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Long-term monitoring of soil quality and trace elements to evaluate land use effects and temporal change in the Wellington region, New Zealand

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A B S T R A C T

Soil quality monitoring is used to assess the soil's ability to maintain agricultural productivity, ecological, and environmental quality. Very few soil quality monitoring studies have reported on multiple samplings over the long-term. Several regional authorities in New Zealand have monitored soil quality since the late 1990s. In the Wellington region, dairy, mixed cropping, market garden, drystock (sheep/beef), horticulture, exotic forestry and indigenous land use systems, and four soil orders have been monitored over 19 years, with up to five repeat samplings per site. This study reports on key soil quality indicators and Cu, Zn, and Cd concentrations. For the most recent sampling per land use, all land use system sites, except drystock, had Zn concentrations below recommended ecological toxicity guidelines. Dairy land use had 21%, and 36% of sites within recommended soil quality target ranges, for Olsen P, and macroporosity, respectively. Compared with indigenous land, across all samplings, Cu concentrations were elevated in horticultural and market gardens sites, while several land uses had lower total nitrogen and higher Olsen P concentrations. Across all samplings, significant increases over time were observed in Zn for dairy, total nitrogen for drystock, and Olsen P for mixed cropping. Significant decreases over time were observed for Cu in forestry, Cd for indigenous and forestry, and bulk density for drystock. No changes over time were detected for macroporosity, anaerobically mineralised nitrogen, or organic carbon. This study shows the programme and our analysis of multiple samplings are valuable for detecting significant trends as an early warning, e.g. Zn and Olsen P changes. The study provides evidence for recommending additional sites for several land uses and increased sampling frequency to ensure future robust statistical analysis. This study included only sites where land use systems did not change, providing a robust basis for detecting change over time, for informing policy, resource and environmental decision-making.

1. Introduction

Widespread and increasing concern over the state of the environment and the impacts of human activities on ecosystem services and functions, highlights the essential need for high-quality, long-term datasets in order to detect environmental change and understand the effects of multiple pressures (Parr et al., 2003; Condron et al., 2014; Johnston et al., 2017). Soil quality is complex with a range of definitions (Bünemann et al., 2018). Soil quality is used to assess the soil's ability to maintain agricultural productivity, and ecological and environmental quality. The definitions of soil quality and soil health have been discussed in the literature with some maintaining that there is a difference, while others use the terms synonymously (e.g. Bünemann et al., 2018; Kibblewhite, 2018). To maintain consistency with early publications (e.g. Sparling et al., 2004), the term soil quality is used for this study, and throughout the paper. Soil quality monitoring programmes are used by regulatory agencies to monitor the impact of land management changes on soil condition to help protect or improve the soil resource (Sparling et al., 2004; McBratney et al., 2014). Soil quality monitoring has led to inclusion of trace elements in policy, such as regional policy statements to regulate environmental management, for example, to limit accumulation of Cd, F, and Zn in soil (Waikato Regional Council, 2016).

While soil quality monitoring programmes exist at regional, state, national, and international scales (Morvan et al., 2008; Saby et al., 2009; Nerger et al., 2016), most have only been undertaken on a one-off basis (Cotching and Kidd, 2010; Arrouays et al., 2012). In other

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cases, considerable effort has gone into the planning of programmes, but they have not been implemented (Huber et al., 2008; Kibblewhite et al., 2010; Bünemann et al., 2018).

Where programmes have re-sampled sites over time (e.g. Cathcart et al., 2008), these are for a single, or in some cases, several land uses or climatic environments. For example, soil quality monitoring in Alberta, Canada, regularly re-sampled the sites of cultivated cropping-land (Cathcart et al., 2008; Caron et al., 2010), while other programmes have re-sampled specific sites over time on multiple land uses (Taylor et al., 2010; Curran-Cournane, 2015; Nerger et al., 2016). Soil monitoring in Bavaria, Germany was limited to grassland sites, but was sampled regularly (Kühnel et al., 2019). The British Countryside Survey, for example, re-sampled sites three times, at approximately 10 year intervals, but the programme ceased in 2007 and had a limited range of four soil indicators (Emmett et al., 2010; Reynolds et al., 2013).

Trace elements can accumulate over time from use of pesticides, animal remedies, veterinary medicines, fertiliser, and manure, but few programmes regularly monitor a range of trace elements. The French National Soil Quality Monitoring Network (Saby et al., 2009), for example, covers the whole of France using 16-km grid sampling, and monitors trace element contamination, but has only been sampled once (Arrouays et al., 2012). In New Zealand, soil quality monitoring by regional authorities typically includes seven or more trace elements, and concentrations above maximum recommended soil guidelines have been recorded (Taylor et al., 2010; Cavanagh, 2014).

This study examines temporal change in soil quality including key trace elements, over the 19-year period (2000–2018 inclusive) of the monitoring programme from the Wellington region of New Zealand. This regional soil quality monitoring programme was developed from an earlier national study across 10 regions (Sparling et al., 2004). The soil quality monitoring programme is intended to serve two main purposes – to assess the quality of soils at sites in relation to land use during a particular sampling period, and to assess any changes over time to inform environmental policy decision making. However, no assessment of temporal changes has been made over the last 19 years, where re-sampling has occurred at intervals of 3 to 10 years. We assessed changes over time for the key soil properties: cadmium, copper, zinc, bulk density, macroporosity, anaerobically mineralised nitrogen, total nitrogen, phosphorus, and organic carbon. We also discuss how results and learnings from this analysis from regular re-sampling on multiple land uses can inform resource management decision-makers and inform future improvements to monitoring programmes.

2. Methods

2.1. Sites, land use and soils in the programme

The Wellington region's monitoring programme commenced in 2000, with sampling across a range of land uses and soil orders. The original monitoring study design had a focus on soil and land use combinations perceived to be a risk to the environment from land use intensification (Sparling and Schipper, 2002; Sparling et al., 2004). The distribution of land use in the region shows monitoring sites tend to be spatially concentrated in areas with intensive land uses on low relief areas within river valley or basins, compared to less intensive land uses in hill country areas (Fig. 1). Land uses monitored in the programme which are considered to be intensive are dairy, mixed cropping, and market gardens (Table 1). Land uses considered less intensive are also monitored: drystock (i.e. sheep/beef, non-dairy), horticulture, and exotic forestry and indigenous vegetation (Table 1). The predominant soil orders monitored (New Zealand classification system (Hewitt, 2010), followed by FAO (Krasilnikov et al., 2009; IUSS Working Group WRB, 2015)), are Brown Soils, (Cambisols/Fragic Cambisols), Gley Soils (Gleyic Fluvisols/Gleysols), Pallic Soils (Fragic Planosols/Luvic Planosols), and Recent Soils (Fluvisols).

Sampling frequency was 3-yearly for dairy, market garden and mixed-cropping, 7-yearly for exotic forestry, horticulture, and drystock, and 10-yearly for indigenous vegetation with sampling occurring in southern hemisphere autumn months of April and May. Within the same land use/farm system, the vegetative cover may be different at the time of sampling; for example, a drystock sampling site may have been under grazed pasture at one sampling and under fodder crop another sampling time. In this paper, we grouped sites based on the land use system, rather than the 'vegetation cover' at the time of sampling.

Details of field methods are reported in Hill and Sparling (2009). Briefly, at each site a 50-m transect was used to take 10-cm-depth soil cores, taken approximately every 2 m. Individual cores were bulked and mixed to obtain a representative sample for chemical and trace element analyses. The 10-cm depth was chosen as reported by Sparling et al. (2004); that study implemented that depth due to standard equipment being available, the depth was the same as required for New Zealand IPCC surface carbon sampling, and similar to the soil fertility depth to 7.5 cm on New Zealand farms. For the soil physical analyses described below, three undisturbed (intact) soil samples were collected along the field transect per site by pressing steel liners (10 cm diameter, 7.5 cm depth) into the soil, and excavating them. Resampling of sites involved visiting the same transect. In some cases, revisiting the same site transect was not possible every time (e.g. due to inaccurate record, accuracy of GPS technology, changes within paddock), in which case the resampling was undertaken at a position nearby in the same paddock, because the paddock is the management unit where farm management is implemented.

