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Effects of Winery Wastewater to Soils on Mineral Properties and Soil Carbon

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Abstract: Winery wastewater (WW) is a high-volume biowaste and, in the context of Marlborough and New Zealand wineries, there is a growing recognition of the need to improve current WW disposal systems to mitigate negative environmental impacts. The application of WW to land is a low-cost method of disposal, that could significantly reduce the environmental risk associated with WW directly entering surface and groundwater bodies. This study analysed elemental concentrations in WW and soils from three Marlborough vineyards across their annual vintage to determine the loading rates of nutrients into WW and the subsequent accumulation effects of WW irrigation on receiving soils. The findings showed loading rates of approximately 1.8 t ha^{-1} yr^{-1} of sodium within WW and a significant increase in soil sodium concentration and pH, attributed to sodium-based cleaning products. A loading rate of approximately 4 t ha^{-1} yr^{-1} of total organic carbon was also identified within WW, however, significant losses in soil carbon, nitrogen, magnesium and calcium concentrations were identified. Focusing efforts to retain key nutrients from WW within soils could provide benefits to New Zealand's wine industry, facilitating increased biomass production in irrigation plots, thereby increasing biodiversity and potentially generating incentives for vineyard owners to contribute to increasing biomass carbon stocks and offset agricultural greenhouse gas emissions.

Keywords: sustainability; wine; wastewater reclamation; sustainable development goal 15; life on land; soil carbon loss

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Citation: Nightingale-McMahon, M.; Robinson, B.; Malcolm, B.; Clough, T.; Whitehead, D. Effects of Winery Wastewater to Soils on Mineral Properties and Soil Carbon. *Land* **2024**, *13*, 751. https://doi.org/10.3390/land13060751

Academic Editor: Amrakh I. Mamedov

Received: 3 May 2024
Revised: 23 May 2024
Accepted: 25 May 2024
Published: 28 May 2024

1. Introduction

By 2030, an aim of the United Nations Sustainable Development Goals (SDGs) is to halve the proportion of untreated wastewater by substantially increasing recycling and safe reuse of industrial wastewater, thereby minimising the release of hazardous material and reducing pollution [1]. Presently, the volumes of industrial wastewater are unquantified, resulting in a limited understanding of the risks or scope for reuse as potential resources. One suitable source for reclamation is winery wastewater (WW) [2], a significant byproduct of the winemaking process, generated from cleaning tanks, washing floors and equipment, rinsing transfer lines, cleaning barrels, managing off wine and product losses, bottling facilities, and operating filtration units [3]. The volumes of WW produced are generally related to the processing capacity of each vineyard. However, estimates vary from 0.2 to 14 litres (L) of WW per L of wine produced, depending on the type of wine and the processing required [4,5]. In New Zealand, it is generally accepted that approximately 10 L of WW is produced per L of wine [6–8], suggesting an approximate national generation of 380,000 m^3 yr^{-1}. [9]. The implementation of effective WW management strategies is considered a critical issue to improve the environmental performance of New Zealand's expanding wine industry as future limitations to water supply are anticipated [10].

As the largest wine-producing region in New Zealand, Marlborough accounts for 74% of the total grape production [11], generating approximately 280,000 m^3 of WW annually [6]. Currently, two methods of WW disposal are available to Marlborough wineries: discharge directly to the municipal wastewater network, or irrigation to land. Disposal through municipal wastewater is not possible for many wineries due to their locality outside the civil network, additionally, the processing required at the wastewater treatment plant has associated costs and the subsequent discharge to sea has potentially harmful environmental impacts [7]. The practice of disposing of WW onto field crops, tree lots, pastures and vineyards through land application is common as a cost-effective method of WW disposal in Marlborough as minimal technical expertise is required [12]. However, there is an environmental risk associated with land application as WW can directly enter surface water and groundwater bodies [13],degrading the resources [7,14] through salinisation, eutrophication and depletion of dissolved oxygen (DO) [15,16]. The deterioration of these water bodies hinders their ability to be used for drinking supply [17,18], water for livestock, and recreational activities [19]. Due to the high organic load, variable salinity and the large volumes produced, it is also crucial to ensure the receiving soils can facilitate the added nutrients, salts and other contaminants [20,21]. Currently, 39 wineries across the Marlborough region discharge their WW to land while the majority discharge their WW to the wastewater network [7]. Compliance monitoring undertaken previously by MDC has observed impaired infiltration, ponding and runoff are apparent, particularly around the autumn vintage season (March to April) [22]. Of the 39 wineries discharging their WW to land in Marlborough, 13 were deemed fully compliant with permitted criteria for WW discharge over the 2021/2022 period, with compliance failures relating to surface ponding, WW sampling inadequacy and parameter exceedances (particularly pH) [23].

