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Trends in soil quality monitoring data in the Waikato region 1995-2018

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Executive Summary

Waikato Regional Council monitors the quality (health) of the major land uses and soil in the region. Initial proof of concept for soil quality commenced in 1995, lead by Landcare Research, proceeded the “500 soils” research project (2000-2003). Regional councils under the Land Monitoring Forum, including WRC, took over monitoring after the initial round. Trends over time can now be analysed as most sites have now been sampled at least 4 times. This report presents data and trends in the seven indicators representing soil quality for the period 1995-2018 and discusses the main soil quality issues facing the Waikato Region.

Changes in soil quality for different soil types and land use types over time and across the Waikato region have been identified. Eleven percent of managed soil quality sample sites (farmed or manged forest but not indigenous bush sites) in the region met all seven indicators in 2018. However, this result is down from a high of 18% in 2006.

The main soil quality issues in the Waikato region are soil compaction, excessive nutrient levels, and loss of soil organic matter (SOM).

Surface compaction is an issue for pastoral, horticultural, arable and forestry land uses in the Waikato. Surface compaction is assessed using the macroporosity indicator. Only 35% of pastoral sites meet the lower macroporosity target of 10% macropores in 2018, but this is an improvement of 10% on the previous five years. This trend needs to continue to meaningfully improve soil quality. However, improvement could be jeopardised if further intensification occurs, especially if this is combined with a couple of wet winters. Although 75% of arable sites meet the lower macroporosity target of 10% (volume of macropores) in 2018, average macroporosity has decreased over the previous five years. Similarly, sites meeting the low macroporosity target for forestry have decreased from 100% to 75% due to tree harvested disturbing the soil and heavy machinery compacting it. These sites should recover over time once the next generation of trees are established. Macroporosity, therefore compaction, has been stable for horticultural sites.

Compaction may result in decreased soil infiltration capacity and generation of surface runoff; increased peak and average stream flows resulting in increased annual flood exceedance probability; transport of contaminants including sediment, nutrients, and pathogens; and localised flooding and bank erosion. In addition, plant uptake of N and P can be reduced in compacted soils due to shallower rooting depths and reduced available N concentrations.

The effects of soil compaction may last for several years unless remedial action is taken. Where compaction is moderate, recovery can be relatively rapid (e.g. within 18 months). However, complete recovery from a more severe compaction or pugging event, with lower macroporosity values, may take many years. Damage to the soil by grazing animals can be minimised by management of livestock and land, including reducing stocking density, moving livestock off wet pasture onto hard standings or into housing, and reducing the length of the grazing season. Precision agriculture techniques should be followed when using machinery for arable and forestry operations and machinery kept off wet soils.

Excessive nutrients are an issue for some pastoral land, horticultural and arable land uses. Results showed average Olsen P and total N in pastoral land in dairy, horticultural and arable land are currently within the excessive or high categories for these indicators. Although the trend in the number of sites meeting the Olsen P and total N targets is stable for pasture and arable land uses, likely due to a combination of economic, social and regulatory pressures. The decrease in the number of sites meeting total N and Olsen P targets for horticulture can be explained to some extent by the conversion of lower intensity apple orchards to higher intensity kiwifruit orchards.

Increased fertility appears to be increasing microbial activity, which is contributing to nitrogen loading onto soils, and specifically soil organic matter, but there also appears to be a concurrent increase in nitrogen losses. Excessive nutrient levels in soils can lead to an increased risk of transfer of N and P to water bodies where they can contribute to changes in water pH, the composition of local biological communities, the formation of algal blooms, or directly impact human and animal health. The greatest risk of P loss is on soils that are poorly drained, have lower structural resilience, or are on slopes, whereas the greatest risk of N loss is on very well- and excessively well-drained soils. When linked together, surface compaction and excessive nutrient concentrations in pasture have been linked to modified soil hydrological behaviour and, ultimately, the deterioration of water quality in ground and surface waters.

Diffuse contamination of surface waters with P and N could be reduced by applying no more than the amount of nutrient sources (e.g fertiliser, supplement, effluent) needed for production, managing critical source areas better, reducing surface runoff and riparian planting.

Loss of SOM is an issue for arable land use. Arable land has the lowest total C average and levels are continuing to decrease despite total C increasing for other land uses. Carbon is lost due to cultivation exposing C in soil aggregates to oxidation, increased microbial activity increasing losses due to their respiration, and reduced inputs of plant residues. As a result, soil biota diversity, N regulation, aggregate stability, infiltration, drainage, and airflow are reduced in arable soils compared with pasture or native soils. The C:N ratio also decreased or narrowed as average total N decreased more slowly than total C, potentially leading to increased risk of N loss.

SOM is considered a key soil attribute as it affects many physical, chemical and biological properties that control soil services such as productivity, the adsorption of water and nutrients, and resistance to degradation. Low SOM is associated with reduced aggregate stability, infiltration, drainage, airflow, microbial biomass, microbial activity, and nutrient mineralisation due to a shortage of energy sources and loss of habitat. Low SOM results in less diversity in soil biota with a risk of the food chain equilibrium being disrupted, which can cause increases in accumulation of toxic substances, plant pests and diseases. Of significance to the Waikato and Waipā catchments is SOM's role in regulating nitrogen in soil. SOM state is assessed by measuring total carbon (total C).

In arable systems, adding manures, applying no more than the amount of nutrients, e.g. fertiliser needed for production, the return of plant material and crop rotation can all help reduce the loss of SOM. However, re-establishment of pasture appears the most practical and cost-effective method of recovering SOM in these systems, but accumulation of soil carbon is slow and recovery of SOM to pre cultivation levels is likely in the range 14-45 years.

Considerable conversion of land from planted radiata pine forest to pasture has taken place on Pumice Soils. Pumice Soils are very 'light' with weak structure and erode easily when disturbed. Impacts of this land use intensification can include loss of soil carbon and SOM, decreased aggregate stability, increased surface compaction and surface crusting with the associated issues of decreased water infiltration and storage, and increased overland flow. Increased overland flow can result in soil erosion and the transfer of nutrients, sediment, pathogens, organic matter, and other contaminants to waterways. The impact of intensification on the biological, physical, and chemical condition of Pumice Soils is likely to be greater than for Allophanic or Granular Soils, as these are both weathered volcanic soils and better suited for pastoral land use.

Soils in 2018 under native vegetation were, on average, acidic (pH 5.1), high in total C (mineral soils 16.7%), low in Olsen P (8 mg/L), low in bulk density (0.52 t/m^3) and had high macroporosity (28% v/v). Soils under production forestry had generally similar characteristics to soils under native but had lower total C (8.4%). Soils under pasture were, on average, less acidic (pH 5.9), had higher Olsen P (45 mg/L) levels and had lower macroporosity (9%) than production forestry and native soils. Soils under horticulture were on average the least acidic (pH 6.6) of the land uses measured, with excessive levels of Olsen P, but were otherwise similar to pasture. Soils under arable land use were the most different from soils under native vegetation. These had low total C (4.5%), indicating loss of SOM, low AMN, indicating little reserve of mineralisable nitrogen due to loss of SOM, very high Olsen P (88 mg/L), indicating excessive fertility, and high bulk density (0.94 t/m^3), indicating compaction. Arable soils were also less acidic (pH 6.2) than soils under other land uses.

The representativeness of the soil quality monitoring sites was assessed. Compared to land area in each land use and soil type category in 2018, the native vegetation, pasture, Brown Soils, Organic Soils, Podzol Soils, Pumice Soils, Recent Soils and Ultic Soils were found to be underrepresented. The representativeness of the dataset could be improved by increasing the number of native sites by 28 to 43 sites and the number of pasture sites by 5. Ideally, these should be from soil orders that are currently under represented; Brown Soil should be increased by 1 site to 19 sites; Organic Soils should be increased by 2 to 7 sites; Recent Soils should be increased by 9 to 15 sites; Podzol Soils should be increased by 9 to 14 sites; Pumice Soil should be increased by 12 to 38 and Ultic Soils should be increased by 4 to 7 sites. The total number of soil quality monitoring sites would need to increase from the 154 currently active in 2018 to about 190-200 for the sampling programme to be representative of all the major land uses and soil orders in the Waikato region.

1 Introduction

Waikato Regional Council (WRC) recognises that the region's economy and people's wellbeing depend on our natural capital, including soils, and has legislative responsibility to manage the soil resource. An established soil quality monitoring programme provides information for State of the Environment (SOE) reporting, policy development, and helps in understanding the interactions between soil and water. The soil quality trend measurements enable assessment of the sustainability of current land use activities and the effectiveness of WRC policies by providing evidence of change or stability.

Soil consists of a complex combination of minerals, organic matter, organisms, air and water. Soils with high soil quality are considered healthy as they support important functions such as agricultural production, water filtration and storage, flood mitigation, nutrient and carbon storage, plant growth, biological diversity, and can act as a barrier to below surface contamination (Ministry of Primary Industries, 2015). Soils with high soil quality are more resilient and durable to the pressures associated with human activities and are quick to recover if damaged. Typically, a soil with high soil quality has low leakage of nutrients and contaminants, low rates of erosion, high levels of biodiversity, will capture and hold water, and can sustain high levels of production. These soils tend to be resistant to disturbance from intense storms and land use change.

Preliminary development of the soil quality programme was carried out in collaboration with Manaaki Whenua – Landcare Research from 1995 with regional coverage achieved by 2005. This programme is aligned with national soil quality monitoring as established and administered through the Land Monitoring Forum (LMF). The quality of the region's soils is assessed by calculating the proportion of sites meeting targets associated with the seven soil quality measures (indicators) and the direction of trends. This report presents trends in the data since 1995 and discusses the main soil quality issues facing the Waikato Region.

1.1 Regional programme objectives

By undertaking soil quality monitoring, WRC will be able to comply with the requirements of the Resource Management Act 1991 and Environmental Reporting Act 2015, can keep our community informed of issues facing our productive land, guide land users in their management practices, and develop well informed and appropriate policies and rules to help address issues as they emerge.

As soils take a long time to form, they should be regarded as a finite resource and natural capital by resource managers. Healthy soils with suitable and sustainable land uses are needed to achieve the Waikato Regional Council's mission to build a Waikato region that has a healthy environment, a strong economy, vibrant communities, while the rural economy can benefit greatly from the sustainable use and management of its soil resources.

1.1.1 The Soil Quality Monitoring Programme has three key objectives:

1. Develop and implement a long-term soil quality monitoring programme that represents the state of soil quality and identifies soil quality changes for different soils and land uses over time and across the region. Results are utilised for State of the Environment reporting and policy development.
2. Develop a database containing soil, site descriptions along with periodic measurements of soil chemical, physical and biological indicators used to monitor changes in soil quality.
3. Provide an early-warning system to identify the effects of primary production land uses on long-term soil productivity and health (physical, chemical, biological). Relate changes in soil quality indicators to land use and land use practices, identifying those having the

greatest impact on soil quality and the wider environment. Track specific, identified issues relating to the effects of land use on long-term soil productivity.

1.2 Reasons for soil quality monitoring

Regional councils are required to comply with the requirements of the Environmental Reporting Act 2015 and to manage natural and physical resources in such a way that enables the purpose of the Resource Management Act (RMA) and its amendments to be achieved. The RMA has a purpose of sustainable management, which incorporates the requirement to maintain the life supporting capacity of land and ecosystems. Soils are living natural capital ecosystems and support a range of life forms, hence the concept of maintaining soil health is embodied in the purpose of the RMA. In addition, Section 35(2) of the RMA requires regional councils to monitor the state of the environment.

The soil ecosystem has multiple roles in the environment including filtering water, regulating nutrients, water and greenhouse gases, maintenance of productivity, and habitat provision (Ministry of Primary Industries, 2015). Poor soil quality results in lower agricultural yields, a less resilient soil and land ecosystems and greater contamination of adjacent water bodies.

Soils can also be viewed in terms of degradation and depletion. Soil degradation is a deleterious change or ecological disturbance to the soil. Soil nutrient depletion occurs when the factors which contribute to fertility are removed or where the conditions which support soil's fertility are not maintained. Degradation and depletion of soils have adverse effects on soil quality, plant productivity, and ecosystem function.

Soils can be degraded in several ways:

1. Structurally, by physical compaction and loss of aggregate stability. Compacted soils are often slow draining, become water-logged when wet resulting in poor aeration which is unsuitable for plant roots and soil animals. Compaction results in lower yields, higher production costs, and reduced profitability. Increased run-off may reduce water quality and accelerate erosion.
2. Through accelerated erosion, when the rate of soil loss is higher than the rate of soil formation. Soil erosion affects SOC dynamics by slaking and breakdown of aggregates, preferential removal of C in surface runoff or wind, redistribution of C over the landscape, and mineralisation of displaced or redistributed C (Lal, 2013).
3. Though elemental imbalance or nutrient depletion, e.g. many Pumice Soils are limited in cobalt and other nutrients.
4. Through soil acidification, salinity, or desertification. These are major causes of degradation in other parts of the world, but very localised in New Zealand.

Depleted soils have lost components essential for healthy plant and soil biology:

1. They may be depleted in nutrients, because nutrient stocks are not being replaced as fast as they are removed.
2. Soils may become too acid for some crops if insufficient lime is applied to counter natural acidification processes.
3. Soils depleted in organic matter have less ability to retain nutrients in the topsoil, are more prone to rapid structural decline, and have less capability to supply plant nutrients from organic reserves. If nutrients are not retained within soils, they can contaminate surface and groundwater.
4. Soils low in biological activity are less able to maintain healthy microbial communities, detoxify wastes, and degrade contaminants.

1.3 Soil quality monitoring programme design

The WRC soil quality monitoring programme is a screening tool or early warning system designed to gather a large amount of information quickly and at a low cost to inform detailed environmental assessment of the region's soils. Currently there are 154 long-term monitoring sites (Table 1, Figure 1). Soil quality monitoring sites were chosen and sampled according to the methods set out in the national guidelines of the LMF manual (Hill & Sparling, 2009). This manual sets guidance for sample size, representativeness, sampling procedures, analytical methods, target values for results, and archiving of samples.

The sites chosen for the WRC soil quality monitoring programme represent dominant soils and land uses, and also include sites on sensitive soils such as peat soils, sites capturing the effects of land use change (e.g. production forestry to pasture) and sites with specific land use practices (such as organic farming). Research (Hill et al., 2003) determined that major Land Use Type and Soil Order contributed to the variability of soil quality indicators at a national scale, although sampling land use and soil combinations of small areas can be justifiable if they are of local concern and/or considered to be "at risk" (Hill et al., 2003). Thus, the sites chosen for the WRC soil quality monitoring programme represent dominant soils and land uses, and also include sites on sensitive soils such as peat soils, sites capturing the effects of land use change (e.g. production forestry to pasture) and sites with specific land use practices (such as organic farming).

Indigenous vegetation has grown at native sites from prehuman times, indicating current soil quality indicator values are at equilibrium for this land use (cover). Therefore, target values for native systems are not defined (Hill & Sparling, 2009; Hill et al., 2003). However, native sites provide valuable baseline information on which the effects of land use change on soil characteristics can be assessed. Additionally, changes in soil parameters over time in indigenous systems can indicate the extent that these systems are being influenced by human activity.

Changes to land use and loss of sites due to a variety of reasons are recorded. New sites are established to preserve and improve the representativeness of the dataset. However, for minor land uses and soil types, the numbers of sites need to be increased to enable statistical analysis. The representativeness of the current soil quality monitoring sites has been assessed in the previous soil quality technical report (Taylor et al., 2017), which found the representativeness of the dataset can be improved by increasing the number of native sites by 33 to 43 sites; Pasture sites by 5 to 59 sites; Brown Soil sites by 1 to 19 sites; Recent Soil sites by 9 to 15 sites; Podzol sites by 9 to 14 sites; Pumice sites by 12 to 39 and Ultic Soil sites by 4 to 7 sites. The total number of soil quality monitoring sites would need to increase to about 190-200 for the sampling programme to be representative of all the major land uses and soil orders in the Waikato region (Table 1). Expanding soil quality monitoring to 190 sites is expected to take several years.

Table 1: Land area of each land use and soil type, and representativeness of the current soil quality monitoring sites in 2018 with recommendations.

Land use	Regional area (ha)	Number of sites	Percent of sites	Percent of region	Number of additional samples recommended
Native	706,000	15	10	28.2	28
Forestry	285,000	19	12	11.4	No change
Arable	11,000	16	10	0.4	No change
Horticulture	2,000	12	8	0.1	No change
Pasture	1,420,000	83	54	56.9	5
Conversion of forestry to pasture	35,000	9	6	1.3	No change
Allophanic	465,280	55	36	18.5	No change
Brown	310,187	18	12	12.3	1
Gley	184,635	20	13	7.4	No change
Granular	172,326	16	10	6.9	No change
Organic (Peat)	108,319	5	3	4.3	2
Podzol	231,409	5	3	9.2	9
Pumice	625,297	26	17	24.9	12
Recent	248,642	6	4	9.9	9
Ultic	110,781	3	2	4.4	4
Total in the region	2,461,800	154			

Soil quality monitoring sites have been re-sampled over time to identify trends for each soil measurement. For continuity, the LMF recommended that a few of the sites be sampled each year, so to resample all the sites in a 5 to 10-year cycle. For the WRC soil quality monitoring programme, about 30 sites (20%) are sampled annually, meaning that it takes five years to sample all 154 current sites. Note that it took several years to build the number of sites to 154 from the initiation of the programme.

Figure 1: Map of soil quality site locations.

1.4 Soil Quality Indicators

Soil quality is the chemical, physical, and biological condition of a soil type for a given land use. There is currently no single measure of soil quality because there are many things about the soil that affect its quality rating – the fertility, physical condition, amount of humus, and biology, while a change in a soil quality characteristic can improve the ability of a soil to provide one type of service but decrease its ability to provide another at the same time. This means there are trade-offs, e.g. increasing nutrients can increase production, but it also increases the risk of loss of nutrients to waterways and associated algal blooms. After preliminary testing (1995-1998) followed by three years of trials (1998-2001), over more than 500 sites, the Land Monitoring Forum agreed on seven key measurements, which are termed indicators (Hill et al., 2003):

1. Olsen P: Olsen P (weight/volume) is the method used to derive the concentration of phosphorous that is available for plant uptake,
2. pH: a measure of soil acidity,
3. total carbon (C): a measure of soil organic matter and carbon stocks,
4. total nitrogen (N): a measure of soil organic matter and nitrogen stocks,
5. anaerobically mineralised N (AMN): a measure of mineralisable nitrogen used to assess soil microbial health and how much organic N is available to plants,
6. bulk density: a measure of physical condition,
7. macroporosity at -10 kPa (shortened to macroporosity for this publication): a measure of soil pores that air and water can use to enter the soil. Compacted soils reduce water or air penetration, restrict root growth and do not drain easily, so have increased potential for run-off carrying sediment, nutrients, and contaminants to surface waters.

Note on Olsen P: Chemical data for Olsen P was converted from a gravimetric basis (weight/weight), as reported by the laboratory, to a volumetric basis (weight/volume) by multiplying by the bulk density

The various properties monitored focus on the dynamic aspects of soil quality and are based on the fitness of the soil for its particular use. For each site, data from the seven key soil quality indicators are compared against target ranges specific to soil order and land use and the number of times a value fails to meet the target ranges recorded. Targets do not exist for native sites as these sites have been under this land use long-term and the indigenous vegetation appears thriving. Values from native sites can be considered ‘background’ or ‘baseline’.

Comparison of soil properties at individual sites over time are also used to assess the extent and direction of change in soil quality characteristics. Overall, soil quality is calculated by the proportion of all indicators that met the target range using the formula:

$P = I / N \times 100$

where P is the proportion of sites not meeting the target for that particular indicator, I is the count of sites not meeting the target range, and N is the total number of sites sampled. Data can also be grouped by land use category to help identify areas of concern, and the proportion of sites meeting or not meeting soil quality targets calculated using the formula:

$Pi = Ic / Ni \times 100$

where Pi is the proportion of sites not meeting the target for that indicator, Ic is the count of sites not meeting the target range, and Ni is the total number of sites sampled for that indicator.

2 Soil quality monitoring programme method

2.1 Sampling and analysis

Soils were sampled at each monitoring site following the methods in the LMF national guidelines (Hill & Sparling, 2009). The same sampling methodology was followed to ensure consistency between results gathered from different regions and over time. The first time a site was sampled, a soil profile pit was dug on site to confirm soil type and to provide a basic soil profile description. For the first and subsequent samplings, a transect on a visually uniform strip of land with at least 10 m clearance from obstructions or constructions was accurately defined to enable relocation for future samples. The LMF manual recommends compositing 25 soil cores over a 50 m transect using a tube auger. In practice, more samples were needed in order to have enough soil left over after analysis to archive. Archived soil samples have proved extremely valuable when testing a change in analytical methods or testing an additional soil measurement on top of the existing seven key indicators. In addition, three undisturbed core samples for physical analyses were taken at 15m, 30m and 45m positions along the transect. These distances were approximate as it was necessary to avoid cow dung and areas not representative of the site. The individually numbered core liner, 75 mm depth by 100 mm diameter was placed on the surface of the soil from which the core sample was to be taken. The number of the liner was recorded in the site notes. The liner was then pressed into the soil, pushing downwards on the ring, e.g. with a block of wood. The field staff then cut round the outer part of the liner with a sharp knife and continue pressing down until the soil was approximately 5 mm below the top of the liner. The liner with the intact core of soil was carefully dug out of the surrounding soil, taking care not to break away the soil from the base of the liner. Excess soil below the bottom of the liner was cut off using a large spatula or knife. The entire liner and core were wrapped with self-adhesive plastic film (kitchen wrap) and packed into a padded crate for transport to the laboratory.

Soils were classified according to the New Zealand Soil Classification (Hewitt et al., 2010). Land use classes used were pasture, forestry to pasture (where land had recently changed from production forestry to pasture), arable (annual cultivation), horticulture (perennial plants left in place), production forestry and native (indigenous vegetation). Initially, the pasture classification was separated into dairy (milking cows) and pasture used for meat or fibre production. However, it has become more difficult in recent times to separate these two classifications as farms have diversified and the indicator results for both these land uses have come together, e.g. dry dairy cows are often run on what was previously only sheep and beef farms.

All analyses were carried out at IANZ-accredited laboratories, Landcare Research and Hill Laboratories, both of Hamilton, in accordance with the Land and Soil Monitoring Manual (Hill & Sparling, 2009). Detail about soil preparation for laboratory analyses and the preferred analytical methods are given in Appendix II of Taylor et al,. (2017). Briefly, the recommended procedures for analyses are:

- **Total C and N** – Analyses using high temperature combustion methods.
- **Soil pH** – measured by glass electrode in a slurry of 1 part by weight of soil to 2.5 parts water.
- **Olsen P** – Extraction by shaking for 2 hours at 1:20 ratio of air-dry soil to 0.5 M $NaHCO_3$ at pH 8.5, filtered, and the phosphate concentration measured by the molybdenum blue reaction using Murphy-Riley reagent.
- **AMN** – estimated by the anaerobic incubation method for mineralisable N. Moist soil is incubated under waterlogged conditions (5 g equivalent dry weight with 10 ml water) for 7 days at 40°C. The increase in ammonium-N extracted in 2 M KCl over the 7 days gives a measure of potentially mineralisable N.

- **Dry bulk density** – Measured on a sub-sample core of known volume dried at 105°C. The weight of the oven-dry soil, expressed per unit volume, gives the bulk density. The bulk density is also needed to calculate porosity.
- **Macroporosity at -10 kPa** - is calculated from the total porosity and moisture retention data: $S_m = S_t - \theta$, where S_m is macroporosity, S_t is total porosity and θ is the volumetric water content at -10 kPa tension.

Notes:
Air-dry, sieved (<2 mm) sub-samples are used for all chemical analyses.
Intact soil cores (triplicate) are used for soil physical analyses.

Chemistry data are normally received from the laboratory on a gravimetric basis (weight/weight), and soil physical data on a weight/volume (bulk density) or volume/volume basis (macroporosity). Chemical data for Olsen P was converted from a gravimetric basis (weight/weight) to a volumetric basis (weight/volume) by multiplying by the bulk density. Note that some New Zealand industry-based Olsen P target values are based on a volumetric basis, but on a 'modified' basis. Further explanation is available in Drewry et al. (2013; 2017).

A physical sample of the soil (air dried, <2 mm) was also stored for reference and for re-analysis if required. Physical samples were stored in screw-top plastic jars, at 18–25°C, with unambiguous identification.

All 150 sites were used to give the overall soil quality picture for the region. However, where the land use had recently changed from production forestry to pasture, indicator measurements were intermediate between those typical of forestry and those typical of pasture (Hermans et al., 2020). Results would have been significantly skewed if these sites were included in one of the pasture categories. Consequently, these eight sites were treated as their own category.

Each indicator measurement has a range within which most soil samples fall. From this process it has been possible to assign a range for each measurement that identifies levels from low, adequate/optimal, and high to excessive (e.g. bulk density is expressed as loose, adequate, or compact, as this is a measure of the weight of soil in a cubic metre). Targets levels for each indicator measurement are set to reflect where negative impacts on the environment could occur and these are based on national guidelines (Sparling et al., 2003a) and updates (Hill & Sparling 2009; Mackay et al., 2013). These targets are presented in Appendix III of Taylor et al., (2017).

Assessment of sites meeting targets could only be done once an adequate number of sites representative of the region had been sampled. It took until 2005 to build up an adequate number of sites so results for sites meeting targets are presented for 2005-2018.

The resulting data are stored in the WRC database system in Microsoft Excel as recommended by the LMF (Hill & Sparling 2009). Measurements were categorised by land use and reported as meeting or not meeting targets. As it took several years to build up the number of sampling sites, these data are presented in the results section for 2005-2018.

