THE EFFECT OF CUTTING AS A FEN MANAGEMENT PRACTICE ON THE INVERTEBRATE BIODIVERSITY OF THE NORFOLK BROADS.

PhD

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2003

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The University of Birmingham

July 2003

1st of 2 files

Introductory material and chapter 1

The remaining chapters and the references are in an additional file

Acknowledgements

Experts David Henshaw[1], Prof. Robert Cameron, Derek Howlett[2] and Robert Marsh helped with identification of spiders (DH[1]), snails (RC and DH[2]) and beetles (RM) respectively, and many volunteers helped to collect the samples. Especial thanks goes to Jim Watts and Mary Chester-Kadwell, for many hours on their knees in mud looking for snails and groping for flooded pitfall traps. Helpers for the vegetation surveys included Trevor Thorley, Gary Kennison, Simon Allen (all of the BA), Amanda Mann (independent volunteer), and John Jackson (of the NWT). Also thanks go to Dominic Vlasto for allowing sampling on his land and to John Blackburn (NWT warden) for help and use of equipment.

Thanks to my supervisors Dr Andrew Pullin and Simon Allen for help and advice throughout the project. Gary Kennison also helped advise in the first year. The Nature Environment Research Council and the Broads Authority provided the essential funding. Problems with the statistics were solved by Dr Allen White, and Dr C Philip Wheater also gave helpful ideas on analysis. Dr Alexander Ramsay adopted the beetle collection, and Dr Domino Joyce gave continued support, and ideas.

Personal thanks go to my husband Nick Kruft Welton for all the support, patience and encouragement. A special thanks to my parents Sheila and Wim Kruft, whose help looking after the elder set of twins made fieldwork in the final year possible. Finally to the twins, Idonea and Melbryn, and Eartha and Saskia, without whom this would certainly have been completed sooner and with fewer missing notes, but who now have a good grounding in entomology. Indeed 'spider' featured in the elder set's earliest zoological vocabulary, along with cat, dog and, for some reason, owl.

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ABSTRACT

This study sought to discover whether summer cutting of fenlands changes the biodiversity of invertebrates in managed areas as compared to control areas. Following preliminary sampling, reedbeds were chosen for the investigation. The invertebrates studied were Mollusca, Araneae and Coleoptera. Species level changes were investigated in order to identify any specific level responses to management.

All the groups studied were shown to be habitat specific and sensitive to management at the species level. Overall biodiversity and similarity, in terms of presence and absence of species within each group, was not shown to be affected by cutting management. There were, however, some year to year changes in biodiversity and similarity for snails and beetles.

All three groups studied contained species which reacted positively to cutting management, increasing in abundance. There were also species in each group which responded negatively to cutting management, decreasing in abundance.

INTRODUCTION

1.1 Aims and Objectives

Wetlands comprise a wide range of habitats, including lakes, rivers, marshes, acid bogs and fens. The Broads, stretching from Norfolk into Suffolk, is Britain's largest range of wetland habitats. The project set out in this thesis was undertaken with the aim of adding to the knowledge of this wetland, its biodiversity and strategies for management.

The pilot study was undertaken to investigate whether different habitats could be grouped together for the purposes of fen management study. The three groups of organisms (Mollusca, Araneae and Coleoptera) were looked at in order to ascertain the extent of the effect of habitat management on each group.

From the results of the pilot study hypothesis were formulated, and these were tested in the main study. The hypotheses stated that snails are management sensitive, and will therefore subsequently decline in numbers and diversity following habitat management. Further that spiders and beetles are not management sensitive and will not therefore be affected by habitat management in terms of their numbers of individuals or diversity.

1.2 Fenland Ecology

Fenland, for the purposes of this project, consists of those open aspect peatland environments which receive flowing groundwater (mincrotrophic). This is not intended to be an absolute definition, more as a loose guideline. There are of course numerous marginal habitats which would challenge varying aspects of this definition – scrubby fenland, or acid bogs which occasionally flood but do not normally receive groundwater for example – but a full discussion on where to draw each dividing line is outside the scope of this thesis. The interested reader is directed to Keddy (2000) for a detailed discussion relating to definitions of wetlands.

Fenland itself is a marginal habitat inhabiting an ecological space between open water and dry land. The ecology is reliant on three major factors. These are water availability, fertility and disturbance. The balance between flooding, erosion and deposition determines the speed of build up of peat.

Changes in the water table, frequency of flooding, depth of flooding, periods when the water table falls below surface level and at what time of year it does so, all play a part in the ecology of the fen. Water extraction for urban and agricultural use is one problem challenging fen ecology, and water quality is another. Tourism, development and agriculture all exert pressure on the water systems. Tourism and leisure activities such as boating and fishing can cause disturbance, erosion of water courses and pollution. Development both for domestic reasons and tourism cause loss of habitat and further pressure to extract water from the water table, lowering it further. Drainage also makes the peat more susceptible to fire and erosion. Dried out peat will oxidise and acidify, again affecting the communities that make up the habitat (Foss and Connell 1998).

Without management, drainage ditches and dykes degrade. They are important for the hydrology of the fens, and contain many specialist freshwater species such as the shining ramshorn snail *Segmentina nitida*

(RDB endangered) and the great raft spider *Dolomedes plantarius* (RDB vulnerable). Ditches and dykes need to be dredged on a rotational basis (Sutherland and Hill 1995) to prevent overgrowth and erosion. Habitat patches are important as refuges during dredging.

