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Long-term Impacts of Environmental Change on the Soils and Vegetation of Snowdonia

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Long-term Impacts of Environmental Change on the Soils and Vegetation of Snowdonia

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Summary

Despite the large quantity of research, there is still little understanding of the long term impact of environmental drivers of change, especially when they act in combination. Understanding the long term impact is vital to enable future mitigation through conservation practices to be successful. This thesis evaluated the long term impact of environmental change drivers on soils and vegetation in Snowdonia National Park, by revisiting historic survey sites. Chapter 3 revisited soil and vegetation plots recorded as part of the International Biological Programme, Chapter 4 revisited soil sampling sites recorded as part of the Field Studies Council Soils of Snowdon study, whilst Chapters 5 and 6 utilised a long term grazing exclusion experiment set up across Snowdonia.

Overall, both the soils and vegetation in Snowdonia showed significant changes when compared to their historic condition. Although in some cases, an increase in soil pH appeared to indicate a recovery from historic acidification, it was apparent that this increase was not large enough to return the soils to their pre 1960s levels. Contrasting results were found for bryophytes and higher plants however, as higher plants indicated more acidic conditions in the present day than previously. Whilst a number of drivers of environmental change were considered, the evidence indicated that acidification, most likely from historic sulphur deposition, was the main driver of the changes identified.

Despite large reductions in sulphur deposition it is evident that the environment has yet to recover from the wide scale acidification that took place. Vegetation is already displaying a negative response to current levels of disturbance, and with climate change predicted to intensify, the resilience of already degraded ecosystems to further disturbance is uncertain, and continued field-based studies that examine the interactive effects of multiple environmental change drivers are required.

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Chapter 1

Introduction

1.1. Introduction and need for research

Global environmental change is having a profound effect on biodiversity resulting in an unprecedented rate of ecosystem alteration. Globally and within a UK terrestrial context, land use, climate change and atmospheric deposition are three of the major drivers of biodiversity change. Experimental studies often investigate individual drivers over the short term resulting in difficulties extrapolating the results to the field.

Long-term observational data sets are essential for investigating the changes that are occurring in natural and semi-natural ecosystems. The need for quality long-term datasets is becoming more urgent as the extent of environmental change gathers pace. By utilising historic data sets, changes in soil chemistry and vegetation community composition and the effects of environmental change drivers, in combination, in natural ecosystems can be investigated. To understand the effect future changes in environmental drivers may have on the environment, an understanding of the response of ecosystems to previous drivers is essential.

The Snowdonia area of North Wales has been the focus of numerous scientific studies over the years. The International Biological Programme (IBP) from 1964 to 1974 investigated the basic processes of biological productivity (Heal & Perkins, 1978), and surveyed in detail an *Agrostis-Festuca* grassland in 1968. Soil chemical analysis has been carried out as part of the Field Studies Council Soils of Snowdon publication (Ball *et al.*, 1969), and a long term experiment, started in 1957, has investigated the effect of grazing regime on soil and vegetation properties (Hughes & Dale, 1970). These studies provide detailed information on the previous condition of the environment and provide an excellent baseline to investigate the long-term impact of environmental change. Since 1995, Snowdon has become a UK Environmental Change Network (ECN) site established to identify the links between biological responses to environmental drivers, and monitors key aspects of climate, pollution and land use. The Snowdon (Yr Wyddfa) ECN site, co-sponsored by the

Countryside Council for Wales and the Welsh Assembly Government, is managed by the Countryside Council for Wales. Along with standard ECN protocols, measurements using the original meteorological equipment from the 1950s were resumed on Snowdon, to link with the historic datasets. These historic and more recent datasets provide an excellent opportunity to investigate the degree of change in the soils and vegetation of Snowdonia.

1.2. Plan of thesis

This thesis is split into 4 experimental chapters followed by a general discussion. The chapters, whilst standing alone, provide evidence for the main drivers of vegetation and soil change within Snowdonia. The overall aim of the thesis is to provide evidence of the changes which have occurred in the vegetation and soils in Snowdonia, and to identify the main driver of this change. The investigative work is presented as four separate scientific papers; therefore some unavoidable repetition of introductory material and references occurs.

The thesis begins with an introduction to the literature, covering the main drivers of environmental change and the effects these exert. Following this, the first two experimental chapters detail the findings of studies that revisited vegetation and soil plots located on Snowdon, forty years after the first studies were completed. Chapter 5 describes a revisitation study to a grazing removal experiment, set up in 1957, with plots located across Snowdonia National Park, whilst chapter 6 investigates the effect that long term grazing removal has had on soil and soil solution chemistry. Chapter 7 includes a general discussion of results from the previous experimental chapters, and addresses the main aim of the research. Conclusions are drawn and further work identified.

1.3. Aims and objectives

The aim of this research was to improve our understanding of natural and man-induced ecological change due to climate, land use and atmospheric pollution changes, and identify long term changes as opposed to short term fluctuations. To achieve this, the aims of the thesis, which are discussed in further detail in each subsequent chapter, were:

1) To identify the changes that have occurred in the vegetation in Snowdonia over the past forty years (Chapter 3)
2) To identify the changes that have occurred in soil chemistry in Snowdonia over the past forty years (Chapter 3 and 4)
3) To identify if changes in vegetation or soil chemistry driven by environmental change is mitigated through grazing removal in Snowdonia (Chapter 5 and 6)
4) To identify the possible causes of the changes identified (Chapters 3, 4, 5 and 6).

1.4. References

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Chapter 2

Literature Review

2.1 Global environmental change

Human activities have profound impacts on the planet (Field, 2001), resulting in the world and its ecosystems undergoing rapid changes (Millennium Ecosystem Assessment, 2005). Humans have altered the supply and turnover of many of the major nutrients that otherwise constrain productivity, composition and diversity of terrestrial ecosystems and as such are causing the earth to undergo rapid environmental changes (Tilman & Lehman, 2001). At the global scale it has been suggested that the five most important determinants of changes in biodiversity are: changes in land use, atmospheric CO_2 concentration, nitrogen deposition and acid rain, climate and biotic exchanges (deliberate or accidental introduction of plants and animals to ecosystems) (Sala *et al.*, 2000). Although, the Millennium Ecosystem Assessment (2005) identified habitat change, climate, invasive species, over-exploitation and pollution as the main drivers of biodiversity change. As a result of human-induced changes in the global environment (Pimm *et al.*, 1995) the global climate is changing at an unprecedented rate (IPCC, 2007b).

In response to this environmental change, biodiversity is also changing rapidly (Sala *et al.*, 2000). Strongly linked to ecosystem processes and societies use of natural resources, biodiversity changes are so large, they are now considered an important global change in their own right (Chapin *et al.*, 1997). The structure and functioning of natural and semi-natural ecosystems are threatened by many human activities, which also pose a threat to the natural variety of plant and animal species (Bobbink *et al.*, 1998).

Currently the effects of environmental change on species composition, diversity and ecosystem functioning are poorly understood. Plants may be limited by a number of variables, including nutrients, herbivores, and their physical environment including climate. Many of these limiting factors are being greatly altered by human actions resulting in large and rapid changes to plant environments (Tilman & Lehman, 2001). Whilst climate change is recognised and accepted as a major threat to global biodiversity (Sala *et al.*, 2000) other anthropogenic causes of environmental change also pose a risk (Brooker, 2006). For example, the emissions of nitrogenous atmospheric pollutants have increased markedly in the past 100 years (Bobbink, 1998), and land use, either intensification of agricultural use or abandonment, is also a significant driver (MacDonald *et al.*, 2000). Individually or in combination, these drivers may cause significant changes in both plant community or soil chemical composition.

2.2 Anthropogenic causes of environmental change

The three main drivers of environmental change within a UK terrestrial context are climate change, atmospheric deposition and land use change (Morecroft *et al.*, 2009). These drivers and their impact on biodiversity, particularly vegetation communities, are briefly reviewed below, with land use considered solely within the context of this study, in terms of grazing regime.

2.2.1 Climate change

On short timescales changes in the natural environment are principally governed by climate. The climate characterises the location and intensity of biological, physical and chemical processes (Beniston, 2005). There is mounting evidence to indicate that the global climate is changing (IPCC, 2007b). Biological responses to warming are evident, suggesting current conservation strategies will need to be revised to account for this changing climate (Hannah *et al.*, 2002a). The conservation of biological diversity is facing new challenges, and a synthesis between biogeography and conservation biology is necessary to respond to these challenges (Hannah *et al.*, 2002b). To be able to predict the influence of humans on climate and recognise how these changes affect ecosystems, an understanding of the processes that lead to climate change and the system that determines the climate on Earth is necessary (Le Treut *et al.*, 2007).

2.2.1.1 The climate system – An overview

Weather is the fluctuating state of the atmosphere and has only limited predictability. It is a result of rapidly developing and decaying weather systems. Climate, on the other hand, is the average weather in terms of its mean and variability over a certain time-span (generally 30 years) and location. Climate has greater predictability but is influenced by external forcings, such as emissions of greenhouse gases and land use changes (Le Treut *et al.*, 2007).

Most organisms, including humans, influence their environment. Since the middle of the 18th Century, humans, however, have started to exert an influence on a much larger scale (Baede *et al.*, 2001; Le Treut *et al.*, 2007). Combustion of fossil fuels, emissions of chlorofluorocarbons (CFCs), and other compounds have not only impacted on the radiative forcing of the Earth, but have also resulted in a depletion of the stratospheric ozone layer (Baede *et al.*, 2001; Le Treut *et al.*, 2007). Urbanisation and human forestry and agricultural practices, all examples of land use change, have affected the physical and biological properties of the Earth's surface, which in turn can change radiative forcing and impact on regional and global climate (Baede *et al.*, 2001; Le Treut *et al.*, 2007), although the main effect of land use changes and agricultural practices on climate change is through emissions of N_2O and CH_4 from livestock and cultivated soils (Smith *et al.*, 2007).

2.2.1.2 Detection of climate change and attribution of causes

Understanding the recent trends in both global and regional climate informs predictions of the future climate. The warming of the global climate is indisputable (IPCC, 2007b). Between 1905-2005 the earth's climate has warmed by approximately 0.74°C (IPCC, 2007b) with two main periods of warming, between 1910 and 1945 and from 1976 onwards (Walther *et al.*, 2002). The rate of warming from 1976 onwards has been double that of 1910-1945 resulting in a greater rate of warming than at any other time during the past 1000 years (IPCC, 2001). Prior to 2007, the ten warmest years on record in the UK had all occurred in the previous 13 years (Jenkins *et al.*, 2007).

The longest continuous temperature record in existence is the Central England Temperature (CET), monthly series, which began in 1659 (Manley, 1974). Data for this series is obtained from a number of stations, including Rothampsted (Hertfordshire), Pershore (Worcestershire) and Stonyhurst (Lancashire) (Jenkins *et al.*, 2007). Comparison of the monthly temperatures from the CET and other stations distributed across the UK shows high correlation, indicating the applicability of this data set extends beyond central England (Croxton *et al.*, 2006). Since the 1980s the CET has increased by about one degree Celsius, which is a far more rapid rise in temperature than that shown by the global average land-surface temperature (Jenkins *et al.*, 2007). Scotland and Northern Ireland however, have shown only a rise of approximately 0.8°C since 1980. No similar long-term data set of temperature exists for Wales, with the existing data set only running since 1914 (Jenkins *et al.*, 2007).

A number of causes, both natural and man-made are thought to be responsible for the changes in global temperature. Natural variation in the earth's climate due to volcanic eruptions and changes in the output of the sun are in part responsible (Jenkins *et al.*, 2007). However, models using these factors alone, poorly replicated the recent change in global temperature. Only when anthropogenic factors such as greenhouse gases and sulphate aerosols were added to the model, could the recent temperature rise be replicated (Stott *et al.*, 2000). The rapid rise in the temperature shown by the CET is likely due to a significant influence from man-made greenhouse gases (Jenkins *et al.*, 2007). Karoly & Stott (2006) also found a significant influence of anthropogenic factors on the recent warming of the CET. Although temperature series do exist for Scotland, Ireland and Wales, only the CET time series is long enough and robust enough to enable attribution of causes to be made. Analysis of the gridded data available for Wales, however, does show a similar trend to that of the CET (Jenkins *et al.*, 2007).

An increase in the global mean temperature is also predicted to increase the frequency of extreme weather events, such as heat waves, droughts and floods. Across the world, the duration and intensity of these events are increasing (Meehl *et al.*, 2007). Predictions from climate models suggest that under a warmer future climate, there will be dryer summers and wetter winters. Northern Europe is predicted to experience more severe wet extremes, with intense and heavy downpours interspersed with longer relatively dry periods (Meehl *et al.*, 2007). It is likely that a decrease in the daily temperature range will result in fewer frost days, directly impacting on the growing season, which is predicted to increase as the climate warms (Meehl *et al.*, 2007). Within Europe extreme events are already being observed with increased frequency, as extraordinarily high precipitation levels led to flooding in central Europe during the summer of 2002, and in 2003 many parts of Europe were affected by a heat wave (Trenberth *et al.*, 2007). Extreme weather events do occur naturally and cannot always be directly attributed to anthropogenic climate change, however, the duration of these events in recent years does suggest an association (Trenberth *et al.*, 2007). These extreme events may pose more of a threat to terrestrial ecosystems than an increase in the average global temperature (Walther *et al.*, 2002).

2.2.1.3 Climate change effects on organisms

The effects of climate change on a broad range of organisms with diverse geographical distributions are becoming evident (Walther *et al.*, 2002). Over the past few decades, several models have been developed to predict the effect of climate change on biodiversity (Willis & Bhagwat, 2009). Responses of organisms to climate change is being detected in the phenology and physiology of organisms, the range and distributions of species and the composition of and interactions within communities (Walther *et al.*, 2002).

2.2.1.3.1 Phenology

Phenology is the study of seasonal plant and animal life cycle events driven by environmental factors, particularly climate (Menzel & Fabian, 1999). It is one of the simplest approaches with which to track changes in the ecology of species in response to recent climate change (Walther *et al.*, 2002). As a result of many species being easily identifiable, many long term phenological datasets exist (Walther *et al.*, 2002). Changes in phenology can create asynchronies between trophic pairings, which may reduce individual fitness and increase the risk of population extinctions and biodiversity loss (Thackeray *et al.*, 2010). Many of the spring activities of animals and plants, such as first flowering (Parmesan & Yohe, 2003; Parmesan, 2006), first singing (Walther *et al.*, 2002) and leaf unfolding are particularly sensitive to temperature (Menzel & Fabian, 1999) and are occurring progressively earlier in the year. Phenological change may vary among species, functional groups and trophic levels however (Thackeray *et al.*, 2010). Nearly all spring phenophases correlate well with the temperature of the proceeding months, although later onset of autumnal phenophases is far more heterogeneous (Walther *et al.*, 2002). There is an indication that the growing season is being extended (Menzel & Fabian, 1999) with an additional eight days at the beginning and four in the declining phase (Myneni *et al.*, 1997). In Northern and Central Europe, a correlation between spring events, such as the start of vegetation period also correlates with the North Atlantic Oscillation (NAO) which quantifies winter climatic conditions (Ottersen *et al.*, 2001). Within the UK, the timing of spring and summer events has become earlier across a range of taxa and is widespread across different habitats (Thackeray *et al.*, 2010). Greater response to winter warming has been found in early blooming and herbaceous species than late-blooming and woody plants (Post & Stenseth, 1999). Disruption of the phenological timing of different trophic levels may have catastrophic consequences for important ecosystem services, particularly food production (Thackeray *et al.*, 2010).

2.2.1.3.2 Range shifts

It is generally accepted that climate regimes influence species' distributions with species displaying different thresholds to temperature and precipitation. As warming occurs, it is thought that these 'climatic envelopes' will shift towards the poles or towards higher altitudes (Walther *et al.*, 2002) altering plant species distributions (IPCC, 2007a). At increasing altitudes, mountains have a reduction in area, which may suggest that mountain ecosystems are likely to exhibit greater sensitivity to climate change (Theurillat & Guisan, 2001; Diaz *et al.*, 2003) and therefore they provide good systems in which to detect and assess the ecological impacts of climate change (Beniston, 2003).

A recent global assessment of the impact of climate change on mountain ecosystems suggested that mountains will experience the greatest warming (Walther *et al.*, 2005) and in the 21st century this will be two to three times greater than that experienced during the 20th century (Nogues-Bravo *et al.*, 2007). Kelly & Goulden (2008) found an increase in elevation of dominant plant species which was attributed to climate change. Mountain top species are among the first to have become extinct due to recent climate change (Parmesan, 2006). One of the first effects to occur will be a modification of the competitive relationships between plant functional types (Theurillat & Guisan, 2001), with some plants gaining a competitive advantage with increased temperatures (Zavaleta *et al.*, 2003). However, infertile grasslands may be more resistant to the effects of climate change than other more sensitive habitats (Grime *et al.*, 2008).

Species distribution models (SDMs) have been developed in the last decade to aid the investigation of the potential impacts of climate change on plant species distributions (Randin *et al.*, 2009). In Western Europe, predictions of a loss of 60% of species in European Mountain ranges may indicate a disproportionate sensitivity to climate change (Thuiller *et al.*, 2005). Similarly, a high resolution study of alpine and subalpine woody plants predicted a loss of 40-50% of plant species extinctions due to climate change (Dirnböck *et al.*, 2003). Although other studies have found a relative low rate of loss, between 2% and 5% of habitat loss, but nearly 40% of species were predicted to lose more than 90% of their suitable habitat (Guisan & Theurillat, 2000). SDMs that incorporate climate and topographical heterogeneity (such as altitude) predict higher persistence of suitable habitat (Randin *et al.*, 2009). Whilst these studies provide an indication of the possible effects of climate change on species extinctions, there is a large amount of uncertainty associated with these models. These uncertainties have been highlighted by Thomas *et al.* (2004), as predictions of extinction risk varied between methods. Although, when the fertilisation effect of increased carbon dioxide is taken into account this reduces the impact from rising temperatures (Willis & Bhagwat, 2009).

2.2.2 Atmospheric pollution

Deposition of atmospheric pollutants is widely recognised as an important driver of environmental change in semi-natural ecosystems (Van Der Wal *et al.*, 2003). Nitrogen (N) and sulphur (S) are the primary air pollutants causing acidification, and in the case of N, eutrophication.

2.2.2.1 Trends in S and N deposition

UK emissions of S peaked in the 1970s, and have since reduced dramatically, with a 91% reduction in pollutant S in the UK between 1970 and 2007 (RoTAP, 2011). Deposition of N species from the atmosphere is the main source of N in terrestrial semi-natural ecosystems. Emissions of nitrogen oxides (NO_x) and their reaction products (NH_4^+, HNO_3 and NO_3^-) and ammonia (NH_3) are responsible for N deposition (Asman *et al.*, 1998). In recent years, however, reactive N has increased globally by 120% and continues to increase every year (Galloway *et al.*, 2008). Despite UK N emissions declining slightly from their peak in the early 1990s, no reduction in N deposition has been detected (RoTAP, 2011). An increase in the anthropogenic production of reactive N and increased NH_3 emissions due to the intensification of agriculture, have rapidly increased the levels of NO_x

and NH_x emitted to the environment in the past century. This has led to widespread increases in N deposition rates, and global anthropogenic N inputs are likely to increase even further in the future (Rowe *et al.*, 2005; RoTAP, 2011). N deposition is unlikely to change between 2000 and 2010, with areas that already exceed the critical load likely to remain so, thereby failing to achieve the targets set by the Gothenburg Protocol (Rowe *et al.*, 2005). The amount of reactive N created by humans and the primary points of loss to the environments are well understood. However, there is a lack of understanding of the rate of N accumulation in environmental reservoirs, which is problematic because of the possible cascading effects of accumulated N in the environment (Galloway & Cowling, 2002).

2.2.2.2 Acidification of the environment

In the UK uplands, the recovery from acidification in both upland surface waters (Monteith & Evans, 2005) and soils (RoTAP, 2011) is clearly evident. The effects of N deposition are strongly modified by soil factors, with greater resilience being shown by sites with higher pH (Clark *et al.*, 2007). Inputs of S and N can cause reductions in soil pH with a subsequent loss of base cations (Horswill *et al.*, 2008), which can ultimately lead to shifts in plant species composition (Smart *et al.*, 2005; Maskell *et al.*, 2010; Stevens *et al.*, 2010). A decline in soil acidity by the 1980s (Billett *et al.*, 1990; Farmer, 1995; Adamson *et al.*, 1996), and 1990s (Kuylenstierna & Chadwick, 1991) had been detected, however, studies completed more recently have since found an increase in soil acidity and signs of recovery from acidification (Emmett *et al.*, 2010). Recovery appears to be more evident in less acidic mineral soils than in more acidic, organic rich soils however, (Emmett *et al.*, 2010), which is in agreement with available soil solution data (RoTAP, 2011).

Many species are unable to tolerate a reduction in soil pH with evidence suggesting increased S and N deposition are correlated with acidification and a reduction in species richness (Horswill *et al,*. 2008; Maskell *et al.* 2010; Stevens *et al.,* 2010). Vegetation differs in its response to deposition and this is dependent on habitat (Maskell *et al*., 2010). Crawley *et al.* (2005) found a rapid reduction in species richness when the pH of soil fell below 4.5. Similarly, Roem *et al.* (2002) found germination of heathland species significantly reduced below pH 5 and acidification was the most important factor in reducing species richness. Macro- and micronutrient concentrations decline with lower pH as these are the first ions to leach from the soil (Stevens *et al.,* 2010). As the number of limiting soil resources increase, niche dimension decreases resulting in a decrease in plant species numbers (Harpole & Tilman 2007).

2.2.2.3 Effect of N in the environment

The natural nutrient balance in the environment is disturbed when reactive N in any form accumulates in pools in the atmosphere, the soil or water. Detrimental environmental effects of this accumulation include eutrophication, acidification, and species composition change. The buffering capacity of many soils will prevent the effects of increased levels of reactive N in the environment being observed for a substantial amount of time (Rowe *et al*., 2005). Modelling N cycling is difficult as many processes are biologically mediated, and it is intimately linked with several other elements, including carbon and phosphorus (Rowe *et al*., 2005).

Deposition rates vary depending with topography with upland areas receiving disproportionately more N deposition compared with lowlands (Kirkham, 2001). This is in part due to high precipitation levels in the uplands, and the high ionic concentration of hill cloud (NEGTAP, 2001). The species composition of vegetation is determined by the availability of nutrients (Bobbink *et al*., 1998) and in most upland soils, N is the limiting factor for growth, and therefore these plant communities are particularly sensitive to N deposition (Lee & Caporn, 1998). Many plant species are adapted to low nutrient conditions and can only compete successfully on low nutrient soils (Bobbink *et al*., 1998). Atmospheric deposition of N has the potential to reduce plant species richness by allowing plants adapted to nutrient poor conditions to be out-competed, thereby reducing the number of plant species in a given area (Stevens *et al.*, 2004). It has been suggested that N deposition is responsible for the declining species diversity and encroachment of nitrophilious lowland species into upland areas that has been recently detected in British vegetation (Smart *et al.*, 2003; Stevens *et al.*, 2004; Smart *et al.*, 2005; Carey *et al.*, 2008). Various studies have demonstrated the significant ecological effects experimental N addition can have on a range of habitats (Power *et al.,* 1998, Roem *et al.,* 2002). Although invaluable in elucidating mechanisms of N deposition, inputs are often unrealistically high, and attempt to determine the effects of long-term inputs over the short term (Dupre *et al.,* 2010; Maskell *et al.*, 2010). Response to N deposition has been found to vary with habitat. Stevens *et al.* (2004) found a negative relationship with N deposition and species richness in acid grasslands and Maskell *et al.* (2010) found similar responses in heathlands, although no correlation has been found in calcareous grasslands (Maskell *et al.*, 2010; Van Den Berg *et al.*, 2010).

2.2.3 Grazing as a driver of ecosystem change

Grasslands are dependent on the activity of grazing animals (Watkinson & Ormerod, 2001) and the role herbivores play in mediating plant species richness is a critical issue in nature conservation and the management of biodiversity in grasslands (Olff & Ritchie, 1998). The major management activity in temperate grasslands is grazing, either for livestock production or conservation objectives (Bullock *et al.*, 2001). Free-ranging sheep grazing is an integral part of range management in the uplands of Britain (Welch & Rawes, 1964). The start of the 20th century saw a large increase in agricultural production within the UK, with a fivefold increase in sheep numbers found in the Peak District leading to degradation of the environment (Dallimer *et al.*, 2009). More recently, agricultural policy has resulted in a reduction in grazing animals, with agricultural abandonment leading to habitat degradation, particularly in agriculturally marginal mountain areas (MacDonald *et al.*, 2000) and such effects may be exacerbated by interaction with other environmental drivers of change (Van Der Wal *et al.*, 2003).

2.2.3.1 Vegetation response to grazing

Grazing can either increase or decrease spatial heterogeneity of the vegetation pattern depending on the grazing pattern (Adler *et al.*, 2001). By selectively grazing, herbivores exert direct effects on plants, by for example reducing plant dispersal potential, and this may play a large role in the ability of these vegetation communities to respond to climate change (Theurillat & Guisan, 2001). Grazing can also increase dispersal potential through seed movement by animals either internally (Pakeman & Small, 2009) or externally (Manzano & Malo, 2006). Grazing can also exert indirect effects through preferential grazing and mediating competitive interactions between species (Milne & Hartley, 2001).