2.2 Supporting analysis

Other analyses were carried out at Landcare Research, AgResearch, Plant and Food Research and the University of the Waikato stable isotope unit on the same samples. These analyses provided information that helped interpret the trends in soil quality data. Some analyses were obtained as part of the same method that produced the soil quality indicator described above, e.g. extractable nitrate and ammonium are derived as part of AMN; the C:N ratio was derived from total C and total N. Aggregate stability was found to be a useful supporting indicator for soil structure for arable soils. However, it is only regularly carried out for arable soils or where a soil disturbance (erosion, landscaping etc) has occurred. These methods are included in Hill &

Sparling (2009) and Appendix II of Taylor et al., (2017). Mixed modelling was not used for aggregate stability due to regular sampling being limited to cultivated or disturbed sites. Instead, a 5-year rolling average was plotted to allow for annual fluctuations.

Other analyses were carried out as investigations into other potential soil quality indicators, e.g. hot water carbon and nitrogen (HWC and HWN), and ^{15}N. HWC and HWN provide a measure of labile soil organic matter; the organic matter fraction most readily available for microorganisms, and they can be used to assess the stability and dynamics of soil carbon. HWC and HWN are more sensitive to environmental pressures than total C, so suggested for monitoring longer-term trends in organic matter (Ghani et al., 2003). HWN provides a measure of potentially available N.

The method described by Ghani et al. (2003) was used for HWC and HWN. Briefly, air-dried soil samples were extracted with distilled water (1:10 soil : water ratio) in an 80°C water bath for 16 hours. Mixed modelling was not used for HWC and HWN as data have been collected for a shorter time, with the full number of sites not sampled until 2010. Instead a 5-year rolling average was plotted to allow for annual fluctuations.

^{15}N in soil was developed to identify N sources, transformations and cycling. A representative subsample of the air-dried soil sample was obtained by passing the whole sample through a stainless-steel riffle (which split the sample) until the desired subsample size was reached (about 5 g). Any obvious plant material was then removed with tweezers before fine grinding using an agate mortar and pestle. Samples were analysed for N isotopes using a Europa Scientific 20-20 Stable Isotope Analyzer at the University of Waikato Stable Isotope Unit, where overall reproducibility of laboratory measurements is ~0.2 ‰. The natural abundance of ^{15}N is expressed in delta notation (δ) as parts per thousand (‰) using the following equation (after Coplen, 2011):

$$\delta^{15}N = R(^{15}N/^{14}N)_{sample} - R(^{15}N/^{14}N)_{air} - 1$$

where R $(^{15}N/^{14}N)_{sample}$ is the ratio of $^{15}N/^{14}N$ in the sample and $R(^{15}N/^{14}N)_{air}$ is the ratio of $^{15}N/^{14}N$ in atmospheric N_2 (the international standard).

2.3 Statistical analysis

An analysis was carried out as part of the previous soil quality technical report (Taylor et al., 2017). An additional three more years data has now been added to the dataset and the same method of statistical analysis used. Each indicator was assessed for statistical trends using linear mixed modelling with random splines overall (shortened to "mixed modelling" in the results and discussion sections), by soil order and by land use. Data for four of the indicators (total C, total N, Olsen P and AMN) required log transforming to get approximate constancy and normality of the residual variation (Appendix I Statistical analysis). The back-transformed estimates were "bias-corrected" to make this the same as the overall arithmetic mean of all the original values in the data. Data calculated by this method are presented in the results section for 1995-2018.

Random terms were used to give an appropriate structure to the error terms, allowing for the correlations arising from repeated observation of the same sites. Random smoothing spline terms were used in the model for three reasons: firstly, they allow us to fit trend terms that are data driven, not requiring the choice of a particular functional form, secondly, they allow us to estimate the trends as if every site had been measured at every date, and finally they allow modelling the serial correlation between successive observations on the same sites (Appendix I).

For the anerobically mineralised N indicator only, the addition of the last three years data has given rise to a significant non-linear (spline) time component being estimated now, whereas before it was nearly zero. Therefore, the modelled lines have become more jagged compared with the previous report.

The accuracy of estimated values in any one year was improved by utilising the information from sites before and after each time period, thus increasing the effective sample size (Appendix I). Data calculated by this method are presented in the results section for 1995-2018.

Data was analysed for supporting analysis using the breakdown and one-way ANOVA function of Statistica 13, while plots were produced using the boxplot function.

3 Soil quality results

Results are first presented for the overall dataset, then by soil order and land use, to identify the contribution these make to driving change. Soil quality results for 35 sites monitored in 2018-19 are presented in Appendix II and results for all 154 sites are presented in Appendix III.

Overall, 11% of soil quality sampling sites from land in farming or production forestry in the region, weighted for land area, met the target values for all seven indicators in 2018, similar to the percentage of sites meeting the target values for all seven indicators for the preceding eight years (Figure 2). However, this is down from a high of 18% in 2006. To meet the target values for all seven indicators the soil must fall within the adequate, optimal, or normal range as shown in Table 3 for each of the indicators. There has been a corresponding increase in the number of sites failing to meet the target values for one or two indicators over the same time period. On an area weighted basis, the main soil quality issues for productive land in the Waikato region occur on pastoral land (Figure 3). Note that native sites represent 30% of the land area and all sites are in the satisfactory range for all seven indicators, while pasture sites represent 58% of the land area and 3% of sites are in the satisfactory range for all seven indicators. Further discussion of the soil quality issues identified are presented by land use in the discussion section.

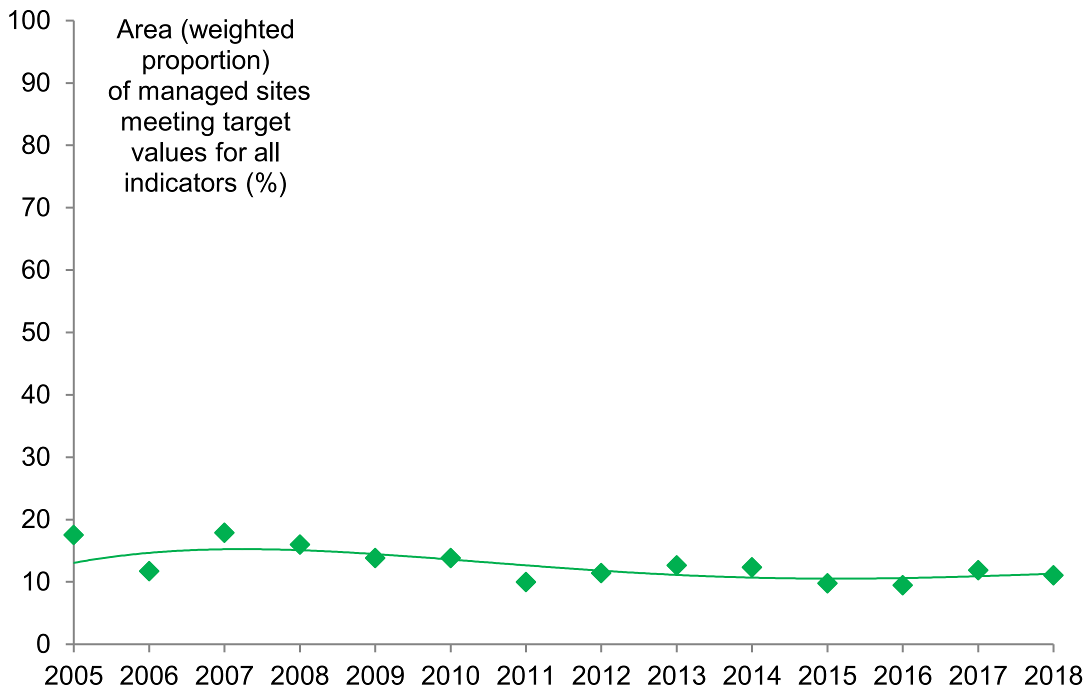

Figure 2: Proportion of sites for all land uses meeting the target values for all seven soil quality monitoring indicators from 2005 to 2018 (% number of sites weighted for land area).

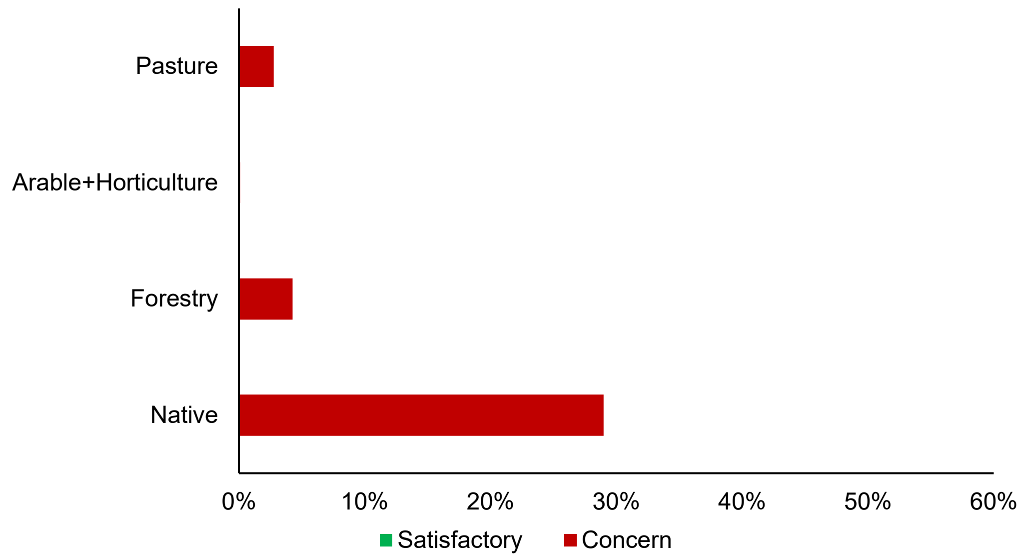

Figure 3: **The proportion of sites weighted for land area in the satisfaction range for all seven indicators as of 2018.**

Results for 2018

Soils from soil quality sites active in 2018 under indigenous vegetation were on average acidic (pH 5.1), high in total C (mineral soils 16.7%, Organic Soils 49%), low in Olsen P (8 mg/L), low in bulk density (0.52 t/m3) and had high macroporosity (28%). In comparison, soils under pine forestry had generally similar characteristics to soils under indigenous vegetation but had lower total C (8.4%). All the pine forest sites were a monoculture of Pinus radiata. In agriculture, crop and pasture species diversity has been shown to increase total C compared to monocultures and the increased C storage is a direct function of the microbial community (Liang et al., 2017; McDaniel et al., 2014). Whether this behaviour also applies to forests appears to still need investigating as only a few studies have addressed forest SOC stocks, but it has proved challenging to disentangle effects of species diversity in natural forest conditions (Vesterdal et al., 2013). On the other hand, total C stock differs among tree species and the lower total C content in pine forest sites compared to indigenous vegetation sites may simply be a species identity effect (Vesterdal et al., 2013).

Soils under pasture were on average less acidic (pH 5.9) than forest soils, due to additions of lime, with higher Olsen P (45 mg/L) most likely due to additions of phosphate fertiliser, and lower macroporosity (9%) due to the treading of hooved animals.

Soils under horticulture were on average the least acidic (pH 6.6) of the land uses measured, with excessive levels of Olsen P, but were otherwise similar to pasture.

Soils under arable use were most different from soils under indigenous vegetation. These had low total C (4.5%), indicating loss of SOM, low AMN indicating little reserve of mineralisable nitrogen due to loss of SOM, very high Olsen P (88 mg/L) indicating excessive fertility, and high bulk density (0.94 t/m3) indicating compaction. Arable soils were also slightly acidic (pH 6.2).

3.1 Changes in bulk density 1995-2018.

The mixed modelling results showed no significant change in bulk density for the overall data (Figure 4), but there were significant ($p<0.05$) differences in linear trend when the results were assessed by soil order ($p=0.009$, Figure 5). Bulk density had decreased in Brown and Granular soils by an average of 0.004 and 0.005 t/m^3, respectively, over the last five years. There was a significant difference, but only at $p<0.1$, when results were assessed by land use. Bulk density for arable land increased by an average of 0.005 t/m^3 (p=0.099, Figure 6). Changes in other land uses were not significant.

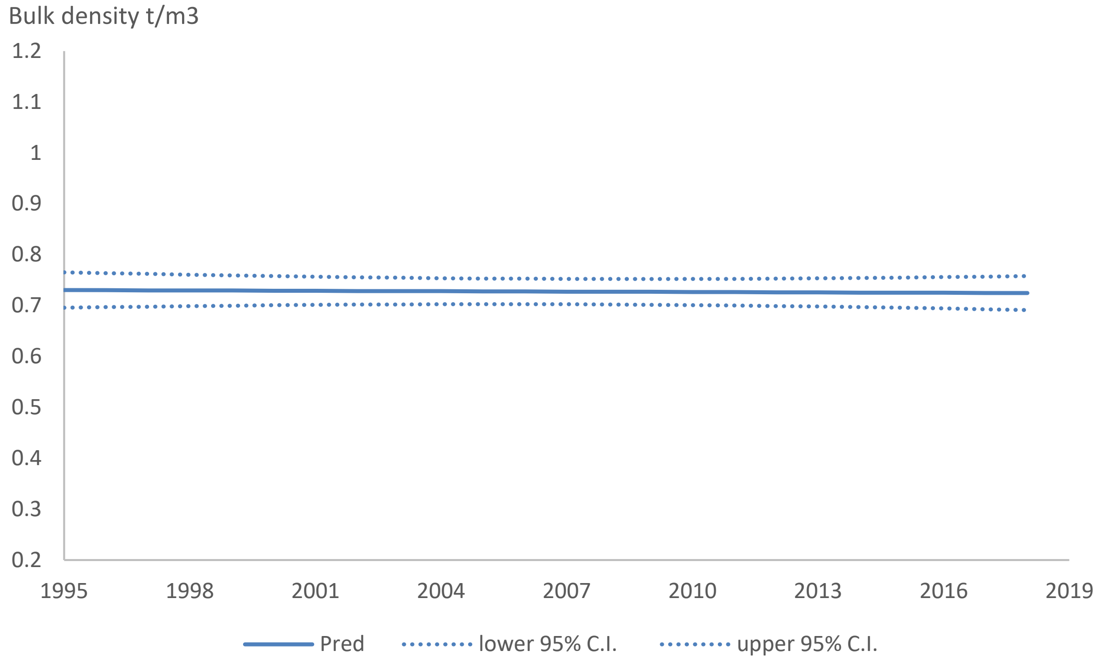

Figure 4: **Change in mixed modelling average bulk density 1995-2018 for all sites (all land uses and soil orders) with 95% confidence limits.**

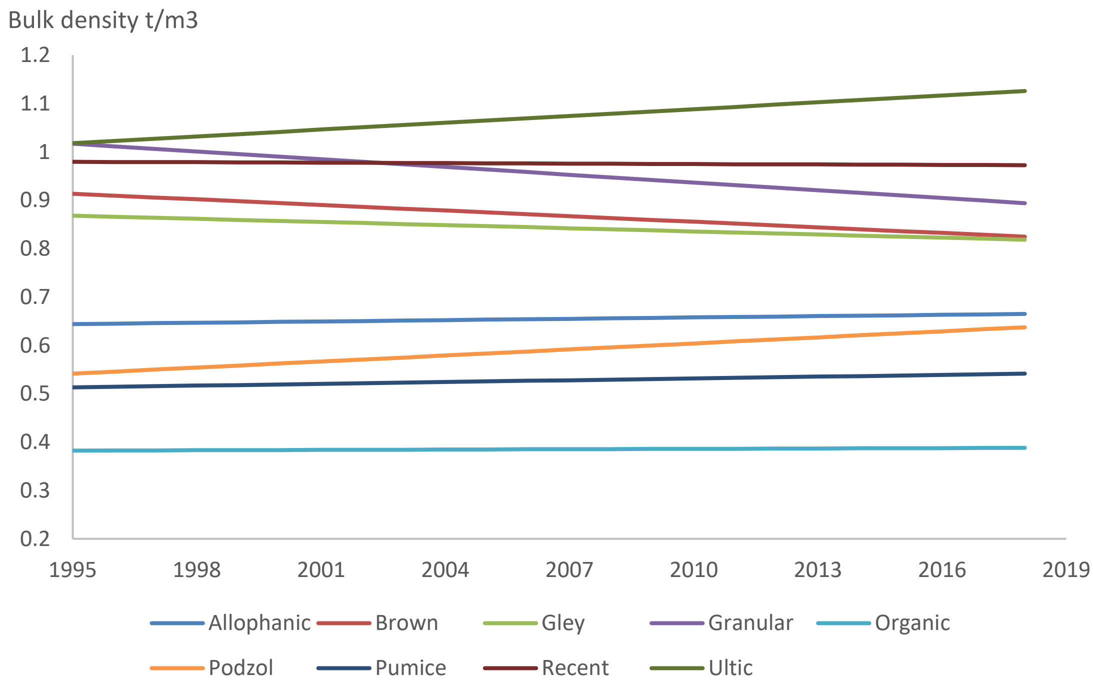

Figure 5: **Change in mixed modelling average bulk density 1995-2018 by soil order.**

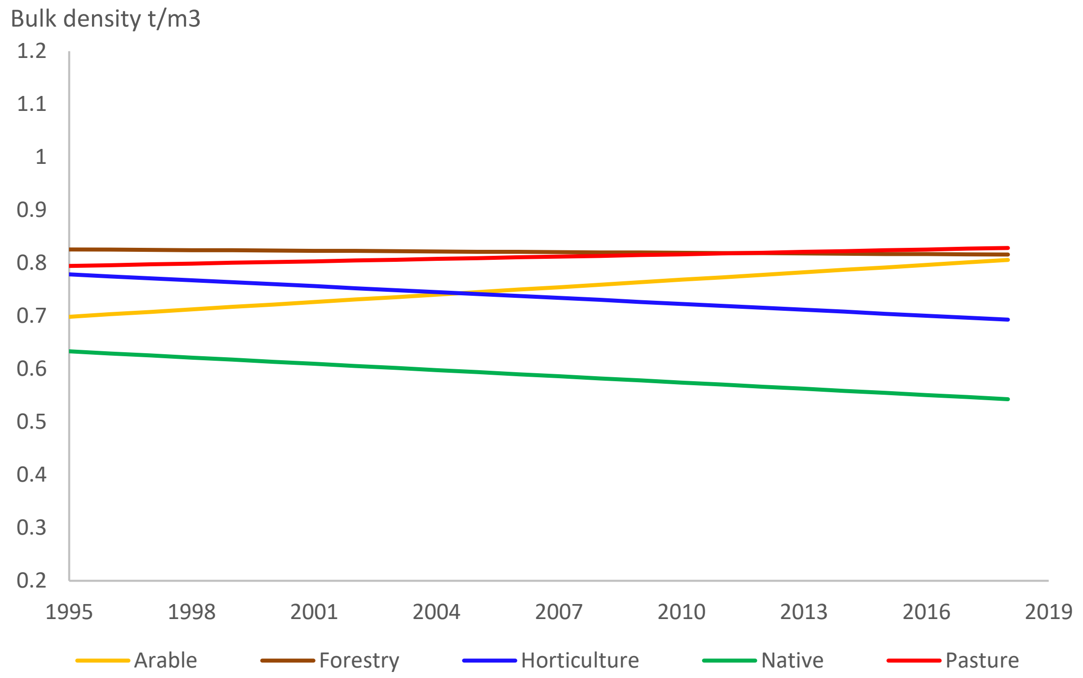

Figure 6: **Change in mixed modelling average bulk density 1995-2018 by land use.**

Measurements below lower targets for bulk density were considered to indicate loose soil, while results above upper targets were considered to indicate compaction. All sites (100%) meet the upper bulk density targets (1-1.4 t/m3 depending on soil type, Appendix III of Taylor et al., (2017); data not shown) but a considerable percentage of forestry sites did not meet the lower bulk density targets (0.2-0.7 t/m^3 depending on soil type, Appendix III of Taylor et al., (2017); Figure 7) due to forestry being a land use that can reduce erosion on low bulk density soils. However, the percentage of forestry sites meeting the low bulk density target has increased over time.

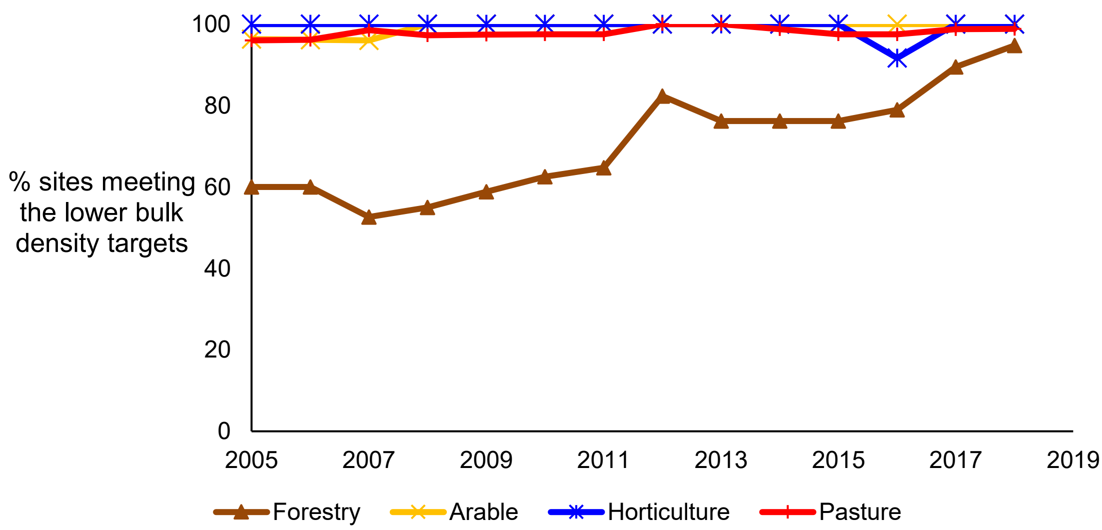

Figure 7: **Change in percent of sites meeting the lower bulk density targets 2005-2018 by land use. All native sites meet targets (100%) in all years so are not shown.**

3.2 Changes in macroporosity @ -10 kPa 1995-2018.

The mixed modelling macroporosity results overall showed no evidence of a trend with time (Figure 8) and differences in pattern or linear trend between soil orders and land uses was not significant (Figures 9-10). The apparent "wave" in the curve is an artifact within measurement error (Appendix 1). Although not significant, there appears an overall decrease in macroporosity for arable and horticultural land (over the 23 years). However, modelling the last 5 years data shows macroporosity has been stable or has increased.