The level of nutrients in and coming into the system (Keddy 2000 refers to this as substrate fertility) also helps to determine the vegetation communities present. Pollution e.g. from herbicides, pesticides or industry and eutrophication e.g. from sewerage or fertilisers all have major impacts on peatland specialists. Peatland habitats are naturally nutrient poor and specialists are adapted to these conditions. Eutrophication severely effects the biodiversity of the affected area (Tolhurst 1997). Much fenland specialist floral species are poor competitors when fertility in a fen is increased, compared to pioneers such as the common nettle (*Urtica dioica*).

Disturbance can be intense and short lived, such as fire or mowing, followed by a period of recovery. The frequency of this sort of disturbance is crucial to the ecology. Few species can survive intense disturbance of this sort on a regular basis. Most species, however, rely on it in the long term as it allows regeneration and controls succession to scrub. Low intensity, continuous disturbance can be an important factor controlling the environment to its benefit, or a chronic problem degrading it. Disturbance at this level includes processes such as grazing and trampling, which can be beneficial in one season or habitat, but destructive in another.

Succession can be a problem if disturbance is too infrequent or not effective at controlling scrub. As trees recolonise the habitat they change the hydrology of the fens, using up water and preventing rainfall reaching the ground. They also shade out areas of the fen, changing light availability. They act as natural barriers reducing airflow and changing the microclimate. Importantly scrub adds litter to the fen or bog and this enriches it. Trees also increase the number of available niches for wildlife, which on a limited scale can be a good thing, but on a larger scale the wetland is changed and fen specialist species are pushed out (Whild *et al* 2001).

To restore degraded fenland the degradation must be reversed. The water table must be raised again and succession to scrub reversed. Any influx of pollutants should be stemmed (Foss and Connell 1998). Succession can be reversed or slowed or stopped using grazing at an appropriate stocking density, or cutting. Cutting can take place in the summer or in the winter. There are advantages and disadvantages to each strategy and they should be carefully considered before being applied. Prior management history should be taken into account and the regime not changed unless the habitat. is already degraded (Foss and Connell 1998). Many studies show that restored wetlands are superior in quality to recreated wetlands, (e.g. Doshi *et al in press*) for any number of parameters from biodiversity to hydrology to attractiveness for migrating birds. This in itself is an incentive to maintain and improve existing degraded wetland habitats in preference to trying to recreate new ones.

1.3 Historical Ecology and Land Use

Historically fens and reedbeds were a local resource and were maintained by the day to day use of the local population. Scrub species were kept at bay as the wood from young trees was used in many ways. Willow

(*Salix* spp) and alder (*Alnus glutinosa*) were used for example in basket weaving and as fire-wood. Willow is still used as the tree of preference for making cricket bats. Peat was extracted at a sustainable rate for use as a fuel, and this slowed the build up of reed litter, allowing the regeneration of reedbeds. Reed itself (Figure 1.1), and saw-sedge were used for thatching, and there has recently been an upsurge in demand for this commodity, with a revival in the traditional practices (Hawke and José 1996). Other species such as bog myrtle (*Myrica gale*) were also used in a variety of ways ranging

Figure 1.1 Reed is still used as a material for thatching. It is a commodity for which reedbeds have been traditionally managed for centuries.

from an insect repellent to a protection from witches (Simpson *et al* 1996).

More botanically diverse fens provided rush (*Juncus* spp) and sedge (*Cladium mariscus*) for flooring, thatching, fodder and bedding (Figure 1.1). The marshes and wet meadows were used for grazing in the summer after cutting. Some areas, known as washes, were grazed in the summer and flooded in the winter as a form of flood protection (Sutherland and Hill

1995). More recently reedbeds have been planted as a form of water filter (Hawke and José 1996, Hudson 1992) to help mitigate the effects of eutrophication from agricultural run-off or sewage treatment plants.

1.4 Historical Ecology of the Broads

The Broads are the largest stretch of wetland in England with 125 miles (200km) of navigable waterways stretching between Norwich, Stalham, Lowestoft and Beccles. As recently as 1960 it was thought that the broads themselves were natural phenomena, but studies by Dr Joyce Lambert (Bartlett 1993) and corroborating evidence, such as the vertical rather than sloping sides to the lakes, changed opinion. The broads originated between the 9th and 13th centuries, and were formed initially by generations of Norfolk inhabitants digging peat for fuel. This activity became commercialised in the Middle Ages when an Abbey acquired the rights to peat-cutting. The demand for peat must have been huge. Documents show that one monastery in Norwich alone used 200,000 bales of peat per year, and Norwich Cathedral Priory accounts show 400,000 turves burnt a year (Bartlett 1993). Within 200 years nine million cubic feet of peat had been extracted. These gradually flooded to become the broads present today.

For many centuries natural succession from fen to fen carr woodland was kept at bay using the traditional practices of peat and turf cutting, reed and sedge cutting for thatching (Figure 1.1), collecting litter for use as cattle bedding and the harvesting of marsh and fen hay as winter feed for cattle. Cutting was originally done by hand using a scythe. Many areas of the Broads were traditionally managed as grazing marsh for livestock or as refuges for game birds, such as the pheasant (*Phasianus colchicus*). With the reduction of such traditional management in the latter half of last century much fenland has been lost to woody scrub and wet woodland and over the past 50 to 80 years scrub regeneration has spread unabated. With the spread of scrub and the draining of many marshes, important wetland and fenland habitats have shrunk, threatening many wetland specialists such as the marsh harrier (*Circus aeruginosus*), the bittern (*Botaurus stellaris*), Cetti's warbler (*Cetti cetti*), the swallowtail butterfly (*Papilio machaon*) (Figure 1.2), the hen harrier (*Circus cyaneus*), the great water parsnip (*Sium latifolium*) and the fen orchid (*Liparis loeselii*).