Typically, the composition of WW varies based on the type of wine produced [24], the season [13] and the bespoke nature of the WW treatment systems employed by each vineyard [25]. Due to the high quantities of grape juice and solids, WW is typically acidic [15] and contains high concentrations of potassium (K) [26,27]. In addition, high sodium (Na) concentrations are often observed, most of which result from the use of alkaline cleaners used for sanitation, i.e., sodium percarbonate ($2Na_2CO_3{\cdot}3H_2O_2$), sodium carbonate (Na_2CO_3) and sodium hydroxide (NaOH) [28], resulting in the Na concentration dependent on sanitation operations rather than the volume or type of wine processed. When applied to soils, the relative molar concentration of Na or K compared to the divalent cations of magnesium (Mg) and calcium (Ca) is expressed as the Na adsorption ratio and K adsorption ratio (SAR and PAR, respectively), in $\text{mmol}_c\ L^{-1}$. A high value for SAR and PAR present in WW can cause the accumulation of Na and K [7,9,29], leading to the dispersal and subsequent loss of Mg and Ca ions from the soil exchange complex [30], resulting in soil nutrient deficiencies in these elements. High SAR and PAR values have also been shown to change soil structure by increasing clay dispersion and reducing the distribution of mesopores and micropores [31,32]. This impedes water and root penetration [33], ultimately leading to a reduction in plant growth. The decrease in soil infiltration rates can also result in surface runoff and soil erosion [34], subsequently remobilising pollutants into surrounding environments [35].

Attempts to mitigate the accumulation of Na in soils resulting from WW irrigation have led to the encouragement for industry to change to K-based caustic cleaning products (e.g., KOH) [26]. Previous research in Marlborough has shown that the soil dispersion risk under WW irrigation is low due to the high silt content in the native soils [29]. However, the accumulation of Na and K in soils under WW irrigation has been observed [7]. Grape residues also cause WW to be high in organic compounds such as sugars, ethanol [5], organic acids, phenolic compounds [36] and higher molecular weight organic compounds [28]. The decomposition of these organic compounds leads to elevated biological and chemical oxygen demand (BOD and COD, respectively), particularly during the grape harvest period in autumn (March to April) [20,37]. While elevated BOD and COD may hinder soil and plant function [13], the addition of the organic constituents also provides significant increases in soil carbon (C) and a slow release source of other nutrients [14,28]. International evidence suggests that the rates of WW to land application could be kept within a potentially beneficial range for key soil and plant nutrients including C [15,25], K [26], phosphorus (P) [38] and some cases, nitrogen (N) [5,39]. However, the limited records on the volumes of WW applied in Marlborough's vineyards result in inaccuracies when approximating the total nutrient loading rates [20] and the impacts on plant growth are not known.

The aim of this research was to quantify the loading rates and subsequent accumulation of WW nutrients in soil C, N, P, K and Na, as well as the effects on soil pH. We hypothesised that the application of WW would lead to an increase in soil C stocks, in both the topsoil and subsoil as the organic C substrates within WW are assimilated into receiving soils. Increases in total N, P and K were also hypothesised, suggesting a potential enhancement of soil fertility by positively affecting the soil's nutrient availability [14]. However, we hypothesised that any enhancement in soil fertility would be hindered by increases in Na, with expected rates of accumulation higher than those for K.

2. Materials and Methods

2.1. Site Descriptions

Two sets of soil and WW samples were collected in spring (October) 2021 and autumn (April) 2022 from three separate vineyards (Site A, Site B and Site C) in Marlborough, New Zealand (Figure 1). Samples were obtained under the conditions of anonymity of land ownership and therefore, Figure 1 identifies only the general locality of the three vineyards. Due to the topography, the average annual rainfall within the Marlborough region is highly variable; however, the sampling region, inland from Blenheim typically receives 600–800 mm of rainfall per year [40]. The soil at Site A and Site C are described as Typic Fluvial Recent Soils (RFT), and the soil at Site B is classed as Immature Orthic Brown Soil (BOI) [41]. RFT and BOI soils are taxonomically related to Typic Ustifluvent and Dystric Ustochrept soils [42,43], respectively, and are weakly developed and formed in alluvial sand, silt or gravel [44]. These soils are generally well drained with very low vulnerability to water logging in non-irrigated conditions, however, have a moderate or high structural vulnerability and are therefore susceptible to leaching [45]. Results from particle size analyses of soils confirmed that there were no significant differences in soil texture between Site A and Site C where soils were predominately sandy loam ranging to silt loam [43]. Site B's soil texture was significantly different from that at Sites A and C with higher silt and clay contents, predominantly classified as a silt loam.

The site coverage across the WW irrigation plots, where soil samples were collected, was similar for all three sites, largely consisting of grass cover with a small number of sporadic gum trees (*Eucalyptus camaldulensis*) at Site B and C. Georeferenced historic aerial photographs held by Marlborough District Council (MDC) indicates that the WW irrigation plots were established between 2005 and 2012, 1996 and 2004 and 2012 and 2016 for Site A, B and C, respectively [46]. The WW irrigation plots for all three sites were approximately 5.1 ha, 1.0 ha and 1.6 ha, respectively. Annual WW volume generation was recorded at Site B at the discharge point (July 2021 to June 2022) as 6661 m^3. The WW volumes at Site C were estimated to be 10,000–12,000 m^3 over the same time period, based on water usage throughout the year as measured at the vineyard's water take. The annual WW application rate from Site B can be therefore estimated at approximately 670 mm and Site C can be estimated between 625 mm and 750 mm. It was not possible to estimate WW volumes for Site A during the 2021/2022 period.