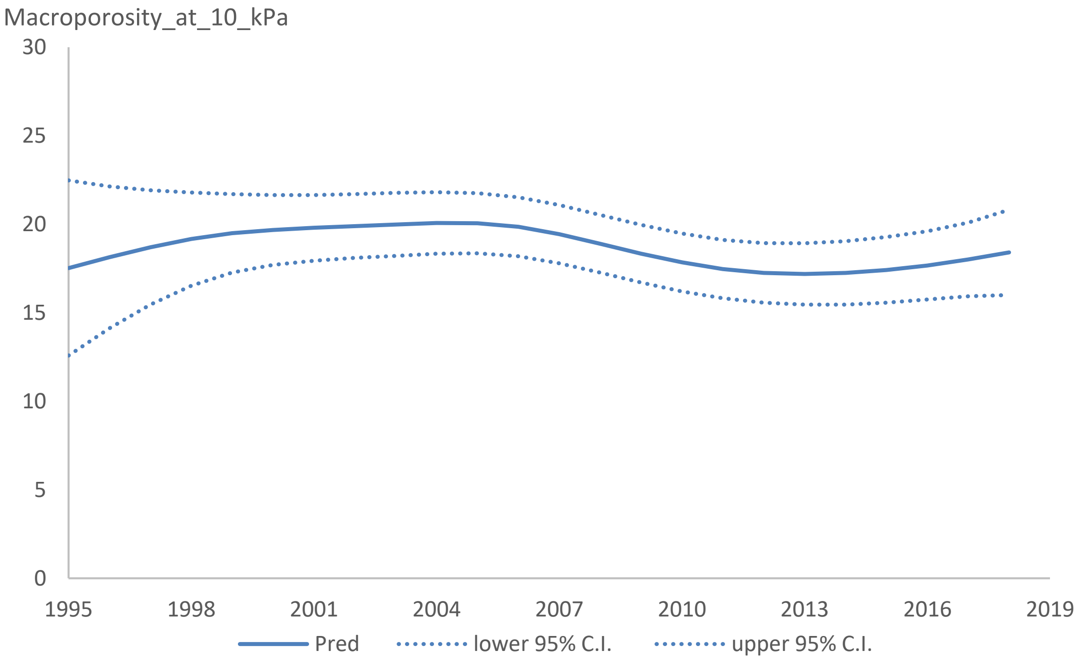

Figure 8: **Change in mixed modelling average macroporosity @ -10 kPa (pores > 30 µm) 1995-2018 for all sites (all land uses and soil orders) with 95% confidence limits.**

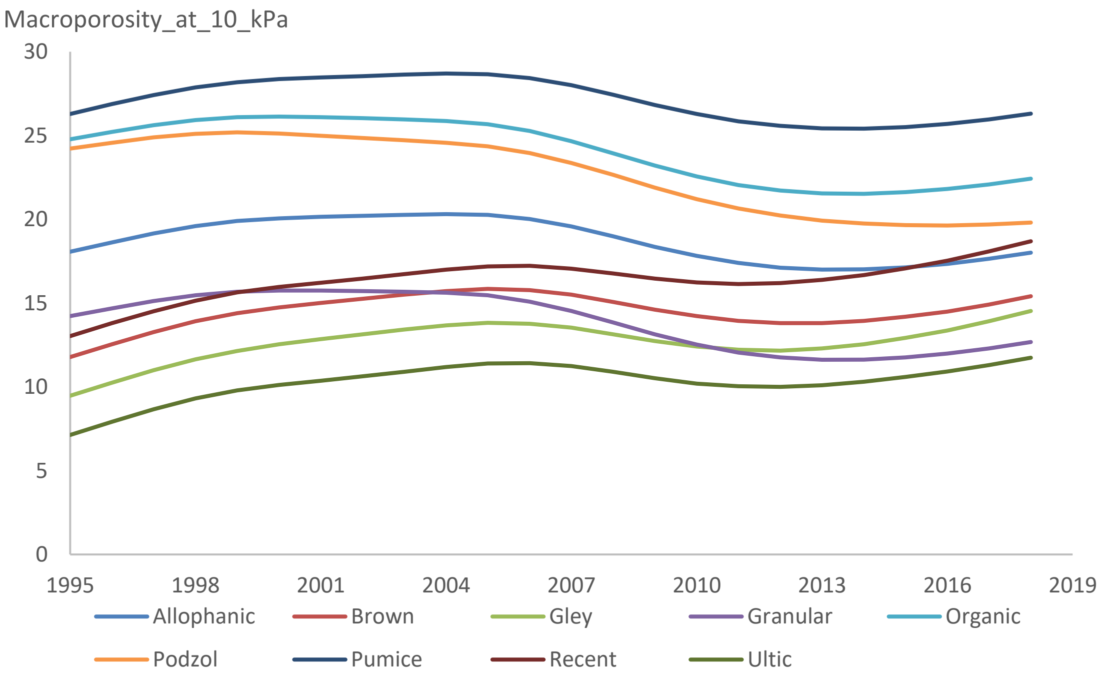

Figure 9: **Change in mixed modelling average macroporosity @ -10 kPa (pores > 30 µm) 1995-2018 by soil order.**

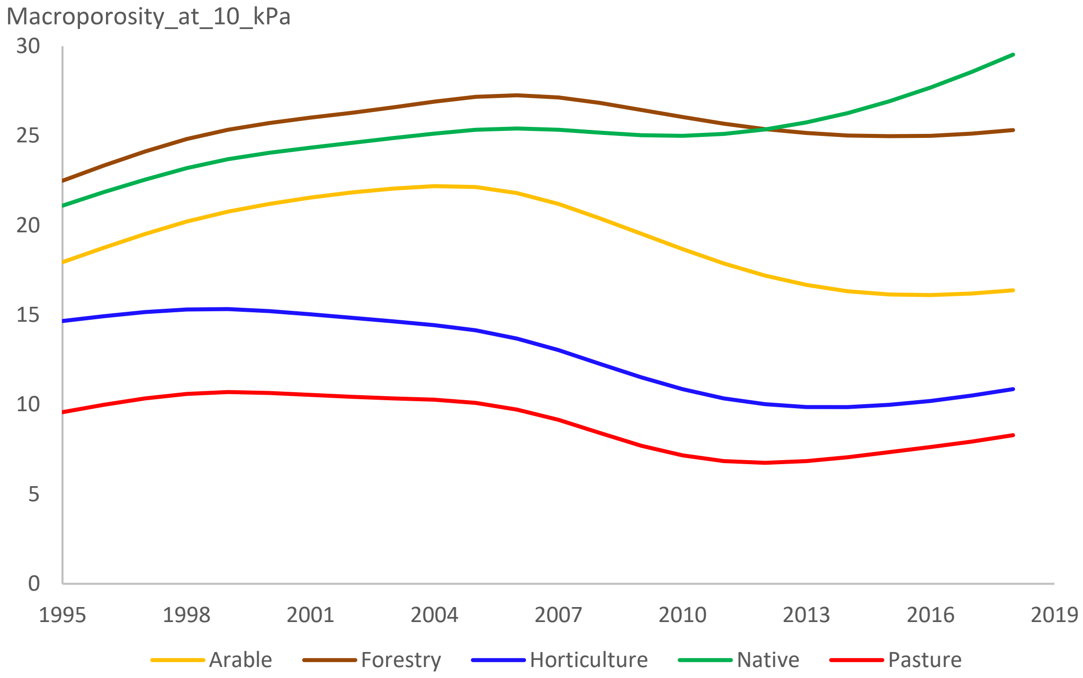

Figure 10: **Change in mixed modelling average macroporosity @ -10 kPa (pores > 30 µm) 1995-2018 by land use.**

Measurements below lower targets (8% for forestry, 10% other land uses, Appendix III of Taylor et al., (2017)) are considered to indicate compaction, while results above upper targets (30%, Appendix III of Taylor et al., (2017)) are considered to indicate loose soil.

The percentage of sites meeting the lower targets decreased over time for all productive land uses (Figure 11), and the percentage of sites meeting the upper target increased for arable and forestry (Figure 12). Soils under horticulture showed the largest decrease in average macroporosity and in meeting the lower macroporosity target of 10% macropores. Soils under pasture already had low macroporosity and only 35% of pasture sites meet the low macroporosity target in 2018, although this was an improvement on the previous three years.

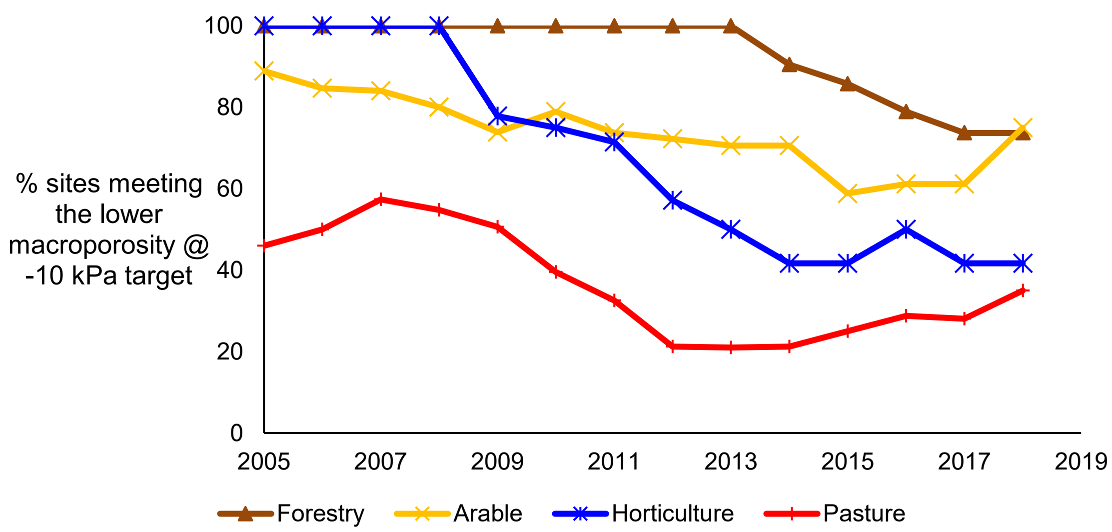

Figure 11: **Change in percent sites meeting the lower macroporosity @ -10 kPa target, 2005-2018 by land use. All native sites meet targets (100%) in all years so are not shown.**

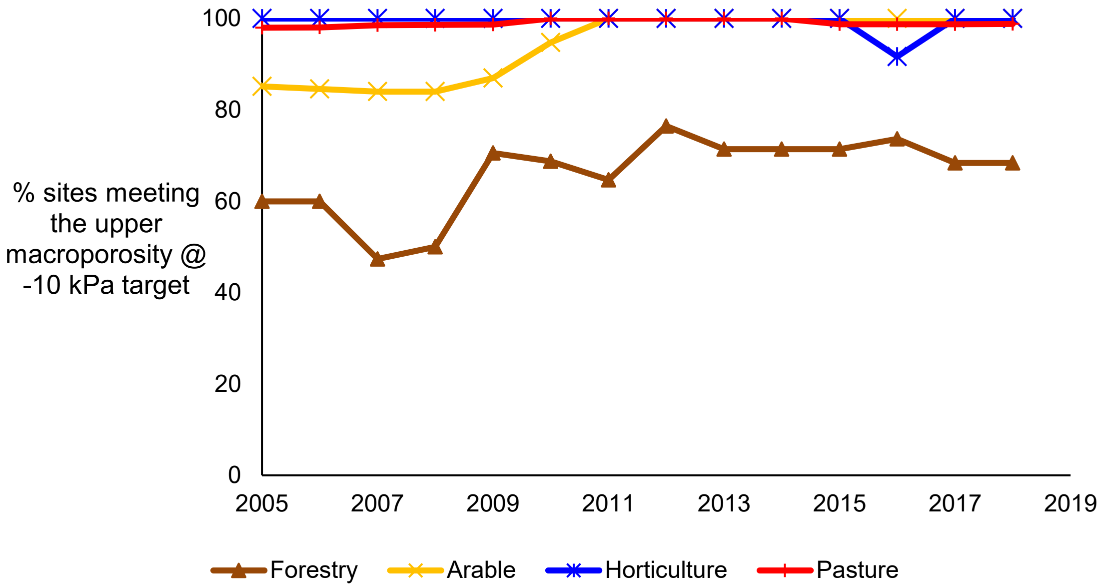

Figure 12: **Change in percent sites meeting the upper macroporosity @ -10 kPa target, 2005-2018 by land use.**

3.3 Changes in Olsen P 1995-2018.

The mixed modelling Olsen P results show a non-linear, increasing trend over time for the overall dataset (Figure 13) that remained a consistent pattern with soil order or land use (Figures 14 and 15). Olsen P is reported on a volumetric basis (weight/volume) basis.

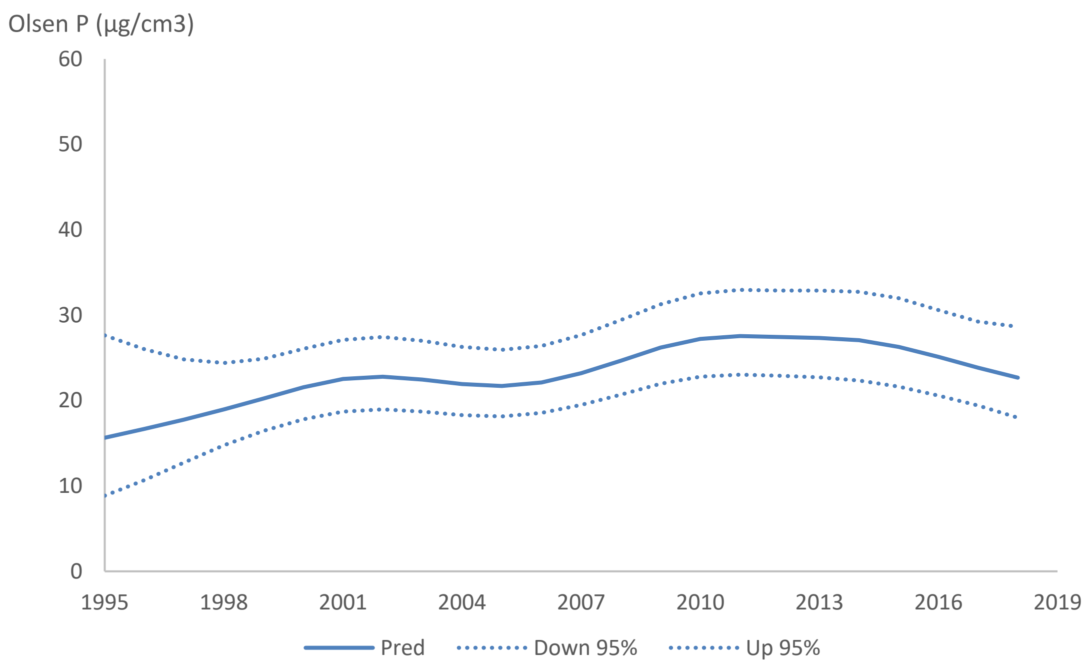

Figure 13: **Change in mixed modelling average Olsen P 1995-2018 for all sites (all land uses and soil orders) with 95% confidence limits.**

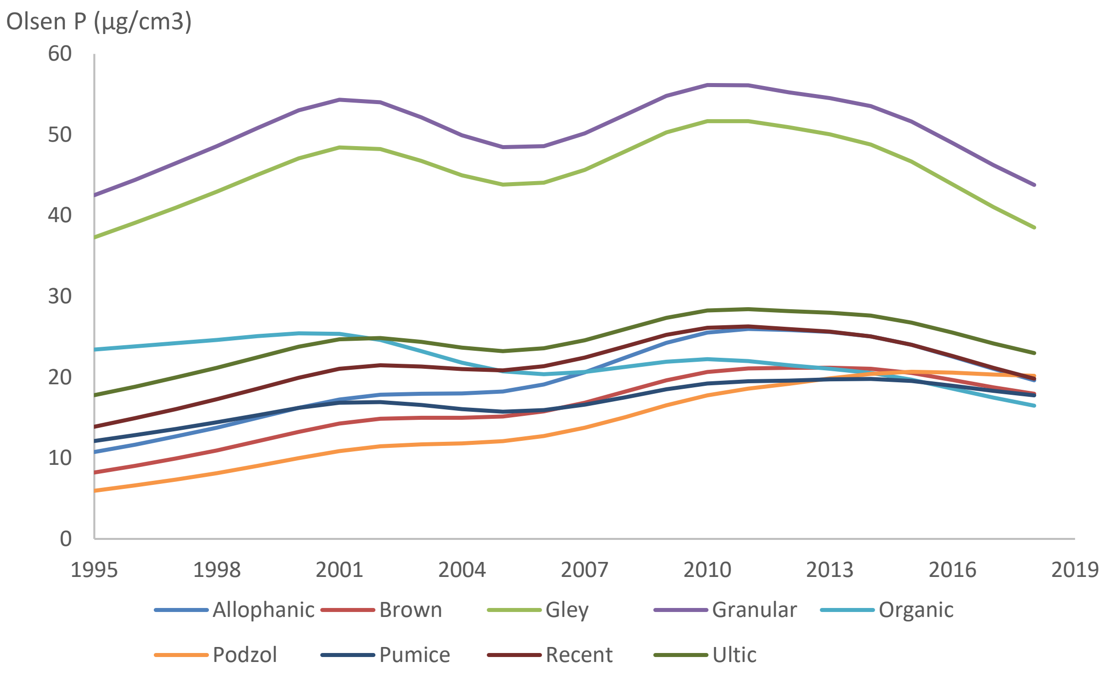

Figure 14: Change in mixed modelling average Olsen P 1995-2018 by soil order.

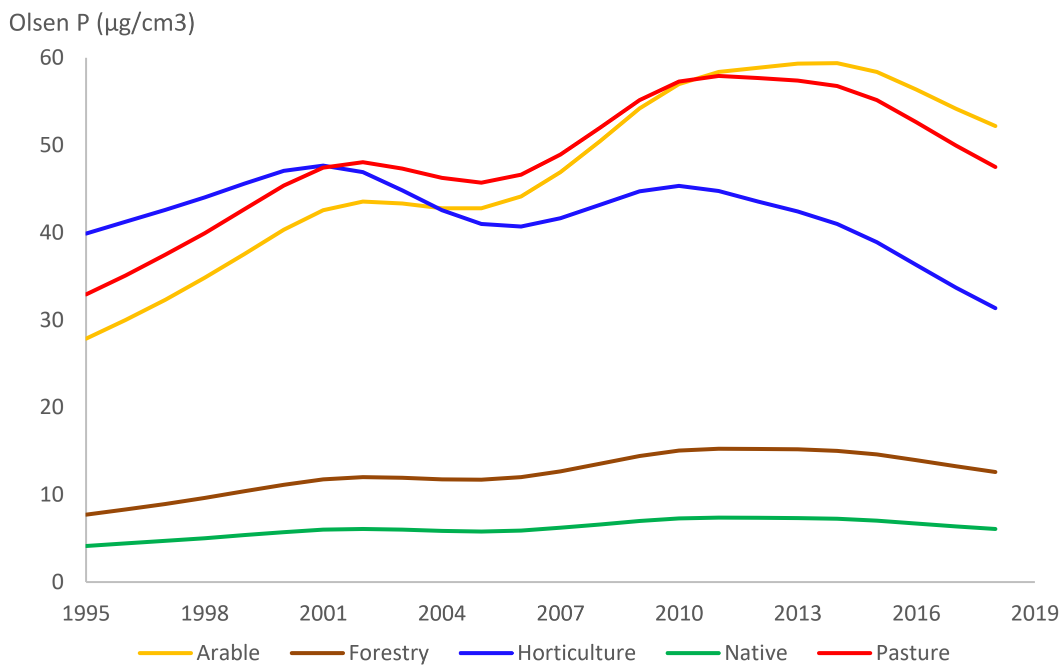

Figure 15: Change in mixed modelling average Olsen P 1995-2018 by land use.

Values below lower targets (5-25 mg/L depending on land use and soils type; Appendix III of Taylor et al., (2017)) are considered to indicate deficiency in P, while measurements above the upper target (50 mg/L, Appendix III of Taylor et al., (2017)) are considered to indicate increased risk of losing P from land. Note that Olsen P (volume/weight) measured for soil quality here is derived from Olsen P measured gravimetrically multiplied by bulk density. So, values should be considered a good indication rather than a direct measurement of Olsen P (volume/weight).

The percent of sites meeting the high target for Olsen P (Olsen P < 50 mg/L) in soils under arable and horticulture has stabilised after declines in meeting this target (Figure 16). Pasture and forestry remain stable. Interestingly, although average Olsen P increased slightly for forestry, even taking into account the decline in recent years, Figure 15), there was an increase in the percent of sites not meeting the lower Olsen P target, i.e. deficient P status (Figure 17). Similarly, more horticultural sites failed to meet the lower Olsen P target. Interestingly, these sites were all apple orchards.

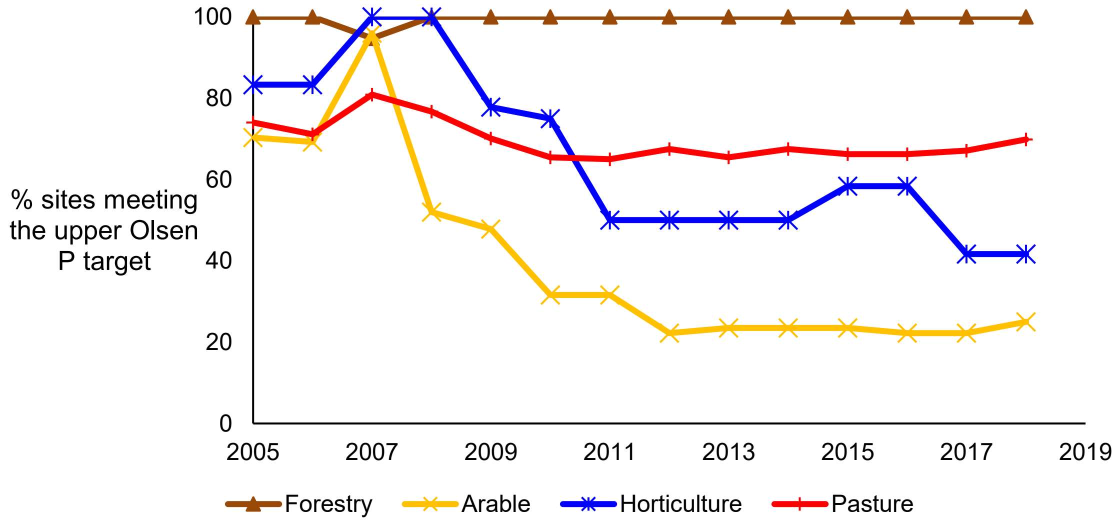

Figure 16: **Change percent sites meeting the upper Olsen P target 2005-2018 by land use. All native sites meet targets (100%) in all years so are not shown.**

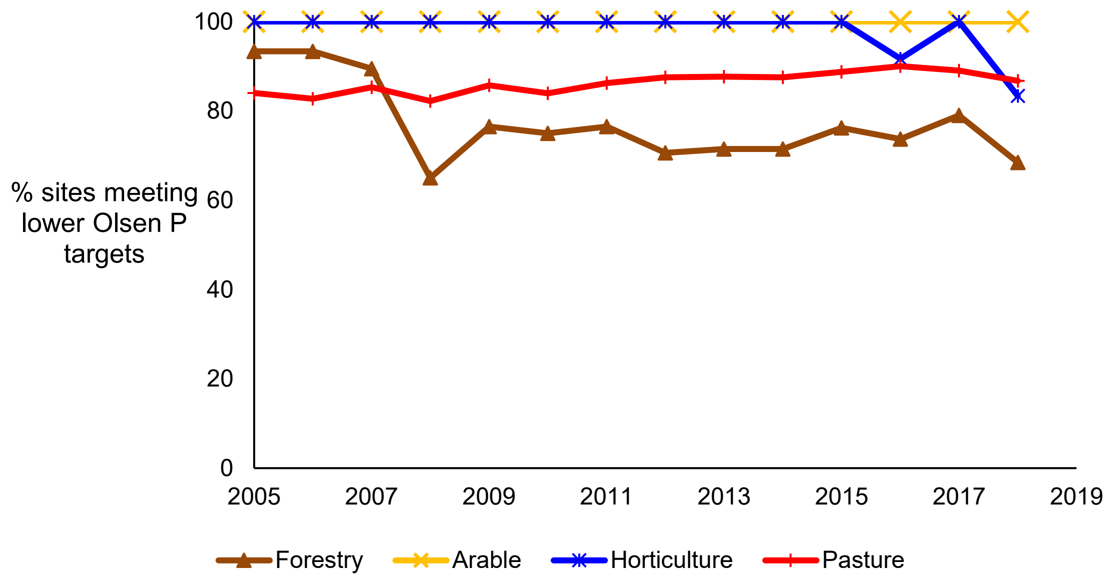

Figure 17: **Change percent sites meeting the lower Olsen P target 2005-2018 by land use.**

3.4 Changes in total N 1995-2018.

For the overall dataset, mixed modelling total N results showed a nonsignificant ($p<0.05$), non-linear trend over time (Figure 18), although an apparent and nominal upward trend can be seen. There were no significant differences ($p<0.05$) in pattern or linear trend between soil orders (Figure 19). However, the linear trend did vary significantly ($p<0.05$) between land uses (Figure 20). Notably, arable declined, while all the other land uses showed an increase over time, with significant ($p<0.05$) increases for native and pasture land uses. The reasons for this behaviour are assessed in the discussion section.

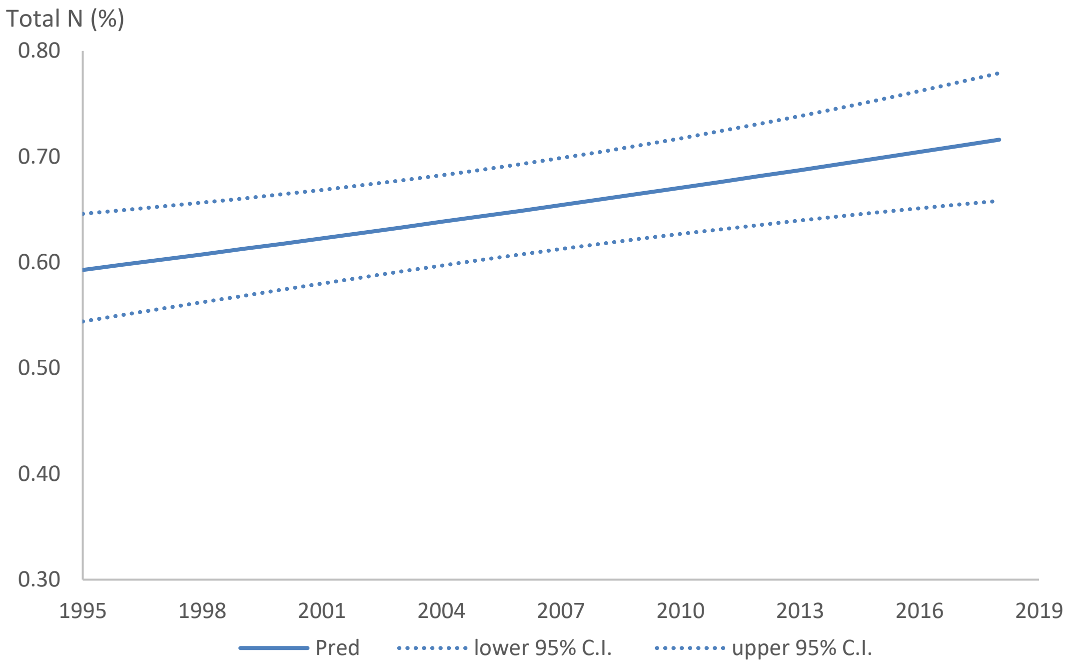

Figure 18: **Change in mixed modelling average total N 1995-2018 for all sites (all land uses and soil orders) with 95% confidence limits.**

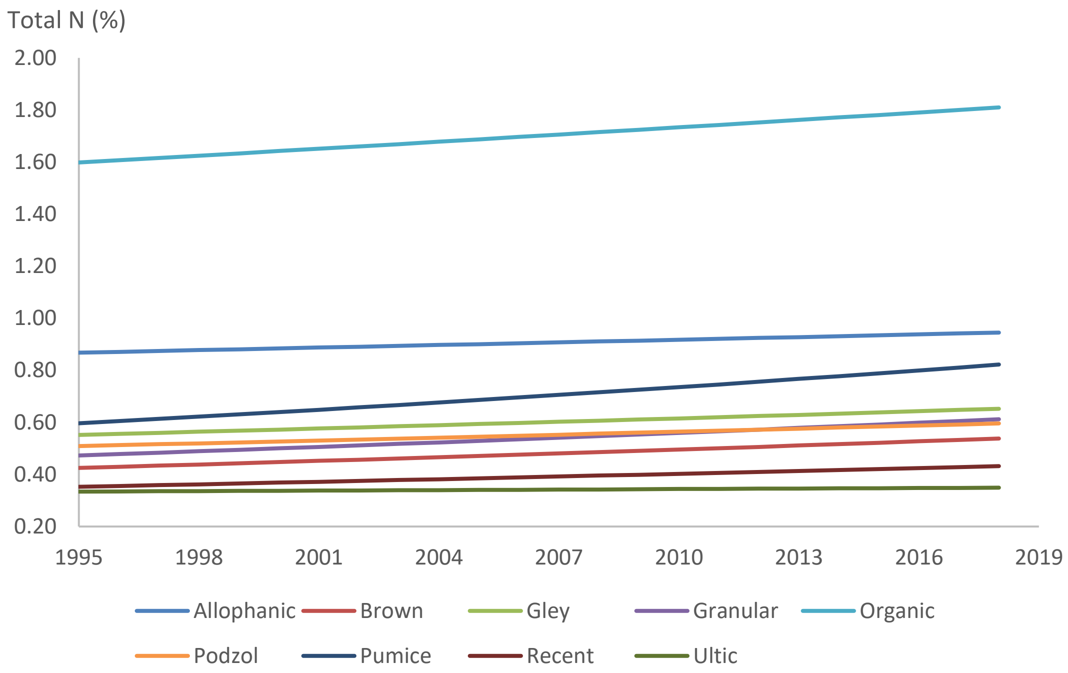

Figure 19: **Change in mixed modelling total N 1995-2018 by soil order.**

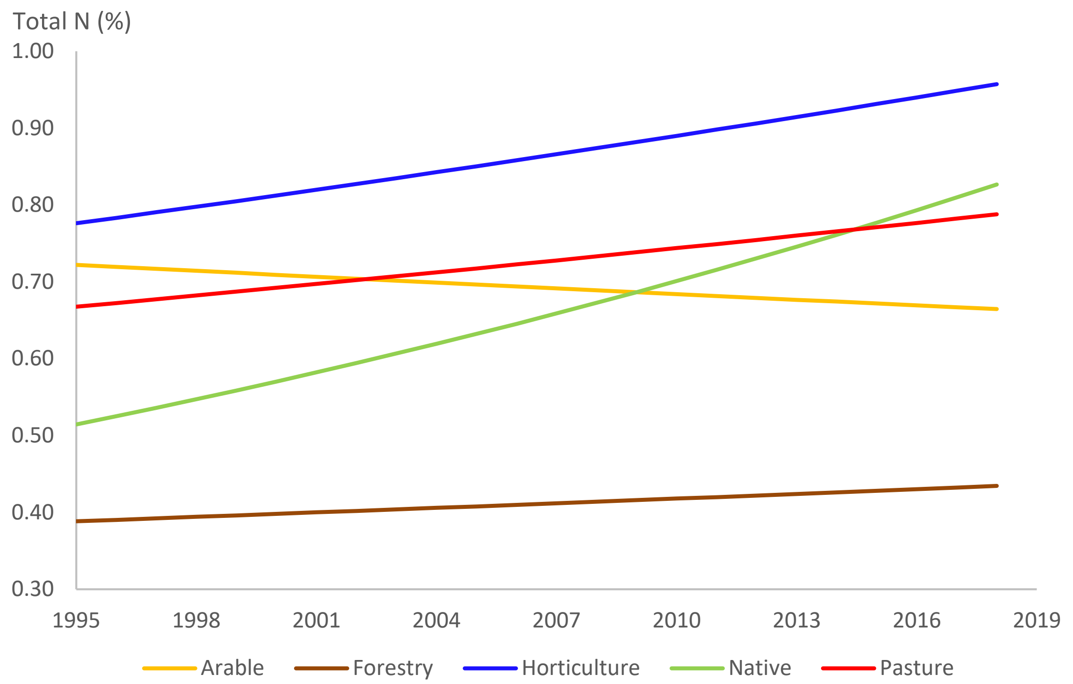

Figure 20: Change in mixed modelling total N 1995-2018 by land use.