2.2. Soil properties

The programme monitors 11 trace elements (As, Cd, Cr, Cu, Pb, Ni, Zn since 2000; Fe, Mn, U, F since 2013), and the soil quality indicators pH, bulk density, macroporosity, anaerobically mineralised nitrogen, total nitrogen, Olsen phosphorus, organic carbon, aggregate stability (tilled sites), and, since 2012, hot water-extractable C and N. We have focussed on a subset of the key soil properties, specifically cadmium (Cd), copper (Cu), zinc (Zn), bulk density, macroporosity, anaerobically mineralised nitrogen (AMN), total nitrogen (TN), Olsen phosphorus, and organic carbon (OC). This is because Cd, Cu, and Zn are commonly elevated in agricultural soils due to inputs from fertilisers, fungicides and animal remedies respectively. Macroporosity is the percentage of large soil pores responsible for soil drainage and aeration, and has been identified as an indicator of soil compaction (Drewry et al., 2004a; Rab et al., 2014). Olsen P is an indicator of plant-available P, OC is an estimate of organic matter content, and TN has been shown to be a useful surrogate measure of soil N supply (Shepherd et al., 2015). AMN estimates how much organic N is potentially mineralisable and can be considered biologically active, and is also used as a surrogate measure of the microbial biomass (Hart et al., 1986; Sparling et al., 2004; Hill and Sparling, 2009).

Soil chemical analyses, except for trace elements, were carried out by the Manaaki Whenua – Landcare Research environmental chemistry laboratory (Manaaki Whenua – Landcare Research, 2019). The composite soil samples for chemical analyses were well mixed, air-dried and sieved (<2 mm). Total N and OC were determined by dry combustion of air-dry, finely ground soil, AMN by anaerobic 7-day incubation at 40 °C, and Olsen P was measured and reported gravimetrically (Blakemore et al., 1987; Manaaki Whenua – Landcare Research, 2019). Total recoverable trace elements were analysed by inductively coupled-mass spectrometry (ICP-MS) following concentrated nitric/hydrochloric acid digestion, by Hill Laboratories, Hamilton. From the three undisturbed (intact) soil samples per site, a 3-cm deep subsample ring was taken to determine bulk density, and macroporosity (pores >30 μm) measured at −10 kPa matric potential on ceramic tension plates, with remaining soil for particle density used in the macroporosity calculation (Gradwell, 1972; Hill and Sparling, 2009) by the Manaaki Whenua – Landcare Research soil physics laboratory in Hamilton.

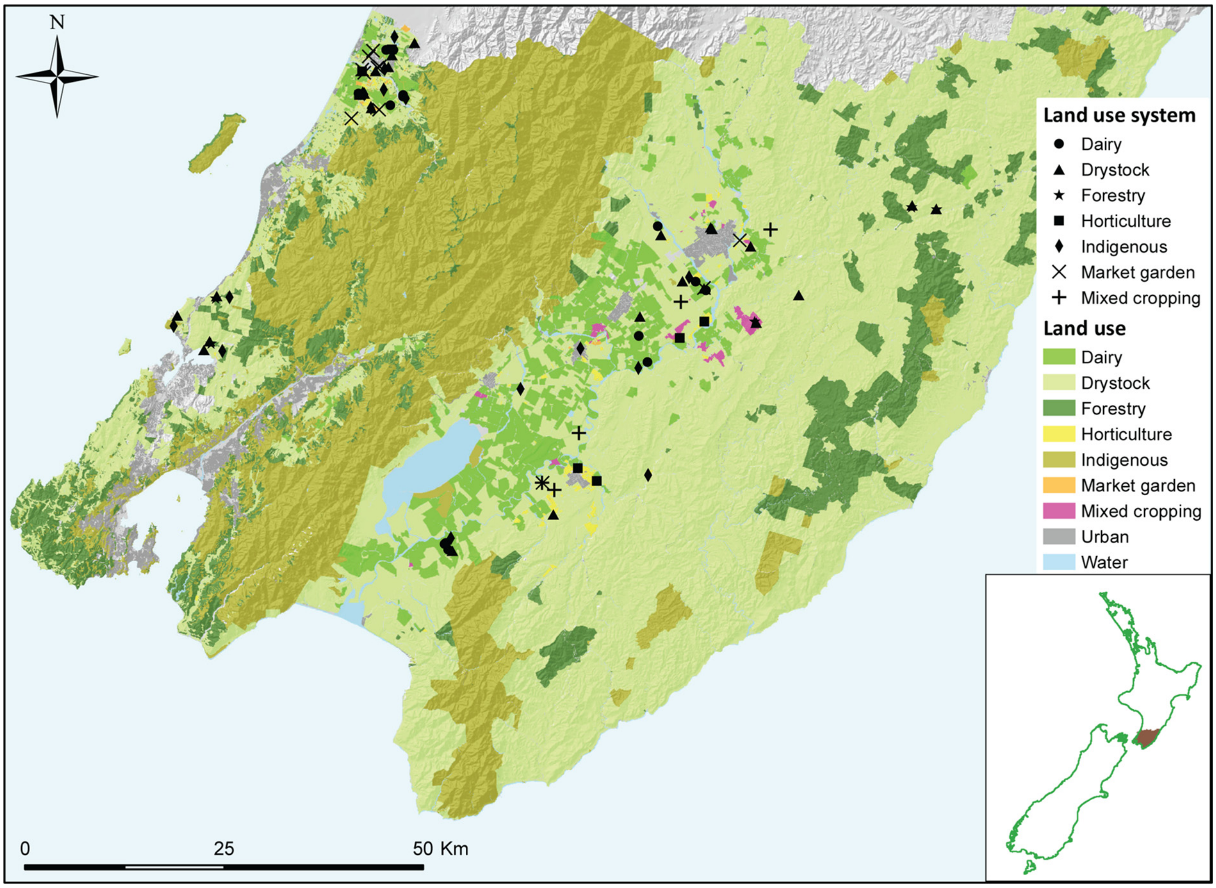

Fig. 1. Wellington region land use map and distribution of soil quality monitoring sites.

Table 1
Land use system classifications used in the analysis.

Land use system	Description
Mixed cropping	Extensive arable cropping (e.g. cereals, maize, grass seed)
Dairy	Milking platform (effective area including pasture and crop for grazing lactating cows)
Drystock	Sheep, beef, deer, dairy runoff (for non-lactating cows or young stock)
Forestry	Exotic forestry (e.g. *Pinus radiata* plantation)
Horticulture	Orchards, vineyards
Market garden	Intensive vegetables (for human consumption)
Indigenous	Indigenous forest and vegetation

2.3. Environmental guidelines, target ranges, and land use map

To provide a perspective on the quality of the soil, results from the most recent sampling were compared with relevant target ranges or guideline values. Target ranges for nitrogen, phosphorus, carbon and physical properties have been developed to balance productivity requirements with environmental concerns for monitoring the 0–10 cm soil depth, and sourced from Hill and Sparling (2009), and Mackay et al. (2013). For trace elements, soil toxicity guideline values for the protection of ecological receptors (Cavanagh, 2019) were also used. For cadmium trigger values within the Tiered Fertiliser Management System, TFMS (http://www.fertiliser.org.nz/Site/resources/tools.aspx), which is used to manage accumulation of cadmium in productive soils by imposing increasingly stringent fertiliser applications with increasing soil cadmium, were used.

A regional land use map was constructed by Greater Wellington Regional Council (GWRC) using GIS, council records, and LUCAS (Ministry for the Environment, 2019) data to provided context to soil quality monitoring results used in this study.

2.4. Data preparation and analysis

Data were available from 2000 to 2018 inclusive (19 years) for 118 sites. Up to five repeat-samplings per site were available. There were too few sites to determine temporal trends for Melanic and Allophanic Soils, so these, and all sites with only one sampling date were removed from the analysis. Further, only sites with the same persistent land use or 'land-use system' for all samplings in the period were used (Table 1). Thus, 23 sites were removed due to changes in the land use system (e.g. from dairy to cropping). After removal of these sites, 281 rows of data for 82 unique sites, across four soil orders (Table 2), were available for different soil properties.