Figure 1.2 The UK distribution of the swallowtail butterfly is restricted to the Norfolk Broads.

With the advent of the industrial revolution, land management practices such as reed cutting and livestock grazing started to die out, whilst over the same period the population has increased dramatically, putting pressure on limited resources (e.g. increased water extraction) and increasing disturbance. Additionally the tourist industry has increased exponentially. All this has lead to the decline of the fens and habitat loss. Bibby *et al* (1989) estimate that up to 40% of reedbed has been lost since 1945. Norfolk has just 2500ha of fen remaining (Madgwick *et al* 1994), yet this is the largest area of wetland in Britain. The Broads Society (www) suggests that 60% of the 5225 ha (12,900 acres) of fenland in the Broads has sallow willow and alder encroachment, turning it into wet carr woodland. The BA's own figures agree, estimating that around 2000 ha of the remaining 5000 ha of undrained fen is currently clear of carr woodland (www).

One of the ongoing problems the BA has had to face is the progressive abandonment of fens and marshes matched with the increase in urbanisation and intensive farming practices. This loss of traditional management practices, coupled with the increased pressure on the Norfolk water table from the increase in population and tourism has meant that a lot of the Norfolk landscape has converted through natural succession to wet woodland.

One of the biggest problems in the Broads is scrub regrowth due to the change in land use patterns (Tolhurst 1997). Scrub regrowth has been aided by drainage of vast tracts of Norfolk and East Anglia for agricultural and urban development. Drainage and lowering of the water table damages the bog or fen and allows different communities of plants and animals to develop.

1.5 Conservation Management

Only recently have wetlands been managed for their conservation value. According to Keddy (2000) "hydrology and fertility are the two key factors that determine the kinds of wetlands found in a landscape". Different hydrological regimes lead to different types of wildlife and consequently much wetland management is aimed at maintaining the hydrological variation found in wetland areas and reducing the rate of eutrophication, litter and nutrient build up (Keddy 2000). Reedbeds are defined by Wheeler (1992) as containing more than 75% *Phragmites* spp.. Commercial reedbeds generally comprise more than 90% *Phragmites* spp.. If reedbeds are left unmanaged the build up of litter allows them to dry out and revert to scrub and carr woodland (Haslam 1972). Conservation management therefore focuses on the early successional stages for reedbed and fens. A reedbed managed by summer cutting and shallow summer flooding will tend towards a more floristically diverse tall herb community, whereas managing using winter cutting and summer flooding encourages a more monoculture reedswamp community.

In general it is not advised that a traditional management regime should be changed, as the species present on the site are those that are well adapted to that management, particularly if the practice has been carried out over many years, and has not lapsed. Where there has been no management for several years, or inadequate management then restoration of the reedbed or fen may be needed. Adjusting the hydrology of the site can encourage reed, increase litter breakdown, facilitate cutting and provide aquatic habitat for wildlife (Hawke and José 1996). Reed prefers to grow in water averaging in depth from surface level to 20cms deep (Burgess *et al* 1995), however reed may not be the main consideration. Flooding to 20cms encourages the bittern (*Botaurus stellaris*) but summer flooding can kill milk parsley (*Peucedanum palustre*) the food plant of the caterpillar of the swallowtail butterfly (*Papilio machaon*), and is also detrimental to soil invertebrates. Allowing parts of the fen to revert to scrub (suggested amount 15% around the margins of the site, Hawke and José 1996) can encourage a range of invertebrates, and also the endangered Cetti's Warbler (*Cetti cetti*), but this requires a drier site than that preferred by the bittern.

Keddy (2000) claims that "Europeans accept intensive management (e.g. cattle grazing, peat cutting, mowing), whereas North Americans tend to prefer natural controlling factors (erosion, fire and flooding). A critique of Keddy's wide-ranging statement and its possibly over-generalised management theory is beyond the scope of this introduction, however it is true to say that many European wetlands have been maintained or restored by intensive management practices.

Mowing or cutting vegetation will slow eutrophication by preventing litter build up, so long as the litter is removed from the site. Leaving piles of cuttings by the edge of the site can make good refuges for invertebrates but cutting should not be used to fill ditches, hollows or left at the edge of carr scrub as this is detrimental to invertebrates. Tussocks are important overwintering habitats for beetles and spiders (Rushton *et al* 1990) and should not be damaged by mechanical mowers. Summer mowing should be carried out after the ground bird nesting season (late July or August), and can be followed up with grazing. Fens dominated by sedge (*Cladium mariscus*) benefit from a 3-4 year cutting rotation. Cutting stimulates new buds and can provide temporary open habitat as well as discouraging reed encroachment if summer cut. It can be carried out on a range of timescales. Short rotation (single or double wale) in winter is best for commercial reed (Hawke and

José 1996, Sutherland and Hill 1995), whereas longer rotation (3-15 years) is better for conservation. Older reed, with many dead stems in amongst the regrowth provides cover for birds, such as the reed warbler (*Acrocephalus sciraceus*) and the dead stems themselves provide overwintering sites for invertebrate larvae such as the twin-spotted wainscot (*Archanara geminipunctata*).