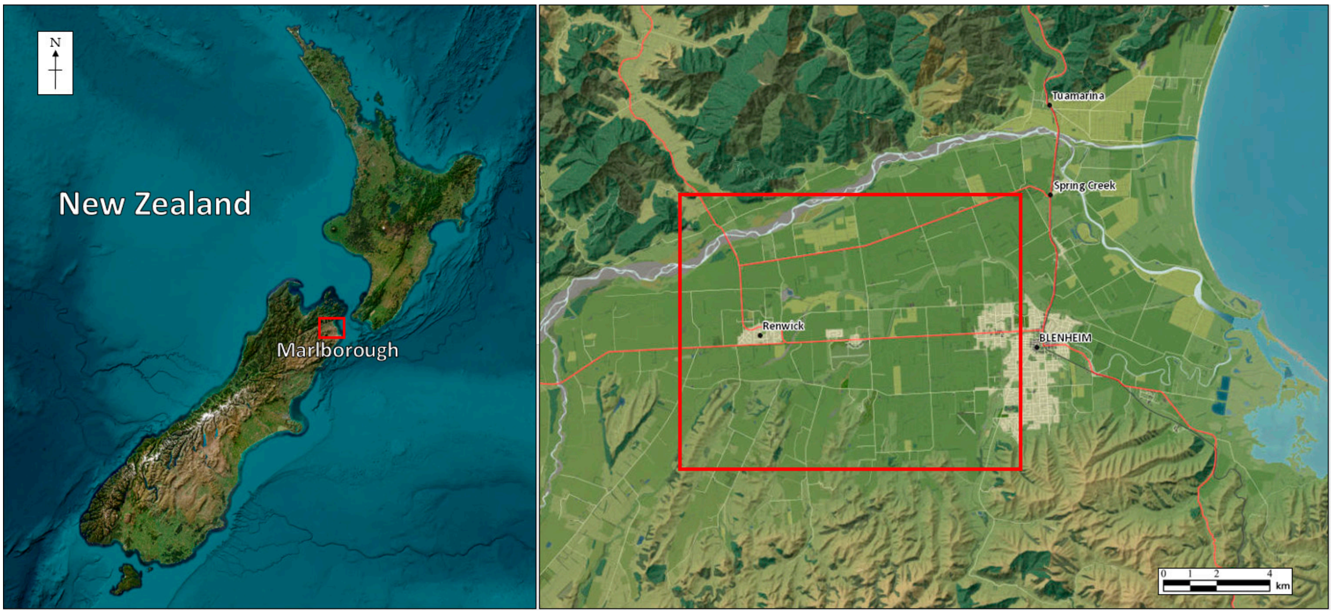

Figure 1. Map of Marlborough region on the northern coast of the South Island of New Zealand. Red squares indicate the general locations of the three sites where samples were collected for the measurements.

2.2. Sample Collection

During the spring 2021 sample round, a total of 30 soil pits, 10 per site, were hand-dug using a stainless-steel spade to a maximum depth of 600 mm. Five pits were excavated within each of the WW irrigation plots, the locations of which were chosen randomly within each WW irrigation plot and an additional five pits were excavated on soils deemed to have not been irrigated with WW (control), adjacent to each irrigation plot where land cover was the same. Soil samples were collected from four depths of 0–50 (referred to as 25 in Figures), 300, 450 and 600 mm, at each pit using a stainless-steel trowel. This resulted in the collection of 120 soil samples that were transported to the laboratory on ice, where they were kept at <10 °C prior to analysis. During the autumn 2022 sampling period, a total of 30 soil samples, 10 per site, we collected from 0 to 50 mm depth (referred to as 0 in Figures) using a stainless-steel trowel. Again, five samples were chosen randomly within the WW irrigation plot, and five control samples were taken from non-irrigated soils adjacent to each WW irrigation plot. This resulted in the collection of 30 surface samples, resulting in 150 soil samples in total over the two sampling periods.

The WW samples were collected in 20 L sealed plastic containers from each of the three wineries discharge points during both the spring and autumn sampling periods and transported to the laboratory to be stored (<10 °C) prior to analysis. A WW sample was unable to be taken from Site B during the autumn period due to unscheduled maintenance of the vineyard's WW treatment system.

2.3. Chemical Analyses of Soil and WW Samples

Soil water content was determined by weighing 10–20 g of fresh soil and oven drying at 105 °C for 24 h [47]. Soil pH was measured using an HQ 440d Multi-Parameter Meter (HACH, Loveland, CO, USA) from soil solutions with 1:2.5 g sieved (<2 mm) soil and deionised water, shaken for 1 h in a horizontal shaker at 120 cycles min^{-1} and left overnight to equilibrate [47]. pH and electrical conductivity (EC) of the WW were measured using the same meter. Total C and N concentrations of sieved (<2 mm) soils were determined by total combustion in a C: N analyser (LECO CN828, St. Joseph, MI, USA). Microwave digestion was used prior to analyses of the following element concentrations: Na, Mg, P, K, and Ca. Digestion was carried out using 5 mL HNO_3 reagent with a 1:50 Milli-Q dilution on 0.5 g of sieved (<2 mm) soil and 0.5 g of WW. Element concentrations in the digests were determined using inductively coupled plasma mass spectrometry (ICP-MS, 7500cx, Agilent Technologies, Santa Clara, CA, USA). Standard reference materials (SRMs) were included at 30 sample intervals, blanks were analysed during each digestion batch (15 sample intervals), and duplicates and triplicates were included at 10 and 20 sample intervals, respectively. WW samples were run in triplicate. Recoveries of SRMs ranged from 93 to 132% with the exception of one SRM where recovery was 150 and 202% for K and Na, respectively. Laser diffraction measurements were undertaken for each soil sample to determine particle size distribution (PSD) using a high-definition particle size analyser (Micrometrics Saturn DigiSizer II, Norcross, GA, USA). Total organic carbon (TOC) concentrations in the WW were determined using IR detection combustion analysis (Elementar TOC-TN_{b} $_{C}$ube analyzer, Hanau, Germany).