As responses to management may evolve over many years, long-term studies of grazing removal are important (Hill *et al.*, 1992), but, few long term studies on the effect of grazing exist (Bullock *et al.*, 2001). The longest running experiment which started in 1955, on Moor House National Nature Reserve, provides an insight into the effects of grazing removal, although this experiment lacked any formal control plots until 1961 (Welch & Rawes, 1964). Indications from long-term grazing experiments show a rapid decrease in low growing plant species and increased occurrence of shrubs after the removal of grazing (Hill *et al.*, 1992). Intensively grazed grasslands are dominated by fast-growing plant species (Bardgett *et al.*, 1998) whilst grasslands with low grazing levels are dominated by slow-growing plants (Klumpp *et al.*, 2009). Grazing can increase grassland biodiversity, by maintaining semi-natural ecosystems with a diverse sward structure thereby supporting a wider range of species (MacDonald *et al.*, 2000) and this diversity can be lost when grazing is removed. Welch & Rawes (1964) identified large changes in the vegetation composition after only eight years of grazing removal, with a reduction in species recorded from ninety to sixty-seven species.

2.2.3.2 Below ground assemblage responses

Information on the abundance and activity of soil organisms that are responsible for organic matter decomposition and nutrient turnover is required to fully understand the dynamics and functional mechanisms driving vegetation change in hill grasslands (Bardgett *et al.*, 1997). Grazing exerts a number of indirect effects on nutrient flow through grasslands including redistribution, changes in cycling rates, increased uptake by grazed plants and increased loss of N through NH_4 volatilisation (Holland & Detling, 1990). There is also evidence to suggest that long-term changes in grazing pressure can impact on soil pH (Marrs *et al.*, 1989). Herbivores consume the majority of above-ground vegetation, which results in a rapid return of plant available nutrients to the soil surface as dung and urine (Bardgett *et al.*, 1993). Short term exclusion of sheep has been shown to increase the total abundance of fungal hyphae, and decrease the abundance of microarthropods, whilst the long term removal of sheep resulted in reductions in microbial activity (Bardgett *et al.*, 1993). Grazing can also exert effects on soil solution chemistry. Although there is currently a lack of information on the effects of grazing on soil solution in the British uplands, Helliwell *et al.* (2010) have recently shown little effect of grazing on soil solution chemistry in the Scottish uplands. Recent studies have begun to utilise plant functional trait analysis to enable predictions of grazing on vegetation communities.

2.2.3.3 Using plant traits to measure grazing effects

The search for general rules for associating species and environmental conditions has been the focus for community ecologists for decades (Lavorel & Garnier, 2002), as it is hoped that species niches provide information about the environment (Hill *et al.*, 2000). This has focussed, in recent years, more closely on the identification of easily-measured, universally applicable predictors of the ecosystem function and responsiveness to change (Diaz *et al.*, 2004). The evidence is growing to support the existence of such predictors, as single traits or sets of co-occurring traits of plants (Diaz *et al.*, 2004).

A widely used system for categorising plant traits is that of Grime's (1974) Competitor-Stress tolerator- Ruderal (C-S-R) system. It has power in its predictions and simplicity in its assumptions and is particularly proficient at balancing the two (Hunt *et al.*, 2004). Originally developed through inference it was later validated through bottom-up statistical synthesis (Grime *et al.*, 1997; Diaz *et al.*, 2004). Grime's (1974) theory suggests that species have life-history strategies that are a combination of the three plant functional types (Bullock *et al.*, 2001). C-S-R scores provide an indication of external factors that may affect plant growth, namely competition between species for resources, stress through lack of resources and disturbance, for example through grazing (Grime, 1974). Strategies have been calculated for individual plants, although it is only recently that Hunt *et al.* (2004) have produced a tool to allow calculation of the functional type of a vegetation community as a whole. Bullock *et al.* (2001), Carey *et al.* (2008) and Stevens *et al.* (2010) have all utilised this tool to investigate changes in vegetation composition as a response to environmental drivers of change. Using trait analysis increases understanding of the mechanisms of plant response to grazing and may enable predictions to be made of changes in species composition under different grazing regimes (Bullock *et al.*, 2001). This may become particularly relevant to estimate the effects of any grazing intensity changes resulting from agricultural policy changes in the future.

2.3 Revisitation studies as a way of detecting long-term environmental change

Long-term observational data sets are essential for determining the degree to which anthropogenic drivers of change are impacting on natural and semi-natural ecosystems. The need for quality long-term datasets is becoming more urgent as the extent of environmental change gathers pace (Parr *et al.*, 2003). Studies that record species occurrence and cover in permanent plots allow separation of short term fluctuations and long term trends in vegetation change, but are costly to maintain (Bakker *et al.*, 1996). The use of long-term archived biological records have been advocated as an alternative to long-term plot studies (Bennie *et al.*, 2006), and these allow observation of long term changes in vegetation (McCollin *et al.*, 2000; Ross *et al.*, 2010). ‘Revisitation’ studies can produce high quality data, and these can indicate the nature and magnitude of vegetation change over very long periods, as long as plots can be relocated with sufficient accuracy (Ross *et al.*, 2010). A lack of measurement consistency can cause problems interpreting results, therefore it is imperative that methods are kept consistent between sampling dates, allowing real environmental change to be detected rather than methodological variability (Morecroft *et al.*, 1997; Beard *et al.*, 1999; McCollin *et al.*, 2000). Linking information from long-term monitoring and revisitation studies can provide robust estimates of vegetation and environmental changes (Dodd *et al.*, 1995; Kahmen *et al.*, 2002). A number of revisitation studies have been completed (McCollin *et al.*, 2000; Kahmen *et al.*, 2002; Bennie *et al.*, 2006; Ross *et al.*, 2010) however these primarily focus exclusively on vegetation change. This makes it difficult to attribute observed change to environmental drivers, particularly where (as in the case of atmospheric deposition) the effects of these drivers are mediated through changes in soil abiotic conditions. The UK Countryside Survey (Carey *et al.*, 2008) is one of the few revisitation studies in which both soil and vegetation change have been recorded. Even this survey, however, includes soil chemical observations on only a small proportion of vegetation survey sites, spread across all UK habitat types, with soil chemical data available only from 1978. Additional, integrated soil and vegetation revisitation studies over long periods, and focused on specific habitats, should thus provide additional insight into the extent and causes of vegetation change in upland ecosystems.

2.4 Environmental Change Network Snowdon/Yr Wyddfa Study site

This thesis presents data collected from Snowdonia National Park/ Parc Cenedlaethol Eryri and Snowdon/Yr Wyddfa. This study area was chosen due to its importance for biodiversity and history of scientific monitoring and research. Monitoring programmes, such as the Environmental Change Network and the International Biological Programme have provided substantial data on key environmental change drivers offering an excellent opportunity to investigate the long-term impacts of drivers on soil and vegetation change.

2.4.1 Snowdonia National Park

Designated as a National Park in 1951, Snowdonia was the first and largest to be established in Wales. Snowdon or Yr Wyddfa, the highest peak in Wales, occurs in Snowdonia National Park. Snowdonia contains many areas of conservation importance both nationally and internationally. There are 15 Special Areas of Conservation, 5 Special Protection Areas, 3 Ramsar sites, 107 Sites of Special Scientific Interest and 21 National Nature Reserves, as well as the only Welsh UNESCO World Biosphere Area (SNPA, 2010). One of the key qualities of Snowdonia is the biodiversity, with many nationally and internationally important species of flora and fauna flourishing in the Park. However, pressures from climate change, pollution, changing farming and forestry practices and invasive species are mounting and implementing appropriate management plans are essential to maintain the biodiversity and conservation status of the area (SNPA, 2010).

2.4.2 Environmental Change Network

Due to a long history of research, Snowdon became an Environmental Change Network (ECN) site in 1995. The UK ECN was established to identify the links between biological responses to environmental drivers, and monitors key aspects of climate, pollution and land use. The ECN (ECN, 2011) aims to:

- Establish and maintain a selected network of sites within the UK from which to obtain comparable long-term datasets through the monitoring of a range of variables identified as being of major environmental importance;
- Provide for the integration and analysis of these data, so as to identify natural and man-induced environmental changes and improve understanding of the causes of change;

- Distinguish short-term fluctuations from long-term trends, and predict future changes;
- Provide for research purposes, a range of representative sites with good instrumentation and reliable environmental information.

Figure 2-1 Location of UK ECN sites (Morecroft *et al.*, 2009)

The twelve terrestrial sites (Figure 2-1) within the network represent a wide variety of habitats, management regimes, climates, and levels of air pollution. These sites were chosen due to the history of research that existed on each site, and the stability of management and ownership (Morecroft *et al.*, 2009). Standard protocols (Sykes & Lane, 1996) are used across the network, with the methodology developed in the late 1980s (Morecroft *et al.*, 2009). On Snowdon, along with the standard protocols, measurements using original meteorological equipment from the 1950s were resumed in 1995. The long-term monitoring of environmental change drivers at the site provides the context to this study.

2.4.2.1 Trends in climate

Morecroft *et al.* (2009) completed analysis of the trends of ECN variables across the network of sites for a range of physical parameters that were most likely to influence biodiversity. Annual mean air temperature have shown an increase of 1.2 °C over 15 years (lower bound 0.8 °C, upper bound 1.6 °C) for the five upland sites (Figure 2-2, a). Turner *et al.* (2007) report increases in year-round weekly measurements of 0.48 ± 0.13 °C, 1.27 ± 0.10 °C and 1.45 ± 0.11 °C in the maximum, minimum and mid-range temperatures, respectively, in relation to the mean values for 1966-77, for Snowdon for the years 1995-2006. Snowdon receives high amounts of precipitation with over 3000 mm of rain per year (Figure 2-2, b), and receives by far the greatest amount of rainfall per year across all ECN sites. Although highly variable between years, there is an upward trend in precipitation levels on Snowdon (Figure 2-2, b).

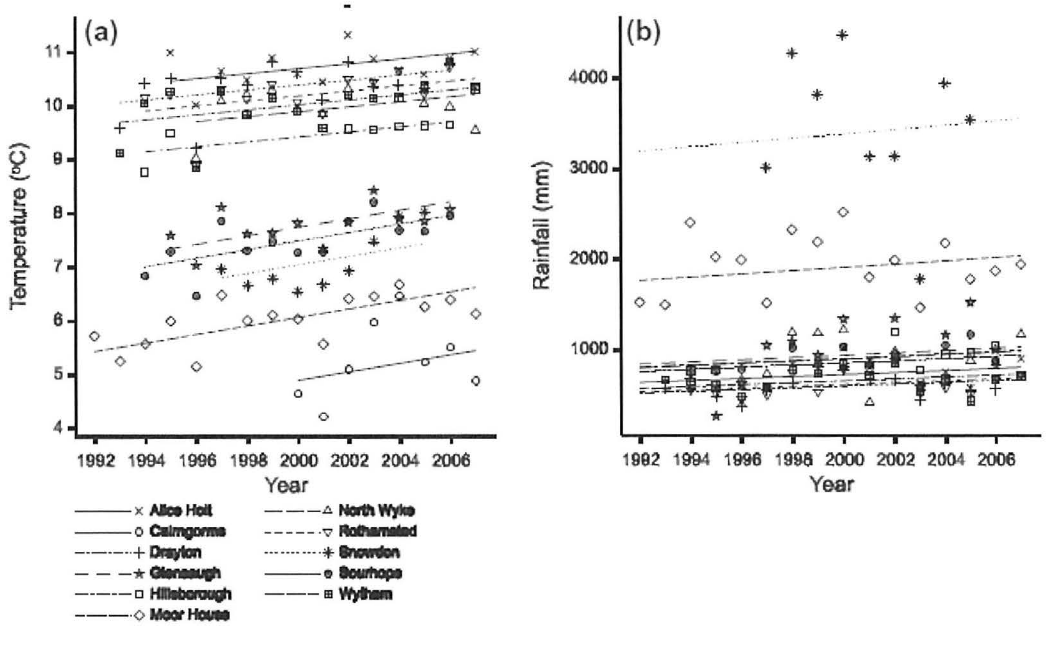

Figure 2-2 Trends in (a) annual mean temperature and (b) precipitation at different ECN sites (Morecroft *et al.*, 2009)

2.4.2.2 Trends in air pollution

Precipitation chemistry has remained relatively stable on Snowdon since 1995 with little change apparent in sulphate, nitrate or ammonium concentrations; however, a significant increase in pH and decrease in volume weighted $x\mathrm{SO}_4^{2-}$ were detected (Figure 2-3). For grassland sites, Snowdon had the highest estimated total N deposition (22 kg N $m^{-2}y^{-1}$) across the network, with the greatest proportion of total deposition made up of wet deposition of NO_3^- and NH_4^+ (Morecroft *et al.*, 2009).

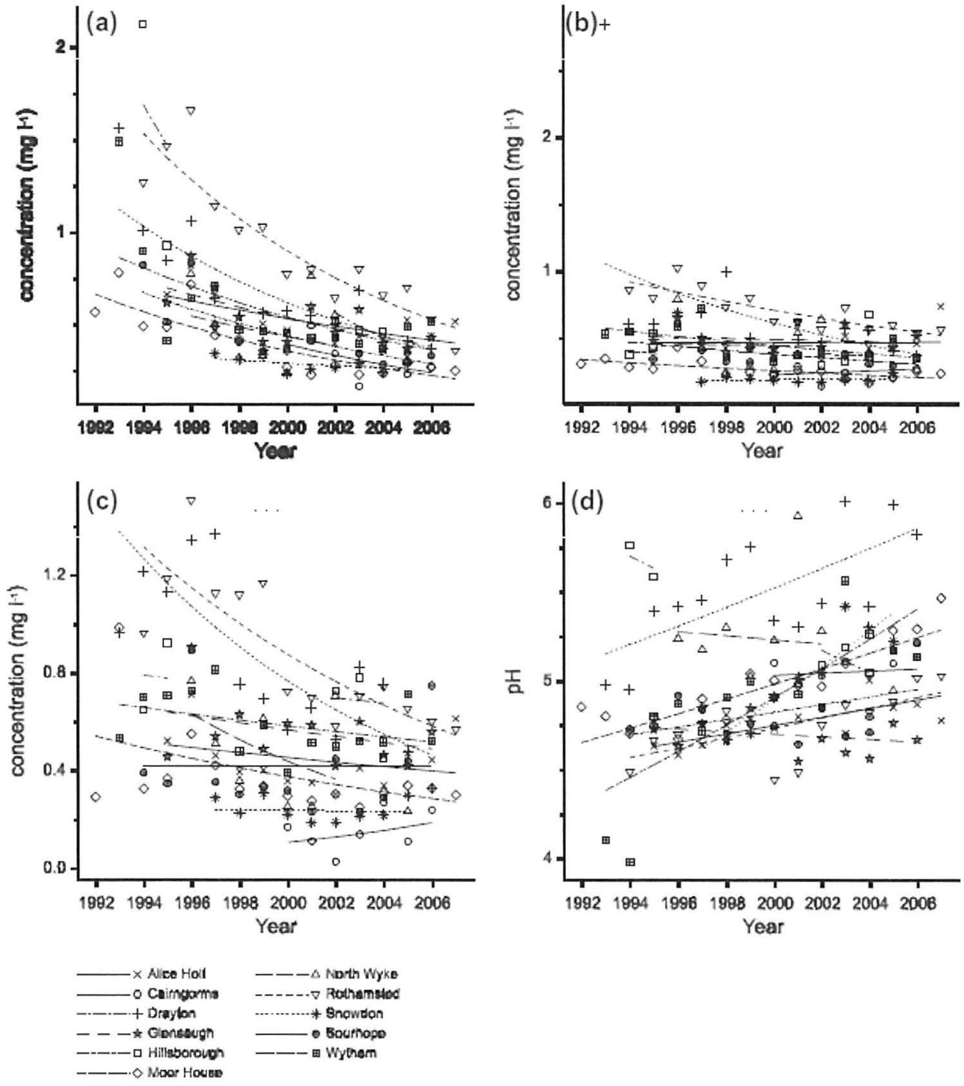

Figure 2-3 Temporal trends in precipitation chemistry at different ECN sites: (a) sulphate; (b) nitrate; (c) ammonium; (d) pH (Morecroft *et al.*, 2009)

2.4.2.3 Trends in Vegetation

Mean species richness (Figure 2-4) of vegetation within the ECN vegetation plots showed no significant change at the Snowdon site. There was a significant decrease in Ellenberg *N* indicating a decrease in species associated with high fertility habitats and an increase in Ellenberg *W* indicating an increase of species associated with wet habitats (Morecroft *et al.*, 2009). Other vegetation indices showed no significant trends on Snowdon suggesting a fairly stable plant community.

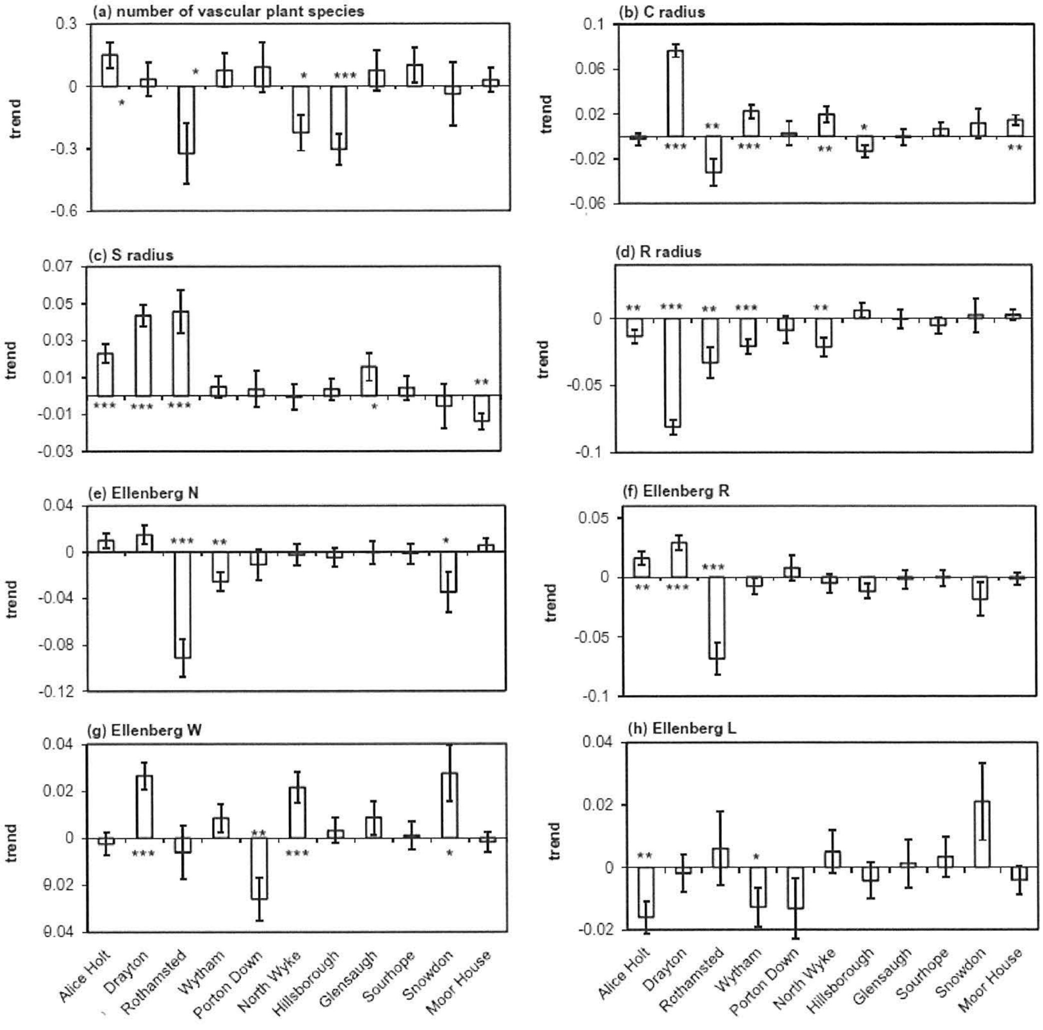

Figure 2-4. Trends in plant species richness and indices of plant functional type at ECN sites. Data are linear trends in 'fine grain' ECN plot data in the period 1993-2006, using annual and three yearly data combined. Error bars represent standard error. **p*<0.05, ***p*<0.01, ****p*<0.001 (Morecroft *et al.*, 2009)

2.4.2.4 Trends in grazing

In North West Wales major changes to the sheep grazing practices occurred around 1900 after which over wintering of wethered sheep (castrated rams) was stopped (Hughes *et al.*, 1973). Overall grazing intensity decreased as a result of these changes initially, but a steady increase was seen thereafter with an increase in sheep numbers by the mid 1970s (Hughes *et al.*, 1973). Only limited sheep grazing has occurred between October to April, and this has largely been attributed to stray ewes (Hill *et al.*, 1992). In recent years sheep numbers have decreased due to changes in the EU Common Agricultural Policy, and this pattern is seen both on Snowdon and across North Wales (Hughes *et al.*, 1973; Ministry of Agriculture, Fisheries and Food, Pre-1979; Welsh Office, 1979-1998; National Assembley for Wales, 1999-present, Turner *et al.*, 2007) (Figure 2-5).

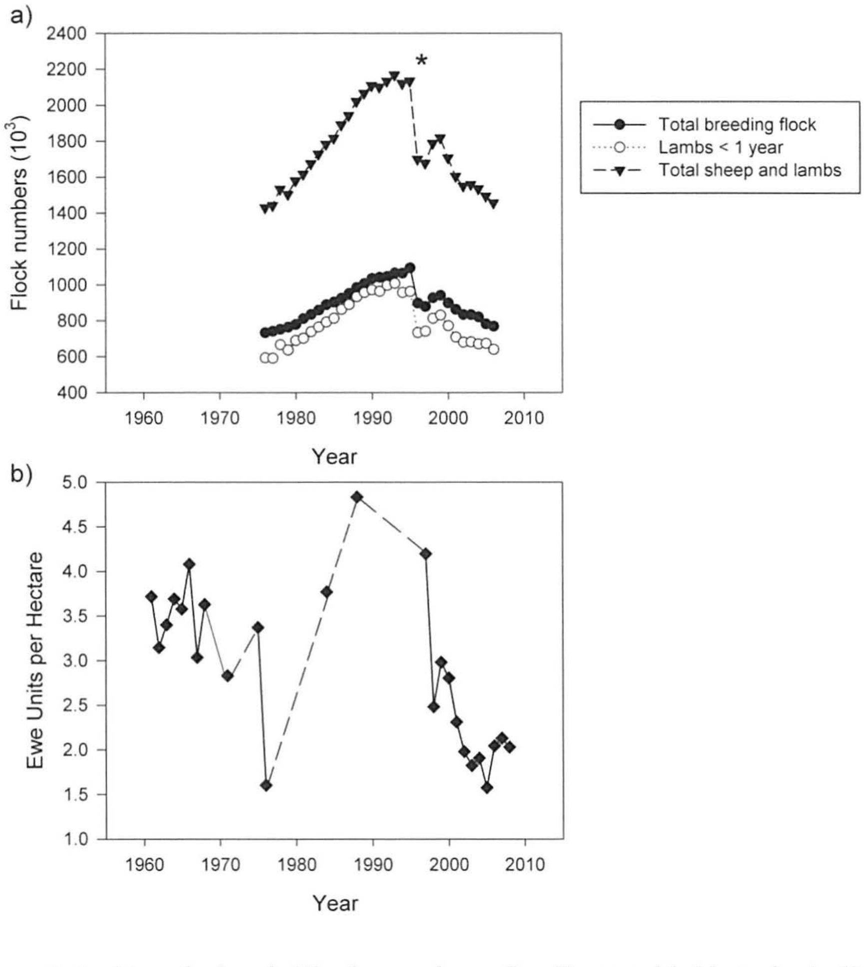

Figure 2-5 Trends in a) Flock numbers for Gwynedd (Asterix indicates year in which reporting area reduced from Gwynedd county to North Wales Unitary Area (Sources: Ministry of Agriculture, Fisheries and Food, Pre-1979; Welsh Office, 1979-1998; National Assembley for Wales, 1999-present) and b) Ewe units per hectare for plots recorded on Snowdon (Source Dale & Hughes, 1978; Turner *et al.*, 2007).

2.5 Summary

It is clear the earth is undergoing unprecedented changes in drivers of environmental change (Pimm *et al.*, 1995) and both globally (Sala *et al.*, 2000; Millennium Ecosystem Assessment, 2005), and within a terrestrial UK context (Morecroft *et al.*, 2009) the three main drivers of biodiversity change are land use, climate change and atmospheric deposition. Mountain ecosystems are highly sensitive to change (Diaz *et al.*, 2003), however, vegetation indices on Snowdon in recent years appear relatively stable (Morecroft *et al.*, 2009). Understanding the long-term effects of environmental drivers is important to allow projections of the effect of future changes in environmental drivers. 'Revisitation' studies over long periods may provide additional insight in to the extent and causes of soil and vegetation change in upland ecosystems that can inform future policy and management. This thesis presents the results from revisitation studies completed to investigate the long-term impacts of changes in environmental drivers on the soils and vegetation of Snowdonia.

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Chapter 3

Identifying drivers of species compositional change in a semi-natural upland grassland over a 40 year period

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3.1 Abstract

Question: Few long term studies exist with integrated vegetation and soil composition data, coupled with detailed environmental driver records. Can changes in community composition in an upland grassland be identified by revisitation after a forty year period and allow the main environmental drivers of the change to be identified?