Grazing on the other hand is a different approach to the same problem. Cattle provide a variable vegetation structure at low stocking density (not more than 0.5 cows/ha), whereas sheep grazing tends to be uniform. Horses crop closely in places, but have latrine areas which quickly become rank. The Irish Peatland Conservation Council (Foss and O'Connell 1998) suggests stocking densities of 0.2 ponies/ha in the winter rising to 0.3/ha in the summer. Light grazing in the summer months between July and October avoids many of the flooding problems and associated welfare issues found on very wet sites in the winter. Sheep should be stocked between 0.25 and 0.37/ha. Overstocking can be potentially very damaging to fens, especially if the fen is wet. Mechanical damage, from trampling and footprints is known as poaching. Limited poaching can be useful as it opens up the ground for regenerating species. The fen violet (*Viola persicifolia*), for example requires disturbed soil for germination.

Burning is a management practice that has been used in the past to help regenerate badly neglected wetlands. A study by Ditlhogo *et al* (1992) found no significant difference between the effects of cutting and cool burning on invertebrates when the burn was carried out on wet fen. Many authors (e.g. Cowie *et al* 1992, Foss and O'Connell 1998, Hawke and José

1996, Sutherland and Hill 1995) urge caution when applying this practice, particularly on drier areas or in summer months.

1.6 Conservation Management in the Broads

The BA is responsible for managing around 30,000 ha (74,000 acres) of Broadland. Management for local biodiversity has helped to encourage an increase in tourism in the Norfolk Broads. The Broads are an oasis of wetland habitat and as such provide a home for many specialist species. Over 250 species of plants alone inhabit the Broads, many of which are confined to this region. It also provides an essential stop over habitat for migrating birds such as the osprey (*Pandion haliaetus*), which nests in Scotland. Fenland in the Broads occupies 5225 ha (12,900 acres) although 60% has scrub – willow and alder - encroachment, turning it into wet carr woodland.

One of the main duties of the BA is to conserve and enhance the natural beauty of the Broads. This includes protecting and restoring and where possible improving habitats capable of supporting quality wetland communities. Current management in the Broads includes restoring the fens to their former (1920s and earlier) open aspect by reducing the amount of scrub and fen carr woodland in the area to around 15%. Farmers are encouraged to use traditional practices in environmentally sensitive areas, to minimize damage, drainage and disturbance.

1.7 The Broads Authority Project

The Broads Authority faces a huge task of trying to restore degenerated fens. The University of Birmingham (UoB) and the Broads

Authority (BA) in Norfolk have worked together for a number of years on projects such as the Large Copper reintroduction programme. These links led to the set up of an integrated project on fen management in 1997. The BA received the go ahead that year to investigate the effects of using a new machine in fen management. This machine, the fen harvester (Figure 1.3), had been used with some success on the continent for several years, but it's precise effects on the environment had not been tested. The machine itself is a caterpillar-tracked combine harvester suitable for cutting huge swathes of the fen at once.

The fen harvester

Figure 1.3 The fen harvester is designed to cut fen and reedbeds. It has caterpillar tracks to spread the weight of the machine, limiting the damage to the peat surface.

The use of the fen harvester would solve a lot of current management problems, or rather lack of management problems. In particular it would address the problem of limited resources being available to fund the labour intensive practices by which the Broads were traditionally managed. It cuts large areas of land quickly using little manpower compared to traditional methods, thus significantly reducing the costs of this type of management. The fen harvester however is large and heavy and potentially destructive. It is designed to have low ground-pressure and to be driven over the fens to cut, collect and process the vegetation without causing much mechanical damage to the peat surface (Hawke and José 1996), however this had not been scientifically tested.

The BA is trying to remedy the situation with little manpower and limited resources. To recreate the habitat of the past large areas of the fen need to be managed. This could be done using the fen harvester, or alternatively the introduction of grazing animals to the fen could be the answer. First it is essential to know how cutting and grazing affect the fen.

The BA set about cataloguing the effects of the fen harvester on various aspects and habitats in the Broads. The cutting regimes the BA decided to test included the height of the cut, and the difference between the mechanical fen harvester compared with a hand worked device called a Bücher mower. They also set up experiments to test the difference between summer and winter cuts. The entire range of vegetational habitats present in the Broads was studied and habitats were chosen ranging from pure reedbed to mixed fen to eutrophic fen to sedgebed. Additionally they also decided to test the effectiveness of grazing stock as a fen management tool.

The Irish Peatland Conservation Council (Foss and Connell 1998) describes mowing as "an essential management tool in maintaining a fen habitat" but in the same breath warns that for wet sites "the passage of machinery is likely to do more damage than good". Bearing this quandary in mind the Broads Authority (BA) devised this study to quantify the exact effect of mowing using heavy machinery compared to mowing using a hand pushed mechanical reciprocating mower (Bücher mower).