2.4.1. Statistical analysis

When measuring soil environmental properties over time, several factors determine the value and its change over time. First, an effect due to the characteristics of the soil sample such as the sample depth, soil order, or land use, as well as a possible effect due to the year of the measurement. Second, a random effect that expresses the variability of the soil property between different sites with the same soil characteristics (e.g. the same soil order and land use). The random effect captures the fact that different sites may have different levels of soil property and the change over time is expected to be small around the average for the site. A component of uncertainty is expected, due to measurement error or unrecorded variables.

Table 2
Number of unique sites and temporal samples, per land use system, for each soil order.

Unique sites			Soil order		
Land use system	Brown	Gley	Pallic	Recent	Total
Dairy	3	2	6	3	14
Drystock	6	1	10	7	24
Forestry	3	0	0	2	5
Horticulture	3	2	3	0	8
Indigenous	6	1	3	5	15
Market garden	1	2	2	6	11
Mixed cropping	0	2	2	1	5
Total	22	10	26	24	82
Temporal samples					
Land use system	Brown	Gley	Pallic	Recent	Total
Dairy	15	10	30	15	70
Drystock	18	3	29	21	71
Forestry	9	0	0	6	15
Horticulture	9	6	9	0	24
Indigenous	12	2	6	10	30
Market garden	5	10	7	24	46
Mixed cropping	0	10	10	5	25
Total	68	41	91	81	281

All the observed soil properties have a lower limit of zero, and their probability distributions tend to be skewed, with a variance that increases with the mean. For these reasons, a logarithmic transformation was applied to each soil property value to stabilise the variance, in accordance with standard practice (Faraway, 2006). A square-root transformation was found to be more suitable for macroporosity. The model for each soil property (using bulk density, ρ, as an example) was:

$$\log \rho_{i,j,k} = \overline{\log \rho} + L_i + S_j + \text{Site}_k + \Delta_i t + \epsilon$$

where $\rho_{i,\,j,\,k}$ is the bulk density for land use i, soil order j, and site k, $\overline{\log\,\rho}$ is the overall mean value of the log-transformed bulk density for some reference land use and soil order, L_i is the change in log-transformed bulk density as a result of changing from the reference land use to land use class i, S_j is the change in log-transformed bulk density as a result of changing from the reference soil order to soil order j, while ϵ is the uncertainty or error, assumed to be Gaussian distributed.

The value of Δ_i is the rate at which the log transformed bulk density changes for each year, specific to each land use class i.

Finally, the term Site_k is the random effect associated with site k.

No interaction was included in the model between land use and soil order, since not all soil orders were sampled for each land use, although the available soil orders were approximately balanced across each land use class. The temporal change was limited to within each land use class, since each property was expected to change with respect to that variable and not soil order. The complexity of the model was thus limited by the available data.

Using the model above, the important result from the analysis is to determine which of the Δ_i is statistically significant, since this would indicate that the corresponding soil property in that land use has a statistically significant temporal effect.

The resultant linear mixed-effects model (Pinheiro and Bates, 2000) was solved using maximum likelihood. Although described in terms of bulk density, the model can be extended to any of the properties. For all properties except macroporosity, the log-transformed version of the property was used as the response. For macroporosity, the square root is used. For land use, indigenous land use was chosen as the reference since it can be considered a control, as management is not normally applied to that class. For soil order, the choice is less obvious, so Brown Soils were chosen. The model was kept the same for all properties. Finally, although the model is presented in terms of the log (or square root) transformed response, the estimates of soil properties were back-transformed to standard units. The predictions include the estimation of a 95% prediction interval, which is the range of values into which a new observation will fall, with a certain probability (Knowles and Frederick, 2019). Finally, we note that the analysis covers the change in soil properties over 2000–2018, so predictions of soil properties before and after this period should be treated with some caution. Predictions would be compromised if management practices changed from those used in the analysis period, and predictions will have increased uncertainties outside 2000–2018.

2.4.2. Statistical model fit and outliers

Each model was checked for conformance to the assumptions of the linear mixed effects model, such as residual-versus-fitted plots and checks that residuals and random effects are Gaussian distributed. Failures in any such checks would suggest problems with the form of the model. None of the diagnostic checks suggested problems in the form of the model.

To examine the model fit per indicator, Table 3 shows R^2 values for each of the models, defined as in Nakagawa and Schielzeth (2013). While R^2 values are well-understood for linear models, there is less agreement in the literature for application for the mixed effects models used here. The definition from Nakagawa and Schielzeth (2013) defines two R^2 values: one (the marginal R^2) gives the proportion of the variance explained by the fixed effects alone; the second (the conditional R^2) gives the proportion of the variance explained by both the fixed and random effects. Table. Shows that the variance explained using the fixed effects model only is at best modest (the largest R^2 value is 0.57), but the variance explained by fixed and random effects is much higher (the largest value is 0.91). This suggests that adding the random effect due to the site is very important in providing a good model fit.

For Cd, 27 values were below the detection limit of the laboratory method and these values were replaced by half of the minimum laboratory detection limit for the statistical analysis. If these values were discarded, then the conclusions for the temporal trends are unchanged but the fixed effects for dairy, horticulture and market garden are significantly different from indigenous land. This is because the greatest proportion of the replaced Cd values are in the indigenous and forestry systems, with 30% and 50% in each, respectively.

For the Cu model, two horticulture site concentrations were highly influential, with estimated mean changes over time of 3.12 and 1.03 mg kg^{-1} $year^{-1}$ after two re-samplings, compared with the mean rate of change in this land use class of 0.13 mg kg^{-1} $year^{-1}$. Similarly, for the Zn model, two sites were highly influential, with apparent changes in concentration of between −2.4 and − 7.6 mg kg^{-1} $year^{-1}$. For Cu and Zn, these highly influential sites were removed, since they had an unusually large effect for a relatively small number of sites.

3. Results

3.1. State and comparison with target ranges or guideline values

Summary statistics including the current state (median) for soil properties (Table 4), and a comparison with target ranges for

Table 3
Summary of the model fit per indicator.

Indicator	Marginal R^2	Conditional R^2
Cd	0.26	0.78
Cu	0.51	0.91
Zn	0.38	0.87
Macroporosity	0.36	0.54
Bulk density	0.45	0.81
AMN	0.53	0.73
TN	0.53	0.88
Olsen P	0.37	0.88
OC	0.57	0.90

Marginal and conditional R^2 values defined as in Nakagawa and Schielzeth (2013).

Table 4
Minimum, median and maximum for each soil indicator, for the most recent soil sampling.

Statistic	Land use system	Cd (mg kg^{-1})	Cu (mg kg^{-1})	Zn (mg kg^{-1})	Bulk density (Mg m^{-3})	Macroporosity (% v v^{-1})	TN (%)	AMN (mg kg^{-1})	OC (%)	Olsen P (mg kg^{-1})
Minimum	Dairy	0.13	6.0	32	0.88	2.3	0.26	70	2.6	27
	Drystock	0.05	4.0	28	0.81	3.1	0.31	56	3.0	9
	Forestry	0.04	3.4	15	0.85	14.7	0.21	30	3.5	6
	Horticulture	0.15	8.2	47	0.76	7.9	0.24	47	2.4	25
	Indigenous	0.04	2.6	18	0.63	8.5	0.19	75	2.9	6
	Market garden	0.09	13.1	52	1.04	9.6	0.14	14	1.3	38
	Mixed cropping	0.18	3.5	31	1.10	3.2	0.26	46	2.4	24
Median	Dairy	0.42	14.0	80	1.15	7.7	0.48	212	4.6	62
	Drystock	0.18	9.0	49	1.03	11.8	0.44	121	4.8	34
	Forestry	0.08	5.2	35	0.96	25.1	0.35	52	5.6	13
	Horticulture	0.25	21.1	56	1.06	12.8	0.46	103	5.8	80
	Indigenous	0.09	10.1	53	0.92	16.0	0.51	145	6.8	19
	Market garden	0.28	24.0	92	1.31	17.0	0.19	37	2.0	159
	Mixed cropping	0.20	7.3	56	1.22	13.6	0.30	58	3.1	47
Maximum	Dairy	0.63	25.0	97	1.41	20.1	0.87	353	9.8	96
	Drystock	0.50	20.0	177	1.21	19.3	0.87	222	10.6	96
	Forestry	0.12	7.6	47	1.00	30.8	0.46	56	7.7	88
	Horticulture	0.51	93.0	75	1.23	25.4	0.76	150	8.5	148
	Indigenous	0.21	16.8	100	1.19	34.2	0.91	277	13.3	106
	Market garden	0.46	98.0	112	1.42	26.0	0.37	145	4.9	219
	Mixed cropping	0.23	13.6	75	1.43	18.2	0.33	75	3.4	76

Sites used had no land use system change. Data for 82 unique sites were used as described in section 2.4.

productivity without environmental damage, or trace element toxicity guideline values for the protection of ecological receptors (Table 5), for the most recent sampling, for sites that had no change in land use system during the monitoring period, are presented per land use system in this section.

