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Resilience of upland soils to long term environmental changes

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ABSTRACT

The effect of long-term changes in land-use, pollution deposition and climate change on upland soils was evaluated by resurveying a large set of sites in a mountain landscape in the UK, which were initially sampled forty years ago. Unexpectedly, despite the length of time between sampling dates, no significant changes in pH, soil exchangeable base cations or C and N percentage content by weight were observed across a range of soil type and parent material. This suggests that the soils have been relatively resistant to the large changes in the environmental pressures experienced in the past forty years, which include a 1.5 °C increase in mean temperature; the peak of UK sulphur deposition in around 1970, followed by ~90% deposition reduction; long-term increases in nitrogen deposition; and major changes in grazing intensity. These results suggest that upland soils may be considerably more resilient to the future environmental changes than many previous assessments have suggested.

Keywords:

Climate change

Nitrogen

Montane

Grazing

Land use

Atmospheric pollution

Soil cores

Soil chemistry

1. Introduction

Mountain ecosystems are considered particularly sensitive to environmental changes (Thuiller, 2007) and may provide effective systems in which to detect and assess the ecological impacts of climate change (Beniston, 2003). Some of the greatest changes in land-use, and strongest climatic warming, have been experienced in Central and Northern European mountain ecosystems due to the complex variability of snow, ice and temperature extremes (Hagedorn et al., 2010). However, the impact of these changes on soils, and their role in 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). Because deposition rates vary with topography, upland areas receive disproportionately more S and N deposition compared with lowlands (Kirkham, 2001). These inputs can cause reductions in soil pH and a loss of base cations (Reuss and Johnson, 1986), ultimately resulting in shifts in plant species composition (Smart *et al.*, 2005; Horswill et al., 2008; Maskell et al., 2010; Stevens et al., 2010). Sulphur emissions in the UK peaked around 1970, and subsequently declined, by 80% between 1986 and 2007 (RoTAP, 2012). Soil resurveys and a small number of soil solution monitoring sites show 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 in upland semi-natural ecosystems (e.g. Davies et al., 2005; RoTAP, 2012).

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 and Lehman, 2001). The formation of reactive N has increased globally 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 and Caporn, 1998), particularly where vegetation is adapted to 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 phosphorus (P) limited they are likely to become more readily N saturated (Phoenix et al., 2004), resulting in increased leaching of pollutant N, or they may accumulate much of the N in biomass and soil organic matter (Aber et al., 1989; Phoenix et al., 2004). In general, carbon-poor ecosystems such as montane habitats on thin soils are more susceptible to N saturation, because they have a low capacity to store deposited N in organic matter (Evans et al., 2006a). In the UK, N deposition over the last 20 years has remained fairly stable (RoTAP, 2012), 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, N inputs 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, the largest terrestrial C stocks lie below ground (Bradley et al., 2005). N addition has been found to be positively correlated with increases in soil organic matter in both moorland and forest ecosystems (Evans et al., 2006b; de Vries et al., 2009) and Kirby et al. (2005) observed a positive correlation between N deposition and soil organic matter across the UK. 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 the long-term decomposition rate 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 directly 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 or lost via denitrification. 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 reported a loss of topsoil C between 1978 and 2003, 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), and later analyses of these data suggest that only 10% of the observed changes could be attributed to climate, with the majority of soil C loss attributed to changes in land use and management (Smith et al., 2007; Kirk and Bellamy, 2010), possibly augmented by reducing soil acidity in the uplands (Evans et al., 2007). Land use change has also been identified as responsible for the greatest changes in soil C (Stevens and Wesemael, 2008). Furthermore, other long term soil resurvey studies in the UK have observed either a slight increase (Kirby et al., 2005), or no significant change in soil organic matter over similar time periods (McGovern et al., 2011; Emmett et al., 2010).

The Snowdon area of North Wales has the highest elevation within the southern UK, and a wealth of historic and current environmental data are available. Snowdon itself has an iconic status as the highest mountain in Wales, and provides a diverse range of ecosystem services, ranging from provision of drinking water, food from livestock grazing and energy from hydropower, to recreation and culturally significant biodiversity (Dick et al., 2011). Ball et al. (1969) provided 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 and Perkins, 1978) and climatic measurements (Perkins, 1978) in the area. Additional land use measurements on the intensity of grazing were also carried out (Dale and Hughes, 1978). More recently, part of the Snowdon massif became a site within the Environmental Change Network (ECN), which monitors environmental drivers, particularly climate, pollution and land-use, and the biological responses of habitats and key taxa (Morecroft et al., 2009). ECN protocols for plant, animal, soil and water monitoring (Sykes and Lane, 1996) have been conducted since 1995, along with measurements using the original IBP meteorological equipment 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 following their 1990s peak, increases in temperature since the 1970s, and decreases in atmospheric SO_2 and NO_2 (Lloyd et al., 2012). These changes are compared to longer-term national- or regional- scale trends in these environmental drivers in Fig. 1, which illustrates the magnitude of changes that have occurred since the original soil survey.

In this study, we present the results of a resurvey of soil chemistry at a large set of sites first visited in 1968 as part of a survey by Ball et al. (1969). 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 the detailed environmental records for the site.

2. Materials and Methods

2.1. Sample site

Sample sites, located across the Snowdon massif, were relocated in autumn 2008 using original GB six figure Ordnance Survey grid references. Precise sampling sites were further identified by detailed location, relief, altitude and soil descriptions in the original report. 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. Each sample site was categorised by underlying geology and soil type so that the investigation could also test whether the response of soil chemistry to environmental drivers was consistent.

