

The impact of coniferous afforestation and deforestation on the chemical properties of soil

BSc Geography



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200463461

Word count: 9828

Acknowledgements

With thanks to Gareth Browning of the Forestry Commission, and Judith and Andy Weston, tenant farmers of the National Trust land adjacent to Ennerdale forest, for granting me permission to collect soil samples. Many thanks to all laboratory staff for their help and advice when analysing the soil. Thanks also to my dissertation tutor, Pippa Chapman for all her valuable help and support, my DSG, and my parents for helping me collect all 50 soil samples!

Abstract

This study was undertaken in Ennerdale, Cumbria, in the west of the Lake District, UK. The rewilding scheme 'Wild Ennerdale' ultimately aims to return previously coniferous afforested land back to its semi-natural wild origins, which in turn should improve deteriorated soil quality. Much of the UK upland moorlands were afforested with fast growing, high yielding exotic coniferous tree species, such as Sitka spruce, to meet the high demand for timber during the First World War. This led to a significant reduction in native, slower growing deciduous tree species. To establish whether an improvement in the chemical properties of soil is gained during the initial stages of 'Wild Ennerdale', the effects of both coniferous afforestation and deforestation are to be examined, with a moorland site in Ennerdale acting as a control site. Soil acidity, comprising of soil pH and exchangeable acidity; exchangeable base cations calcium, magnesium, potassium, and sodium; and available nitrogen, are the soil chemical properties specifically examined. All chemical properties (bar exchangeable magnesium and sodium), were found to decrease from their initial state after coniferous afforestation. Coniferous deforestation was then found to exacerbate these poor soil conditions, despite an overall increase in soil pH. Overall, it could be established that after a period of approximately ten years, coniferous deforestation in Ennerdale actually led to the deterioration of soil quality. Coniferous deforestation is only the first phase of 'Wild Ennerdale', however, so the effect the reforestation of native, broadleaf deciduous species has on soil chemical properties should be examined in the future. This would truly establish whether 'Wild Ennerdale' is in fact improving soil quality in terms of its soil chemical properties. This knowledge would be essential in providing reasoning for the initiation of similar schemes to take place in the future.

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1. Introduction

1.1 Rationale

1.1.1 Moorland

Much of the UK consists of upland moorland areas, which resemble to the highest extent, what the landscape would have looked like centuries ago, before any type of afforestation began. The majority of moorland areas in the UK are located in the north and west (BBC, 2012), which today are most commonly used for sheep grazing (Cumbria Hill Farming, 2008). UK upland areas receive high rates of precipitation, contributing to their acidic nature by adding and maintaining the supply of hydrogen ions to the soil (Reynolds *et al.*, 1988). Acidic soils can be identified at a glance, by the presence of acid loving plants such as heather and bilberry. Both coniferous afforestation and deforestation lead to the disturbance of moorland soil, profoundly changing soil properties, and subsequently affecting the whole ecosystem, including both its flora and fauna (Smal and Olszewska, 2008; Macdonald *et al.*, 2009). Although there are very few completely natural soils remaining on Earth, due to the widespread reoccurring change in land use, upland areas of the UK best represent semi-natural soil conditions (Gerrard, 2000), and so are often used as control sites in similar studies.

1.1.2 Afforestation

The UK is covered by approximately three million hectares of forest, equating to about 12% of the UK's land cover, making it 'one of the least densely forested countries in Europe' (Forestry Commission, 2012). Of this, roughly 1.7 million hectares are coniferous forest, with the remaining 1.3 million hectares comprising of deciduous forest (Forestry Commission, 2012).

Due to the high demand of wood during the First World War, mainly for the production of boats, UK timber stocks rapidly depleted (Convery and Dutson, 2008). Native tree species, such as oak, alder, and birch, would have been the main source of wood, dominating the UK landscape from as early as Mesolithic times (Wild Ennerdale, 2006(a)). It was only after the First World War ended that forest management began (Fisher and Binkley, 2000; Richter and Markewitz, 2001). For example, during the 1920s,

due to the resulting widespread clearance of deciduous forest, the newly established Forestry Commission, rapidly acquired and managed previous upland grassland sites across the UK as coniferous plantation forests (Neal *et al.*, 1992; Wild Ennerdale, 2006(a); Convery and Dutson, 2008; Berthrong *et al.*, 2009). This rapid increase in coniferous afforestation was dominated by the plantation of the faster growing, high yielding, exotic coniferous tree species such as Sitka spruce, which would have quickly become dominant (Neal *et al.*, 1992; Convery and Dutson, 2008). By the mid 1990s, there had been a dramatic increase in coniferous forest cover not only in the UK, but throughout the world (Fisher and Binkley, 2000; Macdonald *et al.*, 2009). Despite the Forestry Commission's aim to substantially increase timber production and sustain yields for the future, coniferous afforestation has consequently led to significant changes in soil chemistry across an ecosystem level (Reynolds *et al.*, 1988; Oxbrough *et al.*, 2006; Convery and Dutson, 2008).

1.1.2.1 The effect of coniferous afforestation on the chemical properties of soil

Soil acidity; soil pH and exchangeable acidity

Land use change leads to extensive alterations in the chemical properties of soil, particularly soil acidity (Strobel *et al.*, 2001; Olszewska and Smal, 2008; Macdonald *et al.*, 2009). Although much of the UK upland environment is already acid sensitive, acute changes in vegetation cover have been found to significantly increase soil acidity (Neal *et al.*, 1992; Grieve, 2001). In turn, this acidification has affected numerous soil chemical properties, such as the regulation and subsequent availability of essential elements, directly influencing plant growth (Cresser *et al.*, 1993; Ashman and Puri, 2002; Brady and Weil, 2008).

The presence of a coniferous tree canopy has led to considerable acidification and long term changes in soil chemistry, due to the capture of atmospheric pollutants (Neal *et al.*, 1992; Forest Authority, 1998; Grieve, 2001; Richter and Markewitz, 2001; Berthrong *et al.*, 2009). The scavenging of acidic pollutants leads to their absorption by trees, and storage in soil, where they might subsequently be leached into the groundwater, resulting in significant implications for water quality too (Forest Authority, 1998; Ashman and Puri, 2002; Brown *et al.*, 2010). The widespread input of highly acidic pine needles onto the forest floor creates an extremely acidic litter, aiding soil acidification (Neal *et al.*, 1992; Fisher and Binkley, 2000; Gerrard, 2000). The weathering of minerals can also contribute

to soil acidity (Strobel *et al.*, 2001). The more acidic the soil, the higher the concentration of hydrogen ions, which in turn causes the release of aluminium ions, highly toxic to both plants and animals (Goulding and Stevens, 1988; Rowell, 1994; Brady and Weil, 2008). Soil acidity is strongly related to other soil chemical properties, predominantly the regular leaching of soil minerals (Cresser *et al.*, 1993).

Exchangeable base cations

Coniferous afforestation often takes place on poor, particularly wet soil such as peat or gley, whereas better soil is used for agricultural crop production (Forest Authority, 1998; Fisher and Binkley, 2000). As such, forest soils often contain many stones, which although act as a major primary source of nutrients when weathered, also increase drainage due to the formation of large pore spaces (White, 1997; Fisher and Binkley, 2000; Gerrard, 2000; Richter and Markewitz, 2001). The weathering of rock provides by far the largest source of base cations to the soil, however, the presence of a tree canopy again leads to the capture of atmospheric cations which also contribute to base cation reserves in the soil, accumulating over time (Johnson and Lindberg, *in* Adriano and Havas, 1989; Richter and Markewitz, 2001; Berthrong *et al.*, 2009). The often steep slopes on which afforestation may take place can also influence drainage rates, leading to the leaching of base cations out of a soil, enhanced by soil acidity which ultimately dominates soil chemistry and the processes occurring within the soil (Reynolds *et al.*, 1988; Cresser *et al.*, 1993; Fisher and Binkley, 2000).

Nutrient cycling processes give soil under forests unique characteristics compared to other soils (Fisher and Binkley, 2000). Deep tree roots are important for nutrient uptake, transporting and recycling nutrients to upper soil horizons, where a high demand for base cations is essential for tree growth and survival (Forest Authority, 1998; Fisher and Binkley, 2000; Grieve, 2001; Berthrong *et al.*, 2009). Tree roots can also be beneficial in stabilising the soil by providing physical support, and combined with the accumulation of leaf litter and consequent production of organic matter, tend to prevent the erosion of soil, in turn conserving base cations (Forest Authority, 1998; Fisher and Binkley, 2000).

Available nitrogen

The main source of nitrogen to the soil is from atmosphere, where firstly it has to be 'fixed' into a form available to plants (White, 1997; Gerrard, 2000). Mineralisation is the

process by which nitrogen is converted from its organic form, to inorganic forms that can be readily taken up by plants (Bardgett, 2005; Chapman, *in* Holden, 2008). Plants therefore take up nitrogen in the forms of ammonium and nitrate, however, nitrite is toxic to plants even at small concentrations (Brady and Weil, 2008). The process of nitrification (converting ammonium (NH_4^+) to nitrite (NO_2^-) and finally to nitrate (NO_3^-)), has been found to lead to the acidification of soil, via the production of hydrogen ions (Richter and Markewitz, 2001; Bardgett, 2005; Brady and Weil, 2008). Usually the more acidic the soil, the lower the supply of macronutrients (Cresser *et al.*, 1993; Chapman, *in* Holden, 2008). The rate at which nitrification occurs, is controlled by important soil characteristics such as aeration and pH. Nitrogen is an essential macronutrient required in large quantities by plants, which is why coniferous afforestation is known for the depletion of available nitrogen reserves within the soil (Rowell, 1994; Richter and Markewitz, 2001; Brady and Weil, 2008).

1.1.3 Deforestation

In tropical areas, deforestation of any forest type is very much recognised as having negative impacts on the environment, especially the soil, and significantly contributing to the increase in atmospheric carbon dioxide, and therefore global warming (Saikh *et al.*, 1998; Malhi *et al.*, 2002). Although overall decreasing since the beginning of the century, deforestation rates of the Amazon rainforest for example, still resulted in a loss of around 6,000 square kilometres of forest between 2000 and 2006 (Butler, 2010). Despite deforestation contributing largely to global warming in the tropics, this study considers solely the effects of UK coniferous deforestation on the chemical properties of soil.

In the UK, coniferous deforestation has substantially increased in recent decades. With a reduction in the demand for UK timber supplies, due to the high production from European countries, there is less pressure to produce timber, so fast growing coniferous species need not dominate UK forests (Wild Ennerdale, 2006(a)). This has allowed rewilding schemes such as 'Wild Ennerdale' to take place, reintroducing more natural, native deciduous species to the UK landscape in an attempt to return it to its original 'wild' roots, and in turn improve soil conditions (Wild Ennerdale, 2006(a)).

Coniferous deforestation is likely to increase further over the next decade, particularly in areas such as mid Wales and Scotland, as coniferous afforestation plantations which took place post Second World War are reaching peak maturity (Anderson *et al.*, 1990; Neal *et al.*, 1992). An increase in the desire for recreational activities within forests has also prompted the change in forest type composition, from the uniform coniferous plantations, to the more natural, slow growing picturesque deciduous forests (Wild Ennerdale, 2006(a)).

1.1.3.1 The effect of coniferous deforestation on the chemical properties of soil

Soil acidity; soil pH and exchangeable acidity

The removal of trees has been recognised to decrease soil acidity, due in large, to the dramatic reduction in the capture of acidic atmospheric pollutants, and organic matter content (Neal *et al.*, 1992; Cresser *et al.*, 1993; Smal and Olszewska 2008). This increase in soil pH should subsequently reduce the amount of aluminium ions in the soil, and technically lead to an increase in exchangeable base cations, because aluminium will no longer dominate cation exchange sites (Rowell, 1994). However, other more dominant processes occurring in the soil often lead to differing results.

Exchangeable base cations

Despite coniferous deforestation leading to a reduction in soil acidity, and so increasing the potential for more exchangeable base cations, other factors such as leaching play a huge role in the loss of exchangeable base cations in the soil (Gerrard, 2000). As deforestation leaves the soil exposed, precipitation leads to significant amounts of leaching, which combined with a reduction in leaf litter, and therefore organic matter content, result in the loss of exchangeable base cations from a soil (Gerrard, 2000; Ashman and Puri, 2002). The harvesting of trees has the most significant effect on exchangeable base cation reserves in the soil, as a significant amount are stored in the wood of trees, so are removed when the tree is (Reynolds and Stevens, 1998).

Available nitrogen

Again, uptake by plants and leaching play a huge role in the loss of available nitrogen from the soil. Despite the ability of positively charged ammonium ions (NH_4^+) being held on cation exchange sites in the soil, the negatively charged nitrate (NO_3^-) and nitrite (NO_2^-)

ions are not, making them highly susceptible to leaching (Cresser *et al.*, 1993; Rowell, 1994; Bardgett, 2005; Brady and Weil, 2008).

1.1.4 Summary

In recent decades, the realisation that coniferous afforestation leads to a deterioration in soil quality has been widely recognised. The initiation of rewilding schemes such as ‘Wild Ennerdale’, ultimately aim to undo the effects of coniferous afforestation, by firstly deforesting coniferous plantations, and eventually reforesting these areas with natural, native deciduous species, in an attempt amongst others to improve soil quality. This study primarily aims to analyse whether the initial stages of ‘Wild Ennerdale’ are working, by accessing the effects coniferous deforestation has on the chemical properties of soil. This is imperative in recognising the success of this rewilding scheme, and possible existence of schemes like this in the future. Previous studies have rarely examined the effects of both coniferous afforestation and deforestation within the same area, which also makes this study somewhat unique.

1.2 Aim

To determine the impact coniferous afforestation and deforestation has on the chemical properties of soil.

1.3 Objectives

- To evaluate the effects of coniferous afforestation and deforestation on soil acidity; soil pH and exchangeable acidity.
- To examine the effects of coniferous afforestation and deforestation on exchangeable base cations; calcium, magnesium, potassium, and sodium.
- To determine the effects of coniferous afforestation and deforestation on available nitrogen.