2.4. Data Analyses

Statistical analyses were performed using R version 4.3.2 [48] with the residuals plotted to test the assumptions of data normality and homoscedasticity. Data were log10, sqrt or cube root transformed where assumptions were not met. A two-tailed unpaired *t*-test was used to compare element concentrations at each depth, for each site individually. A three-way analysis of variance (ANOVA) was carried out for soil variables with depth, irrigation and site used as independent variables with interactions. Where there were significant differences ($p < 0.05$) between group means, Tukey's HSD (honestly significant difference) post-hoc tests were calculated using the package multcomp, version 1.4-25 [49]. Data visualization was made using the package ggplot2 version 3.5.1 [50].

3. Results

3.1. WW Volumes and Loading Rates

WW characteristics between Site A, B and C were consistent with the most notable variation occurring between the autumn and spring sampling rounds. With the exception of pH (which was more alkaline during the spring sampling period than in autumn), the highest WW analytical results were recorded during the autumn 2022 period (Table 1). Total P concentration at Site B was 1.1 mg/L in WW during autumn, while P concentrations were <0.3 mg/L for both sampling periods at Sites A and C. The mean SAR and PAR values were 16.0 and 5.7 $mmol_c$ L^{-1}, respectively. There were no visual signs of ponding or runoff following WW application at any of the sites during the sampling periods.

Table 1. Winery wastewater discharged to land at Site A, B and C from October 2021 and April 2022, excluding Site B April 2022, standard error of the mean in brackets, $n = 5$. A WW application rate of 670 mm yr^{-1} has been used to calculate nutrient loading rate.

Analyte	Mean Concentration	Range	Nutrient Loading Rate (kg ha^{-1} yr^{-1})
pH	7.1 (0.5)	5.5–8.2	-
Electrical conductivity (dS/m)	2.2 (0.3)	1.2–3.1	-
Total organic carbon (mg/L)	597 (178)	91–1020	4000
P (mg/L)	0.1 (0.1)	0–1.1	0.7
Na (mg/L)	265 (21.1)	108–433	1776
Mg (mg/L)	2.8 (0.4)	0.4–5.5	19
Al (mg/L)	10.0 (1.0)	3.1–19.8	67
K (mg/L)	163 (19.5)	74.5–280.0	1092
Ca (mg/L)	16.3 (2.0)	3.9–29.7	109
Sodium Adsorption Ratio ($mmol_c$ L^{-1})	16	14.1–19.2	-
Potassium Adsorption Ratio ($mmol_c$ L^{-1})	5.8	5.7–7.3	-

3.2. Properties of Soils under WW Irrigation

The irrigation of WW increased the pH across all sites and depths compared to values for the controls (Figure 2). The most marked increases were at Site C where all soils across all depths within the WW irrigated plot were alkaline (pH > 7). Total soil C and N concentrations within WW irrigated soils showed significant decreases at Site C compared to the controls, while a significant increase was recorded in soil C at Site A at the 300 mm depth. Soil N concentrations were below the laboratory detection limits for a number of the subsoil (300–600 mm) samples collected, hence, standard errors are not shown. Significant increases in total soil P concentrations were measured in subsoils (300–600 mm depth) at Sites A and C and this trend was observable but not significant at Site B where variances were high.