2.2. Chemical analyses

Chemical analyses were carried out following the methods originally used by 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. C 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. P was extracted using a 1:40 soil to 0.5M acetic acid ratio and analysed using the molybdate blue colorimetric method.

2.3 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.

3. Results

Results of the soil resurvey indicate that there have been only small changes in the measured chemical soil properties on Snowdon during the last forty years (Table 1). Between the two sampling dates (1968 and 2008) there was a slight decline in pH, by an average of 0.08 units. Similarly, there was a small decrease in total soil exchangeable base cations. These results both suggest some acidification of the soil by deposition of acidifying compounds during the intervening period, rather than any recovery in response to falling S deposition levels since the 1970s, although changes were not statistically significant. Individual responses of soil types varied with no consistent trends in exchangeable base cations found.

The direction of change in pH appears to be dependent on the original pH value of the soil. Acid soils have increased in pH and are now less acidic than initial values; whereas the previously less acidic soils have shown a decrease in pH (Fig. 2).

Similarly contrasting results were found for base cation concentrations, with increases in the previously more acidic soils and decreases in the less acidic soils.

The C and N content of the soil appears to have increased, particularly in brown podzolic soils. On the other hand, the C:N ratio has shown little change. An increase in extractable P was also found. However, none of these results were statistically significant. These changes all suggest a small increase in soil total nutrient stocks, consistent with the long-term accumulation of organic matter. However, the absence of a reduction in C:N ratio implies that this may not necessarily equate to an increase in soil fertility.

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

To investigate if there were significant differences between individual soil types or soils on the same underlying geology, the percentage change 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 (Fig. 3). No significant differences were found in any of the soil chemical variables measured although there was some variation between soil types. No significant difference was found between the years within individual soil categories.

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 (Fig. 4). One group however, the rhyolitic and bedded pyroclastic series, did show a significant increase in N content between 1968 and 2008 (p = 0.018).

4. 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 an overall reduction in soil pH (albeit very small), in contrast to these studies, and to a similar study completed on Snowdon based on a different dataset covering a smaller area (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 our original survey took place. By the time the initial samples from the other, UK-wide studies were collected, in the late 1970s, there had already been a drop in S emissions of approximately 20% (NEGTAP, 2001). In general, there is a lag between deposition and environmental effects (Monteith and 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 yet occurred. 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). The initial samples in the UK wide studies may have captured the peak in acidification exhibited by UK soils, whereas this study pre-dates it. This may be why the soils appear to have acidified further rather than indicated a recovery.

Looking more closely at the response of soil pH, it appears that the previously less acidic soils show larger changes in pH. The initially more acidic soils showed an increase in soil pH, suggestive of recovery from acidification while the previously less acidic soils are still showing a decrease in soil pH. 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 capacity of acid and alkaline soils differs, with poorly buffered acid soils tending to respond relatively quickly to changes in deposition, whereas the larger buffering capacity of alkaline soils will lead to lags between the deposition peak and the peak of acidification ('Damage Delay Time'; Hettelingh et al., 2007). This could explain the contrasting changes in pH of acid and alkaline soils on Snowdon during a period in which S deposition has peaked, and then fallen, but still remains above the critical load. The absence of consistent recovery in soil pH across Snowdon, despite substantial reductions in the UK deposition acidifying compounds (Figure 2), highlights the considerable length of time it may take for ecosystems to recover 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 was observed in soil exchangeable base cation concentrations, although there was high variation and no clear trends emerged, consistent with the results for soil pH. The extent of base cation depletion is largely determined by the type of soil and the rate of weathering of the underlying geology. In coastal areas, deposition of base cations from sea-salt can occur, resulting in the alteration of cation exchange equilibria within the soil (e.g. Evans et al., 2001), with greater deposition occurring at times of increased storm events, which tend to coincide with large positive values of the winter North Atlantic Oscillation (NAO) index (Hurrell, 1995;).. Despite a higher winter NAO index in 2008 compared to 1968 (December-March means derived from http://www.cru.uea.ac.uk/~timo/datapages/naoi.htm), which might indicate greater seasalt inputs prior to the recent survey, the Na content of the soil was actually higher in 1968. Therefore there was no evidence to suggest that frequency of antecedent storm occurrence had a major impact on the soil chemistry within the sampling years. Few other long term studies have reported data for exchangeable base cation concentrations, so there is little measurement data against which to evaluate these results. However modelling studies conducted in similar locations have suggested that recovery in soil pH will be more rapid than recovery in base saturation (e.g. Evans, 2005). A recent study on Snowdon incorporating only brown earth soils above pumice-tuffs of relatively high base status found significant reductions in soil exchangeable base cation concentrations, concurrent with an increase in pH, suggesting that recovery from acidification following S deposition reductions may not yet have led to the effective recharge of soil base cation status (McGovern et al., 2011). The underlying geology was not recorded as part of the current study, but it is likely that the greater variability of soil and bedrock type across the study area contributed to the observed variability of base cation changes.