Measurements below lower targets (0.1% for forestry and 0.25% for other land uses, Appendix III of Taylor et al., (2017)) are considered to indicate depleted nitrogen soil status, while measurements above the upper target (0.7%, Appendix III of Taylor et al., (2017)) are considered to indicate increased risk of losing N from land.

Generally, for the total N indicator, there is an analogous relationship between mixed modelling results from 2005-2018 with changes in the percentage sites meeting the total N upper target (Figure 21). Increases in total N in soils under pasture and horticulture were reflected in declines in the percentage of sites meeting the high target; declines in total N in soil under arable were reflected in an increase the percentage of sites meeting the high target. However, from 2011, a small percentage of arable sites no longer met the lower target for total N (Figure 22). Before that, all sites met the lower total N target.

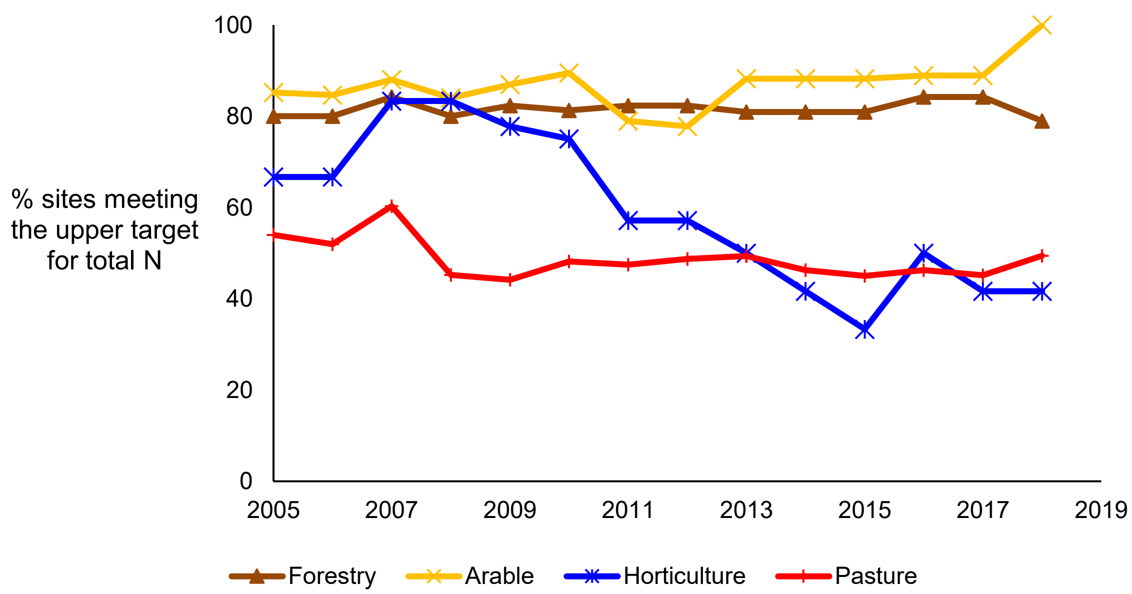

Figure 21: Change in percent sites meeting the upper target for total N 2005-2018 by land use. All native sites meet targets (100%) in all years so are not shown.

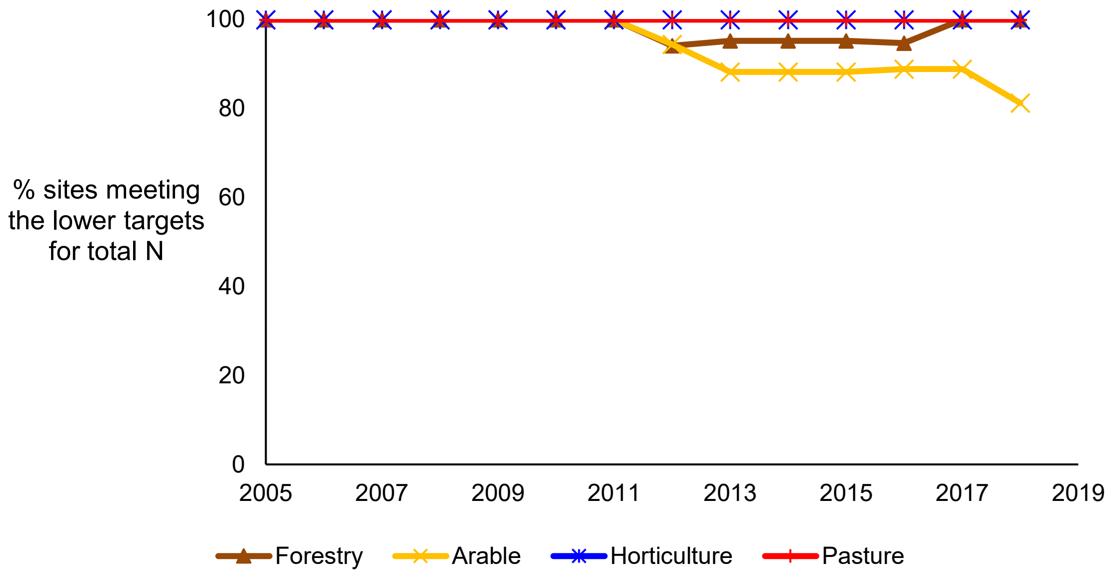

Figure 22: Change in percent sites meeting the lower target for total N 2005-2018 by land use.

3.5 Changes in AMN 1995-2018.

For the overall dataset, mixed modelling results showed there was a significant ($p<0.05$) non-linear trend over time (Figure 23). There is a significant (significant ($p<0.05$) decline in AMN across all soils and land uses over the last 5 years (Figures 24 and 25).

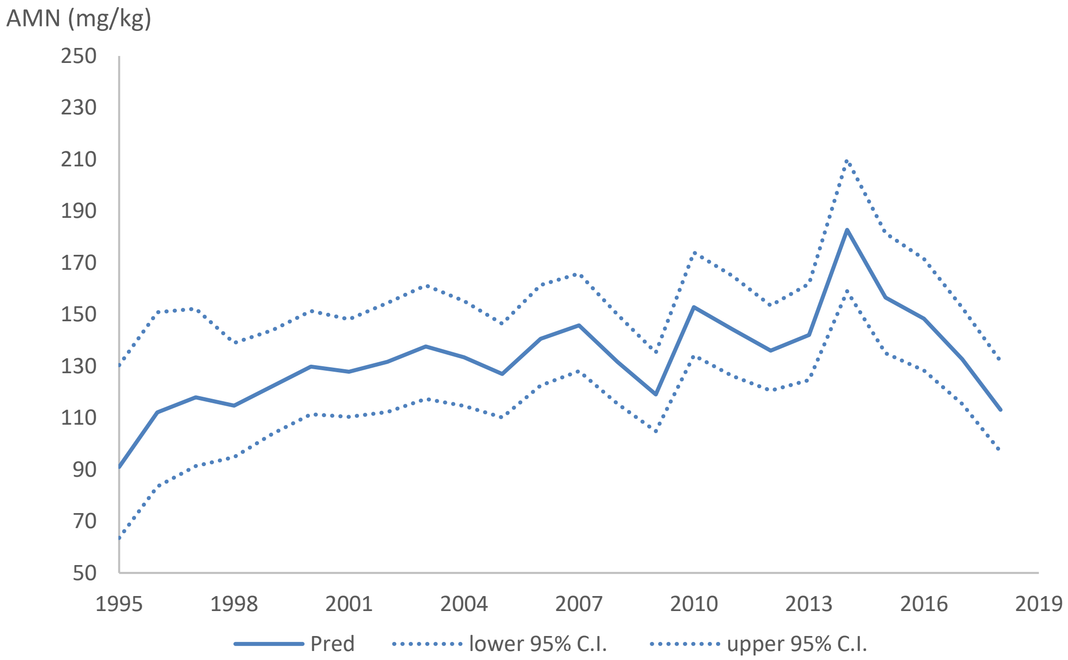

Figure 23: Change in mixed modelling average AMN 1995-2018 for all sites (all land uses and soil orders) with 95% confidence limits.

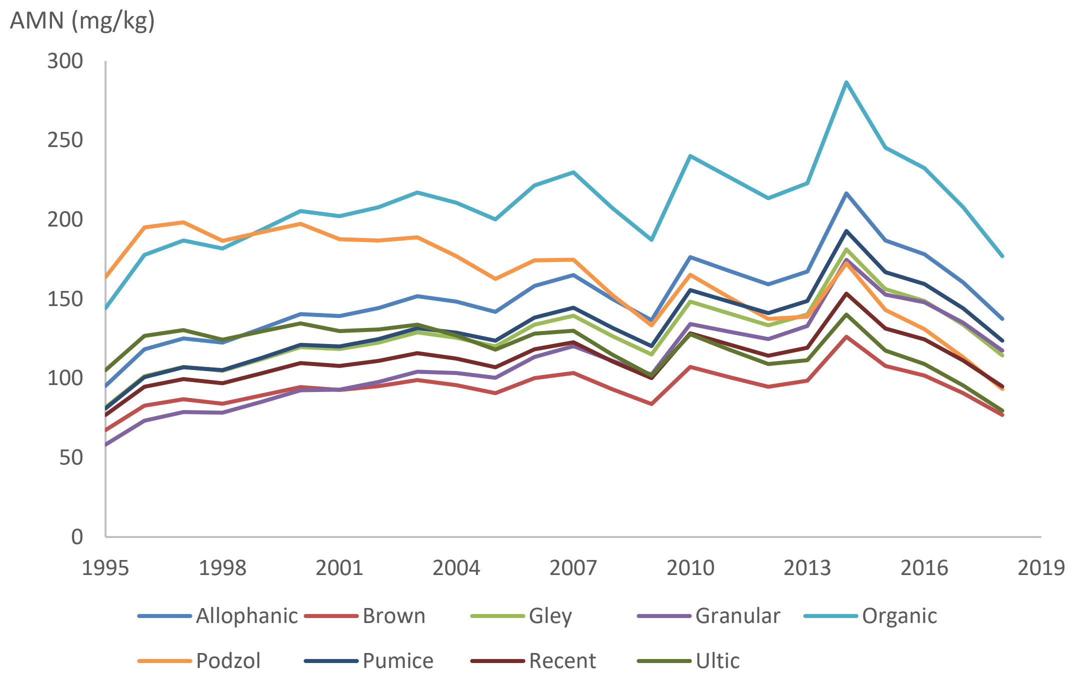

Figure 24: **Change in mixed modelling average AMN 1995-2018 by soil order.**

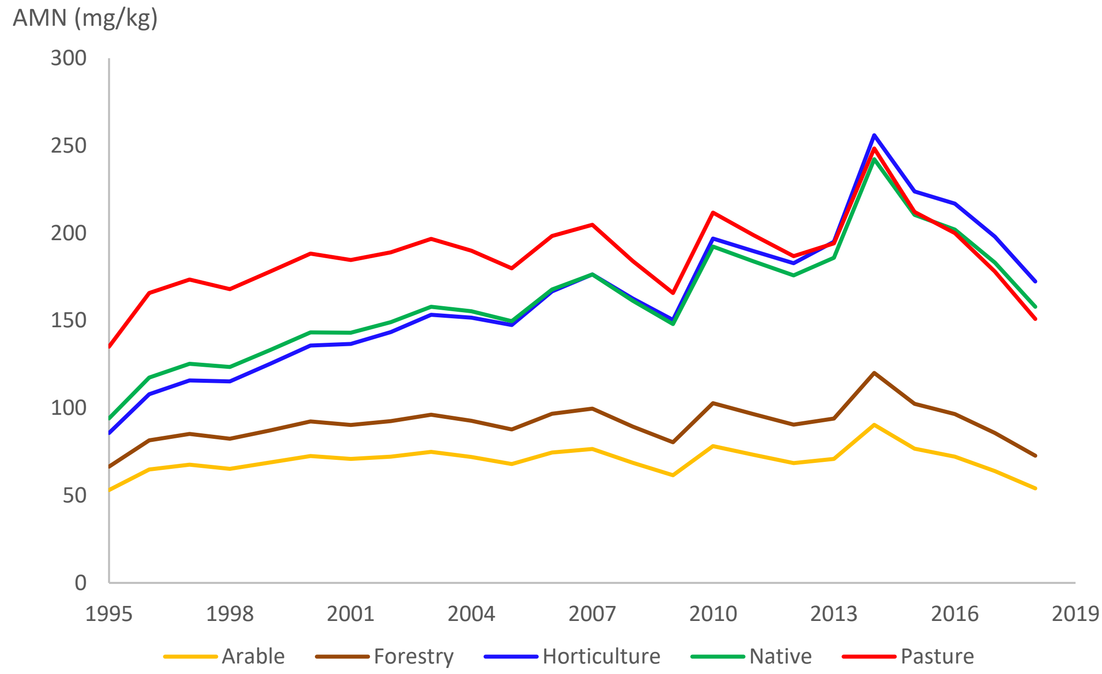

Figure 25: **Change in mixed modelling average AMN 1995-2018 by land use.**

Measurements below lower targets (50 mg/kg for pasture, 20 mg/kg for other land uses, Appendix III of Taylor et al., (2017)) are considered indicative of low levels of nitrogen that can be potentially mineralised from soil organic matter, which also relates to microbial activity. There is currently no upper target for AMN.

Consistent with the mixed modelling results, arable showed a decline in meeting the AMN (low) targets, one pasture site did not meet targets in 2018, while 100% of all other land uses met targets (Figure 26).

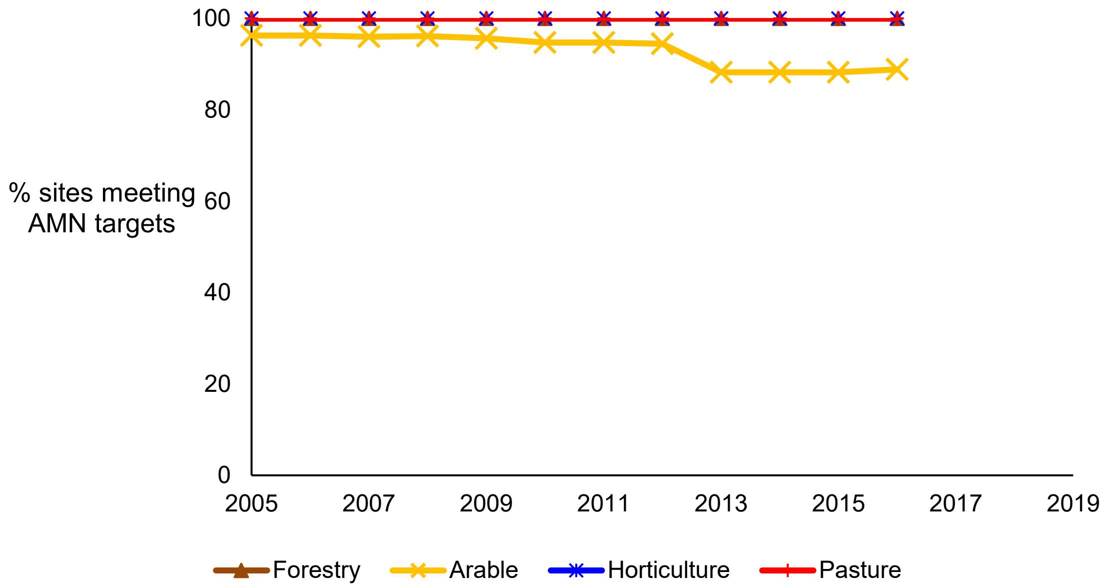

Figure 26: **Change in percent sites meeting the target for AMN 2005-2018 by land use. All native sites meet targets (100%) in all years so are not shown.**

3.6 Changes in Total C 1995-2018.

For the overall dataset, mixed modelling results showed no significant ($p<0.05$) non-linearity in the pattern over time (Figure 27). The linear trend over time varies significantly between land uses ($P=0.008$, Figure 28) with a notable significant decline of 1.0% per year for arable while all other land uses increased (Figure 29). There were no significant differences in pattern or linear trend between Soil Orders

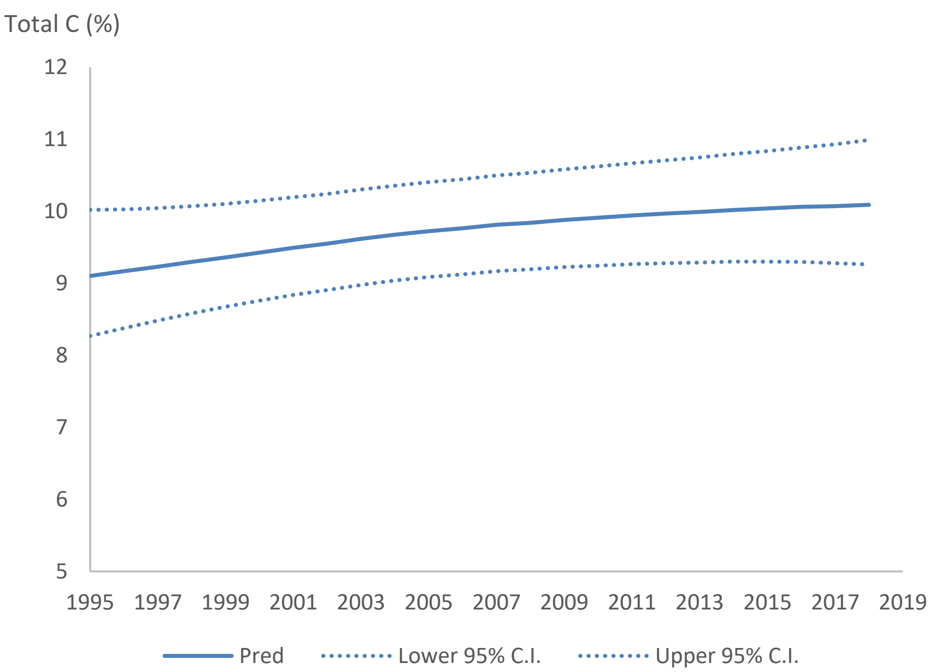

Figure 27: **Change in mixed modelling average total C 1995-2018 for all sites (all land uses and soil orders) with 95% confidence limits.**

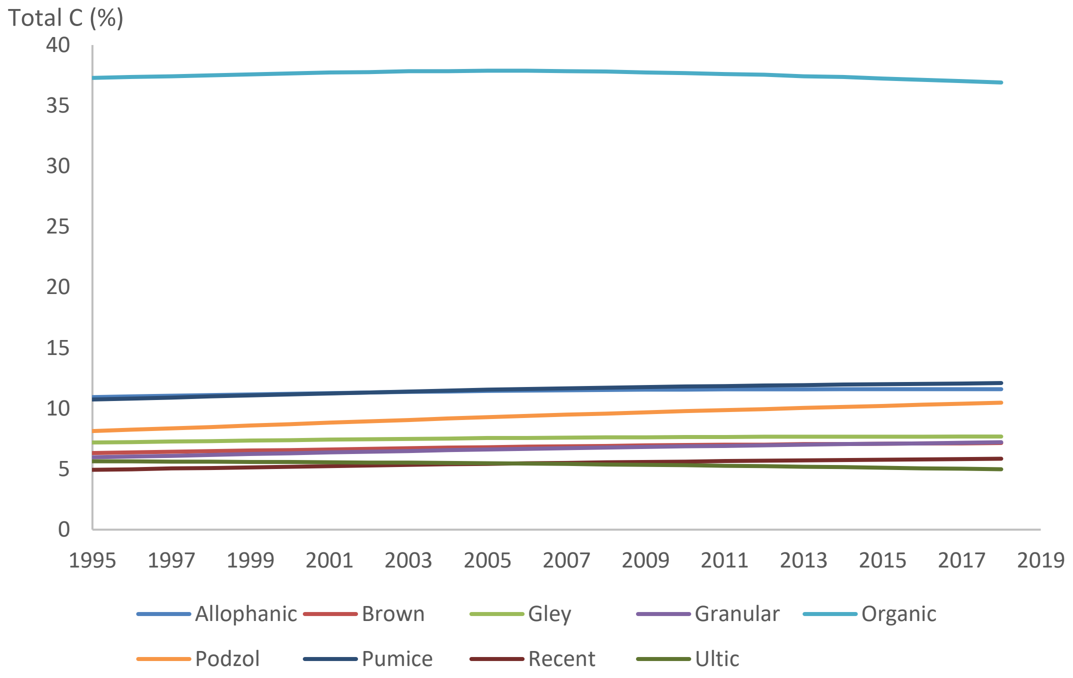

Figure 28: **Change in mixed modelling average total C 1995-2018 by soil order. Note the change in scale due to Organic Soils.**

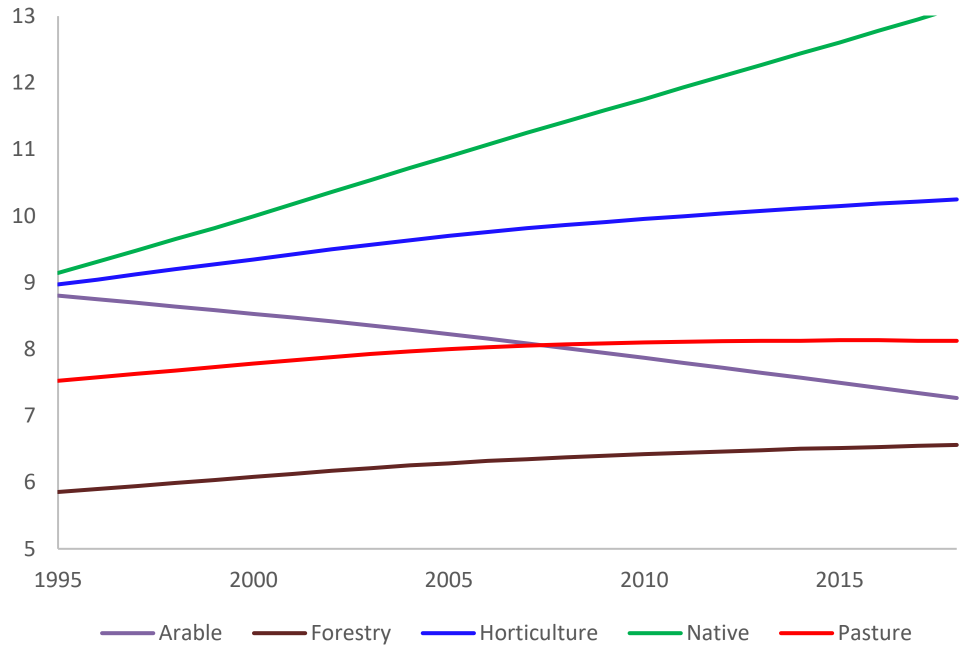

Figure 29: **Change in mixed modelling average total C 1995-2018 by land use.**

Measurements below lower targets (2-3% depending on soil type – note that Organic soils must have 18% C to be classified as Organic so could never fail to meet the target, Appendix III of Taylor et al., (2017)) are considered indicative of depleted soil carbon and soil organic matter. There is no upper target for total C.

Consistent with the modelling data, the number of arable sites meeting targets for total C decreased (Figure 30). Also of note are forestry sites, where some sites decreased after harvest leading to their not meeting targets but they appear to have recovered, so now meet total C targets.