All land use system sites, except drystock, had Zn concentrations below toxicity guidelines (Table 5). Although one horticultural site had Cu of 93 mg kg^{-1}, which is just below the 100 mg kg^{-1} guideline, other sites were well below the guideline. With the exception of two dairy sites (with Cd 0.6 and 0.63 mg kg^{-1}), all sites had Cd concentrations within Tier 0 of the TFMS (Table 5).

Most land use system sites had bulk density values within target ranges, while all land uses had fewer sites within the macroporosity target range than for bulk density (Table 5). The land use system with the lowest compliance was dairy, where only 36% of sites were within the macroporosity target range, compared with 71% of drystock sites, while median macroporosity was lowest for dairy land use (Table 4).

All land use system sites, except for market gardens, had AMN values within target ranges (Table 5). Market gardens had the lowest percentage of sites (36%) within the target range for TN. Most non-indigenous sites were above the upper-target range for Olsen P values. Market garden sites had very high Olsen P concentrations (Table 4), with five sites exceeding 150 mg kg^{-1}. Only 21% of dairy sites were within targets for Olsen P, with 29% of sites ≥80 mg kg^{-1} (double the upper-target range value). All mixed cropping, dairy, drystock, and forestry sites were within carbon targets (Table 5).

3.2. Influence of land use and soil order

This section presents the statistical modelling assessment of the influence of land use and soil order on soil properties. A summary of the statistical modelling for the effect L_i for each land use system (Table 6), and S_j for soil order (Table 7) is provided. Comparisons are with the indigenous land use system or Brown Soil reference classes.

Cu concentrations were elevated in horticultural and market gardens sites (Table 6), and Recent Soils (Table 7). The dairy land use system had elevated Cd concentrations (Table 6). For soil physical properties, all non-indigenous land use systems, and Pallic and Recent Soils, had elevated bulk density, while reduced macroporosity was observed for four land use systems, and Recent Soils (Table 6, Table 7).

For AMN, forestry, horticulture, market garden, and mixed cropping land uses (Table 6), and Pallic Soils, had lower concentrations (Table 7). Four land use systems had reduced TN values compared with indigenous land use (Table 6), while Pallic and Recent Soils had reduced TN compared with Brown Soils (Table 7). Dairy, horticulture, and market garden systems had elevated Olsen P concentrations (Table 6). Compared with the indigenous reference, all land use systems had reduced OC percentage (Table 6), while Pallic and Recent Soils had lower OC percentage (Table 7).

3.3. Change over time

This section presents the statistical modelling assessment of change over time. A summary of the statistical modelling for the temporal change Δ_i is presented in Fig. 2. Significant increases over time were observed for Zn in dairy, TN for drystock, and Olsen P for mixed cropping (Fig. 2). Significant decreases over time were observed for Cu in forestry, Cd for indigenous and forestry, and bulk density for drystock (Fig. 2).

The change in Zn per year for dairy land use was 1.4 mg kg^{-1} yr^{-1} (95% CI 1.1, 1.8). The mean change in Olsen P over time for the mixed cropping farm system is 4.6 mg kg^{-1} yr^{-1} (95% CI 2.4, 9.3). Changes over time for individual sites for each land use are also presented for Zn (Fig. 3) and for Olsen P (Fig. 4). The supplementary material shows Cu and Cd changes over time, and a summary table of the statistical modelling.

4. Discussion

4.1. Land use, soil order, and comparison with targets or guidelines

Trace elements may accumulate in soils through intentional or unintentional application of agrichemicals. Nationally and internationally, it

Table 5
Target range or guideline value for individual soil properties, and percentage of sites within target range or below guideline value, for the most recent sampling of each site. Sites with no land use system change were used.

Indicator	Unit	Land use system						Reference
			Cropping	Pasture	Forestry	Horticulture	Market garden	
Target range								
Bulk density	(Mg m^{-3})		Pallic and Recent Soils (0.4–1.4), other soils (0.7–1.4)					1
Macroporosity	(% v v^{-1})		10–30	10–30	8–30	10–30	10–30	2
AMN	(mg kg^{-1})		>20	>50	>20	>20	>20 [A]	1
TN	(%)		0.25–0.7	0.25–0.7	0.1–0.7	0.25–0.7	0.25–0.7	1
Olsen P [B]	(mg kg^{-1})		20–40	20–40[C]	5–30	20–40	20–40[C]	2
OC	(%)		Recent Soils (>2), other soils (>2.5)					1
Guideline values								
Cd			Tier 0 up to 0.6 mg kg^{-1}, and Tier 1 up to 1.0 mg kg^{-1}					3
Cu			100 mg kg^{-1}					4
Zn			170 mg kg^{-1}					4
Sites within target range (%)		Cropping	Dairy	Drystock	Forestry	Horticulture	Market garden	
Bulk density		80	93	100	100	100	91	
Macroporosity		60	36	71	80	63	82	
AMN		100	100	100	100	100	82	
TN		100	71	88	100	75	36	
Olsen P		40	21	29	80	38	18	
Carbon		100	100	100	100	88	45	
Sites below guideline (%)								
Cd		100	86	100	100	100	100	
Cu		100	100	100	100	100	100	
Zn		100	100	96	100	100	100	

Pasture target includes dairy and drystock. Macroporosity at −10 kPa. [A] included as cropping. [B] Units were not clearly identified in Mackay et al. (2013), but Hill and Sparling (2009) state mg kg^{-1} should be used so mg kg^{-1} are the units used from our analysis and measurement. [C] Hill country Olsen P target value is 15–20, with same comments as in note C; three sites in this analysis classed as hill country. References: 1, Hill and Sparling (2009); 2, Mackay et al. (2013); 3, MAF (2011); 4, Cavanagh (2019), 95% protection level for ecological receptors.

is commonly reported that orchards and vineyards typically have higher levels of trace elements due to use of fungicides containing Cu, than other land uses (Gaw et al., 2006; Komarek et al., 2009; Marzaioli et al., 2010; Duplay et al., 2014). In the Wellington region, no horticultural sites exceeded the 100 mg kg^{-1} maximum guideline for Cu (though one orchard approached that value), and most horticultural sites were below 25 mg kg^{-1}. For comparison, a quarter of horticultural sites in the Waikato region of New Zealand exceeded the 100 mg kg^{-1} Cu guideline (Taylor et al., 2010), and this may be due to a longer period of more than 80 years in horticulture, compared to our study. The climate in the Wellington region is cooler and drier than in Waikato, which may not favour fungus growth as much as other regions, thus requiring less control. Zn concentrations in soils of the Wellington region are considered typical of values published for other regions (Taylor et al., 2011; Taylor, 2016), and New Zealand pastures (Alloway, 2008).

Cadmium is a recognised contaminant of many phosphate fertilisers, and the elevated Cd in dairy land in our study likely reflects both the use of superphosphate fertiliser, which typically has higher Cd concentrations than other P fertilisers used in market gardening for example, and application history. Additionally, older superphosphate fertiliser had higher cadmium concentrations (Roberts et al., 1994). The increased Cd concentration in the dairy land use agrees with results reported by Cavanagh (2014) and Abraham (2018). However, Taylor et al. (2010) also reported greater Cd concentrations in horticultural soils than other land uses, including dairy.

