are within the Lulworth (Povington) Ranges on areas previously heavily disturbed by tank exercises. The size of the experimentally disturbed plots was limited due to concerns expressed by site managers about the degree of apparent destruction being inflicted on vegetation stands of conservation importance. However, areas trampled by livestock can be much larger than the experimental plots, for example at areas of focal interest such as ponds. Possibly such larger-scale disturbance could potentially provide suitable conditions for regeneration from the spore bank. Further, larger scale experimental work at carefully selected extinct sites could clarify this.

5.5.2 REGENERATION OF OTHER SPECIES

In addition to the species already discussed above, seed banks were found for 35 species, one of which - *Gentiana pneumonanthe* - was found at sites where it was not present in the undisturbed sward. For 10 of these species, there were significantly more seedlings in disturbed plots than in undisturbed plots. *Sphagnum* species may also have regenerated from a spore bank (Sundberg & Rydin, 2000). However, cover reduced overall, and it was not possible to differentiate vegetative *Sphagnum* from *Sphagnum* regenerating from spores in the field. Sundberg and Rydin (2002) suggest that regeneration from spores may be important in *Sphagnum* following disturbance, although nutrient release from litter and cover from vascular plants may be needed. Note that the experiments did not necessarily show which species were present in the seed bank, but which of those species present in the seed bank would regenerate under these particular conditions. Exposure at different times of year, for example, may produce different results (e.g. Thompson, 2000). All seedlings were considered to have regenerated from the seed bank except *Salix* spp, although these were significantly more abundant in disturbed plots. Most *Salix* species are recorded as having a transient seed bank (Thompson *et al.*, 1997). Seeds are dispersed from April, and seed was frequently noted on most of the heathland sites studied (Plate 5.3). It is likely that the recorded seedlings were a result of seed rain, and had not regenerated from a buried seed bank.

There were some differences between trampling and turf inversion treatments, although this did not result in significant differences in the number of seedlings for any species.

Trampling had less impact on the cover of the most abundant species (e.g. *Molinia caerulea*, *Eriophorum angustifolium*, *Erica tetralix*, and *Narthecium ossifragum)* but resulted in higher cover of standing water than digging. Microtopographical variation also appeared greater in trampled plots. Results from Chapter 4 suggest that both these factors potentially enhance seedling emergence, while the higher degree of shading may not be detrimental to seedling emergence (although possibly affecting subsequent establishment and growth). There were more seedlings in trampled plots than dug plots, although the high variation between sites and between plots on sites meant that this difference was not significant.

Plate 5.4 *Salix* spp. seed caught in *Erica tetralix* shoots on Stoborough Heath, Purbeck, in May 2000.

5.5.3 IMPORTANCE OF BURIED SEED BANKS FOR THE CONSERVATION OF WET HEATH AND VALLEY MIRE

It was not possible within the scope of this study to record the effect on plant community composition of regeneration from the buried seed in wet heaths and valley mires, because this is also dependent on subsequent seedling survival and development. However, it is possible to make some predictions about the potential for enhancement of degraded wet heath and valley mire sites from the buried seed bank through livestock trampling. At the sites studied, although species diversity had reduced since the 1930s (Byfield & Pearman, 1994), the vegetation was still defined as wet heath or valley mire. Vegetation changes had not been sufficient for the vegetation community description to change (e.g. to scrub or sallow carr), and so the role of livestock trampling in enhancing vegetation, rather than changing vegetation type, is considered here.

Regeneration from seed banks may lead to a change in plant community composition if species' representation in the seed banks is different from their representation in the exiting vegetation. Therefore the differences in seed bank persistence between species can be exploited to increase species diversity. The presence of an extensive long-term persistent seed bank for *Calluna vulgaris* is well-established (e.g. Gimingham, 1972) and similar seed banks have been recorded for *Erica tetralix* (Pywell *et al.,* 1997) and *Erica cinerea* (Thompson & Band, 1997). Evidence of a seed bank for *Erica ciliaris* was also found in this study. In contrast, *Molinia caerulea* is considered to have only a short-term persistent seed bank (Thompson *et al.,* 1997), while *Carex panicea,* both *Drosera* species, *Polygala serpyllifolia*, *Potentilla erecta*, *Rhynchospora alba, Ulex galli* and *U. europaeus* are recorded as having at least short-term and in some cases long-term persistent seed banks. Therefore, once the dominant *Molinia caerulea* has been reduced, for example by grazing (Chapter 3) or by trampling, it is likely to be replaced by those species with longer-term persistent seed banks, which includes characteristic species of wet heath and valley mire that are likely to be target species for restoration.

However, non-target species may also regenerate – *Ulex* spp. seedlings were more abundant in disturbed plots, and *Betula* sp. seedlings were also found. Both *Betula* and *Ulex* spp. have persistent seed banks (Thompson *et al.,* 1997). Given a sufficiently frequent reccurrence of trampling, the seed banks of these species may be depleted, as they take much longer to reach maturity than the majority of the likely target species. However, colonisation via seed rain may also change species composition once gaps have been opened by trampling. Again, this may not necessarily be by target species, particularly if these are rare and source populations are distant. The increase in *Salix* spp. in disturbed plots in this study clearly demonstrated this, and recruitment of this species can continue provided that the external seed source persists.

The intensity and repetition of trampling (or other disturbance) will also influence changes in vegetation composition. If seed germination is stimulated, but seedlings repeatedly never reach maturity and produce seed with which to replenish the seed bank, clearly the population will become extinct both above and below ground. The continual enhancement of rare populations through seed bank exposure may in fact degrade a community if appropriate conditions for establishment are not also fulfilled (Parker *et al.,* 1989; Bakker *et al.,* 1996). This may be an advantage of using livestock as a tool to facilitate regeneration from the seed bank at degraded sites, rather than mechanical means such as turf stripping. For example, cattle trampling can create favourable conditions for the germination of characteristic wet heath and valley mire species (Chapter 4), and consideration of habitat descriptions for extant sites suggests that requirements for subsequent establishment may also be met through livestock grazing and trampling (also see Chapter 4). The variation created by livestock in microtopography, water cover, canopy cover (see Chapter 3) and also through patchy use of habitats (see Chapter 2) mean that the requirements of a variety of species are more likely to be met across space and time.

Deliberately stimulating species regeneration in this way presents an interesting problem for conservationists – to what extent, and how frequently, should regeneration from the seed bank be stimulated? Typical seed bank species show bursts of population growth in response to opening in the canopy, and then decline – therefore conservation managers may wish to wait until this decline is evident before trying to enhance regeneration, rather than try to maintain individual populations at constant level.

Another important source of regeneration in this study was buried root fragments. *Eriophorum angustifolium, Genista anglica, Molinia caerulea, Narthecium ossifragum, Rhynchosphora alba, R. fusca* and *Trichophorum cespitosum* were all observed regenerating vegetatively from buried roots or buds. This was particularly noticeable in trampled plots, where there were generally more roots and shoots remaining. Perennials from habitats such as wetlands commonly use vegetative propagation to survive disturbance and environmental fluctuations (Parker *et al.,* 1989), and terms such as propagule bank (e.g. Van der Valk & Davies, 1979) and bud bank (e.g. Putwain & Gillham, 1990) have been used to describe reserves of dormant regenerative stages other than seeds. In his study of the seed bank of boreal bogs, Jauhiainen (1998) noted that germination from seed was low, and many of the species present regenerated through vegetative expansion from adventitious or buried propagules. Vegetative propagules may behave functionally as seeds, however their longevity is not generally known (Leck, 1989). Although the greatest densities of seeds recorded from any habitat so far have been from an Appalachian bog (McGraw, 1987), vegetative regeneration may be more important in valley mire, and to a lesser extent, wet heath. The reasons for this were apparent in valley mire plots, where disturbance to *Sphagnum* spp. together with the roots and shoots of vascular species that form the surface of valley mires, resulted in a wet mush. The surface was often flooded, and there was no solid substrate or intact *Sphagnum* providing a medium for seed germination. In slightly drier mires and on wet heaths there was generally sufficient peat present to prevent this. No seedlings were found in the wettest valley mire treatment plots, and even vegetative regeneration was limited even 18 months later, when plots were still predominantly bare. In marshland, Van der Valk & Davies (1979) concluded that the potential of the propagule bank to contribute to peatland regeneration was rather poor, and that vegetative growth is rather more important than reproductive strategy. (See also discussion in Chapter 4 of the reproductive strategies of wet heath and valley mire species), and this is likely to be the case for valley mires.

5.5 Conclusions

- The re-establishment at extinct sites of three species associated with wet heaths - *Cicendia filiformis, Radiola linoides* and *Pinguicula lusitanica* - was successfully achieved by simulated livestock trampling;
- Vegetative regeneration from roots and buds may be more important than regeneration from seed for longer-living species or those characteristic of more stable, wetter mires. Vegetative regeneration was shown to be stimulated by livestock trampling for *Genista anglica.*
- In addition to facilitating initial regeneration, livestock presence may provide appropriate conditions for the ongoing establishment of wet heath and valley mire species, leading to the successful re-establishment or expansion of populations;
- Propagule banks were not demonstrated for *Drosera anglica, Hammarbya paludosa*, *Rhynchospora fusca* or *Lycopodiella inundata* in this study. Further work is required to assess the importance of propagule banks in the conservation of *R. fusca* and *L. inundata* on wet heath sites;
- Regeneration from propagule banks is a dynamic process, and populations may not appear stable. Attempting to maintain stable above-ground populations of rare species may lead to their extinction both above and below ground.

6. Summary and general discussion

6.1 Introduction

Livestock are used increasingly as a management tool on heathland sites to encourage diversity in both species composition and structure. Although work has been done previously on both the behaviour and effects of livestock on upland heathland, grazing of lowland heath has only been studied in detail in the New Forest. The New Forest is a unique and very different system to the majority of lowland heaths in the UK. The aim of the research described in this thesis was to investigate the use of livestock as a conservation management tool, particularly to restore diversity on heaths where successional processes had led to a decline in species. The ways in which domestic livestock use lowland heathland sites in general and how they affect wet heath and valley mire vegetation in particular were explored. Habitat selection, general behaviour and diet were studied in both cattle and pony herds grazing on lowland heathland sites in Purbeck, in Dorset, southern England (Chapter 2). Changes in community composition and structure together with changes in environmental variables following reintroduction of cattle were explored experimentally (Chapter 3). The potential effect of livestock on the germination of selected characteristic wet heath and valley mire species was investigated and considered in the context of the species habitat requirements (Chapter 4). The potential of livestock trampling to expose buried seed banks and promote regeneration of the species studied in Chapter 4 was investigated experimentally both in glasshouses and in the field (Chapter 5).

6.2 Livestock behaviour

Three cattle and four pony herds were studied on four lowland heaths, all of which are nature reserves where livestock had been reintroduced as a management tool. The management aims were to increase species diversity, reduce dominant and invasive species such as purple moor-grass *Molinia caerulea* and scrub, and to enhance overall structural diversity. Chapter 2 considered whether livestock used sites in a way that might achieve such objectives.

6.2.1 OVERALL HABITAT SELECTION

All livestock groups showed non-random use of sites, and used particular habitat types while avoiding others. Habitats supporting a high cover of fine grasses or young heather *Calluna vulgaris* were positively selected over those supporting coarser grasses or other woody species. The considerable amount of time spent foraging suggests that this was a major determinant of habitat selection, and this is supported by seasonal changes and by observations of dietary selection (see below).

Vegetation management was observed to affect habitat selection. Previous studies have shown general avoidance of dry heath communities. However, further division of dry heath according to heather structure showed that dry heath types supporting young *Calluna* (building dry heath and tracks) were positively selected. These areas had been created by various management techniques including mowing and litter stripping which resulted in an increase of young *Calluna* growth. At one site, degenerate dry heath characterised by old leggy heather bushes was abundant enough to be classified separately, and this habitat was preferentially selected in autumn for resting, although largely avoided for the rest of the season.

Other seasonal changes were the notable decrease in use of wet heath and valley mire over winter, which was generally compensated for by an increase in use of acid grassland, restoration heath or dry heath. Changes in the use of acid grassland were probably dependent on the area of acid grassland available, as where this was limited there was greater use of dry heath habitats. Other changes were more site-specific, for example, use of woodland by cattle reduced in autumn at one site, but increased at another. Differences between herds were also apparent, for example, use of restoration heath by New Forest ponies at Hartland peaked in mid-winter, but remained little used by Exmoor ponies at this time.

6.2.2 USE OF HABITATS

The extent to which habitats were used for particular behaviours varied, although habitats were never used exclusively for one behaviour. Habitats used predominantly for foraging were those with abundant grass cover such as acid grassland, valley mire, wet heath, and tracks, plus restoration heath, saltings and saltings margins where these were present. Of these, acid grassland, restoration heath, tracks and saltings margins were positively selected for foraging. Of the less grassy habitats, management resulting in a higher cover of grasses and/or younger growth of *Calluna vulgaris* led to an increase in foraging (e.g. on building dry heath, managed scrub and woodland). For resting, wet habitats were generally avoided and building dry heath, mature dry heath, woods and managed scrub/wood were selected. Tracks, roads and dry heath were the only habitats used predominantly for moving, although several habitats were positively selected. Livestock generally moved along small paths when moving across dry heath, and were only observed moving through vegetation when foraging for dwarf gorse. Dunging was generally in proportion to habitat use, although there was a tendency for dunging to be higher around resting locations, possibly resulting in the transfer of nutrients and plant species between foraging and resting habitats.

Seasonal changes in habitat use for particular behaviours included a reduction in use of valley mire and wet heath for foraging in autumn/winter. Again, other changes were more herd-specific – for example, use of dry heath for foraging increased in cattle at Arne in 1999, but did not increase in other herds.

6.2.3 DIET

On all habitats, livestock generally positively selected grasses and avoided other species. *Calluna vulgaris* was positively selected on building dry heath where young shoots were available. Scrub species were occasionally eaten, but never positively selected, with the exception for dwarf gorse *Ulex minor*, which was positively selected in autumn by cattle at one site. Otherwise, there was little seasonal variation in species selection within habitats. Supplementary feeding had a very strong impact on dietary selection, and little else was selected when supplementary feed was available. There was some indication that scrub browsing increased when supplementary feeding was carried out, but it was not clear whether this was as a result of changes in dietary requirements, seasonal changes in availability, or the fact that supplementary feeding was carried out in a location with abundant scrub, and all grasses in the area were heavily trampled and covered in dung.

6.2.4 FACTORS AFFECTING HABITAT SELECTION

Overall, habitat selection was largely determined by the availability of suitable forage, generally grasses. Resting requirements were also important, and inclusion of resting periods in analyses notably increased the selection for dry habitats. Locations used for resting were also closer to shelter than would be expected by chance, and seasonal changes in habitat selection (such as the increase in use of degenerate dry heath for resting at Arne, of the increase in use of woods at Hartland) were related to shelter.

Habitat use was also affected by the distribution of habitat patches. Less favoured patches located between preferred patches were necessarily used in travelling between them. Although site-specific, this increased the use of (mature) dry heath at all sites, which was used predominantly for moving. The accessibility of habitat patches was important. For example, seasonal variation in use of particular habitat patches was influenced by the decrease in ranging distance in winter, with fewer habitat patches away from the main foraging areas being used.

The location of water sources was also important, as livestock generally visited a water source at least once in each 24-hour observation, and therefore had to move to habitat patches where water was available. Supplementary feeding also had a considerable effect on habitat selection, and when this took place livestock rarely moved away from the feeding areas except to drink. The only time when interactions between individuals within a herd were observed to have an effect on habitat selection was during supplementary feeding. Less dominant individuals were forced away from the feed and were observed to forage in adjacent scrub and woodland rather than move away from the rest of the herd to a more preferred habitat. Direct interactions between livestock species and breeds were only ever observed in particularly small habitat patches, or at drinking sites. For example, Exmoor ponies appeared dominant over New Forest ponies, and both were dominant over cattle. This occasionally resulted in the less dominant herd changing habitat patch (although they tended to move on to another patch of the same habitat type).

6.2.5 DIFFERENTIAL BEHAVIOUR

Although general trends were apparent across the suite of sites and herds studied, there were differences between them. As discussed in Chapter 2, these may have been due to differences in livestock type and/or site characteristics. Notably, ponies spent considerably more time foraging than cattle, and therefore removed a greater biomass. However, cattle are heavier, and the impact of trampling was presumably greater. They also spent more time lying down, and again their greater body mass is likely to have resulted in more vegetation damage. At Hartland Moor, New Forest ponies (who travelled shorter distances) were less selective in their habitat use than both cattle and Exmoor ponies, and made most use of habitats other than the preferred acid grassland, including wet heath and valley mire. However, cattle made more use of dry heath than ponies. Cattle ate more *Calluna* than ponies when on dry heath, but otherwise dietary selection was similar between livestock groups. There were more total observations of ponies browsing scrub than cattle, although overall selection was similar.

There were also differences between breeds, types and sites. For example, Exmoor ponies were more selective than New Forest ponies, and lactating cattle were more selective than non-lactating cattle. All livestock at Hartland appeared considerably more selective than the cattle at Arne, who, although spending much of their time on acid grassland, spent a roughly equivalent time on saltings, and most time on building dry heath. There was also some variation in habitat selection between individuals within herds. Site-specific characteristics such as habitat patch size and distribution also influenced habitat selection (see section 6.1.4).

6.2.6 LIVESTOCK BEHAVIOUR AND IMPACTS ON VEGETATION

These behavioural observations suggest how livestock will affect the vegetation on complex heathland sites such as those studied. Firstly, there will be a differential impact on different habitats. Biomass removal, trampling and dunging will be greatest on grassy habitats and will remain high throughout the year. The effects are likely to be evenly distributed spatially (although ponies may cause more patchy effects by use of particular areas for latrines, which are then avoided for grazing). The impact on managed dry heath and tracks will also be relatively high, although it may show more seasonal variation. The impact on dry heath will be relatively low, and will be largely confined to localised damage to *Calluna* plants through trampling and lying, plus nutrient enrichment and the introduction of grassland species through dunging around resting sites. Use of wet heath and valley mire is likely to be limited, although foraging may be high where the vegetation is dominated by grasses. Trampling damage will also be greatest around pools used for drinking.

Within each habitat, impacts will be different for different species. Grasses will experience the greatest biomass reduction, as they are preferentially grazed. Young *Calluna* plants may be eaten, but older bushes and other ericaceous species are more likely to be damaged by trampling or crushed by lying than grazed. *Ulex minor* may be selectively eaten in autumn, but otherwise scrub species are unlikely to be preferentially selected, although this will vary according to livestock type.

However, impacts also relate to the susceptibility of vegetation types or species, not just how much they are used, and this was explored for wet heath and valley mire in particular.

6.3 Impacts of grazing on heath vegetation

6.3.1 VEGETATION COMMUNITY RESPONSE

The impact of cattle on wet heath and valley mire was explored in Chapter 3. The results broadly support the un-replicated and mainly opportunistic studies reviewed in Chapter 1 (e.g. Cox, 1998; Sanderson, 1994; Clarke, 1988). Cattle had a marked effect on both wet heath and valley mire communities within three years. Changes in vegetation structure were apparent after the first grazing season, and differences were reinforced over the subsequent two seasons. Changes in environmental variables, which have not previously been studied on wet heath and valley mire, were particularly striking. Livestock trampling increased variation in surface microtopography on both vegetation types and resulted in a greater cover of surface water and bare ground. Substrate compaction on wet heath was increased by trampling, but decreased in the valley mire. However, changes in species composition were slow. Cattle grazing reduced the dominance of graminoids such as *Molinia caerulea*, and increased the diversity of forbs, while *Sphagnum* species appeared vulnerable to trampling. Changes in ericaceous species were more subtle, but cattle grazing was associated with a slight decline in cover. Valley mire appeared more sensitive to grazing than wet heath, even though grazing pressure on valley mire was less than that on wet heath (Chapter 2). This was partly a consequence of the greater initial cover of graminoids (which were selectively grazed by cattle), and partly because the wetter substrate was more susceptible to trampling.

Results from both wet heath and valley mire showed that the cattle created a diversity of gap types through both biomass removal and trampling. Species responses to these changes at Coombe Bog may take longer than the three years of this project to become apparent. Longer-term work is needed to establish to what extent livestock grazing will lead to changes in overall vegetation type. For example, at stream and pool-side sites at both Hartland Moor and Stoborough six to eight years of grazing appears to be shifting thick *Molinia caerulea* - dominated swards towards short, grassy and relatively herb-rich swards similar to the streamside lawns found in the New Forest (e.g. Byfield & Pearman, 1994; Wright & Westerhoff, 2000).

6.3.2 INDIVIDUAL SPECIES' RESPONSES

The wet, open conditions resulting from livestock presence can provide suitable germination and establishment conditions for a suite of characteristic heathland species (Chapter 4). Emergence in the field from sown seed of great sundew *Drosera anglica*, petty whin *Genista anglica* and pale butterwort *Pinguicula lusitanica* was increased by artificial cattle trampling. However, only *D. anglica* also responded to biomass removal through cutting. Glasshouse experiments confirmed that the locally increased water levels enhanced germination of these three species, and yellow centaury *Cicendia filiformis*, but artificial shading was not shown to be important. Allseed *Radiola linoides* was not affected by either increased water levels or increased green shading. However, with the exception of the woody, long-lived *G. anglica*, apparent habitat requirements for these species plus marsh clubmoss *Lycopodiella inundata*, bog orchid *Hammarbya paludosa* and brown-beak sedge *Rhynchospora fusca* suggest that the absence of shading may be important for subsequent establishment and growth, as all are found in un-shaded locations

Trampling will also expose existing propagule banks, and may lead to the regeneration of annual and short-lived perennial species absent in the above-ground vegetation (Chapter 5). Populations of *Cicendia filiformis, Pinguicula lusitanica* and *Radiola linoides* were successfully regenerated by experimental trampling at sites in which they had been extinct from the above ground vegetation. *Genista anglica* also forms a buried seed bank, but none was discovered by experimental trampling. There was some indication that the creation of canopy gaps may, however, stimulate vegetative expansion in this species. The existence of a buried seed/spore bank was not demonstrated for *Drosera anglica, Lycopodiella inundata, Hammarbya paludosa* or *Rhynchospora fusca.* However, anecdotal evidence suggests that *R. fusca* regeneration from a buried spore bank may occur.

Differences in life-cycle characteristics can be used to explain these results. As unproductive habitats subject to relatively low levels of disturbance, valley mires are likely to support species that rely on vegetative expansion, reducing the need for resource expenditure on seed production (Grime *et al.,* 1988). Exposure of seed banks through trampling will not play a prominent role in the conservation of populations. However, trampling may expose propagules other than seeds and promote vegetative expansion, and this was shown for *Genista anglica.* Long-lived species such as *G. anglica* are also less reliant on seed production for persistence. However, annuals and short-lived perennials reliant on seedling recruitment, and particularly those characteristic of disturbed habitats such as *Cicendia filiformis, Pinguicula lusitanica* and *Radiola linoides*, are more likely to create a persistent seed bank. A different conservation strategy is therefore needed for different wet heath and valley mire species, and in Chapter 4, some characteristic wet heath and valley mire species are grouped according to potential grazing management aimed at maximising population.

6.4 Implications for the conservation management of lowland heaths

Specific questions relating to livestock grazing on heathland were considered in previous chapters. To realise the potential of this work in informing conservation action, it needs to be considered within the framework of the wider issues of heathland grazing. This section uses the results presented in previous chapters to consider to what extent specific conservation objectives are likely to be met through livestock grazing and trampling, and how this can be manipulated to optimise results.

6.4.1 POTENTIAL FOR ACHIEVEMENT OF MANAGEMENT OBJECTIVES

The key objective for using livestock in the conservation management of lowland heath is the creation or maintenance of conditions suitable for the persistence, growth or re-colonisation of populations of desired heathland flora and fauna species. This is likely to be achieved through the halting of successional change (control of invasive species) and creating a disturbance regime which leads to variation in vegetation composition and structure. Can livestock grazing and trampling as described in this study achieve this?

Control of invasive species

Without disturbance, wet heath and mire are likely to become dominated by the competitive grass *Molinia caerulea*, whose tussocky growth and deciduous habit will lead to an increase in organic matter and change in surface water level, allowing subsequent scrub colonisation and eventual development to carr woodland (Chapter 1). Similarly, on dry heath, invasion by scrub is the first step to conversion to woodland. On many dry heaths, dominance by wavy hair-grass *Deschampsia flexuosa* is also increasingly a problem, although this is not common on Dorset heaths.

Livestock preferentially select grass species on heathland habitats (Chapter 2). At appropriate stocking rates, this can reduce the dominance of invasive grasses such as *Molinia caerulea*. This was seen at Coombe Bog, where a reduction in *M. caerulea* cover was associated with cattle grazing (Chapter 3), and has been reported at a number of other sites (Chapter 1). However, grass species were introduced to dry heath communities through cattle dung, although their persistence was not studied. The subsequent grazing regime will be one of the factors influencing whether species introduced through dung become established. Variable livestock pressures were not studied in this project, but the literature shows that high or low grazing may both lead to an increase in grasses at the expense of ericaceous and herbaceous species on upland heathland (Chapter 1), and this is likely to occur on lowland heathland. Grazing alone may not be suitable for controlling invasive grasses on eutrophied sites, for example, grazing of *Deschampsia flexuosa* was not considered adequate to control this species on eutrophied dry heaths in the Netherlands (e.g. Bokdam & Gleichman, 2000),.