Winter cutting doesn't disturb breeding animals such as birds, nor does it interfere with the seed production and flowering of the fen plants. However access to sites may be difficult as often water levels are higher during the winter. Summer cutting in general has the advantage that it allows diversification of communities by reducing standing crop (Hawke and José 1996) although Gryseels (1989) found the vegetation remained species poor despite a change in the composition of species. If the plan is to reduce nutrient content of the fen then removing the vegetation when it is at its highest would be desirable. Different vegetation types react differently to summer of winter cutting - some species are stimulated to better growth following cutting but others are eradicated. Whereas most species tend to withstand regular cutting, reed (*Phragmites australis*) does not, and sites should be cut in the winter if managing for reed (Tolhurst 1997).

An alternative method of managing large areas of the fen easily is to graze using cattle, ponies or sheep. Different grazers effect the vegetation in different ways. Comparisons can be made between cattle, Welsh and Konik ponies (a.k.a. Konig ponies) (Figures 1.4 and 1.5), sheep and red deer in the way that they use a site. Ponies tend to be more selective in their choice of grazing, for example, and leave a more patchy, variable habitat. Welsh and Konik ponies use the habitat is slightly different ways. Cattle, whilst still producing an irregular habitat, tend to reduce the height of vegetation more uniformly over the whole area (Tolhurst 1997). Cattle also cause more structural change to the soil as they are larger, heavier animals (typically

400kg to a pony's 150kg). This affects the species that live or grow on grazed sites. Previous studies such as Zulka *et al* (1997) noted that catch rates, when sampling spiders, are affected by management such as grazing. Grazing alters the habitat structure and creates numerous microhabitats. Zulka *et al* (1997) surmised that habitat structure was an important influence on the numbers of spiders in the habitat.

Figure 1.4 Konik ponies were used to graze the site at Hickling.

Grazing is low maintenance - very little manpower and little equipment is needed, and it can be all year round depending on how productive the fen is, i.e. the nutrition available to the animals and whether supplementary feeding is required. It is also dependent on how waterlogged the fen is, which affects the animals' welfare. Grazing leaves natural habitat patches if not over grazed. One problem is working out the right stocking density for the fen in question. This is not an absolute value and may change with the seasons and experience of the animals. Animals experienced in fens will forage better and more efficiently that animals new to a site. The breeds should be hardy. There can be problems of trampling, and enrichment from dung and urine (Foss and Connell 1998) and ultimately the decision to graze a site for conservation purposes must be taken individually based on the importance of the site in question.

1.8 Measuring Biodiversity

Biodiversity is a buzz word that has become well used in the past decade. Most measures of biodiversity revolve around 'how many' and 'how different' things are in one area compared to things in another. However, if biodiversity is defined as the "irreducible complexity of all life" as in Williams *et al* (1994), then biodiversity cannot be reduced to one parameter

Figure 1.5 Welsh ponies were used to graze the site at Broad Fen.

and therefore "total biodiversity ... is not directly measurable" (Margules and Williams 1994).

Simplistically a measure of species richness alone may seem to quantify biodiversity and if this were true it would allow easy comparisons between sites and groups. However Humphries *et al* (1995) cite an example where two sibling species of daisy would be considered less diverse than a species of daisy and a columbine. This shows that biodiversity has a component of taxic diversity, or species composition. The example could be expanded to note that two species of daisy plus the columbine would be considered yet more diverse, and so biodiversity is some function of both species richness, and species composition.

Other factors need to be considered. Harper and Hawksworth (1994) consider a system to be more diverse if the species in it are equitable i.e. if there are equal abundances of species, or even more commonly if the species closely follow a Poisson distribution (Hammond 1994), rather than a system where there is one dominant species, and many, relatively rare, non-dominant species.

Many other measures of diversity have been used or suggested. Such measures include the number of endemics, the complexity of the habitat, the length or complexity of the food chains, trophic level diversity, life-style diversity, evolutionary potential and functional diversity (see Gaston 1996, Harper and Hawksworth 1994). As Norton (1994) points out: "it appears that scientists can offer a very large number of possible 'diversity measures', but that these measures cannot be aggregated into a unique measure of the diversity of the system". Only by measuring every parameter could we gain some deep insight into the biodiversity of a system. These parameters, however, stand apart, and attempts to combine them have been singularly unsuccessful as the separate measures are non-additive. They measure different things - they are “apples and oranges”. Adding more apple-iness will not compensate for a lack of orange-iness. Further, if this ‘whole of biodiversity’ is to be measured then comparisons cannot meaningfully be made across areas and between sites. This leads us full circle back to the fuzzy, irreducible concept of biodiversity with multivariate boundaries.

Use of the concept of biodiversity is severely limited by the inability to sum the various measures of it. In order to make ‘biodiversity’ tractable a simplification must be made, and the ‘best’ or most meaningful measure of biodiversity chosen. Whatever measure is ultimately chosen places value on that parameter (Williams 1996). When designing a study, for example, care must be taken to ensure that the data to be collected truly reflects the aspect of biodiversity that is to be measured and compared across sites. For example, presence/absence records alone will give no information about relative abundances, and unless absences are recorded as positive absences (i.e. looked for but not found), then range size data cannot be compiled, nor meaningful rarity scores estimated.

The most commonly measured surrogate of character richness (the current accepted currency of biodiversity) is probably species richness but this is by no means the only one, even though it is too often used as though it encompassed the entirety of biodiversity on its own (Gaston 1996). Species richness captures many of the facets of biodiversity, (Gaston 1996), and the strong relationship between character richness and species richness greatly reduces the demand on the data, and therefore the cost of any study. Species richness is therefore often used as a baseline to biodiversity studies. If species richness is to be used as a measure then it is necessary to have some globally accepted concept of what constitutes a species.