As was found in the Countryside Survey 2007 (Emmett et al., 2010) and previous Snowdon surveys (McGovern et al., 2011), this study has shown no significant change in the C content of the soil, although there was a small increase in percentage C content . It has been suggested previously that warming (Bellamy et al., 2005) and changes in land-use and management can affect soil C (Smith et al., 2007; Kirk and Bellamy, 2010). Soils were sampled from a wide area of Snowdon, so it is not possible to ascertain the exact changes in grazing density for each site, but it is likely that the broad trend of decreasing grazing density has occurred throughout the area. 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 contributed 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 associated organic matter accumulation within the soil (e.g. De Vries et al., 2009). 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.

Overall, the soils on Snowdon appear to have been remarkably robust through a period of major environmental change. Despite a temperature increase of around 1.5 °C, no significant changes attributable to climate change could be detected. As there are currently only two time points of data, it is not possible to identify whether the rate of change of soil chemistry has varied throughout that period. With snapshot studies such as this, it is also possible that fluctuations in values could occur between collection times but remain undetected. Repeating this study in the future would potentially allow identification of patterns of change, albeit at a coarse resolution. More intensive ongoing monitoring of soil and soil solution chemistry at the Snowdon ECN site should permit assessment of the interannual variability of soil chemistry, and its shorter-term response to climate, deposition and land-use variations in future.

5. Conclusion

The absence of significant changes in soil composition on Snowdon, over a 40 year period in which major changes in key environmental drivers have occurred, provides important insights regarding the sensitivity of soils, and upland ecosystems, to long-term environmental change. Many studies have reported recent changes to soil composition in a UK context, based on large-scale surveys, but our more intensive study of a single location suggests a surprising degree of stability over a longer period. One explanation for this lack of change is that upland soils have been resilient to the (large) environmental pressures to which they have been exposed during the last century. A second interpretation is that soils have been damaged by previous changes, principally acidification, and have not yet recovered. A final possibility is that further damage occurred during the decades following the initial survey in 1968, and that subsequent recovery has been sufficient to reverse this damage, but not to return the system to pre-acidification conditions. This last interpretation appears most consistent with the results of more recent UK-level surveys showing increases in soil pH at broader scales over shorter periods. Site specific monitoring of changes in environmental drivers and their impacts, using historic and existing data, has an important role in providing detailed support to broader-scale survey-based approaches, and in elucidating the interactive effects of multiple pressures on specific ecosystems.

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References

Aber, J.D., Nadelhoffer, K.J., Streudler, P., Melillo, J. 1989. Nitrogen saturation in northern forest ecosystems. BioScience 39:378-386.

Ball, D.F., Mew, G., MacPhee, W.S.G.. 1969. Soils of Snowdon. pp. 69-107. Field Studies.

Bellamy, P. Loveland, P., Bradley, R., Lark, R., Kirk, G. 2005. Carbon losses from all soils across England and Wales 1978-2003. Nature 437**,** 245-248.

Beniston. M. 2003. Climatic Change in Mountain Regions: A Review of Possible Impacts. Climatic Change 59**,** 5-31.

Bradley, R.I., Milne, R., Bell, J., Lilly, A., Jordan, C., Higgins A. 2005. A soil carbon and land use database for the United Kingdom. Soil Use and Management 21**,** 363-369.

Brasher, S., Perkins, D.F.. 1978. The Grazing Intensity and Productivity of Sheep in the Grassland Ecosystem. In: Production Ecology of British Moors and Montane Grasslands (eds. Heal, O.W. and Perkins, D.F.). Springer-Verlag, Berlin.

Curtis, C.J., Evans, C.D., Helliwell, R.C., Monteith, D.T.. 2005. Nitrate leaching as a confounding factor in chemical recovery from acidification in UK upland waters. Environmental Pollution 137, 73-82.

Dale, J., Hughes, R.E. 1978. Sheep Population Studies in Relation to the Snowdonia Environment. In: Production Ecology of British Moors and Montane Grasslands (eds. Heal, O.W. and Perkins, D.F.). Springer-Verlag, Berlin.

de Vries, W., Reinds, G.J., Gundersen, P., Sterba, H. 2006. The impact of nitrogen deposition on carbon sequestration in European forests and forest soils. Global Change Biology 12**,** 1151-1173.

de Vries, W., Solberg, S., Dobbertin, M., Sterba, H., Laubhann, D., van Oijen, M., Evans, C., Gundersen, P., Kros, J., Wamelink, G.W.W., Reinds, G.J., Sutton, M.A. 2009. The impact of nitrogen deposition on carbon sequestration by European forests and heathlands. Forest Ecology and Management 258**,** 1814-1823.

Diaz, H.F., Grosjean, M., Graumlich, L. 2003. Climate Variability and Change in High Elevation Regions: Past, Present and Future. Climatic Change 59**,** 1-4.

Dick, J., Andrews, C., Beaumont, D.A., Benham, S., Brooks, D.R., Corbett, S., Lloyd, D., McMillan, S., Monteith, D.T., Pilgrim, E.S., Rose, R., Scott, A., Scott, T., Smith, R.I., Taylor, C., Taylor, M., Turner, A., Watson, h. 2011. A comparison of ecosystem services delivered by 11 long-term monitoring sites in the UK environmental change network. Environmetrics **22**, 639-648.

Emmett, B.A., Reynolds, B., Chamberlain, P.M., Rowe, E., Spurgeon, O., Brittain, S.A., Frogbrook, Z., Hughes, S., Lawlor, A.J., Poskitt, J., Potter, E., Robinson, D.A., Scott, A., Wood, C., Woods, C.. 2010. Countryside Survey: Soils Report from 2007 Technical Report No. 9/07. p. 192. NERC/Centre for Ecology and Hydrology.