2. Methodology

2.1 Study area

Soil samples were collected from three differing sites in Ennerdale. Ennerdale lies in the west of the Lake District National Park, northwest England (*Fig. 2.1; 2.2*) (Fryer, 1981), partly owned by the Forestry Commission. Its remote valley, which gradually narrows from west to east, spans a length of nine miles, has a widest point of three and a half miles, and covers a total area of 4,711 ha (Wild Ennerdale 2006(b); Convery and Dutson, 2008). Some of the Lake District's highest summits surround Ennerdale Valley, for example; 'Green Gable, Great Gable, Pillar, Kirk Fell and Steeple' (Convery and Dutson, 2008; Cumbria Hill Farming, 2008). Ennerdale has an altitudinal range of 100 - 900 m above sea level, and receives high levels of rainfall per year (1524 - 1778 mm) (Cumbria Hill Farming, 2008). The underlying geology of Ennerdale is made up of the 'Skiddaw Slates Group of Ordovician mudstones and siltstones' (Ennerdale Historic Landscape Survey, 2003). Sheep grazing governs the upland areas of Ennerdale, whereas forest cover dominates lower altitudes (Convery and Dutson, 2008; Cumbria Hill Farming, 2008). Ennerdale Water is located in Ennerdale Valley (*Fig. 2.1*); a two and a half mile long by one mile wide lake, with a deepest point measuring 44.81 metres (Wild Ennerdale, 2006(b); Cumbria Hill Farming, 2008). Much of Ennerdale is designated as either a Site of Special Scientific Interest or Special Area Conservation (Convery and Dutson, 2008).



Figure 2.1. View of Ennerdale, with Ennerdale Water visible to the right of the photograph (taken during the preliminary visit).

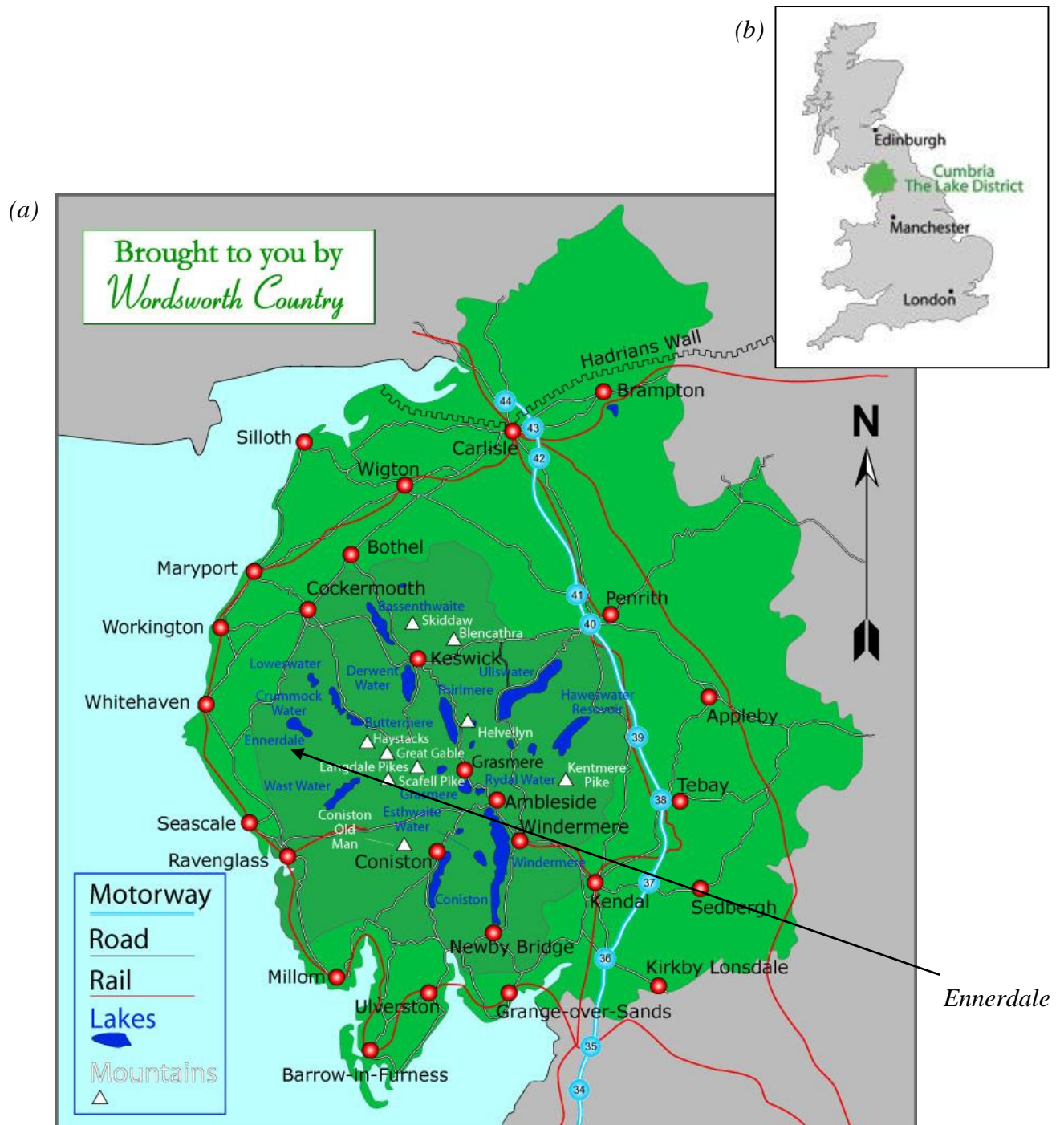


Figure 2.2. (a) Map to show location of Ennerdale within Cumbria, (b) location map to show Cumbria within the UK (Wordsworth Country, 2012).

Coniferous afforestation of Ennerdale began in 1925 rapidly increasing in subsequent decades, and in turn significantly deteriorating soil quality (Wild Ennerdale, 2006(a); Convery and Dutson, 2008). Since the Forestry Commission became partners with the National Trust and United Utilities, focus has been placed on the rewilding initiative ‘Wild Ennerdale’, set up in 2002 with a primary aim to produce ‘a more naturally evolving landscape’ (Convery and Dutson, 2008). ‘Wild Ennerdale’ ultimately intends to, in the long run, reforest previously coniferous afforested areas with native deciduous species that

would have once dominated the area, in an attempt amongst others to improve soil quality, the aesthetic landscape, and water quality, as forested systems are tightly coupled with river ecosystems (Convery and Dutson, 2008). This has led to a shift in forest type dominance within Ennerdale, from numerous highly managed coniferous forested areas, to increasing areas of natural native broadleaf forests (Wild Ennerdale, 2006(a)). Not only does ‘Wild Ennerdale’ seek to rewild the landscape, but also to ‘provide socio-economic benefits for the local community’ (Convery and Dutson, 2008).

2.2 Study sites

To determine the impact coniferous afforestation and deforestation has on the chemical properties of soil, three differing sites within Ennerdale were used; a moorland, a forest, and a deforested area. Sites were selected after a preliminary visit was undertaken, and after correspondence from Gareth Browning of the Forestry Commission. The moorland site, located next to Ennerdale forest (*Fig. 2.3; 2.4(a)*), acted as a control site representing initial soil conditions before coniferous afforestation took place (owned by the National Trust, tenanted by farmers Judith and Andy Weston). The forested site (*Fig. 2.3; 2.4(b)*) was used to identify the effects coniferous afforestation has had on the soil, and finally, the deforested site (*Fig. 2.3; 2.4(c)*) was used to identify the effects coniferous deforestation has had on the soil (both latter sites owned by the Forestry Commission). Site descriptions and abbreviations are classified in *Table 2.1*.

Table 2.1. Site descriptions and classification.

Site abbreviation	Site type	Site description
S1	Moorland	Predominantly grassland, with the presence of billberry, and some heather (<i>Fig. 2.4(a)</i>).
S2	Forest	Coniferous spruce forest made up of predominantly Sitka spruce (<i>Fig. 2.4(b)</i>).
S3	Deforested	Deforested coniferous spruce forest. Felling took place in approximately 2000 (<i>Fig. 2.3</i>). Tree stumps and branches were still present, left to decompose in situ (<i>Fig. 2.4(c)</i>).

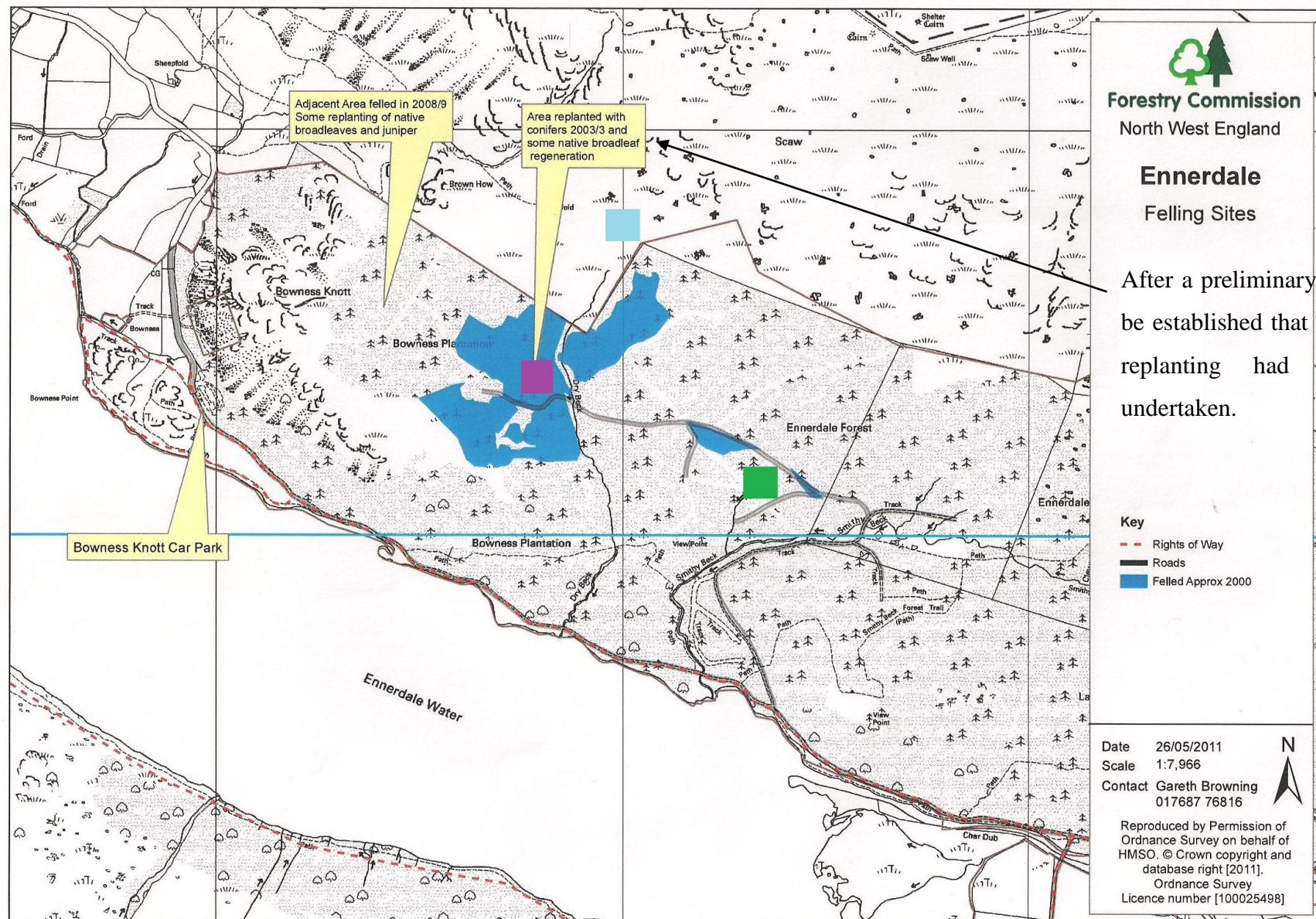


Figure 2.3. Map from Gareth Browning of the Forestry Commission, to show deforested areas within Ennerdale. Additional coloured squares represent the three sites; Moorland Forest Deforested (not to scale).

(a)



(b)



(c)



Figure 2.4. Photographs of each site (taken during the preliminary visit when weather conditions were better);(a) moorland (centre of photograph), (b) forest, and (c) deforested.

2.3 Soil collection

After a preliminary visit to Ennerdale was undertaken, soil sampling was carried out on September 11th 2011. At each site, fifteen soil samples were collected at a depth of approximately 0.1 m (which encompassed both the organic and mineral horizons), using a bulb planter. Prior to soil collection, a sampling strategy was developed to include a representative area of each site (Macdonald *et al.*, 2009). A fifteen metre by fifteen metre sampling grid was drawn for each site; dividing the sampling area into 225 individual one metre by one metre squares, each square numbered chronologically (*see Appendix 2*). Before collection, twenty five random numbers (between one and 225) were generated for each site using Microsoft Excel. The first fifteen numbers were to be used as the squares at which a soil sample would be collected; with an extra ten numbers to be used in case soil collection could not be taken at a square due to obstruction (i.e. the presence of a tree or large rock) (*see Appendix 2*). For consistency, soil samples were taken from the bottom left hand corner of each selected square. Once collected, individual soil samples were stored in airtight bags and kept in a refrigerator whilst awaiting further analysis.

2.4 Lab analysis

Each soil sample was divided roughly into three; one third kept refrigerated (for analysis that required field moist soil), and two thirds dried (as the majority of analysis was undertaken on dry soil). Soil samples were dried at 40°C in the oven for about ten days, (due to soil being extremely wet when collected). Each sample was then ground and passed through a two millimetre sieve. Soil texture was determined using the ‘feel method’ (Chapman, *in* Holden, 2008) (*see Appendix 1 for details*).

2.4.1 Analysis using field moist soil

Available nitrogen (ammonium (NH₄), nitrate (NO₃) and nitrite (NO₂)), was determined using field moist soil, with 1 M KCl used for extraction, at a ratio of 1:5 moist soil:solution (Rowell, 1994). An AAS (atomic absorption spectrometry) machine was used for analysis, recording NH₄, NO₃ and NO₂. Results were then converted to mg/kg⁻¹ of soil.