The greater bulk density in non-indigenous land use systems is consistent with other studies (Taylor et al., 2010; Curran-Cournane, 2015) and is likely to be attributable to compaction of soil from vehicles and animal grazing (Drewry et al., 2008; Houlbrooke et al., 2011). Our study shows soil compaction is common in the dairy land use, a similar finding identified in other studies (Taylor et al., 2010; Curran-Cournane, 2015; Ministry for the Environment and Stats NZ, 2018), but contrasts results of Clark et al. (2007), who reported most dairy sites had macroporosity values within the target range used in their study (8–20%).

Several land use systems had lower AMN concentrations and TN than indigenous vegetation. However, typically there is greater soil

Table 6
Estimated regression parameters for land use system with respect to the reference (indigenous). The values for each land use system are the change in the transformed response as the land use system changes from the reference, and where significant are shown by (*).

Response	Sigma[1]	Intercept[2]	Change for land use system from intercept					
			Drystock	Dairy	Forestry	Horticulture	Market garden	Mixed cropping
Log (Cu)	0.209	2.21	−0.102	+0.159	−0.444	+0.751 (*)	+0.971 (*)	−0.364
Log (Zn)	0.140	3.991	−0.136	+0.0247	−0.454 (*)	+0.131	+0.206	−0.179
Log (Cd)	0.302	−1.602	−0.0248	+0.753 (*)	+0.0251	+0.310	+0.162	−0.00304
Log (Bulk density)	0.0833	−0.1987	+0.22 (*)	+0.203 (*)	+0.164 (*)	+0.202 (*)	+0.335 (*)	+0.419 (*)
Sqrt (Macroporosity)	0.711	4.617	−1.02 (*)	−1.62 (*)	+0.0922	−0.770 (*)	−0.432	−1.480 (*)
Log (AMN)	0.351	5.085	−0.147	+0.0157	−0.952 (*)	−0.356 (*)	−1.29 (*)	−0.735 (*)
Log (TN)	0.173	−0.4713	−0.252 (*)	−0.103	−0.427 (*)	−0.174	−0.931 (*)	−0.688 (*)
Log (Olsen P)	0.346	2.971	+0.0199	+0.73 (*)	−0.466	+1.05 (*)	+1.36 (*)	−0.0212
Log (OC)	0.168	2.192	−0.379 (*)	−0.332 (*)	−0.315 (*)	−0.358 (*)	−1.06 (*)	−0.873 (*)

[1] The sigma is the residual standard error of the residuals.
[2] The intercept is the estimated mean for the indigenous land use system class with Brown Soil order.

Table 7
Estimated regression parameters for soil order with respect to the reference class (Brown Soils). The values for each soil order are the change in the transformed response as the soil order changes from the reference, and where significant are shown by (*).

Response	Sigma[1]	Intercept[2]	Change for soil order from intercept		
			Gley	Pallic	Recent
Log (Cu)	0.209	2.21	+0.257	−0.0748	+0.44 (*)
Log (Zn)	0.14	3.991	+0.248 (*)	−0.0307	+0.336 (*)
Log (Cd)	0.302	−1.602	+0.0714	−0.0505	−0.103
Log (Bulk density)	0.0833	−0.1987 (*)	+0.0169	+0.0938 (*)	+0.158 (*)
Sqrt (Macroporosity)	0.711	4.617 (*)	−0.427	−0.317	−0.405 (*)
Log (AMN)	0.351	5.085 (*)	−0.0929	−0.201 (*)	−0.192
Log (TN)	0.173	−0.4713 (*)	−0.0926	−0.198 (*)	−0.376 (*)
Log (Olsen P)	0.346	2.971	+0.245	+0.115	+0.267
Log (OC)	0.168	2.192 (*)	−0.150	−0.250 (*)	−0.458 (*)

[1] The sigma is the residual standard error of the residuals.
[2] The intercept is the estimated mean for the indigenous land use system class with Brown Soil order.

AMN and TN in pasture compared with cultivated soils, e.g. market gardens, as pastoral soils have urine and dung-N input. AMN and TN can also decrease in cultivated soils, relative to pasture, as higher frequency of cultivation can be associated with reduced levels of soil carbon (Curtin and McCallum, 2004; Sparling and Schipper, 2004; Marzaioli et al., 2010; Curtin et al., 2017; Bai et al., 2018). We had a similar finding in our study. However, as noted by Powlson et al. (2014), many studies also show the additional carbon due to no-till practices is relatively small. Stevenson et al. (2015) reported that land use classes do not always have well-defined 'boundaries' due to a range of land (and possibly N) management practices associated within land uses. Similarly, pastoral and cropping land uses have been shown to have high variability in AMN and TN (Curtin and McCallum, 2004; Taylor et al., 2010; Curtin et al., 2017).

All land use systems in our study had lower OC content than indigenous vegetation, a similar observation reported in other studies (McIntosh et al., 1997; Taylor et al., 2010; Curran-Cournane, 2015). Similarly, Sparling and Schipper (2004) reported OC was the lowest for cropping and tussock grassland than in other land uses, at 0–10 cm depth. Soil OC typically decreases with land use intensification, such as increased frequency of cultivation (Marzaioli et al., 2010; Akinsete and Nortcliff, 2014; Curtin et al., 2017; McNally et al., 2017; Schipper et al., 2017). However, carbon stocks to 30 cm depth were reported by Ministry for the Environment (2010) to be greatest in pastoral land. Therefore, OC levels vary considerably, spatially and with management practices and depth, even within land uses such as grazed pasture (Curtin et al., 2017; Mudge et al., 2017; Whitehead et al., 2018). Grazed dairy pastures are typically reported to be within OC target ranges (e.g. Clark et al., 2007; Curran-Cournane, 2015), including OC targets determined by Hill and Sparling (2009) based on development of production and environmental response functions for New Zealand soil quality indicators (Lilburne et al., 2004; Sparling et al., 2008).

In our study Olsen P in many non-indigenous sites exceeded the upper target value, suggesting P concentrations are above agronomic requirements, as also observed elsewhere (Cotching and Kidd, 2010; Taylor et al., 2010; Curran-Cournane, 2015; Gourley et al., 2015; Ministry for the Environment and Stats NZ, 2018). High P concentrations pose a risk to water quality at some sites (Burkitt et al., 2010; Curran-Cournane et al., 2011; Hart and Cornish, 2016). However, Olsen P concentrations range widely between sites as shown by Clark et al. (2007), but in contrast to our study, they had dairy site median values within agronomic targets for three of five catchments.

Many monitoring studies typically have focused on land use. Our study was predominantly focussed on land use, but also included soil order. For example, Pallic and Recent Soils had greater bulk density than Brown Soils. Brown Soils have also been shown to be generally well structured, and therefore be naturally more resilient to soil physical damage than Pallic Soils (Greenwood and McNamara, 1992; Hewitt and Shepherd, 1997; Drewry et al., 2000). Pallic Soils had greater bulk densities than Brown Soils, irrespective of land use in our study, with similar conclusions elsewhere (McIntosh et al., 1997; Drewry et al., 2000). The higher OC observed in Brown Soils is suggested to partly arise from their formation in higher rainfall areas (McIntosh et al., 1997; Hewitt, 2013), and contributes to their relative structural stability. Recent Soils are typically young weakly weathered soils which have had less time to accumulate organic matter, and contain lower OC and TN compared with other soil orders (Curran-Cournane et al., 2013). Studies have also reported OC content is typically an inherent soil property, for well-developed soils, explained more by soil order rather than by land use (e.g. Sparling and Schipper, 2002; Cotching and Kidd, 2010), and therefore related to inherent (pedological) soil properties, e.g. specific surface area (McNally et al., 2017; Kirschbaum et al., 2020). In contrast, however, OC content can also be modified by land use and management, e.g. depleted by cultivation. Our study showed Brown Soils had greater TN than the other soils. Similarly, Brown Soils had greater TN content than Pallic and other soils elsewhere (McIntosh et al., 1997; Curran-Cournane et al., 2013), but in contrast to OC, Sparling and Schipper (2002) reported that land use had the same effect on TN as soil order.