It is not clear to what extent domestic livestock grazing can be used to manage regenerating tree and scrub species. Grazing has excited much interest as an alternative to mechanical and chemical control techniques, as these can be both expensive and time consuming. For example, the current Tomorrow's Heathland Heritage project in Dorset allows from £700 ha^{-1} to £2500 ha^{-1} (depending on scrub cover) for scrub clearance (Moore, 2000). However, initial overheads for grazing projects are also high - approximate costs for using 10 goats over two years to clear scrub from 10ha on a heath in Surrey totalled £1217 ha^{-1} (R. McGibbon, pers comm.). This involved confining browsing livestock within a relatively small area in which scrub was dominant. Using free ranging livestock which are grazers rather than browsers is likely to be far less effective. At the sites studied in Purbeck, cattle and ponies generally avoided mature and degenerate dry heath (where scrub invasion is high), and rarely browsed scrub species. It seems unlikely that livestock will have much impact on scrub populations at these sites. However, other livestock types have been used with varying degrees of success to reduce scrub at sites elsewhere (see Lake *et al.*, 2001), and the example of the New Forest shows that at sufficiently high densities, stock may prevent scrub invasion. Discussion with site managers and casual observation also suggest that livestock may also prevent or reduce re-growth from scrub that has been previously managed by cutting. This area merits further study.

Provision of suitable conditions for persistence, growth or regeneration of populations of characteristic heathland species

The possible impacts of grazing and trampling on heathland Species of Conservation Concern (Biodiversity Information Group, 2001) were assessed in Lake *et al.* (2001), and therefore this work is not repeated here. It was shown in Chapters 4 and 5 that livestock trampling and grazing could provide suitable conditions for the regeneration and possibly the persistence of a number of scarce wet heath and valley mire species. One of the key elements in this was the creation of a diversity of gap types. Similarly, the SoCC assessments showed that heathland species require a diversity of conditions. For example, species such as coral necklace *Illecebrum verticillatum* and Hampshire purslane *Ludwigia palustris* require very heavily grazed and poached swards. A relatively open sward is necessary for heath lobelia *Lobelia urens,* but this may be susceptible to heavy grazing during flowering. Similarly a range of requirements is found on other heath types such as the dry grass heath of the Brecks in East Anglia, where species like spring speedwell *Veronica verna* and Breckland thyme *Thymus serpyllum* require a very short sward, but field wormwood *Artemisia campestris*, another short sward species, only persists where grazing is excluded.

This diversity of requirements is not limited to plant species, and is even more complex in insects. For example, insects such as the heath bee fly *Bombylius minor*, the silver-studded blue butterfly *Plebejus argus* and the heath tiger beetle *Cicindela sylvatica* require bare areas for nest building, the warm microclimate, or for hunting respectively. However, the beetle *Cryptocephalus coryli,* the dingy mocha moth *Cylophora pednularia* and the cranefly *Nephrotoma sullingtonesis* are dependent on heathland scrub (birch *Betula* spp, willow *Salix* spp. and Scot's pine *Pinus sylvestris* respectively). Many species also have highly specialised requirements, for example the Purbeck mason wasp *Pseudopipona herrichi* needs bare clay near a water source for its burrows, and provisions its nest with the caterpillar of a particular moth species, which itself prefers short *Erica cinerea* dominated dry heath (Roberts & Else, 1997). Therefore, livestock grazing and trampling must have a variable impact if the requirements of a variety of such species are to be met. Given the large number of species in question (e.g. over 60 insect species of conservation concern alone), it will be difficult to establish to what extent the impacts of grazing and trampling will be beneficial for each species. Establishing whether livestock can produce a diversity of conditions provides a short cut to assessing whether grazing and trampling will on balance be beneficial to the heathland flora and fauna. The specific impacts studied on wet heath and valley mire vegetation have been discussed above. Here the potential for the creation of diversity within heathland sites as a whole is approached by considering livestock behaviour.

On a site-scale, diversity encompasses (i) the different habitat types present, (ii) differences in the species composition of individual habitat patches, and (iii) diversity in vegetation structure within or between habitat patches. The potential role of livestock in halting succession (thus loss of particular habitat types) has been considered above. Here, the potential for this to vary across a site and the potential for the creation of variation within habitat types are considered. Diversity may be achieved through grazing and trampling if this is heavier in some areas than others. For example, if differential grazing pressure is achieved on wet heathland communities, livestock may reduce dominance of *Molinia caerulea* and promote the regeneration of a diversity of plant species in some areas, while elsewhere leaving more tussocky *M. caerulea* growth suitable for, e.g., smooth snake *Coronella austriaca* and adder *Vipera berus*. Intermediate pressure may optimise conditions for species such as the Red Data Book large marsh grasshopper *Stethophyma grossum*, found on open quaking bogs where there is some tussocky *Molinia caerulea*. On dry heath, livestock may increase the bare ground vital to a number of invertebrates species at resting sites or paths, while avoiding remaining areas and thus leaving reptile sites undisturbed.

So the success of livestock in creating a diversity of conditions depends on non-random use of sites. Non-random use of habitat types was shown in Chapter 2. Use of habitat patches may also be non-random. For example, use of a patch of habitat that is generally avoided may be increased if the patch is located between two preferred habitat types. Or it may be decreased if a patch is partially cut off from the rest of the site and access is difficult. Intensity of use by livestock may also vary within a patch. Use of dry heath is likely to be highly patchy. For example, on the Purbeck heaths, both cattle and ponies showed a strong preference for grasses within the heathland sward (Chapter 2), and this preference may result in increased grazing of heather surrounding grassy patches (e.g. Hester & Baillie, 1998). Similarly, use of mature dry heath was largely confined to paths, and therefore grasses and *Calluna* alongside paths were more grazed and trampled than those further from the paths. On wet heath, grazing and trampling may be greatest around drinking pools, as was seen at Arne. The deepest areas of valley mires were avoided on all sites. Livestock grazing alters the sward, and this itself may increase patchy use. Grazing will reduce the sward height and promote young growth, which may be more nutritious and therefore preferred by livestock. Livestock are therefore likely to concentrate on areas where they have changed the sward to the optimum height and quality (Fryxell, 1991). This was seen at Hartland, where particularly tall, tussocky *Molinia caerulea* patches were avoided and the stock tended to use areas they had already grazed. So livestock behaviour suggests that there is the potential for a diversity of conditions to be created, and this may be further enhanced through exploitation of the factors causing variable habitat use.

6.4.2 MANIPULATION OF LIVESTOCK BEHAVIOUR TO ACHIEVE MANAGEMENT OBJECTIVES

The precise impact of livestock grazing and trampling on a particular site will be dictated by the livestock type, stocking rate, grazing season, the habitat types present, the state of the vegetation (i.e. amount of forage available), the distribution of habitat patches and the location of tracks, water and shelter. While making site-specific predictions about the likely impact of livestock difficult, these factors may be manipulated to fine-tune grazing to maximise the potential for meeting management targets. The importance of livestock type and stocking rate in heathland conservation grazing have been reviewed in Lake *et al.* (2001). These are mentioned in passing here, but discussion is largely limited to the application of results presented in this thesis.

Site infrastructure

There are a number of ways in which site infrastructure may be adjusted to influence livestock behaviour. The first of these is the size of the grazing unit. It was seen in Chapter 2 that variation in habitat use within a herd was relatively low (although there were exceptions to this). Therefore, even if stocking rates are the same within two grazing units, one small and the other large, the variation in grazing intensity may be greater at the larger site. Herding behaviour may mean that the greater number of animals in the larger site is not necessarily reflected in a greater dispersion of animals across the site (provided forage does not become limiting). Thus, at the larger site, grazing pressure may be higher on those areas of habitat frequented by the herd. Likewise, the greater area available to the herd at the larger site will mean that it is more likely that some vegetation patches will be grazed rarely or not at all (however, this will be dependent on a number of factors including optimum herd size for the livestock in question, the distribution and size of habitat patches across the grazing units, and whether forage becomes limited at particular habitat patches). It suggests that free-range grazing may not be a suitable tool for small sites unless enough is known about the distribution and requirements of other species present. On small sites, there will be much less diversity in the impact of grazing and trampling. Livestock are therefore more likely to have a detrimental effect on some species present, which may themselves be more vulnerable on a small site.

The second way in which sites may be adapted to influence livestock behaviour is in the inclusion of various habitat types within the grazing unit. In some areas of Britain, heaths are still part of a cultural landscape including pasture or other grazing land (such as saltings), arable land, scrub, and woodland, with which their use was once integrated (Webb, 1998). While there is little reason to include agriculturally improved land within heathland grazing units, other semi-natural habitat types such as unimproved grassland, saltings, scrub, or wood are sometimes still available. Selective habitat use by livestock means that inclusion of these habitats will influence the grazing and trampling pressure on heath habitats. In general, this is likely to decrease pressure on heath habitats, which may not be considered desirable. Work in the Netherlands (Bokdam & WallisDeVries, 1992) has shown that cattle perform better where they have a greater choice of habitat types. Inclusion of non-heath habitats may make grazing possible (particularly year round) on sites where the heath habitats alone could not adequately support a herd. In addition, inclusion of particular habitat patches may in fact increase pressure on heath habitats. For example, in this study the ponies at Godlingston Heath made most use of both dry and wet heath habitats while travelling between two patches of acid grassland, one at either side of the site. Exclusion of one of these patches would probably have reduced the time ponies spent on the heath habitats.

There are also practical reasons for the inclusion of different habitat types. An adjacent area of acid grassland may be included within the overall grazing scheme to provide layback land to which livestock may be removed if the principle habitats become overgrazed, or stock lose condition or become ill. It may also provide a suitable site for supplementary feeding (see below). In some cases, site managers prefer to have a particular field to which the stock return regularly in order to facilitate checking and treatment.

Related to grazing unit size and character is the issue of fencing, which at most sites is a prerequisite of grazing. One approach is to fence habitats within the site separately so that the timing and density of stock grazing can be carefully controlled in each (e.g. Harris & Jones, 1998). The alternative is to create an extensive system with only a perimeter fence, allowing stock to move between habitats at will (such as at the sites studied in Purbeck). The first approach provides a high degree of control, and can be used where livestock presence may impact detrimentally on other species of conservation concern such as ground nesting birds, or where a site manager wants to control the timing of grazing impacts (such as on flower-rich maritime heath) or concentrate livestock activity in a particular areas (for example for scrub control). It may also be used in conjunction with knowledge of livestock behaviour to focus activity on (or away from) particular areas according to management aims. Preferred habitats such as acid grassland can be opened up to decrease pressure elsewhere, or they may be shut off to increase pressure. Opening particular patches may also be used to influence livestock behaviour on adjacent patches (see discussion of Godlingston above). More extensive systems tend to be used where there are less specific requirements, and offer a more natural grazing system as livestock behaviour is not inhibited. They may lead to a greater range of grazing pressures at a given stocking rate and therefore increased ecological diversity (see above).

Livestock use of sites may also be manipulated through the location of shelter and water sources. Resting locations were found to be closer to shelter on the Purbeck heaths, and there was also some indication that sheltered areas were generally used more in winter. This suggests there is the potential to provide (or remove) shelter to influence use of an area. For example, scrub sheltering a resting site on degenerate dry heath could be removed to discourage use of that location. Similarly, a copse left on wet heath near a water source may encourage stock to linger on the wet heath. Water sources were used at least daily by livestock on the Purbeck heaths, and herds often moved specifically to reach them. This suggests that habitat use may be influenced by the location of troughs or pools, which can therefore be positioned, dug or fenced off according to management aims. Poaching around pools is considered important for species such as lesser flea-bane *Pulicaria vulgaris*, pillwort *Pilularia globulifera* and pennyroyal *Mentha pulegium* (e.g. Byfield & Pearman, 1994) and trampling could be increased to improve conditions for such species by shutting off alternative water sources. In a similar way there is potential for fencing to be used to manipulate trampling pressure. For example, fencing and gates could be used to create 'pinch points', focussing trampling around a particular areas and creating suitable conditions for species such as yellow centuary *Cicendia filiformis* (Chapter 4), pygmy rush *Juncus pygmaeus* or three-lobed crowfoot *Ranunculus tripartitus* (R. Lansdown pers comm.).

Livestock at Hartland Moor showed a strong positive selection for tracks. Although positively selected for a number of behaviours, tracks (and mown firebreaks) were used predominantly for moving. Both cattle and ponies were also observed to stick largely to paths when crossing areas of dry heath. This suggests that tracks or paths could be created to influence livestock use of dry heath communities in particular, keeping them away from areas of concern for reptiles, or introducing them to areas otherwise little used. In fact, vegetation management in general could be used to help influence ranging behaviour. On all the sites studied, livestock were found to select preferentially managed vegetation over the equivalent unmanaged vegetation. Burning has been noted to increase grazing pressure at other sites (e.g. Bullock & Pakeman, 1997), and was traditionally carried out to increase forage availability (Allchin, 1997). Management such as cutting or burning can be used to attract livestock to an area. Again, this could also be used to increase use of adjacent habitat patches, or attract livestock away from areas where trampling or grazing may have an adverse effect.

Livestock type and management

Livestock type will have a strong influence on the outcome of grazing. There are considerable behavioural difference between species and between breeds (Section 2.1). In addition, there are likely to be differences between individuals depending on age (e.g. Provenza & Balph, 1987;1988)), gender (e.g. Gordon, 1989c) and background (e.g. Gill, 1987; Biquand & Biquand-Guyot, 1992; de Beaulieu, 1998). The exploitation of these differences and interactions between livestock types in conservation grazing on heaths is reviewed in Lake *et al.* (2001).

The impact of livestock on heathland is also dependent on the actual grazing and trampling pressure. This is often estimated as stocking rate (livestock units $ha^{-1} yr^{-1}$). Current stocking rates used for conservation grazing on lowland heathland sites vary between 0.03 and 0.50 LU $ha^{-1} yr^{-1}$ (Lake *et al.*, 2001). Although stocking rate may be useful for broad comparisons of livestock impact, the impact of livestock at any particular stocking rate will vary according to livestock type, vegetation characteristics, site characteristics and husbandry, and therefore its usefulness in predicting vegetation change is limited. Utilisation rate, as used in studies in the uplands (e.g. Armstrong *et al.*, 1997) may be more useful, but has not yet been applied to lowland heaths.

There is evidence that livestock on heaths were traditionally shepherded (Chapter 1), often being brought off the heaths at night to facilitate nutrient transfer onto arable land, or being moved around the heath to maximise forage intake. Although still technically feasible, daily shepherding is unlikely to be carried out today. However, where sites are divided into smaller grazing units, livestock can be moved between these as described above, on a diurnal, weekly or seasonal basis as is appropriate. In particular, seasonal changes in livestock behaviour may also be exploited according to management aims. For example, on the Purbeck sites, grazing on wet heath and valley mire was greatest in late summer, therefore grazing at this time will maximise the impact on this habitat. Scrub browsing was greatest in winter, when little other forage is available, and this would be the optimal time to impact on scrub, particular as there is some suggestion than gorse browsing in summer may stimulate growth more than it reduces it (Gallet & Roze, 2001b). Use of dry heath by cattle for foraging and resting increased in autumn, and stock should be on site during this period if an impact is required on these habitats. Conversely, concerns for reptile sites, which may be vulnerable to trampling (e.g. Mouton & Corbett, 1999), may mean that stock are removed before they start to disperse over dry heath in the autumn. Changes in ranging behaviour are also important, as livestock are likely to move further and use a greater number of habitat patches in summer compared to winter. Changes in herding behaviour may also be seasonal - the Hartland cattle herd split into more groups in late summer, than early summer or autumn, therefore grazing and trampling pressure was most dispersed over the site at this time.

It is common farming practice to provide supplementary food for livestock maintained outdoors through the winter. In most cases supplementary feeding is undesirable when grazing is carried out on lowland heathland for conservation benefits. Any form of supplementary feed may concentrate feeding, trampling, poaching, dunging and urination in small areas around the feed. The feed itself may provide inputs of nutrients and other chemicals into the surrounding soil, and introduce plant propagules. The presence of supplementary feed may provide a disincentive to livestock to use more nutritious forage present on the site (such as *Ulex* spp. in winter)(R. Ekins, pers. comm.). However, judicial supplementary feeding may allow grazing to continue on vegetation whose nutritive value is insufficient to maintain the well-being of livestock. In such situations, the problems of intensified trampling and nutrient input may be minimised by restricting feeding to an area of limited conservation interest. Mineral supplements in the form of licks are often used due to the low mineral status of heathlands (e.g. Bokdam & WallisDeVries, 1992) and protein feed blocks can be used to stimulate stock to increase their intake of roughage. Because supplement blocks alter behavioural patterns (Chapter 2) there is the potential for this to be used beneficially to manipulate patterns of site use, attracting livestock into areas they may otherwise ignore. The difference between different kinds of supplementary feed should be considered – for example hay is likely to alter behaviour more strongly than a mineral lick, as livestock will spend longer eating hay than a mineral lick.

This project has been approached from the angle of the impact of livestock grazing on heaths, and therefore has not considered any agricultural implications (many conservation grazing schemes also have agricultural objectives e.g. (Grayson 1997; 2000). Neither has it directly addressed animal welfare. Livestock managers need to be particularly aware of welfare issues in conservation grazing schemes, as there may be conflicts between animal welfare and grazing requirements. This is most likely when grazing particularly nutrient poor vegetation (such as heathland swards) or when looking for an impact on species that are not selected by the livestock unless preferred forage is not available. In particular, the concept of extensive grazing schemes where the aim is to allow free-ranging stock to be as self-maintaining as possible and replicate the ecological role of extinct large wild herbivores (e.g. Van Wieren, 1989; WallisDeVries, 1993; WallisDeVries, 1995), raises welfare concerns. This system has been tried in the Netherlands (e.g. Bruinderink & Kuiters, 2000), and it seems unlikely that the welfare compromises would currently be found acceptable in Britain. Further discussion of the Dutch model can be found in Kampf (2000). However, extensive grazing schemes on large sites where stock are by no means wild but are nonetheless free to behave naturally are carried out in Britain, for example at the sites studied in Purbeck. Tolhurst (2001) provides a comprehensive review of welfare requirements and legal obligations in the context of nature conservation grazing.

6.4.3 OTHER ISSUES

Livestock grazing can provide a versatile tool with which to create a diversity of conditions on heathland sites, and may be focussed to achieve particular management aims. However, grazing may not be suitable at all sites. There are a number of situations in which constraints to grazing may be prohibitive, and a review of practical constraints (such as urban reserves, lack of layback land, fencing problems) can be found in Lake *et al.*, (2001). Grazing is likely to be variable and impacts are not precisely predictable. This means that monitoring must form an integral part of any grazing regime. Grazing is also unlikely to provide the solutions to all management problems on heathland sites, and should not be considered to preclude the necessity for other management activity. For example, it is unlikely that grazing livestock will reduce scrub cover on sites with grassy habitats available (although particular livestock types in particular circumstances may achieve this – such as goats fenced into a patch of scrub). Similarly, the effect of grazing livestock on heather structure has not been directly studied on lowland heaths. Although livestock behaviour suggests that a diversity of heather types may be created or sustained through grazing and trampling, this remains to be tested. It is likely that grazing in conjunction with cutting or burning will achieve the best results. Heaths were historically used for grazing livestock in combination with associated activities such as burning to increase forage availability, in addition to sod and vegetation cutting, mineral extraction and other activities (Chapter 1). So grazing alone did not shape the ecology of heathlands, and there is no reason to expect that management through livestock alone will achieve conservation objectives today. Other management techniques will be required in addition to grazing to optimise conservation outcomes – it would be a mistake to consider grazing schemes as panacea for heathland management. This is particularly the case where restoration rather than maintenance management is being carried out.

However, there are a number of non-ecological benefits that grazing management can confer which differentiate it from other management techniques, and these are worth mentioning briefly. Firstly, there are possible socio-economic benefits. Grazing schemes, particularly those with concurrent agricultural objectives, play a key role in the reintegration of heathlands into the cultural landscape within which they arose (although it should be noted that in some cases agricultural objectives may compromise conservation objectives). Firstly, grazing requirements mean that other parts of the landscape are likely to be incorporated within a grazing scheme (for example for winter grazing). This has the potential to lead to the gradual functional reintegration of heaths into the surrounding landscape. Heathlands are particularly vulnerable to destruction because of their redundancy and isolation within the contemporary landscape, and as semi-natural habitats they will always be vulnerable to changes in human management. Linking them with other habitats will reduce this isolation. Grazing schemes also have the potential to link heathlands back into the local community – and any such link must enhance the sustainability of heathland management. This will happen where local graziers, agricultural suppliers and stock-people are employed. In addition, because the agricultural viability of conservation schemes is currently questionable despite conservation support payments (e.g. Grayson, 2000), value-adding techniques are increasingly used to enhance profits. For example, this may be through adding premiums on sourced products (e.g. 'local', 'organic' or 'conservation'), farm gate sales, or use of local farmers markets. These are generally approaches that require integration with the local community. Heathlands are more likely to be conserved (and enjoyed) if people understand them and feel a link with them.

The second advantage of grazing management is that it promotes a particular conservation perspective. Since the advent of biodiversity planning in the UK following the 1992 Biodiversity Convention, nature conservation has become increasingly targeted at particular species and habitats. In effect, attention has been drawn away from ecological processes and focussed on the outcomes of these processes, units with which conservation success can be measured. While in many cases this has been beneficial for the individual species or habitats involved, there are times when it is inappropriate, and a more process-orientated view would be helpful. Take as an example the regeneration of 'extinct' species from the seed bank (discussed in Chapter 5). *Pinguicula lusitanica* is a species that apparently forms abundant seed banks (Chapter 5), and flowers prolifically following management, but then its population dwindles. The tendency may therefore be to repeat management regularly to ensure a constant above ground population. But how can we decide which stage of this plant's life cycle should be targeted for conservation? Perhaps this species evolved to spend a significant amount of time in the form of seeds, but there is not much room for this eventuality in the target-orientated evaluation of its conservation. In fact, for some species, constant management may deplete the seed bank and lead to a longer- term decline in populations. Such a species needs a conservation system that values the process of its re-colonisation from the seed bank as much as its resulting above-ground presence. Livestock grazing is compatible with such as system. It is complex and variable, and although this means it is less predictable than other management options, it allows greater diversity of processes to occur. Using such a management technique will help remind us that perhaps natural processes are as important as the outcomes of these processes, while helping to ensuring that they are allowed to operate.

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Appendix I Arne Cattle 2000

Data relating to the 24-hour Arne cattle follows in 2000 referred to in Chapter 2 are presented here.

Routine

Figure 1 shows behaviour during each 24-hour follow period. Figure 2 shows the routes taken during each 24-hour observation period.

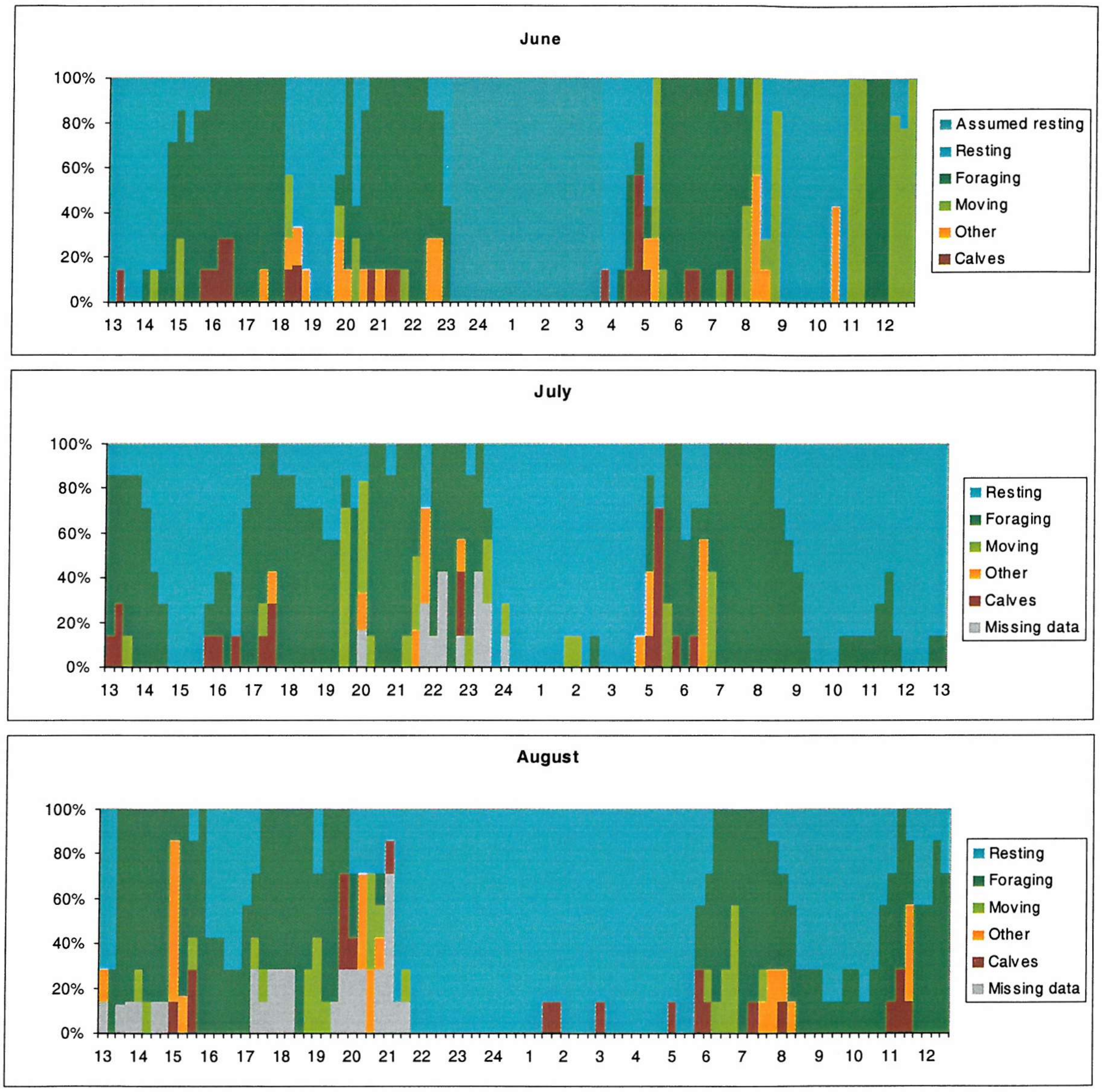


Figure 1 Behaviour during each 24-hour observation periods by cattle at Arne in 2000. The proportion of the herd carrying out each behaviour is shown on the vertical axis, and the time in hours along the horizontal axis. 'Assumed resting' represents night-time periods when the herd was not followed.