2.4.2 Analysis using dry soil

Soil pH was determined in CaCl₂ solution, ‘to minimise calcium release from the soil exchange complex’, using a 1:2.5 soil:solution ratio (Rowell, 1994; Gerrard, 2000). Soil pH is defined on a negative logarithmic scale, expressing the concentration of hydrogen ions in a solution ($\text{pH} = -\log \{H^+\}$) (Gerrard, 2000; Ashman and Puri, 2002; Cresser *et al.*, 2003). Exchangeable acidity is a measure of the exchangeable aluminium and hydrogen ions present in a soil solution (Brady and Weil, 2008). It was determined using titrations, where extraction initially took place with 1M KCl using a 1:10 soil:solution ratio (Rowell, 1994). Results were converted to cmol kg^{-1} . Exchangeable base cations; calcium, magnesium, potassium and sodium, were determined by analysis using an ICP-AES (inductively coupled plasma atomic emission spectroscopy) machine, after extraction at a 1:25 soil:solution ratio using 1M ammonium chloride (as soil pH was below 5) (Rowell, 1994). Results were converted to cmol kg^{-1} . Base saturation could then be calculated (with all units in cmol kg^{-1}) using the following formula (Brady and Weil, 2008);

$$\text{Base saturation (\%)} = \frac{\sum \text{Ca} + \text{Mg} + \text{Na} + \text{K}}{\sum \text{Ca} + \text{Mg} + \text{Na} + \text{K} + \text{H} + \text{Al}} \times 100$$

Moisture content was calculated after soil was oven dried for 24 hours, in crucibles at 105°C. Loss on ignition (LOI) could then be determined after the crucibles were put in a muffle furnace at 500°C for a further 24 hours, to burn off remaining water and organic matter (Rowell, 1994). For each laboratory test, all samples were mixed and analysed randomly to ensure no bias.

2.5 Statistical analysis

To determine the correct statistical analysis test to use for either a significant difference or relationship between data, normality of the data was first established using the Anderson-Darling test, which tests whether the data set follows a normal distribution (Dytham, 2003). The probability that the set of data is normally distributed is represented by the *P*-value. If $P < 0.05$ the data is not normally distributed, it is significantly different to a normal distribution, so a non-parametric test should be used (Fowler *et al.*, 1998; Dytham, 2003). Data transformations were undertaken to see if data could be normalized, and

therefore allow for a parametric, more accurate test to be used (Fowler *et al.*, 1998). This was not observed however, so original data was used in all statistical testing.

In testing for a significant difference between data sets at the three sites, either a parametric one-way ANOVA test (normally distributed data), or a non-parametric Kruskal-Wallis test (not normally distributed data) was used. A one-way ANOVA (analysis of variance) tests if the means of ≥ 3 categorical groups, in this case the three sites, are significantly different from each other (Dytham, 2003; Ennos, 2007). A significant result ($P < 0.05$), at a 95% confidence level, indicates that at least one pair of groups has significantly different means, however, it is not stated which groups this is, so a post hoc test may then be undertaken (Dytham, 2003; Ennos, 2007). Fisher's pairwise comparison post hoc test identifies the pairs of sites at which a significant difference between means is observed (Dytham, 2003). The non-parametric equivalent of a one-way ANOVA test is the Kruskal-Wallis test, which instead compares the medians of ≥ 3 categorical ranked groups (Fowler *et al.*, 1998; Dytham, 2003). A P -value < 0.05 again indicates a significant difference between at least one pair of groups, however, it is not identified which pair. As there is no post hoc test available for Kruskal-Wallis, a pairwise Mann-Whitney U test was used, which compares the medians of two ranked groups only (Dytham, 2003; Ennos, 2007). In this case, Site 1 was paired with Site 2, Site 2 with Site 3, and Site 3 with Site 1 (*see Table 2.1 for site classification*). A significant difference between group medians is indicated by a P -value < 0.05 .

In testing for a significant correlation between continuous variables, either a parametric Pearson's Product Moment Correlation (normally distributed data), or a non-parametric Spearman's Rank test (not normally distributed data) was used. Pearson's Product Moment Correlation is used to test for a correlation between two variables, the statistic r result ranging from -1 to 1, perfect negative correlation to perfect positive correlation, respectively (Dytham, 2003), where $P < 0.05$ indicates a significant correlation. Spearman's Rank was used if data were not normally distributed, where data sets were first ranked and then Pearson's Product Moment Correlation undertaken (Dytham, 2003). Again a P -value < 0.05 indicates a significant correlation, with an r value determining correlation strength. Statistical tests are summarised in *Table 2.2*. Although replicates for individual soil samples were taken (two per site), they were not used in statistical testing as replicate results proved not to differ greatly from original results. Relevant graphs were produced from statistical analysis.

Table 2.2. Summary statistical test table.

	Normally distributed data	Not normally distributed data
To test for a significant difference between all three sites.	ANOVA	Kruskal-Wallis
Post hoc test/equivalent, to identify which sites are significantly different from each other.	Fisher's F	Mann Whitney U test
To test for a significant correlation between variables.	Pearson's Product Moment Correlation	Spearman's Rank

3. Results

3.1 Basic soil characteristics and LOI

Site classification and basic soil characteristics are presented in *Table 3.1*. Soil texture is of extreme importance as it has a great influence on the chemical properties of soil, so should be considered when interpreting results (Ashman and Puri, 2002; Brady and Weil, 2008).

Table 3.1 Basic soil characteristics.

Site	Site number	Texture	pH	LOI
Moorland	Site 1 (S1)	Clay loam	3.41	74.18
Forest	Site 2 (S2)	Clay	3.23	80.61
Deforested	Site 3 (S3)	Sandy clay loam	3.34	21.28

Loss on ignition (LOI) again plays an important role in determining soil chemical properties. LOI varied between sites, being highest in the forest (80.61), and considerably lower in the deforested site (21.28) (*Table 3.1; Fig. 3.1*). Mann Whitney U tests revealed there to be a significant difference between S2 vs. S3 ($P < 0.0001$), and S3 vs. S1 ($P < 0.0001$), but not between S1 vs. S2 ($P = 0.0971$).

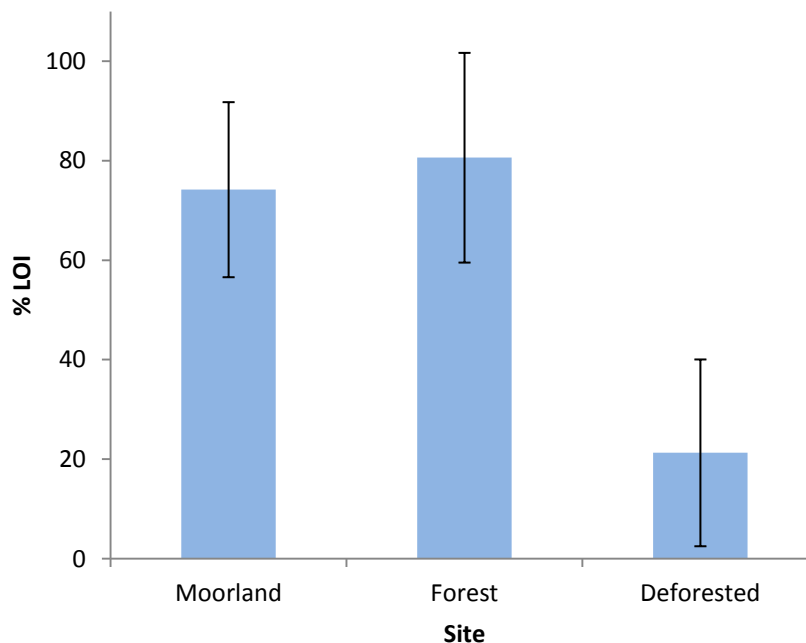


Figure 3.1 Average loss on ignition at the three study sites (error bars represent standard deviation, $N = 15$ per site). Moorland S1; Forest S2; Deforested S3.

3.2 Soil pH variations between sites

Soil pH varied between sites (*Fig. 3.2*), with the highest pH (3.41) observed in the moorland, and lowest in the forest (3.23). The moorland had the most variability between samples (0.63), compared to the forest and deforested sites, 0.42 and 0.37 respectively. It also had the largest median (3.44), compared to the forest (3.19) and deforested sites (3.27). After ANOVA, Fisher's post hoc test revealed a significant difference between S1 vs. S2 only.

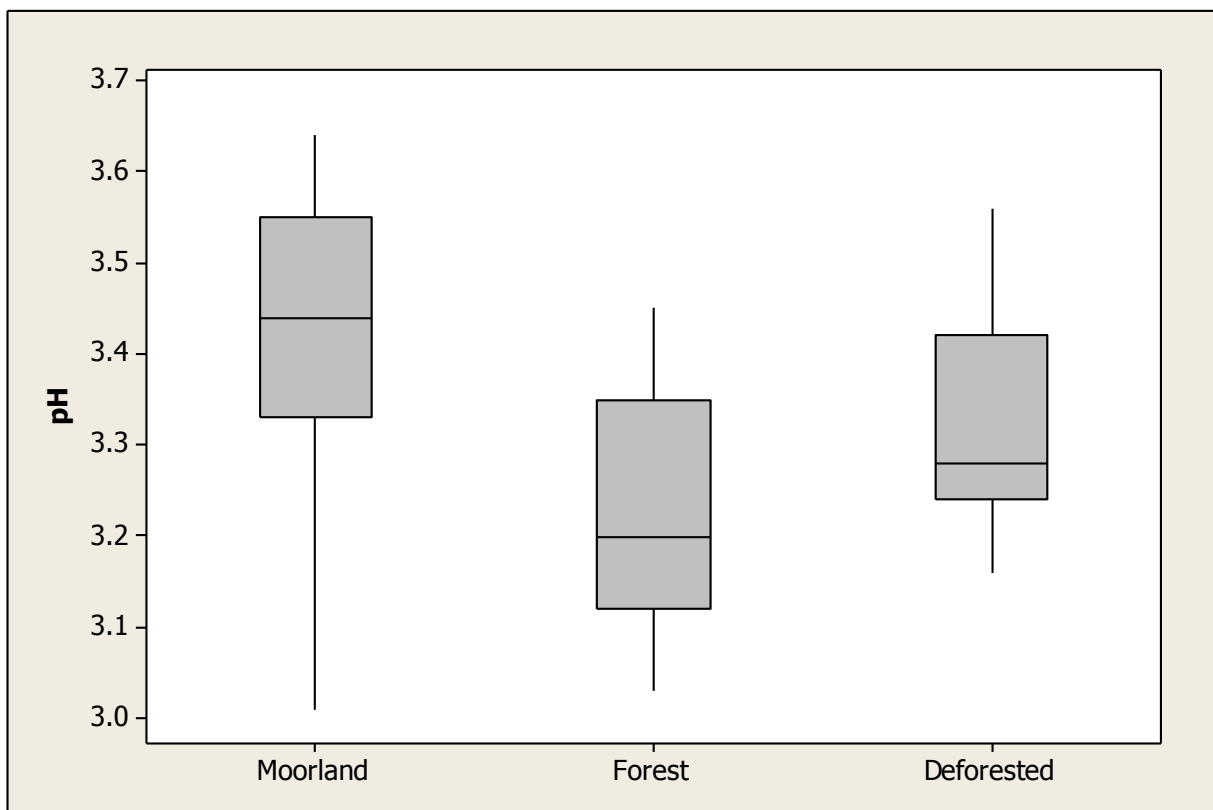


Figure 3.2 Box plot to show range, interquartile range and median (Dytham, 2003) for pH at the three study sites ($N = 15$ per site). Moorland S1; Forest S2; Deforested S3.

3.3 Exchangeable acidity variations between sites

Exchangeable acidity (exchangeable aluminium & exchangeable hydrogen ions) ranged from 11.69 cmol kg⁻¹ to 7.31 cmol kg⁻¹, moorland site to deforested site (*Fig. 3.3*). Mann-Whitney U tests revealed a significant difference between S1 vs. S2 ($P = 0.0055$) and S3 vs. S1 ($P = 0.0011$), but not between S2 vs. S3 ($P = 1.0000$).

Individual exchangeable aluminium varied the least between sites (3.60 cmol kg⁻¹). Again, S1 vs. S2 ($P = 0.0008$) and S2 vs. S3 ($P = 0.0001$) were found to be significantly different, with S3 vs. S1 posing no significant difference at $P = 0.4068$. Individual exchangeable hydrogen ions were also found to be significantly different between two sets of site pairs, however in this case, S2 vs. S3 ($P < 0.0001$) and S3 vs. S1 ($P < 0.0001$) were observed to be significantly different, but not S1 vs. S2 ($P = 0.5897$).