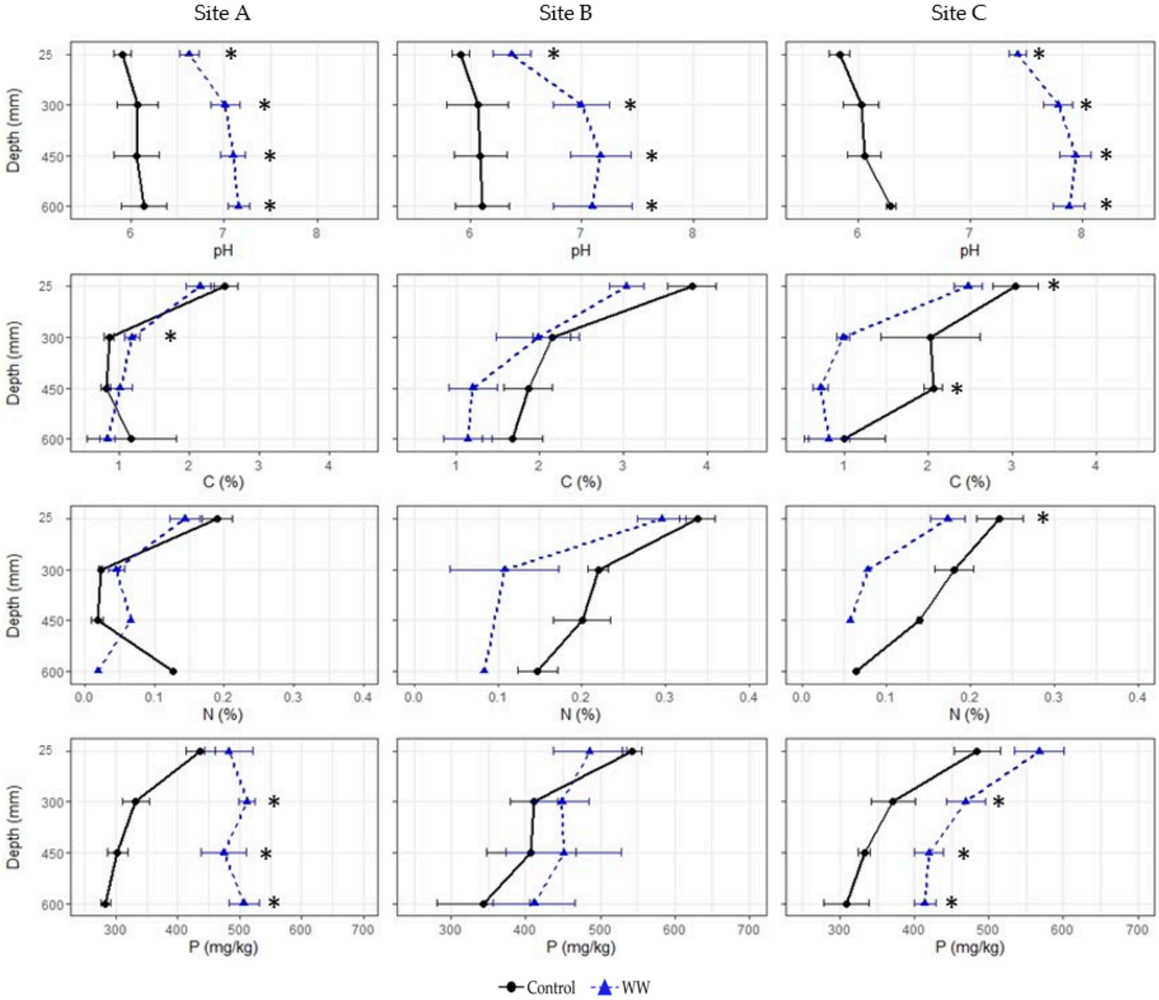

Figure 2. Concentrations of pH and total elements in the soil profile (0–600 mm depth) at Site A, B and C in spring 2021 for WW irrigated plots and non-irrigated control plots. Values shown are means and standard errors ($n = 10$ at 0 mm depth, $n = 5$ for 300–600 mm depth). Asterisks indicate significant differences between treatments at $p \leq 0.05$ using a two-tailed unpaired *t*-test.

Under WW irrigation, concentrations of Na were greater in soils at all depths compared with the non-irrigated plots at Sites A and C (Figure 3). Significant differences in Na concentration were measured between WW irrigated soils and controls at the surface (0–50 mm depth) for Site B, and this trend was observable but not significant in for the remaining depths where variances were high, similar to that for P concentrations. There were no significant increases in soil K at any site with the exception of Site C at 450 mm depth (Figure 3). Significant decreases in Ca concentrations were recorded at Sites B and C at 0–50 mm and 450 mm depths, respectively. Significant decreases in Mg concentrations were recorded at Sites A and B at 0–50 and 450 mm depths, respectively.