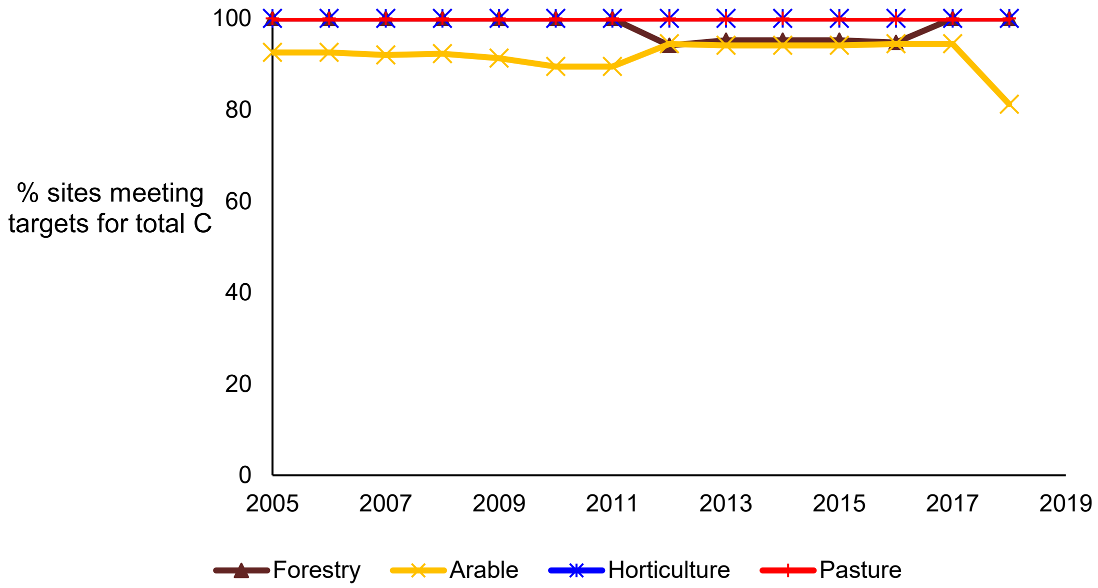

Figure 30: **Change in percent sites meeting the target for total C 2005-2018 by land use. All native sites meet targets (100%) in all years so are not shown.**

3.7 Changes in pH 1995-2018.

For the overall dataset, mixed modelling results showed evidence of a non-linear pattern over time (Figure 31). Decreasing linear trend differences between soil orders were significant (p=0.052, Figure 32), e.g. for Allophanic and Recent Soils, soil pH had decreased by 0.036 and 0.056 per year over the last 5 years, respectively. In comparison, there were significant differences in linear trend between land uses (p=0.033, Figure 33).

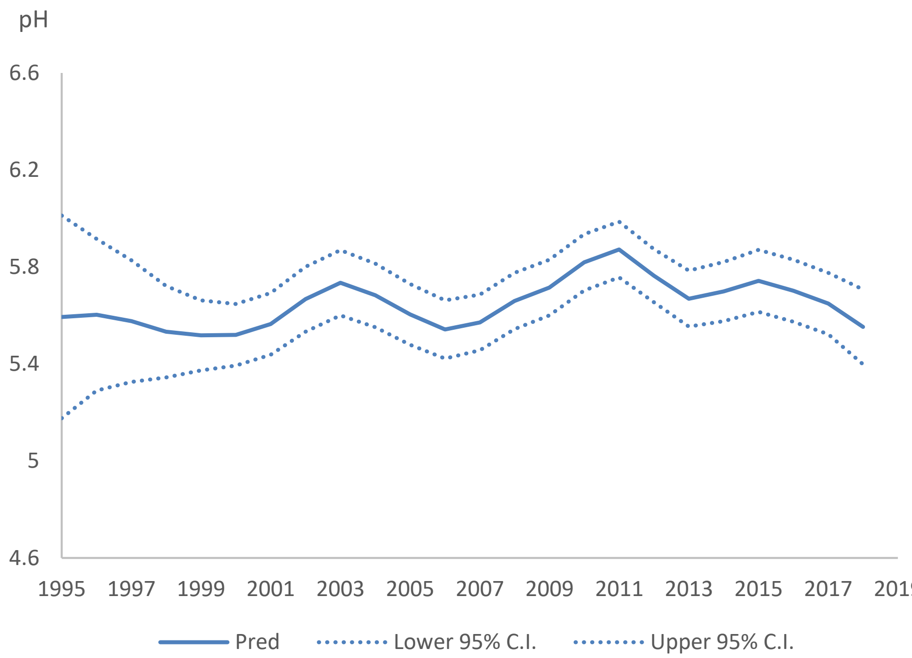

Figure 31: **Change in mixed modelling average pH 1995-2018 for all sites (all land uses and soil orders) with 95% confidence limits.**

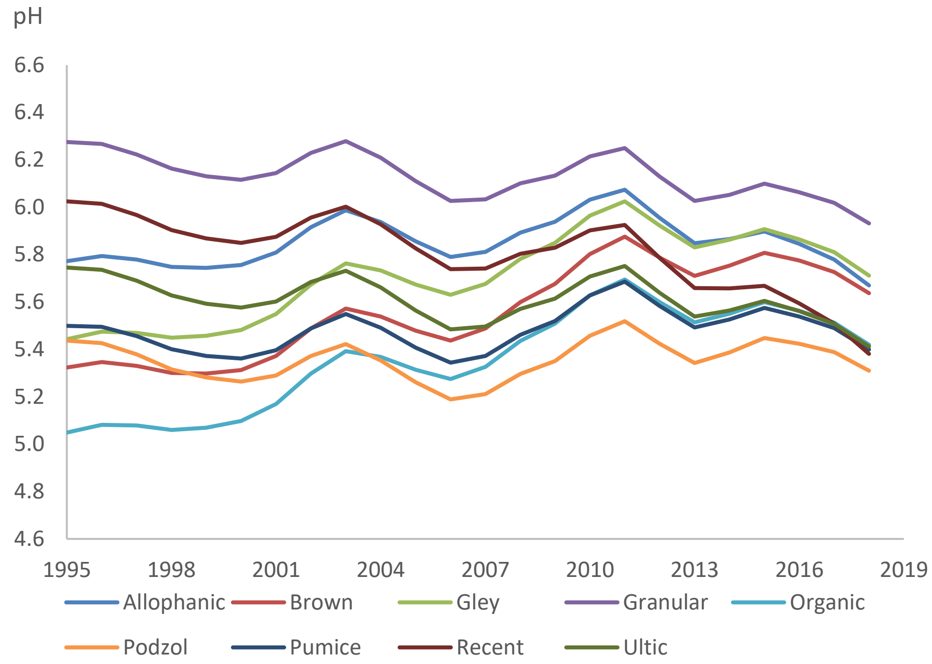

Figure 32: **Change in mixed modelling average pH 1995-2018 by soil order.**

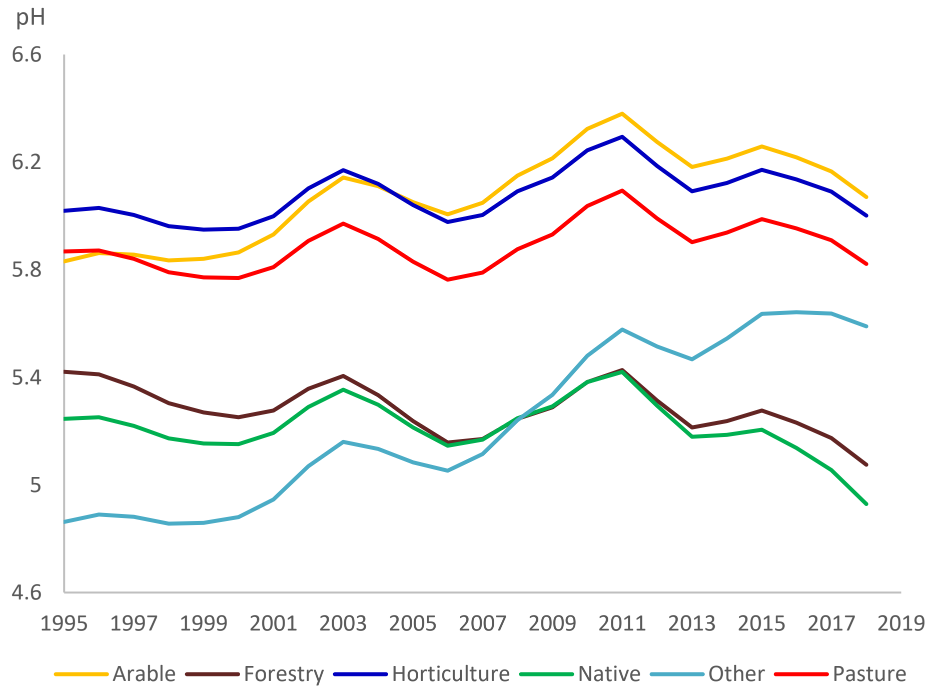

Figure 33: **Change in mixed modelling average pH 1995-2018 by land use.**

Measurements below the lower targets (3.5-5.5 dependent on land use and soil type; Appendix III of Taylor et al., (2017)) are considered indicative of increased acidification, while those above the upper target (6.6-7.6 dependent on land use and soil type; Appendix III of Taylor et al., (2017)) are considered indicative of increased alkalisation.

Consistent with the mixed modelling results, there appeared little change in the percent of sites meeting the lower pH target, except for a slight decrease for pasture (Figure 34). All sites met the upper targets (data not shown).

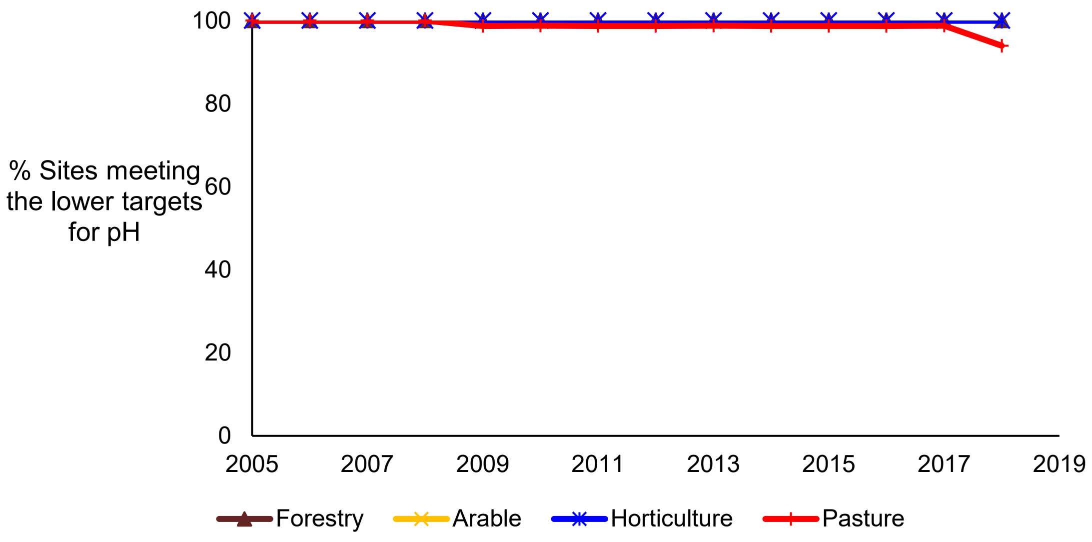

Figure 34: **Change in percent sites meeting the lower target for pH 2005-2018 by land use. All native sites meet targets (100%) in all years so are not shown.**

3.7.1 Supporting analyses

There were significant ($p<0.05$) differences by ANOVA in the C:N ratio with land use. Land where fertiliser was applied (arable, horticulture and pasture) had lower C:N compared with native or forestry land uses (Figures 37, 38). The C:N ratio derived from total C and N had declined for all land uses except forestry.

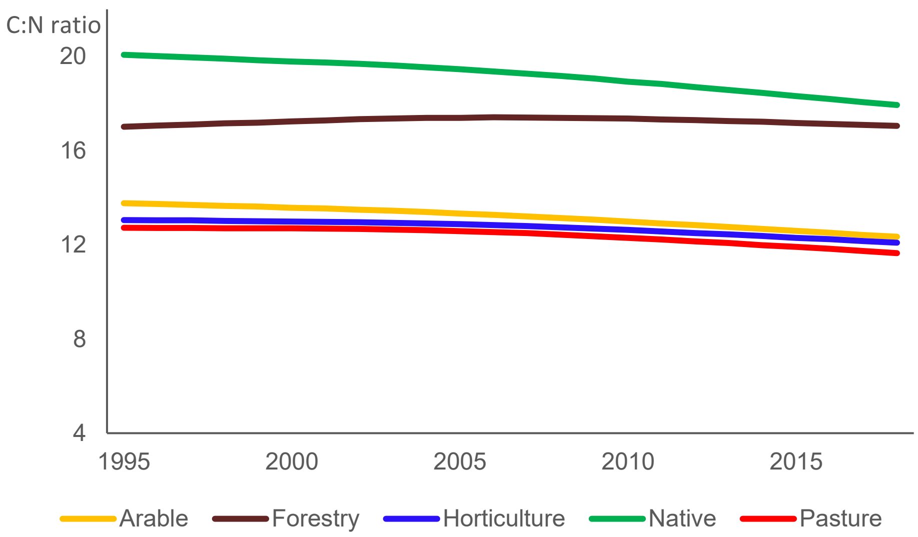

Figure 35: **Change in mixed modelling average C:N ratio 1995-2018 by land use.**

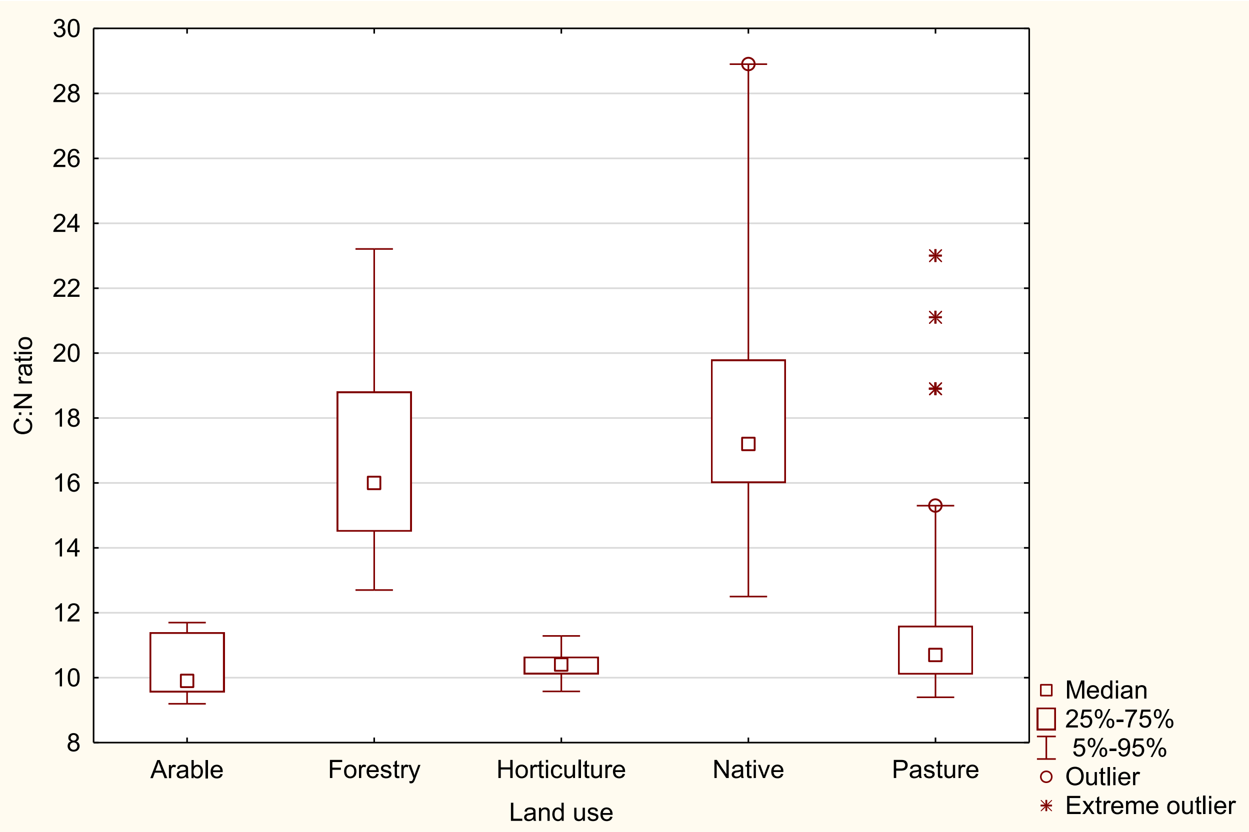

Figure 36: **Boxplots of the C:N ratio for native, forestry, arable, horticulture and pasture land uses from soil quality sites in 2018. Boxes are quartiles, whiskers are 95% confidence.**

There were significant (p<0.05) differences by ANOVA in aggregate stability with land use (Figure 39). Arable land had considerably lower aggregate stability than native, forestry, horticultural and pastoral land uses. In addition, several sites originally in pine forest were converted to pasture in 2008. Aggregate stability declined considerably with conversion but increased once pasture was established. However, levels have not yet returned to previous or to levels for long-term pasture.

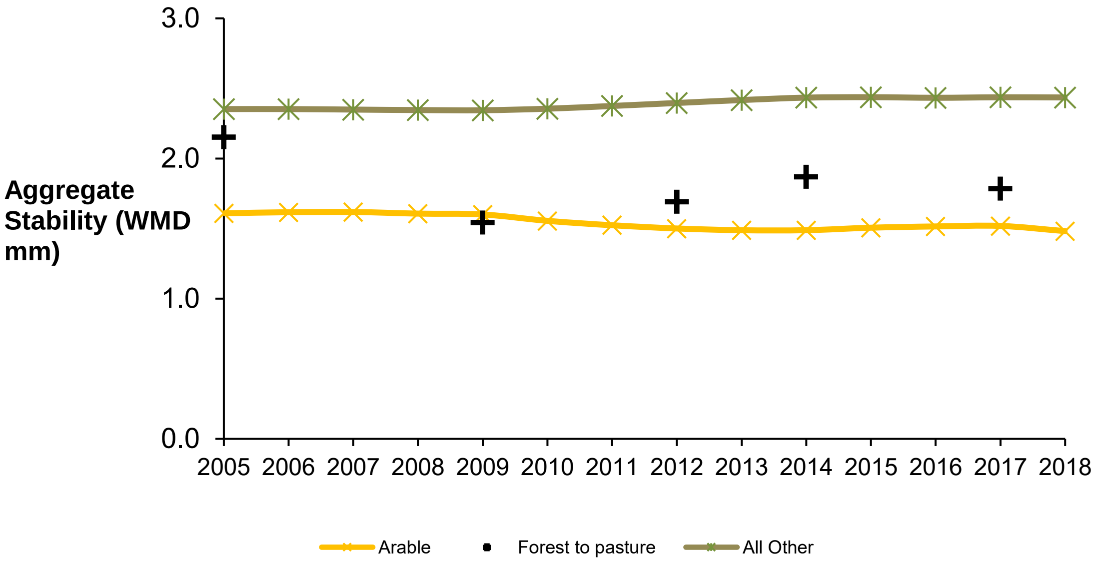

Figure 37: **Change in 5-year rolling average aggregate stability for arable and sites converted from forest to pasture compared with all other land uses.**

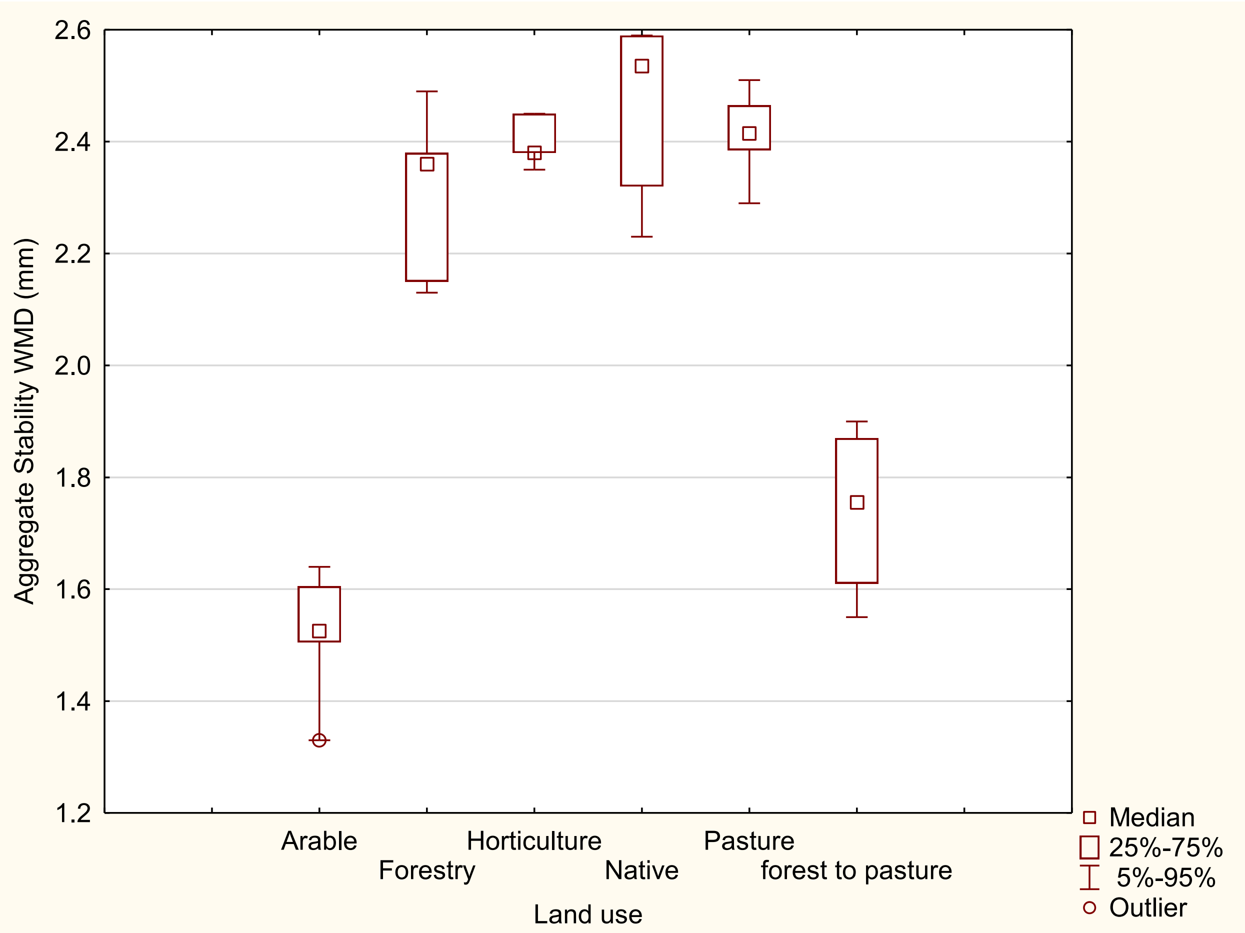

Figure 38: Boxplots of aggregate stability for native, forestry, arable, horticulture, pasture and pine forest converted to pasture land uses from soil quality sites in 2018. Boxes are quartiles, whiskers are 95% confidence.

Analysis of HWC and HWN was carried out on all soil quality samples from 2005 and on earlier samples where archived samples existed. This test is currently being investigated as a replacement for the AMN soil quality indicator. Results were amalgamated into a five-year rolling average to allow for annual fluctuations and to include a full set of sampling sites for trend analysis. Results showed significant ($p<0.05$) differences by ANOVA in HWC and HWN for land use (Figures 41-44). Native sites had significantly ($p<0.05$) higher HWC values followed by pasture, forestry, horticulture, and arable had significantly ($p<0.05$) lower HWC values (Figure 42). There was a significant ($p<0.05$) decline in HWC for pasture between 2010 and 2018 (Figure 41) but changes for other land uses were not significant.