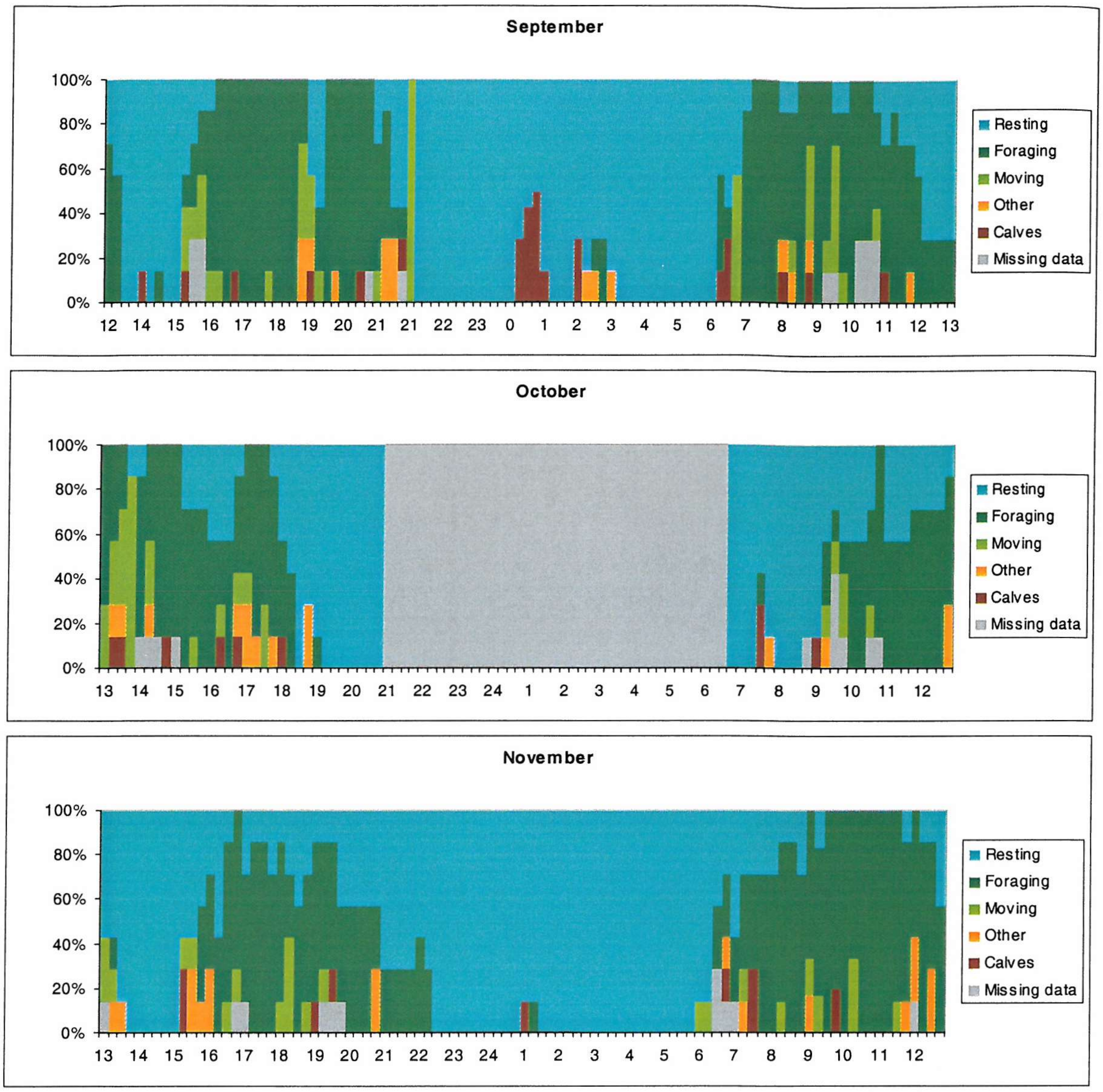


Figure 1 cont'd.

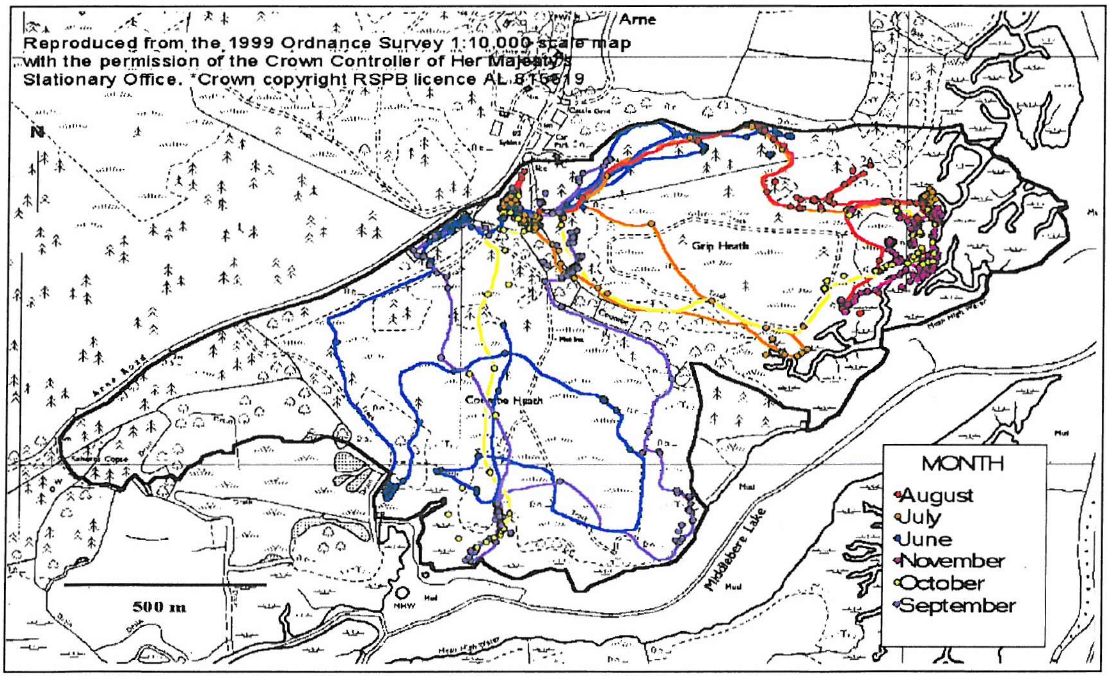
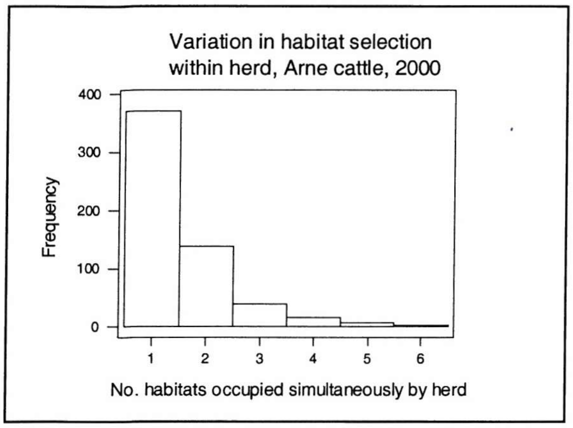


Figure 2 Routes taken by cattle at Arne, 2000, over each 24-hour period.

Individual variation



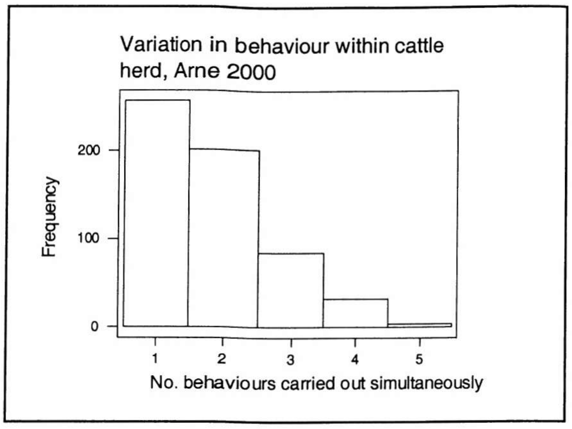


Figure 3. Variation in habitat selection measured as the number of habitats that the herd occupied simultaneously (including calves).

Figure 4. Variation in behaviour measured as the number of habitats which the herd occupied simultaneously.

24-hour habitat selection

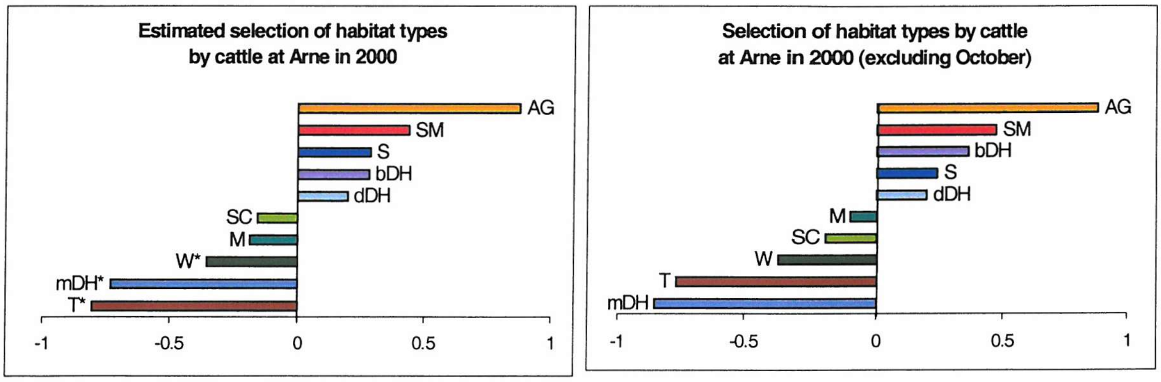


Figure 5. Habitat selection expressed as Jacobs' indices using pooled data over the grazing season. Habitat use was non-random (chi square = 16556.4, df = 9, p<0.001). * denotes habitats for which overall selection was significantly different from zero (Wilcoxon signed rank test, P < 0.05). See Table 2.2 for abbreviations.

During the October observation the herd moved overnight. Habitat selection was therefore calculated using estimated observation numbers. The herd grazed for three hours less than in the remaining 24-hour observations. Data from 1999 and from the Hartland herd suggest that total foraging time remained constant throughout the season. They were located in degenerate dry heath (dDH) just inland from the saltings at daybreak, and did not move to forage for about 3 hours. As the cattle had wet, muddy legs, it was assumed they had grazed on the saltings overnight. Therefore, 3 hours of foraging observations on the saltings were substituted for 'assumed resting' observations. The remaining 'assumed resting' observations were split between the evening and morning resting locations

There are only very slight differences in overall habitat selection using the two data sets (Figure 5). The data set including estimated habitat use for October is used for all further analysis as it is considered to give a more accurate representation of overall habitat selection.

Behaviour

Estimated behavioural observations for October were based on averages from the remainder of the 24-hours observations, as these showed a small standard error (Table 1).

Table 1. Estimated behavioural observations for October, for cattle at Arne in 2000

	Resting (%)	Grazing (%)
Mean	50.4	27.4
SE	2.49	2.43

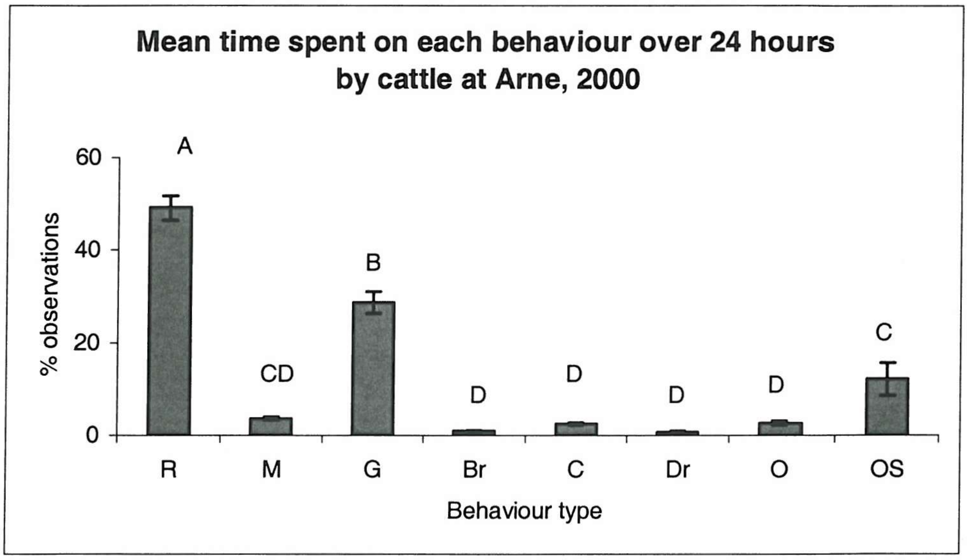


Figure 8. Mean time spent (i.e. percentage of observations) on each behaviour by cattle at Arne, 2000. The difference in mean time spent on each behaviour was significant (GLM, F = 73.04, P < 0.001). Significant differences are denoted by different letters; therefore, behaviours with the same letter show no significant difference. R – resting, M – moving, G – grazing, Br – browsing, C – calves, Dr – drinking, O – other, OS, out of sight.

Behaviour in calves was similar (Figure 9), although less time was spent grazing than by cows. No calves were observed browsing. Data for October were excluded, as calf behaviour could not be reliably estimated in the three hours assumed to be spent grazing by the cattle.

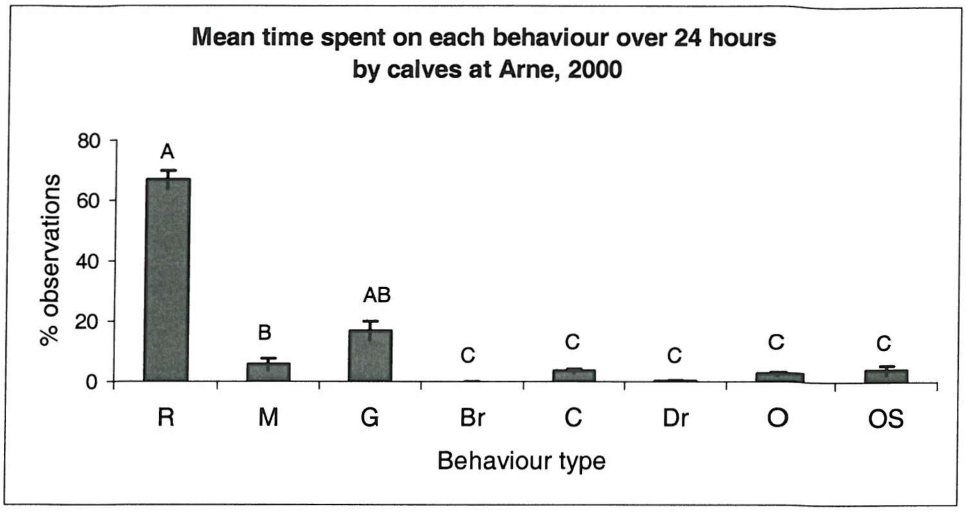


Figure 9. Mean time spent (i.e. percentage of observations) on each behaviour by calves at Arne, 2000. The difference in total time spent on each behaviour was significant (GLM, F = 14.09, P < 0.001). Significant differences are denoted by different letters; therefore, behaviours with the same letter show no significant difference. R – resting, M – moving, G – grazing, Br – browsing, C – calves, Dr – drinking, O – other, OS, out of sight.

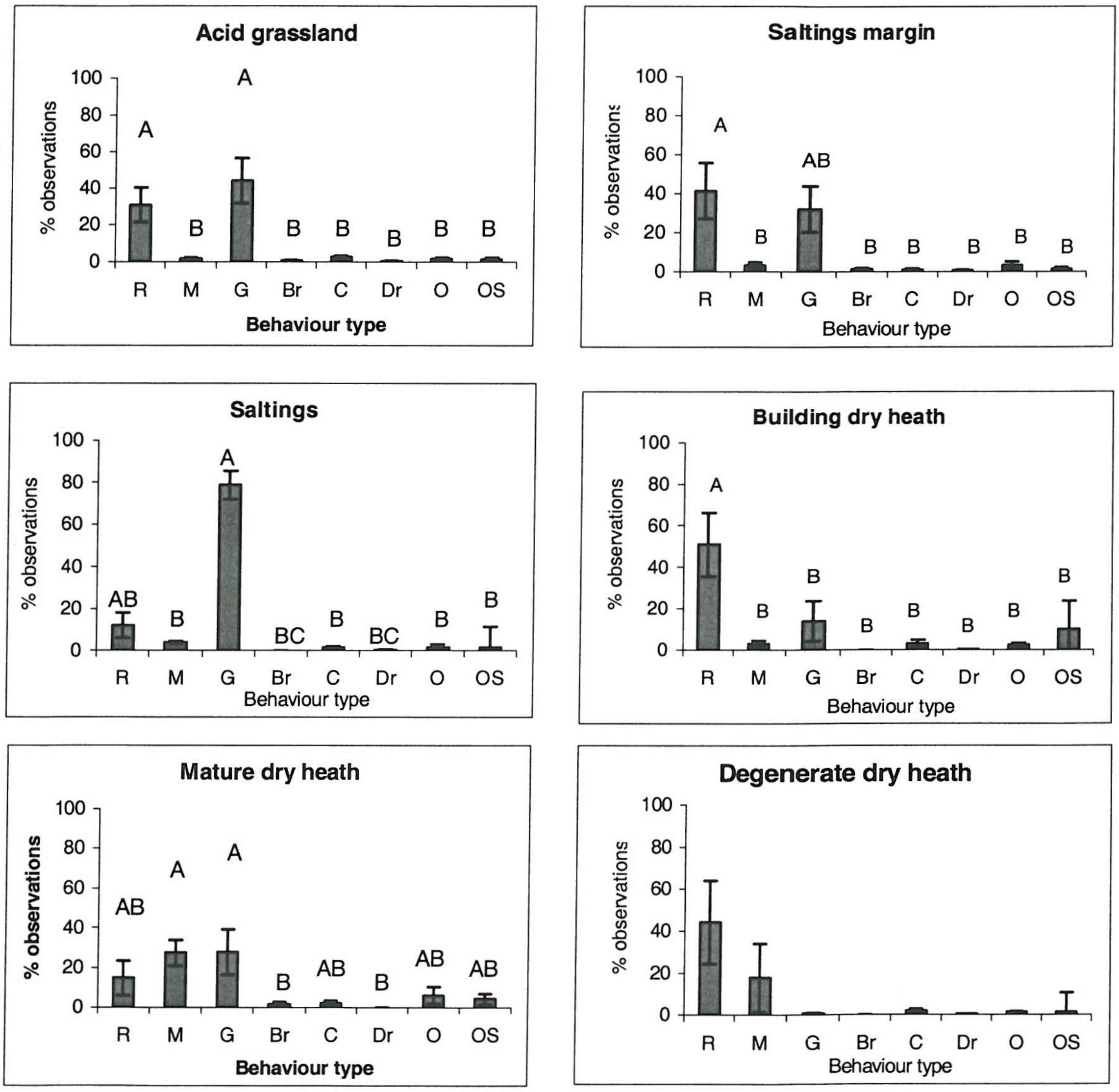


Figure 10. See legend overleaf.

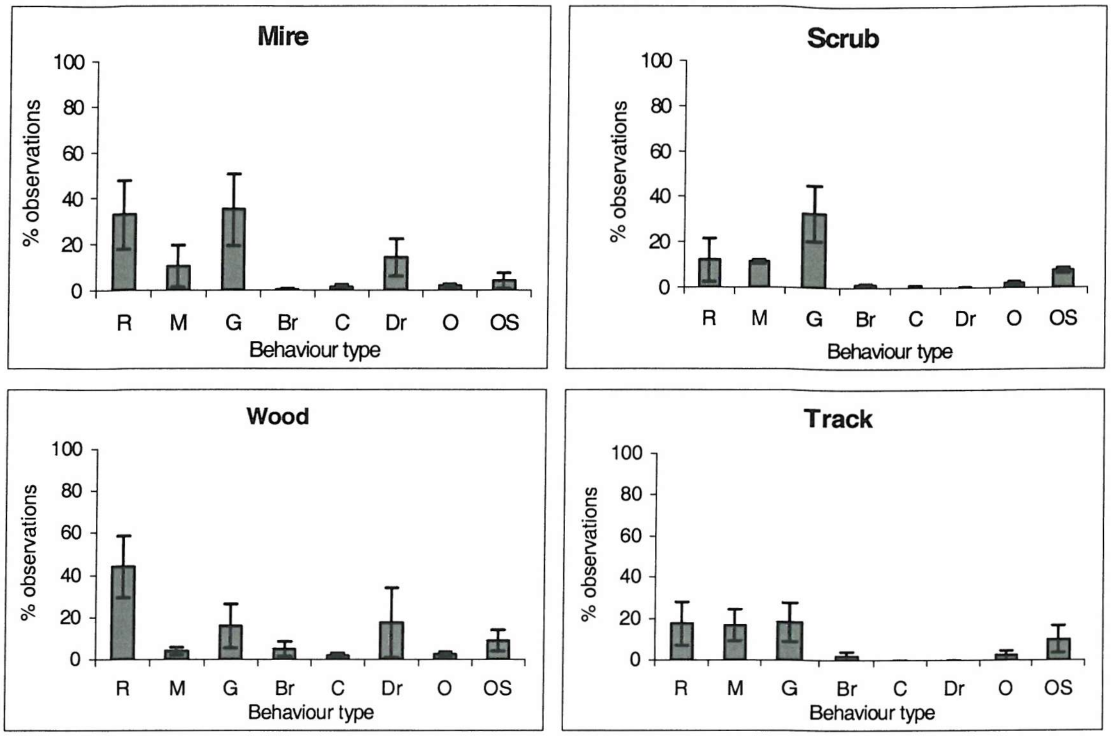


Figure 10. Mean time spent on each behaviour (% of observations) in different habitats. Significant differences are denoted by different letters, therefore behaviours with the same letter show no significant difference. Where letters are omitted no significant difference between behaviour times was found. AG: F = 14.17***; S: F = 18.04***; SM: F = 3.86***; bDH: F = 56.62***; mDH: F = 3.97***; dDH, W, M, T, SC – differences not significant at P = 0.05. *** denotes $P < 0.001$. For behaviour types see Fig.9.

Different habitats were used to a different extent for each behaviour (Figure 10). Behaviour in each habitat was similar to that in 1999, with the exception of SM, which was most used for resting, and mDH, of which greater relative use was made for resting and moving.

Selection of habitats for key behaviours

Acid grassland (AG) and saltings margin (SM) were highly selected for each behaviour. Building and degenerate dry heath (bDH, dDH) were positively selected for resting, whereas saltings (S) and and mire (M), were positively selected for foraging. Tracks (T) was positively selected for moving. (Figure 11).