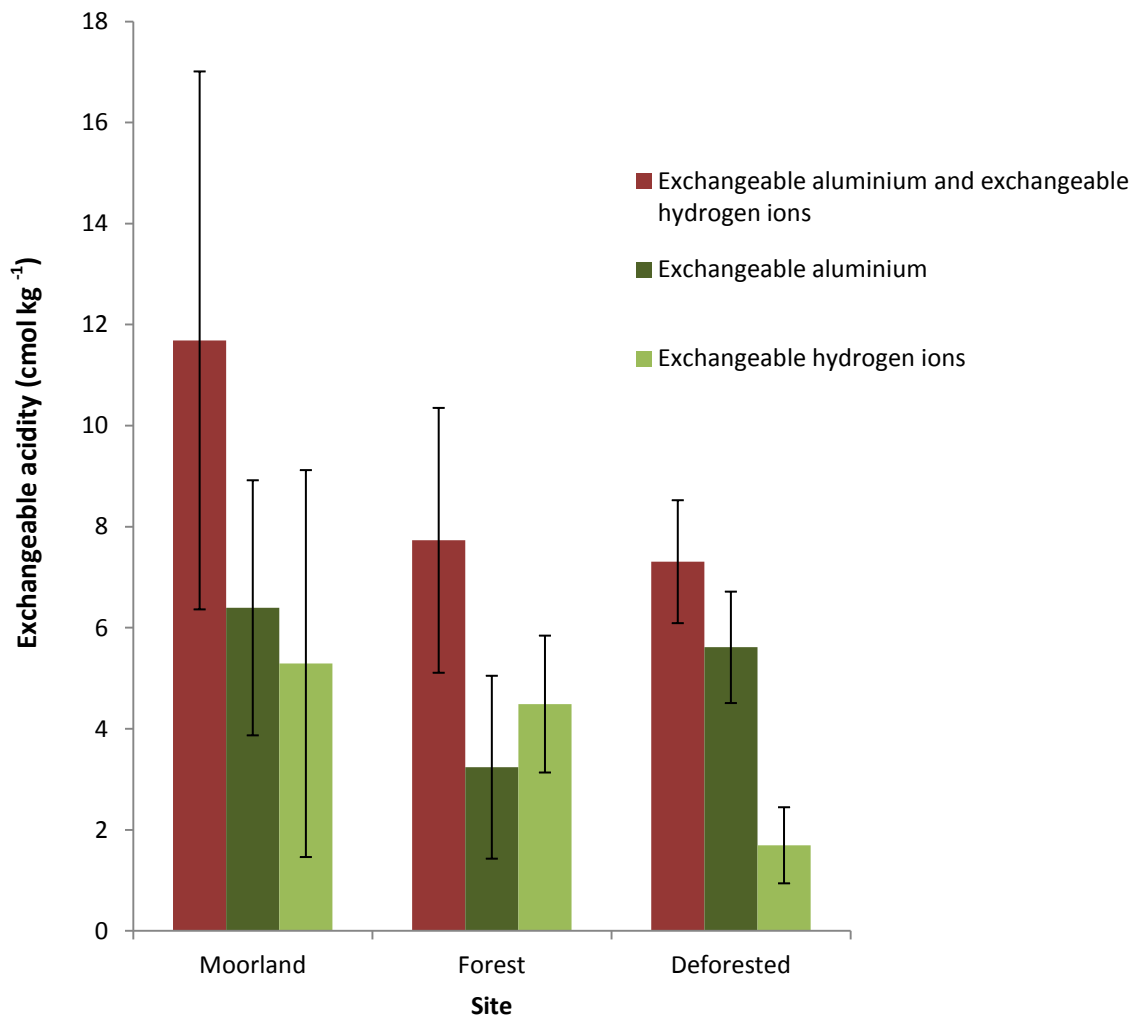


Figure 3.3 Average exchangeable acidity components at the three study sites (error bars represent standard deviation, $N = 15$ per site) Moorland S1; Forest S2; Deforested S3.

3.4 Exchangeable base cation variations between sites

Exchangeable calcium varied between sites (*Fig. 3.4*), with the moorland having the highest value of 3.3705 cmol kg^{-1} , and the biggest range between samples (5.67 cmol kg^{-1}), whereas the deforested site had the lowest value of 1.2730 cmol kg^{-1} , and the smallest range (3.00 cmol kg^{-1}). A significant difference was observed between; S2 vs. S3 ($P = 0.0005$) and S3 vs. S1 ($P = 0.0004$), when Mann Whitney U tests were carried out. S1 vs. S2 was not significant different ($P = 0.3401$).

Exchangeable magnesium (*Fig. 3.4*) was also lowest from the deforested area (1.1605 cmol kg^{-1}), but considerably higher from the forest (6.0299 cmol kg^{-1}). Median values followed the same pattern, being highest from the forest (6.18 cmol kg^{-1}), and lowest from the deforested area (0.76 cmol kg^{-1}). Mann-Whitney U tests revealed a significant difference between all sites (S1 vs. S2 $P = 0.0279$, S2 vs. S3 $P = 0.0310$, S3 vs. S1 $P = 0.0004$).

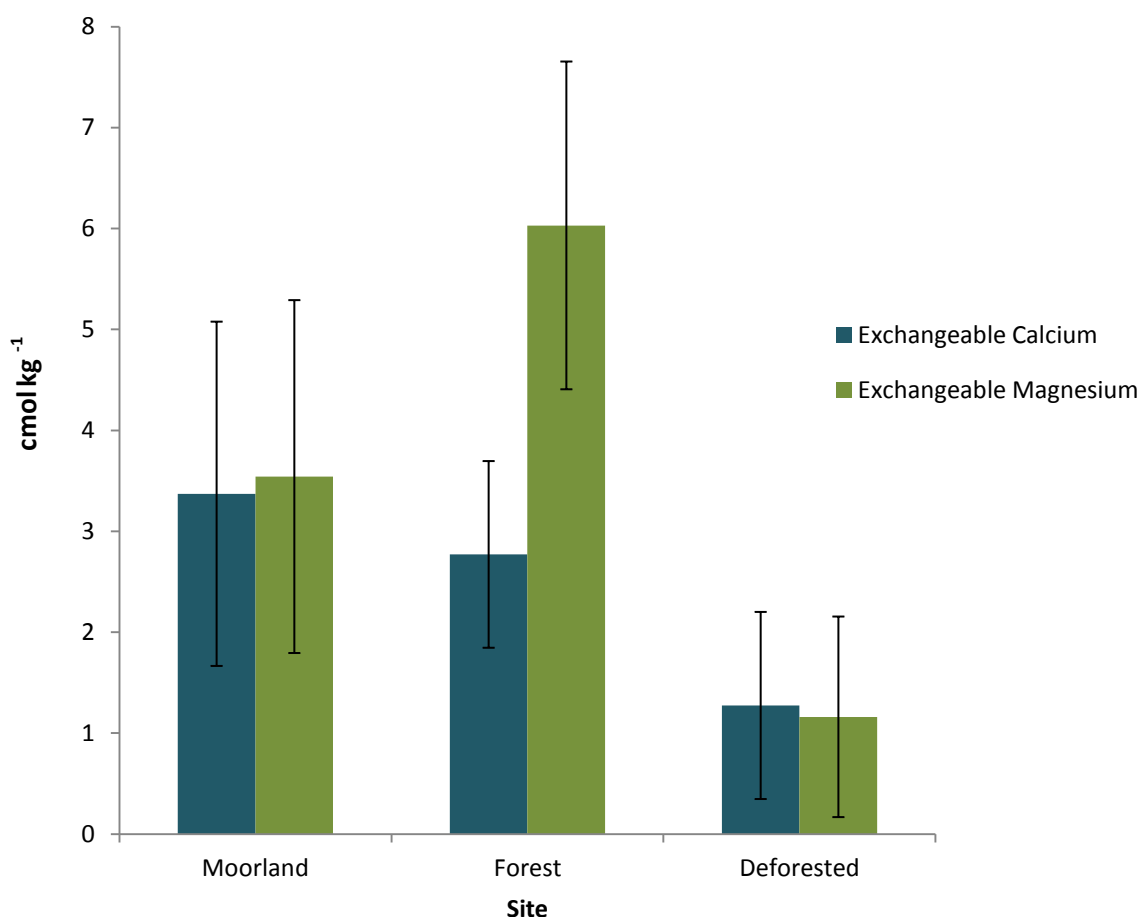


Figure 3.4 Average exchangeable base cations calcium and magnesium at the three study sites (error bars represent standard deviation, $N = 15$ per site). Moorland S1; Forest S2; Deforested S3.

Exchangeable potassium varied between sites (*Fig. 3.5*), with the moorland soils containing the highest concentration, whereas soils from the deforested site had the lowest concentration ($1.1011 \text{ cmol kg}^{-1}$ and $0.5811 \text{ cmol kg}^{-1}$ respectively). Mann-Whitney U tests revealed a significant difference between all sites (S1 vs. S2 $P = 0.0279$, S2 vs. S3 $P = 0.0310$, S3 vs. S1 $P = 0.0028$), the paired deforested and moorland sites having the most difference.

Exchangeable sodium was highest from the forest ($1.8975 \text{ cmol kg}^{-1}$), and lowest from the deforested site ($0.7390 \text{ cmol kg}^{-1}$) (*Fig. 3.5*), with the forest having the most variability between samples, and the deforested site having the least ($0.3711 \text{ cmol kg}^{-1}$ and $0.2197 \text{ cmol kg}^{-1}$ respectively). After ANOVA, Fisher's pot hoc test revealed all sites to be significantly different from each other.

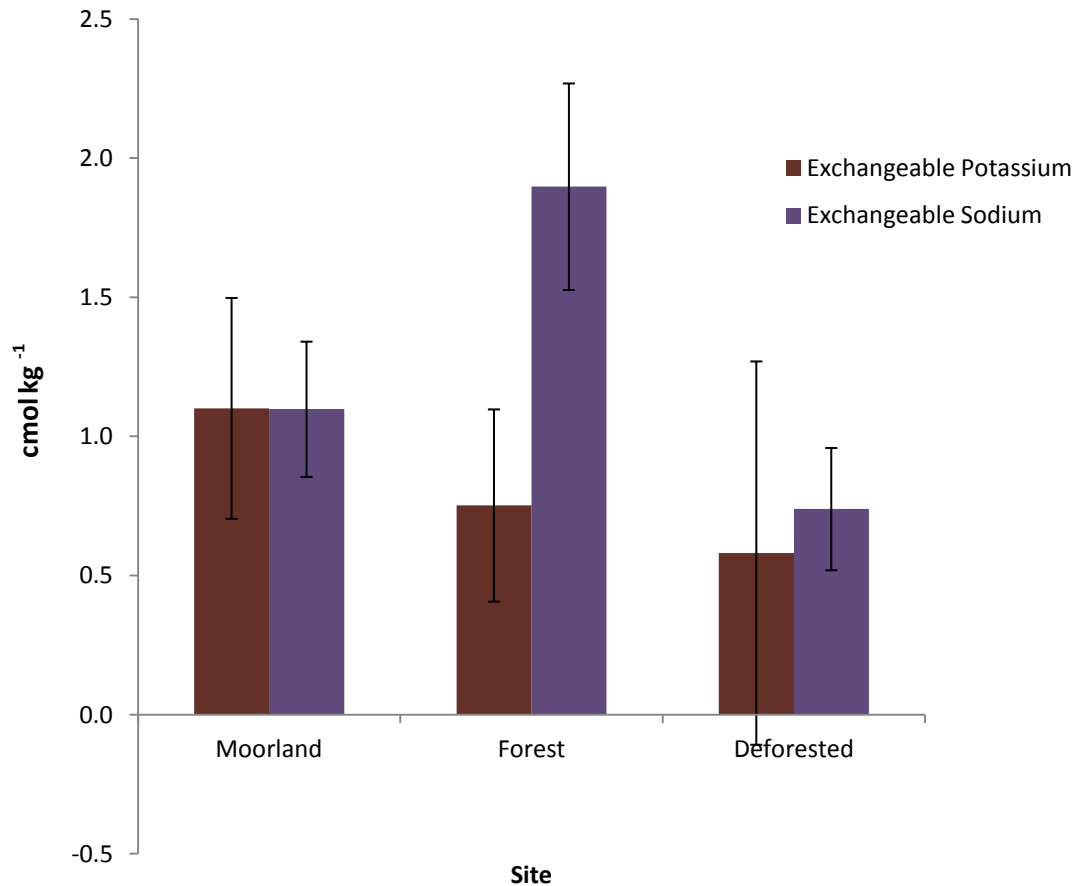


Figure 3.5 Average exchangeable base cations potassium and sodium at the three study sites (error bars represent standard deviation, $N = 15$ per site). Moorland S1; Forest S2; Deforested S3.

3.5 Base saturation variations between sites

Base saturation was found to be highest in soils collected from the forest, and lowest in soils collected from the deforested site (*Table 3.2*).

Table 3.2 Base saturation at all three study sites.

Site	Site number	Base saturation (%)
Moorland	Site 1 (S1)	44
Forest	Site 2 (S2)	60
Deforested	Site 3 (S3)	34

3.6 Available nitrogen variations between sites

Ammonium concentration showed lots of variability between individual samples at each site; the highest being 20.60 mg kg⁻¹ in the deforested site, 16.44 mg kg⁻¹ in the moorland, and 12.56 mg kg⁻¹ in the forest. Ammonium concentration varied between sites (*Fig. 3.6*), with Fisher's post hoc test revealing a significant difference between S1 vs. S2, and S3 vs. S1 after an initial ANOVA test.

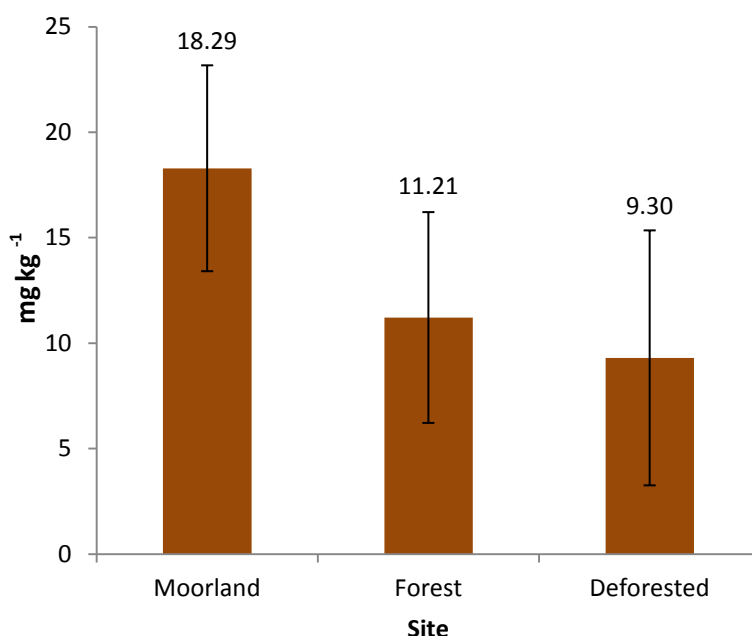


Figure 3.6 Ammonium concentration at the three study sites (error bars represent standard deviation, N = 15 per site). Moorland S1; Forest S2; Deforested S3.

Nitrate concentration was by far the lowest in the forest, compared to the other two sites (*Fig. 3.7*). A significant difference was observed between S1 vs. S2 ($P = 0.0225$), and S2

vs. S3 ($P = 0.0007$), but not between S3 vs. S1 ($P = 0.0815$) when Mann Whitney U tests were carried out.