4.2. Change over time

Monitoring the change of soil properties over time can be useful to help assess the impacts of different land management practices, or whether they are needed. For change over time, multiple samplings are more robust than only two samplings. In analysis of multiple samplings of the same site, as in our study, site conditions (e.g. soil) do not change, or are chosen so that changes are unlikely, or are minimal (e.g. land use system as in our analysis). The changes in soil properties at a site are therefore less likely to be the result of variations in site conditions or management. Multiple samplings are preferred over a single repeated measurement because there is a chance that a single detected change is due to random variation, which is less likely when significant change is detected using additional samplings. Despite this, very few soil quality monitoring programmes internationally have had more than a one-off sampling (Arrouays et al., 2012; Bünemann et al., 2018; Smith et al., 2020), from the same sites for multiple land uses, e.g. the study of Curran-Cournane (2015) had two samplings per land use, while grassland sites were sampled five times and in a soil carbon study (Kühnel et al., 2019). In a study of environmental trace elements in Swiss soils, Desaules et al. (2010) recommended improving measurement quality and increasing frequency of sampling to detect reliable trends.

With the exception of one land use for Zn and Cu, and two for Cd, there was an absence of significant changes over time for these trace elements in the other land use systems. The absence of a statistically significant change over time is not necessarily bad, as it depends on reasons for the absence, e.g. a stable situation, as long as there are enough monitoring sites to detect change. Where a significant trend is found, the importance depends on the direction. When a trend is detected what is important is whether the magnitude of the trend is actually meaningful from an environmental or a production perspective, and its relativity to guideline values. The observed increase in Zn in dairy land use suggests ongoing accumulation in soil Zn. The change in Zn per year was equivalent to mean change of 4.2 mg kg^{-1} over the 3-year sampling period, and shows that the statistical analysis is sensitive. In practice, this increase would largely fall within the sampling and analytical variability, although over 20 years it suggests a mean increase

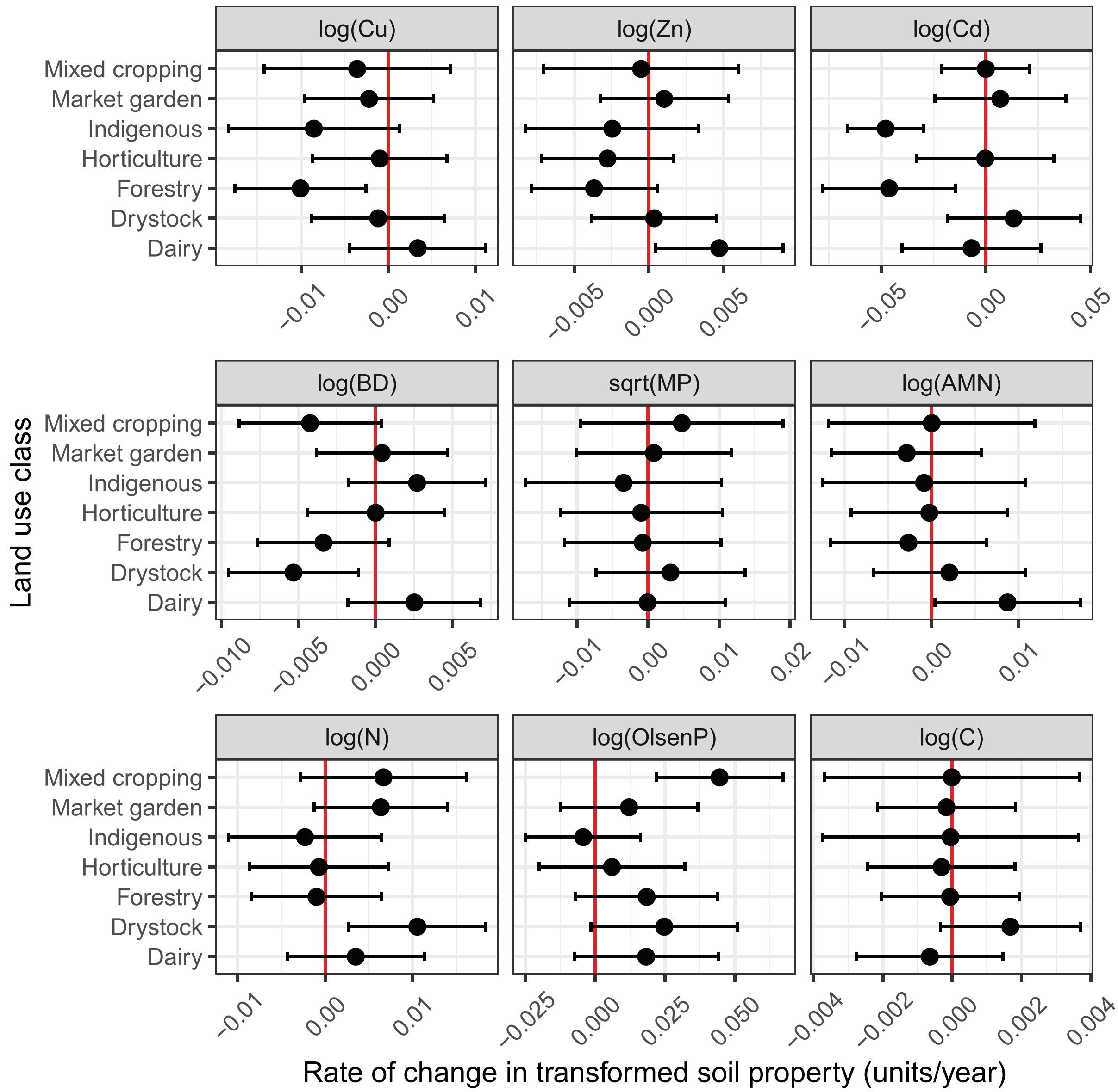

Fig. 2. Rate of change in transformed soil properties over time (units year^{-1}) for each land use system class. The filled circles give the mean estimate of the rate of change for that land use system, while the error bars indicate the 95% confidence interval for the estimate. Note that all but one property uses a log transformation as the response while a square-root transformation is used for macroporosity.

of 28 mg kg^{-1}. The accumulation could be attributable to use of animal health products for facial eczema treatment (Taylor et al., 2010). Continued accumulation of Zn has potential negative impacts on soil quality if concentrations increase beyond what was observed in this study, such as Zn toxicity to soil microbial function and plants, induced copper deficiency, and antibiotic resistance in soil bacteria (Kim and Taylor, 2017; Heydari, 2020).

Cavanagh (2014) and Abraham (2018) reported no clear trends in soil Cd concentration over time for land uses monitored by regional authorities and fertiliser companies in New Zealand. Evidence of trends in soil Cd varies, with indications of a plateau and levelling for Cd concentrations at a long-term sheep-grazed trial site (McDowell, 2012), in contrast to a recent study at that site that showed Cd concentrations continuing to increase over time (Gray et al., 2017). While the modelling fit was good for the Cd and Cu changes over time in the forestry sites, this should be treated with caution as only five sites were monitored. Further sites would give more confidence in this result, as a significant trend with few samples is less powerful, since there is a greater risk that replacing one of the measurements could change the significance compared with a larger dataset.