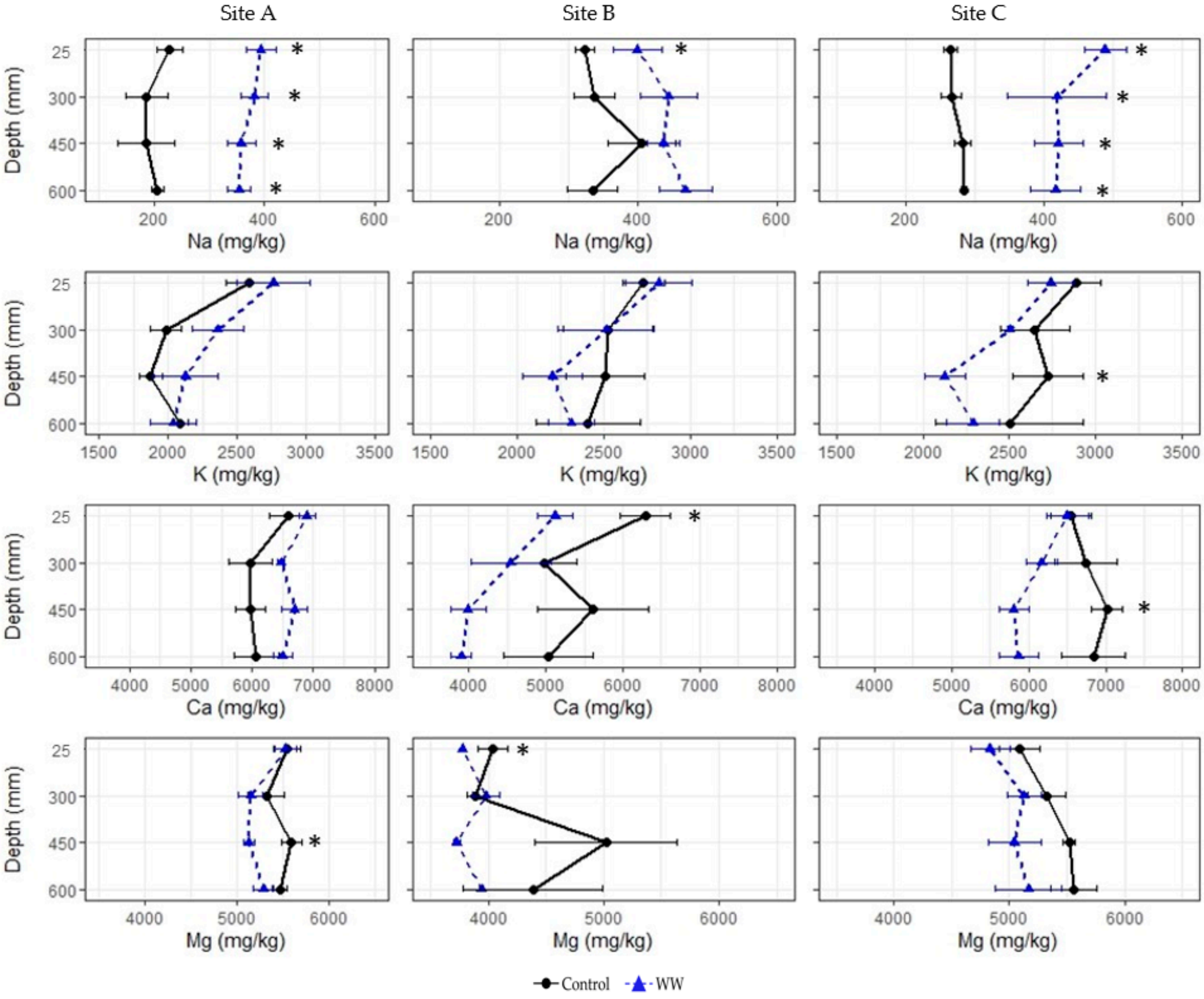

Figure 3. Concentrations of total elements in the soil profile (0–600 mm depth) at Site A, B and C in spring 2021 for WW irrigated plots and non-irrigated control plots. Values shown are means and standard errors ($n = 5$). Asterisks indicate significant differences between treatments at $p \leq 0.05$ using a two-tailed unpaired t-test.

Depth had a significant effect ($p > 0.05$) on all measured soil parameters, with the exception of Na. Significant differences ($p > 0.05$) between sites were identified for all measured soil parameters with the exception of Mg. Significant differences between the interactions of site and treatment were also identified for total soil N, K, Na, Ca and soil pH. Irrigation with WW decreased ($p < 0.05$) soil C concentrations relative to the controls (mean concentrations of 1.7% and 2.3% for the WW irrigated and control soils, respectively). Significant decreases were also recorded in soil N concentration (mean 0.17% and 0.21% for WW irrigated soils and control soils, respectively), although the C/N stoichiometry was consistent between WW treated and control soils, indicating a tight coupling of soil C and N cycling. Significant increases were identified in pH (mean values 7.1 and 5.9 for the WW irrigated soils and control soils, respectively), and Na (mean values 449.3 mg/kg and 281 mg/kg for the WW treated and control soils, respectively). A significant increase in soil P was also identified (mean values 513 mg kg^{-1}, and 418 mg kg^{-1} for the WW-treated and control soils, respectively). However, it is unclear how much of the increased P was available to plants. Significantly lower concentrations of soil Ca were measured in the WW-treated plots (5764 mg/kg) compared with concentrations in the control plots

(6292 mg kg^{-1}) with similar differences in soil Mg concentrations (5167 mg kg^{-1} and 6000 mg kg^{-1} for the treated and control soils, respectively). No significant differences were measured in soil K concentrations (2104 mg kg^{-1} and 2502 mg kg^{-1} for the treated and control soils, respectively).

Significant linear relationships ($p < 0.05$) between soil pH and total soil C, N and Na concentrations were identified when all soil samples were combined (Figure 4). The relationship between soil pH and soil N concentration was significant for all soils however, the negative relationships were only significant ($p > 0.05$) for the WW-treated soils (Figure 4B).