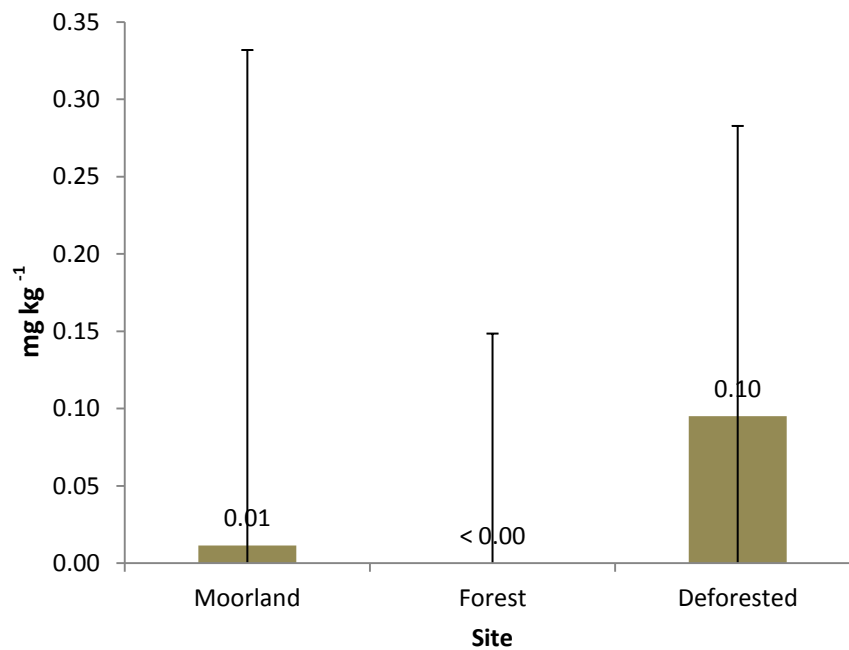


Figure 3.7 Nitrate concentration at the three study sites (error bars represent standard deviation, $N = 15$ per site). Moorland S1; Forest S2; Deforested S3.

3.7 Correlation between pH and exchangeable base cations

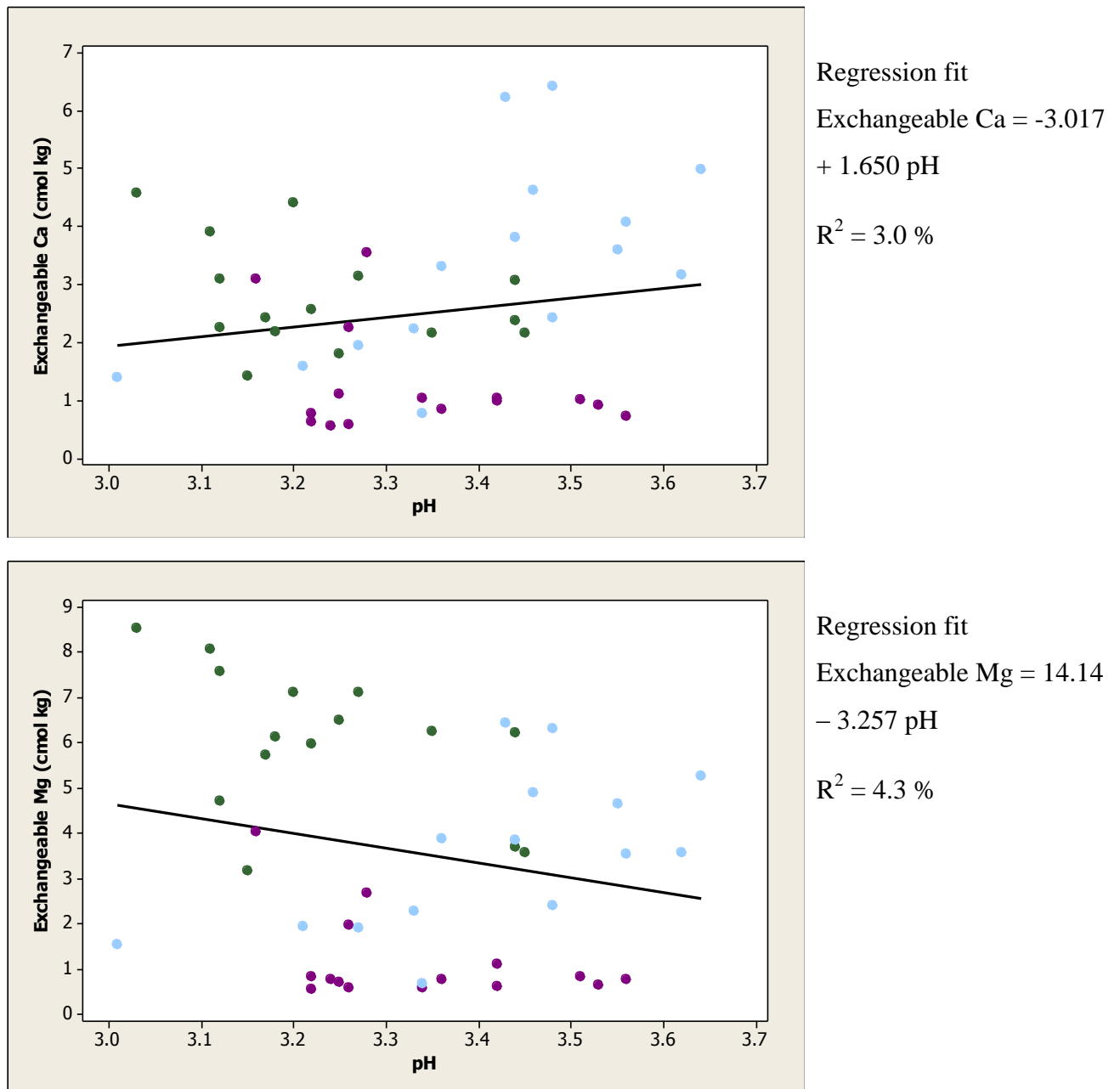


Figure 3.8 Correlation between soil pH and exchangeable base cations calcium (Pearson correlation = 0.128, $P = 0.404$, $N = 45$) and magnesium (Pearson correlation = -0.186, $P = 0.222$, $N = 45$). ■ Moorland ■ Forest ■ Deforested.

Spearman's rank was used to test for a significant correlation between soil pH and all exchangeable base cations. Although not proven significant, a positive correlation was observed for both exchangeable calcium, and exchangeable potassium (exchangeable potassium being more highly correlated with a Pearson correlation value = 0.228) (Fig.

3.8; 3.9). Exchangeable magnesium and exchangeable sodium on the other hand were negatively correlated, exchangeable sodium more so with a Pearson correlation value = -0.253, although again, neither proved significantly correlated with soil pH (Fig. 3.8; 3.9). Individual soil samples collected from each site were generally clustered together.

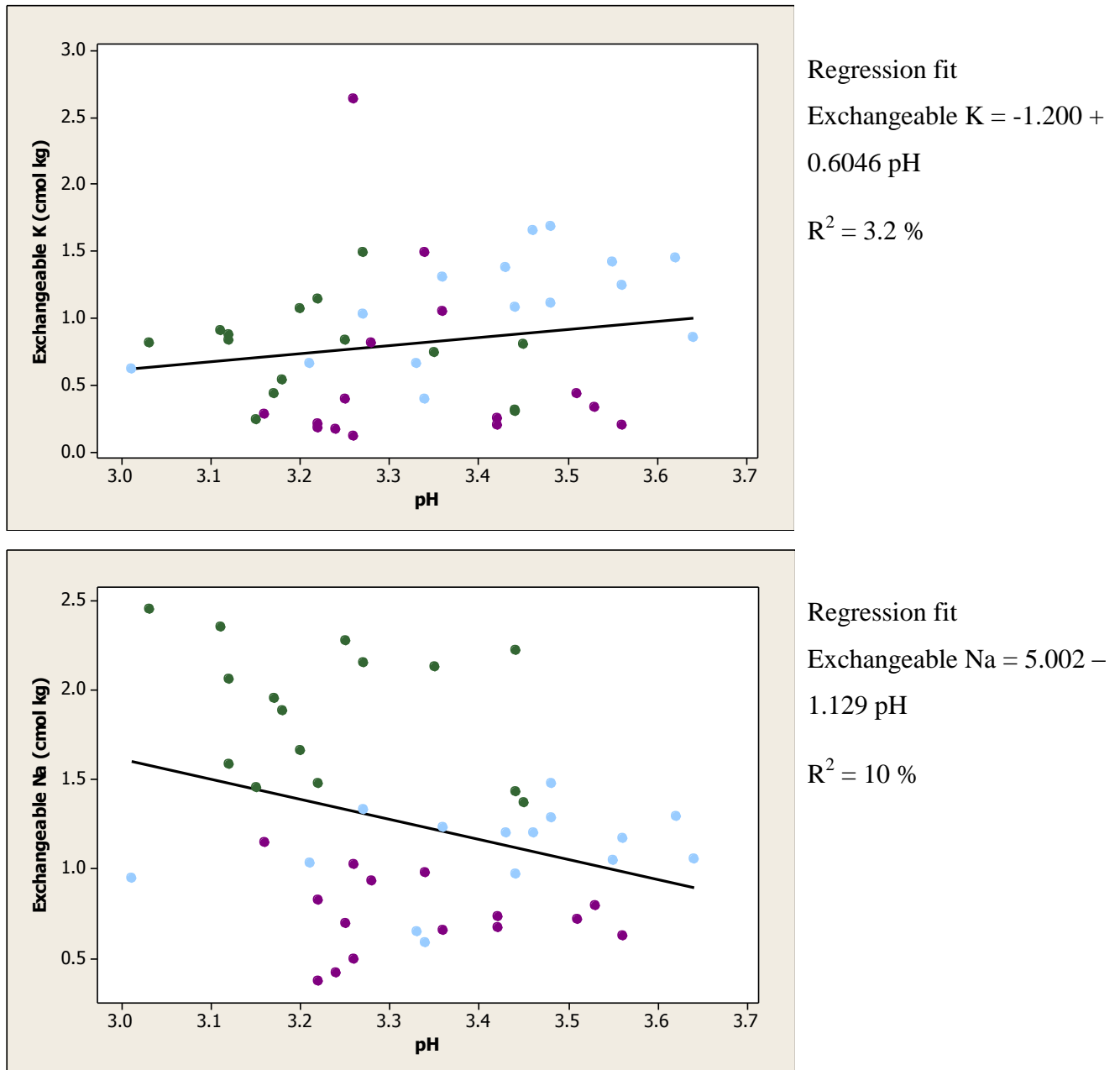


Figure 3.9 Correlation between soil pH and exchangeable base cations potassium (Pearson correlation = 0.228, $P = 0.132$, $N = 45$) and sodium (Pearson correlation = -0.253, $P = 93.000$, $N = 45$). ■ Moorland ■ Forest ■ Deforested.

4. Discussion

4.1 Soil acidity; soil pH and exchangeable acidity

4.1.1 Moorland

UK upland moorland soils, such as those at Ennerdale, are typically acidic due to the high amounts of rainfall they receive annually (in the case of Ennerdale, 1524 – 1778 mm), and the preceding rate at which chemical weathering reactions occur (Likens *et al.*, 1977; Reynolds *et al.*, 1988; Cumbria Hill Farming, 2008). It could therefore be expected that in Ennerdale, the moorland control site would already be of an acidic nature, due to the high input of hydrogen ions to the soil, (indicated by the presence of bilberry and heather). This was demonstrated by the highest concentration of exchangeable hydrogen ions observed in the moorland soil (*Fig. 3.3*), and a low pH of 3.41, emphasising the soils acidity. The weathering of minerals in the soil also proves to be an influential factor controlling soil pH (Strobel *et al.*, 2001), very much dependant on rates of rainfall.

Soil acidity encompasses both soil pH and exchangeable acidity; the two being intricately linked (Brady and Weil, 2008). Despite the moorland soil having the highest soil pH, it unexpectedly had the highest concentration of exchangeable hydrogen ions. This was surprising as pH is a measure of hydrogen ions on a negative logarithmic scale, so it could be expected that the soil with the lowest pH, would hence have the highest concentration of exchangeable hydrogen ions (Rowell, 1994; Brady and Weil, 2008). The high concentration of exchangeable hydrogen ions could, however, explain the highest concentration of exchangeable aluminium, owing to hydrogen ions causing the release of the highly toxic aluminium ions (Rowell, 1994). Aluminium is particularly harmful to plants and animals, specifically freshwater organisms, because of its toxicity (Grieve, 2001). It is deemed to be most responsible for inhibiting the extension of particularly sensitive plant roots, leading to a somewhat expected deficiency in the supply of essential nutrients to a plant (Cresser *et al.*, 1993; Rowell, 1994; Gerrard, 2000; Ashman and Puri, 2002).

4.1.2 Forest

Coniferous afforestation of moorland soils has been shown to expectedly exacerbate soil acidity (Barton *et al.*, 1999; Gerrard, 2000; Smal and Olszewska, 2008). As could be assumed, soil collected from the forest in Ennerdale resulted in the lowest, most acidic pH

of 3.23 (*Fig. 3.2*), proving to be significantly different to that of the moorland soil. Studies by both Berthrong *et al.*, (2009), and Smal and Olszewska (2008) found too, that afforestation led to more acidic soil. The presence of a coniferous tree canopy plays a vital role in aiding soil acidification compared to the short, sparse vegetation, such as bilberry, present on moorland (Barton *et al.*, 1999). A coniferous tree canopy also leads to greater soil acidification compared to that of a deciduous tree canopy, because of the larger surface area and height coniferous trees comprise (Reynolds *et al.*, 1988; Neal *et al.*, 1992; Barton *et al.*, 1999). This enables the capture, and absorption of industrially produced acidic atmospheric pollutants, such as nitrous oxides and sulphur dioxide, which are released into the soil via stemflow and throughfall, producing nitric and sulphuric acids (Reynolds *et al.*, 1988; Neal *et al.*, 1992; Gerrard, 2000; Grieve, 2001; Smal and Olszewska, 2008). Even in areas of low pollution, such as Ennerdale, these pollutants accumulate gradually over time, eventually leading to the enhancement of soil acidity (Barton *et al.*, 1999). Johnson and Lidberg (*in* Adriano and Havas, 1989) also found the effect of atmospheric deposition to be a significant factor in soil acidification.