Location: Snowdon, Wales, UK.

Methods: Changes in plant community and soil composition were assessed by resurveying an upland *Agrostis-Festuca* grassland in 2008, forty years after the original survey. PCA analysis and ecological indicators were used to determine changes in plant community composition. Redundancy analysis (RDA) allowed the impact of soil chemical composition on the vegetation community to be assessed.

Results: A significant shift in community composition was found between years. A 35% reduction in species richness and an increase in the grass:forb ratio, suggest significant ecosystem degradation. Indicator values suggested acidification of the community with an increased acidity preference of species recorded in 2008. However, soil pH measurements showed that soil pH had increased. RDA suggested that the main shifts in species composition were correlated with an increase in pH and a reduction in soil exchangeable base cation concentration. Clear ecosystem responses to climate, land use change or nitrogen enrichment were not observed.

Conclusions: Shifts in vegetation and soil composition are clearly identifiable after forty years. The shifts in community composition are consistent with ecosystem degradation due to acidification during the period between surveys. Ecological indicator values and soil chemical composition displayed differing degrees of change. Whilst soils appear to be recovering from historic effects of sulphur deposition, vegetation community composition changes appear to lag behind those in soil chemistry.

Nomenclature: (Stace, 1997)

Keywords: Sulphur deposition, Environmental Change Network, plant community, revisitation, soil chemistry, grazing, and climate change.

3.2 Introduction

Ecosystems are currently changing at an unprecedented rate (Pimm *et al.*, 1995). Land use, climate change and atmospheric deposition are three of the major drivers of biodiversity change globally (Sala *et al.*, 2000) and specifically within a UK terrestrial context (Morecroft *et al.*, 2009). Individually or in combination, these drivers may cause significant changes in biodiversity (Walther *et al.*, 2002) potentially resulting in altered ecosystem functions, and changing the ability of ecosystems to adapt to future environmental pressures (Chapin *et al.*, 2000).

Mountain ecosystems are highly sensitive and vulnerable to change (Diaz *et al.*, 2003) and therefore provide good systems in which to detect and assess the ecological impacts of climate change (Beniston, 2003). Over the last century, global mean surface temperature increased by 0.6 ± 0.2 ^{o}C (Houghton *et al.*, 2001), with mountain areas experiencing the greatest increases (Walther *et al.*, 2005). Warming in mountainous areas may result in a shift of communities towards higher altitudes (Walther *et al.*, 2002), and mountaintop species are among the first to become extinct due to recent climate change (Parmesan, 2006). One of the first effects to occur will be a modification of the competitive relationships between plant functional types (Theurillat & Guisan, 2001), with some plants gaining a competitive advantage with increasing temperatures (Zavaleta *et al.*, 2003). However, infertile grasslands may be more resistant to the effects of climate change than other more sensitive habitats (Grime *et al.*, 2008).

Ecosystems may respond differently to climate change, when the effects and interactions of other anthropogenic factors are taken into account (Tylianakis *et al.*, 2008). These factors may be more of a threat to grasslands than changes in climate (Grime *et al.*, 2008). Grasslands are dependent on the activity of grazing animals (Watkinson & Ormerod, 2001) and grazing can either increase or decrease spatial heterogeneity of the vegetation pattern depending on the grazing pattern (Adler *et al.*, 2001). Grazing can either reduce plant dispersal potential, through selective grazing (Theurillat & Guisan, 2001), or increase dispersal potential by seed movement by animals, either internally (Pakeman & Small, 2009) or externally (Manzano & Malo, 2006). The start of the 20th century saw a large increase in agricultural production within the UK, with a fivefold increase in sheep numbers found in the Peak District, leading to degradation of the environment (Dallimer *et al.*, 2009). More recently, agricultural policy has resulted in a reduction in grazing animals, with agricultural abandonment leading to habitat degradation, particularly in agriculturally marginal mountain areas (MacDonald *et al.*, 2000), and such effects may be exacerbated by interaction with other environmental drivers of change (Van Der Wal *et al.*, 2003).

Atmospheric deposition is widely accepted as a driver of environmental change (Van Der Wal *et al.*, 2003) and the primary air pollutants causing acidification and eutrophication are nitrogen (N) and sulphur (S). In the UK, S emissions peaked at very high levels around 1970, and have since decreased dramatically; between 1986 and 2007, pollutant S in the UK declined by 80% (RoTAP, 2011). This represents by far the greatest chemical change that has taken place in the UK uplands over this period, and has led to clear evidence of recovery from acidification in both upland surface waters (Monteith & Evans, 2005) and soils (RoTAP, 2011). On the other hand, reactive N has increased globally by 120% and continues to increase each year (Galloway *et al.*, 2008). Although UK N emissions have declined slightly from their peak in the early 1990s, as yet there has been no measured reduction in N deposition (RoTAP, 2011). Evidence is mounting to suggest that N is responsible for recent changes in British vegetation, including declining species diversity in grasslands, and the encroachment of nitrophilous lowland species into upland areas (Smart *et al.*, 2003; Stevens *et al.*, 2004; Smart *et al.*, 2005; Carey *et al.*, 2008). Deposition rates vary depending on proximity to industrial, transport and agricultural sources, and on topography, with upland areas receiving more N deposition compared with lowlands (Kirkham, 2001). Although S deposition has decreased substantially, much of this change has occurred in the last two decades, and the impact of historic deposition levels on plant species distributions may still be evident (Stevens *et al.*, 2010). It is therefore important to account for historical acidification when analysing vegetation change (Horswill *et al.*, 2008; Morecroft *et al.*, 2009).

The effects of N deposition on vegetation are strongly modified by soil factors (Maskell *et al.*, 2010). Greater resilience is shown in sites with higher pH (Clark *et al.*, 2007). Many species are unable to tolerate a reduction in soil pH so acidification due to S and N deposition can result in reduced species richness (Horswill *et al.*, 2008; Maskell *et al.*, 2010; Stevens *et al.*, 2010). Acidification can also affect the availability of macro- and micronutrient concentrations (Stevens *et al.*, 2010). As the number of limiting soil resources increase, niche dimension decreases resulting in a decrease in plant species numbers (Harpole & Tilman, 2007). Various experimental studies have demonstrated the significant ecological effects experimental N enrichment on a range of habitats (Power *et al.*, 1998; Roem *et al.*, 2002). Experimental studies involve the artificial short-term enhancement of N inputs, and the extent to which observed responses mirror those due to ambient N deposition over the longer term remains uncertain (Dupre *et al.*, 2010; Maskell *et al.*, 2010). Historical datasets can therefore provide important complementary information, although few studies exist for which both vegetation and soil measurements are available.

In recent years, a number of ecosystem ‘revisitation’ studies have been completed (McCollin *et al.*, 2000; Kahmen *et al.*, 2002; Bennie *et al.*, 2006; Ross *et al.*, 2010), but these often focus exclusively on vegetation change. This makes it difficult to attribute observed change to environmental drivers, particularly where (as in the case of atmospheric deposition) the effects of the drivers are mediated through changes in soil abiotic conditions. The UK Countryside Survey (Carey *et al.*, 2008) is one of the few revisitation studies in which both soil and vegetation change have been recorded. Even this survey, however, includes soil chemical observations on only a small subset of vegetation survey sites, spread across all UK habitat types, with the earliest soil chemical measurements made in 1978. Integrated soil and vegetation revisitation studies over long periods, and focussed on specific habitats, may thus provide additional insight into the extent and causes of vegetation change in upland ecosystems.

The Snowdon area of North Wales has the highest elevation within the southern UK. A wealth of historic and current environmental data exists for the area. The International Biological Programme (IBP) from 1964 to 1974 investigated the basic processes of biological productivity (Heal & Perkins, 1978), and surveyed in detail an *Agrostis-Festuca* grassland in 1968 (Perkins, 1978). Soil chemical analysis was also carried out on the same location (Ball, 1978), along with climatic measurements (Perkins, 1978) and assessment of land use (Brasher & Perkins, 1978). Since 1995, Snowdon has become a UK Environmental Change Network (ECN) site established to identify the links between biological responses to environmental drivers, and monitors key aspects of climate, pollution and land use (Morecroft *et al.*, 2009). Along with standard ECN protocols (Sykes & Lane, 1996), measurements using the original IBP meteorological equipment were resumed, to link with the original IBP dataset. Analysis of these data from Snowdon has revealed significant shifts in key drivers of biodiversity change. Grazing intensity has declined from 51 ±1.66 (values are expressed as mean ± s.e.m. throughout) sheep present in 1961-1968 to 33 ±2.83 sheep present in 1997- 2008. Annual mean air temperature (based on year round weekly measurements) has increased by 0.48°C ±0.13, 1.27°C ±0.10 and 1.45°C ±0.11 in the maximum, minimum and mid-range temperatures respectively in relation to the mean values for 1966-1977. Atmospheric SO_2 has shown a large decline consistent with the UK wide reductions, and NO_2 has shown an increase from an average concentration of $2.3 \mu g m^{-3}$ to $4.1 \mu g m^{-3}$ since 1997 (Turner *et al.*, 2007).

In this study, we present the results of a revisitation survey of both soil chemistry and vegetation at plots first surveyed in 1968 as part of the IBP. The aims of the study were to (i) quantify any change in the plant community and soil chemical composition that has occurred on Snowdon over a 40 year period; and (ii) identify the main drivers of these changes based on analysis of observed changes, supported by detailed environmental records for the Snowdon site.

3.3 Materials and Methods

A montane calcifugous grassland (UK National Vegetation Classification (NVC) class U4 *Agrostis-Festuca* grassland; (Rodwell, 1992)) located at Llyn Llydaw, Snowdon, Wales, UK (lat 53^{o} 05'N, long 4^{o} 02'W), 488 metres above sea level, surveyed in the summer of 1968 as part of the International Biological Programme (Perkins, 1978), formed the baseline data for this study. Snowdon has highly variable geology, soils and climatological conditions and this is reflected in the composition of vegetation communities (Perkins, 1978). Snowdon experiences high levels of precipitation throughout the year and is one of the wettest locations within the UK (Perkins, 1978). The study site is located on base-rich well drained soils (Typical Brown Earth) overlying pumice-tuffs of the Bedded Pyroclastic Series, which have a relatively high base status (Ball, 1978).

3.3.1 Vegetation Survey

The survey area, measuring 0.74 ha, was split into thirty-two plots $232m^2$ in area (Perkins, 1978). Each plot was further split into four subplots for recording purposes. Within each subplot, vegetation was recorded at three points along four transects (two horizontal and two vertical) spaced 3.05 metres apart using a grid point method (five pins (35cm x 2mm spaced 10cm apart) were recorded at each point). 60 pins were recorded in each subplot. Presence/absence of species were recorded at each pin, with presence occurring if any live part of a species was touching the pin. All plants (vascular plants and bryophytes) were identified to species level. The four subplots were combined to give a total plot value for analysis.

Original plots were relocated in the spring of 2008, using remaining plot markers and photographs. To minimise sources of error between the surveys, vegetation was recorded at the same time of the year using the original method. The original point quadrat and pins were relocated, and used in the resurvey. Percentage cover was calculated from the number of hits per pin. Where there was doubt over the original identification of some plant species (*Festuca* spp. and *Rumex* spp.) these were grouped to genus level.

To determine the impact of environmental change on the plant community, cover weighted Ellenberg values adjusted for British plants (Hill *et al.*, 1999) and C-S-R functional signatures (Hunt *et al.*, 2004) were calculated for each plot. Ellenberg values provide an indicator of a plants realised niche (Hill *et al.*, 1999) based on species preference in terms of nutrient levels (EbN), acidity (EbR), moisture (EbM) and light (EbL), while C-S-R scores provide an indication of external factors that may affect plant growth, namely competition between species for resources, stress through lack of resources and disturbance for example, through grazing (Grime, 1974). Indicator values for climatic conditions have been calculated for vascular plants (Hill *et al.*, 2004) and bryophytes (Hill *et al.*, 2007) by averaging the mean climate of the 10-km squares where they occur in the British Isles. Baseline climate summaries from the UK Climate Impacts Programme were used for the climate data for the 10-km squares (Hulme & Jenkins, 1998). Daily weather measurements from individual meteorological stations were interpolated and averaged over the period 1961 – 1990 to construct these baseline summaries (Barrow *et al.*, 1993). Cover weighted indicator values for January mean temperature (T_{Jan}), July mean temperature (T_{Jul}) and annual precipitation for plants (Hill *et al.*, 2004) and bryophytes (Hill *et al.*, 2007) were also calculated for each plot to indicate the average climatic conditions in the UK in which the plants are found.

Following the protocol of Stevens *et al.* (2010), a soil acidity preference for each plot was calculated using data from Grime & Lloyd (1973). This index was tailored to a threshold of pH 5, which is close to the point of aluminium mobilization (Grime *et al.*, 2007; Stevens *et al.*, 2010) and below which only species tolerant of aluminium toxicity are usually found (Grime *et al.*, 2007). This index allows identification of species with a higher preference for acid habitats and may indicate acidification through atmospheric deposition (Maskell *et al.*, 2010; Stevens *et al.*, 2010). Where data were not available for a species, it was excluded from this analysis.

3.3.2 Soil Survey

Soil samples were collected in summer 2009 to coincide with the original sampling timescale. 5cm diameter soil cores were taken to a depth of 15cm from each four of the subplots, and bulked per plot. Chemical analyses were carried out following the methods of Ball *et al.* (1969) to minimise error due to analytical methods currently in use. Briefly, analysis was completed on soil that passed through a 2mm sieve after air drying (35°C) and light grinding. Soil pH was measured using a Hanna instruments pH 209 pH meter in a solution of 1:2.5 soil to water ratio. Carbon was calculated from loss-on-ignition (Ball, 1964). Total N was determined using the Kjeldahl method on oven-dry soil (105°C) using a Foss 2300 Kjeltec analyzer unit. Exchangeable cations were extracted using a 1:40 soil to neutral 1M ammonium acetate ratio and analysed using a Perkin Elmer AAnlyst 400 Atomic Absorption Spectrometer. Phosphorus was extracted using a 1:40 soil to 0.5M acetic acid ratio and analysed using the molybdate blue colorimetric method.

Original data analysis revealed considerable local heterogeneity across the plots and identified five groups of plots (subsequently labelled A to E) (Perkins, 1978). Soil data from the 1969 survey were only available as an average of the initial grouped plots (A to E), so to allow comparison, data from individual plots in the 2009 survey were averaged according to original groups (Ball, 1978; Perkins, 1978).

3.3.3 Statistical analysis

To assess any change between sampling years, analysis was carried out in R (Version 2.11.0, R Development Core Team, Vienna, Austria) using paired t-tests. Data were transformed where necessary to meet assumptions of normality and homogeneity of variance.

To investigate if a shift in vegetation community composition had occurred, a Principal Components Analysis (PCA) was run in CANOCO (CANOCO, version 4.54, Plant Research International, Wageningen, the Netherlands). The relationship between vegetation and soil chemistry composition was analysed by Redundancy Analysis (RDA) in CANOCO. RDA allows direct analysis of vegetation composition in relation to specific environmental variables (Ter Braak, 1995; Ter Braak & Smilauer, 2002). For the RDA ordination analysis, individual plot vegetation data were classified into the original five groups (A to E) according to available soil data and original ordination analysis to ensure comparability between datasets. All soil variables were tested for significant contribution to the explanation of the variation in the vegetation composition data with the Monte Carlo permutation test ($P \leq 0.05$) associated with the forward selection subroutine in CANOCO.

3.4 Results

3.4.1 Community composition analysis.

A significant shift in plant composition occurred between the 1968 and 2008 survey years, with the plots clustering strongly according to year in the PCA (Figure 3-1). The first two axes explained 68.6% of the total variation of the species composition. Eight plant species showed large changes (Figure 3-1.). Of these *Agrostis capillaris* and *Nardus stricta* showed the greatest change with an average percentage cover increase of 15.41 ±0.88 and 2.67 ±0.52, respectively. In total, 15 plant species recorded in 1968 were not recorded in 2008, and an additional five plant species were recorded for the first time in 2008. The vegetation condition measures (Table 3-1) and indicator values (Table 3-2) show a significant decline in species richness ($t=15.75$, $P<0.001$), with the number of species per plot decreasing from 36.8 ±0.65 to 23.8 ±0.63 between survey years. Species richness has declined in both vascular plant and bryophyte communities (Table 3-2).

Table 3-1 Vegetation condition measures calculated from the vascular plant composition of plots sampled in 1968 and 2008. CSR Grimes Score Individual Components: C = competitor score, S = stress score, R = ruderal score. Mean plot values are displayed and values in parentheses are one standard error of the mean (*n*=32). Asterisk indicate degree of significance of difference calculated with paired t tests * $P<0.05$, ** $P<0.01$, *** $P<0.001$

Year	**1968**	**2008**	***P***
Grass:forb ratio	2.13 (0.07)	2.81 (0.12)	***
Soil acidity preference	0.45 (0.01)	0.51 (0.01)	***
Grimes score component			
C	0.30 (0.004)	0.33 (0.01)	***
S	0.38 (0.01)	0.37 (0.01)	*
R	0.33 (0.004)	0.30 (0.01)	***

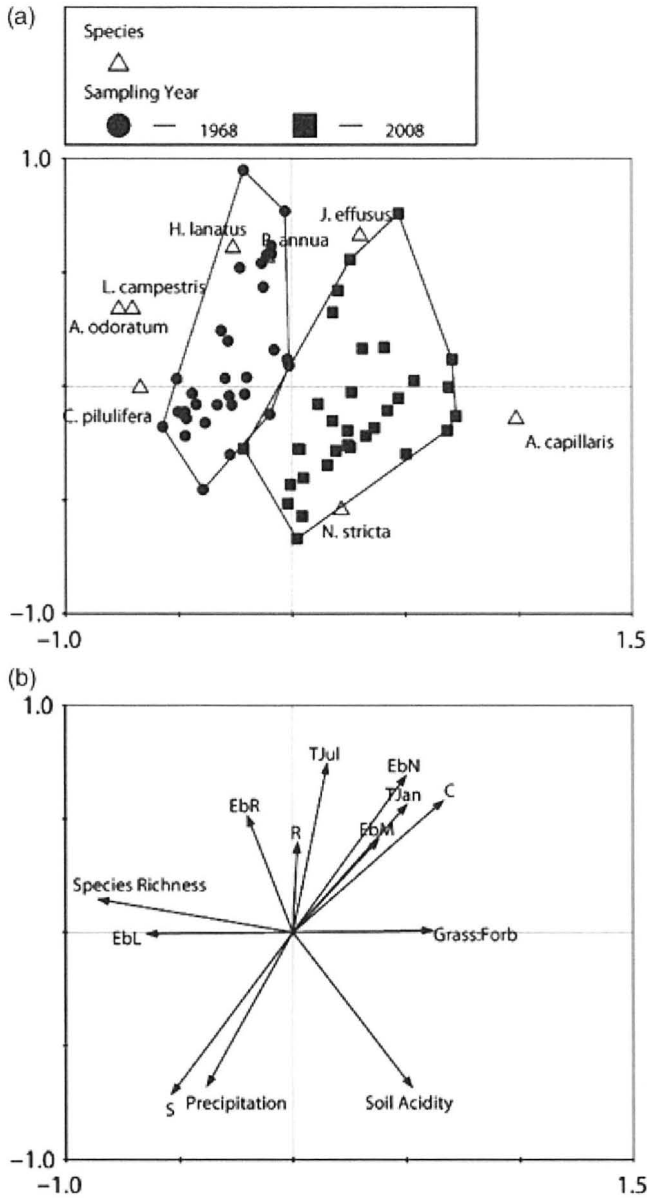

Figure 3-1. (a) PCA of vegetation community composition of the individual 32 plots to investigate variation in composition between sampling years 1968 and 2008. Displayed are the species that have shown the greatest change in percentage cover: A. capillaris=*Agrostis capillaris*, A. odoratum=*Anthoxanthum odoratum*, C. pilulifera=*Carex pilulifera*, H. lanatus=*Holcus lanatus*, J. effusus=*Juncus effusus*, L. campestris=*Luzula campestris*, N. stricta=*Nardus stricta*, and P. annua=*Poa annua*. (b) Vegetation condition measures used as supplementary environmental variables in the PCA analysis. These were calculated as weighted averages from the vegetation composition of individual 32 plots. Abbreviations as follows: EbL=Ellenberg Light, EbR=Ellenberg Reaction, EbM=Ellenberg moisture, EbN=Ellenberg Nitrogen, TJan=Temperature January, TJul=Temperature July, Precipitation=Annual Precipitation, Soil Acidity=Plant Soil Acidity Preference, Grass:Forb=Grass:Forb ratio, C-S-R=Grimes Score Individual Components: C=competitor score, S=stress score, R=ruderal score

3.4.2 Vegetation condition measures and indicator values.

There was a significant increase in the grass:forb ratio (t=8.41, $P<0.001$), from 2.13 to 2.81 between 1968 and 2008. Significant shifts in the soil acidity preference, indicates that the plant species present today are more indicative of acidic habitats than in 1968.

The functional signature of the vegetation composition did not change significantly between years and maintained a C-S-R strategy. There were significant shifts in the individual components, with a significant fall in the R-component ($P<0.001$) which has led to a corresponding gain in the C-component ($P<0.001$). There was a smaller, but still significant ($P<0.05$) decrease in the S-component. This suggests there was less disturbance with increased competition between plants for resources in 2008 than in 1968.

Results from the EbR scores and the soil acidity preference showed a high level of agreement, with species composition in 2008 for both indicators showing a higher preference for more acidic soils. EbN has shown little change between the two dates, however, suggesting no change in site nutrient status. EbL levels have decreased significantly (t=4.56 $P<0.001$) suggesting an increase in the potential canopy height. EbM has increased significantly although the same relationship was not found for the indicator value of precipitation.

Comparison of the weighted averages for the Ellenberg values for the vegetation composition split into vascular plants and bryophytes reveals some interesting differences. Vascular plants are indicating an environment which is more fertile and acidic, whereas the bryophytes indicate an environment of lower fertility and acidity. The change in the EbR value for the bryophyte community, however, was not significant. Plot values for EbL differed between plant and bryophyte responses, with plants showing a decrease in EbL suggestive of increased canopy height potential, however, bryophytes showed an increase in EbL suggesting canopy height is lower and greater light is reaching the surface.

Table 3-2 Indicator values calculated from vegetation composition surveyed in 1968 and 2008. Values were calculated on all data and then broken down into vascular plant and bryophyte constituent parts. Mean plot values are displayed and values in parentheses are one standard error of the mean (n=32). Values are weighted according to percentage cover. Asterisk indicate degree of significance of paired t test n.s. $P \geq 0.05$, * $P<0.05$, ** $P<0.01$, *** $P<0.001$

Vegetation condition measures	**Vascular plants and bryophytes**			**Vascular plants**			**Bryophytes**		
	1968	**2008**	***P***	**1968**	**2008**	***P***	**1968**	**2008**	***P***
Species Richness	36.78 (0.65)	23.75 (0.63)	***	26.00 (0.50)	16.84 (0.49)	***	10.78 (0.35)	7.19 (0.28)	***
Ellenberg Fertility	3.76 (0.04)	3.77 (0.04)	n.s.	3.80 (0.04)	3.87 (0.04)	***	3.62 (0.03)	3.16 (0.04)	***
Ellenberg Reaction	4.50 (0.02)	4.40 (0.02)	***	4.58 (0.02)	4.42 (0.02)	***	4.23 (0.05)	4.33 (0.05)	n.s.
Ellenberg Light	6.60 (0.014)	6.55 (0.01)	***	6.72 (0.02)	6.55 (0.02)	***	6.12 (0.04)	6.57 (0.03)	***
Ellenberg Moisture	5.35 (0.02)	5.40 (0.02)	*	5.40 (0.02)	5.44 (0.02)	*	5.18 (0.04)	5.14 (0.02)	n.s.
January Temperature (Tjan) (°C)	3.478 (0.002)	3.484 (0.003)	*	3.514 (0.002)	3.510 (0.003)	*	3.35 (0.004)	3.32 (0.01)	***
July Temperature (TJul) (°C)	14.413 (0.004)	14.397 (0.01)	***	14.42 (0.003)	14.42 (0.004)	n.s.	14.39 (0.01)	14.28 (0.01)	***
Annual precipitation (mm)	1122.04 (1.10)	1122.62 (1.42)	n.s.	1113.47 (0.94)	1113.88 (1.13)	n.s.	1152.37 (2.05)	1177.81 (3.82)	***

3.4.3 Soil Chemical Analysis

The chemical composition of the soils has changed significantly in nearly all variables measured. Soil pH has shown a significant increase (t=10.43, $P<0.001$) (Table 3-3) between 1969 and 2009 and is now above the point at which aluminium mobilisation is thought to occur. There have been significant declines in base cations, with K, Na, and Mn showing the largest declines. Therefore higher concentrations of soil exchangeable base cations were associated with lower pH values. There has also been a decline in soil carbon content, however this is not significant and values between groups varied more than between years. Soil N content has shown a significant decline (t= 5.7735 $P<0.01$).