Although there was a reasonable model fit for bulk density, and no change over time for other land use systems, the improvement (reduction) in bulk density over time for drystock systems should be treated with caution, for the reasons described previously. Soil physical properties, including bulk density, are reported to improve after summer drying (Drewry et al., 2004b). Although Curran-Cournane et al. (2013) reported they could not rule out climate variability for their samplings as a possible causal factor of change, a seasonal factor is an unlikely explanation for the bulk density change in our study, as mean soil moisture between our samplings was similar in the drystock system. In contrast to other studies (e.g. Curran-Cournane et al., 2013; Curran-Cournane, 2015; Taylor et al., 2017), our study did not detect significant changes in macroporosity over time for any land use, but the model fit as assessed by conditional R^2 was the lowest of all our models. Other

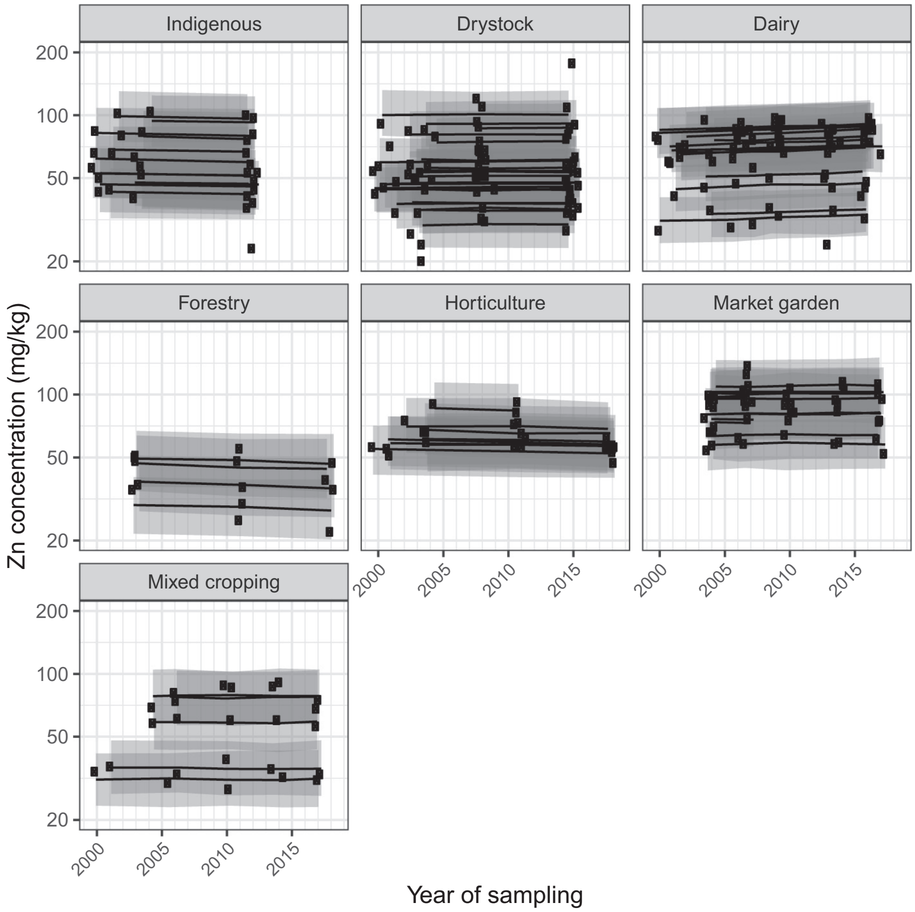

Fig. 3. Temporal predictions from the log-transformed Zn concentration model, grouped by land use system. Points indicate field data and lines indicate mean predictions for a specific site. Unfilled points are outlier Zn values not used in the model. The shaded region is plus-or-minus one standard deviation from the mean prediction for a given site. Note that the values on the vertical axes are back-transformed values in mg kg^{-1} on a log scale axis.

studies reported significant changes in macroporosity over time, and therefore found this property to be a useful indicator of change over time, particularly over the 0–10 cm depth (Drewry et al., 2004b; Monaghan et al., 2005; Taylor et al., 2017), but these studies had much more frequent samplings (2–4 times per year for up to four years) than those in our study.

Soil Olsen P is commonly measured by agronomic advisors and fertiliser representatives as the basis for fertiliser application recommendations in pasture (0–7.5 cm depth) and cropping (0–15 cm). As such, whether there is a change over time in Olsen P depends on the extent to which farmers actively manage fertiliser application based on this soil property. The observed increase in Olsen P concentrations in mixed cropping system over time suggests growers are not managing fertiliser application based on this property and crop requirements and are applying P fertiliser beyond agronomic need. Parfitt et al. (2014) reported significant increases in P fertility on flat land for dairy and drystock. More recently, McDowell et al. (2019) also identified significant increases in soil Olsen P concentrations (0.08 to 1.15 mg P L^{-1} yr^{-1}) over 2002–2014, across New Zealand. Different studies have suggested different factors may influence fertiliser application, but research is needed on farmer soil testing and nutrient management decision-making processes (e.g. Daxini et al., 2018; Lobry de Bruyn, 2019). Taylor et al. (2017) suggested P application was also influenced by commodity prices, while Parfitt et al. (2012) suggested economic return, drought, and the use of nutrient budgeting influenced changes in farm N fertiliser use during 1990–2010. However, McDowell et al. (2019) suggested that isolating probable causes for regional and national trends in nutrient mangement and environmental impact remains a challenge.

4.3. Implications for regional and national soil quality monitoring programmes

Soil quality monitoring programmes are science-based soil management tools to assess soil ecosystem health (Curran-Cournane, 2015), to provide an early warning of change (Desaules et al., 2010; Taylor et al., 2010), and detect adverse changes (Arrouays et al., 2012). They are intended to inform land managers (and regulators) about the changing state of the environment (Desaules et al., 2010; Bünemann et al., 2018), and inform resource management policy (e.g. Waikato Regional Council, 2016). In New Zealand, national reporting on soil quality occurs every three years (Ministry for the Environment and Stats NZ, 2018), but regional scale data are not available in all regions, and their data require development to enable national consistency to ensure they are reliable and have sound statistical methodology to meet government statistical criteria (Statistics New Zealand, 2007; PCE, 2019).

Of particular importance in soil quality monitoring programmes is the ability to detect change over relevant temporal and spatial scales, i.e. with adequate precision and statistical power from programme designs (Arrouays et al., 2012). The absence of a statistically significant change over time may indicate stable conditions if sample size is sufficient. The sample size requirements for soil quality at a national level analysed by Hill et al. (2003), provides some guidance, with a sample size per land use recommended to be >30 samples per region

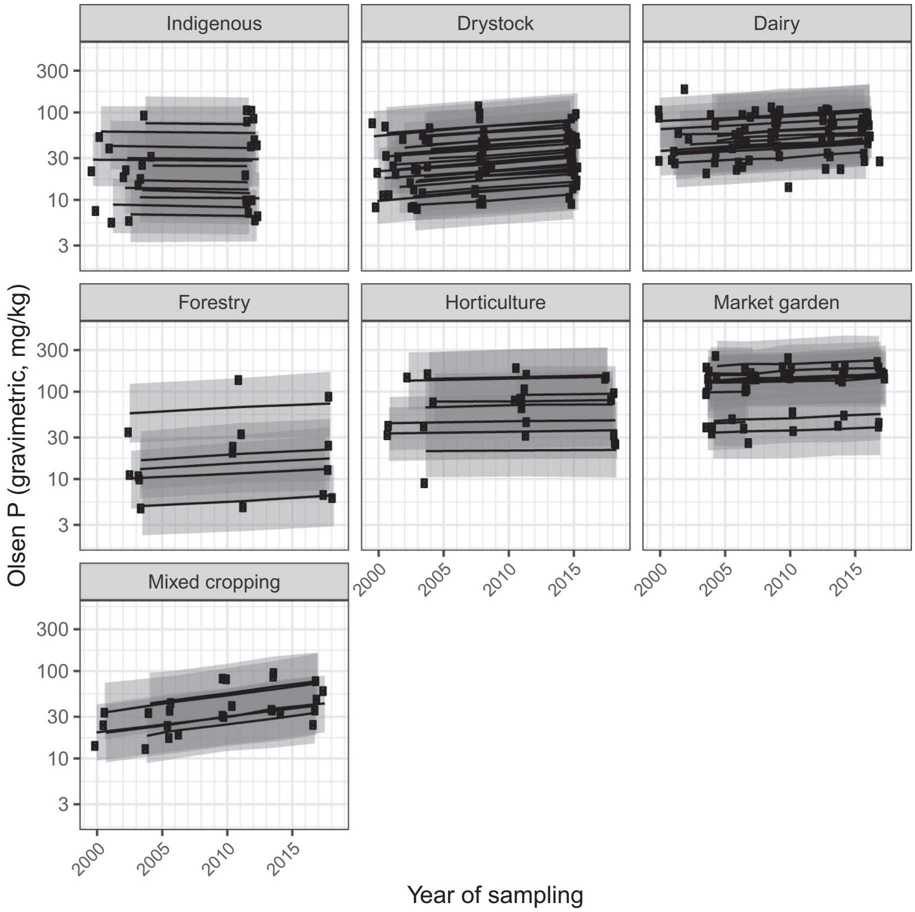

Fig. 4. Temporal predictions from the log-transformed Olsen-P model, grouped by land use system. Points indicate field data and lines indicate mean predictions for a specific site. Unfilled points are outlier Olsen-P values not used in the model. The shaded region is plus-or-minus one standard deviation from the mean prediction for a given site. Note that the values on the vertical axes are back-transformed values in mg kg^{-1} on a log scale axis.