Soil acidity not only arises from humanly produced pollutants, but also various natural processes such as organic matter breakdown, and the widespread input of acidic pine needles onto the forest floor (Neal *et al.*, 1992; Rowell, 1994; Fisher and Binkley, 2000; Gerrard, 2000; Ashman and Puri, 2002), which has inevitably occurred in Ennerdale. The production of carbonic acid by microbial respiration, also leads to the intensification of soil acidity (Gerrard, 2000). Differing tree species produce unique litter comprising of different chemical composition (Strobel *et al.*, 2001). It was expected that a reduction in soil pH would lead to an increase in exchangeable acidity (Rowell, 1994), however, this was not evident, as overall exchangeable acidity decreased (*Fig. 3.3*). The clay texture of the forest soil is also thought to have been a significant factor in decreasing pH, with Oxbrough *et al.*, (2006) concluding that peatlands were most sensitive to afforestation.

Soil acidity itself determines numerous soil characteristics, most notably nutrient availability. Exchangeable aluminium interferes greatly with the movement and transfer of exchangeable base cations in a soil, causing a variety of problems (Rowell, 1994; Ashman and Puri, 2002). Soil acidity has been shown in the past to contrast strongly between sites of different land use (Richter and Markewitz, 2001), also proving to be highly related to the regular leaching of soil minerals (Cresser *et al.*, 1993; Rowell, 1994).

4.1.3 Deforested

Soil pH increased from forest soil to soil collected from the deforested site (*Fig. 3.2*), even though a significant difference was not obtained. This does, however, very much indicate that the removal of trees has been an influential factor in reducing soil acidity. Neal *et al.*, (1992) also observed that tree canopy removal in mid Wales led to a decrease in soil pH, which they suggested was due to a dramatic reduction in the capture of acidic pollutants.

The loss of organic matter due to deforestation reduces the amount of carbonic acid produced in the soil (as organic matter contains a high carbon content), and therefore acidity. This was found to be true in the study by Smal and Olszewska (2008), where soil pH increased as organic matter content decreased. The lowest concentration of exchangeable acidity was observed in soil collected from the deforested site (*Fig. 3.3*) even though soil pH was not the highest at this site, which again was unexpected as it does not follow past literature.

4.2 Exchangeable base cations

4.2.1 Soil texture

Soil texture plays a huge role in determining the majority of chemical soil properties, namely the movement and retention of exchangeable base cations (Cresser *et al.*, 1993; Ashman and Puri, 2002). Compared to clay particles (< 0.002 mm), sand particles (0.02 – 2 mm) within a soil consequently create large pore spaces, resulting in little capability to retain water or nutrients (Brady and Weil, 2008; Chapman, *in* Holden, 2008). Exchangeable base cations are therefore often easily leached out of a soil, contributing to the low nutrient content of sandy soils (Gerrard, 2000; Brady and Weil, 2008). This is evident at Ennerdale, with soil collected from the deforested site, consisting of a sandy clay loam texture, having the lowest concentration of exchangeable base cations (*Fig. 3.4; 3.5*). Sandy, well drained and aerated soil is often also prone to erosion (Gerrard, 2000).

Clay particles, on the other hand, are much more compactable than sand particles, due to their small size, and so create considerably smaller pore spaces, which pose poor drainage (Gerrard, 2000; Brady and Weil, 2008). This, combined with their large surface area and electrical charge, makes clay particles particularly good at absorbing water, and nutrients,

which accounts for clay dominated soils being nutrient rich (Gerrard, 2000; Brady and Weil, 2008). This was observed at Ennerdale, with clay dominated soil from both the moorland and forest, comprising high concentrations of exchangeable base cations (*Fig. 3.4; 3.5*). This does mean, however, that waterlogging of clay and peatland soil is extremely common, with different tree species tolerating waterlogging better than others (Paterson and Mason, 1999; Gerrard, 2000; Brown *et al.*, 2010). For example, the roots of Sitka spruce are particularly sensitive to waterlogging (Paterson and Mason, 1999).

4.2.2 Cation exchange capacity and base saturation

The cation exchange capacity (CEC) of a soil refers to the total amount of exchangeable cations that it can absorb (Brady and Weil, 2008). The CEC of soil is expected to change as vegetation does (Saikh *et al.*, 1998), although this sometimes may not be evident until several decades after afforestation (Olszewska and Smal, 2008). CEC is highly influential upon soil texture, specifically the presence of negatively charged colloidal material, namely clay particles (Chapman, *in* Holden, 2008). Soils dominated by clay particles, and those which contain high organic matter content (moorland and forest soil in Ennerdale), therefore have a higher CEC compared to sandier soils (those collected from the deforested site in Ennerdale), which comprise of little negatively charged colloidal material (Cresser *et al.*, 1993; Brady and Weil, 2008; Chapman, *in* Holden, 2008). This could explain the significantly higher concentration of exchangeable base cations observed at the moorland and forested sites, compared to the deforested site (*Fig. 3.4; 3.5*).

Base saturation is highly related to the CEC of a soil, indicating the ‘proportion of exchange sites occupied by base cations’, where a higher base saturation represents a more fertile soil (Gerrard, 2000; Chapman, *in* Holden, 2008). Soil collected from the deforested site in Ennerdale had a significantly lower LOI (an indication of organic matter, and therefore negatively charged colloidal material) than soil collected from the other two sites (*Table 3.1; Fig. 3.1*). This is similar to Clark *et al.*, (2011), who also found peaty soil to have higher LOI values. This probably explains the low base saturation level, and hence concentration of exchangeable cations obtained from the deforested site (*Table 3.2; Fig. 3.4; 3.5*).

The CEC of a soil, and base saturation are somewhat influenced by soil acidity (Gerrard, 2000). The more acidic a soil is the higher the concentration of exchangeable aluminium ions, which in turn dominate cation exchange sites because of their high ionic composition

(Goulding and Stevens, 1988). This decreases the absorption of exchangeable base cations, most noticeably exchangeable calcium and exchangeable magnesium (Rowell, 1994; Ashman and Puri, 2002; Chapman, *in* Holden, 2008). As the CEC and base saturation of a soil increases, it can therefore be expected that soil pH will increase too (Brady and Weil, 2008). This was found by Olszewska and Smal (2008), where a decrease in base saturation was observed after afforestation. In this study, however, this expected relationship was not observed, as the soil from the forest had the highest base saturation of 60%, but the lowest soil pH.

4.2.3 Moorland

The main source of exchangeable base cations to moorland soil is by the direct weathering of rock (Likens *et al.*, 1977; Cresser *et al.*, 1993). This was found particularly true of potassium by Goulding and Stevens (1988). The high rates of rainfall Ennerdale receives, will therefore rapidly weather the underlying ‘Ordovician mudstones and siltstones’ (Ennerdale Historic Landscape Survey, 2003), readily releasing base cations into the soil. This is likely to explain the high concentrations of exchangeable calcium and exchangeable potassium in the moorland soil, compared to lower concentrations of the same base cations at the other sites (*Fig. 3.4; 3.5*).

4.2.4 Forest

The capture and accumulation of atmospheric cations by coniferous trees, over time contributes to an increase in exchangeable base cation supply in the soil (Johnson and Lindberg, *in* Adriano and Havas, 1989; Richter and Markewitz, 2001; Berthrong *et al.*, 2009). Atmospheric cation inputs reach the soil via precipitation, and absorption by the trees, originating from various sources, including terrestrial dust, volcanic emissions and oceanic spray (Likens *et al.*, 1977; Gerrard, 2000). The presence of a tree canopy could perhaps explain the significant increase in concentrations of exchangeable magnesium and sodium in soils collected from the forest, compared to the moorland site (*Fig. 3.4; 3.5*). This increase in exchangeable sodium could also be present because sodium is the base cation which is the least preferred to be absorbed by cation exchange sites, as it is ‘not essential to plant biochemistry’ (Cresser *et al.*, 1993; Berthrong *et al.*, 2009). Exchangeable magnesium results obtained in this study differed to those found by Berthrong *et al.*, (2009), where afforestation was not proved to have a significant effect on increasing soil magnesium concentration. The clay texture of forest soil could also suggest

the retention and increase of exchangeable base cations magnesium and sodium (Ashman and Puri, 2002).

Plants rely on macronutrients present in the soil for growth and survival (Ashman and Puri, 2002). Coniferous afforestation leads to a reduction in base cations from the soil as they are readily taken up by trees, where they are stored in leaf litter and plant biomass (Barton *et al.*, 1999; Gerrard, 2000; Grieve, 2001; Olszewska and Smal, 2008; Berthrong *et al.*, 2009). Exchangeable calcium and potassium concentrations in the soil decreased after coniferous afforestation (*Fig. 3.4; 3.5*), suggesting their rapid uptake by coniferous trees. A decrease in calcium, magnesium, and potassium were found after afforestation in studies by Johnson and Lindberg, *in* Adriano and Havas, (1989), Olszewska and Smal, (2008), and Berthrong *et al.*, (2009) under the afforestation of *Pinus* species. The decrease in exchangeable calcium and potassium could also be explained by the increase in soil acidity. Exchangeable base cation concentrations would decrease, as exchangeable aluminium and exchangeable hydrogen ions would likely dominate cation exchange sites (*Fig. 4.1*) (Gerrard, 2000; Brady and Weil, 2008). This was also found to be true by Johnson and Lindberg, *in* Adriano and Havas, (1989) who concluded that a decrease in exchangeable base cations was due to an increase in soil acidity, via acid deposition.

4.2.5 Deforested

Nutrients in a soil become depleted when their removal is higher than their addition (Gerrard, 2001). Coniferous deforestation in Ennerdale led to the significant decrease in all exchangeable base cations, from that of the forested site. Saikh *et al.*, (1998) also concluded that forest clearance was responsible for the significant decrease in exchangeable base cations in the soil, whereas Johnson, (1992) stated that ‘most studies show no significant change with harvesting’, contrasting to results from this study. The uptake and storage of exchangeable base cations by coniferous trees undoubtedly played a huge role in their reduction from the soil, removed when trees were harvested (Goulding and Stevens, 1988). After all, there is a strong relationship between the recycling of elements in soil and plants (Richter and Markewitz, 2001).

Leaching is the process most responsible for the loss of exchangeable base cations from a soil, with ‘cations with the least affinity for exchange sites’ being lost most easily (Gerrard, 2000). Precipitation levels play a significant role in the leaching of a soil (Ashman and Puri, 2000), and with no forest canopy to intercept precipitation,

exchangeable base cations would be frequently washed out. This could create distinct nutrient rich horizons lower down in the soil profile (Ashman and Puri, 2002). As well as leaching, soil erosion can reduce exchangeable base cation concentrations, predominantly in the topsoil where a high majority of exchangeable base cations are concentrated (Ashman and Puri, 2002).

As previously established, soil texture plays a huge role in the loss of exchangeable base cations from a soil. Soil collected from the deforested site, consisting of a sandy clay loam, would pose larger pore spaces than that of the other two clay dominated sites, thus allowing rapid water movement through the soil, and permitting dissolved exchangeable base cations to be easily lost (Paterson and Mason, 1999; Gerrard, 2000; Ashman and Puri, 2002; Brady and Weil, 2008). The often poor quality, stony soils where afforestation takes place is particularly poor in retaining nutrients, and so would likely enhance leaching (Fisher and Binkley, 2000; Gerrard, 2000).

Leaching is known to aid the acidification of soil (Cresser *et al.*, 1993; Olszewska and Smal, 2008). Again, like in the forest soil, a reduction in exchangeable base cations would suggest an increase in soil acidification, exchangeable aluminium and exchangeable hydrogen ions dominating the exchange complex (Gerrard, 2000; Richter and Markewitz, 2001), however, this was not observed at Ennerdale.

4.3 Correlation between soil pH and exchangeable base cations

In this study, despite no significant correlations being observed between soil pH and exchangeable base cations, slight trends were evident (*Fig. 3.8; 3.9*). As soil pH increased, so too did the concentration of exchangeable calcium and potassium, although only very slightly. An increase in exchangeable base cations as soil pH increases would be expected, as the higher the soil pH, the less exchangeable aluminium and hydrogen ions present in the soil. This would enable a higher potential for base cations to be held onto the exchange sites of negatively charged colloidal material, as demonstrated in *Fig. 4.1* (Brady and Weil, 2008; Chapman *in* Holden, 2008).

Exchangeable magnesium and sodium on the other hand decreased as soil pH increased (*Fig. 3.8; 3.9*). This was not expected as these results do not fit in with previous literature (Brady and Weil, 2008; Chapman, *in* Holden, 2008). Again, soil texture and LOI, thus

base saturation could perhaps partly explain these surprising correlations, as they account for the low amount of negatively charged colloidal material present in the deforested soil, limiting cation exchange sites. This, however, should be investigated further in the future.

It was evident that individual soil samples collected at each site were closely grouped together, displaying trends of their own. Soil collected from the moorland site displayed the most expected trends, exchangeable base cations increasing with soil pH, although these may still not be proved significant if tested on their own.

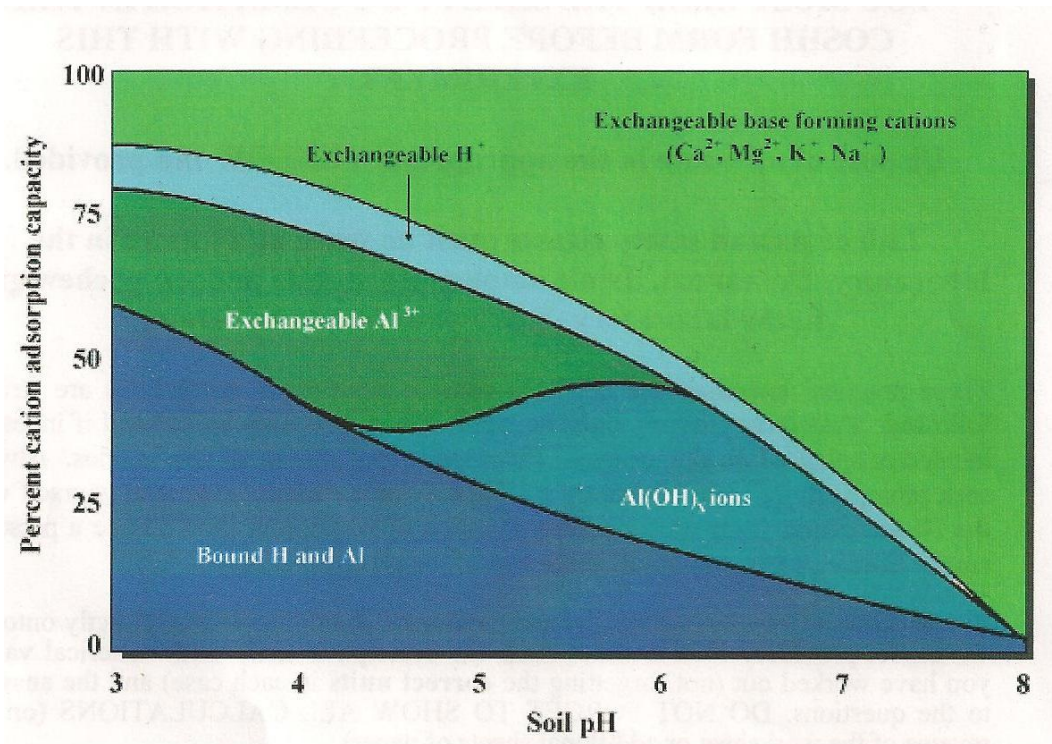


Figure 4.1. Diagram taken from Brady and Weil, 2008 to show the relationship between soil pH and exchangeable cations held on to cation exchange sites within a soil.