Evans, CD, Monteith, DT and Harriman, R. 2001. Long-term variability in the deposition of marine ions at west coast sites in the UK Acid Waters Monitoring Network: impacts on surface water chemistry and significance for trend determination. The Science of the Total Environment 265, 115-129.

Evans, C.D.. 2005. Modelling the effects of climate change on an acidic upland stream. Biogeochemistry 74**,** 21-46.

Evans, C.D., Reynolds, B., Jenkins, A., Helliwell, R.C., Curtis, C.J., Goodale, C.L., Ferrier, R.C., Emmett, B.A., Pilkington, M.G., Caporn, S.J.M., Carroll, J.A., Norris, D., Davies, J., Coull, M.C. 2006a. Evidence that soil carbon pool determines susceptibility of semi-natural ecosystems to elevated nitrogen leaching. Ecosystems, 9, 453-462.

Evans, C.D., Caporn, S.J.M., Carroll, J.A., Pilkington, M.G., Wilson, D.B., Ray, N., Cresswell, N.. 2006b. Modelling nitrogen saturation and carbon accumulation in heathland soils under elevated nitrogen deposition. Environmental Pollution 143**,** 468-478.

Evans, C.D., Freeman, C., Cork, L.G., Thomas, D.N., Reynolds, B., Billett, M.F., Garnett, M.H., Norris, D.. 2007. Evidence against recent climate-induced destabilisation of soil carbon from 14C analysis of riverine dissolved organic matter. Geophys. Res. Lett.. 34**,** L07407.

Galloway, J.N., Townsend, A.R., Erisman, J.W., Bekunda, M., Cai, Z., Freney, J.R., Martinelli, L.A., Seitzinger, S.P., Sutton, M.A. 2008. Transformation of the Nitrogen Cycle: Recent Trends, Questions, and Potential Solutions. Science 320**,** 889-892.

Hagedorn, F., Mulder, J., Jandl, R.. 2010. Mountain soils under a changing climate and land-use. Biogeochemistry 97**,** 1-5.

Hall, J., Heywood, L., Smith, R. 2004. Trends in critical loads exceedances for acidity and nutrient nitrogen for the years 1995-97, 1998-2000, 1999-2001 & 2010. Report and Figures to Defra. Available from http://cldm.defra.gov.uk/Reports_CLDM.htm.

Hettelingh, J.P., Posch, M., Slootweg, J., Reinds, G., Spranger, T., Tarrason, L. 2007. Critical Loads and Dynamic Modelling to Assess European Areas at Risk of Acidification and Eutrophication. Water, Air, and Soil Pollution 7**,** 379-384.

Horswill, P., O'Sullivan, O., Phoenix, G.K., Lee, J.A., Leake, J.R.. 2008. Base cation depletion, eutrophication and acidification of species-rich grasslands in response to long-term simulated nitrogen deposition. Environmental Pollution 155**,** 336-349.

Hurrell, J.W. 1995. Decadal trends in the North Atlantic oscillation: regional temperatures and precipitation. Science 269, 676-679.

Janssens, I.A., Dieleman, W., Luyssaert, S., Subke, J.A., Reichstein, M., Ceulemans, R., Ciais, P., Dolman, A.J., Grace, J., Matteucci, G., Papale, D., Piao, S.L., Schulze, E.D., Tang, J., Law, B.E. 2010. Reduction of forest soil respiration in response to nitrogen deposition. Nature Geoscience 3, 315-322.

Kirby, K.J., Smart, S.M., Black, H.I.J., Bunce, R.G.H. 2005. Long term ecological change in British woodland (1971-2001). English Nature Research Reports No. 653.

Kirk, G.J.D., Bellamy, P.H.. 2010. Analysis of changes in organic carbon in mineral soils across England and Wales using a simple single-pool model. European Journal of Soil Science 61**,** 406-411.

Kirk, G.J.D., Bellamy, P.H., Murray Lark, R. 2010. Changes in soil pH across England and Wales in response to decreased acid deposition. Global Change Biology 16, 3111–3119.

Kirkham, F. 2001. Nitrogen uptake and nutrient limitation in six hill moorland species in relation to atmospheric nitrogen deposition in England and Wales. Journal of Ecology 89**,** 1041-1053.

Klumpp K, Fontaine S, Attard E, Le Roux X, Gleixner G, Soussana JF. 2009**.** Grazing triggers soil carbon loss by altering plant roots and their control on soil microbial community. Journal of Ecology 97, 876-885.

Lee, J., Caporn, S. 1998. Ecological effects of atmospheric reactive nitrogen deposition on semi-natural terrestrial ecosystems. New Phytologist 139, 127-134.

Lloyd, D., Turner, A., Skates, J., Easter, J., Bowmaker, V. 2012. Snowdon Environmental Change Network: 15 Years of monitoring on Yr Wyddfa/Snowdon. Countryside Council for Wales, Bangor.

Maskell, L.C., Smart, S.M., Bullock, J.M., Thompson, K., Stevens, C.J. 2010. Nitrogen deposition causes widespread loss of species richness in British habitats. Global Change Biology 16, 671-679.

McGovern, S., Evans, C., Dennis, P., Walmsley, C., McDonald, M. 2011. Identifying drivers of species compositional change in a semi-natural upland grassland over a 40 year period. Journal of Vegetation Science 22, 346–356.