Table 3-3 Chemical composition of soil samples collected as part of the IBP survey in 1969 and resurvey in 2009. Data were grouped into five groups according to those identified in the initial IBP analysis. Mean values are displayed and values in parentheses are one standard error of the mean ($n=5$). Asterisk indicate degree of significance of paired t test n.s. $P\geq 0.05$, * $P<0.05$, ** $P<0.01$, *** $P<0.001$

Year	**1969**	**2009**	**P**
pH	4.74 (0.02)	5.12 (0.04)	***
Exchangeable cations (me 100g^{-1})			
Ca	1.97 (0.05)	1.44 (0.07)	***
K	0.42 (0.04)	0.28 (0.02)	**
Na	0.49 (0.01)	0.21 (0.01)	***
Mg	0.84 (0.04)	0.72 (0.01)	*
Mn	0.23 (0.03)	0.01 (0.001)	**
Extractable P_2O_5 (mg 100g^{-1})	2.94 (0.57)	2.10 (0.84)	n.s.
C (calculated wt %)	5.22 (0.40)	4.68 (0.43)	n.s.
Total N (wt %)	0.52 (0.03)	0.40 (0.03)	**
C/N ratio	10.10 (0.22)	11.61 (0.28)	**

3.4.4 Soil and Vegetation community analysis

Within the RDA, plant communities clustered according to sampling year, when both plant communities and soil chemical variables were analysed together (Figure 3-2). The variation explained by the first two axes was 83.9%, with the main shift in vegetation groups between years along the first axis. Soil exchangeable base cation content was significant in explaining the variation in plant composition between years ($P \leq 0.05$, Monte Carlo), and P_2O_5 explained significant variation within years ($P \leq 0.05$, Monte Carlo). A lack of significance in other variables may be a result of strong correlations with variables already included in the model. On average, the amount of variation explained by each individual soil variable (excluding all other environmental variables) was 0.40 (range 0.17-0.59).

Soil exchangeable base cations and pH were associated with the first axis, with opposite orientations reflecting the divergent pattern of rising pH and falling base saturation since 1969, noted above. Unexpectedly, soil pH showed the opposite relationship to the soil acidity preference and EbR values, with a higher soil pH in 2008 contrasting with a species composition suggestive of acidic conditions (particularly among vascular plants). P_2O_5 was associated with the second axis but showed no significant change between years.

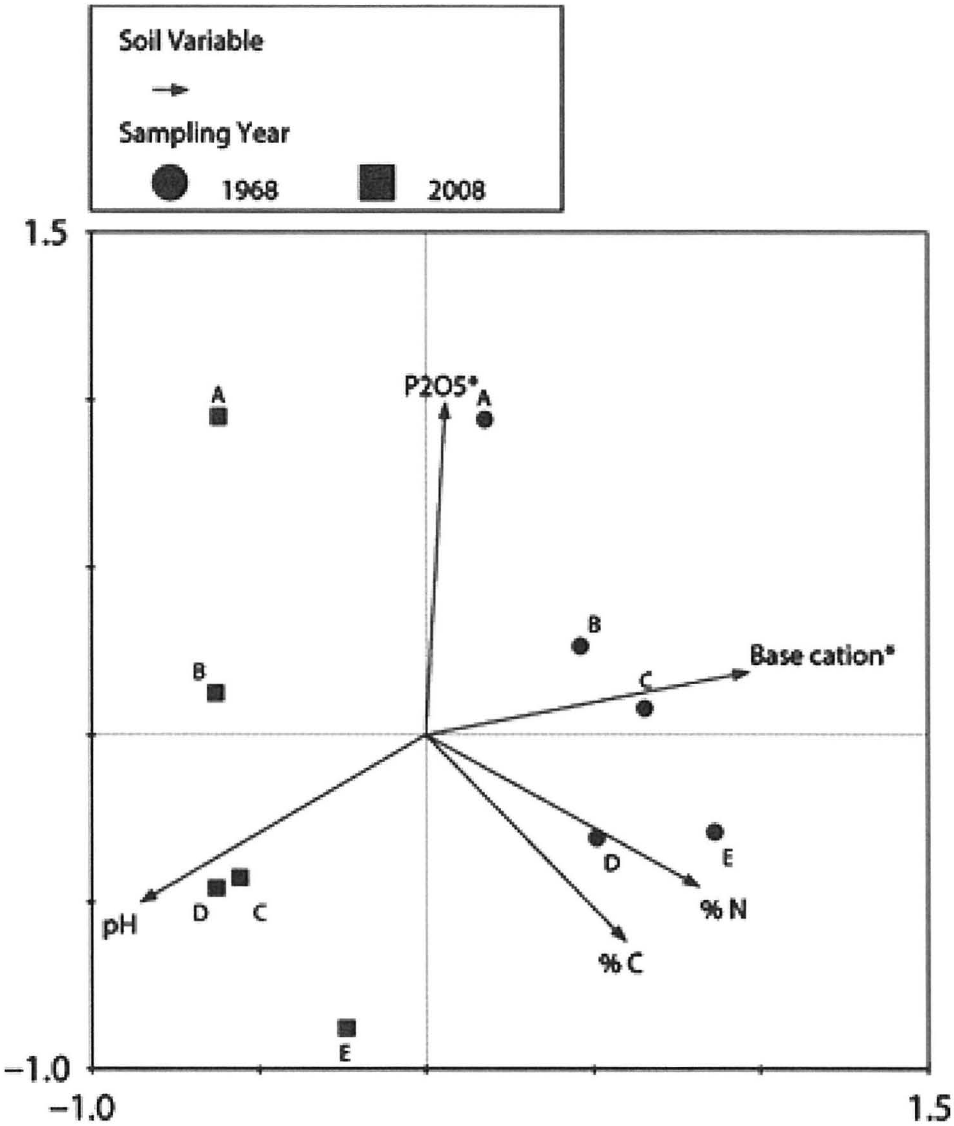

Figure 3-2 RDA of vegetation composition and accompanying soil chemical data sampled in the original IBP survey and the resurvey. Vegetation and soil data classified according to the five groups of vegetation composition (a to e) identified in the initial IBP survey. Asterisks indicate that the soil chemical variable contributed a significant amount in describing the variation in vegetation plots using the forward selection subroutine in CANOCO ($P \leq 0.05$, Monte Carlo).

3.5 Discussion

3.5.1 Degradation of the Grassland

This study has identified marked shifts in vegetation composition since the late 1960s in an *Agrostis-Festuca* grassland on Snowdon with accompanying changes in soil chemistry. There has been a considerable loss of species richness between the two surveys, with concurrent increases in grass:forb ratio. These both suggest degradation of the grassland (Carey *et al.*, 2008). No significant change in EbN, an increase in C:N ratio and a decrease in soil N content suggests that eutrophication is not the main driver of vegetation change found in this study. This is in agreement with Stevens *et al.* (2010), despite a small shift in the S-component (Hunt *et al.*, 2004). In relation to soil organic matter content, no significant loss of soil carbon was observed, consistent with Emmett *et al.* (2010) rather than Bellamy *et al.* (2005).

3.5.2 Impact of Historic Acid Deposition

Historic and current levels of acidification can have major effects on soil chemistry and have been implicated in the loss of species richness (Stevens *et al.*, 2004; Dupre *et al.*, 2010; Maskell *et al.*, 2010). Although levels of S deposition have decreased significantly, N deposition rates may still be high enough to cause nitrate leaching (and therefore acidification) and could continue to cause reductions in species richness. The rate of recovery of soils is currently incompletely established, however an increase in soil acidity was found by a number of long-term re-sampling studies first surveyed between 1950 to 1974 and resurveyed in the early 1990s (Billett *et al.*, 1990; Kuylenstierna & Chadwick, 1991; Farmer, 1995; Adamson *et al.*, 1996), but at this time the UK uplands were still strongly affected by S acidification. Since then, recovery from acidification has been marked in the surface waters draining UK upland areas (e.g. Davies *et al.*, 2005). Available soil solution data suggest a similar recovery trend in deeper mineral soil horizons, but with less evidence of change in surface organic horizons (RoTAP, 2011). Repeat soil surveys as part of the Countryside Survey showed a progressive increase in soil pH from 1978 to 1998 to 2007 in less acidic mineral soils, but again showed less evidence of change in more acidic, organic rich soils (Emmett *et al.*, 2010). The 2009 measurements are based on bulked 0-15 cm analyses of a relatively high pH mineral upland soil, and the increase in pH appears consistent with these other data. The initial IBP study coincided with the peak in UK sulphur deposition (RoTAP, 2011) and the increase in observed soil pH between 1968 and 2008 is thus probably attributable to the subsequent decline in S loadings, although it is unlikely that pH increased monotonically through the intervening 40 years. Soil acidification and recovery typically lag relative to changes in deposition, due to the capacity of soil processes to buffer changes in acid anion loadings through processes including cation exchange and sulphate retention (Galloway *et al.*, 1983).

As well as direct effects of acidification, indirect effects on vegetation may occur through interactions between soil pH and plant nutrient availability, as base cations are readily leached from acidified soils (Horswill *et al.*, 2008; Stevens *et al.*, 2009). Reductions found in base cations are consistent with other studies (Blake *et al.*, 1999; Horswill *et al.*, 2008; Stevens *et al.*, 2009). However, there is an apparent lack of recovery in soil base cations associated with the increase in pH. Although depletion of base cations and acidification of soils has been attributed to long-term inputs of N under experimental conditions (Horswill *et al.*, 2008), S rather than N has been by far the dominant driver of acidification in the UK and elsewhere to date (Curtis *et al.*, 2005; RoTAP, 2011). Dynamic modelling studies have suggested that soil pH recovery will be more rapid and complete than recovery in base saturation (e.g. Evans, 2005), due to the slow replenishment of depleted base cation reserves by soil weathering. Areas of high S deposition are associated with areas of high N deposition (NEGTAP, 2001), so attributing the acidification of semi-natural ecosystems to one or other source is difficult, although Dupré *et al.* (2010) found that cumulative

N deposition explained more of the variation in species richness than cumulative S deposition.

3.5.3 Differing responses for Bryophytes, Vascular plants and Soil

Atmospheric deposition and soil conditions respond over different time scales to changes in S emissions. Bryophytes respond directly to atmospheric inputs, and are therefore likely to show a rapid response to changing pollution levels. The increased EbR values associated with bryophytes between the 1968 and 2008 surveys are therefore consistent with the reduction in S emissions over the same period. Vascular plants, on the other hand, are more responsive to changes in soil conditions, which may lag relative to deposition by years to decades (Galloway *et al.*, 1983). Taking account also of the expected lag in vascular plant composition relative to changes in soil chemistry, it is therefore possible that the vascular plants were, in both surveys, still reflecting the impacts of previous pollutant inputs; in 1968 this would have been the cleaner conditions of the preceding decades, whereas in 2008 it may have been the more polluted conditions of the 1970s-1990s. If correct, this lag in response of vascular plants would explain the differences found in Ellenberg values between vascular plants and bryophytes. This finding also highlights the importance of assessing the ecosystem as a whole when assessing biodiversity change, and the inherent difficulties of trying to relate vegetation changes to environmental drivers over the relatively short periods associated with most monitoring activities. Appreciation of this lag in vascular plant recovery also has important implications for conservation management and monitoring on many upland sites impacted by atmospheric deposition, with bryophyte composition potentially providing a better indicator of future site condition.

3.5.4 The Role of Revisitation Studies in Environmental Monitoring

This study demonstrates the important contribution revisitation studies can have on explaining the effects of a changing environment on natural and semi-natural ecosystems. Monitoring studies provide information on temporal changes, but few truly long-term datasets exist, which is problematic for ecosystem properties, such as vegetation composition, which respond over long periods. Revisitation studies provide the potential to determine ecosystem change over very long periods. While they also present challenges in attributing observed change given the multiple and non-linear changes in environmental drivers likely to have occurred over the intervening period, the extended timescale provided by such studies has the potential, in combination with the other approaches described, to significantly aid understanding of long-term ecological response to environmental change. The Snowdon study has particular value, as it is one of the few locations where there are detailed historic measurements of soil chemistry, vegetation composition, and site specific drivers of environmental change, such as climate and grazing at a single location.

3.6 Conclusions

This study has uncovered evidence of long-term, large scale changes in semi-natural calcifugous grassland in the UK, which appear to indicate long-term ecosystem degradation. Investigating both the vegetation and soil has suggested that different components of the ecosystem respond to changes over different time scales. At the Snowdon site, observed changes appear to be largely attributable to atmospheric deposition of pollutants. Soils are likely to recover slowly, and perhaps incompletely, from the effects of long-term acidification, with implications for further changes in vegetation composition, biodiversity and ultimately ecosystem functioning. To allow conservation management strategies to be developed, consideration of all drivers of change is essential. Vegetation is already displaying a negative response to current levels of disturbance, and with climate change predicted to intensify, the resilience of already degraded ecosystems to further disturbance is uncertain, and continued field-based studies that examine the interactive effects of multiple environmental change drivers are required.

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Chapter 4

Resistance of upland soils to long term environmental changes

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4.1 Abstract

Three of the greatest global pressures facing the environment are changes in land use, pollution and climate change. These pressures are particularly acute in mountain ecosystems where they act simultaneously and at greater magnitude than at lower altitudes, and their impact on montane soils is poorly quantified. The effect of changes in land use, pollution deposition and climate change on upland soils was evaluated by revisiting sites in a mountain landscape in the UK, initially sampled forty years ago. Unexpectedly, despite the length of time between sampling dates, no significant change in pH, soil exchangeable base cations, carbon or nitrogen was found across a range of soil type and parent material. This suggests the soils have been relatively resistant to the large changes in the environmental pressures experienced in the past forty years, which include climatic warming, acidification and major changes in grazing intensity, although it does not exclude the possibility of non-linear changes during the intervening period, such as soil acidification and recovery. We therefore conclude that upland soils may be more resilient to future environmental change than has previously been suggested.

4.2 Introduction

Mountains cover approximately 25% of the global land surface and are recognized for containing ecosystems that are rich and highly diverse (Diaz *et al.*, 2003). Mountains are particularly sensitive to changed conditions (Thuiller, 2007) and as such provide good systems to detect and assess the ecological impacts of climate change (Beniston, 2003). The greatest change in land use and strongest climatic warming is currently being experienced by mountain ecosystems (Hagedorn *et al.*, 2010). However, the impact these changes may have on soils and their role in the cycling of carbon and nutrients is largely unknown (Hagedorn *et al.*, 2010).

Soil biogeochemistry can also be significantly affected by acidifying inputs of sulphur (S) and nitrogen (N) (Morecroft *et al.*, 2009) and as deposition varies with topography, upland areas receive disproportionately more acid deposition compared with lowlands (Kirkham, 2001). These inputs can cause reductions in soil pH and a subsequent loss of base cations (Horswill *et al.*, 2008), ultimately resulting in shifts in plant species composition (Smart *et al.*, 2005; Maskell *et al.*, 2010; Stevens *et al.*, 2010). S emissions in the UK peaked around 1970, followed by a major decrease; between 1986 and 2007 alone, pollutant S deposition in the UK declined by 80% (RoTAP, 2011). Emerging evidence from soil resurveys, and a small number of soil solution monitoring sites, suggests that decreasing acid deposition has resulted in an increase in soil pH across the UK (Kirby *et al.*, 2005; Morecroft *et al.*, 2009; Emmett *et al.*, 2010; Kirk *et al.*, 2010). This trend is also evident in higher-frequency long-term monitoring data from surface waters draining from upland semi-natural ecosystems (e.g. (Davies *et al.*, 2005; RoTAP, 2011).

Humans have greatly impacted the rates of supply of many of the major nutrients that constrain the productivity, composition and diversity of terrestrial ecosystems and as such are causing rapid environmental changes (Tilman & Lehman, 2001). At a global scale, the creation of reactive N has increased by 120% and continues to increase each year (Galloway *et al.*, 2008). Within the UK, acidic and calcareous grasslands are threatened by increases in N deposition (Lee & Caporn, 1998) particularly where these systems contain vegetation dependent on low soil fertility (Willems *et al.*, 1993). The fate of pollutant N within a system and its effects on N budgets is not fully understood at present. If these systems are at least partially P limited they are likely to become more readily N saturated (Phoenix *et al.*, 2004). Systems may either become readily N saturated resulting in increased leaching of pollutant N or they may accumulate much of the N (Phoenix *et al.*, 2004). In the UK, deposition over the last 20 years has remained fairly stable (RoTAP, 2011) and there is little clear evidence of progressive N saturation of terrestrial or freshwater ecosystems during this time (Curtis *et al.*, 2005; Emmett *et al.*, 2010). However, levels of N input to semi-natural ecosystems remain far above background levels and budget studies suggest that much of this N is accumulating in soils (Morecroft *et al.*, 2009). This may have considerable consequences for the recovery of the ecosystem in the future if deposition levels were to decline (Phoenix *et al.*, 2004).

Increased N deposition may also impact on other nutrients, particularly carbon (C). In most regions of the world, including the UK, soils hold the largest terrestrial C stocks (Bradley *et al.*, 2005). N deposition has been found to be positively correlated with increases in soil organic matter in both forest and moorland ecosystems (Kirby *et al.*, 2005; De Vries *et al.*, 2006; Evans *et al.*, 2006; Evans *et al.*, 2007; De Vries *et al.*, 2009) and the increased rate of accumulation is thought to occur through two mechanisms: Firstly, through an increase in plant biomass and increased litter production and secondly through a reduction in decomposition of organic matter (e.g. Waldrop *et al.*, 2004; Reay *et al.*, 2008; Janssens *et al.*, 2010). Additional C is not fixed however, if the additional N inputs are immobilised within the soil in a form which is inaccessible to plants (De Vries *et al.*, 2006) or if systems are N saturated, so that additional N is leached to surface waters.

Understanding whether soils will become long term sinks or sources of C is dependent on understanding the N cycle within semi-natural ecosystems (De Vries *et al.*, 2006). A resurvey of UK soils recorded a loss of topsoil C and attributed this to climate change (Bellamy *et al.*, 2005). However the attribution of the decrease to climate change has been questioned (Smith *et al.*, 2007). More recent analyses of these data suggest that approximately 10% of the observed changes were due to climate and the majority of soil C loss was due to changes in land use and management (Smith *et al.*, 2007; Kirk & Bellamy, 2010), possibly augmented by reducing soil acidity in the uplands (Evans *et al.*, 2007). Other studies have also suggested the greatest changes in soil C are due to changes in land use (Stevens & Wesemael, 2008). In contrast to Bellamy *et al.* (2005) other long term soil resurvey studies have found either a slight increase (Kirby *et al.*, 2005) or no significant change in soil organic matter (Emmett *et al.*, 2010; McGovern *et al.*, 2011).

The Snowdon area of North Wales has the highest elevation within the southern UK and a wealth of historic and current environmental data. Ball *et al.* (1969) provides detailed information on the distribution of soil types across Snowdon and their chemical composition in 1968. Concurrently the International Biological Programme (IBP) completed an assessment of land use (Brasher & Perkins, 1978) and climatic measurements (Perkins, 1978) in the area. Additional land use measurements were also carried out (Dale & Hughes, 1978). More recently, the area has become a site within the Environmental Change Network (ECN) with standard ECN protocols (Sykes & Lane, 1996) conducted since 1995. The ECN was established to identify the links between biological responses to environmental drivers and monitors key aspects of climate, pollution and land use (Morecroft *et al.*, 2009). Additional measurements using the original IBP meteorological equipment were resumed in 1995, to enable comparison with the original IBP data. This monitoring has revealed significant shifts in key drivers of soil change, including reductions in grazing intensity, an increase in temperature, decreases in atmospheric SO_2 and an increase in NO_2 (Turner *et al.*, 2007). A recent resurvey of soil and vegetation data from the small relatively uniform IBP site on Snowdon has shown shifts in soil composition indicative of acidification from S (McGovern *et al.,* 2011).

In this study, we present the results of a revisitation survey of soil chemistry from sites first visited in 1968 as part of the *Soils of Snowdon* study. The aims of the study were: i) to quantify any change in the soil chemical composition that has occurred on Snowdon during this 40 year period; and ii) to identify the main drivers of these changes based on analysis of observed environmental changes, using the detailed environmental records for the site.

4.3 Materials and Methods

Sample sites were relocated in autumn 2008 using original GB six figure Ordnance Survey grid references. Sampling sites were further identified by detailed location, relief, altitude and soil descriptions. To account for any error in exact site relocation, three replicates were taken at each site, spaced one metre apart. Each replicate consisted of three 5 cm diameter cores, taken to a depth of 15 cm and spaced 30 cms apart. After collection, cores were split into horizons and bulked, to provide the three individual replicates per original sampling site.

Chemical analyses were carried out following the methods of Ball *et al.* (1969) to minimise any systematic errors that might result from changes in analytical methods since the original survey was undertaken. Briefly, analysis was completed on soil that passed through a 2mm sieve after air drying (35^{o}C) and light grinding. pH was measured using a Hanna instruments pH 209 pH meter in a solution of 1:2.5 soil to water ratio. Carbon was calculated from loss-on-ignition (Ball, 1964). Total nitrogen was determined using the Kjeldahl method on oven-dry soil (105^{o}C) using a Foss 2300 Kjeltec analyzer unit. Exchangeable cations were extracted using a 1:40 soil to neutral 1M ammonium acetate ratio and analysed using a Perkin Elmer AAnlyst 400 Atomic Absorption Spectrometer. Phosphorus was extracted using a 1:40 soil to 0.5M acetic acid ratio and analysed using the molybdate blue colorimetric method.

4.3.1 Statistical analysis

To assess any change between sampling years, analysis was carried out in R (R Development Core Team, 2010) using paired t-tests. To test if underlying geology or soil type influenced soil chemical changes, the difference between the two years was calculated and analysed using an ANOVA. Data were transformed where necessary to meet assumptions of normality and homogeneity of variance.

4.4 Results

There have been only small changes in the chemical variables measured in the soils of Snowdon after forty years (Table 4-1). There has been a slight decline in pH, by an average of 0.08 units. Similarly, there has been a decrease in total soil exchangeable base cations. These results are consistent with soil responses to deposition of acidifying compounds during the intervening period but were not statistically significant. Individual responses of soil exchangeable base cations varied, however, partly as a function of the original pH value of the soil. Acid soils appear to have increased in pH and are now more alkaline than initial values; whereas the more alkaline soils have shown a decrease in pH (Figure 4-1). Similar results were found for base cation concentrations, with increases in the acid soils and decreases in the alkaline soils.

The C and N content of the soil appears to have increased, particularly in brown podzolic soils (Figure 4-2). However the carbon/nitrogen (C/N) ratio has shown little change. Some increase in extractable phosphorus was observed, but overall changes were not significant (P = 0.846). The changes suggest a small increase in soil total nutrient levels although the absence of a reduction in C/N ratio implies that this may not necessarily equate to an increase in soil fertility.

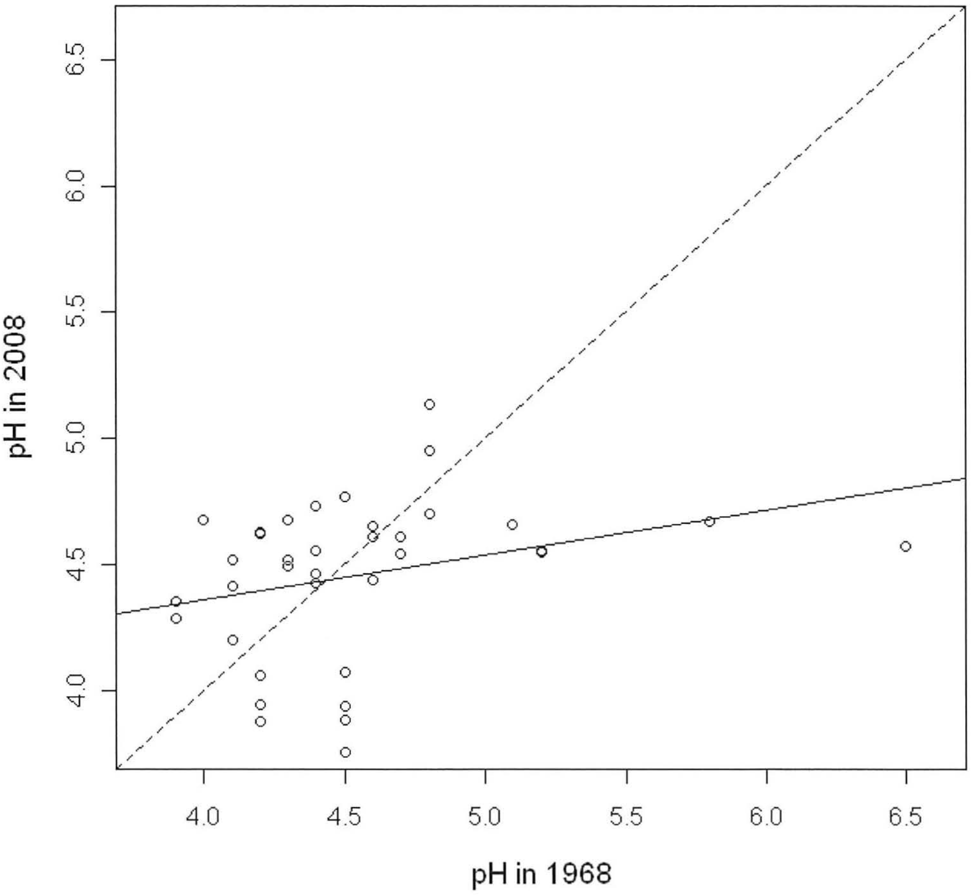

Figure 4-1 Soil pH in 1968 plotted against soil pH in 2008. Solid line represents the trend in the relationship based on a linear regression. Dashed line represents no change. *n*=36.