4.4 Available nitrogen

4.4.1 Moorland

Nitrogen is an essential macronutrient for plants, taken up in the forms of ammonium and nitrate, but with nitrite being toxic to plants (Cresser *et al.*, 1993; White, 1997; Brady and Weil, 2008). As nitrogen 'does not occur to any useful extent in primary minerals' (Cresser *et al.*, 1993), its input to the soil via fixation from the atmosphere is essential. Nitrogen in the soil is capable of varying substantially over decades (Richter and

Markewitz, 2001). Soil collected from the moorland site had a significantly higher concentration of ammonium compared to the other two sites (*Fig. 3.6*). Nitrate concentration did not follow this pattern, however (*Fig. 3.7*).

4.4.2 Forest

Concentrations of both ammonium and nitrate decreased significantly from the moorland site to the forested site. This could be expected since because nitrogen is a macronutrient essential for the growth and survival of plants, afforestation hence leads to a significant decrease in both available forms of nitrogen in forest soil (Rowell, 1994). Richter and Markewitz (2001), stated that pine forests are known for the high concentrations of available nitrogen they uptake, with Berthrong *et al.*, (2009) also finding that along with soil carbon, soil nitrogen decreased after afforestation when compared to soil collected from native grasslands. They concluded that this was probably due to plant uptake. A decline in soil nitrogen was also observed by Smal and Olszewska (2008), again, thought to be due to the rapid uptake by fast growing coniferous tree species. A decline in nitrate concentration from the mineral horizon was observed by Stevens and Hornung (1988), proved true of this study also, as nitrate concentration in forest soil was extremely low at $< 0.00 \text{ mg/kg}^{-1}$.

Soil acidity can play a significant role in the amount of available nitrogen present in a soil. Usually the more acidic a soil, the lower the macronutrient supply (Cresser *et al.*, 1993; Chapman, *in* Holden, 2008). Again, the more acidic the soil the more hydrogen ions expected to be present, thus leading to an increase in exchangeable aluminium ions likely dominating cation exchange sites (although this was not found to be true of this study). In theory though, this would lead to a decrease in the positively charged ammonium ions (NH_4^+) that can be held on cation exchange sites (as they will be taken up by exchangeable aluminium) and so they could be readily leached from a soil. Nitrification rates can also influence soil acidity (Richter and Markewitz, 2001), due to the production of hydrogen ions in ammonium (Brady and Weil, 2008).

4.4.3 Deforested

Ammonium concentration was lowest in soil collected from the deforested site (*Fig. 3.6*). This could be expected due to its uptake and retention by coniferous trees, accumulating in the wood and bark, and therefore being lost during deforestation (Stevens and Hornung,

1988; Richter and Markewitz, 2001; Smal and Olszewska, 2008). Nitrate concentration, however, was surprisingly highest in soil collected from the deforested site (*Fig. 3.7*), increasing from that of the forested site. Stevens and Hornung (1988) also found nitrate concentration to increase after deforestation, although reasons for this are unknown.

Nitrate too is extremely susceptible to leaching. Positively charged ammonium ions (NH_4^+) can be held on cation exchange sites, but negatively charged nitrate (NO_3^-) and nitrite (NO_2^-) ions cannot, leading to them being easily leached from the soil (Cresser *et al.*, 1993; Rowell, 1994; Brady and Weil, 2008). Due to the soil collected from the deforested site being of a sandy clay loam texture, leaching rates would have been high. As soil carbon and soil nitrogen are indicators of soil acidity, a decrease in their concentrations suggest an overall decrease in soil fertility (Berthrong *et al.*, 2009). This appears to be true of this study, as when a decrease in ammonium was observed in soil collected from the deforested site, a decrease in all exchangeable base cations was also observed (*Fig. 3.4; 3.5; 3.6*).

4.5 Limitations

There are several limitations which, if this study were to be conducted again, should be considered. Firstly, there was probably not enough soil samples collected from each site to truly represent their area, given their relative size. Number of soil samples permitted to analyse in the laboratory was restricted, however, along with time. Individual soil samples collected at each site sometimes showed much variability, with Cresser *et al.*, (1993) stating that even over small distances, soil pH can vary by up to two units. The more samples used in analysis would therefore reduce this variability. It would also have been interesting to examine the effects of coniferous afforestation and deforestation on the physical properties of soil as well. Individual soil samples were comprised of both organic and mineral soil horizons, this combination of different horizons could inevitably have lead to a misinterpretation of the true soil conditions that are in fact present for individual horizons, so in the future should be analysed separately.

Insufficient time had elapsed after the coniferous forest was deforested to truly examine the effects coniferous deforestation has had on the chemical properties of soil. This study was conducted to provide only an indication of results, however. In terms of significantly

changing soil properties, ten years is not considered a long time. Smal and Olszewska, (2008) suggested that significant changes in soil properties may not be observed until several decades after afforestation.

5. Conclusion

Overall, it can be concluded that both coniferous afforestation and deforestation led to substantial changes in the chemical properties of soil. With the moorland site in Ennerdale acting as a control site, it could be established that coniferous afforestation led to a reduction in soil pH, making the forested site in Ennerdale the most acidic of the three examined. Coniferous deforestation could then be recognised to increase soil pH, reducing its acidity, due to the loss of the coniferous tree canopy. Exchangeable acidity results, however, were not as expected and so did not interlink with soil pH results. Despite coniferous afforestation leading to the lowest soil pH, exchangeable acidity was not the highest. This would have been predicted with decreasing soil pH, due to the increasing exchangeable hydrogen ions, and in turn exchangeable aluminium ions. Coniferous deforestation in fact surprisingly led to the overall lowest exchangeable acidity (exchangeable aluminium and exchangeable hydrogen ions combined).

Compared to the moorland control site, coniferous afforestation led to varying concentrations of exchangeable base cations. Exchangeable magnesium and sodium increased, probably due to their capture from the atmosphere and thus accumulation by the coniferous tree canopy. Exchangeable calcium and potassium on the other hand decreased, which was undoubtedly due to their considerable increase in uptake by the coniferous trees for growth and survival. Despite coniferous afforestation leading to a variety of both increases and decreases in individual exchangeable base cations, coniferous deforestation led to the significant reduction in all exchangeable base cation concentrations. It could be predicted, however, that differing soil texture between sites, namely the sandier deforested soil, played a huge role in this. No significant relationships were observed between soil pH and exchangeable base cation concentrations, which did not coincide with past literature.

Ammonium and nitrate results differed in their concentration pattern, making it difficult to establish a true pattern for overall available nitrogen. Coniferous afforestation led to a decrease in both forms of nitrogen compared to the moorland control site, suggesting tree uptake was a dominant factor in reducing this macronutrient. This was accompanied by the highest LOI, representative of organic matter. Coniferous deforestation, however, proved to be much more variable, leading to a decrease in ammonium concentration, but

an increase in nitrate concentration. The lowest LOI was observed at the deforested site, though its sandy clay loam soil texture should be considered.

Taking everything into account, it can be established that coniferous afforestation led to an overall decrease in soil quality, as almost every chemical property measured (bar exchangeable magnesium and sodium) decreased from that of the moorland site. Coniferous deforestation, however, seems to have exacerbated these poor conditions, despite an overall increase in soil pH. Soil pH is nevertheless such an influential factor for numerous chemical soil properties that despite a lack of significant increases in exchangeable base cations and available nitrogen, this increasing soil pH result is somewhat encouraging.

The realisation of the negative effects coniferous afforestation has on soil chemical properties, has, and will continue to grow with the help of influential organisations such as the Forestry Commission, National Trust, and United Utilities. Despite coniferous deforestation only leading to a positive increase in soil pH, it should be remembered that coniferous deforestation is only the first step in improving soil conditions at Ennerdale under the 'Wild Ennerdale' rewilding initiative. Reforestation of native broadleaf deciduous species is the next stage of 'Wild Ennerdale', attempting to return the landscape back to its once semi-natural state, and so analysis of reforested areas such as these should, in due course be analysed too. Hopefully, this will see the eventual desired return of exchangeable base cations and available nitrogen to the soil, aiding soil improvement in the future. Therefore, as a continuation of this study, the effect of deciduous reforestation on the chemical properties of soil within Ennerdale should be examined, to see if there is an improvement in soil quality, preserving this fundamental resource.

To further this study, long term monitoring of soil from the same coniferous deforested site within Ennerdale could be undertaken, say every decade, to establish a real pattern in the changing chemical properties of soil, until reforestation of the site commences. The lack of significant results suggests that it takes longer than a ten year period for soil chemical properties to actually improve, which can be expected seeing as soil takes hundreds, if not thousands of years to form. The effect both coniferous afforestation and deforestation have on other environmental factors could also be considered, to widen our knowledge of the ecosystem as a whole. The tightly coupled integrated river ecosystem, for example, could be taken into account, specifically water quality and macroinvertebrate community

composition. Many studies have considered the effect coniferous afforestation has on river ecosystems, but the effects of coniferous deforestation are lacking, especially studies which combine the two. Whole ecosystems should benefit from improved soil conditions, enabling a greater number of species, both florally and faunally, to be supported. Although not specifically looked at within this study, soil is particularly important as a store of carbon, which if frequently released could contribute significantly to global warming (Chapman, *in* Holden, 2008). This is another aspect of soil chemistry that could be investigated, to further our knowledge and understanding of potential future climate change, enabling us to recognise just how influential coniferous deforestation is on soil carbon reserves.

6. References

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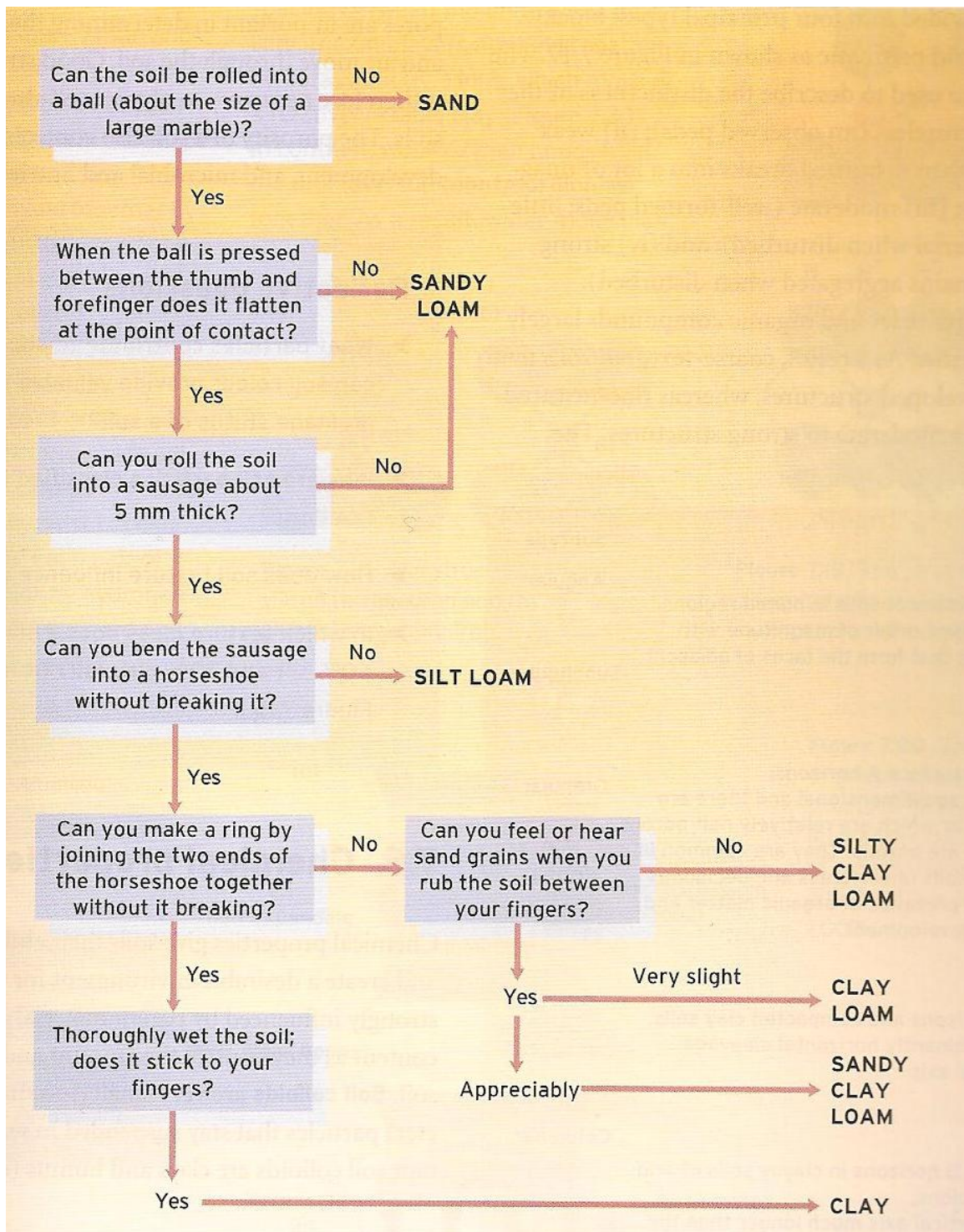
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7. Appendix

Appendix 1



Appendix 1.1. Soil texture 'feel' test (Chapman, in Holden, 2008).