Monteith, D.T., Evans, C.D.. 2005. The United Kingdom Acid Waters Monitoring Network: a review of the first 15 years and introduction to the special issue. Environmental Pollution 137, 3-13.

Morecroft, M.D., Bealey, C.E., Beaumont, D.A., Benham, S., Brooks, D.R., Burt, T.P., Critchley, C.N.R.., Dick, J., Littlewood, N.A., Monteith, D.T., Scott, W.A., Smith, R.I., Walmsley, C., Watson, H. 2009. The UK Environmental Change Network: Emerging trends in the composition of plant and animal communities and the physical environment. Biological Conservation 142, 2814-2832.

NEGTAP. 2001. Transboundary air pollution: acidification, eutrophication and ground-level ozone in the UK. Department of the Environment, Transport and the Regions.

Parker, D.E., Legg, T.P., Folland, C.K.. 1992. A new daily Central England Temperature Series, 1772-1991. International Journal of Climate, 12, 317-342.

Perkins, D.F. 1978. Snowdonia Grassland: Introduction, Vegetation and Climate. In: Production Ecology of British Moors and Montane Grasslands (eds. Heal, O.W. and Perkins, D.F.). Springer-Verlag, Berlin.

Phoenix, G.K.., Leake, J.R., Read, D.J., Grime, J.P., Lee, J.A.. 2004. Accumulation of Pollutant Nitrogen in Calcareous and Acidic Grasslands: Evidence from N Flux and 15N Tracer Studies. Water, Air, and Soil Pollution 4, 159-167.

R Development Core Team. 2010. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Reay, D.S., Dentener, F., Smith, P., Grace, J., Feely, R.A.. 2008. Global nitrogen deposition and carbon sinks. Nature Geoscience 1**,** 430-437.

Reuss, J. O., Johnson, D. W. 1986. Acid Deposition and the Acidification of Soils and Waters, Springer Verlag, New York.

RoTAP. 2009. Acidification, Eutrophication, Ground Level Ozone and Heavy Metals in the UK. Defra, London

Smart, S.M.., Bunce, R.G.H., Marrs, R., LeDuc, M., Firbank, L.G., Maskell, L.C., Scott, W.A., Thompson, K., Walker, K.J.. 2005. Large-scale changes in the abundance of common higher plant species across Britain between 1978, 1990 and 1998 as a consequence of human activity: Tests of hypothesised changes in trait representation. Biological Conservation 124**,** 355-371.

Smith, P., Chapman, S.J., Scott, W.A., Black, H.I.J., Wattenbach, M., Milne, R., Campbell, C.D., Lilly, A., Ostle, N., Levy, P.E., Lumsdon, D.H., Millard, P., Towers, W., Zaehle, S., Smith, JU.. 2007. Climate change cannot be entirely responsible for soil carbon loss observed in England and Wales, 1978–2003. Global Change Biology 13**,** 2605-2609.

Stevens, A., Wesemael, B.V.. 2008. Soil organic carbon dynamics at the regional scale as influenced by land use history: a case study in forest soils from southern Belgium. Soil Use and Management 24**,** 69-79.

Stevens, C.J., Thompson, K., Grime, J.P., Long, C.J., Gowing, D.J.G.. 2010. Contribution of acidification and eutrophication to declines in species richness of calcifuge grasslands along a gradient of atmospheric nitrogen deposition. Functional Ecology 24, 478-484.

Sykes, J.M., Lane, A.M.J. 1996. The UK Environmental Change Network: Protocols for standard measurements at terrestrial sites. Stationery Office, London.

Thuiller, W. 2007. Biodiversity - Climate change and the ecologist. Nature 448, 550-552.

Tilman, D., Lehman, C.. 2001. Human-caused environmental change: Impacts on plant diversity and evolution. Proceedings of the National Academy of Sciences 98**,** 5433-5440.

Waldrop, M.P., Zak, D.R., Sinsabaugh, R.L.. 2004. Microbial community response to nitrogen deposition in northern forest ecosystems. Soil Biology and Biochemistry 36, 1443-1451.

Willems, J.H., Peet, R.K., Bik, L. 1993. Changes in chalk-grassland structure and species richness resulting from selective nutrient additions. Journal of Vegetation Science 4, 203-212

Figure Captions

Fig. 1. a) Mean annual temperature residual compared to 1966 -1977 mean, based on Central England Temperature Record (Parker et al., 1992; http://www.metoffice.gov.uk/hadobs/hadcet/), ; **b)** total sheep grazing numbers for the County of Gwynedd (adjusted for boundary changes in 1996 based on area change); **c)** UK total sulphur emissions and **d)** UK nitrogen emissions from the UK National Atmospheric Emissions Inventory (http://naei.defra.gov.uk**); e)** Mean temperature residual measured at the Snowdon ECN automatic weather station compared to the mean of an earlier 1966 -1977 monitoring period at the same location; **f)** mean weekly sheep numbers at the ECN site; **g)** mean annual wet and dry sulphate concentrations and **h)** mean annual total inorganic nitrogen concentrations in precipitation between 1996/7 and 2011 at the Snowdon ECN site. (Lloyd et al., 2012)

Fig. 2. Measured soil pH in 2008 versus measured pH in 1968. Solid line represents 1:1 line (i.e. no change in pH), dashed line represents best fit based on linear regression ($R^2 = 0.11$, $p = 0.05$, n=35).

Fig. 3.

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.

Fig. 4.

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.