There was little difference in the amount of variation within each survey year for each of the soil chemical variables measured; only potassium (K) showed a large increase in variance in 2008. Because K comprises only a small part of the exchangeable base cation pool, which is dominated by calcium (Ca) (Table 4-1), total soil exchangeable base cations showed little difference in variance between years.

Table 4-1. Results of paired t-tests to determine the effect of sampling year on soil chemical variable. Mean values ± one standard error of the mean are displayed (*n*=36) ((‡) *n*=35). Significant *P* values are shown in bold.

		Exchangeable cations (me 100g^{-1})						**Extractable**	**C**	**Total N**	**C/N**
Year	**pH**	**Ca$^{(‡)}$**	**K**	**Na**	**Mg**	**Mn**	**Total**	**P_2O_5 (mg 100g^{-1})**	**(calculated wt %)**	**(wt %)$^{(‡)}$**	**Ratio$^{(‡)}$**
1968	4.54 ±	0.95 ±	0.55 ±	0.81 ±	1.37 ±	0.26 ±	3.911 ±	2.99 ±	19.75 ±	1.40 ±	13.09 ±
	0.086	2.711	0.072	0.083	0.258	0.062	0.440	0.459	2.029	0.125	0.713
2008	4.46 ±	0.83 ±	1.08 ±	0.36 ±	1.40 ±	0.03 ±	3.72 ±	3.29 ±	20.72 ±	1.60 ±	13.07 ±
	0.052	2.349	0.290	0.030	0.171	0.006	0.441	0.534	1.557	0.119	0.728
P	0.340	0.749	**0.004**	**0.000**	0.170	**0.000**	0.814	0.846	0.350	0.143	0.924

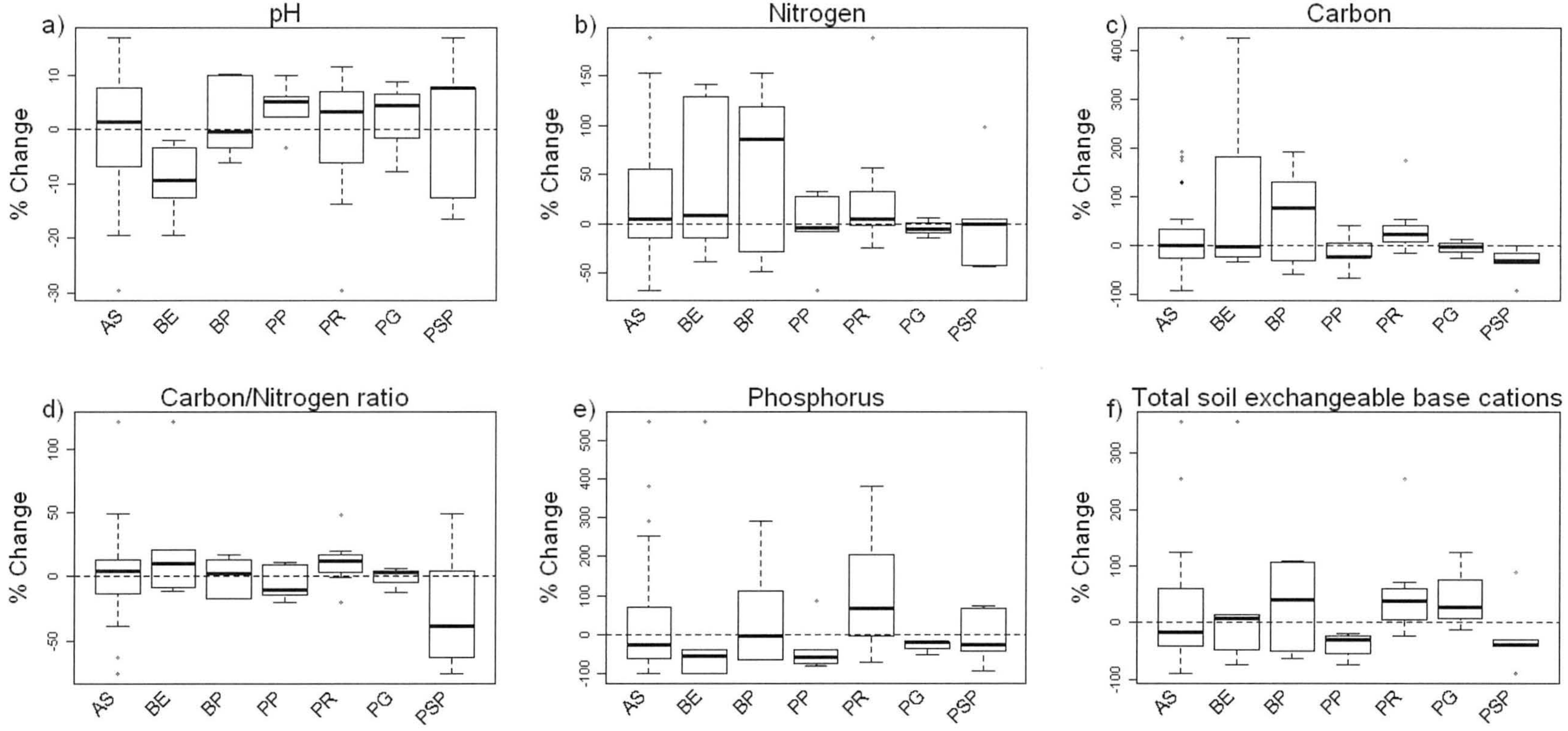

Figure 4-2 Box plots (median, 25th- and 75th- percentile values, min and max values) of the percentage change relative to 1968, by soil type of a) pH, b) Nitrogen, c) Carbon, d) Carbon/Nitrogen ratio, e) Phosphorus, and f) Total soil exchangeable base cations. Only soils with $n \geq 3$ are shown. AS = All soils n=31, BE = brown earth n= 5, BP = brown podzolic n=6, PP = peaty podzol n=5, PR = peat ranker n= 7, PG = peaty gley n= 3 and PSP = peaty soil and peat n=5. Circles represent outliers. Dashed line indicates no change.

To investigate if there were significant differences between individual soil types or soils on the same underlying geology, percentage changes in the variables measured relative to their concentration in 1968 were calculated. Groups were only included within the analysis if they contained three or more replicates. Snowdon has a highly complex geological history, and as such many geological groups were found. Few, however, contained three or more replicates and so most were discarded from the analysis.

There was large variation in the response of the soils, even within the same soil type (Figure 4-2). Although there was some variation between soil types, no significant differences were found in any of the soil chemical variables measured. Within individual soils, no significant difference between years was found.

Similar results were found for underlying geology as were found for soil type. No significant differences existed between geological groups in any soil chemical variable measured (Figure 4-3). One group however, the rhyolitic and bedded pyroclastic series, did show a significant increase in nitrogen content between 1968 and 2008 ($p = 0.02$).

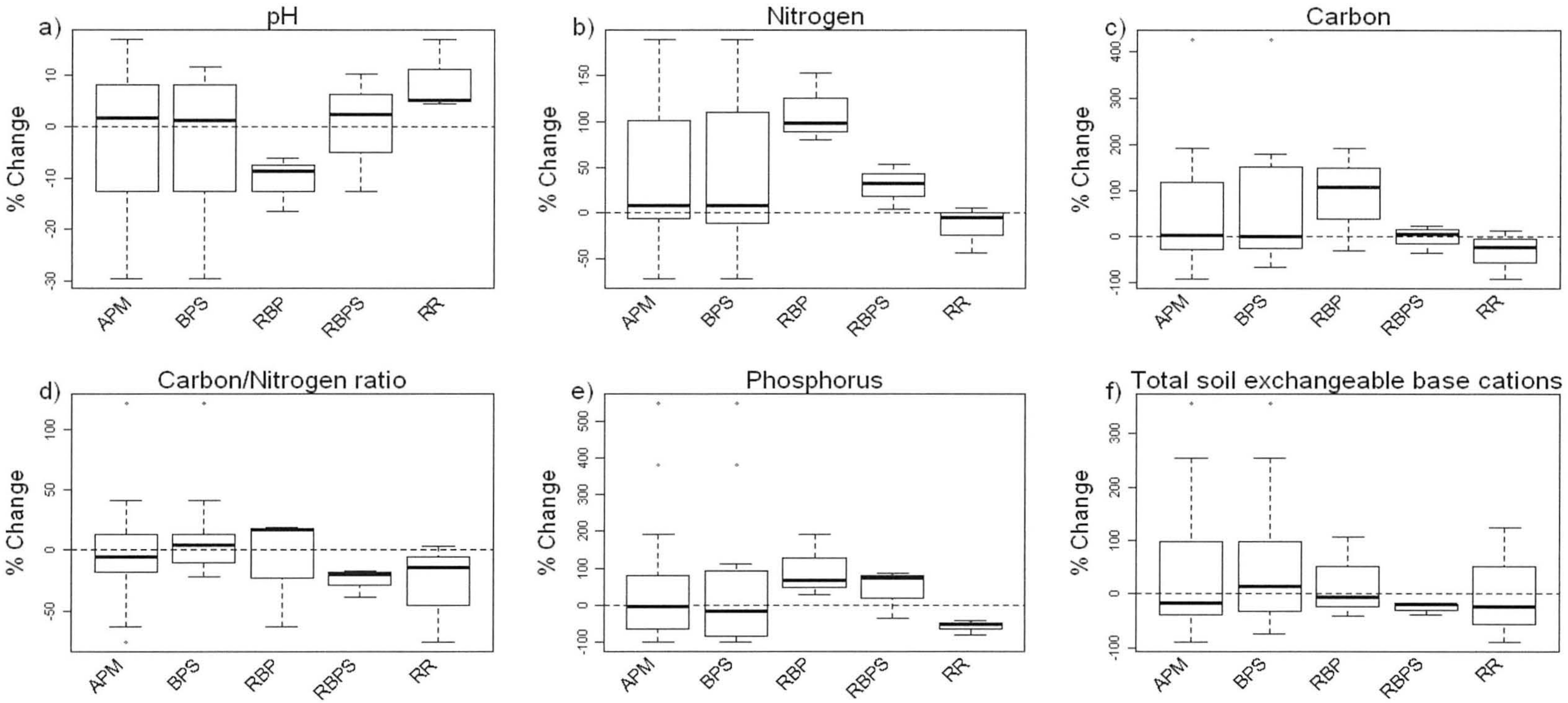

Figure 4-3 Box plots (median, 25th- and 75th- percentile values, min and max values) of the percentage change relative to 1968, by parent material of a) pH, b) Nitrogen, c) Carbon, d) Carbon/Nitrogen ratio, e) Phosphorus, and f) Total soil exchangeable base cations. Only parent materials with $n \geq 3$ are shown. APM = All parent materials *n*=20, BPS = bedded pyroclastic series *n*= 11, RBP = rhyolitic & bedded pyroclastic series *n*=3, RBPS = rhyolitic, bedded pyroclastic series & slate *n*=3 and RR = rhyolitic rocks *n*=3. Circles represent outliers. Dashed line indicates no change.

4.5 Discussion

Despite large scale changes in the environmental conditions on Snowdon, this study indicates that the soils are in a similar condition to that of forty years ago. A number of long term monitoring schemes have identified a recovery in pH of the soil across the UK and attributed this to the reductions in the deposition of acidifying compounds (Kirby *et al.*, 2005; Morecroft *et al.*, 2009; Emmett *et al.*, 2010; Kirk *et al.*, 2010). This study identified little change in soil pH, in contrast to these studies and a similar study completed on Snowdon (McGovern *et al.*, 2011). This apparent contradiction may not be as surprising as it first seems. The peak in deposition of acidifying compounds, namely S, occurred around 1970, just after the original survey took place. By the time the initial samples from the UK wide studies were collected, there had been a drop in S emissions of approximately 20% (NEGTAP, 2001). In general there is a lag between deposition and environmental effects (Monteith & Evans, 2005) and this is reflected in process models of acidification which show a delay between increasing deposition and ecosystem damage, and reducing deposition and ecosystem recovery (e.g. Hettelingh *et al.*, 2007). As the first Snowdon survey was undertaken during a period of rising acid deposition, it is likely that the full extent of soil acidification had not occurred at that time. Similarly, as deposition levels are now falling, soil pH recovery may also be lagged. Furthermore, since deposition levels remain above critical loads for acidification in much of Wales (Hall *et al.*, 2004), only a partial recovery in pH and base saturation can be expected (e.g. Evans, 2005). Our data suggest that, at best, soil pH on Snowdon has only recovered to the levels recorded in the 1960s, at which time soils in this area had already been subject to elevated sulphur deposition for over a century.

Looking more closely at the response of soil pH, it appears that the more alkaline soils show larger changes in pH. Acid soils show an increase in soil pH, suggestive of recovery from acidification, while, alkaline soils are still showing a large decrease in soil pH. Alkaline soils may have been more negatively affected by acidification and a decrease in acid deposition may not necessarily result in an increase in pH (Kirk *et al.*, 2010). The reduction in soil pH may have implications for the plant communities on Snowdon and a number of studies have found links between reductions in soil pH and a loss of species richness (Maskell *et al.*, 2010; Stevens *et al.*, 2010). The buffering capability of acid and alkaline soils will differ greatly and their ability and speed of recovery will also differ. Currently, the rate of recovery of soils from acidification is unknown, and this study highlights the considerable length of time it will take for ecosystems to recovery from the large historic perturbations to their chemical cycles.

Acid deposition affects not only soil pH but also soil exchangeable base cation concentrations. A small decrease has been found in soil exchangeable base cation concentrations although there was high variation within the soils and no clear trends emerged consistent with the results for soil pH. Yet few long term monitoring studies include soil base cation data. The extent of base cation depletion is largely determined by the type of soil and the rate of weathering of the underlying geology. Modelling studies conducted in similar locations have suggested that a recovery in soil pH will be more rapid than recovery in base saturation (Evans, 2005). A recent study on Snowdon incorporating only brown earth soils above pumice-tuffs of high base status found significant reductions in soil exchangeable base cation concentrations and an increase in pH, which suggested acidification and possible signs of recovery from S deposition (McGovern *et al.,* 2011). This underling geology was not sampled as part of the study reported here and highlights the differences in response between different soil types and underlying geology. The interesting fact that other studies have reported negative changes in base cation status within Snowdon soils may suggest that the soils sampled within this study are more resistant to acidification and subsequent base cation leaching.

As was found in the UK Countryside Survey (Emmett *et al.*, 2010), and previous Snowdon surveys (McGovern *et al.*, 2011), this study has shown no significant change in C content of the soil although there was a small increase in percentage C content. In contrast, Bellamy *et al.* (2005) found a significant loss of soil C. This major loss of soil C from the National Soil Inventory has been attributed to changes in land use and management (Smith *et al.*, 2007; Kirk & Bellamy, 2010). There has been no change in land use on Snowdon, although there have been large declines in sheep stocking density more recently (Turner *et al.*, 2007). This study sampled a range of locations, so it is not possible to ascertain the exact changes in grazing density for each site. Increasing grazing has been linked to a loss of soil C and N, as nutrient retention in high disturbance habitats is lowered (Klumpp *et al.*, 2009). Therefore the recent reduction in grazing levels may have led to the increase in soil C and N identified in this study. Alternatively, the increases in soil C and N may be as a result of increased N deposition and subsequent accumulation within the soil. Kirby *et al.* (2005) found a correlation between increases in soil organic matter and N deposition, and Morecroft *et al.* (2009) found evidence of N accumulation in soil. The results from this study are in agreement with this, although distinguishing between the two mechanisms of soil C and N increases is difficult.

4.6 Conclusion

No major changes in the soil chemical composition of the surface horizon in the variables measured were found within this study, over a 40 year period of substantial environmental change. While many studies have reported significant recent changes in the composition of UK soils, our results suggest considerable resilience to change over a longer period. This does not necessarily imply that the soils have not been negatively affected by historic environmental changes in preceding centuries, and could indeed suggest that soils have been slow to recover from previous human impacts. It is, however, difficult to draw conclusions in this respect without knowledge of the pre-industrial ‘reference conditions’. It is also possible that larger changes have taken place, and been reversed, during the interval between the two snapshot surveys. Nevertheless, it is clear that 40 years of intensive land utilisation, elevated nitrogen deposition and climatic change have not led to dramatic overall shifts in soil quality, at an upland site which would be considered relatively sensitive to such changes.

At present, much of the evidence base for the effects of a changing environment on UK soils is derived from national-level monitoring schemes. While this approach is effective at identifying large-scale trends, it can be challenging to disentangle the effects of multiple environmental drivers, operating heterogeneously across the landscape, based on the limited information available at this scale. Intensive studies of smaller areas, while lacking large-scale representation, permit a more detailed assessment of the relationship between well-quantified environmental drivers and ecosystem responses. The process insight provided by these detailed studies therefore provides valuable support to, and potentially a long-term context for, larger-scale soil monitoring schemes.

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Chapter 5

Long-term acidification and grazing effects on species composition in an upland grassland

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5.1 Abstract

Atmospheric deposition is widely recognised as an important driver of environmental change in semi-natural ecosystems. Nitrogen (N) and sulphur (S) are the primary air pollutants causing acidification, and in the case of N, eutrophication. Major changes in levels of pollutant deposition have occurred in the UK uplands and recovery from acidification is evident in upland surface waters and soils. Abiotic and biotic conditions of a grazing removal experiment in the UK uplands were investigated to identify the long-term effects of changing acid deposition and whether the removal of grazing affected the pattern of soil and vegetation change. Grazing treatment had no significant effect on soil pH, although considerable acidification of the soil was identified with little recovery to pre 1957 experimental conditions by 2009. Despite large changes in vegetation composition between grazing treatments, no significant effect of grazing on Ellenberg nitrogen and reaction values or species richness was identified. Species richness and Ellenberg reaction values displayed a gradual decline beginning at the peak of sulphur deposition in 1970 through to 2009, but little change in Ellenberg nitrogen values was found. This paper provides evidence that historic S deposition is still negatively affecting both vegetation and soils and suggests that this legacy needs to be taken into account when considering the future effects of other environmental drivers. This study also illustrates the value of preserving both historical datasets and maintaining grazing exclosures for future use. Without such resources, studies that provide important insights into long-term environmental change would be impossible.

5.2 Introduction

The rates of supply of many of the major nutrients that constrain the productivity, composition and diversity of terrestrial ecosystems have been greatly impacted by humans, and this is resulting in rapid environmental changes (Tilman & Lehman, 2001). Land use changes and atmospheric deposition are two of the major drivers of biodiversity change globally (Sala *et al.*, 2000) and particularly within a UK terrestrial context (Morecroft *et al.*, 2009). These drivers may cause significant changes in biodiversity and alter the ability of the ecosystem to adapt to subsequent environmental change (Chapin *et al.*, 2000).

Atmospheric deposition is widely recognised as an important driver of environmental change in semi-natural ecosystems (Van Der Wal *et al.*, 2003). Nitrogen (N) and sulphur (S) are the primary air pollutants causing acidification, and in the case of N, eutrophication. Long-term acidification can lead to base cation depletion (Horswill *et al.*, 2008), and can ultimately lead to shifts in plant species composition (Smart *et al.*, 2005; Maskell *et al.*, 2010; Stevens *et al.*, 2010). UK emission of S peaked in the 1970s, and have since reduced dramatically, with a 91% reduction in pollutant S in the UK between 1970 and 2007 (RoTAP, 2011). This major change in the upland chemical environment, over a 40 year period, has led to recovery from acidification in both upland surface waters (Monteith & Evans, 2005) and soils (RoTAP, 2011). Many species are unable to tolerate reductions in soil pH (Horswill *et al.*, 2008; Maskell *et al.*, 2010; Stevens *et al.*, 2010), and as the number of limiting soil resources increase, a decrease in niche dimension results in reductions in plant species numbers (Harpole & Tilman, 2007). A decline in soil acidity by the 1980s had been detected (Billett *et al.*, 1990; Farmer, 1995; Adamson *et al.*, 1996), however, studies completed more recently have since found an increase in soil pH and signs of recovery from acidification (Emmett *et al.*, 2010; McGovern *et al.*, 2011). Recovery appears to be more evident in less acidic mineral soils than in more acidic, organic rich soils however, (Emmett *et al.*, 2010; McGovern *et al.*, 2011), which is in agreement with available soil solution data (RoTAP, 2011).

On the other hand, N emissions continued to rise until the 1990s, and have only declined slightly since that time, with no clear reduction in N deposition yet detected (RoTAP, 2011). As a limiting nutrient in most temperate terrestrial ecosystems, N deposition impacts on biological systems in many ways (Bobbink *et al.*, 1998) and it is thought to be responsible for the declining species diversity and encroachment of nitrophilous lowland species into upland areas that have been recently detected in British vegetation (Smart *et al.*, 2003; Stevens *et al.*, 2004; Smart *et al.*, 2005; Carey *et al.*, 2008).

The major management activity in temperate grasslands is grazing, either for livestock production or conservation objectives (Bullock *et al.*, 2001). Free ranging sheep grazing is an integral part of range management in the uplands of Britain (Welch & Rawes, 1964). As responses to management may evolve over many years, long-term studies of grazing removal are important (Hill *et al.*, 1992), but, few long term studies on the effect of grazing exist (Bullock *et al.*, 2001). The longest running experiment which started in 1955, on Moor House National Nature Reserve, provides an insight into the effects of grazing removal (Welch & Rawes, 1964). Indications from long-term grazing experiments show a rapid decrease in low growing plant species and increased occurrence of shrubs after the removal of grazing (Hill *et al.*, 1992). There is also evidence to suggest that long-term changes in grazing pressure can impact on soil pH (Marrs *et al.*, 1989). Grazing can increase biodiversity, by maintaining semi-natural ecosystems with a diverse sward structure thereby supporting a wider range of species (MacDonald *et al.*, 2000), although interactions with atmospheric deposition may alter this response.

Long-term datasets provide the opportunity to investigate the impact which anthropogenic drivers of change have on natural and semi-natural ecosystems (McGovern *et al.*, 2011). Studies that record species occurrence and cover in permanent plots allow separation of short term fluctuations and long term trends in vegetation change, but are costly to maintain (Bakker *et al.*, 1996). The use of long-term archived biological records have been advocated as an alternative to long-term plot studies (Bennie *et al.*, 2006), and these allow observation of long term changes in vegetation (McCollin *et al.*, 2000; Ross *et al.*, 2010). 'Revisitation' studies can produce high quality data, and these can indicate the nature and magnitude of vegetation change over very long periods, as long as plots can be relocated with sufficient accuracy (Ross *et al.*, 2010). Although a number of revisitation studies have been completed, it is difficult to attribute observed changes to environmental drivers as these often focus exclusively on vegetation (McCollin *et al.*, 2000; Kahmen *et al.*, 2002; Bennie *et al.*, 2006; Ross *et al.*, 2010b). This is particularly true where the effects of the drivers (such as atmospheric deposition) are mediated through changes in soil abiotic conditions. One of the few revisitation studies in which both soil and vegetation data has been recorded is the UK Countryside Survey (Carey *et al.*, 2008). However, this is only true for a small proportion of vegetation survey sites and soil data are only available since 1978. A recent study on Snowdon, examining both soil and vegetation data over a forty year period, has indicated a lag between soil and vegetation recovery from S deposition, suggesting a long term ecological legacy from historically high deposition levels (McGovern *et al.*, 2011).

To assess the impact that grazing and atmospheric deposition have on soils and vegetation a revisitation study was undertaken of a long term grazing removal experiment. We evaluate whether the long-term trajectory of changing acid deposition is reflected in measured abiotic and biotic conditions at the site, and whether the removal of grazing has affected the pattern of soil and vegetation change.

5.3 Material and Methods

A large scale grazing experiment, established in 1957 across Snowdonia National Park provided the baseline data for this study (Hughes & Dale, 1970). Snowdon is highly variable in its geology, soils and climate (Perkins, 1978) and as such the sites were established on a variety of soils, ranging from a peaty podzol through to a brown earth (Hill *et al.*, 1992) which supported between 2.5 to 5.0 sheep ha^{-1} respectively (Hughes & Dale, 1970). Sites were located between 350 and 450 metres above sea level. A replicated exclusion experiment was set up at each site, with twelve 2.8m x 3.7m plots. These were arranged in a randomised block experimental design, with three treatments and four blocks. The treatments were as follows:

- Treatment 1: Normal grazing - Sheep allowed free access, most of the sheep are removed from the mountain in winter
- Treatment 2: Summer only grazing - Sheep excluded in winter from October to April
- Treatment 3: No grazing - Sheep excluded throughout the year.

This experimental design was ended in 1981, but all treatments were then ring-fenced, thus converting the summer-only and year-round grazing plots to ungrazed. Of the original experimental design (Hughes & Dale, 1970) only six of the original nine sites were used in the current investigation, due to uncertainty of fencing integrity throughout the intervening period for the other three sites. The six sites used within this study were located in three areas, Llyn Llydaw, Pen-y-Pass and Cwm Idwal.

The original experiment investigated the effect of three grazing treatments on vegetation composition and soil chemistry. Rapid changes in vegetation composition were recorded within the first seven years of experimental grazing removal, although no difference in vegetation composition was found between ungrazed and summer only grazed plots (Hill *et al.*, 1992). After the early rapid changes, *Calluna vulgaris* and *Erica cinerea* achieved dominance within ungrazed plots, and maintained this dominance for many years, whilst grass species remained most abundant in grazed plots (Hill *et al.*, 1992).