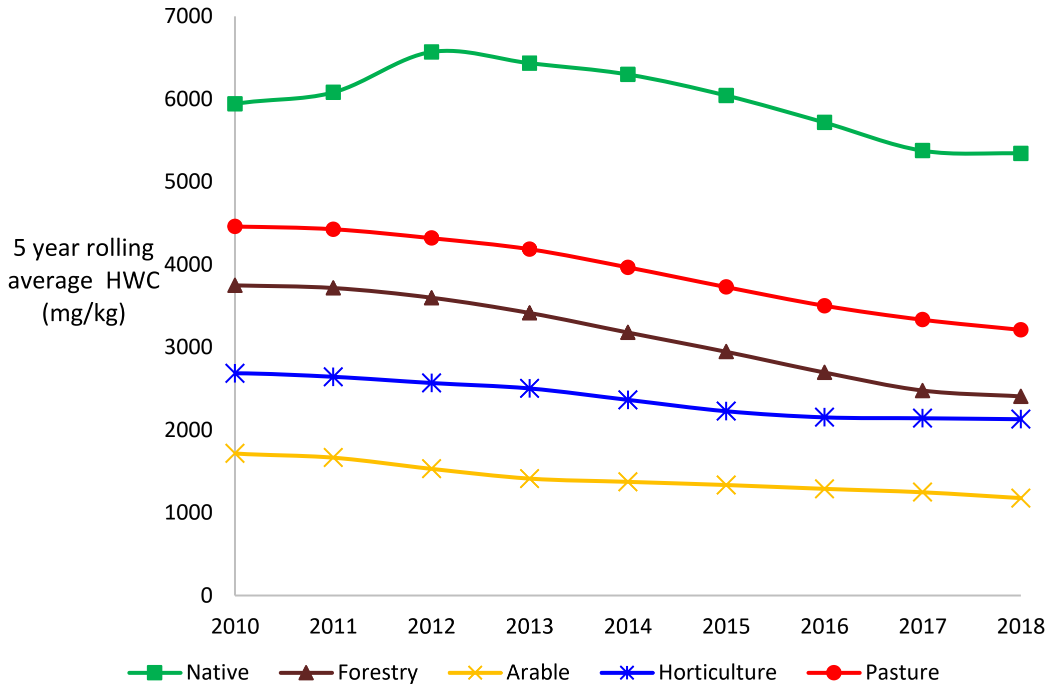

Figure 39: **Five-year rolling average HWC for native, forestry, arable, horticulture and pasture.**

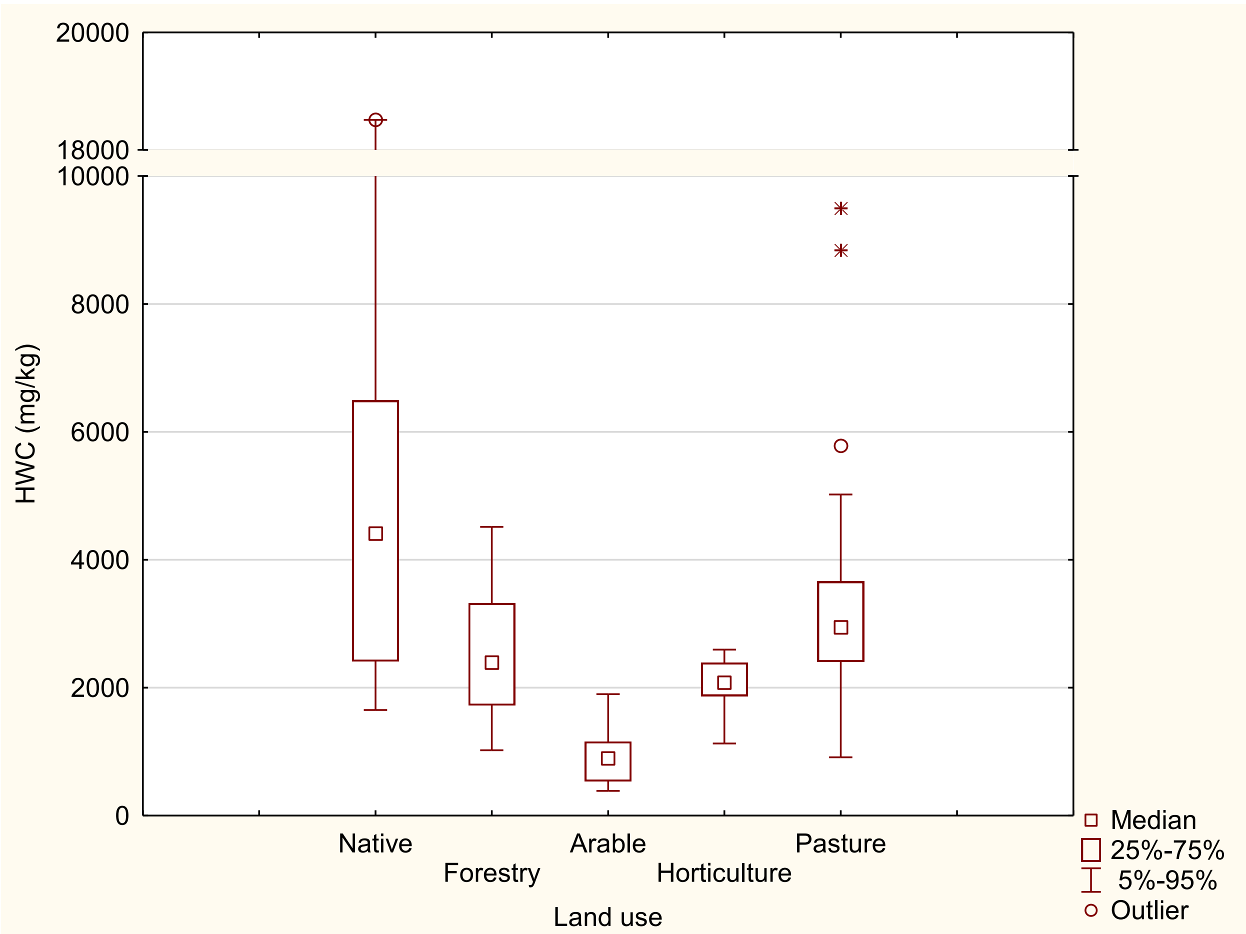

Figure 40: **Boxplots of HWC for native, forestry, arable, horticulture and pasture land uses from soil quality sites in 2018. Boxes are quartiles, whiskers are 95% confidence.**

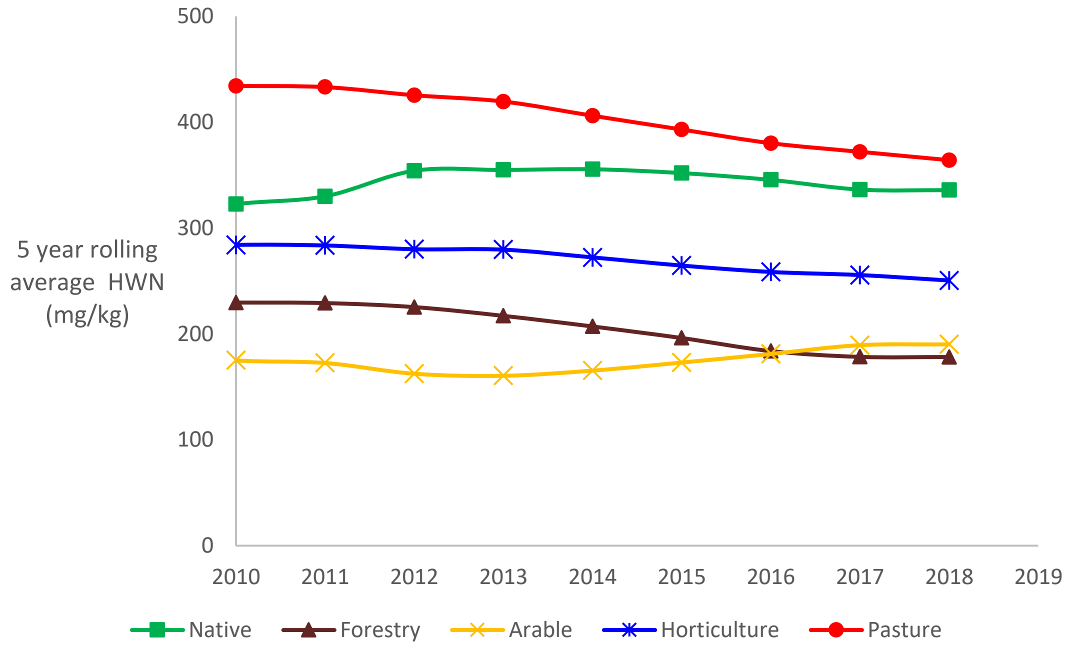

Figure 41: **Five-year rolling average HWN for native, forestry, arable, horticulture and pasture.**

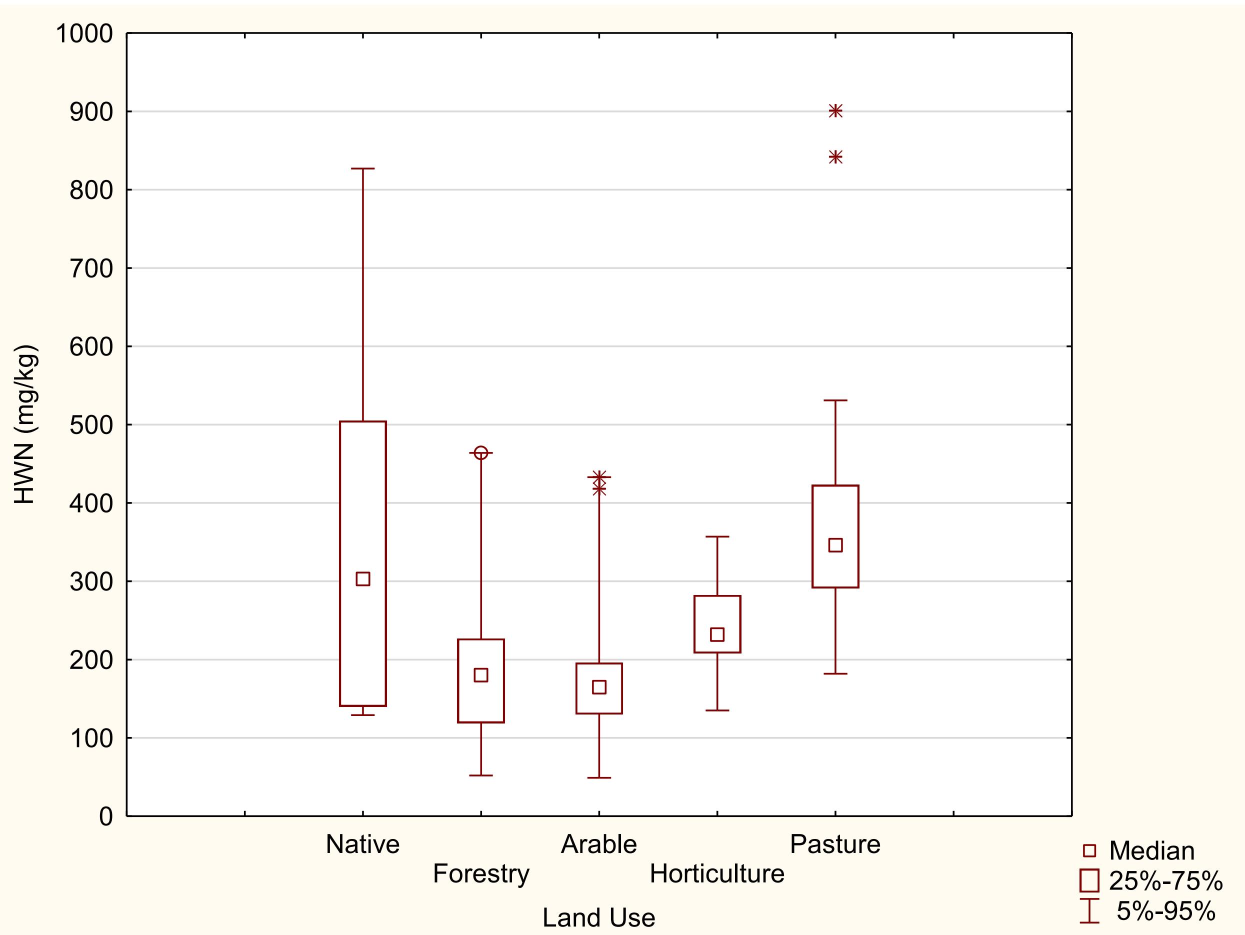

Figure 42: **Boxplots of HWN for native, forestry, arable, horticulture and pasture land uses from soil quality sites in 2018. Boxes are quartiles, whiskers are 95% confidence.**

The order, from highest to lowest, for HWN was different to HWC. Pasture had the highest HWN values, followed by native, horticulture, forestry, and arable had the lowest values (Figure 39). There was a significant ($p<0.05$) decline in HWC for pasture between 2010 and 2018 (Figure 40) but changes for other land uses were not significant.

Some sites had been analysed for change in the minor isotope of nitrogen, ^{15}N, to assess this measurement's potential as an indicator of nitrogen loss over the long-term. Molecules containing ^{15}N are discriminated against in several processes associated with equilibrium and kinetic fractionations. Soils with higher gaseous losses of N tend to have higher $\delta^{15}N$ values (Craine et al., 2015). $\delta^{15}N$ is measured in per thousand (‰). There were highly significant ($p<0.05$) differences by ANOVA in $\delta^{15}N$ for land use. When the land uses were looked at in more detail, arable was significantly higher ($p<0.05$) than horticulture and pasture, which were significantly higher than forestry, which were significantly higher than native (Figure 45).

There was insufficient information to assess trends in $\delta^{15}N$.

$\delta^{15}N$ was compared at paired water irrigated or non-irrigated sites (Figure 44). $\delta^{15}N$ increased with time at both irrigated and non-irrigated sites but the increase was greater at irrigated sites.

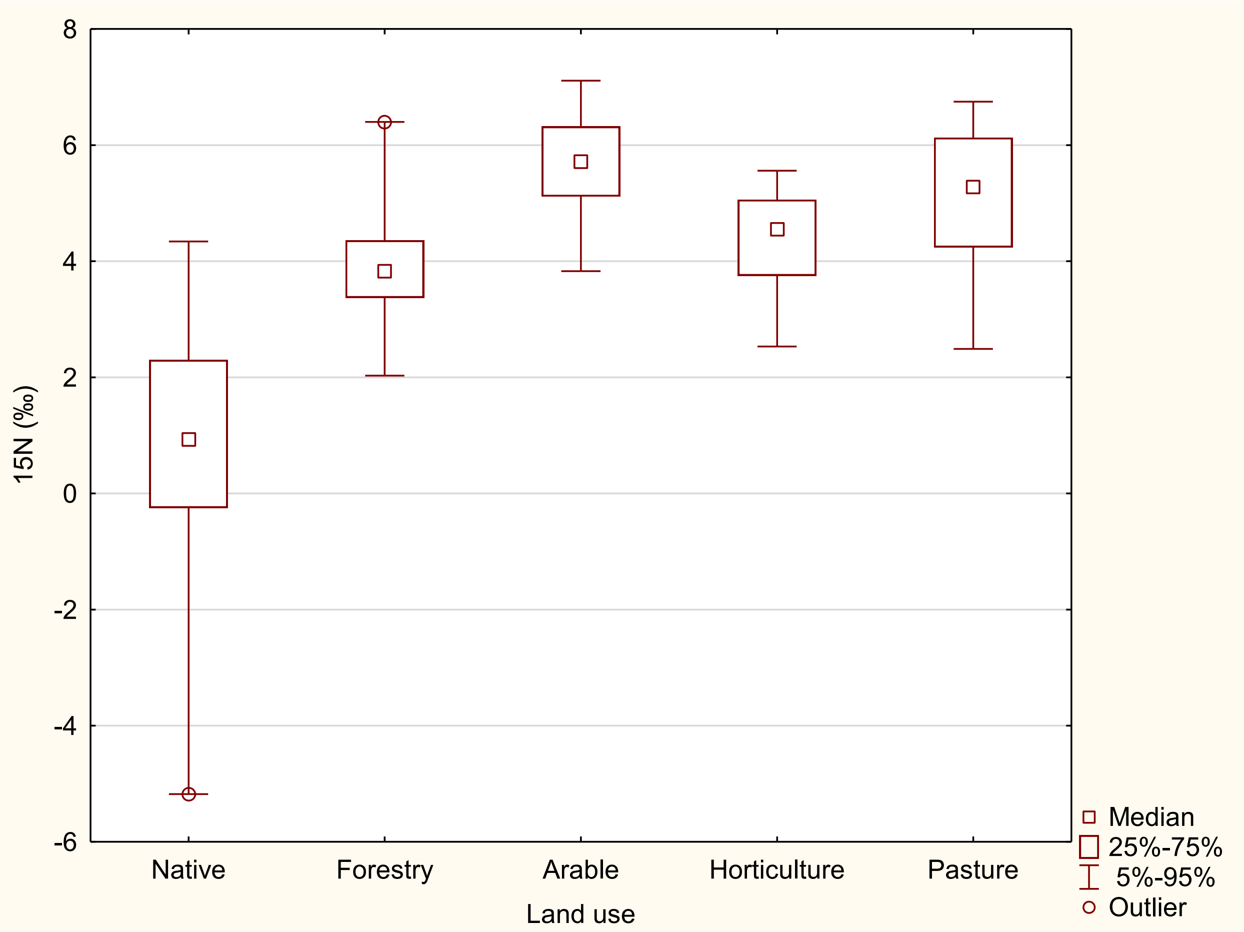

Figure 43: Boxplots of 15N for native, forestry, arable, horticulture and pasture land uses from soil quality sites in 2018. Boxes are quartiles, whiskers are 95% confidence.

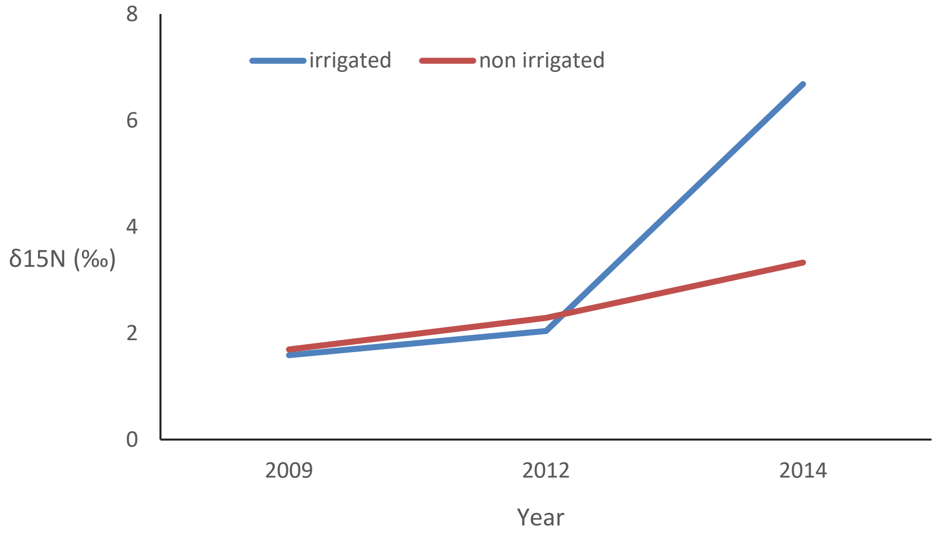

Figure 44: Measurements of change in 15N for irrigated and non-irrigated Taupo Pumice soils.

4 Discussion

Soil is constantly changing. Some processes are natural but human activity can accelerate these processes or initiate new processes resulting in changes to soil quality. These changes can be positive, e.g. increased vegetative coverage can reduce erosion, but many anthropogenic activities increase pressure on the soil resource and reduce the capacity of soils to carry out functions and services. Ministry of Primary Industries (2015) identified six drivers of pressure, agricultural intensification, land use change, climate change and legacy effects from forest clearance, land development, fertiliser application, and cultivation. Monitoring allows assessment of the severity and extent of any adverse effects resulting from these pressures. The European Environmental Agency (2012) attributed widespread soil degradation, leading to a decline in the ability of soil to carry out its ecosystem services, largely to non-sustainable uses of the land. On an area weighted basis, the main soil quality issues for productive land in the Waikato region occurred on pastoral land, but issues also occurred on arable, horticultural, and forestry land (Figure 3). The main soil issues were surface compaction, excess nutrients, and loss of SOM.

4.1 Issue: compaction (indicators: macroporosity, supported by bulk density)

Surface compaction is an issue for pastoral, horticultural, arable and forestry land uses in the Waikato. Macroporosity is considered the better indicator of compaction than bulk density as it is sensitive to structural changes (Drewry et al., 2008; Ball et al., 2007). Pressure on the soil surface forces the soil aggregates closer together, deforming the structure and reducing the soil porosity, bulk density increases and macroporosity decreases, restricting the diffusion of oxygen and the infiltration of water (Hargreaves et al., 2019; Drewry et al., 2008; Bilotta et al., 2007). This in turn can promote the generation of surface runoff, localised flooding and bank erosion (Taylor et al., 2009). Sediment, pathogen, and contaminants in overland flow also increase, especially if the soil is bare of protective vegetation (Bilotta et al., 2007; McDowell et al., 2003a). In addition, plant uptake of N and P tends to be lower in compacted soils due to shallower rooting, reduced available N concentrations and reduced mineralisable N (Hargreaves et al., 2019; Nevens & Reheul, 2003; Lipiec & Stepniewski, 1995).

Increased effects may be seen with increased intensification, e.g. stocking density (Bilotta et al., 2007). Spring pasture relative yield was shown to have a highly significant relationship with macroporosity (Drewry et al., 2004). Loss of macropores reduced spring pasture production and decreased clover growth and clover N fixation (Betteridge et al., 2003, Figure 47). However, macroporosity could recover somewhat over Summer/Autumn (Drewry et al., 2004). Compaction damage of soil under arable crops also decreased crop yield (Van den Akker & Schjønning, 2004; Horn et al., 1995).

Compacted soil may also contribute to global atmospheric warming due to increased emission of CO_2, CH_4 and N_2O from such soils (García-Marco et al., 2014; Ball, 2013; Horn et al., 1995).

Only 35% of pastoral sites meet the lower macroporosity target of 10% macropores in 2018 (Figure 11). However, this is an improvement on the previous five years with an increase in macroporosity over the last five years (Figure 10). This trend needs to continue to meaningfully improve soil quality but improvement could be jeopardised if further intensification occurs, especially if this is combined with a couple of wet winters.

In comparison, 75% of arable sites meet the lower macroporosity target of 10% macropores in 2018 (Figure 11) but average macroporosity has decreased over the last five years (Figure 10). Increased compaction was also indicated by the increased average bulk density (Figure 6). Compaction in arable land can be minimised with the adoption of techniques such as precision agriculture and not driving on the soil when it is wet (Raper, 2005).

Increased compaction was also indicated for horticultural sites by a clear but non-significant decline in macroporosity between 2000 and 2018 (Figure 10). There was also a large decrease in the proportion of sites meeting the lower macroporosity target of 10% macropores for the corresponding period (Figure 11). However, the data from last 5 years shows macroporosity, therefore compaction, has been stable for horticultural sites.

Sites meeting the low macroporosity target for forestry have decreased from 100% to 75%. Several sites have been harvested and the soil disturbed in the process. Ground-based logging equipment may cause soil disturbance by displacing or mixing litter and soil, and/or compacting the soil. In particular, the effects of soil compaction may last for decades unless remedial action is taken, while loss of topsoil may lead to reduced production in subsequent harvests (Murphy et al., 2004). However, new saplings were still establishing on the soil quality sites, which are expected to stabilise to some extent over the next 5-year period.

The reasons as to why compaction first increased and then stabilised more recently are unclear and should be further investigated. Nevertheless, soils under pasture have had low average macroporosity since soil quality monitoring began. When soils are already compacted, it is difficult to compact them further and many pastoral soils appear to have come to a steady state, i.e. they are unlikely to become more compact under the present land use. At compacted pastoral sites, the grazing animals reduce the protective vegetative cover, while hooves physically compact the soil surface (treading).

All sites met the upper target for bulk density (Figure 7), but this result reflects the insensitivity of the bulk density measurement for the light volcanic soils found in the Waikato.

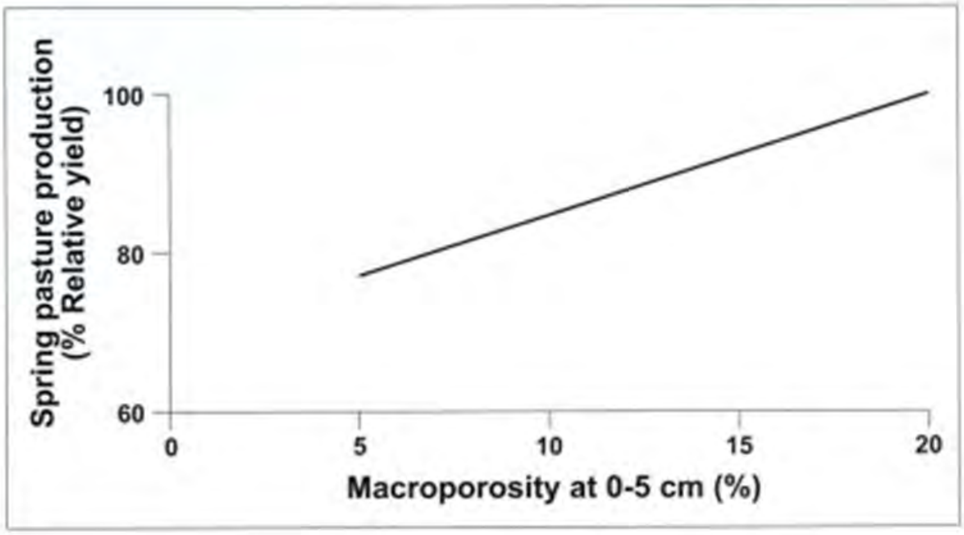

Figure 45: The effect of soil macroporosity on spring pasture production (Betteridge et al., 2003).

Recovery of macroporosity, once pastoral animals were removed, varies depending on soil type, the extent of initial damage, management methods, and climate, but may take anything from weeks to months, or even years (Drewry, 2006). Recovery can be relatively quick for moderate compaction events, e.g. from a macroporosity measurement of 12% to 18% in 18 months (Drewry et al., 2004), although complete recovery from a larger event with lower macroporosity may take many years (Drewry & Paton, 2000).

Damage to the soil by grazing animals can be minimised by management of livestock and land. Bilotta et al. (2007) discussed several options including reducing stocking density, moving livestock off wet pasture onto hard standings or into housing, or reducing the length of the grazing season. In some soils, installing drainage can increase the soils resistance to damage if the watertable can be kept below 500 mm. Tillage and reseeding can break up a surface pan but also accelerate the decomposition of SOM, which could lead to an even worse situation, so should be used with caution.

4.2 Issue: excess nutrients (indicators: Olsen P, supported by total nitrogen)

Nutrients were assessed by two indicators, Olsen P and total N. Olsen P estimates plant available P, thus fertility of a soil, and is a factor in assessing the risk of P loss from soil (McDowell et al., 2003b). Total N measures all the N soils. This measurement provides information on the N fertility status of a soil as well as the amount of soil organic matter, which provides sites for N adsorption.

There were two main risks associated with nutrients in soil, too much and too little. Too much leads to increased risk of transferring nutrients to water bodies where they can contribute to changes in the composition of local biological communities, the formation of algal blooms, or directly impact human and animal health (Buckley & Carney, 2013; Monaghan, 2012). The filtering and buffering capacity of soils is frequently exceeded by excess of nutrients resulting in emission of N, P, or both to the environment. Fertilisation with N significantly increased losses of N in drainage and P in runoff (Monaghan et al., 2005; Edmeades et al., 2020; Figure 48). Excessive and high levels of nutrients in soil are associated with effects in water. Effects can be direct or obtuse, e.g. nutrient driven acidification was reported in the Firth of Thames (Law et al., 2018). Nutrient loads transferred through the Piako and Waitoa Rivers were driving increased phytoplankton blooms, which led to increased heterotrophy and respiration. The resulting additional CO_2 was incorporated into the water column as carbonic acid, thereby decreasing pH and potentially negatively impacting marine organisms (Law et al., 2018).