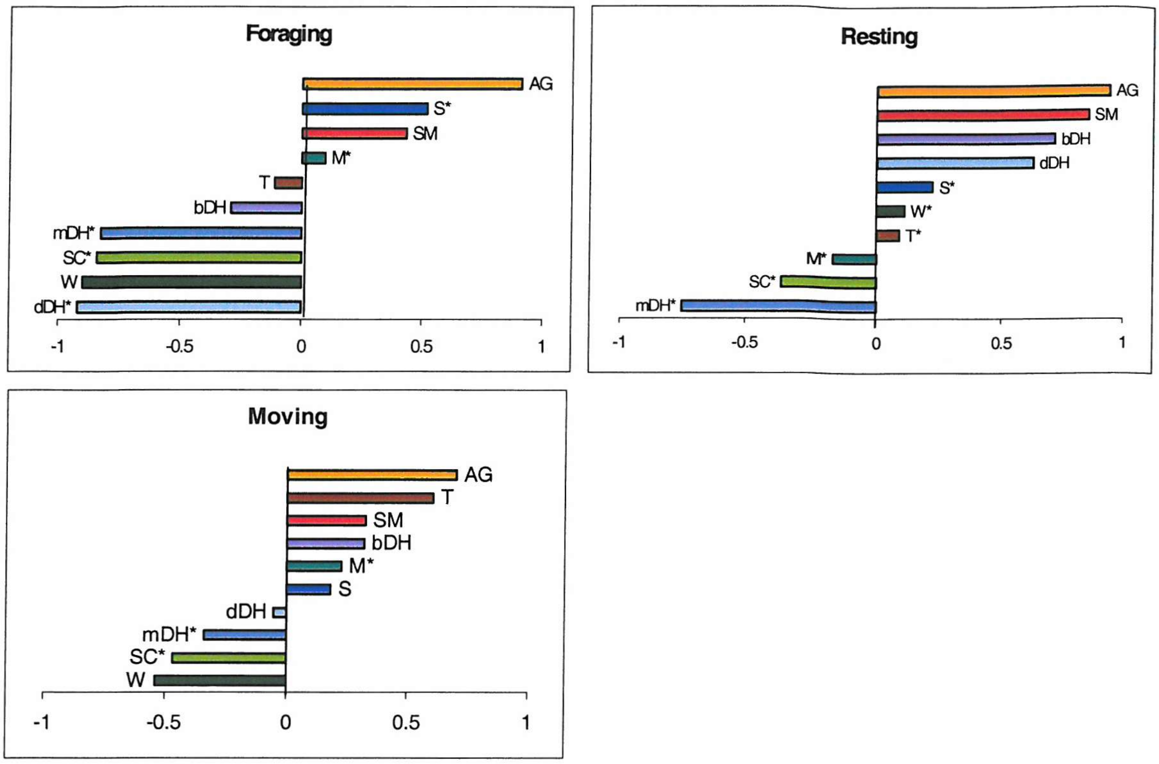


Figure 11. Selection of habitats for resting and foraging, expressed as Jacobs' indices. Habitats for which the Jacobs' index differs significantly from a median of xero (which indicates selection is in direct proportion to availiability) are denoted with * (Wilcoxon signed rank test, P<0.01). See Table 2.2 for habitat abbreviations.

Seasonal changes in total use of habitats for different behaviours

There were no significant changes between summer and autumn in how habitats were used. However, notable trends in habitat selection for key behaviours included increased selection for of degenerate dry heath (dDH) in autumn for resting and foraging, and a corresponding decrease in selection for building dry heath (bDH). Selection for saltings margins (SM) for foraging increased in autumn (Figure 12).

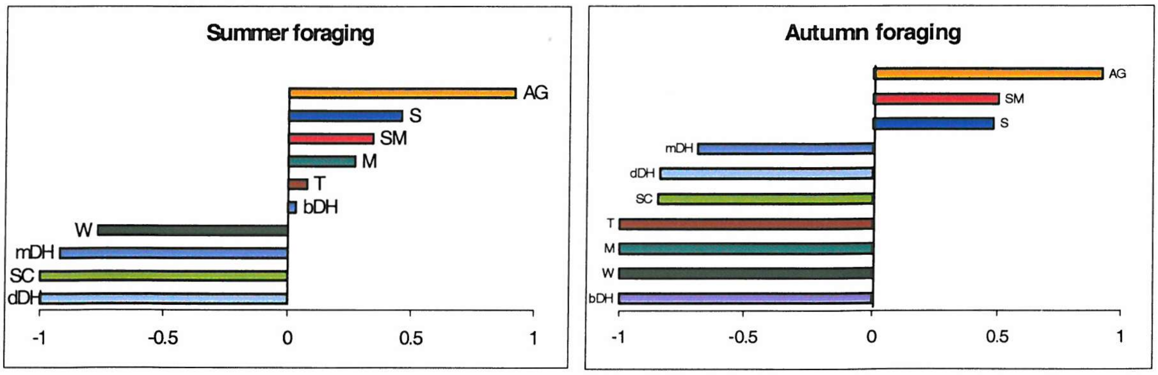


Figure 12. Seasonal selection of habitats for key behaviours by cattle at Arne 2000, expressed by Jacobs' indices. See Table 2.2 for habitat abbreviations. Cont'd overleaf.

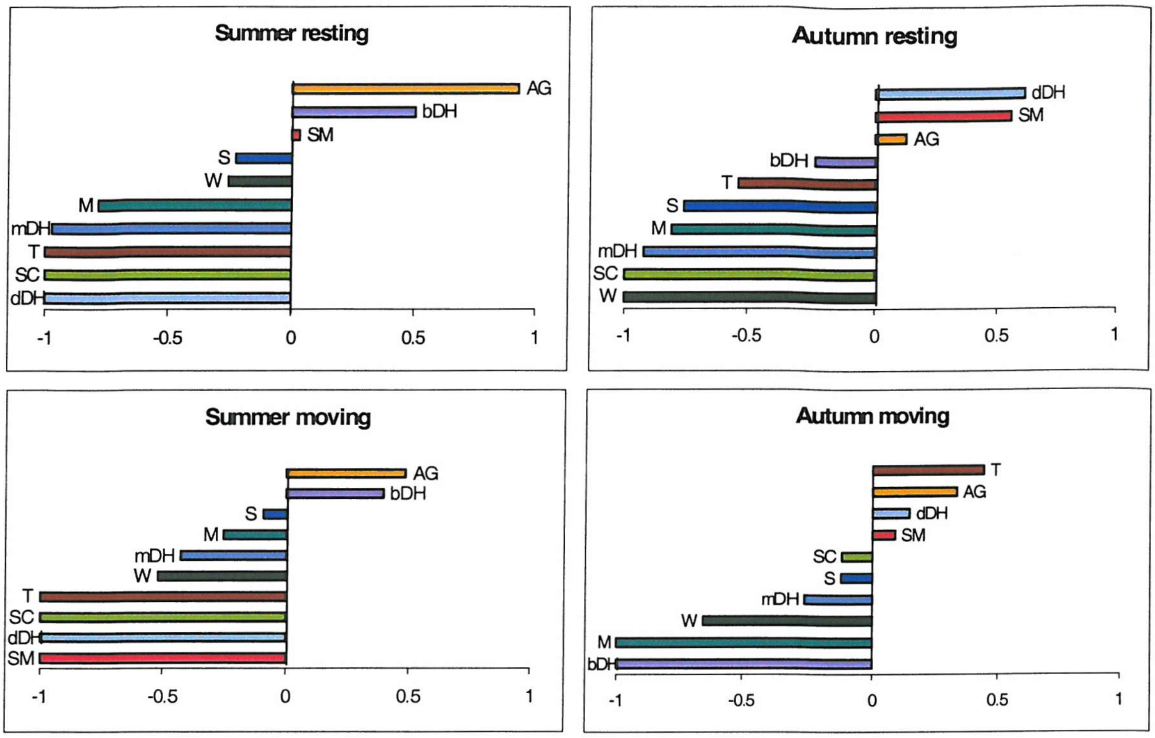


Figure 12 cont'd.

Foraging

Plant species availability and use at Arne in 2000 are shown in Table 2, and plant selection within key habitats is shown in Figure 13. Bite rates in each habitat are shown in Figure 14.

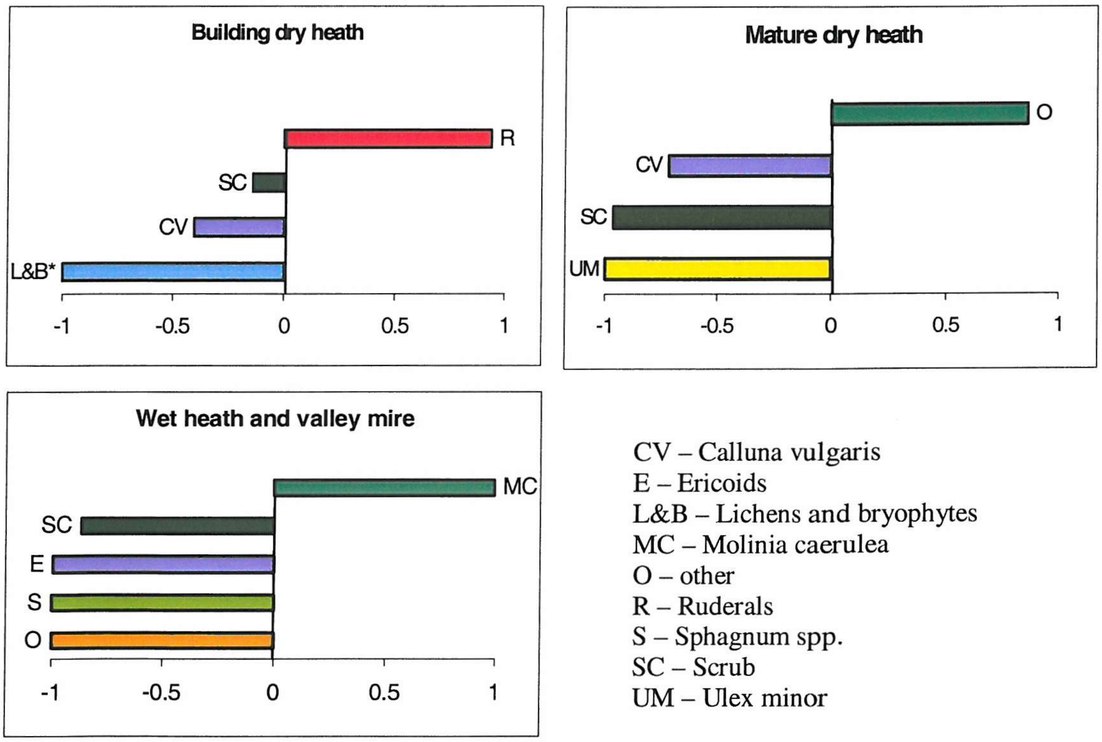


Figure 13 Selection of plant species/categories as quantified by Jacobs' indices. Species for which selection is significantly different from zero are denoted by * (Wilcoxon signed rank test, $P < 0.05$). Tests were not possible for wet heath&valley mire and mature dry heath for which there were only five samples (i.e. individuals). See Table 2 for full species categories BDH: R > all other categories (Mann-Whitney, $P < 0.001$), CV > L&B (Mann-Whitney, $P = 0.03$).

Table 2. Plant species availability and use (measured as the average number of bites of each plant being eaten) at Arne in 2000. dDH is not included due to insufficient data. M – wet heath and valley mire, BDH – building dry heath, MDH – mature dry heath, SC – scrub.

Habitat	**Species**	**% cover mean (SE)**	**% bites**	**Chi square**
M	*Molinia caerulea* (MC)	75.7 (5.1)	99.8	39.67***
	Ericoids (E)	8.0 (2.5)	0.1	
	Other graminoids (O)	9.1 (4.2)	0	
	Sphagnum spp. (S)	6.3 (2.8)	0	
	Pinus sp. & *Ulex europaeus* (SC)	0.9 (0.5)	0.1	
BDH	*Calluna vulgaris* (CV)	48.8 (6.2)	45.9	842.5***
	Lichens & bryophytes (L&B)	39.9 (5.9)	0	
	Colonisers of disturbed ground (e.g. *Rumex acetosella* *Senecio viscosus, Urtica dioica, Digitalis purpurea, Juncus effusus)* (R)	6.3 (4.5)	47.4	
	Other (e.g. *Ulex minor, Pinus* sp., *Erica cinierea, Rhododendron* sp., *Pteridium aquilinum) setacea)* (SC)	5.0 (1.4)	6.7	
MDH	Calluna vulgaris (CV)	92.3 (1.6)	66.3	62.19***
	Ulex minor (UM)	1.5 (0.5)	0	
	Pinus sp (SC)	2.7 (1.2)	0.1	
	Other *(Agrostis curtisii, bryophytes, lichens)* (O)	3.6 (0.8)	33.6	
SC	*Ulex europaeus*	81.2 (5.46)		In sufficient data
	Molinia caerulea	4.0 (2.6)		
	Acid grassland species	11.9 (4.4)		
	Other *(Pteridium aquilium, Rubus fruticosus, Calluna vulgaris)*	2.9 (1.4)		

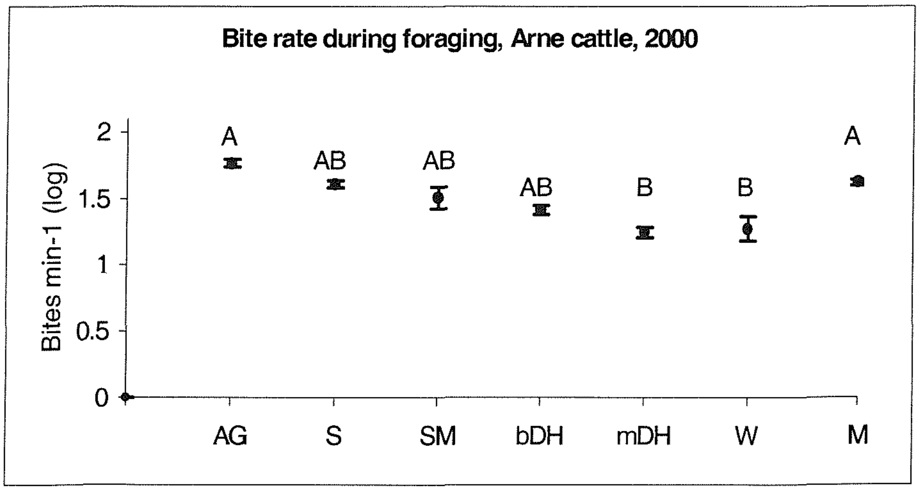


Figure 14. Foraging rates in different habitats. Different letters distinguish groups between which there were significant differences in foraging rate (GLM, F = 10.07, P < 0.001). There were no significant differences between individuals. See Table 2.2 for habitat abbreviations.

Appendix II Hartland Cattle

Data relating to the 24-hour Hartland cattle follows referred to in Chapter 2 are presented here.

Routine

Behaviour during each 24-hour observation period is shown in Figure 1, routes taken over each 24-hour observation period are shown in Figure 2 (see also Figure 2.2b for habitat types). Water was only available in wet heath (WH) and valley mire (VM), except for one trough in AG/DH in the far east of the site. Cattle visited these habitats to drink at least once a day. Although never observed in the central northern section of the site, they were seen in the western section on other several occasions.

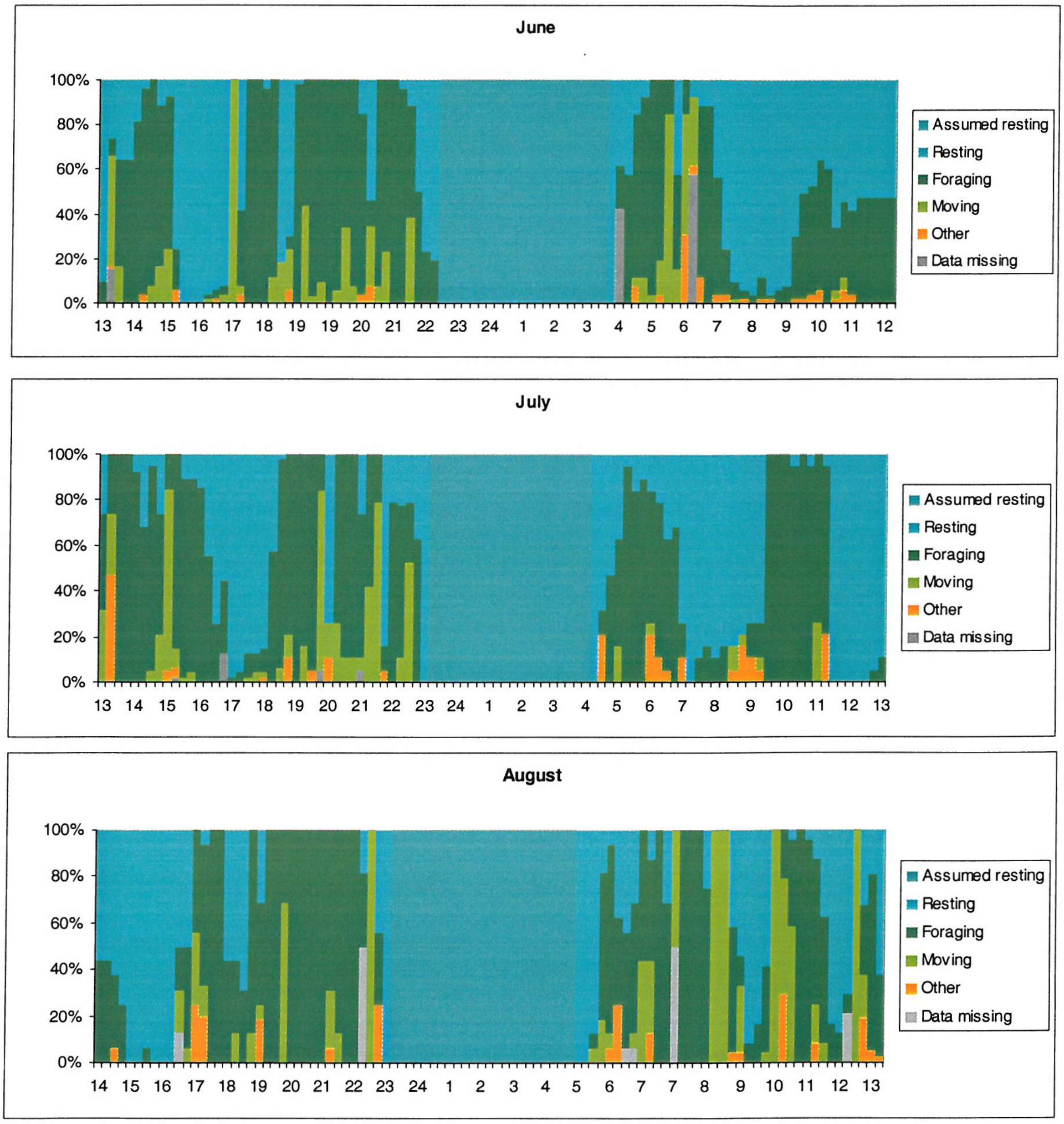


Figure 1. See legend overleaf.

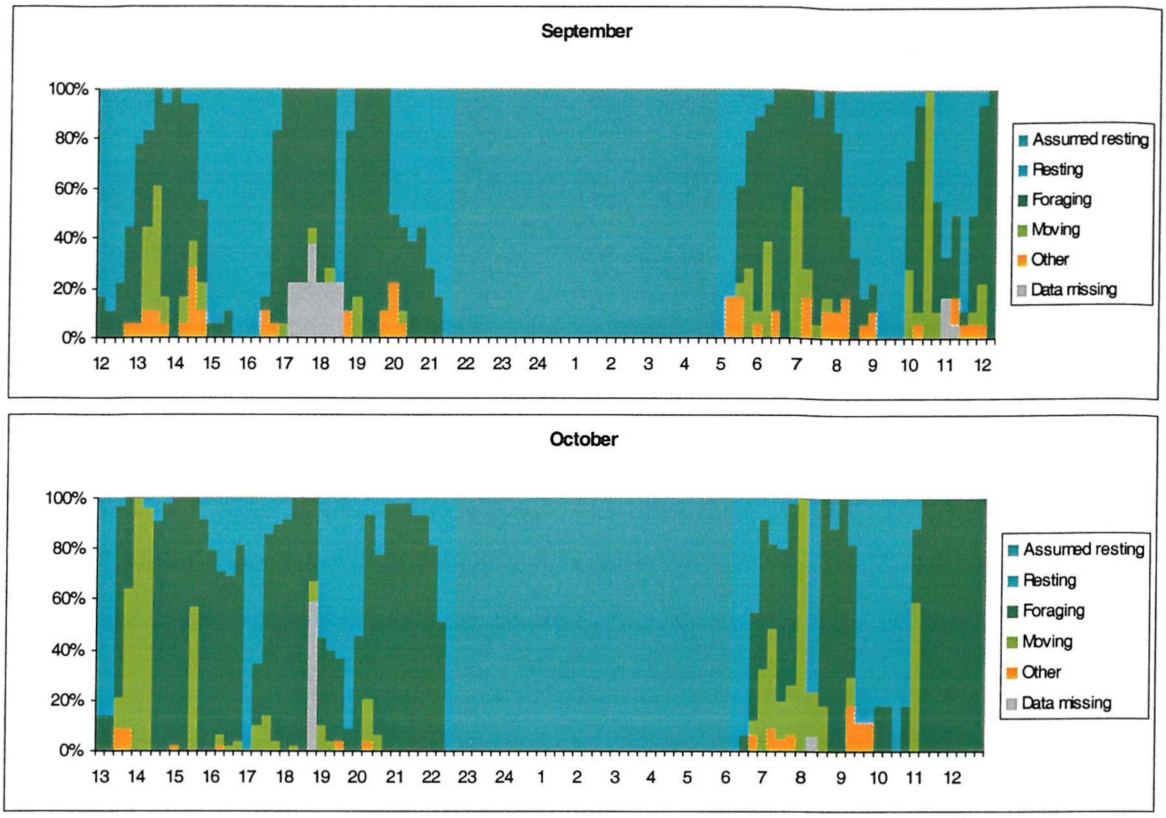


Figure 1 cont'd. Behaviour during each 24-hour observation periods by cattle at Hartland in 2000. The proportion of the herd carrying out each behaviour is shown on the vertical axis, and the time in hours along the horizontal axis. 'Assumed resting' represents night-time periods when the herd was not followed.

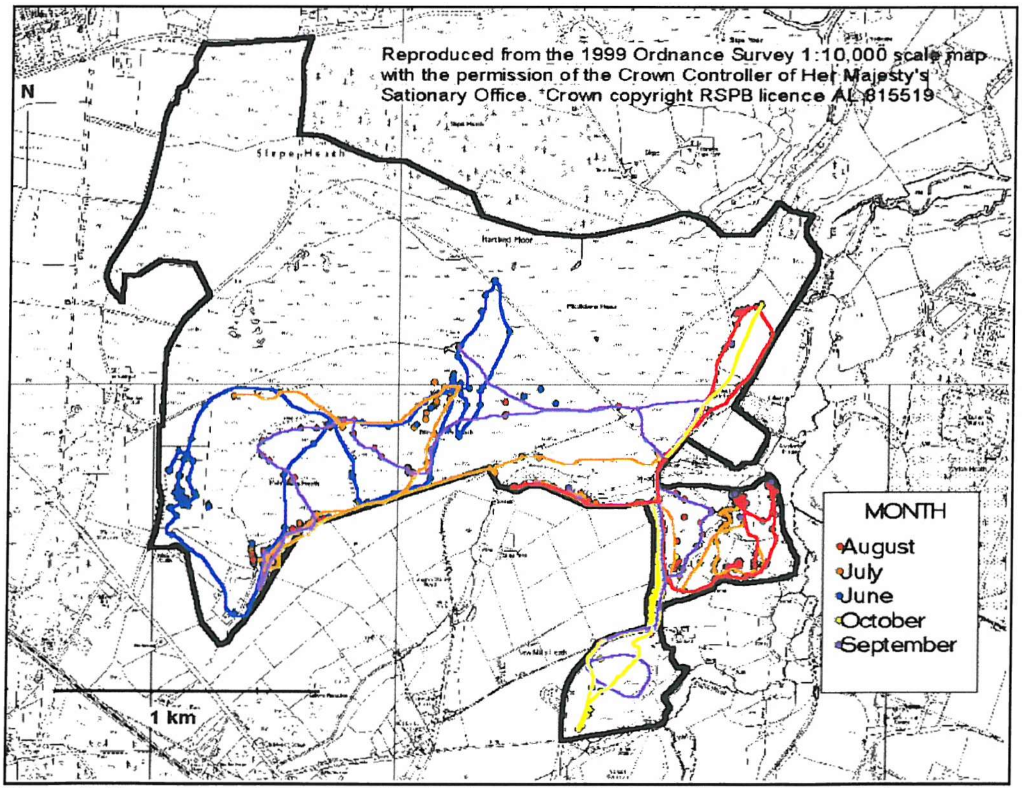


Figure 2 Routes taken by cattle at Hartland over each 24-hour observation periods.