5.3.1 Vegetation Sampling

The vegetation was resurveyed in the summer of 2009. Each replicate of each treatment was relocated within the fence boundary by using old fence posts. To allow comparison with the existing data, the original method of vegetation recording was used within this study. Species cover was recorded using 100 randomly placed pins (35 cm long x 2 mm diameter), in each of the replicates of all treatments. The individual points were located by coordinates given from tables of random numbers. Presence/absence of species were recorded at each pin, with presence occurring if any live part of a species was touching the pin. All plants (vascular plants and bryophytes) were identified to species level.

To determine the effect of grazing on vegetation composition, unweighted percentage cover for the four components of the vegetation (cryptogams, forbs, graminoids, shrubs and trees) were calculated. The vegetation was split into these four components to allow any shift between graminoid dominance to shrub dominance to be detected. To determine the impact of environmental change on the plant community, species richness, and abundance weighted Ellenberg values for Reaction (EbR, considered a measure of acidity preference) and Nitrogen (EbN, considered a measure of fertility preference) adjusted for British plants (Hill *et al.*, 2004) and bryophytes (Hill *et al.*, 2007) were calculated for each replicate plot. Values were calculated for the whole community and separately for vascular plants and bryophytes to investigate the impact of environmental change on these components of the vegetation. The vegetation community was only split into these two categories to allow comparison between vascular plants that rely on soil conditions and bryophytes that rely more on atmospheric conditions.

5.3.2 Soil Sampling

Original soil samples were collected from each plot in 1957, 1968 and 1974. Samples were analysed for pH and carbon (C) in each sampling year and for N in 1974 only. Soil cores of five cm diameter were taken to a depth of 30cm from each replicate, split by horizon and bulked by treatment following completion of vegetation sampling in 2009 and analysed using the methods of Ball *et al.* (1969). Soil pH was measured using a Hanna instruments pH 209 pH meter in a solution of 1:2.5 soil to water ratio. Carbon was calculated from loss-on-ignition (Ball, 1964). Total N was determined using the Kjeldahl method on oven-dry soil (105°C) using a Foss 2300 Kjeltec analyzer unit.

5.3.3 Statistical Analysis

To investigate the effect of grazing treatment on species richness, EbR and EbN, ANOVA analysis (accounting for repeated measures) were run in R, Version 2.11.0 (R Development Core Team, 2010) [R code used for analysis, model<-aov(x~treatment+Error(site/treatment))]. Only data pre-1981, whilst the original design remained intact, were included within the analysis. Data were transformed prior to analysis where necessary to meet assumptions of normality and homogeneity of variance.

The Spearman's rank correlation coefficient was used to investigate the relationship between soil pH, EbR, and species richness to identify any indication of acidification within the vegetation community, and whether this correlated to soil conditions.

5.4 Results

5.4.1 Soil chemistry changes

Although soil pH varied between sites, all sites displayed a trend of acidification from 1957 to 1974 (Figure 5-1, a). Compared to the 1974 measurements, no recovery in pH was evident by 2009 (Figure 5-1, a). Although two sites are located in each locality, soil pH was not simply related to location. Underlying soil type or geology presumably exerts a stronger effect, and in this region show high local heterogeneity. C content of the soil appeared relatively stable throughout the experimental period with little change being detected (Figure 5-1, b). The percentage content of N in the soil appears to have declined from 1.25 ± 0.25 in 1974 to 0.80 ± 0.21 in 2009. This represents a significant ($t=6.60$, $P<0.001$) decline across all sites. There was no significant effect of grazing treatment on measured soil chemistry.

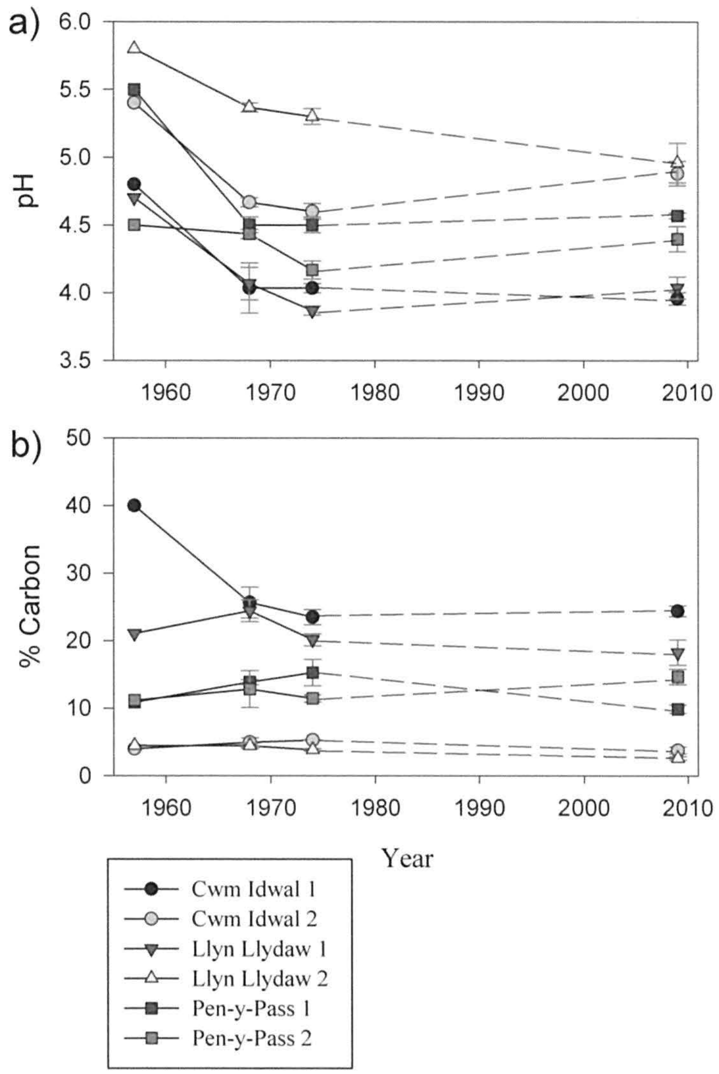

Figure 5-1 Changes in the soil characteristics a) pH and b) % carbon content for each site in Snowdonia National Park over time. Points represent the average across grazing treatments (*n*=3), error bars represent ± 1 s.e.m. The 1957 pre-treatment values are averages for the site (*n*=1).

5.4.2 Vegetation changes as a function of grazing

Large changes in vegetation composition were identified over time, and in relation to the different grazing treatments (Figure 5-2). The ungrazed sites underwent major successional changes, with an increase in shrubs and trees compared to the plots that were grazed prior to 1981. There was also a large decline in graminoid plant cover in the ungrazed sites over the same period. The 'Llyn Llydaw 2' and 'Cwm Idwal 2' sites were the exceptions to this as they have remained dominated by graminoid plants for the duration of the experiment.

Species richness did not respond as strongly as might be expected to the dramatic changes in the community composition due to grazing manipulation (Figure 5-3). Only the species richness of the bryophyte community pre-1981 appeared to be affected by grazing ($F = 8.39$, $P=0.007$), with the ungrazed treatment containing fewer species. No other variable showed a significant difference between the grazing treatments. Greater variation was found between sites in different locations than between grazing treatment so the grazing treatment was disregarded in any further analysis.

5.4.1 Vegetation changes over time

Shifts in the vegetation composition of the grazed plots after 1981 were found to resemble the initial shifts in composition at the start of the experiment in 1957. Species richness (Figure 5-3) appeared fairly stable throughout the 1960s, although bryophyte species richness was more variable than vascular plant species richness. Total species richness dipped just prior to 1970, but appeared to have recovered by the 1980s. Across all plots, there was a small decline in total species richness between 1981 and 2009, but changes were inconsistent between sites. Vascular plants began a gradual decline from the mid 1970s continuing through to 2009. Bryophyte species richness showed a large dip beginning at the end of the 1960s, with a recovery in species richness by 1981. In the vascular, bryophyte and total vegetation communities the plots with the lowest richness have shown an increase by 2009, whilst the plots with the highest richness have shown a decline by 2009. As a result there is a reduction in variation in total species richness between sites.

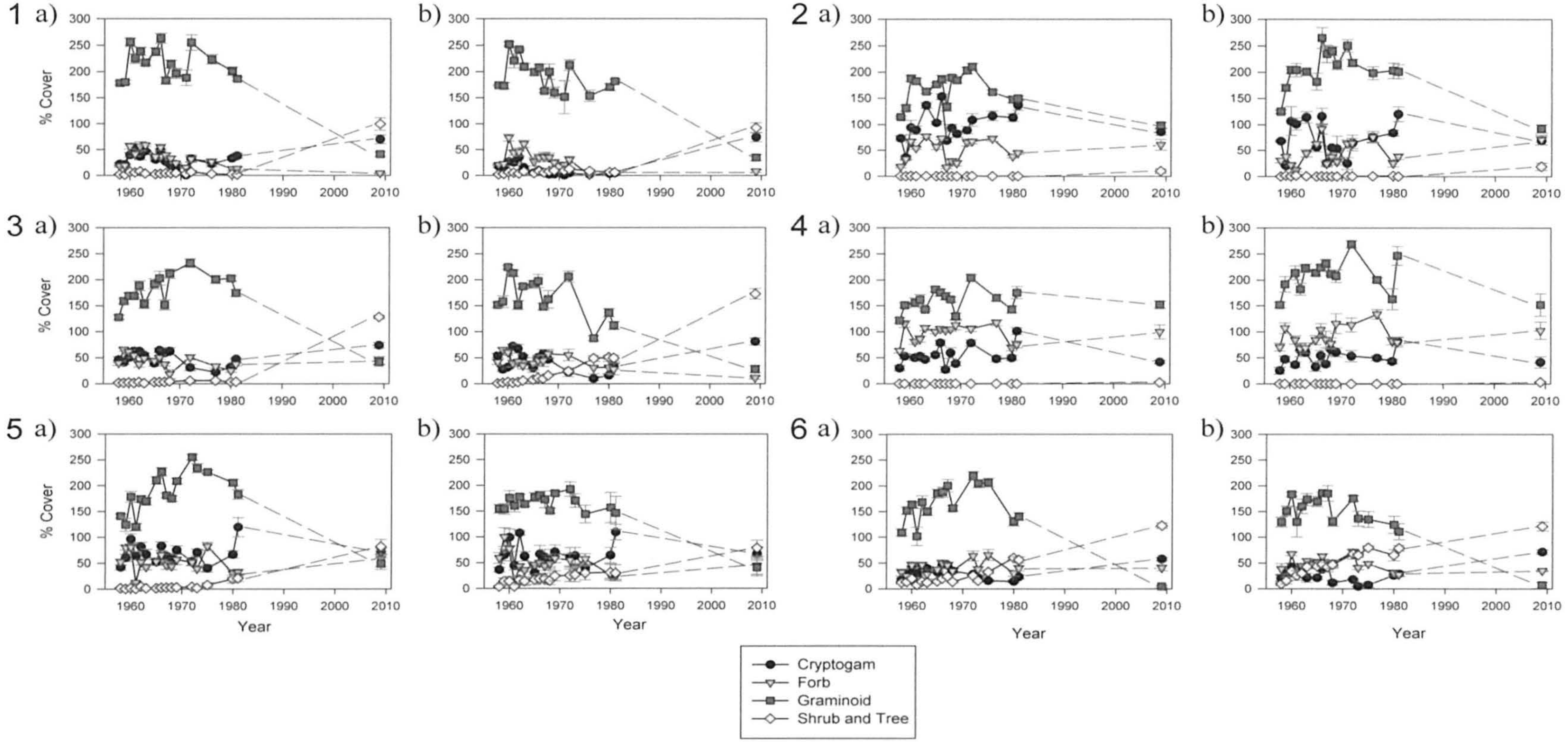

Figure 5-2 Relative percentage cover of vegetation for each site in Snowdonia National Park over time for a) Grazed, b) Ungrazed treatments at sites 1) Cwm Idwal 1, 2) Cwm Idwal 2, 3) Llyn Llydaw 1, 4) Llyn Llydaw 2, 5) Pen-y-Pass 1 and 6) Pen-y-Pass 2. Points represent the average of the treatments at that site; (Grazed (treatment 1 and 2) *n*=8, Ungrazed (treatment 3) *n*=4) error bars represent ± 1 s.e.m..

As might be expected the ranking of the sites in relation to abundance weighted EbR values (Figure 5-4) coincides fairly well with soil pH. All plots show a decline in weighted EbR value in the bryophyte community beginning in 1968, with a subsequent recovery in the early part of the next decade. The same trend is not seen in the vascular plant community however, which displays a gradual decline from the mid 1970s, continuing in most plots through to 2009. The dip in EbR for the bryophyte community corresponds with a decline in percentage cover of cryptogams (Figure 5-2). Although more apparent in the ungrazed treatment this decline was seen in all treatments suggesting this was not caused by grazing.

The EbN values for the total vegetation community appear relatively stable throughout the experiment. In general the vascular plants show an increase in EbN, although this is very small. The pattern within the bryophyte community is less clear. There was a large amount of inter-annual variation within the bryophyte community, with the vascular plant community appearing more stable.

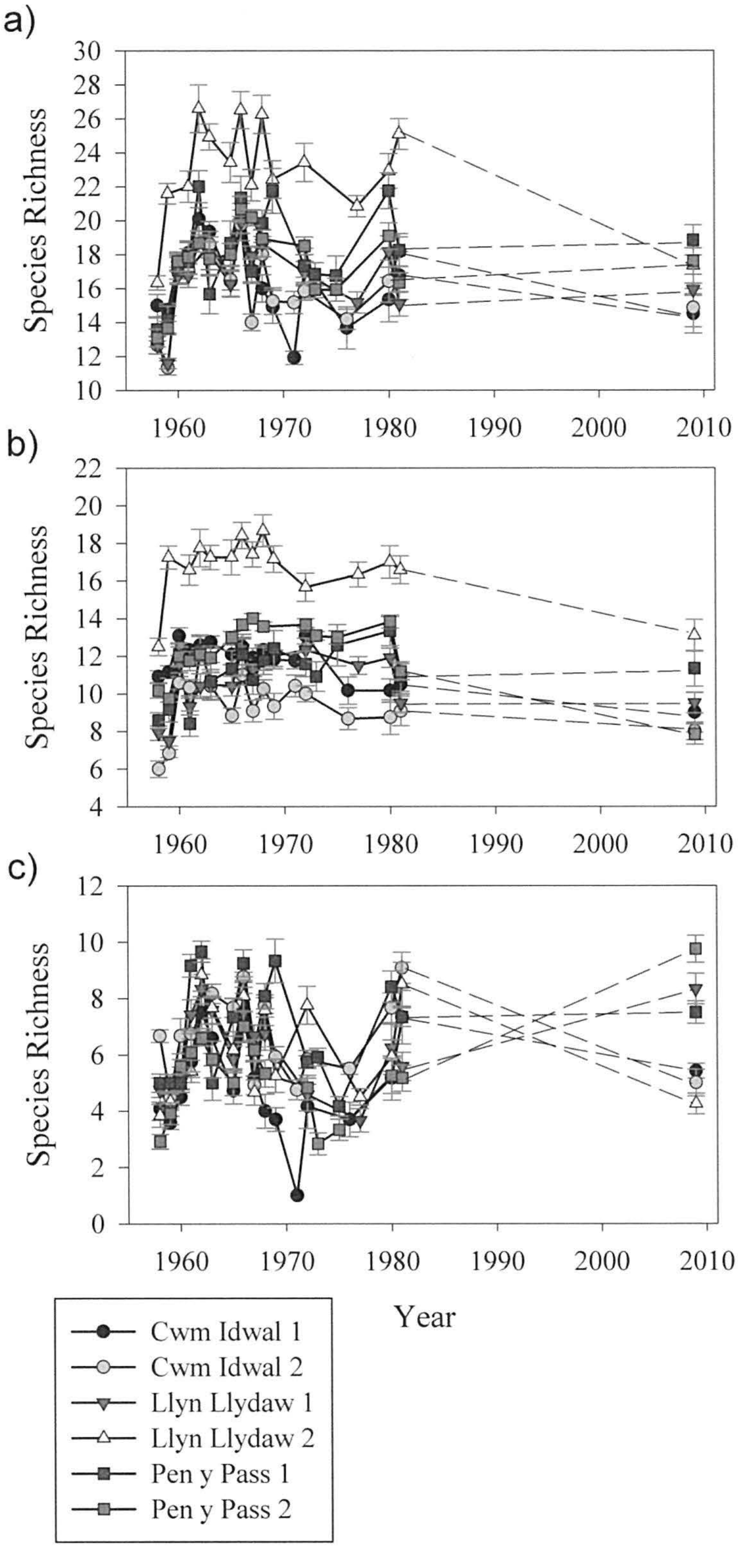

Figure 5-3 Species Richness for all grazing treatments for each site in Snowdonia National Park over time for a) total vegetation community, b) vascular plant community and c) bryophyte community. Points represent the average across all three grazing treatments (*n*=3); error bars represent ± 1 s.e.m..

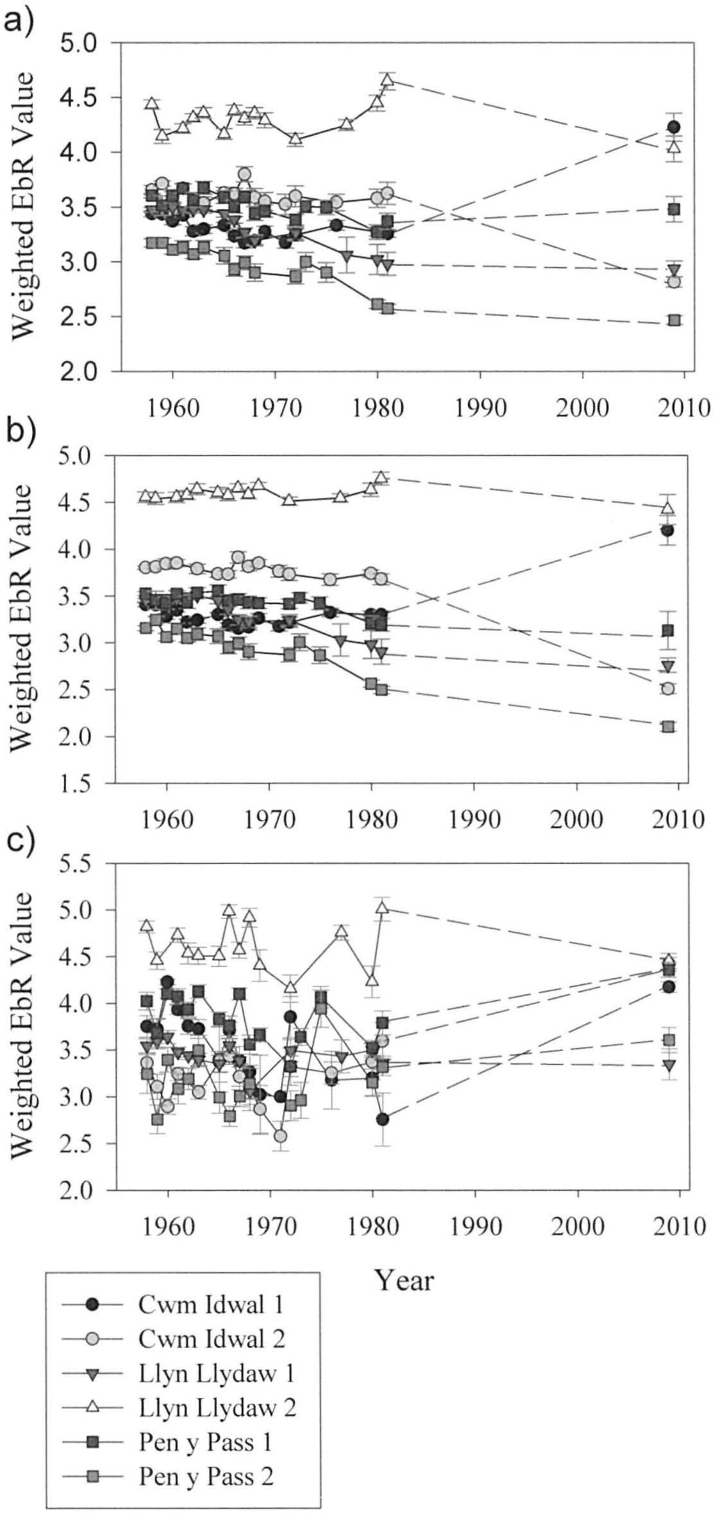

Figure 5-4 Abundance Weighted Ellenberg Reaction (EbR) values for all grazing treatments for each site in Snowdonia National Park over time calculated for a) total vegetation community, b) vascular plant community and c) bryophyte community. Points represent the average across all three grazing treatments (*n*=3); error bars represent ± 1 s.e.m..

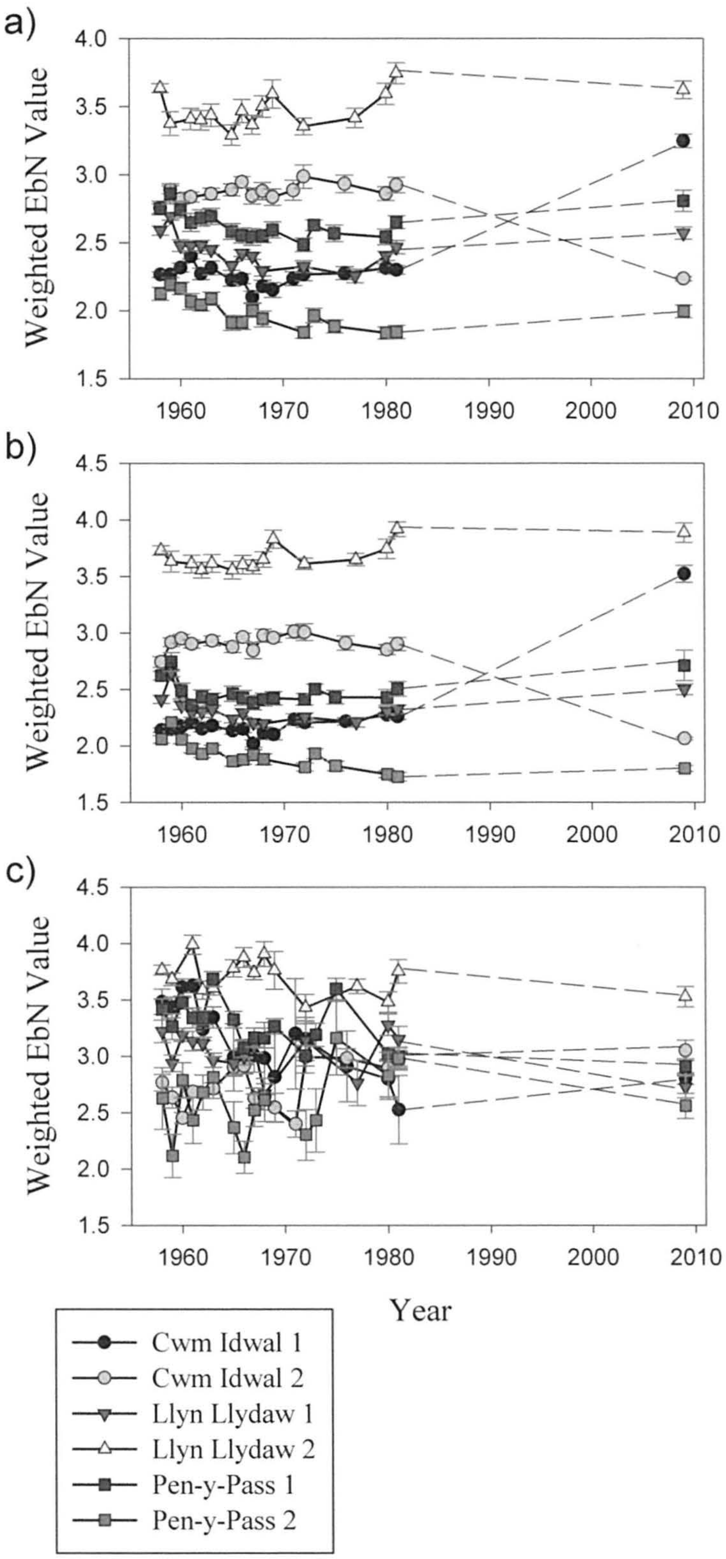

Figure 5-5 Abundance Weighted Ellenberg N (EbN) values for all grazing treatments for each site in Snowdonia National Park over time calculated for a) total vegetation community, b) vascular plant community and c) bryophyte community. Points represent the average across all three grazing treatments (*n*=3); error bars represent ± 1 s.e.m.

5.5 Discussion

This study found no significant difference in soil pH between grazing treatments, contrasting with the results of Marrs *et al.* (1989). Variation between grazing treatments was minimal compared to differences between sites. Interestingly, the differences between grazing treatments for Ellenberg variables and species richness were also small, despite the large changes in the species composition from grass dominated to shrub dominated in the ungrazed plots. Vegetation composition appeared to be more influenced by soil pH than grazing treatment or site location, suggesting that grazing was not the primary driver of the changes identified. The vegetation of the grazed and ungrazed plots were found to have converged by 2009. This finding has also been reported by Hill *et al.* (1992), who found vegetation had converged after nine years of the sites being ring-fenced. As no significant differences between grazing treatment were found whilst the original experimental design remained, the changes identified for EbR, EbN and species richness by 2009 were not caused by the removal of grazing from the previously ungrazed sites. Similarly no grazing treatment effect on soil %C or %N was found. Soil %C from 0-30 cm has remained relatively stable since 1957 despite the removal of grazing. Emmett *et al.* (2010) have also found little change in the soil C content of the soils sampled as part of the UK Countryside Survey. If bulk density did not change, the observed decline would indicate a reduction in the soil total N pool. Yet, %N has declined significantly since 1974. The reasons for this change is unclear, but could be linked to successional changes associated with grazing removal, or the acidification of the site during the 1970s. The decline does suggest there has been no eutrophication of the soils, at least in relation to total N content, within the experimental sites. Despite large changes in vegetation composition between grazing treatments, little effect of grazing on the variables reported here were identified.