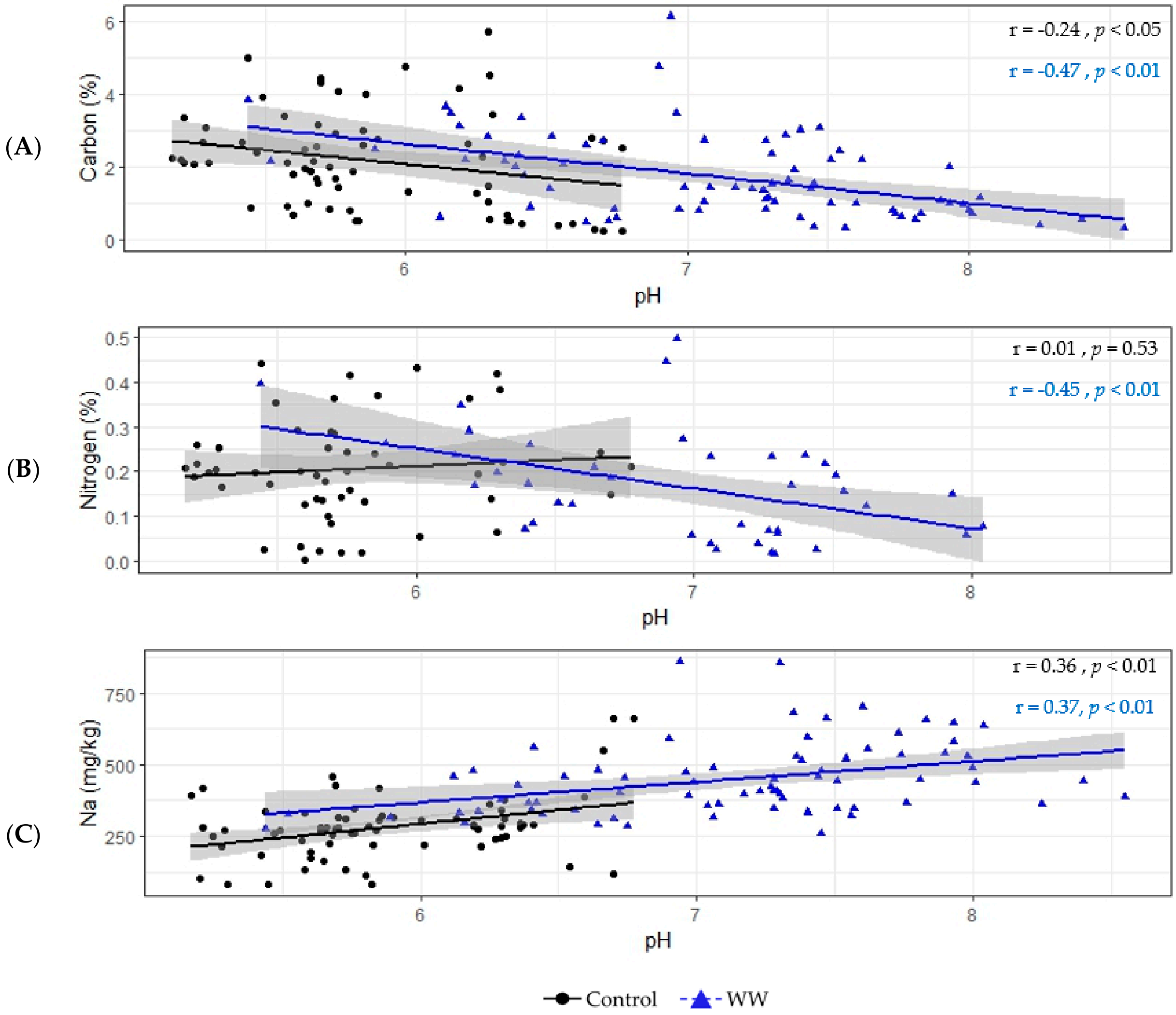

Figure 4. Relationships between soil C vs. soil pH (**A**), soil N vs. soil pH (**B**) and soil Na vs. soil pH (**C**) concentrations at the three sites at 0–600 mm depth. Lines are linear fits for the WW-treated and control soils. The 95% confidence level from the linear models is shown in grey.

4. Discussion

Due to the increases in soil pH and Na, the differences between WW-treated soil and control soil indicate the effects are likely attributable to mineral concentrations within WW as opposed to the increased water supply between treated soils and the non-irrigated control soils. However, the significant differences identified between sites, depth and interactions of site and treatment reflect the complexity of soil interactions and the subsequent accumulation of nutrients from WW. Plant–soil–microbe interactions are likely important factors in the rhizosphere [15,51] and physio-chemical differences such as soil texture [52] and mineralogy will alter accumulation rates between sites and with depth.

The elevated pH identified in WW-treated soils is likely due to the hydrolysis of sodium-based cleaning products [53], evidenced by the correlation identified between soil pH and Na. While the pH of the discharged WW is important, elevated Na concentrations from bicarbonate products have been shown to cause soil pH to increase, even in wastewaters with acidic to neutral pH [54]. The significant increases in soil Na are concurrent with the mean SAR values of 16.0 $\text{mmol}_c\ \text{L}^{-1}$ at the WW irrigated sites and the observed increases in soil Na concentrations could therefore have a large effect on the availability of plant nutrients and plant growth [20], regardless of the pH at the time of discharge [7,9,29]. Studies have shown that salt accumulation and modifications to soil structure can become evident when SAR and/or PAR values exceed approximately 6 and 10 $\text{mmol}_c\ \text{L}^{-1}$, respectively [55]. This is of note as the minimum SAR recorded from the spring sampling was >6 $\text{mmol}_c\ \text{L}^{-1}$, suggesting Na accumulation that could result in soil dispersion occurs in both spring and autumn.