At present there are any number of different concepts, and hence methods of application both within and between groups can be different and even conflicting. Cracraft (1992) adequately demonstrates the difference alternative concepts can make using the Paradisaeidae (birds-of-paradise) as an example. The biological species concept of Mayr (1957) recognises 40-42 species, whereas Cracraft's phylogenetic species concept recognises around 90 different species. Harper and Hawksworth (1994) sum this up nicely when they say that "If the unit of measurement is itself variable, conclusions based on it have necessarily to be treated with considerable caution". Genealogies are not needed if species richness is taken on its own, and this again reduces the demand on data. Phylogenetic differences are not taken into account. Even so species richness can only accurately be measured for very small sample sizes. In practise all taxon biological inventories (ATBIs) are prohibitive in terms of time and expense for all but the smallest studies. Despite this Hammond (1994) argues that they are ultimately the only way forward if biodiversity is to be usefully studied in the future.

It is not always necessary to use an absolute measure of species richness; relative measures (snapshots of biodiversity) can often be applied, which cut down on the time and expense of a survey. However such relative measures can only be used when there is already some idea of how the relative measure relates to the absolute measure (Hammond 1994) within some reasonable margin of error. Using data from previous studies may lead to ambiguous or inaccurate results since it is almost always impossible to know how complete previous surveys were (Hammond 1994). Similarly species richness must be compared over areas of the same size (Gaston 1996) due to the species-area relationship (MacArthur and Wilson 1967) whereby the number of species doubles for every tenfold increase in area (Wilson 1992). This may be easier to say than to do, as similar sample areas may stretch over heterogeneous habitat, thus adding yet another variable to the problem. Hence it may not be possible to accurately extrapolate from one study to a larger study (Colwell and Coddington 1994).

For larger sample areas, higher taxonomic surrogates need to be found. Gaston (1996) and Williams and Humphries (1996) state that species richness correlates positively with higher taxonomic richness, but again higher taxonomic surrogates can only be used with knowledge of how their numbers relate to the absolute diversity (Hammond 1994). The magnitude of the study in question delimits the most appropriate surrogate to use. There is a direct trade off between the ease of carrying out the study and the accuracy and resolution that the surrogate can supply. For example mapping a thousand families in an area will give an idea of the overall character diversity and maps significantly more of the spread of diversity than mapping a thousand species (Williams and Humphries 1996). However mapping the species belonging to those 1000 families would give a much more direct measure though it would take its toll in the cost and duration of the study. However in this study the invertebrates were identified to species level, where possible, so that the relative abundance of particular species (i.e. RDB and Notable species) can be accurately assessed.

1.9 Choosing the Invertebrates

Most studies relating to biodiversity have used vertebrates (Pearson 1994) rather than invertebrates (Niemelä 1997 and references therein) and these have often been proposed as indicator taxa, umbrella taxa and flagship species (e.g. Heywood and Watson 1995, Stork and Samways 1995). Flagship taxa include the familiar species which form the media image of many wildlife charities – pandas, tigers, whales and occasionally invertebrates such as some large butterflies. These flagship species are charismatic popular species used to raise awareness and funds in order to stimulate conservation action. National rare endemic species such as the kiwi are also used as flagship species. Higher predators and larger animals, such as wolves, large cats, elephants and many raptors, like the condor, are often used as umbrella taxa to indicate the overall health of a landscape. By protecting these species, which have large home ranges, the theory is that other, less prominent species, will also be conserved within the same environment. Specific indicator taxa can have more specific correlations, for example, the extent of a prairie dog colony is an indication of the likely numbers of its predator, the endangered black-footed ferret. Key-stone and indicator taxa are often difficult to identify without detailed study in an environment. General indicators of biodiversity have been proposed, but are rarely tested. The most frequently used indicator appears to be floral diversity, though this appears assumed rather than rigorously tested in many studies. Panzer and Schwartz (1998) plants coupled with area size were a useful indicator of invertebrate species richness, but go on to say that this was only 80% accurate, and suggest that a 'shopping basket' approach to indicator taxa would be more appropriate in most cases.

There has however been a growing awareness of the role invertebrates can play in biodiversity studies (e.g. Fagan and Kareiva 1997, Hammond 1994, Miller 1993, Pyle *et al* 1981, Samways 1993, Thomas 1991) as several studies have shown (e.g. Kremen 1994, Pearson and Cassola 1992, Schikora 1994, Zulka *et al* 1997).

Indicator groups should be chosen to reflect the underlying state of the environment. In many studies the species chosen are assumed to reflect the biodiversity directly at each site. Although this may seem intuitively correct it should where possible be demonstrated not assumed (Williams and Humphries 1996). An indicator group should have strong ecological fidelity. It should correlate either positively or negatively with environmental factors, although positive correlations are easier to work with, as absences are hard to prove. The response to disturbance of the indicator group should be reflected in unrelated taxa, hence allowing extrapolation from the indicator group to the rest of the environment.

Table 1.I is a list of some important characteristics an indicator group should exhibit and is based on Brown (1991) with additional categories and annotations as suggested by Pearson (1994).