The Snowdon grazing study is exceptional in a UK context, in that it spans the entire peak of S deposition in the UK (RoTAP, 2011). A number of studies published in the 1990s showed evidence of soil acidification (Billett *et al.*, 1990; Kuylenstierna & Chadwick, 1991; Farmer, 1995; Adamson *et al.*, 1996; Blake *et al.*, 1999). More recent studies have recorded evidence of recovery, in terms of rising pH, since the 1970s (Emmett *et al.*, 2010; Kirk *et al.*, 2010; McGovern *et al.*, 2011). The 50 year duration of the Snowdon grazing study permits this recent recovery to be evaluated relative to the preceding acidification, and suggests that the extent of recovery since the 1970s has been (far) smaller than the acidification that occurred between the preceding two decades, and that mean soil pH at these sites remains approximately half a unit lower than it was in 1957. This incomplete recovery of acidified soils is consistent with predictions from process models such as MAGIC (Wright *et al.*, 2005) which suggest that soils will be slower to recover from acidification and in some cases may even continue to acidify despite reduced deposition. This is in marked contrast to surface waters, which are now showing clear evidence of recovery, in Snowdonia and elsewhere (Davies *et al.*, 2005).

Recent studies have found evidence to support the suggestion that N deposition affects vegetation change through eutrophication (Smart *et al.*, 2005) and acidification (Stevens *et al.*, 2010). Only small changes in EbN were found within this study, and no obvious evidence of eutrophication. This may suggest that if N deposition is affecting the vegetation community it is through acidification. While this could become a greater threat if N deposition continues to increase at the rate seen in previous years (Galloway *et al.*, 2008), it is clear that the major driver of acidification and recovery to date has been S deposition, implying that this has also been the more important driver of observed vegetation change.

Acidification of soil has been shown to have a negative effect on species richness (Stevens *et al.*, 2010). Considerable interactions occur between soil pH and available plant nutrients, as base cations are readily leached from acidified soils (Horswill *et al.*, 2008; Stevens *et al.*, 2009). Although studies have reported an increase in soil pH there was no corresponding recovery in soil exchangeable base cation concentration (McGovern *et al.*, 2011). As the number of limiting soil resources increase, a reduction in plant species numbers has been found (Harpole & Tilman, 2007). In agreement with Stevens *et al.* (2010), this study has found a gradual reduction in species richness within the vascular plant community, with the start of the decline coinciding with the peak in S deposition, and continuing through to 2009. This is also in agreement with the findings of the UK Countryside Survey (Carey *et al.*, 2008).

Stevens *et al.* (2004) have reported a widespread loss of species within the UK as a result of N deposition, as identified by a gradient study across the UK., and have more recently suggested acidification as the mechanism for this (Stevens *et al.,* 2010). Whilst the study by Stevens *et al.* (2004) highlighted the wide spread loss of species, it considered only one time point, and could therefore not pin point when this loss of species occurred. A recent revisitation study highlighted the continued loss of species richness throughout the 1990s across the UK, and although this decline was associated with soil pH, a relationship was not found with N deposition (Van Den Berg *et al.*, 2010). In contrast, this study covers a long period, enabling the time at which the loss of species richness began to be evaluated. This long time scale of evidence suggests that the loss of species richness probably began at least as early as the 1960s, and that it was driven primarily by S deposition, with little evidence of subsequent recovery. For bryophytes a minimum level of species richness coincided with the peak in S deposition. Recovery in species richness is more evident in the bryophyte community than in the vascular plant community, which may be as a result of bryophytes reacting directly to atmospheric conditions rather than indirectly via changes in soil conditions (McGovern *et al.*, 2011). For vascular plants, and the plant community as a whole there appears to be a continued decline in species richness through to 2009 in many sites. Although change is unlikely to have been monotonic during the 28 year period when vegetation surveys ceased, it nevertheless appears clear that the ecosystem has not recovered fully from historic S deposition.

The differing response of the bryophyte and vascular plant communities are evident when looking at the EbR values. The bryophyte community displays a different response to S deposition with large reductions in EbR values in the 1970s, compared to a gradual decline in vascular plant EbR value. Moreover, the bryophyte community has shown greater recovery from peak deposition rates than vascular plants; which are indicating more acidic communities in 2009 than in 1958. Both the vascular plant and bryophyte communities display a significant time lag between acidification of the soil and evidence emerging within the vegetation. Although the soil pH showed the greatest amount of acidification during the 1960s, evidence of vegetation change did not appear until the 1970s. Soil conditions and atmospheric deposition respond to changes in S emissions at different time scales. Bryophytes are likely to show rapid responses to changes in pollution levels as they respond directly to atmospheric inputs, whereas, vascular plants are more responsive to changes in soil conditions. These may display a lagged response relative to deposition by a few years to decades (Galloway *et al.*, 1983). This finding highlights the difficulties with trying to relate vegetation changes to environmental drivers over short periods, and the importance in considering the ecosystem as a whole when assessing biodiversity change. This study also illustrates the value of preserving both historical datasets and maintaining grazing exclosures for future use. Without such resources, studies that provide important insights into long-term environmental change would be impossible.

5.6 Conclusions

Although S emissions have been tackled successfully through legislation, and S deposition has fallen dramatically, the legacy of past ecosystem acidification remains clear. This study suggests that soil pH remains well below pre-acidification levels; bryophytes exhibit a lagged response to deposition changes; and that vascular plant species richness may still be significantly reduced relative to 1960s levels. This acidification legacy has implications for the future status of upland ecosystems, and for our understanding of the impacts of other environmental drivers that often receive far greater attention. For example, whilst many studies reported evidence of negative effects of N deposition in semi-natural ecosystems on a large scale; few of these studies have taken account of the potential confounding influence of S deposition. Similarly, the impacts of climate or land management changes may differ between ecosystems that have been subject to acidification, and those that remain in (or have returned to) an unacidified state. This paper provided evidence that both vegetation and soils are still negatively affected by historic S deposition, and suggests that this legacy needs to be taken into account when considering the future effects of other environmental drivers.

5.6.1 Acknowledgements

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Chapter 6

Grazing removal increases inorganic N leaching in a mountain grassland ecosystem

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6.1 Abstract

Much of the considerable research on the process of nitrogen (N) saturation has been concentrated on forested ecosystems. Yet, the acid sensitive areas of upland Britain are dominated by heathland and acid grassland communities in which the effects of N saturation and nitrate leaching are poorly understood. There is also little evidence concerning the effects of grazing on soil solution in upland Britain. A long-term grazing experiment in the Welsh uplands, allowed the quantification of the effect of soil type and grazing on the chemical parameters of both the soil and soil solution. Significant differences were found, with the type of vegetation (influenced by grazing) being the predominant factor influencing N saturation and nitrate leaching. Significantly more soil nitrate occurred in the ungrazed plots, irrespective of soil type, along with significantly lower pH and higher nitrate leaching in the soil solution. This study provides evidence that grazing and thereby vegetation type affect the ability to utilise increased N availability from N deposition. This may have significant consequences for the future recovery of acidified surface waters; especially where land use and subsequent vegetation change occurs through the withdrawal of grazing extensively within acidified catchments.

6.2 Introduction

Atmospheric concentrations of reactive nitrogen (N) have increased exponentially in the last few decades (Galloway *et al.*, 2008). Elevated inputs have the potential to significantly alter nutrient poor systems throughout much of the world and concern about the cascade of ecosystem impacts has been expressed by scientists, land managers and politicians (Wright *et al.*, 2001). In response to internationally agreed protocols deposition of sulphur (S) has reduced substantially, and N deposition is now the dominant anion in some upland areas (Curtis *et al.*, 2005). Responding rapidly to changes in deposition, mountain ecosystems are extremely sensitive to environmental changes and are particularly at risk (Helliwell *et al.*, 2010) owing to the predominance of low nutrient soils and associated vegetation communities. Historically, nutrient inputs in alpine habitats have been low and therefore enhanced levels of N deposition pose a significant risk (Curtis *et al.*, 2005; Britton & Fisher, 2007). This risk is exacerbated by the higher orographic precipitation and deposition from clouds, mist and fog experienced in montane areas (Fowler *et al.*, 2005). The addition of reactive N has the potential to cause increased growth rates, sward composition change and loss of biodiversity in vegetation adapted to nutrient poor conditions (Evans *et al.*, 2006a). A progressive loss of biodiversity has been found along a gradient of increasing N deposition in a UK study within acidic grasslands (Stevens *et al.*, 2004). A temporal shift towards more nutrient-demanding species has also been found within a repeated national scale vegetation survey in semi-natural upland systems across the UK (Smart *et al.*, 2003).

In addition to the eutrophication effects, excess N can lead to N saturation where inputs exceed biological demands (Aber *et al.*, 1998) and inorganic N (nitrate, NO_3^- and ammonium, NH_4) is only partly retained through uptake and immobilisation (Evans *et al.*, 2006a). N saturation of the soil can lead to leaching of NO_3^- (Rowe *et al.*, 2006) from the soil to surface waters, thus contributing to acidification and the loss of biodiversity (Evans *et al.*, 2006a). NO_3^- is a significant contributor to acidification in the UK (Allott *et al.*, 1995), and is expected to become increasingly important in the future, potentially limiting or reversing surface water recovery from acidification (Curtis *et al.*, 2005).

Increased N flux may also have effects on soil carbon (C) by removing restrictions on microbial growth and respiration thereby increasing the rate of soil C oxidation (Neff *et al.*, 2002). This may also result in a decrease in soil C/N ratio if these processes outweigh the increases in plant growth (Rowe *et al.*, 2006). A weak relationship between soil C/N ratio and N leaching has been found in UK moorland sites along a deposition gradient (Curtis *et al.*, 2004). Evans *et al.* (2006b) observed a similar pattern at a regional scale, although this relationship was complicated by the amount of soil organic matter available to immobilise the N. Soil type may also affect the relationship between C/N ratio and the proportion of N leaching as this is likely to differ systematically between habitat types further confounding the relationship (Rowe *et al.*, 2006).

The interaction between N deposition and land management practices can have significant impacts on the structure and functioning of alpine ecosystems, with the potential loss of N to surface waters and their acidification, resulting in serious consequences for water quality and downstream habitats (Helliwell *et al.*, 2010). The influence of vegetation type on N leaching has not been well defined (Rowe *et al.*, 2006) as the majority of research into the process of N saturation has been targeted on forested ecosystems (Evans *et al.*, 2006a). Yet, most of the acid sensitive areas within the UK comprise heathland and acid grassland communities in which the understanding of N dynamics remains relatively poor (NEGTAP, 2001). Upland ecosystems, although considered to have low nutrient availability, receive significant inputs of N through rain and snow. Depending on their capacity to utilise this additional N, the competitive balance between species may be altered (Alonso & Hartley, 1998).

Land management can have significant effects on the composition of vegetation. The main management activity within temperate grasslands, either for economic or conservation objectives, is grazing (Bullock *et al.*, 2001). In the uplands of Britain, free ranging sheep grazing is an integral part of the management (Welch & Rawes, 1964). It has been shown that grazing is the principal factor controlling vegetation change, with grazing increasing the proportion of grasses and decreasing the proportion of shrubs and trees (Milne & Hartley, 2001). Succession from grasslands to heather often occurs as a result of long-term removal of grazing, demonstrated by long-term sheep exclusion experiments in the uplands (Hill *et al.*, 1992).

Vegetation cover is a reflection of nutrient availability and therefore of the underlying geology and soil type (Milne & Hartley, 2001). Vegetation composition can exert large effects on nutrient cycling, with heathlands characterised by high nutrient turnover time in plants and soil and low levels of plant-available nutrients (Aerts & Chapin, 1999), in contrast to grasslands which are associated with more rapid N cycles and greater plant available nutrient levels (Heil & Bobbink, 1993). Evidence also exists to suggest that long-term changes in grazing pressure can impact on soil pH (Marrs *et al.*, 1989). Although there is currently a lack of information on the effects of grazing on soil solution in the British uplands, Helliwell *et al.* (2010) have recently shown little effect of grazing on soil solution chemistry in the Scottish uplands. An existing long-term grazing experiment in the Welsh uplands, located in an area of high N deposition (RoTAP, 2011), provides a unique opportunity to investigate the effect of land-use (in terms of grazing regime and thereby vegetation composition) and soil type on both soil and soil solution chemistry, along with implications for land management.

6.3 Material and Methods

This study utilised an existing grazing exclusion experiment set up in 1957 in Snowdonia by Hughes & Dale (1970). Six grazing exclosure plots, located in three areas were used in this study. Snowdon is highly variable in its geology, soils and climate (Perkins, 1978) and as such the original sites were established on a variety of soils, ranging from a peaty podzol through to a brown earth (Hill *et al.*, 1992) which supported between 2.5 to 5.0 sheep ha^{-1} respectively (Hughes & Dale, 1970). Sites were located between 350 and 450 metres above sea level. The existing exclosure plots in each area were located on two different soil types, allowing the effect of soil type to be investigated. A grazed plot, of the same size of the ungrazed plot, was established for this study next to each of the six existing ungrazed plots.

6.3.1 Original experimental design

A large scale grazing experiment, established in 1957 across Snowdonia National Park provided the experimental design for this study (Hughes & Dale, 1970). Snowdon is highly variable in its geology, soils and climate (Perkins, 1978) and as such the sites were established on a variety of soils, ranging from a peaty podzol through to a brown earth (Hill *et al.*, 1992) which supported between 2.5 to 5.0 sheep ha^{-1} respectively (Hughes & Dale, 1970). Sites were located between 350 and 450 metres above sea level. A replicated exclusion experiment was set up at each site, with twelve 2.8m x 3.7m plots. These were arranged in a randomised block experimental design, with three treatments and four blocks. The treatments were as follows:

- Treatment 1: Normal grazing - Sheep allowed free access, most of the sheep are removed from the mountain in winter
- Treatment 2: Summer only grazing - Sheep excluded in winter from October to April
- Treatment 3: No grazing - Sheep excluded throughout the year.

This experimental design was ended in 1981, but all treatments were then ring-fenced, thus converting the summer-only and year-round grazing plots to ungrazed. Of the original experimental design (Hughes & Dale, 1970) only six of the original nine sites were used in the current investigation, due to uncertainty of fencing integrity throughout the intervening period for the other three sites. The six sites used within this study were located in three areas, Llyn Llydaw, Pen-y-Pass and Cwm Idwal. For this study, one replicate from treatment 3 was selected in each of the plots to provide the ungrazed plot.

6.3.2 Vegetation cover

Vegetation cover was recorded using 100 randomly placed pins (35 cm x 2 mm) per experimental plot. The individual points were located by coordinates derived from random number tables. Presence or absence of species were recorded at each pin, with presence occurring if any live part of a plant was touching the pin. All plants were identified to species level and split by functional group (cryptogams, forbs, graminoids, shrubs and trees) to analyse the species composition between treatments.

6.3.3 Soil solution sampling and chemical analysis

Four rhizon® soil water samplers (Rhizosphere Research Products, Wageningen, The Netherlands) were placed in an area of homogeneous vegetation within each plot to reduce the possibility of vegetation effects within treatments. The four rhizons® were placed in a square, with two metres between each rhizon®. The same layout was used within both the grazed plots and exclosures, located to ensure water runoff from the opposite treatment did not affect the samplers. The samplers were installed within the top 30cms of the soil at the start of December 2008 to allow the soil to settle before the sampling commenced. Soil solution was extracted in situ and sampled every month from January to December 2009. To ensure sufficient sample was collected and a standard protocol throughout the year, soil solution was sampled over a 48 hour period once a month, using a BD Vacutainer®. Samples from the four rhizons® in each plot were bulked prior to analysis.

pH (Hanna Instruments; Leighton Buzzard, UK pH 209 pH meter) and electrical conductivity (EC) (Jenway 4010 EC meter; Bibby Scientific, Stone, UK) were determined on the soil solution directly. Dissolved organic C (DOC) and total dissolved nitrogen (TDN) in soil solution were measured with a Shimadzu TOC-TNV analyser (Shimadzu Corp., Kyoto, Japan). NH_4^+-N in soil solution was determined colorimetrically by microassay (Downes, 1978; Mulvaney 1996). Cl, NO_3^--N, PO_4-P and SO_4 were determined on a Dionex DX120 Ion Chromatograph (Dionex Corp., Sunnyvale, USA). Cations were measured using a Perkin Elmer AAnlyst 400 Atomic Absorption Spectrometer (PerkinElmer, Inc., Waltham, MA, USA) and these were summed to provide total cation concentrations. Dissolved organic N (DON) was calculated as the difference between TDN and dissolved inorganic N (DIN). The acid neutralising capacity (ANC) was calculated using the method of Helliwell *et al.* (2010) as the sum of base cations minus the sum of strong acid anions i.e. ($Na^+ + Ca^{2+} + Mg^{2+} + K^+ + NH_4^+) - (NO_3^- + Cl^- + SO_4^{2-}$).

6.3.4 Soil sampling and chemical analysis

A 5cm diameter soil core was taken to 50cms from the location of each sampler at the end of the experimental period and split by horizon. After removal of vegetation and roots these were bulked per plot. pH and EC were determined on a 1:2.5 soil:H_2O extract using the same instruments used for the soil solution samples. NO_3^--N and NH_4^+-N were extracted using a 1:4 ratio soil:1M potassium chloride (KCl) extractant and determined colorimetrically by microassay (Downes, 1978; Mulvaney, 1996). Microbial C and N were determined using the fumigation-extraction technique (Brookes *et al.,* 1985, Vane *et al.,* 1987) and calculated using a K_{EC} value of 0.45 and 0.54 respectively. Total C and N of soils were determined using a LECO Truspec® CN Analyser (LECO, Stockport, UK). Exchangeable cations were extracted using a 1:20 soil to 0.5M ammonium chloride ratio and analysed using a Perkin Elmer AAnlyst 400 Atomic Absorption Spectrometer. Exchangeable acidity was determined on a 1:20 soil:1M KCl extract by fixed point titration to pH 8.3 with 0.1M NaOH (Page *et al.,* 1982) using a Metrohm 888 Titrando (Metrohm UK Ltd., Runcorn, UK). Effective cation exchange capacity (CEC) was calculated as the sum of the exchangeable bases (Na^+, Ca^{2+}, Mg^{2+} and K^+) and exchangeable acidity. Base saturation was calculated as the percentage of cation exchange capacity occupied by the base cations (Na^+, Ca^{2+}, Mg^{2+} and K^+). Bulk density was measured using a 100cm^3 metal ring. Percentage moisture was calculated after drying at 80°C until no more weight loss was recorded.

To ensure quality control, standard soils with known chemical concentrations were included within the analysis.

6.3.5 Statistical Analysis

To investigate the effect of grazing and soil type on soil and soil solution chemical parameters, an analysis of variance taking account of the split plot design were run in R (R Development Core Team, 2010). [R code used for analysis, model<-aov(x~soil type*grazing treatment +Error(site/soil type))]. Additional paired t-tests were run to investigate the effect of soil type on soil solution chemistry [model<-t.test(x-treatment, paired=T)]. Soil chemical values are expressed on a dry weight basis. To avoid the errors of pseudoreplication, monthly data were averaged to provide one value per treatment per plot, for each chemical parameter measured.

6.4 Results

6.4.1 Vegetation change

Significant differences in vegetation exist between the grazed and ungrazed treatment plots (Table 6-1). Grazed plots were dominated by graminoids, and contain little cover of shrubs and trees. Ungrazed plots contain significantly more shrubs and cryptogams, with an associated decline in graminoids. Soil type had little effect on percentage vegetation cover with grazing treatment exerting a larger effect.

6.4.2 Effects of Soil Type

The significant effects of soil type were mainly seen within the chemical composition of the soil (Table 6-2) rather than the soil solution. The peaty podzol showed higher exchangeable acidity and higher cation exchange capacity with an associated lower base saturation. pH was lower in the peaty podzol although this was only significant at the $p<0.1$ level. The plots on the peaty podzol also contained higher C concentrations, although again this was only significant at the $p<0.1$ level. A difference in soil C and N content was found between the two soil types although this was only significant at the $p<0.1$ level. Both C and N showed the same pattern however, with the ungrazed plot on the brown earth having higher concentrations than the grazed plot, but with the opposite true for the peaty podzol. This maintains the C/N ratio so that there is only an effect of soil type on C/N ratio and no grazing effect.

Soil type had little effect on the chemical composition of the soil solution (Table 6-3). Although both K^+ and Na^+ showed significant differences between the soil types, total base cation concentration was not significantly different between soil types.

Table 6-1 Weighted percentage cover by plant functional type from grazed and ungrazed treatment plots in Snowdonia. Values represent mean ± sem (*n* = 3). Significant *p* values are shown in bold.

	Brown Earth				**Peaty Podzol**				***p***	
	Grazed		Ungrazed		Grazed		Ungrazed		Soil Type	Grazing
Cryptogam	14.29	± 5.250	19.70	± 3.898	15.795	± 2.231	30.486	± 3.789	0.432	**0.033**
Forbs	29.41	± 3.783	17.73	± 8.941	20.70	± 2.834	17.79	± 3.870	0.608	0.241
Graminoids	54.36	± 3.081	29.25	± 14.367	56.36	± 5.807	15.63	± 8.602	0.351	**0.018**
Shrubs & Trees	1.95	± 1.597	33.31	± 5.095	7.15	± 5.095	36.09	± 15.451	0.752	**0.014**

Table 6-2 Chemical characteristics of soil from grazed and ungrazed treatment plots in Snowdonia. Values represent mean ± 1 sem ($n = 3$). Significant p values are shown in bold.

	Brown Earth				**Peaty Podzol**				p	
	Grazed		Ungrazed		Grazed		Ungrazed		Soil Type	Grazing
C ($g/100g^{-1}$)	4.23	± 1.011	9.17	± 4.937	25.32	± 4.512	18.52	± 4.007	0.060	0.728
N ($g/100g^{-1}$)	0.29	± 0.059	0.61	± 0.273	1.39	± 0.279	1.01	± 0.232	0.098	0.839
C/N	14.18	± 0.848	13.92	± 1.486	18.48	± 0.789	18.33	± 0.246	0.067	0.664
NH_4^+ ($mg/100g^{-1}$)	0.20	± 0.038	0.39	± 0.159	0.15	± 0.032	0.19	± 0.045	0.407	0.159
NO_3^- ($mg/100g^{-1}$)	1.51	± 0.890	5.32	± 0.593	4.11	± 1.368	7.10	± 0.641	0.186	**0.029**
Microbial C (μg/g)	2086.25	± 383.707	1598.27	± 278.282	3678.76	± 1097.284	1621.78	± 229.947	0.434	0.094
Microbial N (μg/g)	157.32	± 32.206	189.06	± 41.471	254.48	± 78.991	188.67	± 58.564	0.469	0.568
pH	4.86	± 0.101	4.66	± 0.198	4.22	± 0.072	4.06	± 0.075	0.058	**0.030**
Total cations ($mEq/100g^{-1}$)	0.95	± 0.134	1.32	± 0.586	1.10	± 0.488	0.61	± 0.038	0.244	0.792
Exchangeable acidity ($mEq/100g^{-1}$)	1.48	± 0.507	2.84	± 1.244	8.67	± 0.224	9.78	± 1.098	**0.012**	0.173
CEC ($mEq/100g^{-1}$)	2.43	± 0.570	4.15	± 0.813	9.77	± 0.703	10.39	± 1.088	**0.011**	0.233
Base saturation (%)	43.44	± 10.439	39.93	± 20.127	10.68	± 4.062	6.02	± 0.833	**0.028**	0.213
Moisture content (%)	1.46	± 0.232	2.22	± 0.977	3.18	± 0.327	3.13	± 0.530	0.106	0.531
Bulk density ($g\ cm^3$)	0.56	± 0.113	0.47	± 0.094	0.13	± 0.010	0.25	± 0.111	0.161	0.827

Table 6-3 Chemical characteristics of soil solution leached from grazed and ungrazed treatment plots in Snowdonia. Values represent mean ± 1 sem ($n = 3$). Significant p values are displayed in bold. Values (except pH and ANC) are shown in μmol per litre.