When too much N and P are present in surface water, algae grow faster than ecosystems can manage. Substantial increases in algae harm water quality, food resources and habitats, and decrease the oxygen that fish and other aquatic life need to survive. Some algal blooms are harmful to humans and animals because they produce elevated toxins and bacterial growth. On the other hand, human infants are vulnerable to nitrate in drinking water, which is often sourced from groundwater or surface water. Conversely, deficient nutrient status reduces plant growth and productivity.

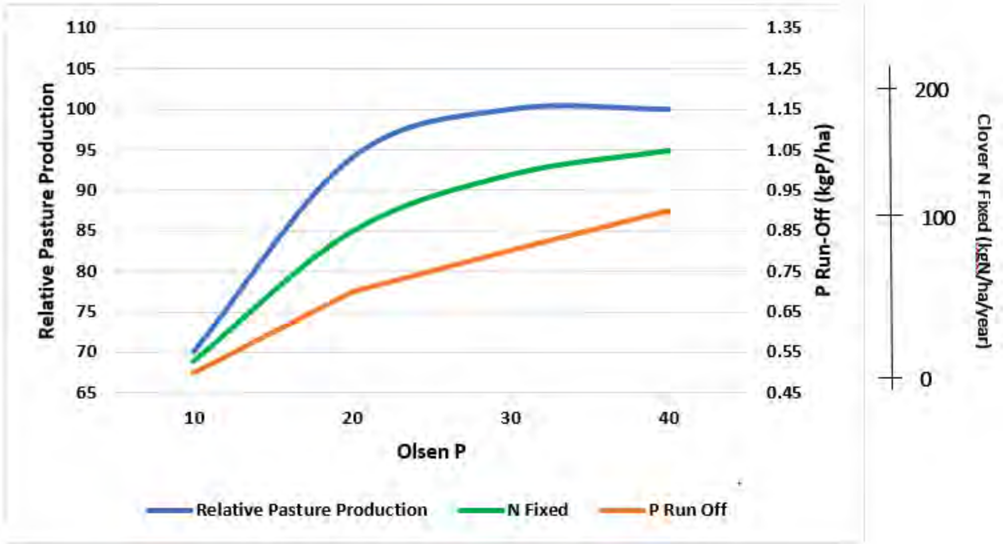

Figure 46: **The estimated effects of increasing Olsen P on relative pasture production, clover N fixation and P runoff (Edmeades et al., 2020).**

Increased nutrient fertility was indicated by increased total N and Olsen P, and resulted in a decrease in the proportion of sites that met targets for pasture, arable and horticulture (Figures 15 & 16, 20 & 21). Results showed average Olsen P and total N in pastoral land are currently within the high or excessive categories for these indicators (Figures 15 & 20). The number of sites meeting the Olsen P and total N targets is stable for pasture and arable land uses, while the trend for horticulture is a decrease in sites meeting these targets. (Figures 16 & 21). The stability of arable and pastoral sites is likely due to a combination of economic, social and regulatory pressures resulting in farmers better managing fertiliser inputs. The decrease in the number of sites meeting total N and Olsen P targets for horticulture can be explained to some extent by the conversion of lower intensity apple orchards to higher intensity kiwifruit orchards.

That both native and pastoral land uses showed significant ($p<0.05$) increases in total N (Figure 20) is surprising as nitrogen should not be applied to indigenous vegetation. The increase in total N for native sites is similar to the increase in total C, suggesting accumulation of stabilised organic matter (Figure 29). However, the total C:total N ratio (C:N ratio) declines for all land uses except forestry (Figure 37). Total C is indicative of organic matter, which retains N as one of its functions (Dick & Gregorich, 2004). The lower the C:N ratio, the more rapidly nitrogen will be released into the soil for immediate crop use but low C:N ratios below 10 have been associated with losses of N (Mackay et al., 2013; Lovett et al., 2002; Tiquia & Tam, 2000). A possible mechanism could be reduced competition for available N between plants and microorganisms with the consequently enhanced decomposition of plant residues by maintaining high microbial activity (Kumar & Goh, 1999). This hypothesis is supported by the AMN, HWC and HWN, and $\delta^{15}N$ results.

- AMN results are generally higher for pastoral sites with high fertility than for other land uses with lower fertility (Figure 25).

- HWC and HWN estimate labile C and soil N mineralisation potential for soil quality samples, i.e. how much C and N are readily available for plants and microorganisms

(Curtin et al., 2017). Pastoral soils had abundant labile C and N available for microbial respiration and subsequent loss as CO_2 (Figures 41-44).

- $\delta^{15}N$ values, which are revealing of N loss with higher values indicative of greater n loss, were higher at intensively farmed sites or sites farmed for a long time or sites receiving irrigation (Figures 43-44). Soil quality monitoring sites converted from pine forest to irrigated dairy farm showed a steady increase in $\delta^{15}N$ values over 5 years (Figure 44). Mudge et al. (2013 and 2016) showed N losses and $\delta^{15}N$ from plots receiving irrigation were higher than those from non-irrigated plots.

Finally, recent work has concluded increased biological activity and detritus from plants is contributing to the accumulation of organic P (Touhami et al., 2020). Thus, it appears likely that increased microbial activity is contributing to nitrogen loading onto soils, and specifically soil organic matter, but there appears to be a concurrent increase in nitrogen loss.

The results presented here are consistent with national data published by the Ministry for the Environment & Statistics NZ (2017), which found nitrogen leaching from agricultural soils was estimated to have increased 29% from 1990 to 2012. Ministry for the Environment & Statistics NZ (2017) also reported nitrate-nitrogen concentration was 10 times higher and dissolved reactive phosphorus concentration was 2.5 times higher in the pastoral land class compared with the native land class for the period 2009–13.

Changes in Olsen P trends, seen in the graphs (Figures 13-15), appear to match with economic pressures, such as world commodity prices, e.g. Olsen P in pasture increases as the world milk commodity price (DairyNZ, 2016) and returns for meat increase. Horticulture may also be affected by the PSA infection in kiwifruit reducing profitability (Ministry of Primary Industries, 2012). The current increasing focus on nutrient budgets is likely reflected in the recent decline in Olsen P.

The accumulation of P has been referred to as legacy P (Motew et al., 2017), which can provide a long-term source of P to plants, as a nutrient and in the wider environment, as a contaminant. There are considerable opportunities on some farms to reduce P runoff and at the same time enhance farm profitability, by mining down the Olsen P to the economic optimal range (Edmeades et al., 2020). Enclosed water bodies, such as lakes can be significantly affected by legacy P and water quality is more vulnerable to heavy rain events when catchments have higher amounts of legacy P (Motew et al., 2017). Increased heavy precipitation is expected with climate change (Intergovernmental Panel on Climate Change, 2014). The greatest risk of P loss is on soils that are poorly drained, have lower structural resilience or are on slopes, while the greatest risk of N loss is on very well drained and excessively drained soils (Monaghan 2012). The air and water quality impacts of the N exports in agricultural systems have been reported as cause for great concern (Davidson et al., 2012). When linked together, surface compaction and excessive nutrient concentrations in pasture have been linked to modified soil hydrological behaviour and, ultimately, the deterioration of water quality in ground and surface waters (Biolotta et al., 2007).

Naturally, reducing both compaction and excessive nutrients can be expected to result in improved ground and surface water quality. Diffuse contamination of surface waters with P and N could be reduced by reducing surplus nutrient flows to groundwater and waterways by reducing surface runoff (overland flow) during high intensity storms and maximising the efficiency of fertiliser use. Methods could include applying no more than the amount of fertiliser needed for production (Buckley & Carney, 2013), managing critical source areas (McDowell & Srinivasan, 2009), and riparian planting (Lee et al., 2003; Parkyn et al., 2003), e.g. a management practice of sowing a low-P-requiring grass in near stream areas has been suggested (McDowell et al., 2014). However, even if compaction and excessive nutrients were removed today there would be a delay or lag until water quality responded.

4.3 Issue: deficient nutrients (indicators: Olsen P, total nitrogen)

Conversely to excess nutrients, deficient nutrients are apparent at some sites. Deficient P nutrient status was apparent at 11% pasture sites (Figure 17), despite increased phosphorous fertility at most and on average for all pasture sites. These sites with P deficit are all on hilly country where topdressing with aircraft may be required. Some of these sites have not met the lower pH target in recent years (Figure 34), suggesting they are not receiving sufficient lime. Low phosphorous status has been recognised as a major factor limiting pasture production on hill country soils (Gillingham et al., 2007; Edmeades et al., 1984) but economic application of fertilisers and lime appears a major challenge.

Total nitrogen was, on average, below the target for total N (0.25% target; 0.22% measured) and deficient for optimum growth for most pastoral and arable plants, but this was only in soil under arable land use. These sites also did not meet the target for total C (average total C 2.4 %) and HWC and HWN are also low at these sites indicating low SOM. How much SOM was lost was calculated by using sites in indigenous forest as a baseline (average total C about 9.4%, total N about 0.58%). The difference between total C measurements for the two land uses indicates about two thirds of the SOM has been lost at these arable sites. Average total N has also decreased from 0.58% to 0.22% because one of the services provided by SOM is nitrogen storage and regulation and this has been reduced with the loss of SOM (Dick & Gregorich, 2004; Kumar & Goh, 1999). The HWC and HWN data also indicate there is low mineralisable N available for plant uptake at these sites. This shortfall needs to be made up by additional inputs of fertiliser.

The interaction between loss of SOM and deficient nitrogen shows the interdependency between different aspects of soil quality. The issue of loss of SOM is further discussed in the next section.

Some monitoring sites were chosen to compare different land management practices. These “paired” sites included water and dairy shed effluent (DSE) irrigated sites and non-irrigated companions. Although the resulting dataset was too small to have confidence in statistical analysis, results were consistent for Readily Available Water (RAW) but not with other physical parameters, such as total available water. Irrigation with either water or DSE resulted in lower values for RAW, while irrigation with DSE also resulted in lower total C values (Figures 35 and 36). The lower total C values was consistent as a mechanism for lower water storage. Drewry et al. (2020) reported irrigation (with associated increased land use intensity) tends to alter soil physical properties, changing them to be like soils formed under higher rainfall, in a review of the effects of irrigation on soil properties. However, there was a lack of consist results across studies and there remains knowledge gaps of the effects of irrigation intensification on soil water storage and movement.

4.4 Issue: Loss of soil organic matter (indicators: Total C, Total Nitrogen)

Soil organic matter (SOM) is fundamentally derived from residual plant and animal material, transformed (humified) by microorganisms and decomposed under the influence of temperature, moisture, aeration and ambient soil conditions (European Environmental Agency 2012). It is considered a key soil attribute as it affects many physical, chemical, and biological properties that control soil services such as productivity, the regulation of water and nutrients, and resistance to degradation (Dick & Gregorich, 2004). SOM is essential for the viability and life-sustaining function of the soil. For instance, organic acids (e.g. oxalic acid), commonly released from decomposing organic residues and manures, prevents phosphorus fixation by clay minerals and improve its plant availability. Also, polysaccharides (sugars) bind mineral particles together into microaggregates. Glomalin, a SOM substance that may account for 20% of soil carbon, glues aggregates together and stabilises soil structure making soil more resistant to erosion, but porous enough to allow air, water, and plant roots to move through the soil.

Several factors are responsible for a decline in SOM, e.g. conversion of grassland, forests and natural vegetation to arable land; deep ploughing of arable soils; drainage and fertiliser use; tillage of peat soils; crop rotations with reduced proportion of grasses; soil erosion; and wild fires. Excess nitrogen in the soil from high fertiliser application rates and/or low plant uptake, high soil temperatures and moist conditions can accelerate soil respiration and cause an increase in the mineralisation of organic carbon, which in turn leads to an increased loss of carbon from soils (European Environmental Agency 2012; Velthof et al., 2011; Kumar & Goh, 1999).

A direct effect of low SOC is reduced microbial biomass, activity, and nutrient mineralisation due to a shortage of energy sources and loss of habitat. In the arable soils of the Waikato region, aggregate stability, infiltration, drainage, and airflow are reduced compared with pasture or native. Scarce SOC results in less diversity in soil biota with a risk of the food chain equilibrium being disrupted, which can cause disturbance in the soil environment (e.g. plant pest and disease increase, accumulation of toxic substances, etc.). Of importance to the Waikato catchment is SOM's role in retaining nitrogen in the soil. Soil acts as a filter and buffer for N, thus protects water and the atmosphere from N pollution, while soil biological activity can make a contribution to fertility (Velthof et al., 2011; Kumar & Goh, 1999). SOM retains N as one of its functions (Dick & Gregorich, 2004).

The significant decrease in total C ($p<0.05$) for arable and use, indicating loss of SOM is particularly notable as total C for all the other land uses increased over time (Figure 29). Also, average total C and HWC (Figure 41) are considerably lower for arable than for any other land use. Total N for arable land declined following total C levels, while all the other land uses showed an increase over time (Figure 20). The C:N ratio also decreased (Figure 37) or narrowed as average total N decreased more slowly than total C. Cultivation can lead to greater loss of C than N, thus narrowing the C:N ratio (Campbell & Souster, 1982). Narrow C:N ratios have been associated with losses of N (Lovett et al., 2002; Tiquia & Tam, 2000) and with decreased mineralisation of N (Janssen, 1996).

In arable systems, changes in SOM tend to be controlled by the amount of organic C supplied in crop residues and the preservation of microaggregates, which protect SOM within them (Kumar & Goh, 1999; Rasmussen et al., 1980). Microaggregates are broken down during cultivation exposing the C within them to oxidation, while plant residues may be minimal if the whole plant is harvested. Irrigation in dryer locations can also assist decomposition of SOM by keeping soil conditions suitably moist for microorganisms to be active (Drewry et al., 2020). The effect of fertiliser N on soil organic matter content and quality is uncertain. Some studies suggest that use of N fertilizer only may result in a decline of SOM content (Velthof et al., 2011).

However, despite the changes observed in total C content, there appeared little change in the percent of sites meeting the total C targets (Figure 30). Similarly, despite declines in total N in soil under arable land the percentages of sites meeting or not meeting the high target (too much N) did not obviously change (Figure 21). In comparison, from 2011, a small percentage of arable sites no longer met the lower target for total N (Figure 22) indicating decreased soil fertility due to declines in SOM. Prior to 2011, all sites met the lower total N target. Also, only arable land showed a decline in meeting the AMN targets, while 100% of sites from all other land uses met targets (Figure 26), again consistent with declining SOM and reduced N fertility under cropping.

Further evidence that loss of soil carbon was impacting the soils ability to store nitrogen was obtained by using hot water extractable carbon (HWC) and nitrogen (HWN) to estimate labile C and soil N mineralisation potential for soil quality samples, i.e. how much C and N are readily available for plants and microorganisms (Curtin et al., 2017). Results showed arable soils had the lowest average HWC and HWN values indicating loss of labile organic matter (Figures 41-44). Arable soils also had the highest $\delta^{15}N$ values indicating loss of nitrogen from the soil (Figure 45).

Maintaining and improving SOM content where cropping is continuous is critical to maintaining soil quality. Long-term studies have consistently shown the benefit of manures, adequate fertilisation, the return of plant material, including legume cover crops and crop rotation on maintaining agronomic productivity by increasing C inputs into the soil (Diekow et al., 2005; Dick & Gregorich, 2004; Kumar & Goh, 1999). However, even with crop rotation and manure additions, continuous cropping usually results in an overall decline in SOM (Reeves, 1997).

Re-establishment of pasture appears the most practical method of recovering SOM for these systems. However, the recovery of carbon and SOM in arable land that included a lightly grazed pasture rotation of four years or longer was very variable, with no sites re-established to the levels found under permanent pasture (Kirschbaum et al., 2017). This result is consistent with studies looking at regaining carbon after erosion, where recovery of SOM has been seen to take 14-45 years (Larney et al., 2016; Sparling et al., 2003b). Hedley et al. (2009) reported carbon accumulated at a mean rate of 4.07 mg/cm^3 per year at sites in the central North Island that had undergone deforestation and conversion to pasture over 20 years, so accumulation of soil carbon should be considered slow.

Similarly, increased P fertility was indicated by increased Olsen P and the proportion of sites that exceeded the high target for Olsen P became greater over time until about 2014. After 2014, Olsen P declined slightly but average levels are still excessive (Figures 15 & 16).

Although the extent of arable land in the Waikato region is small (about 18,000 ha, with about 11,000 ha in maize in 2015), local impacts can be considerable (Taylor et al., 2017). Any shift to a more plant-based food economy will increase agronomic pressures on soil. With more intermittent precipitation in the future, respiration pulses and the associated nutrient release will intensify and become more variable, contributing more to soil biogeochemical cycling and potentially decreasing water quality (Manzoni et al., 2020).

4.5 Conversion of Forestry to Pasture on Pumice soils

Land use intensity and stock density on all soil types has an impact, but it is notable on Pumice Soils where considerable conversion of land from pine plantations to pasture has taken place. Pumice Soils are very 'light' with weak structure and erode easily when disturbed (Paripovic, 2011). Leaving erosion prone soils, such as Pumice Soils, in native bush or planted in production forestry can help to control erosion. Intensification of agriculture, generally, is reported to have negative impacts on water quality, both in New Zealand (Monaghan et al., 2007) and overseas (Taniwaki et al., 2017; Matson et al., 1997). The inherent 'lightness' and weak structure of Pumice Soils may make them more vulnerable to these impacts than Allophanic or Granular Soils, both of which are weathered volcanic soils and more suited to be used for pastoral land use. Paripovic (2011) reported there was increased soil compaction in the A horizon of Pumice Soils on recently converted sites compared to pine forest sites. Also, the plant root depth of much of the pasture on farms recently converted from forest was relatively shallow (about 10 cm), making pasture especially prone to moisture stress during dry periods.

Landscape recontouring of land converted from forestry to pasture was commonly observed while collecting soil quality monitoring samples. Recontouring of land for viticulture in New Zealand has degraded soil structure, lowered subsoil bulk density, and decreased aggregate stability (Scott 2013, Figure 40). Forest to pasture conversion and increasing grazing intensity can both result in loss of soil carbon and SOM (Steffens et al., 2008; Verde et al., 2008; Alfredsson et al., 1998). Nevertheless, SOM will likely recover over 3-4 decades to a new steady state (Schipper et al., 2017; Hedley et al., 2009).

There are also considerable effects on hydrology as peak flows during floods in forested catchments are 20% of those in pastural catchments, while average flow, annual flood exceedance probability, and sediment yield for forests are half those for pasture (Ausseil & Dymond, 2010; Duncan, 1995). The differences in flow can be attributed to greater interception of rain by pine trees and greater soil moisture storage.

In summary, the conversion process can be expected to result in loss of soil carbon and SOM, increased surface compaction and crusting. The increased compaction may result in increased transport of sediment and contaminants with peak-flows causing localised flooding and bank erosion (Taylor et al., 2009).

4.6 Soil acidification (indicator pH)

There were no clear trends with soil pH, the indicator for acidification, and nearly all sites met targets for pH. So, acidification appears not to be an issue at the regional scale for soils in the Waikato region.

4.7 Potentially mineralisable N

The significant ($p<0.05$) decline in AMN across all soils and land uses over the last 5 years (Figures 24 and 25) suggest a reduction in microbial activity and the amount of nitrogen that can potentially be mineralised from the soil. Additional mineral fertiliser may be needed to make up any shortfall in plant requirements.

5 Conclusions

Soil quality changes for different soils and land uses across the Waikato region for 2005-2018 have been identified. Overall, 11% of managed soil quality sampling sites (farmed or manged forest but not indigenous bush sites) weighted for land area in the region met all seven indicators in 2018. This result is similar to the proportion of sites meeting all seven indicators for the preceding seven years but is down from a high of 18% in 2006.

The main soil quality issues identified were surface compaction, excess nutrients, and loss of soil organic matter (SOM).

Compaction (indicators: macroporosity, supported by bulk density)
Surface compaction is an issue for pastoral, horticultural, arable and forestry land uses in the Waikato. Only 35% of pastoral sites meet the lower macroporosity target of 10% macropores in 2018, but this is an improvement of 10% on the previous five years. There is a trend of improvement for pastoral land and this trend needs to continue to meaningfully improve soil quality. However, the trend for arable and forestry land is negative. Intensification for arable and tree harvest for forestry are causative factors.

Excess nutrients (indicators: Olsen P, supported by Total Nitrogen)
Excessive nutrients are an issue for pastoral land, horticultural and arable land uses. However, the trend in the number of sites meeting the Olsen P and total N targets is stable for pasture and arable land uses. The decrease in the number of sites meeting total N and Olsen P targets for horticulture can be explained to some extent by the conversion of lower intensity apple orchards to higher intensity kiwifruit orchards.

Excessive and high levels of nutrients in soil are associated with negative effects in water, such as phytoplankton blooms and decreased pH. The risk of transfer of nutrients to water increases with increased levels of nutrients in the soil. Diffuse contamination of surface waters with P and N could be reduced by applying no more than the amount of fertiliser needed for production, managing critical source areas, reducing surface runoff, and riparian planting.

Loss of soil organic matter (indicators: Total C, Total Nitrogen)
Loss of SOM is an issue for arable land use. As a result, soil biota diversity, N regulation, aggregate stability, infiltration, drainage, and airflow are reduced in arable soils compared with pasture or native soils. The C:N ratio also decreased or narrowed as average total N decreased more slowly than total C, leading to increased risk of N loss. Maintaining and improving SOM content where cropping is continuous is critical to maintaining soil quality but accumulation of soil carbon is slow and recovery of SOM to pre cultivation levels is likely in the range 14-45 years.

Other issues
Conversely to excess nutrients, deficient nutrients are apparent at some sites. Deficient P nutrient status was apparent at pasture sites on hilly country where topdressing with aircraft may be required. Some of these sites are not also receiving sufficient lime. Economic application of fertilisers and lime appears a major challenge at these sites.

Total nitrogen in soil under arable land use was, on average, deficient for optimum growth for most pastoral and arable plants. These sites also did not meet the target for total C (average total C 2.4 %) and HWC and HWN are also low at these sites indicating low SOM with corresponding low capacity for N storage and regulation.

Considerable conversion of land from planted radiata pine forest to pasture has taken place on Pumice Soils. The conversion process can be expected to result in loss of soil carbon and SOM, increased surface compaction and crusting. The increased compaction may result in increased transport of sediment and contaminants with peak-flows causing localised flooding and bank erosion. The impact of intensification on the biological, physical, and chemical condition of Pumice Soils is likely to be greater than for Allophanic or Granular Soils, as these are both weathered volcanic soils and better suited for pastoral land use.

Representativeness
The total number of soil quality monitoring sites would need to increase from the 154 currently active in 2018 to about 190-200 for the sampling programme to be representative of all the major land uses and soil orders in the Waikato region.

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Appendix I - Statistical Analysis

Spline regression often represents a less biased and more efficient alternative to standard linear, curvilinear, or categorical analyses of continuous exposures and confounders. A model was fitted to each response variable that included:

- Linear and non-linear (spline) trends over time
  - Overall
  - Varying by land use
  - Varying by soil order
- Average levels that varied for each combination of land use and soil order
- Random terms to account for average differences between sites and their linear and non-linear trends over time; the last of these accounts for serial correlation within sites.

This model estimated the true trend when the sites sampled change from year to year. The model was simplified for each variable (indicator) to remove unnecessary spline terms but retaining the overall spline and the site splines. The separate splines for land use and soil order were not required for the models for all variables except macroporosity, which required different splines for each land use.

The models were used to

- describe the trends over time
- estimate the true values at 2018 (the last year)
- estimate the site to site and random within site variation

Each indicator was assessed for statistical trends using linear mixed modelling with random splines overall, by soil order and by land use. Four of the variables required log transforming to get approximate constancy and normality of the residual variation (Table 1). The back-transformed estimates were "bias-corrected" to make this the same as the overall arithmetic mean of all the original values in the data. Data calculated by this method are presented in the results section for 1995-2018.

Table AI 1: The sum of the between site and residual (within site and lack of fit of the model) variances expressed as a standard deviation.

	Site + residual SD
Bulk Density	0.1279
pH	0.3528
Macroporosity	5.8966
Total C% (log)	0.3495
Total N% (log)	0.3450
Olsen P (log)	0.7600
AMN (log)	0.3900

The probability that sites will violate the lower and upper limits for each variable was calculated using these models and the predicted 2018 values (shown in the tables in the following sections).

As a check on the model validity, we calculated the observed and expected violations for the 35 sites present in 2018. These showed a good match (Table 2).

Table AI 2: Observed and expected violations for each indicator for 2018 sites.

	Number	Bulk Density	pH	Macroporosity	Total C%	Total N%	Olsen P	AMN	Total
Below lower limit	Expected	5.1	0.2	11.0	1.0	0.0	5.1	0.8	23.2
	Observed	5	0	17	2	0	4	2	30
Above upper limit	Expected	0.0	0.2	4.3	0.0	16.7	8.8	0.0	30.0
	Observed	0	0	4	0	13	9	0	26

The accuracy of estimated values in any one year was improved by utilising the information from sites before and after each time period, thus increasing the effective sample size. The size of the improvement was shown by calculating the effective sample size, e.g. for the 2018 estimated values the effective sample size was calculated as the square of the ratio of the site + residual standard deviation over the standard error of the estimates (Table 3).

Table AI 3: The overall effective sample size over all soil and land use combinations for each indicator present in the data showing improvement over the 35 sites sampled in 2018.

	Overall effective sample size
Bulk Density	209
pH	118
Macroporosity	123
Total C% (log)	223
Total N% (log)	223
Olsen P (log)	169
AMN (log)	147

Effective sample size for any particular soil or land use (or combination of these) were calculated for each variable; they are roughly in proportion to the number of sites in each (Tables 4-10).