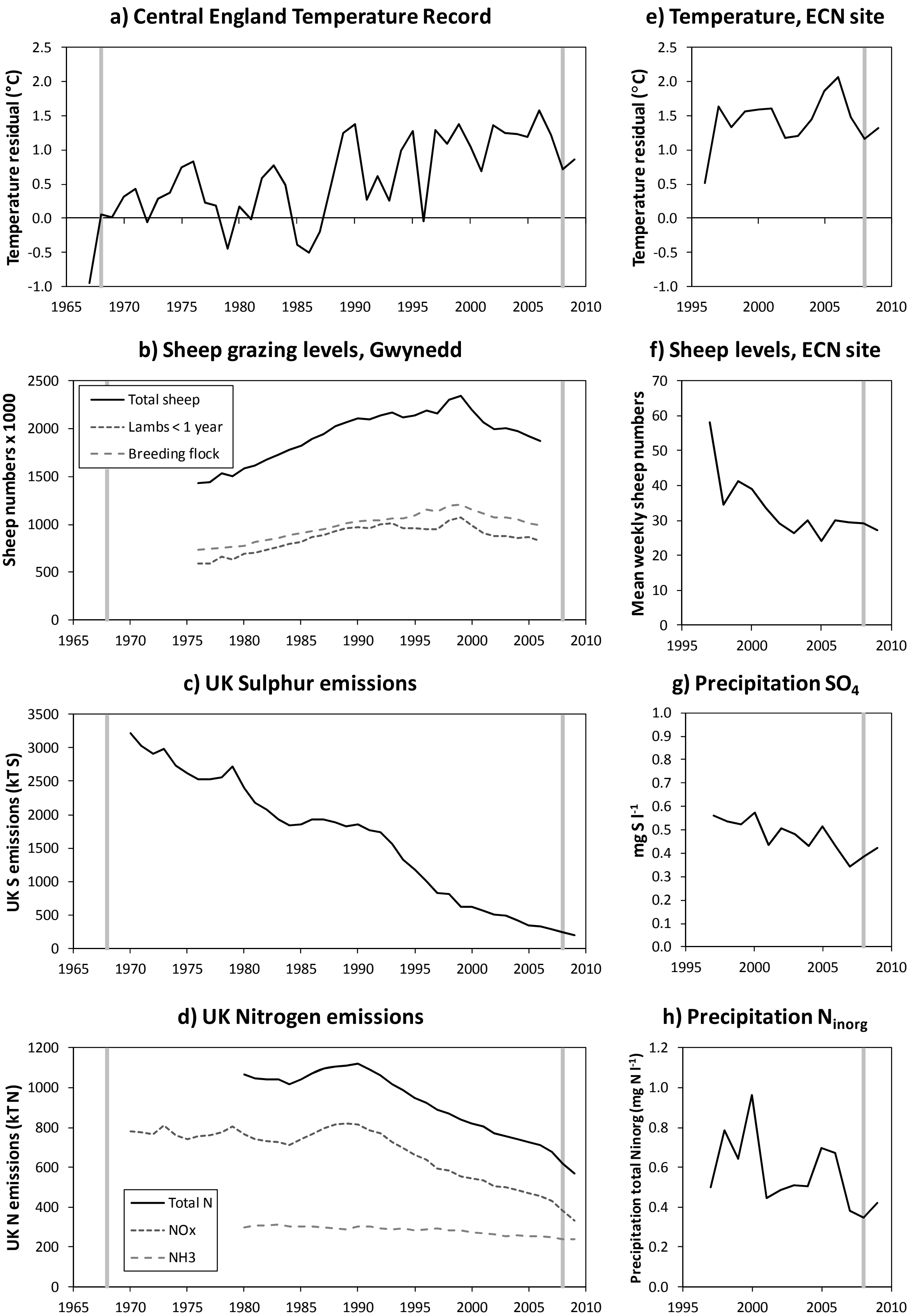
a) Central England Temperature Record
Temperature residual (°C)
e) Temperature, ECN site
Temperature residual (°C)
b) Sheep grazing levels, Gwynedd
Sheep numbers x 1000
Total sheep
Lambs < 1 year
Breeding flock
f) Sheep levels, ECN site
Mean weekly sheep numbers
c) UK Sulphur emissions
UK S emissions (kT S)
g) Precipitation SO_4
mg S l^{-1}
d) UK Nitrogen emissions
UK N emissions (kT N)
Total N
NOx
NH3
h) Precipitation N_{inorg}
Precipitation total Ninorg (mg N l^{-1})

Fig. 1.