Individual variation

Individual variation was not assessed. The Hartland cattle herd split into smaller groups, whose composition was not necessarily constant. The composition of the groups studied changed at least once during each observation, as individuals left or joined the group. Although a core remained, it was not possible to identify all individuals, except the focal animals. A secondary group was considered to have left the herd when it had been 'out of sight' for more than four observations (i.e. one hour), and the observations were amended accordingly. Changes in group composition usually occurred when the herd was moving from one habitat to another, therefore an estimate of the number of habitats occupied by the herd at any one time was complicated by the movement between groups. Since behaviour varies between habitats, the same problem applies to analysis of variation in behaviour. The difficulty in identifying individual animals within the Hartland herd meant that data from one rather than two focal animals was collected during each 24-hour observation period, and therefore no comparison between individuals was possible.

24-hour habitat selection

The variation in herd size throughout 24-hour observation periods (see above) influenced apparent habitat selection, as herd size tended to be largest on acid grassland (AG). Group size generally increased when the study herd arrived on AG, and decreased when the herd left as the group merged with and then separated from other groups already present on AG. Therefore, a single herd observation was used in analysing habitat selection – i.e. the habitat that the majority of the herd occupied at each observation time. Using individual data resulted in a bias away from the habitats where total group size tended to be largest (i.e. AG). In practice, the results from single

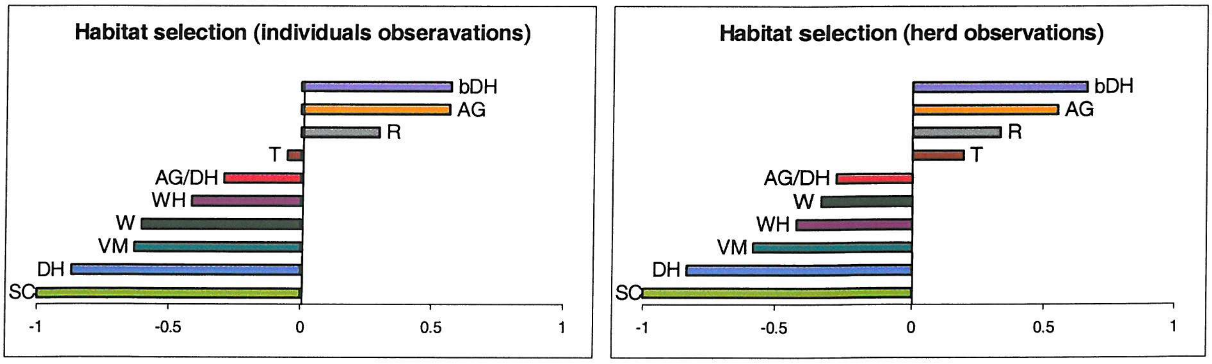


Figure 3. Habitat selection by cattle at Hartland using individuals and herd observations, expressed as Jacobs' indices. Habitat use was non-random (chi square = 51290.71, df = 9, p<0.001, chi square = 1617.59, df = 9, p<0.001).

herd observations were qualitatively very similar to the results from using individual observations to give the proportion of the herd occupying different habitats (Figure 3). The only differences between the results of the two data sets are the change from negative to positive selection for tracks (T), and the increase in the selection for wood (W). As only a proportion of the cattle tended to use these habitat types at any given time, they may be slightly over-represented using the herd observation data. The significance of overall selection was not tested because there were fewer than six samples.

Behaviour

Figure 4 shows the mean time spent on each behaviour by cattle at Hartland.

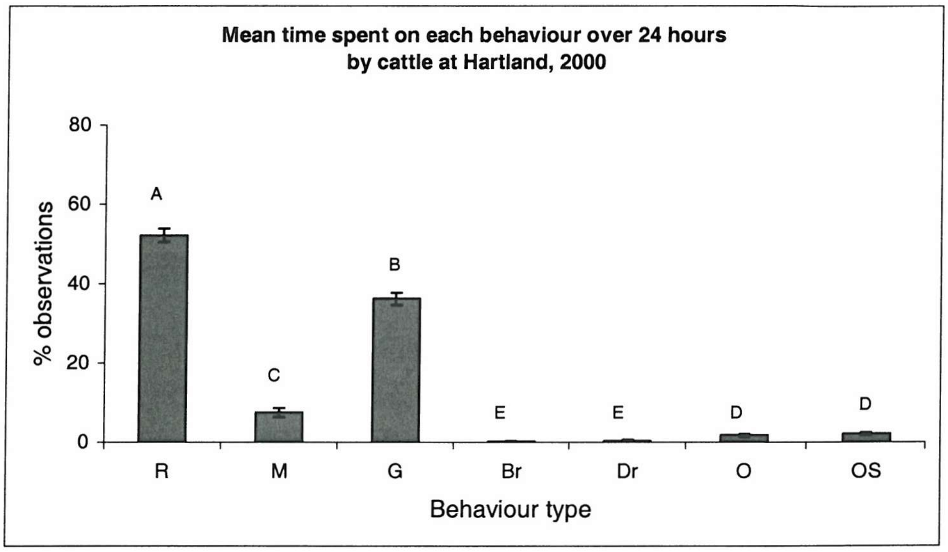


Figure 4. Mean time spent (i.e. percentage of observations) on each behaviour by cattle at Hartland. The difference in mean time spent on each behaviour was significant (GLM, $F = 373.25$, $P < 0.001$). Significant differences are denoted by different letters - behaviours with the same letter show no significant difference.

Behaviour in each habitat

Behaviour in most habitats differed from the behaviour of cattle at Arne in 1999 and is presented in Chapter 2. Behaviour in wood (W) is shown in Figure 5.

Selection of habitats for different behaviours

Building dry heath (BDH) and acid grassland (AG) were the most selected habitats for resting and foraging and were positively selected for moving. Roads (R) and tracks (T) were also positively selected for moving (Figure 6).

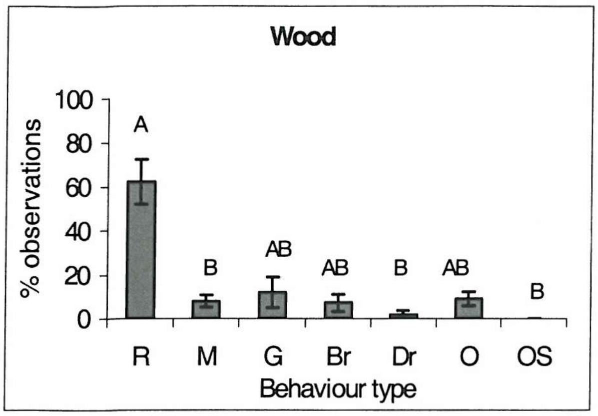


Figure 5. Mean time spent on each behaviour (% of observations) in wood. Behaviours with the same letter show no significant difference. GLM: F = 5.07. R – resting, M –moving, G – grazing, Br – browsing, Dr-drinking, O – other, OS – out of sight.

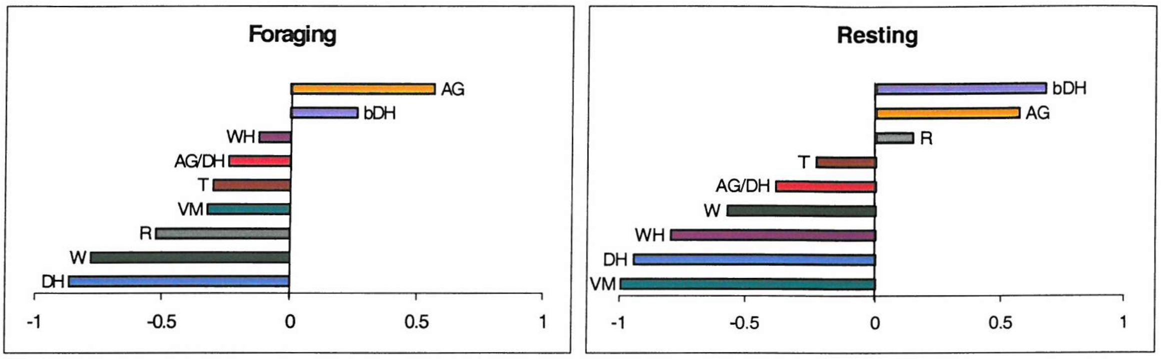


Figure 6. Selection of habitats for resting and foraging by cattle at Hartland, expressed as Jacobs' indices (using total data). See Table 2.2 for habitat abbreviations.

Foraging

Plant species availability and use are presented in Table 1. Plant species were only eaten in proportion to availability in restoration heath (AG/DH). Grasses (*Agrostis curtisii* in building dry heath bDH and dry heath DH, *Molinia caerulea* in bDH and valley mire VM, plus *Phragmites communis* and A. *stolonifera* in VM) was the only positively selected group for all habitats (Figure 7). The only ericaceous species eaten was *Calluna vulgaris* except in bDH, where *Erica tetralix* was very occasionally eaten. Other graminoids eaten in VM included *Trichophorum cespitosum*, *Juncus* spp. And *Carex* spp. Bite rates are shown in Figure 8.

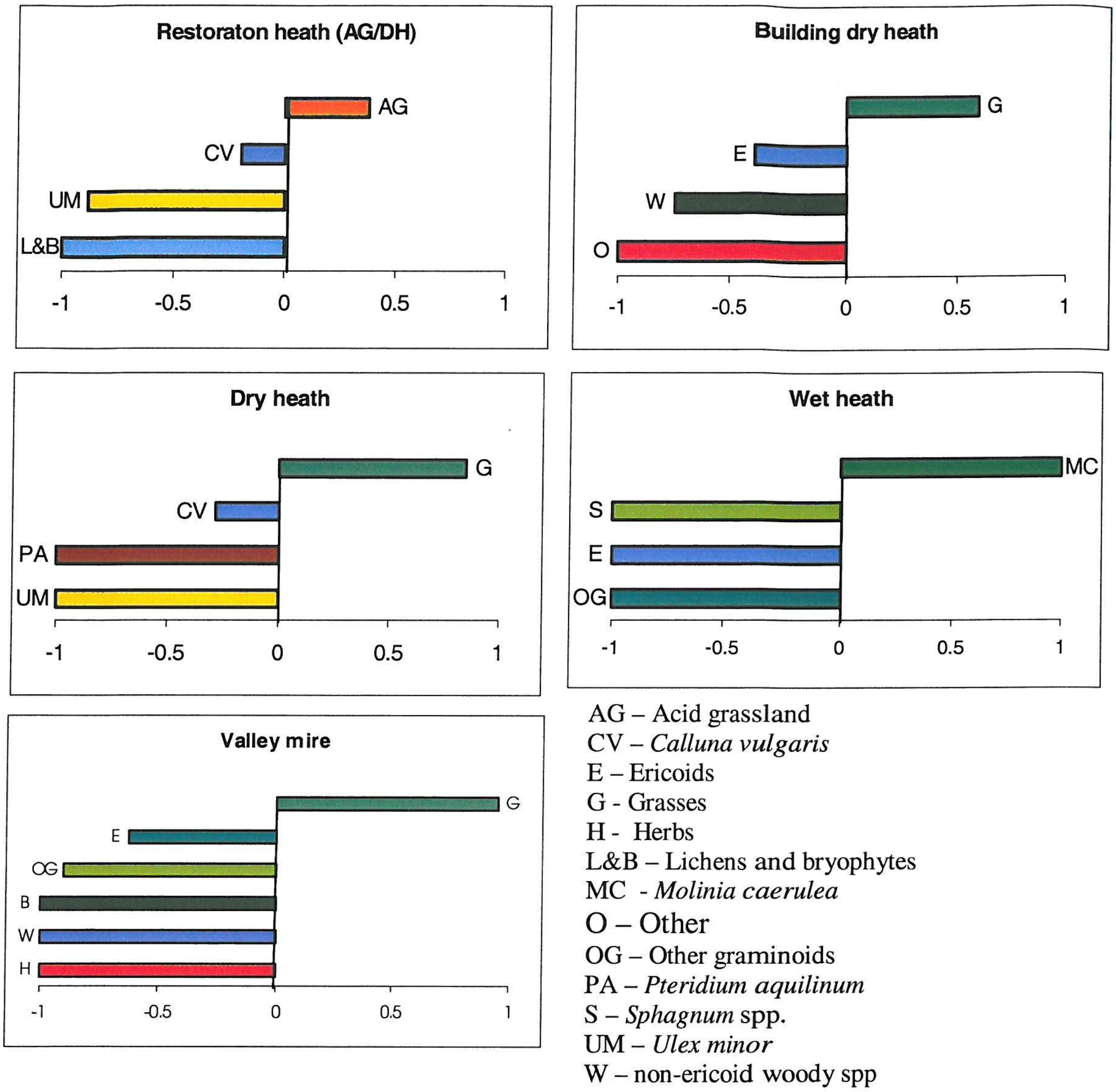


Figure 7. Selection of plant species/categories by cattle at Hartland, expressed as Jacobs' indices. Tests were not possible as there were fewer than six individuals. See Table 1 for full species categories.

Bite rates were highest in grass-rich habitats such as acid grassland (AG), wood (W), restoration heath (AG/DH) and lowest in dry heath (DH) (Figure 8). The foraging rate in W was higher than at Arne. The woods used by cattle on Hartland were wetter than at Arne, and the ground storey was dominated by *Molinia caerulea* rather than the dry heath vegetation found at Arne.

Table 1. Plant species availability and use (thee average number of bites of each plant being eaten) by cattle at Hartland. * = P <0.05, **= P < 0.01, *** -= P < 0.001, NS - P > 0.05

Habitat	Species	% cover mean (SE)	% bites	Chi square
AG/DH	Acid grassland	70.5 (5.3)	84.2	4.49^{NS}
	Ericoids	21.6 (2.2)	15.6	
	Ulex minor	4.1 (3.3)	0.2	
	Lichens & bryophytes	3.9 (2.7)	0	
BDH	Grasses (*Agrostis setacea, Molinia caerulea*) (G)	27.7 (5.3)	60.1	19.49***
	Ericoids (E)	59.2 (5.8)	38.9	
	Other woody spp (W)	6.2 (2.7)	1	
	Other (*Pteridium aquilinum,*	6.2 (2.1)		
	Rumex acetosella, bryophytes) (O)		0	
DH	*Calluna vulgaris* (CV)	92.1 (2.6)	13.1	38.75***
	Ulex minor (UM)	5.5 (2.1)	0	
	Grasses *(Agrostis setacea,*	1.2 (1.7)	86.9	
	Molinia caerulea) (G)		0	
		0.2 (0.7)		
WH	*Molinia caerulea* (MC)	45.9 (4.3)	100.0	42.3***
	Ericoids (E)	42.7 (3.1)	0	
	Other graminoids (O)	8.5 (2.6)	0	
	Sphagnum spp. (S)	2.9 (2.4)	0	
VM	Grasses (*Molinia caerulea, Phragmites communis, Agrostis stolonifera* (G)	50.0 (3.6)	97.5	47.21**
	Other graminoids (O)		2.0	
	Herbs (various) (H)	27.7 (3.5)	0	
	Ericoids (E)	8.4 (2.0)	0.5	
	Other woody spp. (*Ulex* spp.,	1.9 (0.8)		
	Betula sp, *Salix* sp.)	3.2 (0.4)	0	
	Bryophytes (B)			
		9.6 (3.5)		

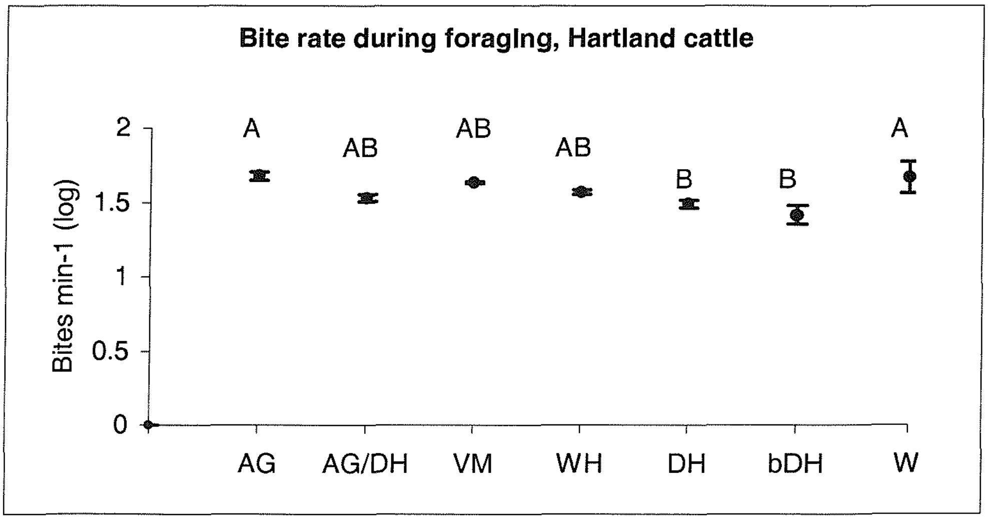


Figure 8. Foraging rates of cattle at Hartland in each habitat. Habitats with significantly different rates are labelled with different letters (GLM, F = 4.76, P = 0.015).

Appendix III Exmoor ponies

Data relating to the 24-hour Hartland Exmoor pony follows referred to in Chapter 2 are presented here.

Routine

Exmoor ponies showed a similar behavioural routine to New Forest ponies (Figure 1). Routes taken during 24-hour observation periods are shown in Figure 2.

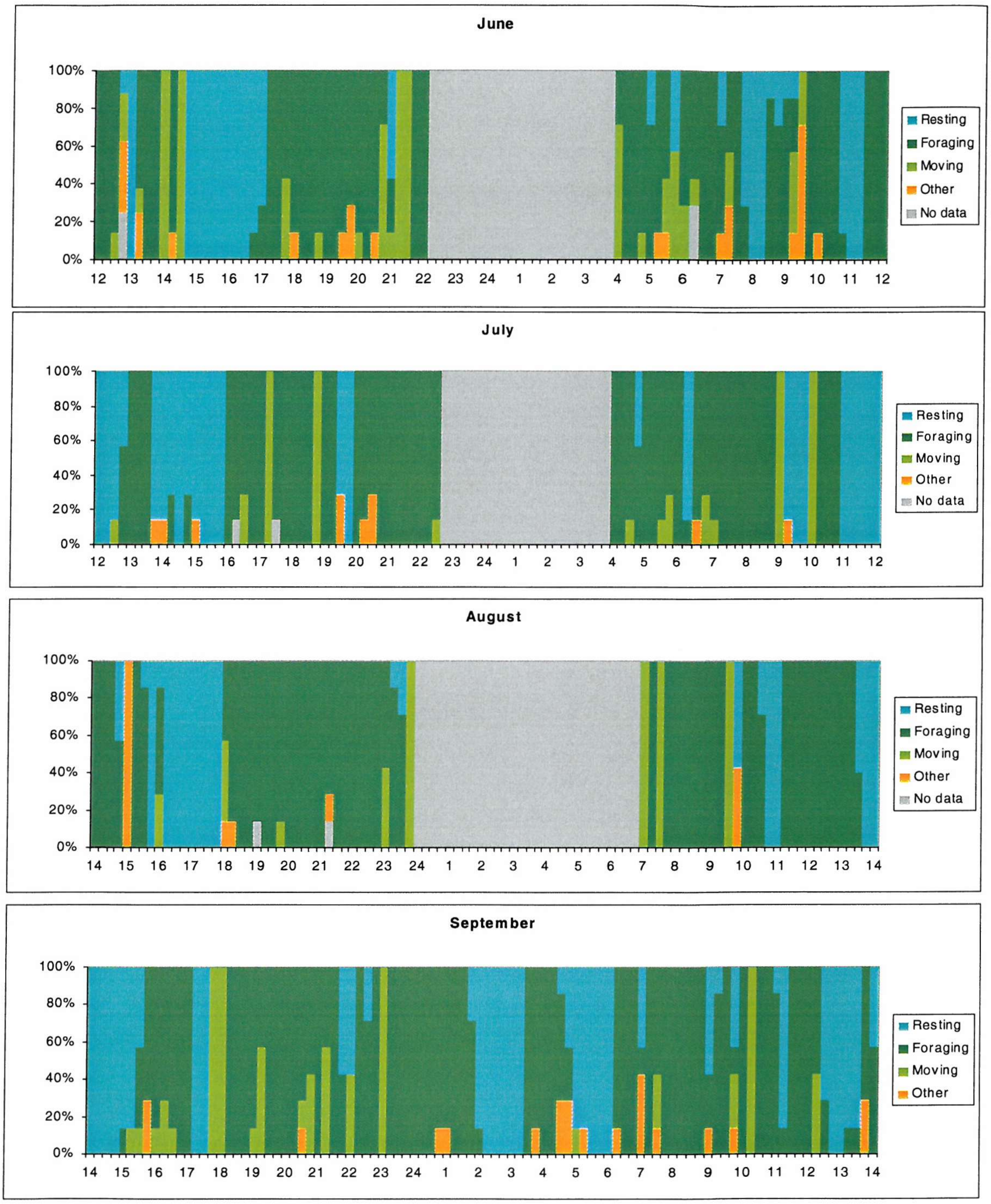


Figure 1. See legend overleaf.

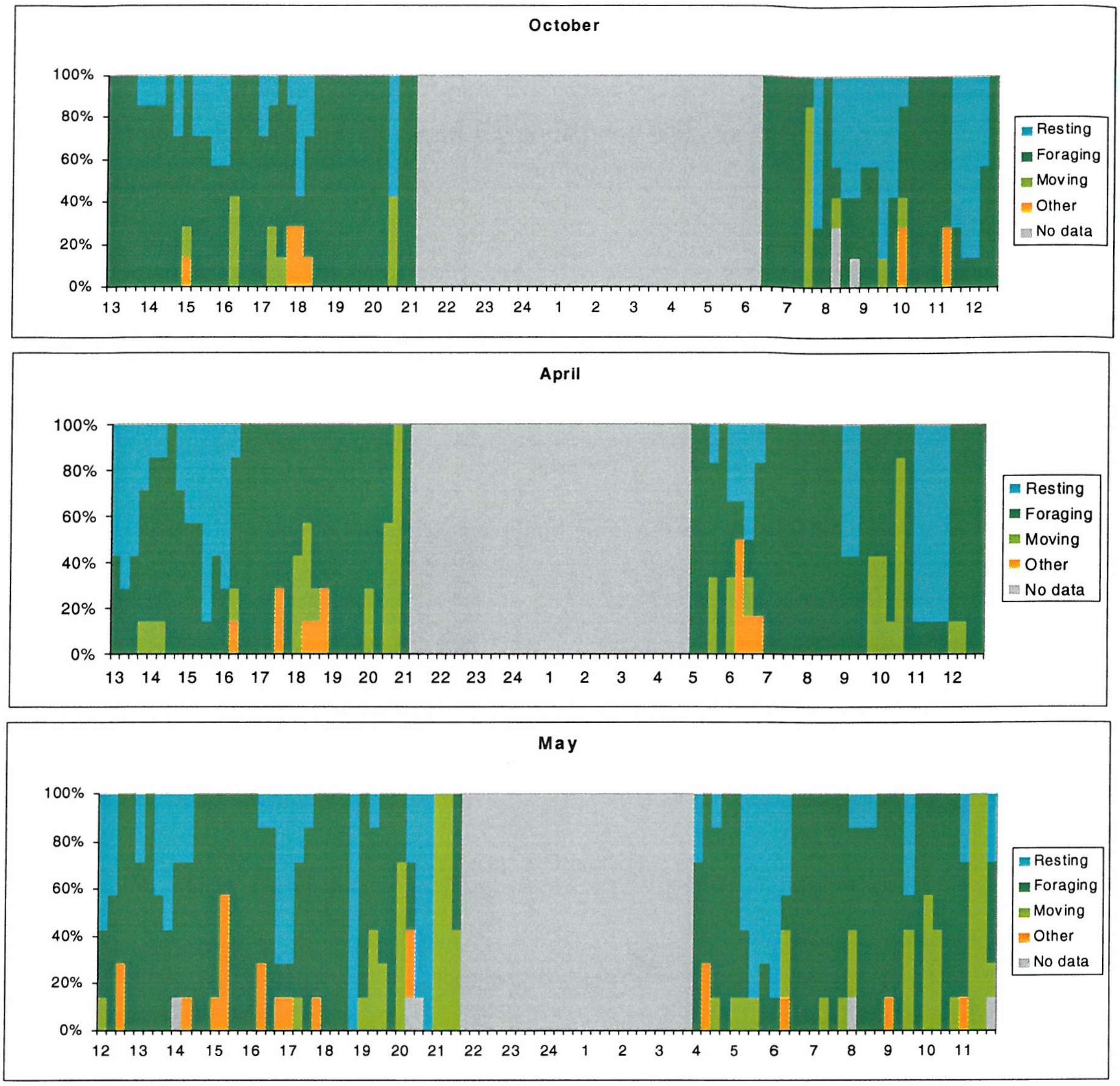


Figure 1 Behaviour during each 24-hour observation periods by Exmoor ponies. The proportion of the herd carrying out each behaviour is shown on the vertical axis, and the time in hours along the horizontal axis.