	Brown Earth		Peaty Podzol		p	
	Grazed	Ungrazed	Grazed	Ungrazed	Soil Type	Grazing
DOC	1819.12 ± 136.22	1527.37 ± 168.74	1852.90 ± 19.23	1896.50 ± 214.62	0.249	0.530
TDN	145.58 ± 25.97	90.54 ±15.93	101.71 ± 6.43	129.74 ± 24.08	0.898	0.557
NH_4^+-N	21.41 ± 2.74	19.72 ± 2.72	17.19 ± 1.57	27.47 ± 9.34	0.842	0.418
NO_3^--N	6.65 ± 3.84	38.69 ± 9.78	7.74 ± 3.56	49.88 ± 16.43	0.764	**0.010**
DIN	28.06 ± 6.58	58.41 ± 12..00	24.93 ± 2.29	77.35 ± 17.62	0.470	**0.016**
DON	117.53 ± 23.60	32.13 ±4.88	76.78 ± 8.32	52.40 ± 8.61	0.488	**0.018**
pH	6.15 ± 0.053	5.97 ± 0.120	5.81 ± 0.005	5.67 ± 0.004	0.149	**0.024**
Total base cations	113.29 ± 19.33	94.11 ± 4.07	85.98 ± 3.00	96.21 ± 9.17	0.245	0.906
SO_4^{2-}	66.25 ± 12.13	82.37 ± 10.47	66.11 ± 14.94	80.80 ± 0.50	0.884	0.229
Cl^-	140.29 ± 12.82	139.25 ± 13.86	140.88 ± 8.99	155.27 ± 8.23	0.666	0.237
ANC	-0.08 ± 0.014	-0.15 ± 0.014	-0.11 ± 0.026	-0.16 ± 0.014	0.331	**0.030**

6.4.3 Effect of grazing

The pH was significantly lower in the ungrazed treatment for both the soil and soil solution. Grazing exerted a significant affect on NO_3^- content within the soil, with lower concentrations in the soil from the grazed treatment than the ungrazed treatment. Similarly NO_3^- content of the soil solution from the grazed treatment was lower than that of the ungrazed treatment. An effect of grazing on ANC within the soil solution was also found.

Grazing had no effect on total N concentration within the soil solution. However, it did effect the relative partitioning between inorganic and organic N within the soil solution. Lower concentrations of DON were found within the soil solution from the ungrazed plots than the grazed plots. The opposite was found for DIN concentrations (Figure 6-1). No significant interaction between soil type and grazing on N leaching was found with both soil types showing the same N leaching pattern. Although large differences in the mean were found for NO_3^- in the soil solution from the peaty podzol (Figure 6-1), this difference was not significant due to similar concentrations found at one site between the grazed and ungrazed treatment plots.

Grazed and ungrazed treatment plots displayed similar patterns in soil solution chemistry over time (Figure 6-2) for pH, NO_3^-, DON and DOC, with less variation between treatments being detected in the second half of the year.

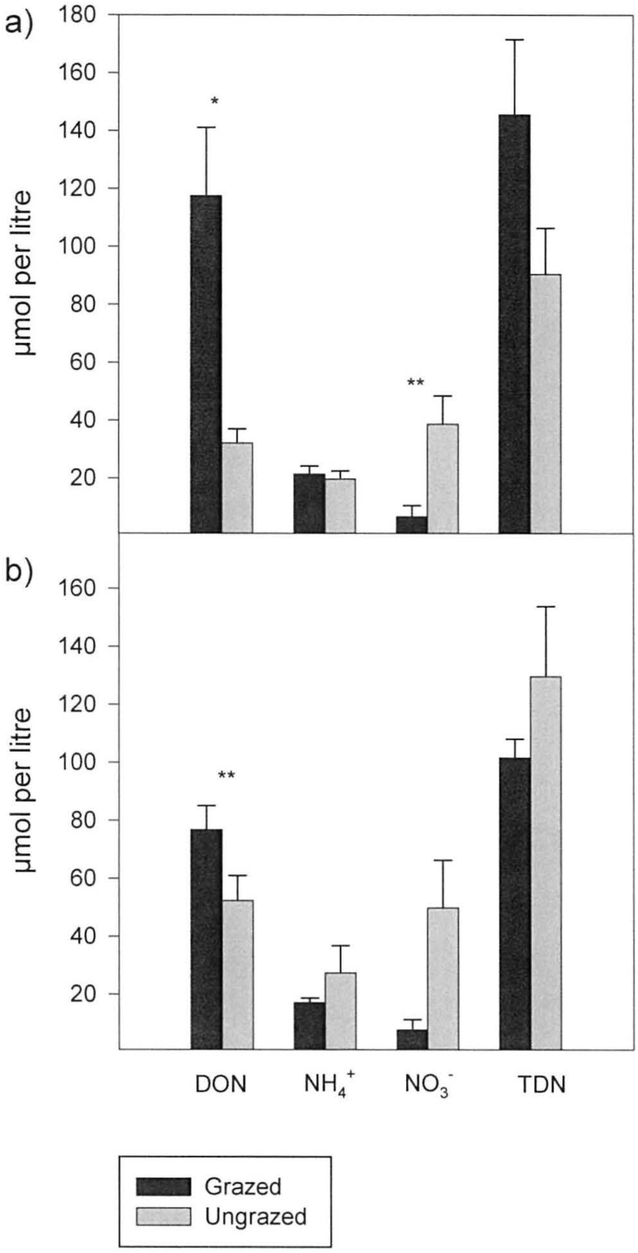

Figure 6-1 Effect of grazing on N partioning in soil solution from treatment plots in Snowdonia for a) Brown earth and b) Peaty podzol. Bars represent the average for the treatment, error bars represent ± 1 s.e.m. n=3. Asterisk indicate degree of significance of difference calculated with paired t tests * P<0.05, ** P<0.01

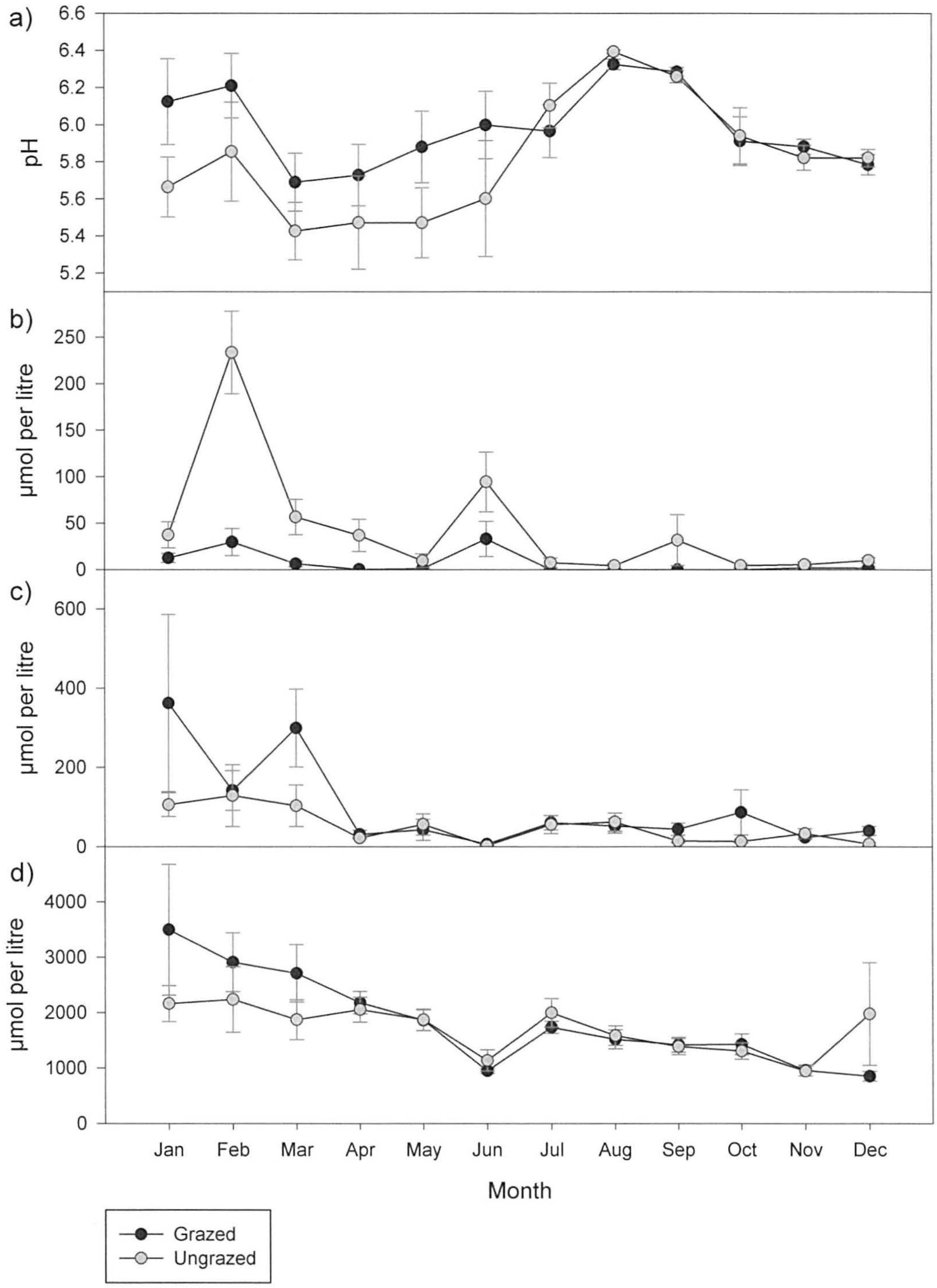

Figure 6-2 Trends in soil solution chemistry over time for a) pH, b) NO_3^- c) DON and d) DOC for grazed and ungrazed treatment plots in Snowdonia. Points represent the average across the treatment and error bars represent ± 1 s.e.m. *n*=6.

6.5 Discussion

The exclusion of grazing livestock over 50 years led to a succession from grassland to heathland communities within the exclosures but there was limited evidence of further succession to woodland, probably due to the lack of suitable propagules in the locality. After 50 years of differing land-use, in terms of grazing, this study has identified some significant differences between the soil and soil solution chemical composition of these upland plots. However, although the C and N values differed between soil types, there was little difference between land-uses. There did not appear to be any evidence of a relationship between soil C/N ratio and N leaching as has been found in a number of other studies (Evans *et al.*, 2006b; Helliwell *et al.*, 2010). Unlike Helliwell *et al.* (2010) and Evans *et al.* (2006b) the plots within this study were all located in areas with similar rates of N deposition, and therefore no N deposition gradient existed, which could explain the lack of relationship. The C/N ratio did not differ with vegetation type in contrast to the suggestion by Rowe *et al.* (2006). Although Rowe *et al.* (2006) investigated the effect of vegetation type on soil C/N ratio and N leaching, it was not possible to carry out a comparison of soil types within the study, and it was suggested as a possible confounding variable. This study was able to contrast two soil types, both present in the same area, with two different vegetation communities, located adjacent to each other, thus providing some control of the variables suggested as confounding factors by Rowe *et al.* (2006). There was little evidence that the soil C/N ratio corresponded with NO_3^- leaching however.

Land-use and the consequent vegetation community differences had a significant effect on N leaching. Higher NO_3^- leaching corresponded to a significantly lower pH, of both soils and soil solution, in the ungrazed heathland plots. The ANC was also lower in these plots, suggesting the ability to neutralise additional acid was impaired. This may have significant implications for the future, if N deposition levels in the UK continue to rise as projected. Concern has been expressed about the effect of NO_3 leaching with suggestions that it may lead to a halt in the recovery and possibly a reversal in the recovery from acidification (Curtis *et al.*, 2005). These plots are located in areas of high N deposition (RoTAP, 2011), and further input my lead to an increase in leaching of NO_3^-, and contribute to the further acidification of catchment surface waters.

Higher DIN within the soil solution does appear to suggest that N saturation may already be occurring in the ungrazed heathland-dominated plots. A greater concentration of NO_3^- within the soil solution of the ungrazed plots is also suggestive of N saturation. There is already concern over the future of grazing in some parts of the uplands with agricultural abandonment occurring (MacDonald *et al.*, 2000), and removal of grazing in some localities may lead to increased succession from grassland to heathland. This may in turn lead to an increase in N leaching from these soils, further exacerbated by predicted increases in N deposition.

It has been suggested that increased N deposition may have effects on soil C by removing limitations to microbial growth, resulting in a decrease in the soil C/N ratio if these processes outweigh increases in plant growth (Rowe *et al.*, 2006). Despite differences in plant growth between land-use types, this study found no difference in microbial C and N of the soil between land-use or soil types.

Heathlands and grasslands differ markedly in their levels of nutrients available to plants (Heil & Bobbink, 1993; Aerts & Chapin, 1999). Heathlands are characterised by low plant nutrient availability and are often limited by N availability. N deposition results in greater inputs and availability of NH_4 which can result in increased N cycling. With the dual effect of promoting nitrification and inhibiting NO_3^- immobilisation, both directly and indirectly, increased inorganic N inputs will result in an accumulation of NO_3^- within the soil and increase the potential for NO_3^- leaching (Curtis *et al.*, 2005). This appears to be evident within the ungrazed heathland plots investigated within this study and this effect is independent of soil type. The faster N cycling of the grassland plots appears to be utilising the increased available N, possibly through growth and repair from being grazed. If grazing reduces substantially, the grassland plots may not be so efficient at utilising this excess N resulting in increased leaching in the future. These results suggest that large scale grazing removal projects that allow succession to heathland in order to meet habitat restoration goals may have knock on, unintended consequences for the capacity of the catchment to recover from acidification, and potentially pose a risk to the ecology of freshwater ecosystems.

6.6 Conclusion

This study has revealed significant differences in the ability of ungrazed heathland and grazed acid grassland to utilise additional N input from deposition. The effects of increased N deposition appear to be mediated through grazing with grazed plots utilising additional N to a greater extent than ungrazed plots, irrespective of soil type. Whilst concern has been expressed about the impact of grazing on upland vegetation communities, little attention has been directed to below ground responses. This study highlights that grazing removal projects for habitat restorations may have greater consequences than intended for upland ecosystems. This paper has provided evidence that grazing impacts on both soil and soil solution chemistry in the uplands which may have implications for the ecology of freshwater ecosystems if grazing removal projects for habitat restoration goals are instigated.

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Chapter 7

General Discussion

7.1 Overview of thesis

As detailed in Chapter 1, this thesis investigated the impact of long-term changes in environmental factors on the soils and vegetation of Snowdonia. The broad aims were to:

1) Identify if any change in vegetation has occurred in Snowdonia over the past forty years
2) Identify if any change in soil chemistry has occurred in Snowdonia over the past forty years
3) Identify if changes in vegetation or soil chemistry driven by environmental change is mitigated through grazing removal in Snowdonia
4) Identify the main drivers of any changes identified

Biodiversity is being lost at an unprecedented rate (Sala *et al.*, 2000), and there is an urgent need to understand the effects of both individual and combinations of drivers of environmental change over the long term. To this end, this thesis evaluated four aims, which are discussed in depth in the individual chapters. All aims were satisfied, and this discussion considers the implications of the results.

7.2 Implications of research

7.2.1 Revisitation Studies

It has been noted that revisitation studies can produce high quality data, as long as experimental plots can be relocated with sufficient accuracy (Ross *et al.*, 2010), and the methodology is maintained to allow detection of real environmental change rather than methodological variability (Morecroft *et al.*, 1997; Beard *et al.*, 1999; McCollin *et al.*, 2000). Throughout the preceding chapters, the original methodology for each of the revisitation studies was maintained, which should allow detection of real change, however, this research has identified an issue with revisiting surveys not designed for monitoring purposes. The success of revisitation studies relies heavily on the methodology and planning leading to the original survey. Whilst it is still valid to repeat surveys, even when the methodology may not have been explicitly set up for monitoring, it does create problems when the objective of the subsequent analysis differs from the original. In Chapter 4, no significant change in the soil chemistry was shown, and whilst this may well indicate a resistance to change, it may be that little change was identified due to the multitude of soil types and underlying geological combinations that occurred within the study. The original study was commissioned to identify the variety of soil types and where they occurred on Snowdon and was therefore designed as an investigation of heterogeneity rather than a scientific investigation (Ball pers. comm.). To this end, the location of survey sites was not designed with any scientific methodology in mind, and samples were collected at random (Ball pers. comm.). Although the survey sites were not permanently marked, and were only recorded to six figure grid references, the detailed descriptions of the sites and soil types were sufficient to relocate with some degree of accuracy. When collecting the soil for Chapter 4, the methodology had to take into account the lack of permanent marking and additional samples in each site were collected to determine the within-site variation, however all laboratory methodologies remained the same.

The problems with plot relocation were negated in the studies described in Chapters 3 and 5 because plots had been either permanently marked or sufficient original plot markers remained to accurately relocate any missing plots. In both Chapters 3 and 5, the original methodologies were utilised and have consequently provided robust data that revealed some striking changes. The investigation reported in Chapter 6, whilst not strictly a revisitation study did make use of a pre-existing experimental set up, again highlighting the opportunities provided by long-term datasets and the importance of preserving both the data and the experiment set up in the field, such as exclosures upon which such studies are dependent. Legitimate problems do exist with revisitation surveys but the information they provide far outweighs these shortcomings and provides valid scientific information to investigate long-term change well beyond the length of typical monitoring studies as demonstrated by the current research.

Constraints on sample collection and existing data also put limits on the analysis that could be carried out on the soil samples. A lack of stratification within the soil between horizons may have masked some of the changes that have occurred. Further stratification may have made it possible to observe leaching of cations from the surface through to the deeper soils. When utilising existing data sets, constraints such as these pose problems that often cannot be overcome. Analysis of additional soil variables may have revealed more information on the changes that have occurred between the survey periods. Whilst this does not negate the valuable information that can be gained from utilising historic datasets it does highlight the importance of appropriate archiving of samples to allow for future analyses.

An additional problem with revisitation studies is the lack of information about the changes that have occurred in the intervening period between surveys. Where only two time points of recording exist, no conclusions can be drawn about the changes in the intervening periods. Vegetation or soil changes are unlikely to have occurred monotonically in the intervening period, and assumptions about the changes can only be made with evidence from other studies. However, the data presented in this thesis can provide information if, in the future, further revisitation of these study sites is undertaken.

7.2.2 Main driver of environmental change in Snowdonia

7.2.2.1 Primary driver of change in soil chemistry

The results of each of the investigations presented in this thesis were remarkable consistent. In agreement with other studies (Emmett *et al.*, 2010), the results reported in Chapter 3 appeared to indicate a recovery in soil acidity as pH had increased. Whilst data in Chapter 4 indicated little change had occurred, this may have masked an initial decline and subsequent recovery which would be in agreement with the data presented in Chapter 3. However, evidence from Chapters 3 and 4 appear to suggest that recovery from soil acidification is occurring (or occurred) whilst the results of Chapter 5 suggests that this increase has been far smaller than the acidification that occurred prior to the mid 1960s. This incomplete recovery of acidified soils is consistent with predictions from process models such as MAGIC (Wright *et al.*, 2005). The data from Chapter 5 pre-date a number of surveys, such as the UK Countryside Survey (Emmett *et al.*, 2010), which have produced evidence of a recovery in soil pH. These data also emphasise an important point in analysis of trends over time, in that the conclusions drawn will depend on the period of time that is selected for investigation.

The investigation described in Chapter 5 studied whether land-use mitigated the effects of other drivers of environmental change but found no significant difference between the soil pH of the different grazing treatments, suggesting grazing was not the main driver of any observed changes. However, the results of Chapter 6 did suggest that land-use may play a role in mitigating future changes in the atmospheric climate as there was evidence of N saturation and increased leaching of NO_3^- in un-grazed plots after 52 years compared to grazed plots, which also correlated with soil pH. This highlighted the importance of considering all possible drivers of environmental change in combination.

The decline in soil pH identified in Chapter 5 coincides with the peak in S deposition that occurred around 1970. With a lack of recovery in total soil exchangeable base cation concentration (Chapter 3) and only a limited recovery in soil pH (Chapter 5), the evidence presented here suggests that the main driver of the changes identified within the soils of Snowdonia over the past fifty years is acidification, most likely from historic S deposition.

7.2.2.2 Primary driver of change in vegetation composition

Evidence from the vegetation composition within the different chapters, showed a high degree of consistency over the main driver of vegetation change in Snowdonia. The results in Chapter 3 indicated that the vegetation on Snowdon is still showing evidence of degradation with lower species richness and an increase in the grass: forb ratio in the resurvey. This correlated well with the results from Chapter 5 which also showed a decline in species richness. Whilst many studies are reporting a negative effect of N deposition on species richness (Stevens *et al.*, 2004; Maskell *et al.*, 2010), the evidence presented in Chapter 5 appears to suggest that this loss of species richness is not a recent occurrence, and has in fact been occurring steadily since the 1970s. Unlike studies that investigate changes along gradients of environmental change drivers, long term data from the same area allows identification of the time when species richness started to decline.

Different components of the vegetation community respond at different rates to reductions in S deposition. The results in both Chapters 3 and 5 suggested that whilst bryophytes appear to have shown a recovery from S deposition, with a recovery in EbR value and species richness, vascular plants are still negatively affected with reduced species richness and more acidic EbR values. As vascular plants respond to soil abiotic conditions and bryophytes respond more directly to atmospheric conditions, this lagged response of vascular plants is important to consider because they will show evidence of past rather than present atmospheric conditions.

Whilst grazing is expected to exert large effects on vegetation composition, little difference in species richness and EbR value was found between grazing treatments in Chapter 5. Chapter 6 did provide evidence that the composition of vegetation will possibly play a role in mitigating future increases in N deposition, but it does not appear to have been the main driver of the changes identified in Chapters 3 and 5. Although identified as a major driver of vegetation change globally, the data presented within this thesis did not identify any changes that could be related to changes in climate. Changes in phenological timing is one of the simplest measures to detect effects of changes in climate and it was not possible to detect this using the data presented within this thesis.

Whilst it is difficult to disentangle the effects of S and N deposition due to correlations in deposition, the evidence presented in this thesis suggested that many of the negative effects on vegetation composition which have been attributed to N deposition (Stevens *et al.*, 2004; Maskell *et al.*, 2010), may in fact be as a result of historic S deposition. Even though controls on S emission were so effective at reducing S deposition, there remains a large, residual effect of acid deposition on this montane environment. It appears that the main driver of the changes identified within the vegetation on Snowdonia over the past fifty years is acidification, most likely from S deposition based on the evidence presented in Chapters 3 and 5.

Overall, both the soils and vegetation indicate that the main driver of the changes identified throughout the preceding chapters is acidification, again most likely as a result of historic S deposition.

7.3 Further work

Despite extensive research that has been undertaken on the effect of specific drivers of environmental change on biodiversity, vegetation composition and soil chemistry there is still a scope for further work to confirm these environmental interactions.

The dataset utilised in Chapter 5 is unique in that it provides long-term data on the changes that have occurred in both the vegetation and soil (with and without grazing) over such a long time period. The collection of soil solution data for these areas and additional soil analyses in Chapter 6 provide an opportunity to undertake some modelling analysis. Currently modelling is being used to support European pollution abatement policy with the use of simple soil acidification and nutrient cycling models (e.g.

MAGIC) combined with field-based empirical relationships with plant species responses (e.g. GBMOVE) (De Vries *et al.*, 2010). Within Europe, integrated soil-vegetation models are used to predict plant species composition as a function of atmospheric deposition of N and acidity. The principle behind this is that dynamic soil modelling (e.g. MAGIC) predicts the changes in soil acidity and water nutrient status whilst the statistical model (e.g. GBMOVE) predicts the vegetation succession or changes in plant species composition in response to the changes in water, nutrient and acidity status using plant species-specific information on habitat preferences (De Vries *et al.*, 2010). The availability of datasets, as presented here, would provide useful evidence of the validity of the model predictions as well as assist development of critical loads for atmospheric deposition of pollutants.

Whilst climate change is considered to be a major global driver of environmental change, no evidence was found within this research for localised effects on Snowdon. This is not to say that Snowdon is not responding, but the nature of the data did not provide the opportunity to adequately investigate this aspect. The most easily detectable effect of climate change on vegetation is the earlier flowering times of the plants (Walther *et al.*, 2002). The data from the studies presented here were always collected at the same time of year (Summer) and no data were collected on flowering time, preventing detection of this effect. The ECN network does collect data on phenology however, and has revealed a progressively earlier flowering time for a number of plant species on Snowdon suggesting that climate is driving some changes. Investigating the spatial distribution of plant species occurrence, particularly montane specialists, may reveal evidence of an upward altitudinal shift of plant species in response to climate change. Using statistical modelling such as that suggested by Damgaard (2009) may enable detection of climate driven changes that would be useful to provide evidence of the effects of climate change in Snowdonia.

7.4 Summary

Over the last 50 years, Snowdonia has undergone large changes in the major environmental drivers of soil chemistry and vegetation composition. However, the major driver of the trends identified were changes in atmospheric deposition of pollutants, particularly S. Evidence from the soils, vascular plants and bryophytes suggest different degrees of environmental degradation and recovery. This highlights the importance of assessing the ecosystem as a whole, as each part contributes complementary information on the state of the environment and portrays a more accurate picture than considering components in isolation. Whilst some soil chemistry parameters (e.g. soil pH) may indicate signs of recovery, others (e.g. total soil exchangeable base cation concentrations) may take far longer to recover but have important implications for vegetation composition. Vascular plants respond more slowly to changes in atmospheric chemistry as they are more reliant on soil chemical conditions, compared to bryophytes that respond more rapidly and are more reliant on atmospheric conditions. This lagged response of vascular plants is important to consider because they will be presenting evidence of past rather than present atmospheric conditions.

Future changes in atmospheric deposition of pollutants, particularly N, may be mitigated through appropriate management of land-use and grazing intensity. The rate of recovery of ecosystems to stressors is currently unknown, and despite strict controls and successful reductions in previous major pollutants, a lack of recovery is evident. Vegetation is already displaying a negative response to current levels of stress, and with climate change projected to intensify, and deposition of reactive N predicted to increase, the resilience of already degraded ecosystems to further perturbation is uncertain. Further field-based studies which examine the interactive effects of multiple drivers of environmental change over long periods of time are required to fully understand the implications of future changes in soil chemistry, vegetation composition, biodiversity and ultimately ecosystem functioning. This in turn will allow conservation management strategies to be developed to mitigate future changes in the environment.

7.5 References

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