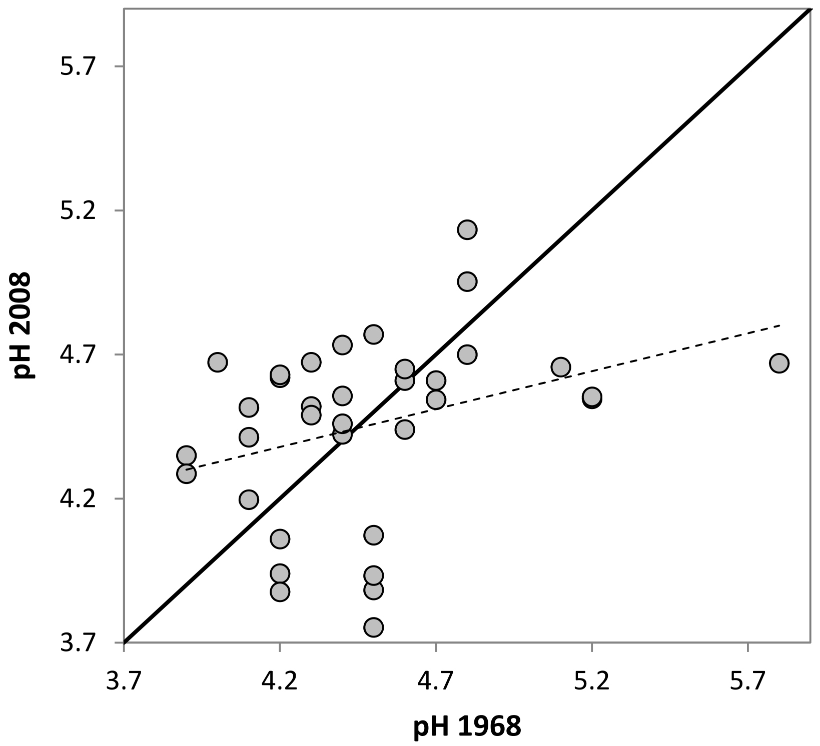

Fig. 2.

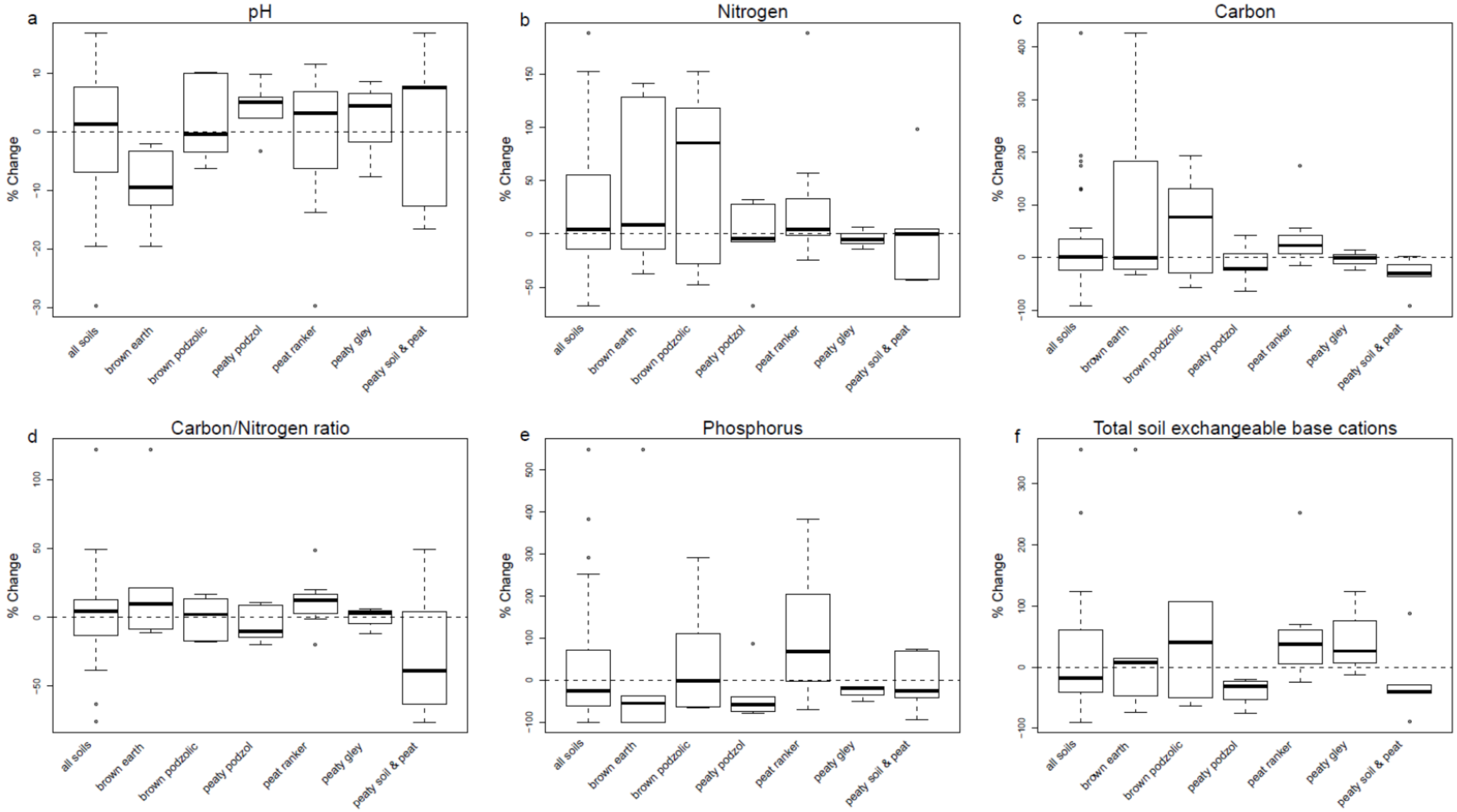
a
pH
b
Nitrogen
c
Carbon
d
Carbon/Nitrogen ratio
e
Phosphorus
f
Total soil exchangeable base cations
% Change
all soils
brown earth
brown podzolic
peaty podzol
peat ranker
peaty gley
peaty soil & peat

Fig. 3.