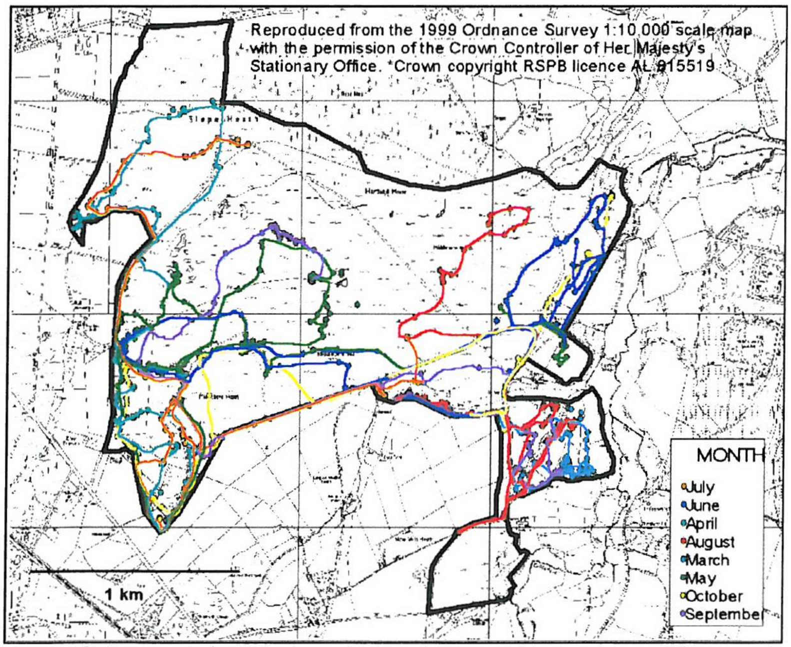


Figure 2. Routes taken during observations. Habitat types can be seen in Figure 2.2b

The Exmoor ponies generally remained on acid grassland, moving over dry heath to reach water sources and, in summer, to forage on wet heath and valley mire.

Individual variation

There was a greater cohesion within the Exmoor pony herd than the New Forest pony herd. It never split into smaller groups, and there was less variation between individual behaviour and habitat choice

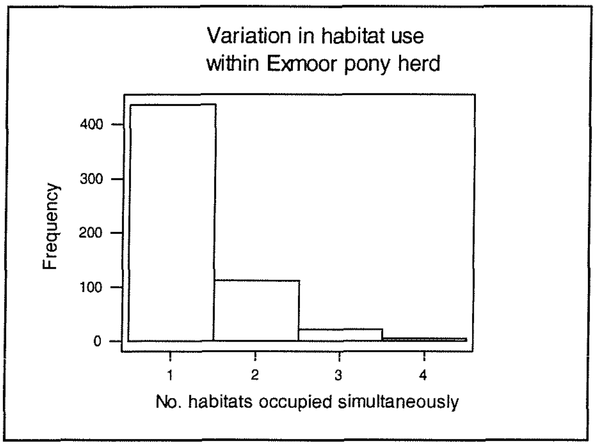


Figure 3. Variation in habitat selection measured as the number of habitats that individuals within the herd occupied simultaneously.

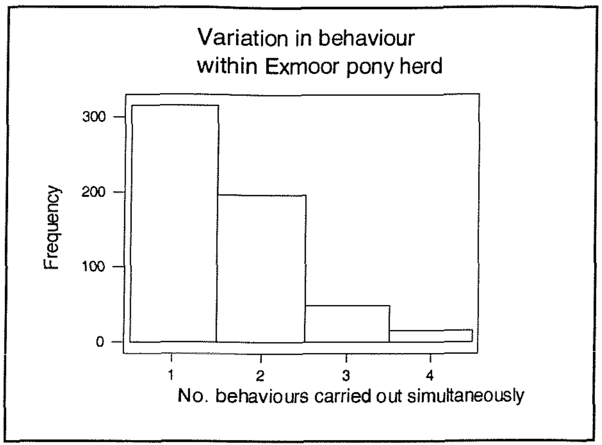


Figure 4. Variation in behaviour measured as the number of habitats that individuals within the herd occupied simultaneously.

Individual habitat use and behaviour

There were no significant differences in habitat selection between individuals. One significant difference was found in behaviour between individuals (see Chapter 2).

24 hour habitat selection

Habitat selection is shown in Chapter 2.

Behaviour

The time spent on each behaviour by Exmoor ponies is shown in Figure 5. Behaviour in habitats in which it differed from the New Forest ponies is shown in Chapter 2. Figure 6 shows the behaviour in the remaining habitats. In each habitat, most time was spent foraging. More time in building dry heath was spent resting than in the remaining habitats. Selection of habitat types for key behaviours is shown Chapter 2, together with seasonal changes in habitat use and selection.

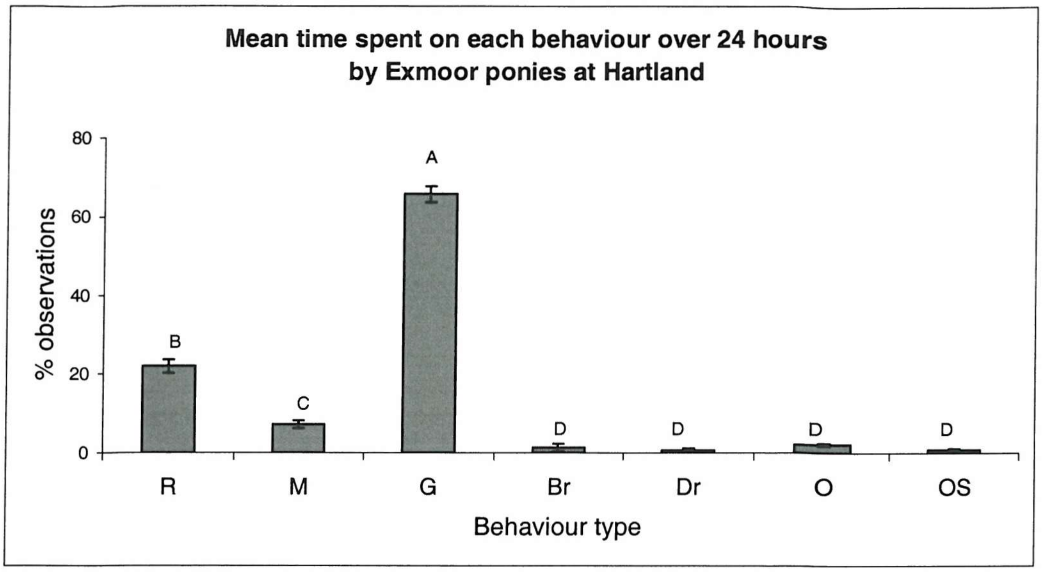


Figure 5 Mean time spent (i.e. percentage of observations) on each behaviour by Exmoor ponies at Hartland. The difference in mean time spent on each behaviour was significant (GLM, F = 361.64, P < 0.001). Significant differences are denoted by different letters, therefore behaviours with the same letter show no significant difference. R – resting, M – moving, G – grazing, Br – browsing, C – calves, Dr – drinking, O – other, OS, out of sight.

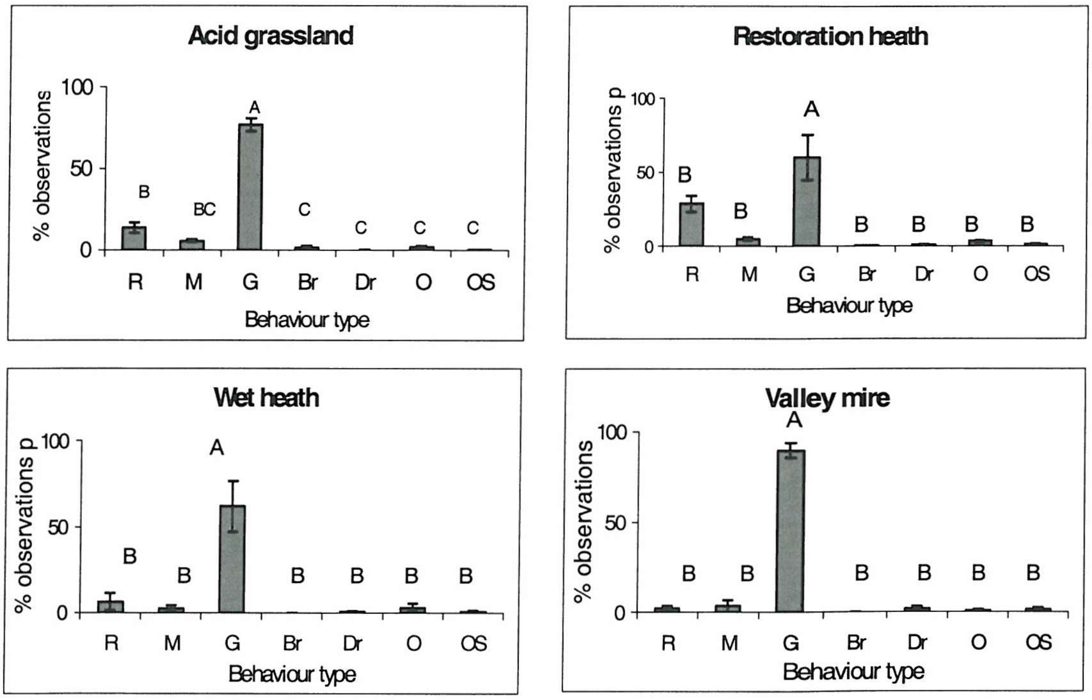


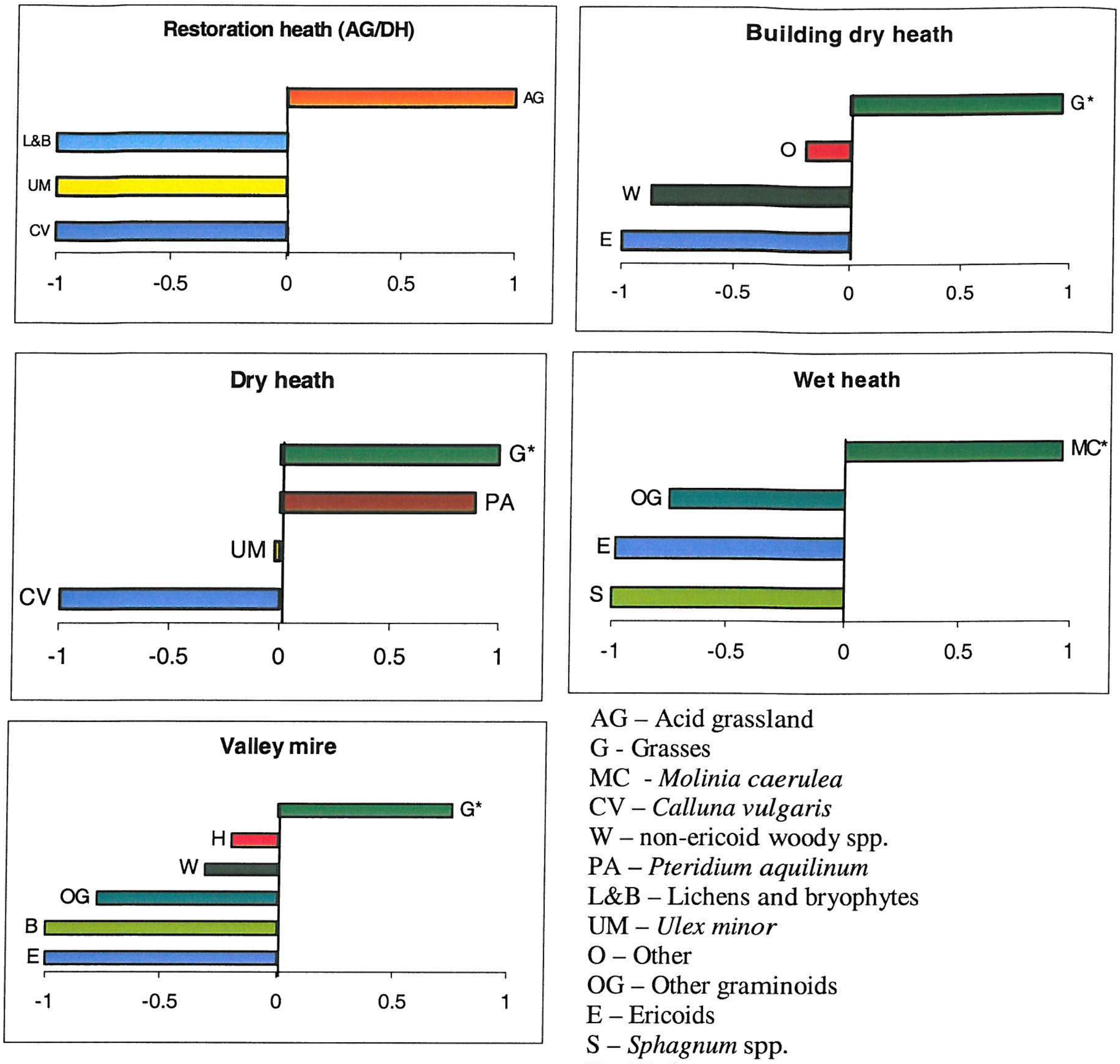
Figure 6. Mean time spent on each behaviour (% of observations) in different habitats. Significant differences are denoted by different letters, therefore behaviours with the same letter show no significant difference. Where letters are omitted no significant difference between behaviour times was found. AG: F = 100.05***; AG/DH: F = 5.03; WH: F = 12.14***, VM: F = 74.7. *** denotes $P < 0.001$. R – resting, M – moving, G – grazing, Br – browsing, C – calves, Dr – drinking, O – other, OS, out of sight.

Foraging

Selection for plant species was not in proportion to availability in any of the habitats tested (Table 1). On building dry heath (bDH) the grasses selected included *Agrostis curtisii, Molinia caerulea,* and *Aira praecox,* 'other' species eaten included *Rumex acetosella.* On dry heath (DH) grasses that were eaten included *Agrostis curtisii* and *Molinia caerulea.* On wet heath (WH), other graminoids species eaten included *Rhynchospora alba* and *Juncus effusus.* On valley mire, in addition to *M. caerulea, Phragmites communis, R. alba, J. effusus, Narthecium ossifragum, Betula pubescens and Myrica gale* were eaten (Figure7). In the few observations of foraging on tracks (T) and roads (R) grasses were always selected.

Table 1 Plant species availability and use (the average number of bites of each plant being eaten). * = P <0.05, **= P < 0.01, *** = P < 0.001. AG/DH – restoration heath, BDH – building dry heath, DH – dry heath, WH – wet heath, VM – valley mire.

Habitat	**Species**	**% cover mean (SE)**	**% bites**	**Chi square**
AG/DH	Acid grassland	70.5 (5.3)	100	9.1*
	Ericoids	21.6 (2.2)	0	
	Ulex minor	4.1 (3.3)	0	
	Lichens & bryophytes	3.9 (2.7)	0	
BDH	Grasses (*Agrostis setacea, Molinia caerulea*) (G)	27.7 (5.3)	95.0	64.7***
	Ericoids (E)	59.2 (5.8)	0.5	
	Other woody spp (W)	6.2 (2.7)	0.5	
	Other (*Pteridium aquilinum, Rumex acetosella, bryophytes*) (O)	6.2 (2.1)	4.0	
DH	Calluna vulgaris (CV)	92.1 (2.6)	3.2	1112.5***
	Ulex minor (UM)	5.5 (2.1)	5.3	
	Grasses *(Agrostis setacea, Molinia caerulea)* (G)	1.2 (1.7)	88.0	
	Pteridium aquilinum (PA)	0.2 (0.7)	3.5	
WH	*Molinia caerulea* (MC)	45.9 (4.3)	98.1	37.5***
	Ericoids (E)	42.7 (3.1)	0.6	
	Other graminoids (O)	8.5 (2.6)	1.3	
	Sphagnum spp. (S)	2.9 (2.4)	0	
VM	Grasses (*Molinia caerulea, Phragmites communis, Agrostis stolonifera* (G)	50.0 (3.6)	86.0	16.5**
	Other graminoids (O)	27.7 (3.5)	4.7	
	Herbs (various) (H)	8.4 (2.0)	5.9	
	Ericoids (E)	1.9 (0.8)	0	
	Other woody spp. (*Ulex* spp., *Betula* sp, *Salix* sp.)	3.2 (0.4)	3.5	
	Bryophytes (B)	9.6 (3.5)	0	



AG – Acid grassland
G - Grasses
MC - *Molinia caerulea*
CV – *Calluna vulgaris*
W – non-ericoid woody spp.
PA – *Pteridium aquilinum*
L&B – Lichens and bryophytes
UM – *Ulex minor*
O – Other
OG – Other graminoids
E – Ericoids
S – *Sphagnum* spp.
H - Herbs

Figure 7 Selection of plant species/categories as quantified by Jacobs' indices. Species for which selection is significantly different from zero are denoted by * (Wilcoxon signed rank test, $P < 0.05$). Tests were not possible for AG/DH, for which there were fewer than six individuals. See Table 1 for full species categories.

<u>Ranking</u>:

BDH: G > all other categories (Mann-Whitney, $P < 0.005$)
DH: G > all other categories (Mann-Whitney, $P < 0.001$)
WH: MC > all other categories (Mann-Whitney, $P < 0.001$)
VM: G > all other categories (Mann-Whitney, $P < 0.001$)

Bite rates were highest in grassy habitats (acid grassland AG, restoration heath AG/DH, tracks T), and particularly low in dry heath (DH) (Figure 8).

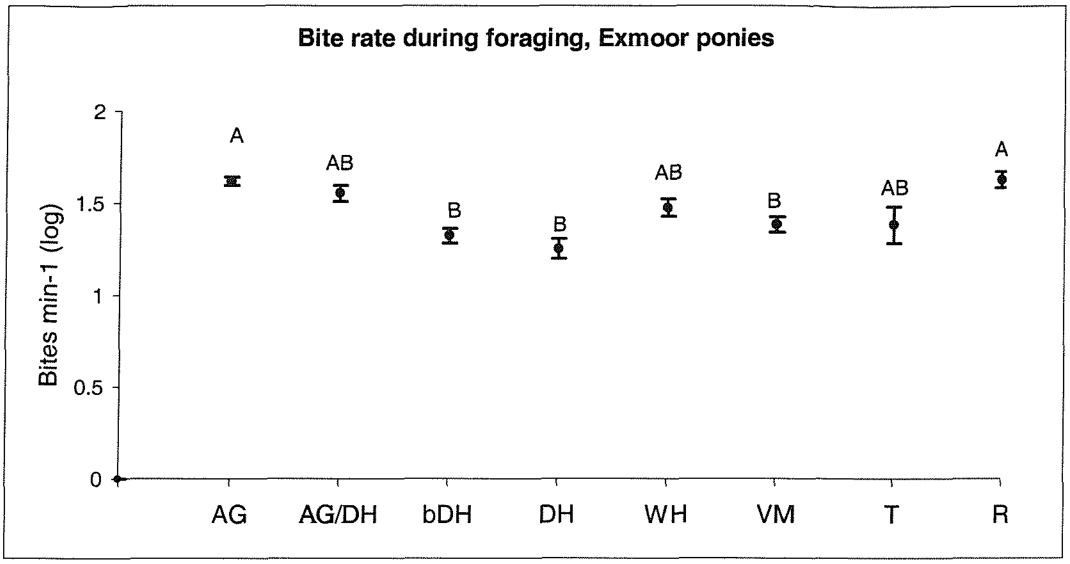


Figure 8 Bite rates of Exmoor ponies in different habitats. Different letters distinguish groups between which there were significant differences in foraging rate (GLM, F = 552, P < 0.001). There were no significant differences between individuals. See Table 2.2 for habitat abbreviations.

Appendix IV Stoborough New Forest ponies

Data relating to the 24-hour Stoborough New Forest pony follows referred to in Chapter 2 are presented here.

Routine

Behaviour during 24-hour observation periods is presented in Figure 1. Herd size varied widely and is therefore given for each month.

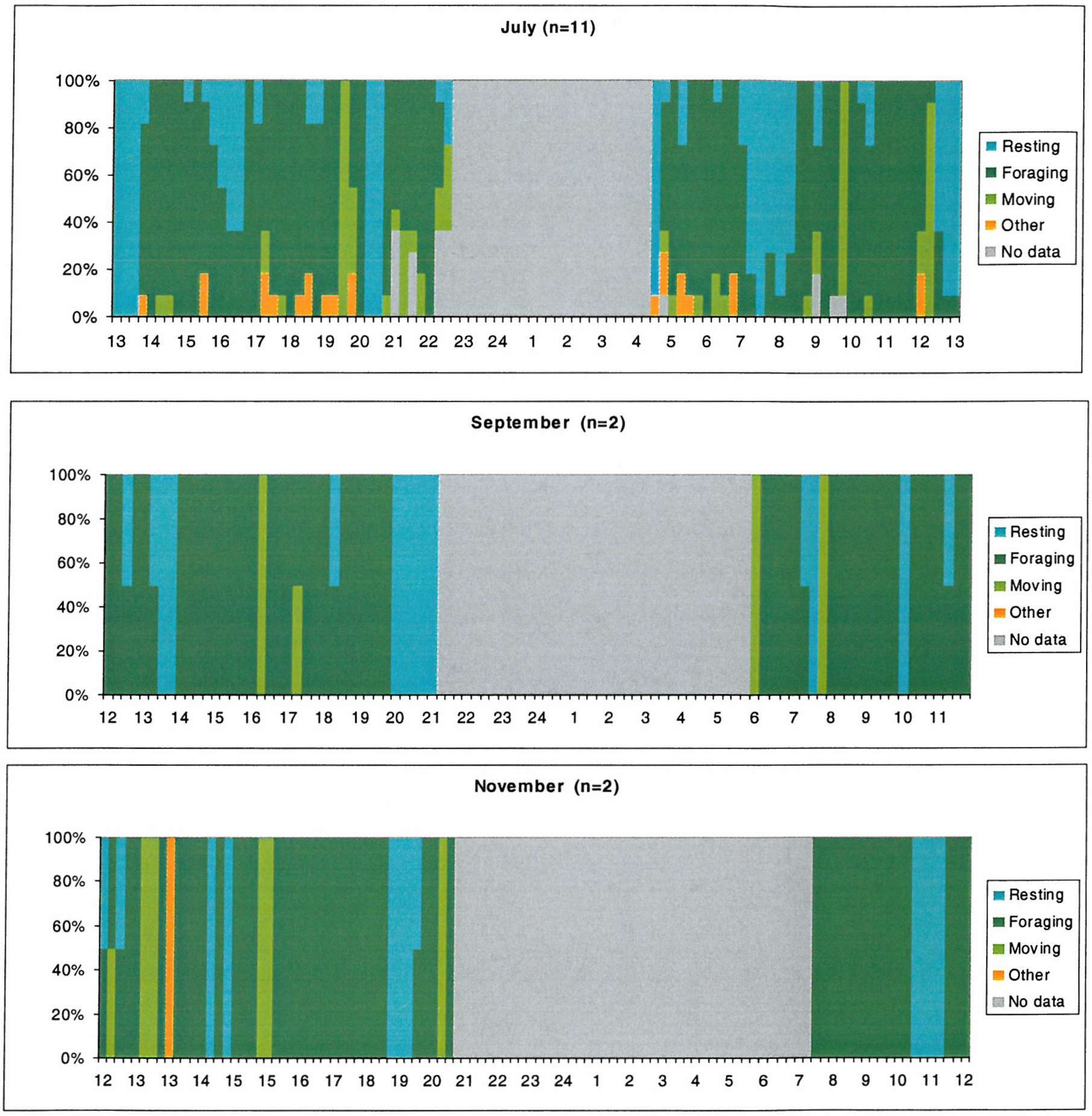
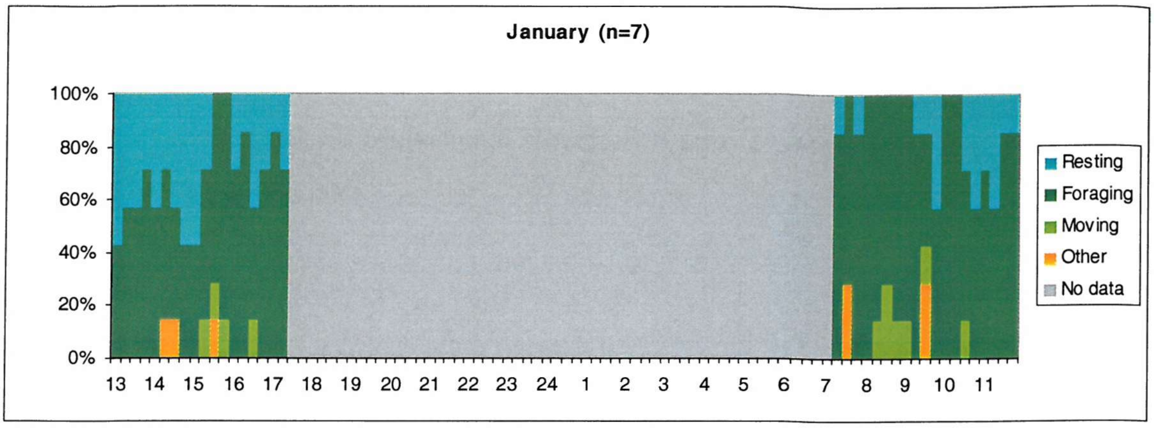
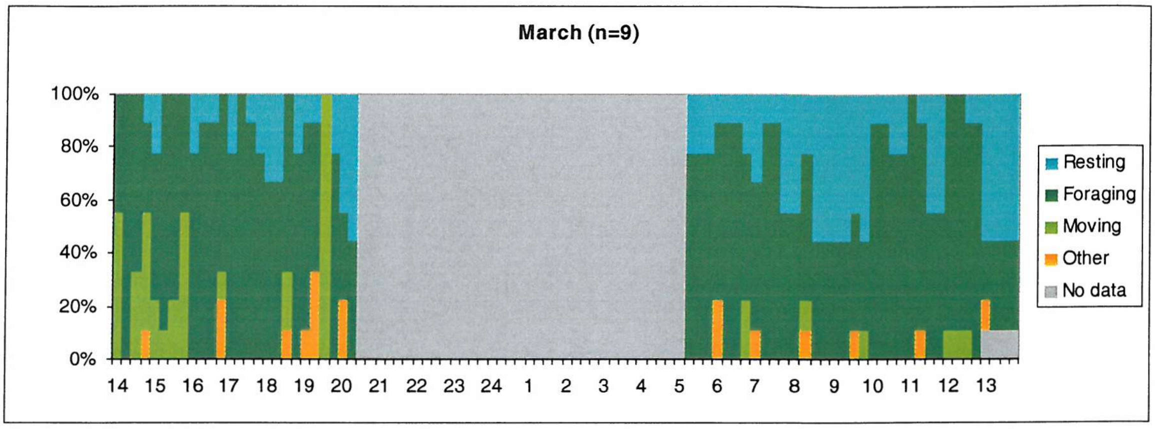
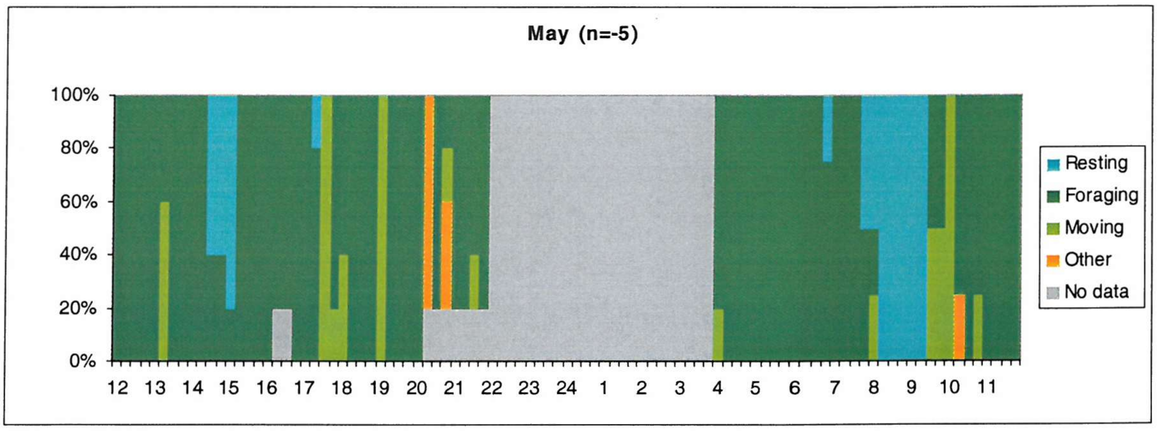


Figure 1. Behaviour during each 24-hour observation periods by New Forest ponies at Stoborough. The proportion of the herd carrying out each behaviour is shown on the vertical axis, and the time in hours along the horizontal axis.