Table AI 4: The effective sample size for soil and land use combinations for bulk density for the 35 sites sampled in 2018.

Effective sample size Bulk density t/m3

Soil_Order	Arable	Forestry	Horticulture	Native	Other	Pasture	Total
Allophanic	13.7	7.0	10.2	5.3		29.8	66.1
Brown	3.6	6.5		3.3		14.7	28.1
Gley	7.6		1.3	2.2	1.3	15.9	28.4
Granular	8.6		3.4	1.2		8.3	21.4
Organic	1.4		0.7	1.2		3.8	7.1
Podzol		2.4		1.0	2.9	2.2	8.5
Pumice	1.6	8.4	0.9	1.6	2.2	24.9	39.6
Recent		1.1		1.2		4.5	6.8
Ultic		1.1				2.0	3.1
Total	36.5	26.6	16.5	17.0	6.4	106.1	209.1

Table AI 5: The effective sample size for soil and land use combinations for macroporosity @ -10 kPa for the 35 sites sampled in 2018.

Effective sample size Macroporosity_at_10_kPa

Soil_Order	Arable	Forestry	Horticulture	Native	Other	Pasture	Total
Allophanic	6.3	5.1	5.1	4.2		14.4	35.1
Brown	2.6	4.3		2.9		8.8	18.5
Gley	3.6		0.5	2.0	0.8	9.0	15.9
Granular	5.6		1.9	1.6		6.1	15.2
Organic	0.5		0.4	1.4		2.7	5.0
Podzol		1.5		0.8	1.3	1.1	4.7
Pumice	1.1	4.9	0.4	0.7	1.1	12.7	20.9
Recent		1.0		1.2		2.8	5.1
Ultic		1.0				1.6	2.6
Total	19.7	17.9	8.3	14.8	3.2	59.2	123.1

Table AI 6: The effective sample size for soil and land use combinations for Olsen P for the 35 sites sampled in 2018.

Effective sample size Olsen P (µg/cm3) (log scale)

Soil_Order	Arable	Forestry	Horticulture	Native	Other	Pasture	Total
Allophanic	11.6	5.8	8.7	4.8		20.6	51.4
Brown	3.3	5.6		2.9		10.8	22.6
Gley	6.8		1.2	2.0	1.2	12.3	23.6
Granular	7.2		3.1	1.1		7.1	18.5
Organic	1.3		0.6	1.1		3.2	6.1
Podzol		2.1		0.9	2.5	2.0	7.4
Pumice	1.5	7.3	0.8	1.3	2.1	17.6	30.6
Recent		1.0		1.1		3.8	5.8
Ultic		1.0				1.8	2.8
Total	31.7	22.8	14.4	15.1	5.7	79.1	168.9

Table AI 7: The effective sample size for soil and land use combinations for Total N for the 35 sites sampled in 2018.

Effective sample size Total N (%) (log scale)

Soil_Order	Arable	Forestry	Horticulture	Native	Other	Pasture	Total
Allophanic	16.1	6.9	11.0	5.2		31.8	71.1
Brown	4.1	7.0		3.2		15.4	29.6
Gley	8.9		1.7	2.1	1.5	16.2	30.4
Granular	8.8		4.0	1.1		8.4	22.3
Organic	1.7		0.7	1.1		4.0	7.5
Podzol		2.6		1.0	3.1	2.5	9.2
Pumice	1.9	9.5	1.1	2.0	2.6	25.6	42.7
Recent		1.1		1.1		4.6	6.8
Ultic		1.1				2.0	3.1
Total	41.5	28.2	18.5	16.8	7.2	110.5	222.7

Table AI 8: The effective sample size for soil and land use combinations for AMN for the 35 sites sampled in 2018.

Effective sample size AMN (mg/kg) (log scale)

Soil_Order	Arable	Forestry	Horticulture	Native	Other	Pasture	Total
Allophanic	8.7	6.1	6.9	4.7		16.2	42.6
Brown	2.9	5.1		3.0		9.7	20.7
Gley	5.3		0.9	2.1	1.1	10.5	20.0
Granular	7.2		2.6	1.3		6.8	17.9
Organic	1.1		0.6	1.2		3.2	6.0
Podzol		1.9		0.9	2.3	1.6	6.7
Pumice	1.3	5.8	0.6	1.2	1.7	13.4	24.0
Recent		1.1		1.1		3.6	5.8
Ultic		1.0				1.8	2.8
Total	26.5	21.0	11.6	15.5	5.1	66.8	146.5

Table AI 9: The effective sample size for soil and land use combinations for Total C for the 35 sites sampled in 2018.

Effective sample size Total C (%) (log scale)

Soil_Order	Arable	Forestry	Horticulture	Native	Other	Pasture	Total
Allophanic	16.4	6.9	11.2	5.2		31.1	70.8
Brown	4.2	7.1		3.1		15.3	29.7
Gley	9.1		1.8	2.1	1.5	16.1	30.7
Granular	8.8		4.1	1.1		8.5	22.6
Organic	1.7		0.7	1.1		4.1	7.6
Podzol		2.7		1.0	3.1	2.6	9.3
Pumice	2.0	9.7	1.2	2.1	2.7	25.3	42.9
Recent		1.1		1.1		4.6	6.8
Ultic		1.1				2.0	3.1
Total	42.3	28.5	19.0	16.9	7.3	109.5	223.5

Table AI 10: The effective sample size for soil and land use combinations for soil pH for the 35 sites sampled in 2018.

Effective sample size pH

Soil_Order	Arable	Forestry	Horticulture	Native	Other	Pasture	Total
Allophanic	6.5	5.3	5.5	4.3		11.8	33.4
Brown	2.5	4.2		2.8		7.2	16.6
Gley	4.1		0.7	2.0	0.8	8.1	15.7
Granular	6.0		2.1	1.4		5.8	15.3
Organic	0.9		0.5	1.3		2.9	5.5
Podzol		1.5		0.8	1.6	1.3	5.2
Pumice	1.1	4.9	0.5	0.9	1.3	10.3	18.9
Recent		1.1		1.1		2.8	5.0
Ultic		1.0				1.6	2.6
Total	21.0	18.0	9.3	14.5	3.7	51.8	118.3

Appendix II - Results of 2018-19 soil quality monitoring

SiteNo.	Soil Order	Land System	Land use	pH	Total C (%)	Total N (%)	Olsen P ($\mu g/cm^3$)	AMN (mg/kg)	Bulk density (t/m^3)	Macropores @ -10 kPa
5	Allophanic	Pasture	Dairy farm	5.66	8.32	0.85	45	146	0.81	3.5
6	Allophanic	Pasture	Dairy farm	5.51	8.58	0.94	17	157	0.74	5.9
21	**Organic**	**Pasture**	**Dairy pasture**	6.52	47.3	2.50	13	287	0.33	8.9
22	**Organic**	**Native**	**Indigenous forest**	4.09	52.8	1.83	2	233	0.11	51.4
26	Pumice	Pasture	Drystock/ Dairy	6.14	5.9	0.44	65	114	0.64	23.2
52	Pumice	Native	Indigenous forest	5.75	12.5	1.00	7	286	0.31	40.4
63	Gley	Pasture	Dairy	5.60	7.76	0.65	21	170	0.65	12.1
64	Gley	Pasture	Dairy	5.60	7.39	0.62	48	158	0.80	7.0
65	Granular	Arable	Cropping	5.79	2.31	0.23	116	16	1.09	22.5
66	Granular	Arable	Cropping	6.65	2.43	0.25	155	19	1.05	21.6
67	Granular	Arable	pasture	5.87	3.85	0.35	41	80	1.22	2.3
69	Granular	Pasture	Drystock Beef	6.35	8.44	0.81	142	206	0.88	4.7
70	Gley	Arable	Maize	6.34	5.55	0.48	55	62	1.00	7.4
73	Granular	Native	Bush	5.62	9.34	0.58	3	173	0.82	14.6
82	Allophanic	Horticulture	Orchard	6.11	6.55	0.58	12	60	0.91	6.9
84	Gley	Native	Urban Reserve	4.21	13.26	0.78	79	105	0.52	31.6
85	Granular	Arable	Cropping	5.87	3.38	0.29	101	37	1.04	19.3
86	Granular	Arable	Cropping	5.78	2.85	0.28	51	26	1.05	21.6
89	**Organic**	**Pasture**	**Dairy**	5.86	47.6	2.52	44	273	0.36	7.3
98	Brown	Pasture	Dairy	5.85	4.38	0.44	69	103	1.14	5.5
99	Gley	Pasture	Dairy	6.21	2.80	0.30	16	68	1.25	2.8
125	Pumice	Pasture	Deer cut & carry	5.89	6.68	0.60	128	198	0.83	9.9
131	Allophanic	Forestry	Forestry	5.42	14.1	1.03	17	144	0.48	28.8
132	Allophanic	Pasture	Drystock	5.36	15.0	1.40	17	243	0.62	9.7
133	Allophanic	Pasture	drystock beef-Bulls	5.95	9.34	0.98	7	174	0.77	3.4
134	Allophanic	Arable	Maize/Tama	5.76	6.32	0.67	29	89	0.87	4.4

135	Granular	Pasture	Drystock beef	6.09	8.28	0.83	22	180	0.83	7.1
137	Brown	Pasture	Dairy	5.58	6.33	0.65	41	177	0.89	6.4
152	Allophanic	Forestry	Production forestry exotic	5.42	18.7	0.91	2	153	0.44	33.1
153	Brown	Forestry	Production forestry exotic	4.11	9.42	0.62	10	51	0.81	12.6
154	Recent	Forestry	Production forestry exotic	5.79	3.67	0.22	11	47	1.15	28.6
155	Allophanic	Forestry	Production forestry exotic	5.35	11.6	0.70	1	127	0.54	34.3
158	Allophanic	Native		4.30	38.39	1.96	6	285	0.32	37.5
159	Allophanic	Native		5.27	15.80	0.72	1	177	0.40	39.3
160	Brown	Native		4.44	12.26	0.62	1	115	0.29	36.8

Appendix III - Results for all 154 soil quality monitoring sites

SiteNo.	Soil Order	Land use type	Land use	pH	Total C (%)	Total N (%)	Olsen P (mg/L)	AMN (mg/kg)	Bulk density t/m^{-3}	%Macropores @-10 kPa
36	Allophanic	Arable	Cropping	6.2	5.2	0.54	100	60	0.95	11.3
37	Allophanic	Arable	Cropping	6.22	5.88	0.59	30	72	0.73	6.0
40	Allophanic	Arable	Maize cropping	5.64	6.52	0.69	36	78	0.71	17.4
61	Granular	Arable	Cropping	6.23	2.68	0.24	76	17.9	1.15	15.9
62	Granular	Arable	Cropping	6.53	2.28	0.19	279	18.9	1.23	11.8
65	Granular	Arable	Cropping	5.79	2.31	0.23	116	16	1.09	22.5
66	Granular	Arable	Cropping	6.65	2.43	0.25	155	19	1.05	21.6
68	Allophanic	Arable	Cropping	6.94	6.54	0.57	84	38	0.79	16.5
70	Gley	Arable	Maize	6.34	5.55	0.48	29	89	1.00	7.4
71	Allophanic	Arable	Cropping	7.16	5.49	0.48	134	37	0.84	12.3
85	Granular	Arable	Cropping	5.87	3.38	0.29	101	37	1.04	19.3
86	Granular	Arable	Cropping	5.78	2.85	0.28	51	26	1.05	21.6
93	Allophanic	Arable	Maize	5.83	4.65	0.50	53	68	0.92	10.1
93	Allophanic	Arable	Maize	5.56	4.56	0.50	34	52	0.79	24.4
94	Allophanic	Arable	Cropping	6.24	5.18	0.53	81	53	0.86	6.2
134	Allophanic	Arable	Maize/Tama	5.76	6.32	0.67	55	105	0.87	4.4
3	Allophanic	Forestry	Woodlot	5.34	9.08	0.62	19	174	0.75	10.7
10	Pumice	Forestry	Plantation forestry	5.50	5.10	0.33	4	79	0.60	31.8
19	Brown	Forestry	Plantation forestry	6.33	6.17	0.48	14	195	0.87	4.6
23	Brown	Forestry	Plantation forest	5.98	4.99	0.28	4	110	0.95	6.6
34	Allophanic	Forestry	Plantation forest	5.62	18.16	1.39	6	268	0.54	6.8
42	Brown	Forestry	Pine	4.71	2.59	0.16	4	32	1.07	9.9
43	Brown	Forestry	Pinus radiata (~8 yrs)	4.80	4.60	0.36	13	76	0.89	17.6
48	Ultic	Forestry	Forestry	4.70	4.13	0.22	7	70	1.06	15.1

57	Podzol	Forestry	Forestry	5.29	5.83	0.31	3	84.0	0.63	31.8
114	Pumice	Forestry	Forestry	4.69	9.76	0.63	49	89.0	0.61	33.4
115	Pumice	Forestry	Forestry	5.09	10.70	0.55	11	144.0	0.41	39.5
118	Pumice	Forestry	Forestry	5.26	6.73	0.29	10	59.0	0.67	27.5
120	Pumice	Forestry	Forestry	5.79	7.11	0.43	25	221.0	0.80	7.3
131	Allophanic	Forestry	Forestry	5.42	14.1	1.03	17	144	0.48	28.8
144	Allophanic	Forestry	Woodlot	4.98	7.07	0.49	36	100	0.84	30.0
152	Allophanic	Forestry	Production forestry exotic	5.42	18.7	0.91	2	153	0.44	33.1
153	Brown	Forestry	Production forestry exotic	4.11	9.42	0.62	10	51	0.81	12.6
154	Recent	Forestry	Production forestry exotic	5.79	3.67	0.22	11	47	1.15	28.6
155	Allophanic	Forestry	Production forestry exotic	5.35	11.6	0.70	1	127	0.54	34.3
60	Allophanic	Horticulture	Orchard	6.13	5.83	0.57	16	150.0	0.85	7.1
80	Allophanic	Horticulture	Orchard	7.12	8.38	0.82	54	139.0	0.68	7.8
81	Allophanic	Horticulture	Orchard	6.68	6.32	0.66	22	137.0	0.72	21.6
82	Allophanic	Horticulture	Orchard to become maize	6.11	6.55	0.58	12	60	0.83	8.5
138	Granular	Horticulture	Kiwifruit Organic	6.55	7.82	0.70	92	398	0.87	6.2
140	Allophanic	Horticulture	Kiwifruit Organic	6.53	11.27	1.04	28	361	0.71	9.1
146	Allophanic	Horticulture	kiwifruit	6.77	9.22	0.90	75	147	0.76	10.8
147	Allophanic	Horticulture	kiwifruit	6.94	7.32	0.70	69	171	0.76	13.2
148	Allophanic	Horticulture	kiwifruit	6.74	8.09	0.83	63	190	0.77	11.2
149	Allophanic	Horticulture	kiwifruit	6.90	6.72	0.64	135	150	0.90	13.7
150	Allophanic	Horticulture	kiwifruit	6.59	9.69	0.93	37	208	0.70	8.9
151	Allophanic	Horticulture	kiwifruit	6.37	7.70	0.77	115	138	0.81	8.7
22	Organic	Native	Indigenous forest	4.09	52.8	1.83	2	261	0.11	51.4
25	Brown	Native	Indigenous forest	5.2	7.3	0.38	2	120	0.82	16.3
28	Podzol	Native	Reserve	4.4	18.2	0.88	11	196	0.57	25.3
38	Allophanic	Native	Indigenous forest	5.45	18.3	1.06	1	297	0.37	28.1
41	Brown	Native	Indigenous forest	5.28	9.92	0.60	1	160	0.65	25.3
44	Recent	Native	Indigenous forest	5.15	5.57	0.37	3	129	0.85	17.6

52	Pumice	Native	Indigenous forest	5.75	12.5	1.00	7	226	0.31	40.4
73	Granular	Native	Bush	5.62	9.34	0.58	3	165	0.82	14.6
84	Gley	Native	Urban Reserve	4.21	13.26	0.78	79	131.0	0.52	31.6
112	Allophanic	Native	Indigenous forest	5.25	8.80	0.53	1	154	0.63	15.7
136	Gley	Native	Native bush	6.30	4.68	0.38	5	89.8	0.94	9.0
145	Allophanic	Native	Senic reserve	5.6	23.4	1.3	2	416	0.27	24.0
158	allophanic	Native		4.30	38.39	1.96	6		0.32	37.5
159	Allophanic	Native		5.27	15.80	0.72	1		0.40	39.3
160	Brown	Native		4.44	12.26	0.62	1		0.29	36.8
9	Pumice	Pasture	conversion Dairy	6.07	6.34	0.45	37	160	0.64	19.5
12	Pumice	Pasture	conversion Dairy	5.59	6.12	0.42	33	111.0	0.64	25.3
13	Pumice	Pasture	conversion Dairy	5.95	7.95	0.58	47	240	0.67	14.4
26	Pumice	Pasture	conversion Drystock/dairy	6.14	5.90	0.44	65	114	0.64	23.2
27	Pumice	Pasture	conversion Dairy	5.19	6.17	0.54	125	156	0.72	22.6
56	Podzol	Pasture	conversion Dairy	6.00	5.76	0.40	27	114	0.85	4.8
141	Pumice	Pasture	conversion Dairy	5.38	7.22	0.54	100	160.0	0.62	29.8
142	Pumice	Pasture	conversion Dairy	5.18	8.64	0.54	127	147.0	0.83	16.1
143	Pumice	Pasture	conversion Dairy	5.04	9.47	0.58	63	169.0	0.74	17.2
5	Allophanic	Pasture	Dairy	5.66	8.32	0.85	45	146	0.81	3.5
6	Allophanic	Pasture	Dairy	5.51	8.58	0.94	17	157	0.74	5.9
21	Organic	Pasture	Dairy	6.52	47.3	2.50	13	287	0.33	8.9
63	Gley	Pasture	Dairy	5.60	7.76	0.65	21	170	0.65	12.1
64	Gley	Pasture	Dairy	5.60	7.39	0.62	48	158	0.80	7.0
89				5.86	47.6	2.52	44	273	0.36	7.3
98	Brown	Pasture	Dairy	5.85	4.38	0.44	69	103	1.14	5.5
99	Brown	Pasture	Dairy maize/pasture	6.21	2.80	0.30	16	68	1.25	1.3
137	Gley	Pasture	Dairy maize/pasture	5.58	6.33	0.65	41	177	0.89	6.4
1	Allophanic	Pasture	Dairy	6.32	10.43	1.07	42	391	0.73	5.6
2	Allophanic	Pasture	Dairy	6.11	8.80	0.87	98	349	0.77	4.8

11	Pumice	Pasture	Dairy	5.57	9.54	0.91	22	214	0.70	7.9
14	Pumice	Pasture	Dairy	5.81	7.18	0.70	80	271	0.71	3.4
15	Pumice	Pasture	Dairy	5.81	7.45	0.63	86	201.0	0.70	15.8
16	Pumice	Pasture	Dairy	5.91	7.58	0.64	63	197	0.80	10.0
17	Pumice	Pasture	Dairy	5.78	8.56	0.75	126	289	0.83	5.5
20	Gley	Pasture	Dairy	5.9	6.4	0.66	65	207	1.08	26.2
30	Podzol	Pasture	Dairy	6.12	6.82	0.53	39	126.2	0.77	3.6
31	Allophanic	Pasture	Dairy	6.7	13.0	1.4	38	340	0.67	42.6
32	Recent	Pasture	Dairy	6.17	6.88	0.70	89	351	0.81	5.2
33	Recent	Pasture	Dairy	5.62	7.33	0.74	62	322	0.81	6.6
35	Allophanic	Pasture	Dairy	5.60	18.18	1.56	8	475	0.58	16.9
46	Gley	Pasture	Dairy	6.01	4.27	0.43	75	144	0.82	7.8
47	Gley	Pasture	Dairy	5.91	6.28	0.62	76	213	1.01	5.8
50	Granular	Pasture	Dairy	5.87	6.89	0.64	62	167	0.95	7.0
74	Organic	Pasture	Dairy	5.41	24.10	1.14	35	140.0	0.54	2.3
75	Gley	Pasture	Dairy	6.14	6.47	0.59	26	83.0	0.91	6.1
76	Gley	Pasture	Dairy	5.99	9.94	0.86	22	129.0	0.77	4.1
77	Allophanic	Pasture	Dairy	6.03	7.34	0.73	43	132	0.86	3.9
78	Organic	Pasture	Dairy	5.75	5.13	0.51	22	38.0	0.74	24.9
91	Allophanic	Pasture	Dairy	6.56	5.83	0.57	46	112	0.99	7.1
92	Allophanic	Pasture	Dairy	6.2	9.2	0.87	20	169	0.73	9.6
95	Allophanic	Pasture	Dairy	6.5	6.4	0.57	70	201	0.94	9.2
96	Gley	Pasture	Dairy	5.6	5.4	0.60	25	90	0.81	14.3
97	Gley	Pasture	Dairy	6.3	3.9	0.41	81	131	1.04	10.3
100	Gley	Pasture	Dairy	6.0	9.0	0.70	41	180	0.91	4.6
106	Gley	Pasture	Dairy	5.76	6.18	0.68	38	137	0.86	3.1
107	Brown	Pasture	Dairy	6.2	4.3	0.41	96	120	1.03	8.8
108	Brown	Pasture	Dairy	6.5	9.8	0.92	134	207	0.86	7.7
109	Brown	Pasture	Dairy	6.2	9.1	0.79	69	170	0.80	6.1

113	Pumice	Pasture	Dairy	5.58	7.03	0.64	41	155	0.82	6.2
116	Pumice	Pasture	Dairy	6.41	9.05	0.76	26	151	0.77	8.4
117	Pumice	Pasture	Dairy	6.40	9.61	0.88	94	197	0.66	20.6
119	Pumice	Pasture	Dairy	5.66	7.99	0.67	51	170	0.80	7.3
121	Allophanic	Pasture	Dairy	5.94	12.60	1.12	13	221	0.70	5.1
122	Recent	Pasture	Dairy	5.78	6.33	0.60	24	183	0.85	3.7
123	Gley	Pasture	Dairy	5.86	10.60	0.91	29	196	0.80	5.8
130	Gley	Pasture	Dairy	6.06	3.62	0.36	70	84.5	1.18	0.5
156	Ultic	Pasture	Dairy	6.50	3.97	0.37	79	86	1.30	4.4
157	Gley	Pasture	Dairy	5.20	9.63	0.84	12	179	0.69	11.8
18	Brown	Pasture	Dry Stock Dairy runoff	6.05	8.01	0.72	10	318	0.81	5.5
69	Granular	Pasture	Dry stock Beef	6.35	8.44	0.81	142	206	0.88	0.9
125	Pumice	Pasture	Deer Farm	5.89	6.68	0.60	128	198	0.83	9.9
132	Allophanic	Pasture	Dry stock	5.36	15.0	1.40	17	243	0.62	9.7
133	Allophanic	Pasture	Dry stock Beef	5.95	9.34	0.98	7	174	0.77	3.4
135	Granular	Pasture	Dry stock Beef	6.09	8.28	0.83	22	180	0.83	7.1
67	Granular	Pasture	pasture with Maize crop	5.87	3.85	0.35	41	80	1.22	2.3
24	Brown	Pasture	Dry stock	5.6	8.4	0.66	11	231	0.76	27.3
29	Podzol	Pasture	Dry stock Beef	6.9	7.9	0.51	31	170	0.51	21.7
39	Allophanic	Pasture	Dry stock	5.68	15.8	1.23	4	243	0.56	3.3
45	Recent	Pasture	Dry stock	5.04	4.60	0.40	23	152	1.04	9.5
49	Ultic	Pasture	Dry stock (beef)	6.08	4.83	0.45	21	152	1.09	6.8
51	Granular	Pasture	DrystockBeef & sheep	5.95	7.72	0.69	16	221	0.82	8.2
53	Gley	Pasture	Cut & Carry grass and chicory	6.88	9.57	0.91	21	240	0.74	13.9
54	Allophanic	Pasture	Cut & Carry grass and chicory	6.80	10.9	1.08	29	232	0.70	10.6
55	Brown	Pasture	Dry stock	6.00	5.81	0.54	19	167	1.00	7.5
72	Brown	Pasture	Dry stock	5.8	11.1	1.06	37	293	0.83	3.3

78	Allophanic	Pasture	Drystock intensive calf raising	6.11	4.94	0.47	26	98.2	0.94	9.0
79	Gley	Pasture	Dry stock	6.57	2.73	0.29	28	52.0	0.89	22.4
88	Allophanic	Pasture	Drystock Sheep and beef	4.85	9.81	0.97	172	201	0.75	25.5
90	Allophanic	Pasture	Drystock	5.9	8.2	0.84	6	181	0.73	22.3
101	Granular	Pasture	sheep/lightly forested	5.3	11.8	1.03	17	239	0.82	3.1
102	Granular	Pasture	Drystock sheep	6.3	14.7	1.4	13	362	0.68	10.2
103	Allophanic	Pasture	drystock - techno beef	5.5	7.9	0.71	81	213	0.99	4.3
104	Granular	Pasture	drystock - techno beef	5.5	8.2	0.72	60	264	0.92	4.0
105	Brown	Pasture	drystock beef	5.91	9.17	0.88	21	217	0.74	6.4
110	Allophanic	Pasture	Dry stock	6.17	10.6	0.99	20	215	0.76	4.1
111	Allophanic	Pasture	Dry stock	5.78	10.7	0.90	5	220	0.70	7.6
124	Pumice	Pasture	Deer Farm	5.36	8.24	0.68	31	148	0.70	27.3
126	Allophanic	Pasture	Deer Farm	5.96	11.5	1.13	43	207	0.69	8.9
127	Allophanic	Pasture	Deer Farm	5.49	12.60	1.24	21	257	0.59	8.2
128	Pumice	Pasture	Deer Farm	5.50	10.7	0.87	13	200	0.63	11.9
139	Allophanic	Pasture	Drystock sheep	5.77	9.86	1.01	20	337	0.71	5.2