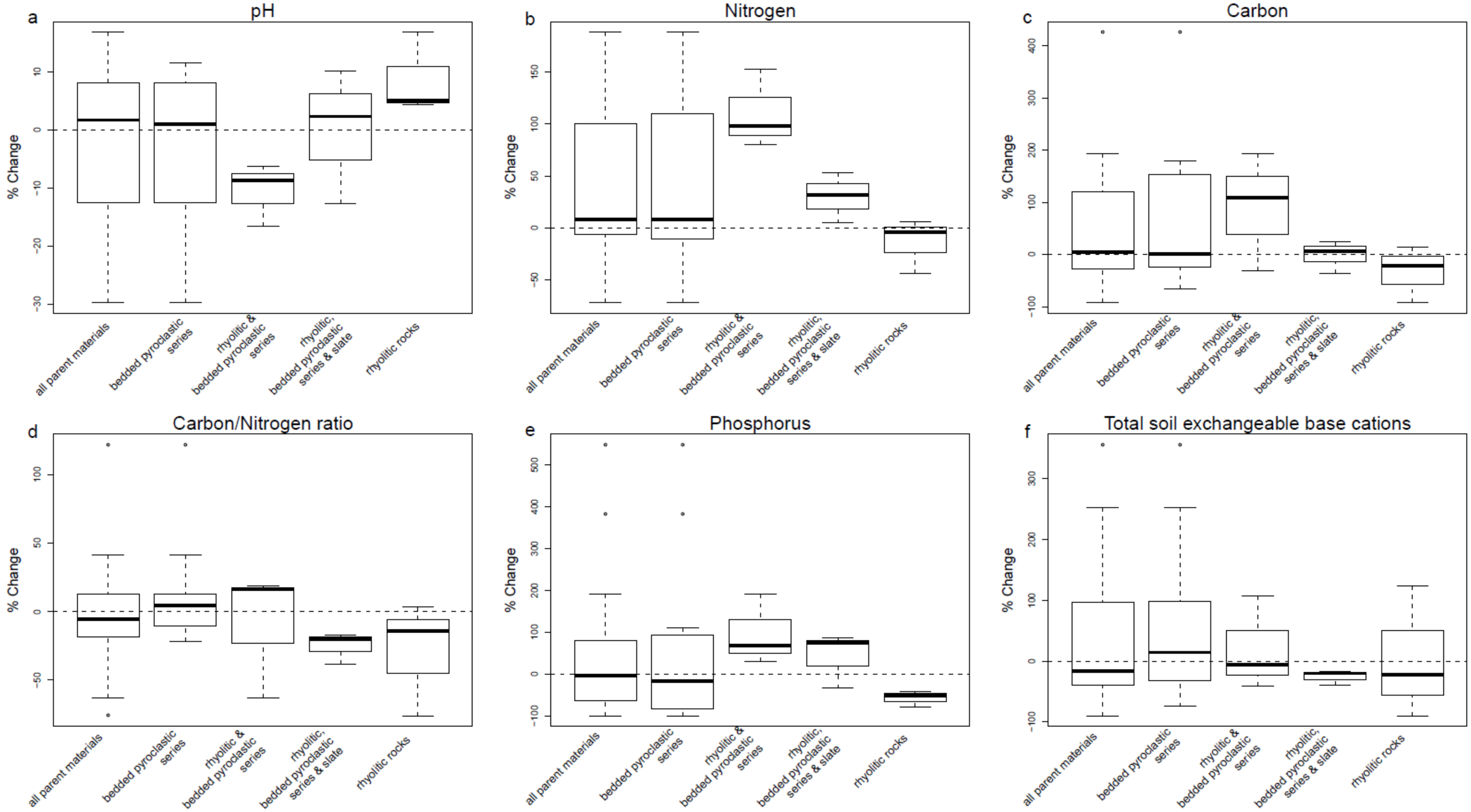
a
pH
b
Nitrogen
c
Carbon
d
Carbon/Nitrogen ratio
e
Phosphorus
f
Total soil exchangeable base cations
% Change
all parent materials
bedded pyroclastic series
rhyolitic & bedded pyroclastic series
rhyolitic, bedded pyroclastic series & slate
rhyolitic rocks

Fig. 4.

Table 1
Results of paired t-tests to determine the effect of sampling year on soil chemical variable. Mean values are displayed and values in parentheses are one standard error of the mean (*n*=36) ((‡) *n*=35). Asterisk indicate degree of significance n.s. $P \geq 0.05$, * $P<0.05$, ** $P<0.01$, *** $P<0.001$.

Year	**Exchangeable cations (me 100g^{-1})**							**Extractable P_2O_5 (mg 100g^{-1})**	**C (calculated wt %)**	**Total N (wt %)(‡)**	**C/N Ratio(‡)**
	pH	**Ca(‡)**	**K**	**Na**	**Mg**	**Mn**	**Total**				
1968	4.54	19.44	0.55	0.81	1.37	0.26	3.911	2.99	19.75	1.40	13.09
	(0.086)	(2.064)	(0.072)	(0.083)	(0.258)	(0.062)	(0.440)	(0.459)	(2.029)	(0.125)	(0.713)
2008	4.46	20.24	1.08	0.36	1.40	0.03	3.72	3.29	20.72	1.60	13.07
	(0.052)	(1.524)	(0.290)	(0.030)	(0.171)	(0.006)	(0.441)	(0.534)	(1.557)	(0.119)	(0.728)
P	n.s.	n.s.	**	***	n.s.	***	n.s.	n.s.	n.s.	n.s.	n.s.