January (n=7)


March (n=9)


May (n=-5)

The ponies were never observed in the northern section of the site, either during the 24-hour observation periods, or on about 20 other casual observations throughout the year. However, three young geldings introduced to the site after May 2001 were repeatedly observed in the northern and central areas. No individuals from the July herd, which used mainly the southern section of the site, remained on site by August. Composition of the subsequent herd changed, but some members remained throughout the rest of the study period. Site use during 24-hour observations can be seen in Figure 2, and compared with habitats in Figure 3.

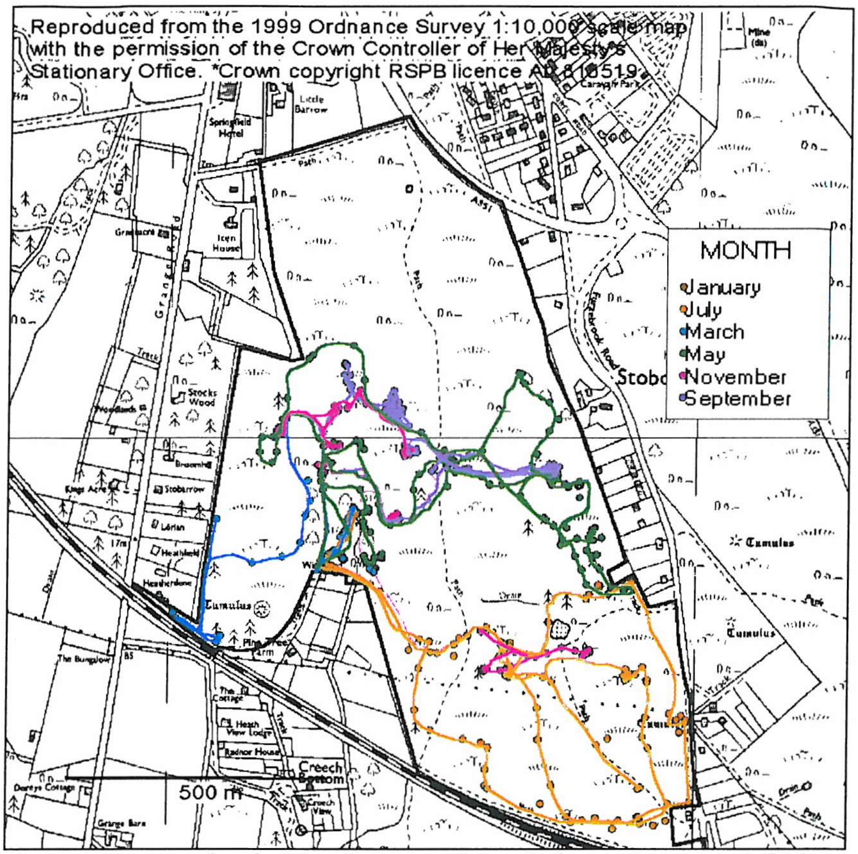


Figure 2. Routes taken by New Forest ponies at Stoborough over each 24-hour period.

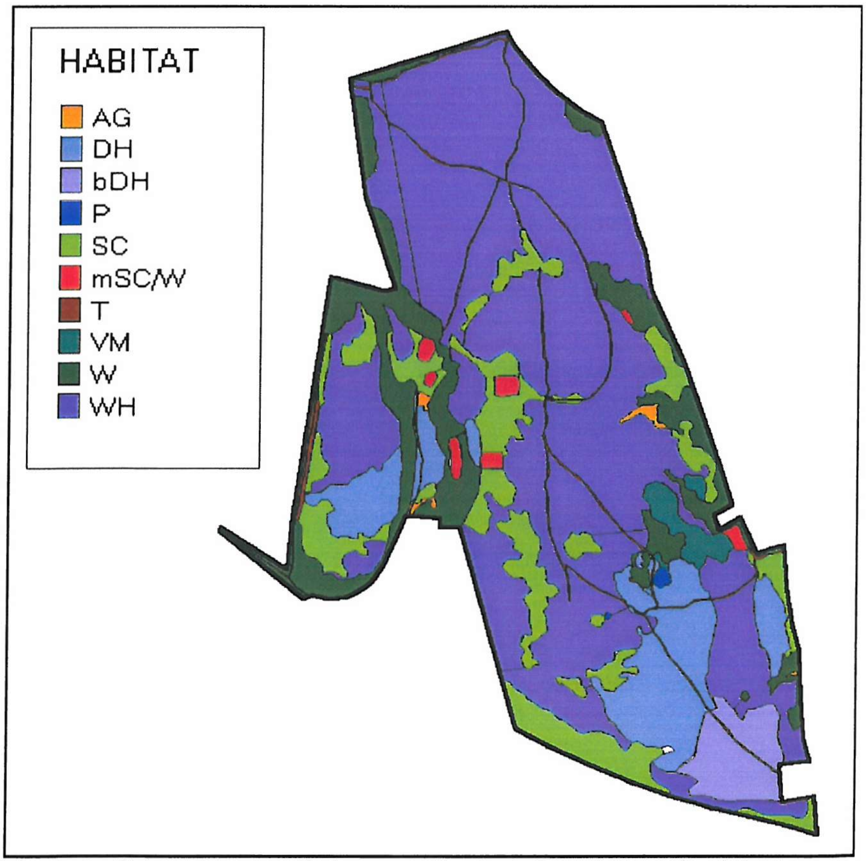


Figure 3. Habitat types on Stoborough Heath. AG – acid grassland, DH – mature and degenerate dry heath, bDH – building dry heath, P = pond, SC – scrub, mSC/W – managed scrub of woodland, T – track, VM – valley mire, W – wood, WH – wet heath.

Individual variation

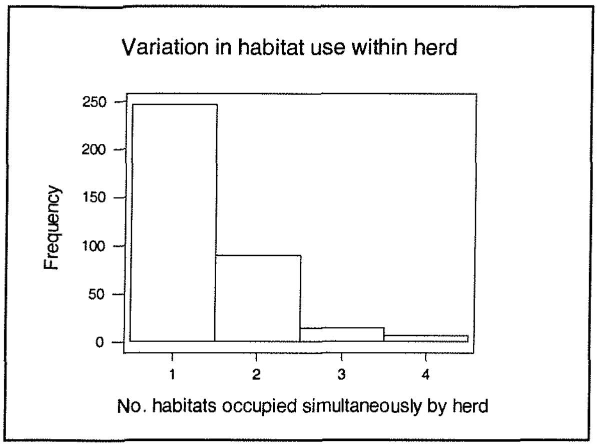


Figure 4. Variation in habitat selection measured as the number of habitats that the herd occupied simultaneously.

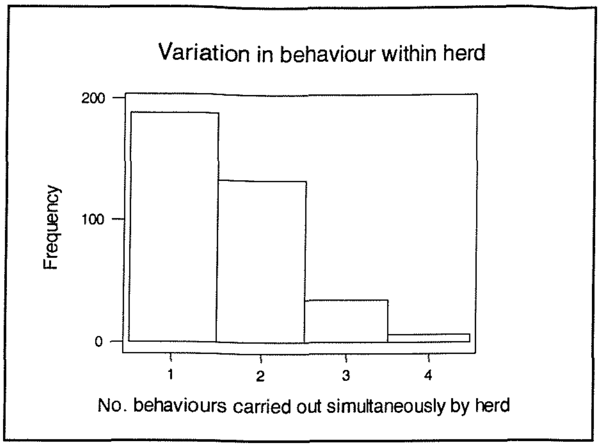


Figure 5. Variation in behaviour measured as the number of behaviours carried out simultaneously within the herd.

Herd composition remained constant throughout each 24-observation, but there was a wide variety in herd size (2-9) between observations.

Individual habitat use and behaviour

There were no significant differences in habitat use between individuals. In contrast to the other livestock groups studied, five out of the six pairs of New Forest ponies at Stoborough showed significant differences in behaviour (see Chapter 2).

24-hour habitat selection and seasonal habitat use

Selection for habitats and seasonal changes are shown in Chapter 2.

Behaviour

As with the other ponies studied, the Stoborough New Forest ponies spent most of their time foraging (Figure 6).

Behaviour in each habitat

Behaviour in habitats where this differs from the Hartland New Forest ponies is shown in Chapter 2. Behaviour in the remaining habitats is shown here (Figure 7). Foraging was the most frequent behaviour in each habitat (Figure 11), and was generally significantly higher than other behaviours. Significant differences were not found where data were limited due to low use of habitat types. Building dry heath (bDH) was included within dry heath (DH). Valley mire (VM) had fewer than 40 observations and was omitted. Trends of particular interest include the unusually high proportion of time spent moving in DH and scratching (included within 'O') in woodland.

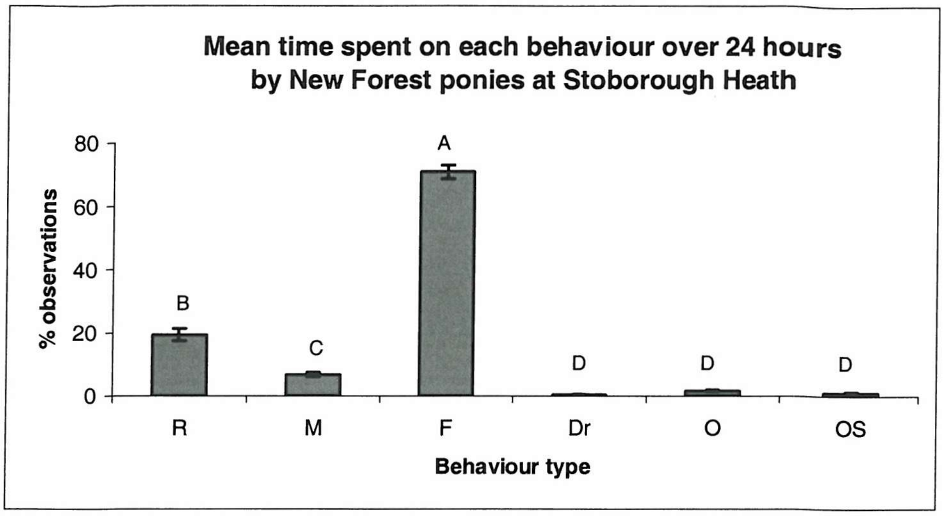
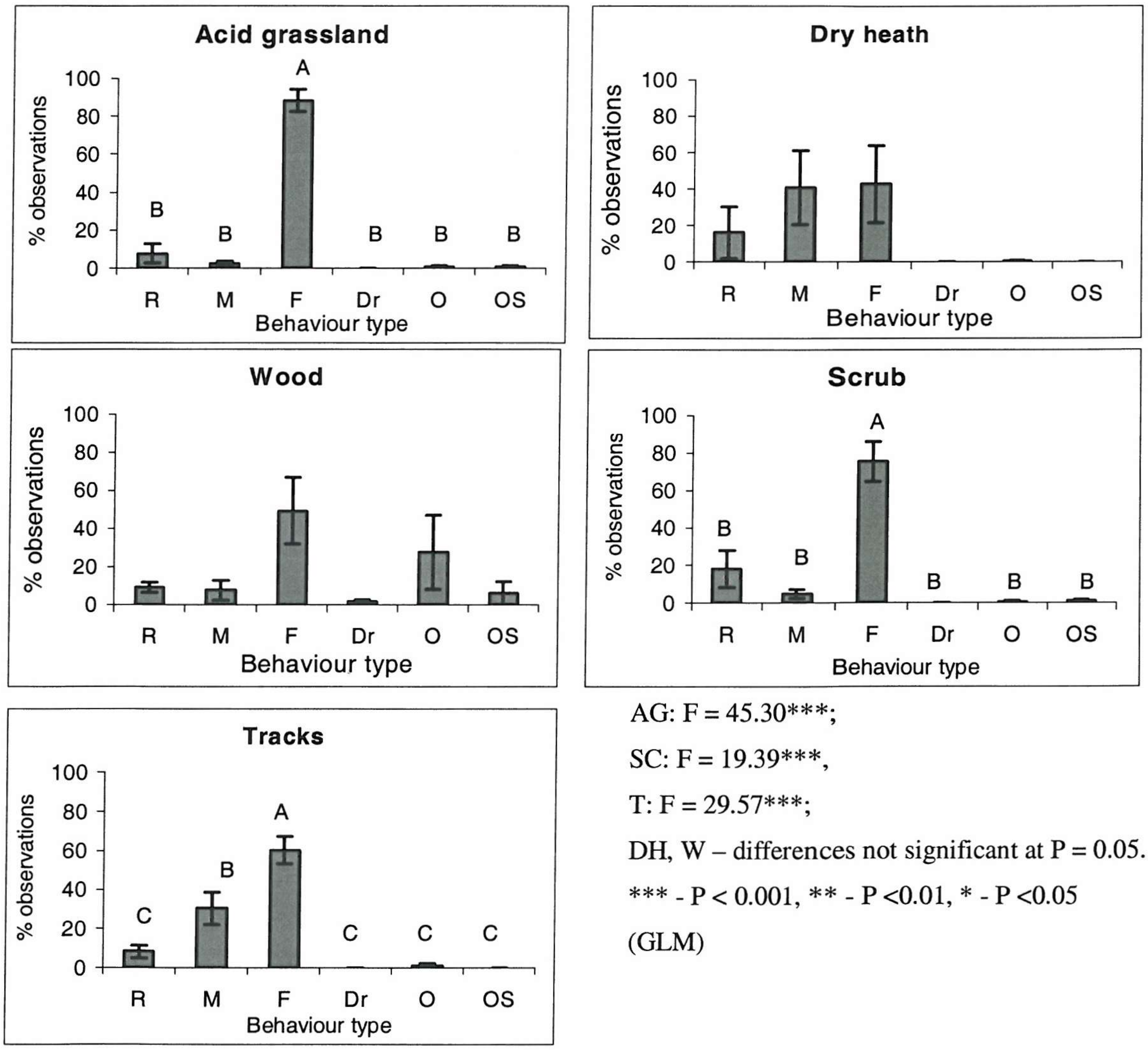


Figure 6. Mean time (i.e. percentage of observations) spent on each behaviour by New Forest ponies at Stoborough. The difference in mean time spent on each behaviour was significant (GLM, F = 222.95, P < 0.001). Significant differences are denoted by different letters, therefore behaviours with the same letter show no significant difference. R – resting, M – moving, F - grazing, browsing and eating hay, Dr – drinking, O – other, OS – out of site.



AG: F = 45.30***;

SC: F = 19.39***,

T: F = 29.57***;

DH, W – differences not significant at P = 0.05.

*** - P < 0.001, ** - P <0.01, * - P <0.05

(GLM)

Figure 7 Mean time spent on each behaviour (% of observations) in different habitats. Significant differences are denoted by different letters. See Figure 6 for behaviour abbreviations.

Selection of habitats for different behaviours

Habitat selection for key behaviours is shown in Figure 8. Selection for individual behaviours generally reflected total habitat selection, with only minor changes in ranking. The relatively high selection for valley mire (VM) for moving was a result of the relatively large July herd moving through VM on one occasion.

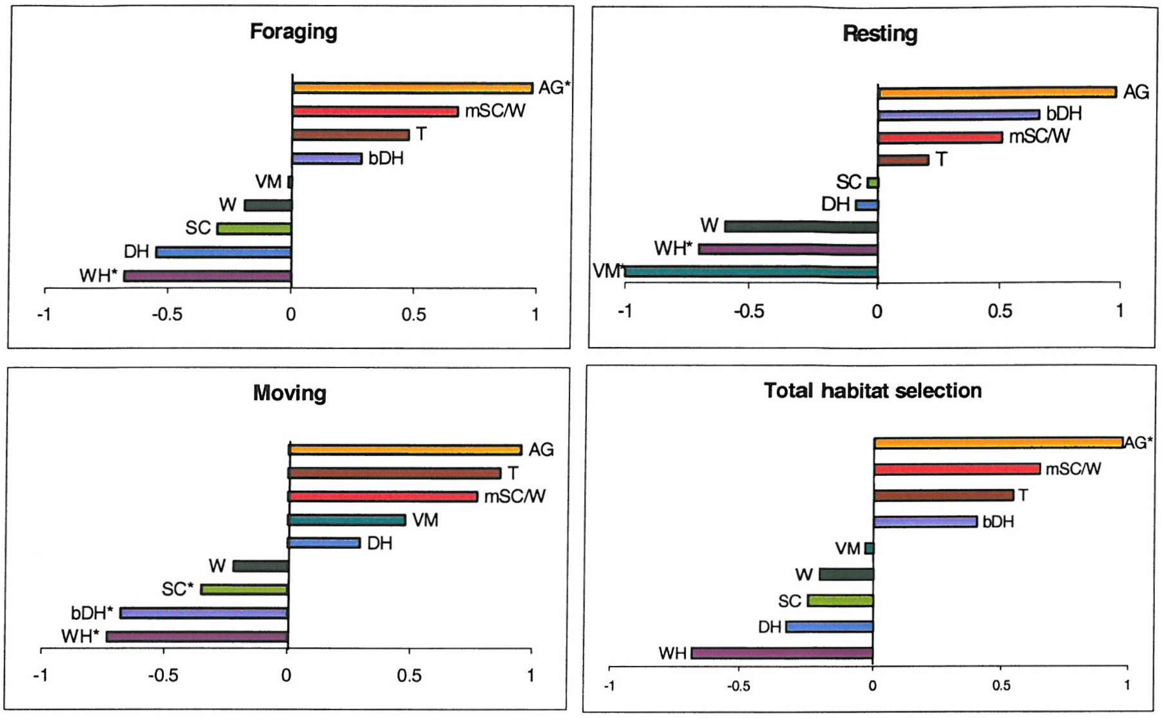


Figure 8 Selection of habitats for resting, foraging and moving plus total habitat selection by New Forest ponies at Stoborough, expressed as Jacobs' indices. Habitats for which the Jacobs' index differs significantly from a median of zero (which indicates selection is in direct proportion to availability) are denoted with * (Wilcoxon signed rank test, $P<0.01$). See Table 2.2 for habitat abbreviations.

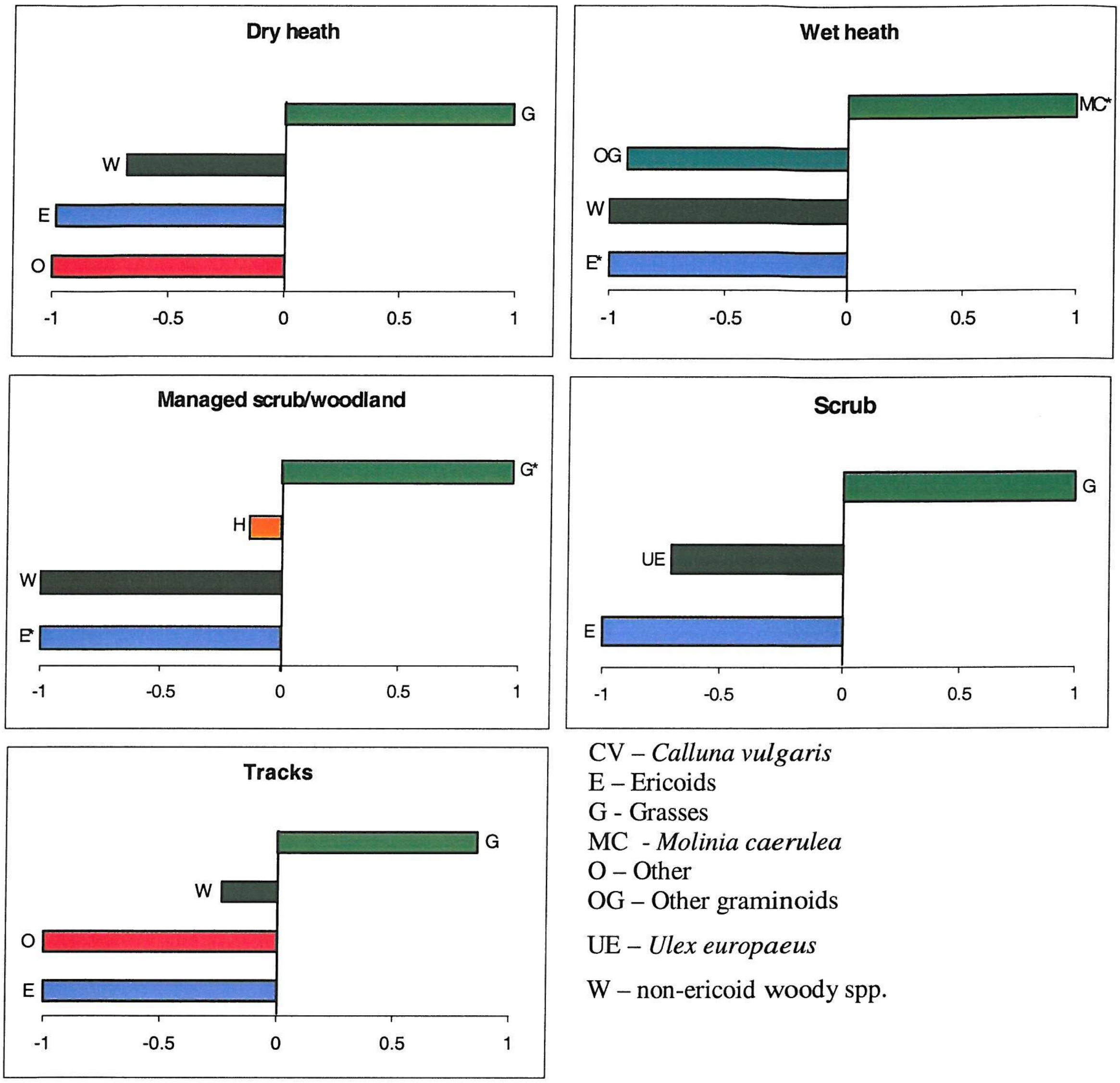
Seasonal changes in behaviours within habitats

Seasonal changes (July, Sept, May compared to Nov, Jan, Mar) in behaviour within each habitat type were not significant. Significant differences between supplementary feeding and non-feeding periods are shown in Chapter 2, together with seasonal trends in habitat selection for key behaviours.