For many of the criteria upon which indicator groups should be chosen invertebrates are well suited. Invertebrates are in general taxonomically and ecologically diverse. Many are relatively sedentary and these tend to have a high ecological fidelity, and consequently are found less often far from unsuitable habitat (vertebrates, particularly the larger ones will roam over wide areas of unsuitable habitat). Linked to this is habitat specificity, which is more precise for many invertebrates than for their vertebrate counterparts (Pearson 1994, Pearson and Cassola 1992).

Table 1.I: Suitability of different invertebrates as environmental indicators.

	Lepidoptera	Diptera	Hymenoptera - not ants	Hymenoptera - Formicidae	Coleoptera	terrestrial Gastropoda	Arachnida - Araneida
taxonomically and ecologically diverse	+	+	+	+	+	+	+
high ecological fidelity	*	*	+	+	+	+	+
relatively sedentary	*	*	*	+	+	+	+
narrowly endemic / well differentiated - specialisation of each popn in a narrow habitat	*	*	*	+	+	+	+
taxonomically well known / easy to identify / stable nomenclature	+	+	+	+	+	+	+
well studied / biology and life history understood	+	*	+	+	+	+	+
popns readily surveyed / abundant & easy to find	+	+	*	*	*	*	+
damped fluctuations / always present in habitat	*	*	*	*	#	*	*
easy to obtain large random samples of spp & variation	*	*	*	+	*	#	*
functionally important in ecosystem	*	*	+	+	+	*	+
response to disturbance predictable, rapid, analysable &linear	*	*	*	*	*	*	+
associates closely with other spp & spp resources / patterns observed are reflected in other related & unrelated taxa	*	*	+	+	+	*	+
potential economic importance - attracts funding	+	*	*	*	+	*	*

KEY: # = not ideal; * = suitable; += ideal

Large samples of invertebrates can be readily collected, (see Collecting the Invertebrates, page 27) allowing good observation and quantification of trends and reducing chance that observed trends are anomalous. They also tend to have fast reproductive turnover, which increases habitat sensitivity and makes changes due to habitat disturbance apparent relatively quickly. Some groups of invertebrates are more functionally important than others and so different groups reflect the overall changes in the ecosystem better than others.

There are problems that should be taken into account before choosing an indicator group. Invertebrates (in comparison to vertebrates) often have widely fluctuating population sizes, both from year to year and from season to season, which may make correlations between times and sites difficult in some cases. In the tropics particularly (less so in Britain) (Stork 1988) there is a problem with the large proportion of unknown species compared to the number of known species (Hammond 1994, Samways 1993). Groups which are better understood in terms of biology and life history make the choice of indicator group better informed and implications of changes more biologically meaningful. Similarly well studied groups tend to have more stable nomenclature, available keys for identification and greater numbers of workers in the research field available to give expert advice.

1.10 Collecting the invertebrates

There are numerous different methods of trapping available. They vary in technique, equipment, efficiency, representativeness, time, cost, effort and composition of species that are caught. In order to sample a variety of different microhabitats within the habitat, a number of complementary trapping methods are needed, and the choice of which ones depends upon the invertebrates to be sampled. There is no one agreed 'best' method, only general principles, and different methods are most appropriate for different groups (see New 1998 for an overview). There are three main styles of sampling – attractants, lie-in-wait and active searching. Figure 1.6 shows volunteer Mary Chester-Kadwell employing an active hunting technique for molluscs.

Attractants rely on a bait of some sort to attract the animal to the trap. Examples include fruit such as bananas or oranges for butterflies (Kremen 1994); faeces or carrion for certain flies and beetles (Williams *et al* 1996); pheromones are available for many insects and can be targeted precisely at the species sought, but tend to only collect one sex; sound e.g. for crickets; light e.g. for fireflies and blacklight traps for moths (Williams *et al* 1996). In this study the range of invertebrates to be collected is too broad to allow any one attractant to be useful. It would be virtually impossible to quantify the results from such trapping methods, and they could not be compared with each other.

Lie-in-wait traps passively collect the invertebrates as they crawl or fly about their habitat. They include pitfall traps (Schikora 1994, Topping and Sutherland 1992), yellow pan traps (Runtz and Peck 1994), malaise traps (Finnamore 1994), flight intercept traps (Williams *et al* 1996), substrate traps e.g. reed nests for bees (Gathman *et al* 1994), water traps and emergence traps (Runtz and Peck 1994). Pitfall traps can be used to effectively collect ground beetles and spiders (Oliver and Beattie 1996). The main drawback of any lie-in-wait sampling method is that it will tend to measure activity rather than absolute density of species (Ottesen 1996, Rykken *et al* 1997) and results should be correlated against an absolute sampling method to give a meaningful snapshot of biodiversity (Gibson *et al* 1992). Dufrêne and Legendre (1997) point out that pitfall traps should be used to compare relative abundances of species between sites and not among species. Additionally, spiders of higher vegetation structure are under represented in pitfall traps (Zulka *et al* 1997).