Appendix 2

Sampling selection grids

211	212	213	214	215	216	217	218	219	220	221	222	223	224	225
196	197	198	199	200	201	202	203	204	205	206	207	208	209	210
181	182	183	184	185	186	187	188	189	190	191	192	193	194	195
166	167	168	169	170	171	172	173	174	175	176	177	178	179	180
151	152	153	154	155	156	157	158	159	160	161	162	163	164	165
136	137	138	139	140	141	142	143	144	145	146	147	148	149	150
121	122	123	124	125	126	127	128	129	130	131	132	133	134	135
106	107	108	109	110	111	112	113	114	115	116	117	118	119	120
91	92	93	94	95	96	97	98	99	100	101	102	103	104	105
76	77	78	79	80	81	82	83	84	85	86	87	88	89	90
61	62	63	64	65	66	67	68	69	70	71	72	73	74	75
46	47	48	49	50	51	52	53	54	55	56	57	58	59	60
31	32	33	34	35	36	37	38	39	40	41	42	43	44	45
16	17	18	19	20	21	22	23	24	25	26	27	28	29	30
1	2	3	4	5	6	7	8	9	10	11	12	13	14	15

Appendix 2.1 Soil sampling grid selection for the moorland site.

Extra random numbers generated: 119, 90, 98, 33, 46, 101, 199, 46, 78, 26, 112, 8, 179, 186, and 60.

211	212	213	214	215	216	217	218	219	220	221	222	223	224	225
196	197	198	199	200	201	202	203	204	205	206	207	208	209	210
181	182	183	184	185	186	187	188	189	190	191	192	193	194	195
166	167	168	169	170	171	172	173	174	175	176	177	178	179	180
151	152	153	154	155	156	157	158	159	160	161	162	163	164	165
136	137	138	139	140	141	142	143	144	145	146	147	148	149	150
121	122	123	124	125	126	127	128	129	130	131	132	133	134	135
106	107	108	109	110	111	112	113	114	115	116	117	118	119	120
91	92	93	94	95	96	97	98	99	100	101	102	103	104	105
76	77	78	79	80	81	82	83	84	85	86	87	88	89	90
61	62	63	64	65	66	67	68	69	70	71	72	73	74	75
46	47	48	49	50	51	52	53	54	55	56	57	58	59	60
31	32	33	34	35	36	37	38	39	40	41	42	43	44	45
16	17	18	19	20	21	22	23	24	25	26	27	28	29	30
1	2	3	4	5	6	7	8	9	10	11	12	13	14	15

Appendix 2.2 Soil sampling grid selection for the coniferous afforested site.

Extra random numbers generated: 8, 17, 23, 194, 54, 1, 170, 201, 15, 186, 117, 182, 141, 158, and 23.

211	212	213	214	215	216	217	218	219	220	221	222	223	224	225
196	197	198	199	200	201	202	203	204	205	206	207	208	209	210
181	182	183	184	185	186	187	188	189	190	191	192	193	194	195
166	167	168	169	170	171	172	173	174	175	176	177	178	179	180
151	152	153	154	155	156	157	158	159	160	161	162	163	164	165
136	137	138	139	140	141	142	143	144	145	146	147	148	149	150
121	122	123	124	125	126	127	128	129	130	131	132	133	134	135
106	107	108	109	110	111	112	113	114	115	116	117	118	119	120
91	92	93	94	95	96	97	98	99	100	101	102	103	104	105
76	77	78	79	80	81	82	83	84	85	86	87	88	89	90
61	62	63	64	65	66	67	68	69	70	71	72	73	74	75
46	47	48	49	50	51	52	53	54	55	56	57	58	59	60
31	32	33	34	35	36	37	38	39	40	41	42	43	44	45
16	17	18	19	20	21	22	23	24	25	26	27	28	29	30
1	2	3	4	5	6	7	8	9	10	11	12	13	14	15

Appendix 2.3 Soil sampling grid selection for the coniferous deforested site.

Extra random numbers generated: 178, 68, 11, 98, 74, 124, 61, 112, 151, 191, 189, 59, 222, 76, and 207.

Appendix 3

		Exchangeable Acidity			Exchangeable Base Cations				Available Nitrogen			
Sampling	pH	H ⁺ + Al ³⁺ cmol kg ⁻¹	Al ³⁺ cmol kg ⁻¹	H ⁺ cmol kg ⁻¹	Ca cmol kg ⁻¹	K cmol kg ⁻¹	Mg cmol kg ⁻¹	Na cmol kg ⁻¹	Ammonium NH ₄ mg kg ⁻¹	Nitrate NO ₃ mg kg ⁻¹	Nitrite NO ₂ mg kg ⁻¹	% LOI
Square	CaCl ₂											
5	3.01	23.8422	10.6488	13.1934	1.3862	0.6160	1.5523	0.9478	11.6125	0.1197	0.0150	60.5008
17	3.56	10.0576	5.8115	4.2460	4.0650	1.2425	3.5486	1.1705	11.9698	-0.0299	0.0199	75.6807
30	3.44	8.6355	5.4855	3.1499	3.8114	1.0752	3.8388	0.9730	21.3344	0.9082	0.0149	72.8403
34	3.48	11.9957	5.0982	6.8975	2.4138	1.1088	2.3939	1.2884	18.5041	-0.3140	0.0199	69.4133
35	3.62	13.8557	6.9169	6.9388	3.1639	1.4482	3.5601	1.2893	25.3981	-0.3544	0.0250	90.2517
43	3.21	23.3381	8.5780	14.7601	1.5831	0.6631	1.9419	1.0281	21.0403	-0.2596	0.0200	58.8691
45	3.33	9.9444	7.3920	2.5524	2.2200	0.6595	2.2654	0.6500	18.0833	-0.0499	0.0199	64.0490
48	3.36	9.3591	5.7661	3.5930	3.3145	1.3078	3.8751	1.2309	18.6790	0.4587	0.0199	77.5367
66	3.43	5.0357	2.5398	2.4960	6.2363	1.3779	6.4444	1.2037	17.9490	-0.0199	0.0347	94.7467
100	3.48	6.6081	2.5820	4.0261	6.4338	1.6836	6.3218	1.4778	23.9839	0.0549	0.0100	93.1052
103	3.55	10.5288	6.5750	3.9538	3.5901	1.4199	4.6582	1.0473	14.9848	0.0150	0.0200	89.4132
125	3.46	10.2532	5.3326	4.9205	4.6368	1.6507	4.8914	1.1985	20.9935	0.0000	0.0198	88.8961
127	3.64	7.9005	3.5847	4.3158	4.9828	0.8483	5.2569	1.0506	24.5818	-0.0100	0.0548	88.5354
155	3.27	12.2457	9.8430	2.4028	1.9538	1.0278	1.9059	1.3273	16.2887	-0.0497	0.0249	56.8150
194	3.34	11.7303	9.7860	1.9442	0.7656	0.3878	0.6634	0.5834	8.9624	-0.2973	0.0198	32.0442
Average	3.41	11.6887	6.3960	5.2927	3.3705	1.1011	3.5412	1.0978	18.2910	0.0115	0.0226	74.1798
Replicates												
45	3.26	10.6663	7.1670	3.4994	2.3546	0.4697	2.2838	0.8533	27.5318	0.0149	0.0645	64.6035
194	3.38	12.4379	10.1608	2.2771	1.0076	0.8307	0.7707	0.8834	11.9382	-0.2339	0.0149	31.5515

Appendix 3.1 Raw data from soil collected at the moorland site.

Sampling Square	pH CaCl ₂	Exchangeable Acidity			Exchangeable Base Cations				Available Nitrogen			% LOI
		H ⁺ + Al ³⁺ cmol kg ⁻¹	Al ³⁺ cmol kg ⁻¹	H ⁺ cmol kg ⁻¹	Ca cmol kg ⁻¹	K cmol kg ⁻¹	Mg cmol kg ⁻¹	Na cmol kg ⁻¹	Ammonium NH ₄ mg kg ⁻¹	Nitrate NO ₃ mg kg ⁻¹	Nitrite NO ₂ mg kg ⁻¹	
4	3.44	5.5958	4.0188	1.5770	3.0669	0.3059	3.6869	1.4322	16.9497	-0.1941	0.0100	51.4853
8	3.12	7.0608	2.3779	4.6830	3.0879	0.8745	7.5822	2.0606	4.9860	-0.3839	0.0299	92.5363
25	3.22	6.8545	2.4211	4.4334	2.5628	1.1383	5.9765	1.4769	5.1914	-0.2089	0.0099	95.1684
32	3.15	15.2695	9.0952	6.1744	1.4183	0.2320	3.1857	1.4557	14.3095	-0.3087	0.0349	78.0841
50	3.25	7.5895	3.1166	4.4729	1.8093	0.8295	6.5174	2.2723	11.9112	-0.0949	0.0250	31.4569
125	3.18	7.1775	2.4750	4.7025	2.1910	0.5311	6.1270	1.8863	17.3295	-0.3231	0.0398	95.6317
126	3.11	4.7711	1.4136	3.3574	3.9107	0.9082	8.0646	2.3532	4.7678	-0.2936	0.0100	95.3386
132	3.44	9.0597	3.1189	5.9408	2.3843	0.3015	6.2303	2.2202	7.1663	-0.2236	0.0248	94.4320
136	3.17	9.7305	2.9218	6.8088	2.4236	0.4308	5.7433	1.9568	7.7659	-0.2788	0.0149	91.1823
138	3.12	8.5473	3.7533	4.7941	2.2637	0.8275	4.7091	1.5874	11.3227	-0.1197	0.0150	92.7906
143	3.20	5.3435	2.4408	2.9026	4.4065	1.0644	7.1187	1.6630	16.8427	-0.1142	0.0149	69.0694
172	3.03	6.0175	2.7789	3.2386	4.5779	0.8072	8.5297	2.4505	12.5224	-0.1700	0.0200	95.8569
184	3.35	9.9480	4.4165	5.5315	2.1591	0.7401	6.2540	2.1277	6.0856	-0.2948	0.0150	83.3909
195	3.45	7.3729	2.6601	4.7128	2.1525	0.7996	3.5885	1.3670	11.8297	-0.2139	0.0149	47.2893
204	3.27	5.6377	1.6042	4.0335	3.1323	1.4887	7.1348	2.1531	19.2361	0.2442	0.0199	95.3107
Average		7.7317	3.2408	4.4909	2.7698	0.7520	6.0299	1.8975	11.2144	-0.1985	0.0199	80.6016
Replicates												
4	3.39	5.2705	2.9281	2.3424	2.9750	0.3011	3.6594	1.2977	9.8730	-0.3033	0.0149	49.5258
32	3.04	15.4911	10.9481	4.5431	1.3809	0.2187	3.1016	1.3977	15.2444	0.7302	0.0497	77.8266

Appendix 3.2 Raw data from soil collected at the coniferous afforested site.

Sampling Square	pH CaCl ₂	Exchangeable Acidity			Exchangeable Base Cations				Available Nitrogen			% LOI
		H ⁺ + Al ³⁺ cmol kg ⁻¹	Al ³⁺ cmol kg ⁻¹	H ⁺ cmol kg ⁻¹	Ca cmol kg ⁻¹	K cmol kg ⁻¹	Mg cmol kg ⁻¹	Na cmol kg ⁻¹	Ammonium NH ₄ mg kg ⁻¹	Nitrate NO ₃ mg kg ⁻¹	Nitrite NO ₂ mg kg ⁻¹	
2	3.34	8.3515	4.9090	3.4424	1.0230	1.4872	0.5918	0.9788	3.0722	-0.1743	0.0100	14.4893
3	3.25	7.4749	5.1467	2.3282	1.1159	0.3928	0.7147	0.6956	5.9220	0.1494	0.0149	16.8994
7	3.26	5.5702	4.0964	1.4738	2.2463	2.6377	1.9776	1.0220	11.9625	0.0894	0.0199	21.2247
9	3.28	5.5305	3.3447	2.1859	3.5587	0.8171	2.6738	0.9304	15.5983	0.2630	0.0347	46.4243
12	3.16	8.5508	6.0553	2.4955	3.0895	0.2781	4.0483	1.1467	22.8431	0.1589	0.0248	81.3111
17	3.22	8.0815	6.4734	1.6082	0.7608	0.2071	0.5540	0.8272	5.1951	0.1245	0.0100	14.1217
33	3.53	7.7092	6.2610	1.4481	0.9205	0.3329	0.6380	0.7968	8.7871	0.0843	0.0298	11.6403
68	3.42	6.6791	5.2494	1.4297	0.9928	0.1932	0.6138	0.7353	2.2471	0.2183	0.0149	10.3826
76	3.36	8.0816	6.7869	1.2947	0.8365	1.0459	0.7581	0.6541	11.8440	0.2739	0.0398	14.6607
99	3.42	7.6784	5.9101	1.7684	1.0279	0.2452	1.1079	0.6731	9.9746	0.2678	0.0198	21.0709
107	3.26	8.1333	6.0897	2.0435	0.5795	0.1185	0.5714	0.4930	11.8689	0.3491	0.0150	11.1716
138	3.24	9.1854	7.3567	1.8288	0.5629	0.1684	0.7610	0.4195	5.5332	0.1293	0.0199	14.4659
156	3.22	5.0318	4.2750	0.7568	0.6336	0.1729	0.8210	0.3726	17.6209	-0.1344	0.0249	11.2710
178	3.51	6.6168	5.7249	0.8919	1.0159	0.4287	0.8129	0.7175	4.0731	-0.0644	0.0248	15.0785
219	3.56	6.9618	6.5254	0.4364	0.7319	0.1910	0.7639	0.6221	2.9664	-0.3096	0.0250	15.0410
Average		7.3091	5.6136	1.6955	1.2730	0.5811	1.1605	0.7390	9.3006	0.0950	0.0219	21.2835
Replicates												
76	3.38	18.5811	5.8916	12.6896	0.8321	0.2884	0.7745	0.6142	3.2845	-0.2396	0.0449	14.4345
99	3.62	8.2752	5.9381	2.3371	1.0300	0.3334	1.1174	0.6358	8.8090	0.2384	0.0199	24.0505

Appendix 3.3 Raw data from soil collected at the coniferous deforested site.