(Hill and Sparling, 2009), but other factors and environmental complexity should also be considered. Increasing the number of forestry and mixed cropping sites in the Wellington region would provide more representative and robust sampling and increase future ability to detect significant changes, as low numbers of sites provides less confidence that any change is robust and real, or the absence of change is not just an artefact of data variability. However, change over time at regional scale has not been evaluated in detail in those studies, so our study provides this knowledge.

The apparent clustering of monitoring sites (Fig. 1) largely reflects the distribution of intensive farming, i.e. dairy and mixed cropping, as the original study design had its focus on risk to the environment from land use intensification (Sparling and Schipper, 2002). Drystock systems in hill country are less intensive. However, eastern hill country drystock sites are spatially under-represented as the remaining drystock sites are mainly on flat land. Similarly, forestry sites are also under-represented in the eastern region. Detecting change over time has been challenging in this regional programme, due to varying sampling frequency, low site numbers for two land uses, and high site variability, but the repeat samplings and our analysis methods have helped. Other considerations include representativeness of spatial coverage and variability of soil quality indicators which our study provides over an extended period. These issues should be considered in any review of the programme. Similarly, other issues such as improved harmonisation of a range of indicators and methods from individual programmes, across administrative boundaries (e.g. Morvan et al., 2008; Arrouays et al., 2012; Smith et al., 2020), should be considered when aggregating regional or national programmes. It may also be useful to consider future monitoring an additional depth such as to 30 cm, as there is typically more variance in surface soil layers than at depth. It is important to report soil properties, particularly OC (e.g. Ministry for the Environment, 2010; Smith et al., 2020) and TN to greater depths than 10 cm, to understand more fully the environmental impacts as well as agronomic benefits. Soil carbon stocks, for example, are also generally evaluated to 30 cm or deeper (McNeill et al., 2014), and a minimum of 30 cm was recommended for evaluating changes in carbon content (Smith et al., 2020). A future challenge will be effects of climate change, so an additional depth could be considered to provide more robust data to evaluate its effects on soil quality, especially carbon. The monitoring programme has contributed to samples for a recent, separate study evaluating soil bacterial communities and soil quality using gene sequencing (Hermans et al., 2020), so consideration of this biological information should be included in future.

Even the definitions of land use class or system, and definitions in spatial data (e.g. Cavanagh et al., 2017) should be considered for consistency when aggregating regions to a national scale. In our study, we considered market gardens to be a separate land use to mixed cropping because different processes occur, e.g. biannual versus much less frequent cultivation, respectively, so we gained knowledge for each of these systems. In contrast, other studies have typically aggregated some land uses, e.g. cropping, market gardens, and horticulture aggregated into a single class (e.g. Ministry for the Environment and Stats NZ, 2018). However, our decision has a trade-off, namely generating knowledge on separate land uses, but compromises site numbers.

There are trade-offs between additional sites and ability of soil monitoring programmes to detect meaningful, but also statistically significant differences. The trade-off includes additional expense incurred for sampling and analysis. There are also trade-offs in monitoring programme designs, e.g. when resampling sites, knowledge of temporal variation is gained, but if more sites were sampled or locations changed, then spatial resolution would improve (Arrouays et al., 2012). In our study, the sampling interval for intensive land use systems is frequent (3-yearly), compared with other soil quality monitoring programmes internationally, and as discussed, frequent sampling has major benefits for detecting statistically significant trends. A key requirement is to have enough sampling sites so that changes in soil properties (if any) could be detected when sampled with the current frequency. Since temporal changes in a key set of soil quality properties in New Zealand land use systems have not been well documented, our study provides some information to estimate what changes could be detected under the same, or an expanded, setting. Since our study was concentrated on a single region, to provide additional sites per land use and soil, the next step would be to combine data from other regions for a comprehensive analysis, to provide additional evidence for decision-makers, and firm guidance for future sampling.

There are very few soil quality monitoring programmes internationally that have had more than a one-off sampling (Saby et al., 2008; Arrouays et al., 2012). The soil quality monitoring programme in Alberta, for example, had four of the 23 national sites, but ceased when funding ended following the 10-year resampling of the sites in 2002/2003 (Government of Alberta, 2020). That programme had been resampled at 5 and 10 years only. Our study shows the Wellington region monitoring programme and its statistical analysis of multiple samplings are valuable for detection of significant trends over time as an early warning, e.g. Zn and Olsen P changes, and comparison with targets and guidelines. Our study provides evidence to show that future monitoring should consider additional site numbers where they are low, and increased sampling frequency to ensure robust statistical analyses can be conducted. The potential for influential points (e.g. as seen in changes of Cu concentration) suggest that maintaining consistent standards of site measurement and laboratory analysis is needed across time and between regions for a national monitoring programme to provide reliable information on soil quality.

Finally, our analysis included only the sites where land use systems did not change during the monitoring period, hence it provided a robust basis for detecting change over time. These learnings and methods could be applied to other programmes internationally. Despite the value of long-term soil quality monitoring that we have shown in this study, very few programmes have endured long-term internationally, so this study adds significantly to the knowledge on soil quality for informing robust policy, resource, and environmental decision-making.

5. Conclusions

Very few soil quality monitoring studies internationally have reported on multiple samplings over the long-term, whereas our study reports five re-samplings. For the most recent sampling per land use for the Wellington region programme, all land use system sites, except drystock, had Zn concentrations below ecological toxicity guidelines, while most sites had Cd and Cu concentrations below these guidelines. Dairy and market garden land use systems had low percentages of sites within the Olsen P target range, indicating concentrations at some sites were above agronomic needs, and of potential environmental risk. Dairy land use systems had a low percentage of sites within the macroporosity target range, indicating soil compaction. Soil compaction has been associated with reduced pasture yield and increased environmental risk of surface runoff and nitrous oxide gas emission.

Across all samplings, compared with indigenous land use, Cu concentrations were elevated in horticultural and market gardens sites, while several land uses had lower macroporosity, AMN, TN, and higher bulk density and Olsen P concentrations. Pallic and Recent Soils had elevated bulk density, lower OC percentage compared with Brown Soils.

Across all samplings over the 19 years, significant increases over time were observed for Zn in dairy, TN for drystock, and Olsen P for mixed cropping land use systems. Significant decreases over time were observed for Cu in forestry, Cd for indigenous and forestry, and bulk density for drystock. No changes over time were detected for macroporosity, AMN, or organic carbon, for the 0–10 cm soil depth. The absence of a statistically significant change over time is not necessarily bad, as it depends on reasons for the absence.

This study shows the monitoring programme and our analysis of multiple samplings are valuable for detecting significant trends as an early warning, e.g. Zn and Olsen P changes. The study provides evidence and discussion for additional site numbers where they are low, and increased sampling frequency to ensure robust statistical analysis, as low numbers of sites provides less confidence that any change is real. This study included only sites where land use systems did not change, providing a sound basis for detecting change over time, and for informing regional resource management policy, land management and environmental decision-making. We recommend this programme continues, because despite the value of long-term soil quality monitoring that we have shown in this study, very few programmes internationally have endured over the long term. To ensure future knowledge needs are able to be met by the programme, we also recommend consideration of additional sites, potentially an additional sampling depth especially for improved integration with other new programmes in New Zealand, such as carbon monitoring. This study adds significantly to our knowledge on soil quality.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.geodrs.2021.e00383.

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