Active hunting techniques include sweepnetting (Johnson 1995), vacuum sampling (Gibson *et al* 1992), leaf-litter sampling (e.g. sieving, Tullgren funnels, Berlese funnels) (Koponen 1994, Longino 1994, New 1998), vegetation beating (Coddington *et al* 1996, Dobyns 1997), canopy fogging using pyrethroids (Perfecto *et al* 1997, Stork 1988), and hand

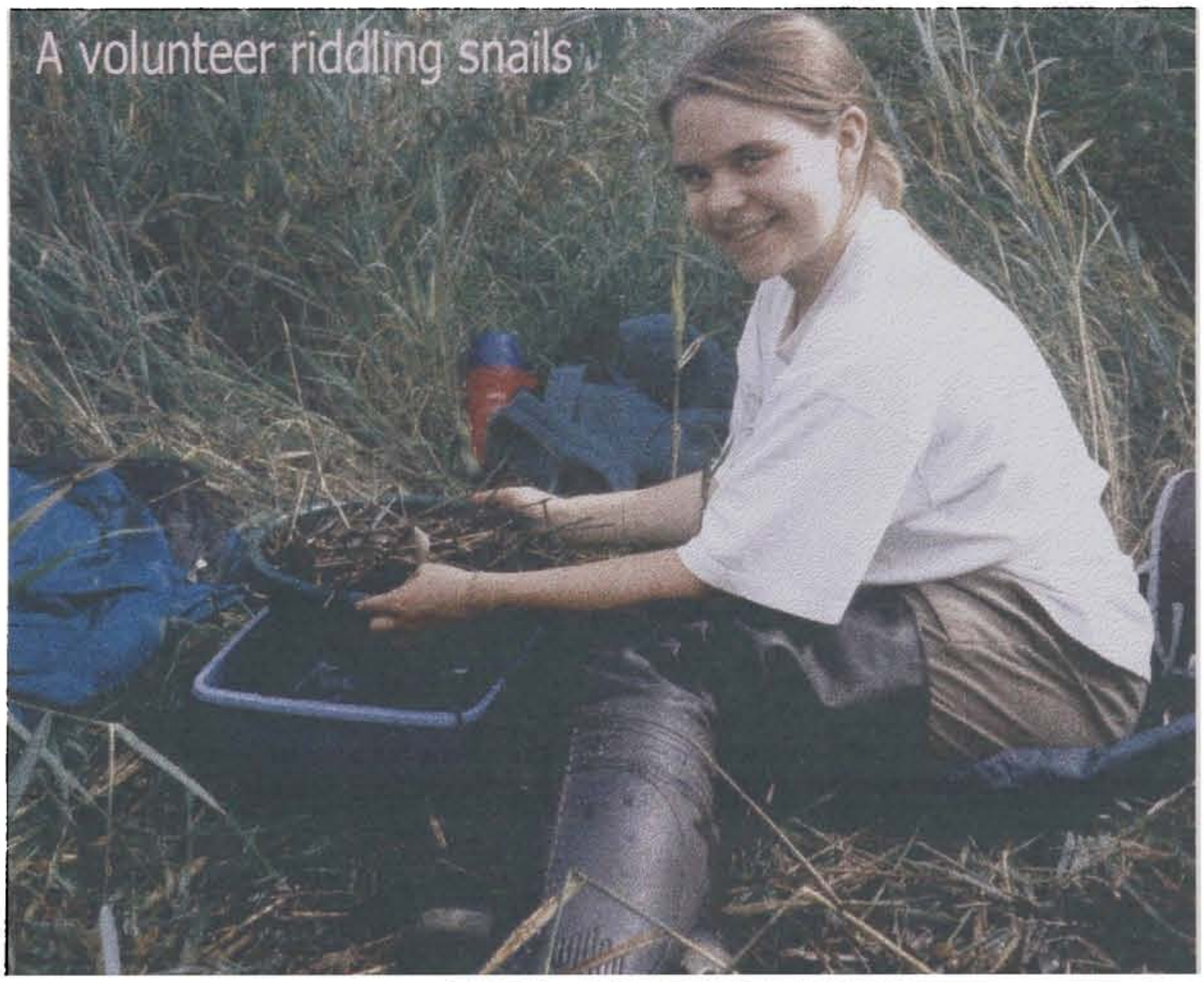

Figure 1.6 A measured amount of reed litter was shaken through a riddle into a tray. The contents of the tray was hand-searched for snails.

searching such as grubbing or pooting. Vacuum sampling gives the closest approximation to an absolute sampling method for species richness and abundance providing the entire catch is analysed (McFerran *et al* 1994, Morris and Rispin 1988), hand searching is an efficient method for collecting large terrestrial gastropods (Ditlhogo *et al* 1992) though riddling or sieving (Figure 1.6) are more effective for smaller gastropods (New 1998, D. Howlett pers. comm.).

Sampling can be done over a period of time either continuously or in concentrated bursts. Either way sampling is a cumulative process and as the number of species collected rises the accumulated total asymptotes to the absolute value. If the cumulative species number is plotted against time it is possible to predict the expected number of species in the environment from the steepness of the curve (Samu and Lövei 1995). Obviously time effort and cost rise with the completeness of the sampling and so a trade off is necessary to ensure enough data is collected to be meaningful without the collecting and identifying becoming intractable. Although absolute or continuous sampling (e.g. ATBIs – all taxon biological inventories) is the ideal, much important information can be obtained from spot sampling.

1.11 Specific Aims

The specific aims of the project were firstly to assess whether the habitats sampled could be grouped together, and to discover whether different sites could be used as markers for similar sites within the same habitat group. Further, whether different management had a stronger or weaker effect than the habitat differences, and whether any differences in effect were found between different groups of invertebrates – specifically snails, spiders and beetles.

The project further sought to discover whether summer cutting of reedbeds changed the biodiversity of these invertebrates in managed areas compared to control areas. Also investigated were species level changes within the invertebrate groups which sought to identify any specific level responses to management.