Foraging

Ponies' selection of plant species differed significantly from availability in all habitats (Table 1). Grasses (*Molinia caerulea, Agrostis curtisii, A. stolonifera, Holcus lanatus*) was the only positively selected plant group in each habitat type (Figure 9). Woody species (*Ulex europaeus, U. minor, Betula pendula* and *Rubus fruticosus*) were eaten in dry heath (DH), tracks (T) and scrub (SC) but were never positively selected. *Cirsium arvense* was eaten in managed scrub/wood (mSC/W) and *Pteridium aquilinum* in scrub

(SC) but neither was positively selected. There were insufficient foraging observation in valley mire (VM) for analysis. Only *Molinia caerulea* was observed being eaten in this habitat.



CV – *Calluna vulgaris*
E – Ericoids
G - Grasses
MC - *Molinia caerulea*
O – Other
OG – Other graminoids
UE – *Ulex europaeus*
W – non-ericoid woody spp.

Figure 9 Average selection of plant species/categories as quantified by Jacobs' indices. Species for which selection is significantly different from zero are denoted by * (Wilcoxon signed rank test, $P < 0.05$). Tests were not possible for DH, SC and T ($n < 6$). See Table 1 for full species categories.

Bite rate was highest in acid grassland (AG) and lowest in scrub (SC) and dry heath (DH). Bite rate in W was high compared to other livestock groups (note that the woodland flora was shorter and grassier at Stoborough compared to other sites). There were no significant differences in step rate between habitats.

Table 1 Plant species availability and use average number of bites of each plant eaten). Species which are significantly positively or negatively selected are in bold text. * - P <0.05, ** P < 0.01, *** - P < 0.001. DH – dry heath, WH – wet heath, T – tracks, SC – scrub, mSC/W – managed scrub/wood.

Habitat	Plant species	% cover	% bites	Chi square
DH	Ericoids (*Calluna vulgaris, Erica cinerea, Erica tetralix*) (E)	80.7 (4.9)	3.3	239.6***
	Ulex spp. (W)	10.0 (1.5)	2.4	
	Grasses (*Agrostis setacea, Molinia caerulea*) (G)	8.5 (1.4)	94.3	
	Other (*Pteridium aquilinum) (O)*	0.8 (0.6)	0	
WH	*Molinia caerulea* (MC)	68.1 (6.8)	94.4	17.0***
	Ericoids (E)	28.0 (7.1)	0	
	Other graminoids (O)	0.7 (0.7)	4.6	
	Ulex spp. (W)	2.4 (1.3)	1.5	
T	Grasses (*M. caerulea, A. setacea, A. capillaris*) (G)	76.2 (8.5)	994.4	7.9*
	Ericoids (*C. vulgaris, E. cinerea*) (E)	16.4 (1.9)	0	
	Ulex Spp. (W)	3.4 (0.7)	5.3	
	Other graminoids (*Carex* spp., *Juncus* spp.) (OG)	4.0 (1.5)	0.3	
SC	Grasses (*Molinia caerulea, Agrostis setacea*) (G)	20.3 (5.1)	100	23.4***
	Ulex europaeus (UE)	68.4 (4.4)	0	
	Ericoids (*Calluna vulgaris, Erica cinerea*) (E)	11.3 (2.9)	0	
mSC/W	Grasses (*Molinia caerulea, Agrostis setacea, Holcus lanatus*) (G)	25.1 (3.4)	90.6	84.2***
	Ericoids (*Calluna vulgaris, Erica cinerea*) (E)	3.8 (1.6)	0	
	Ulex spp. (U)	69.5 (7.2)	5.7	
	Non-ericoid woody spp. (*Salix spp., Rubus frusticosus, Betula pendula*) (OW)	0.1 (0.1)	0.1	
	Other (*Cirsium arvense*) (O)	1.5 (0.7)	1.2	

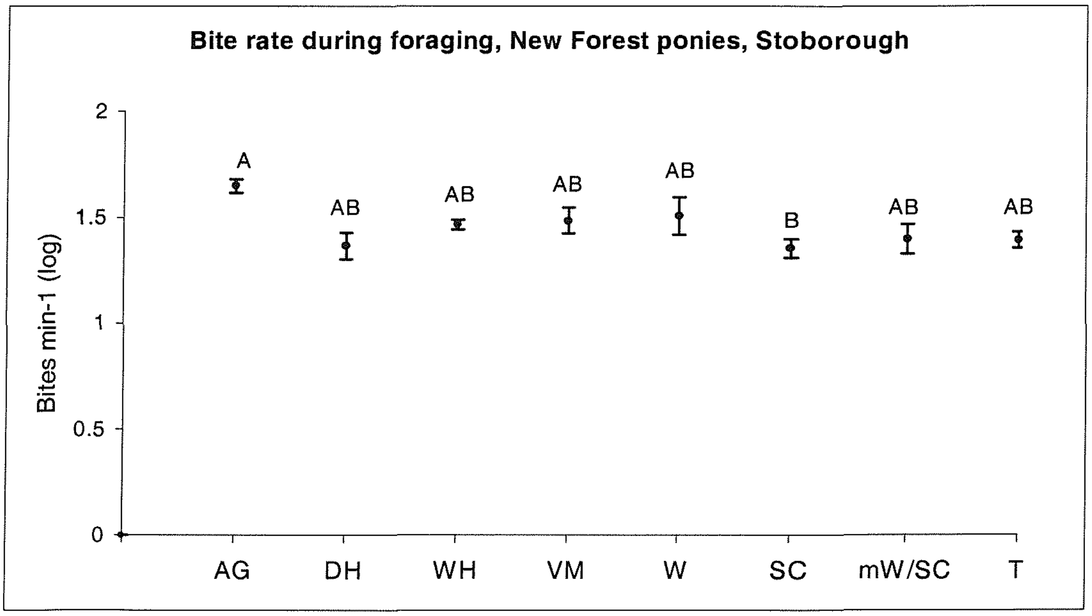


Figure 10 Bite rates in different habitats. Differences between habitats were significant GLM: F = 4.65, P = 0.003. There were no significant differences between individuals.

Appendix V Hartland New Forest ponies

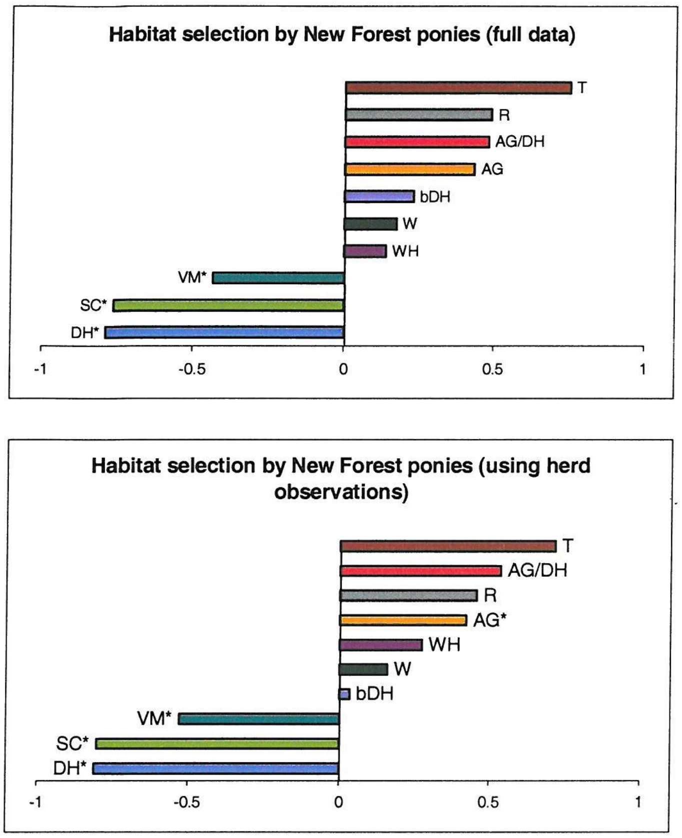


Figure 1. Jacobs' indices for habitat selection by New Forest ponies at Hartland using pooled data over the grazing season (top) and individual observations.

The variation in New Forest pony herd size throughout 24-hour observation periods (see Chapter 2) potentially influenced apparent habitat selection. Group size generally increased when the study herd arrived on AG, and decreased when the herd left as, the group merged with and then separated from other groups already present on AG. This potentially resulted in a bias away from AG and any other habitats where herd size was large (e.g. AG/DH). Therefore, a single herd observation was used in analysing habitat selection – i.e. the habitat that the majority of the herd occupied at each observation time. In practice, the results from single herd observations were qualitatively very similar to the results from using individual observations to give the proportional of the herd occupying different habitats (Figure 1). Consequently, the individual observations were used in subsequent analysis in order to maximise information.

Appendix VI

Table 1 F-values generated by GLM for wet heath and valley mire species at Coombe Bog, Arne CDG – cattle and deer grazed, DG – deer grazed, UG – ungrazed, *** P <0.001, ** P<0.01, NS – not significant (P>0.01) (Chapter 3).

Species	**TREAT**	**BLOCK**	**VEG**	**TREAT * BLOCK**
Bare ground	31.91***	0.99^{NS}	1.47^{NS}	3.69^{NS}
Litter	1.01^{NS}	1.51^{NS}	0.12^{NS}	3.52^{NS}
Calluna vulgaris	0.38^{NS}	6.37**	129.43***	0.42^{NS}
Cladonia ciliata	1.19^{NS}	1.32^{NS}	52.94***	0.56^{NS}
Cladonia portentosa	0.28^{NS}	2.87^{NS}	79.77***	0.11^{NS}
Cladonia uncialis	3.38^{NS}	0.06^{NS}	30.70***	3.38^{NS}
Drosera intermedia	0.55^{NS}	1.65^{NS}	5.00^{NS}	0.42^{NS}
Drosera rotundifolia	0.69^{NS}	2.03^{NS}	6.02^{NS}	4.31*
Erica ciliaris	3.39^{NS}	1.34^{NS}	0.04^{NS}	0.25^{NS}
Erica tetralix	2.62^{NS}	3.91**	152.01***	0.07^{NS}
Eriophorum angustifolium	2.21^{NS}	1.05^{NS}	62.66***	0.84^{NS}
Juncus articulatus	0.81^{NS}	1.25^{NS}	0.51^{NS}	0.17^{NS}
Molinia caerulea	0.34^{NS}	2.19^{NS}	0.03^{NS}	0.70^{NS}
Myrica gale	1.44^{NS}	2.15^{NS}	11.52**	1.44^{NS}
Narthecium ossifragum	1.30^{NS}	4.96**	35.94***	0.57^{NS}
Pinus sylvestris	0.68^{NS}	4.74**	12.94**	1.31^{NS}
Polygala serpyllifolia	1.00^{NS}	2.41^{NS}	3.93^{NS}	1.00^{NS}
Rhynchospora alba	1.17^{NS}	3.86^{NS}	29.47^{NS}	0.28^{NS}
Sphagnum compactum	0.31^{NS}	6.66***	23.83***	0.01^{NS}
Sphagnum cuspidatum	0.17^{NS}	1.73^{NS}	19.11^{NS}	0.17^{NS}
Sphagnum papillosum	1.85^{NS}	5.17**	9.88**	1.85^{NS}
Sphagnum pulchrum	0.22^{NS}	13.11***	29.29***	0.22^{NS}
Sphagnum species (damaged)	0.79^{NS}	1.94^{NS}	19.01***	1.01^{NS}
Sphagnum tenellum	0.99^{NS}	4.21**	47.14***	0.33^{NS}
Trichophorum cespitosum	0.80^{NS}	0.75^{NS}	0.89^{NS}	2.17^{NS}

Appendix VII

Species codes used in Chapter 3 ordination diagrams

Code	Name	English name
Cal vul	*Calluna vulgaris*	Ling
Cla por	*Cladonia portentosa*	-
Cla unc	*Cladonia uncialis*	-
Dro int	*Drosera intermedia*	Oblong-leaved sundew
Dro rot	*Drosera rotundifolia*	Round-leaved sundew
Eri ang	*Eriophorum angustifolium*	Common cotton-grass
Eri cil	*Erica ciliaris*	Dorset heath hybrid
Eri tet	*Erica tetralix*	Cross-leaved heath
Jun art	*Juncus articulatus*	Jointed rush
Mol cae	*Molinia caerulea*	Purple moor-grass
Myr gal	*Myrica gale*	Bog myrtle
Nar oss	*Narthecium ossifragum*	Bog asphodel
Pin syl	*Pinus sylvestris*	Scot's pine
Pol ser	*Polygala serpyllifolia*	Heath milkwort
Rhy alb	*Rhynchospora alba*	White beak-sedge
Sph com	*Sphagnum compactum*	A bog moss
Sph pap	*Sphagnum papillosum*	A bog moss
Sph pul	*Sphagnum pulchrum*	A bog moss
Sph sp	*Sphagnum spp*	Damaged, unidentifiable bog moss
Sph ten	*Sphagnum tenellum*	A bog moss

Appendix VIII

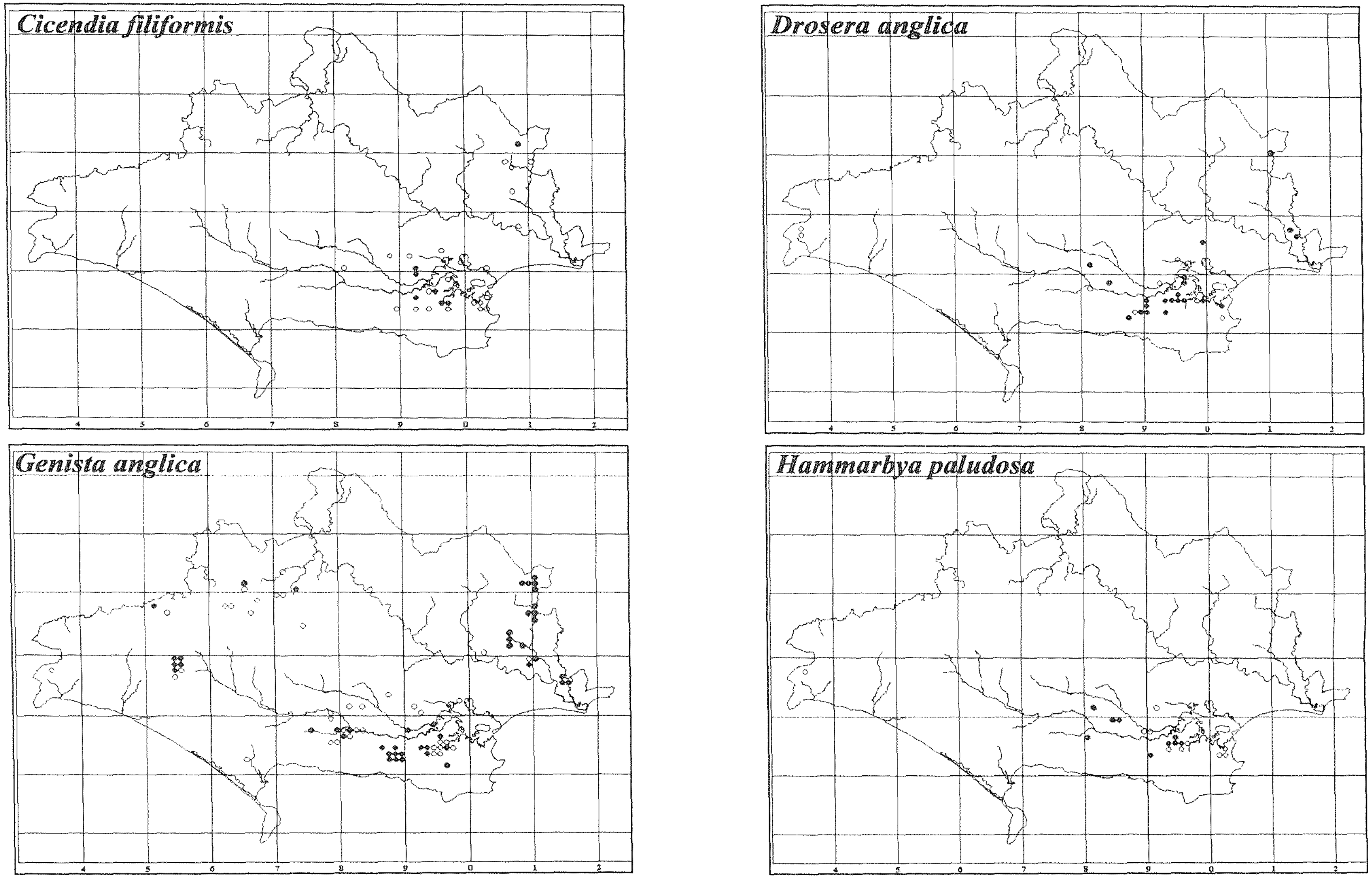


Figure 1 Dorset distributions of species studied in Chapter 4. ○ distribution records pre-1980 ● distribution records 1981-1999.

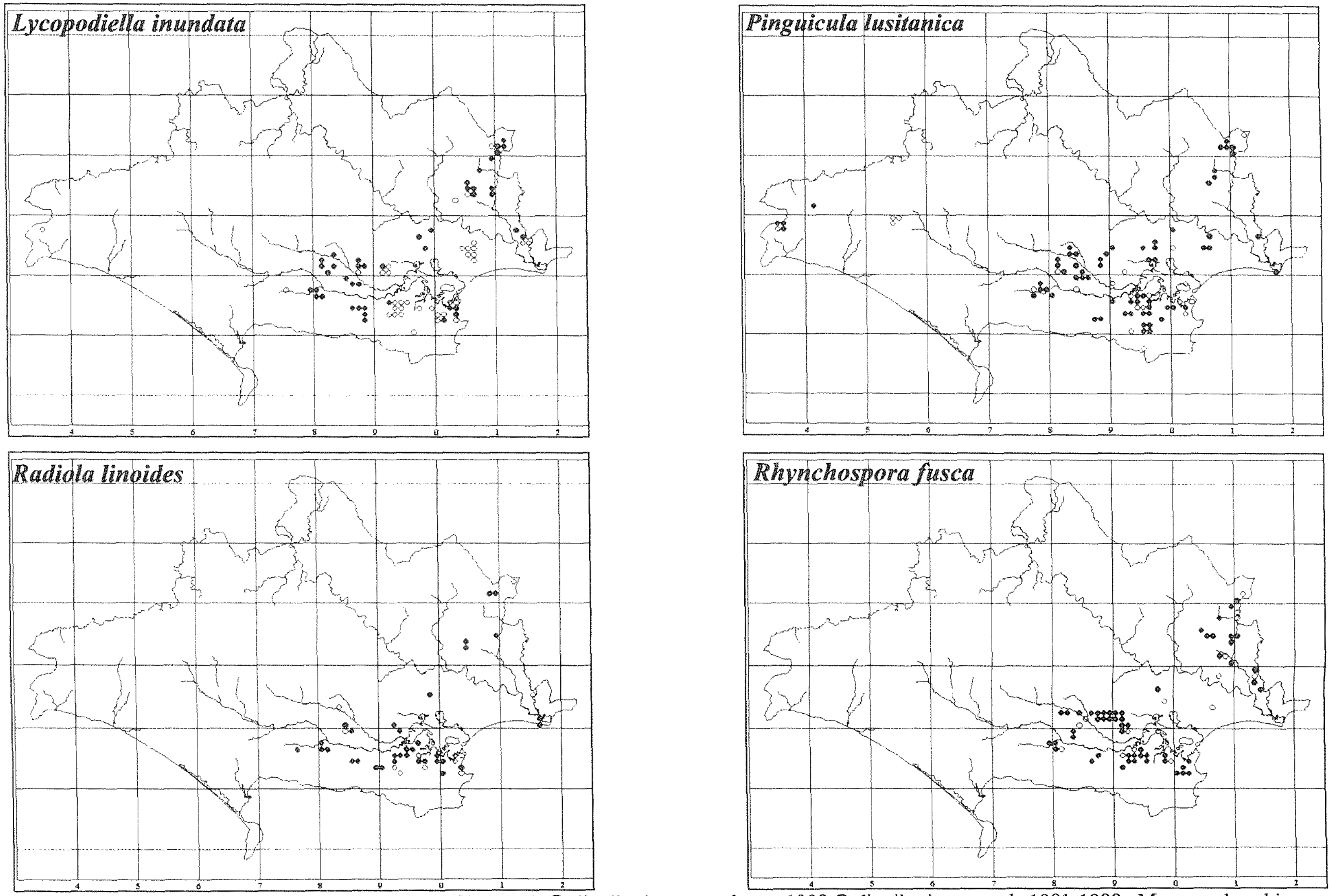


Figure 1 Dorset distributions of species studied in Chapter 4. ○ distribution records pre-1980 ● distribution records 1981-1999. Maps produced in DMAP and kindly supplied by the Dorset Environmental Records Centre.

Appendix IX

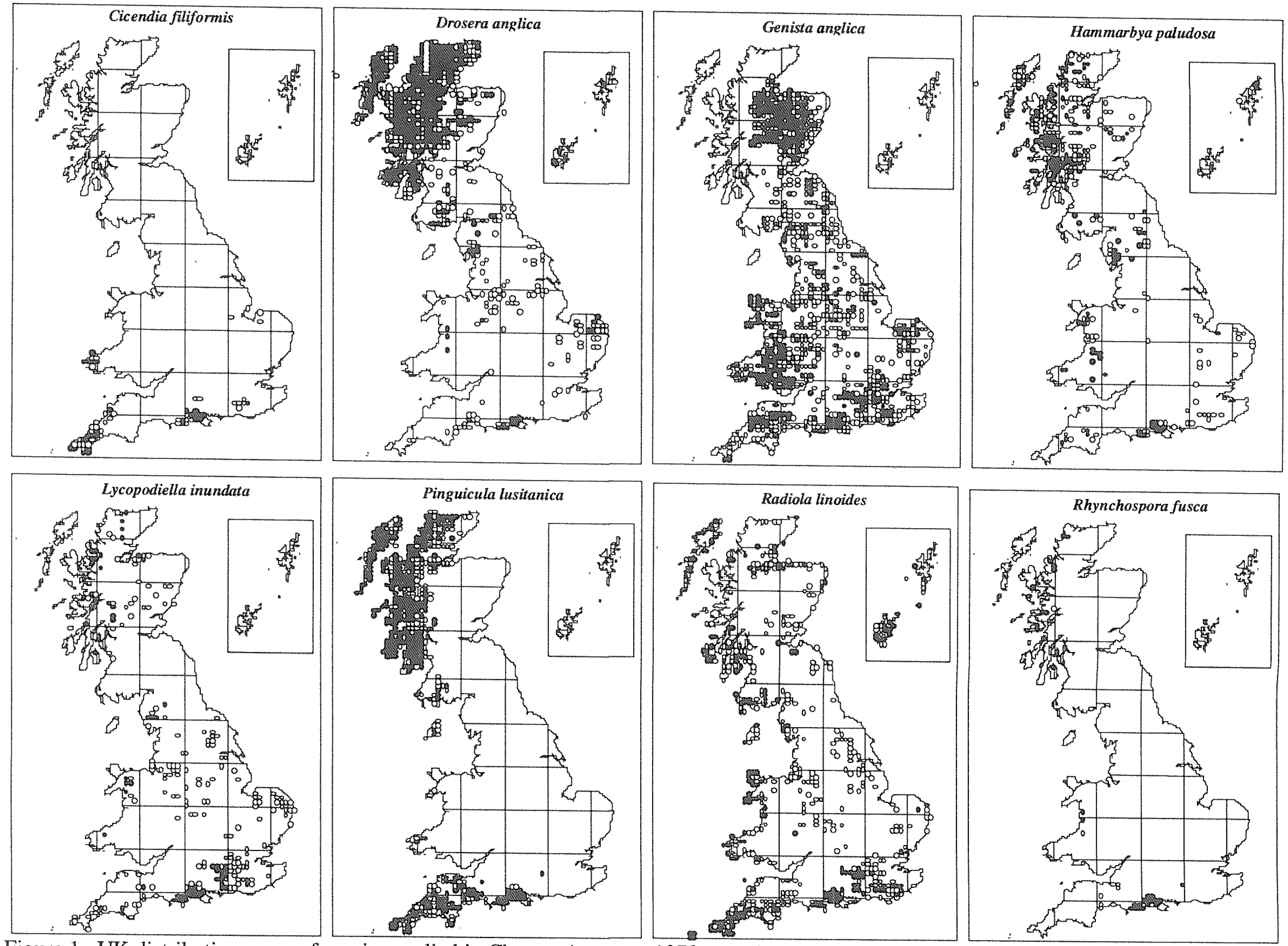


Figure 1. UK distribution maps of species studied in Chapter 4. ○ - pre 1979 records, ● - 1980-1999 records. Maps prepared by the Biological Records Centre, CEH Monks Wood from data collected by the BSBI and others.

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