The soil K findings were consistent with the mean PAR values of 5.8 $\text{mmol}_c\ \text{L}^{-1}$, however, this does contrast with previous reports in Marlborough where exchangeable K (me/100 g) from four WW irrigated sites recorded average increases of 1.3 me/100 g between 1999 and 2011 when compared to control soils [7]. Observable accumulation of K may not be occurring due to competition with Na as shown by the elevated SAR values. These results provide strong recommendations for wineries to reduce their use of caustic cleaning products and to use alternative cleaning chemicals such as potassium hydroxide (KOH) in place of NaOH, resulting in minimising Na concentrations in the final WW effluent [9]. The practice is currently uncommon as the cost of potassium-based products is approximately five to ten times higher than that of Na products [26,56]. However, the findings from this study suggest the potential benefits may offset the cost by subsequently decreasing the soil pH and increasing soil quality by reducing the Na accumulation. The use of soil amendments such as gypsum [57] and polyacrylamide [58] to reduce soil Na is also feasible. However, the application of certain polyacrylamide products can sometimes only provide a short-term solution (1–2 weeks) and application rates often need to be high, particularly using gypsum (tons per hectare) [58].

The decrease in soil C concentrations relative to the controls (mean concentrations of 1.7% and 2.3% for the WW irrigated and control soils, respectively), requires further investigation as the findings contradict earlier studies that recorded significant increases in soil C concentration beneath vine rows [25] and pasture soils [15] under WW irrigation. Significant increases in total organic soil C concentrations were also found at long-term WW disposal sites in Australia [3,59]. However, the increases in one of the soil types may be attributable to the different land use between the soils receiving WW (tree-lot) and the control (permanent grassland). Further analysis using synthetic WW labelled with ^{14}C-lactic acid and glycerol [3,59], found that adsorption of ^{14}C-compounds to soil particles resulted in the removal of 10 to 50% of the added ^{14}C within the first hour after application, while microbial uptake and metabolism were responsible for the removal of less than 5% of ^{14}C. Therefore, the fate of the organic C in WW also warrants further investigation considering the high mean TOC loading rates of 4000 kg $\text{ha}^{-1}\ \text{yr}^{-1}$ at our sites (Table 1).

While the correlations identified between soil pH and concentrations of soil C, N and Na do not necessarily indicate that these were attributable directly to pH, the findings are helpful in identifying recommendations to wineries for practical changes that could be made during WW treatments to potentially improve the retention of key macronutrients. In the case of pH and soil C concentration, the dissolution of organic matter at high pH is common [53], and this is likely attributed to the significantly lower values of soil C. Conversely, soil acidification is considered to be an important contributor to soil C accumulation [60], so the negative relationship suggests that while WW irrigation resulted in significantly lower values of soil C concentration compared with the controls, the effect could be less pronounced or even reversed if soil pH were lowered (<7) within WW receiving soils. The direct adsorption of organic C to soil minerals has also been shown to result in a negative relationship with pH value [61], considering the previous research findings that 10 to 50% of added ^{14}C in WW can be adsorbed to soil [3,59], this suggests that the losses in soil C may be partially attributable to decreases in the mineral bound organic carbon content. Further investigation is needed to determine the processes regulating the absorption of organic C onto soil minerals as this is an important mechanism for the long-term protection of soil C [62,63], while also filtering potentially harmful organic constituents such as fungicides [64] and phenolic compounds [65].

The industry could also stand to benefit from further investigation to address how WW discharge to land accumulates available forms of nutrients P, Mg, K and Ca to potentially promote sustainable vegetation growth. A possible alternative approach for the disposal of WW is to irrigate tree species for woody biomass production that could provide co-benefits for increasing carbon stocks and offsetting agricultural greenhouse gas emissions and potentially generate financial incentives for vineyard owners through participation in New Zealand's emissions trading scheme (ETS). Mean annual rates of carbon sequestration are up to 30 t C ha^{-1} yr^{-1} for native shrub planting within the first 20 years [66]. The effects of irrigating soils with WW may not affect the growth rates of New Zealand native plants as many are adapted to low fertility environments [67]. For example, while the presence of organic compounds and salts in WW could be expected to reduce plant growth [15,68], the salt content and organic loading may not be too dissimilar to the brackish conditions where many New Zealand native species such as *Phormium tenax*, *Coprosma propinqua* and *Cordyline australis* thrive [69]. The effect of WW irrigation on New Zealand native plants is currently unknown, however, as certain native plants have been shown to increase biodiversity, suppress weeds and increase microbial activity in New Zealand vineyards [70], further investigation is warranted.

5. Conclusions

Improvements to the discharge of WW to land through the effective assimilation of nutrients using well-managed irrigation systems may lead to additional benefits to New Zealand's wine industry. Our findings clearly recommend the use of potassium-based caustic chemicals over sodium-based chemicals by wineries to reduce environmental impacts. Additionally, the mean decreases in soil C concentration between WW treated and control soils (approximately 1.5%), is equivalent to the loss of carbon stock of approximately 100 tC ha^{-1}, assuming the change in bulk density is consistent with 600 mm depth in the soil profile. There is a further addition of 4 t C ha^{-1} yr^{-1} in the WW. By focusing on the retention of C and promoting sustainable growth of vegetation, biomass and soil carbon stocks could be increased in an effort to offset agricultural greenhouse gas emissions. By incorporating native vegetation growth within WW irrigation plots, multi-purpose restoration could be encouraged with increases in biodiversity and enhancing cultural significance.

Author Contributions: Conceptualisation, B.R., D.W. and M.N.-M.; methodology, B.R. and M.N.-M.; formal analysis, B.R. and M.N.-M.; investigation, B.R. and M.N.-M.; writing—original draft preparation, M.N.-M.; writing—review and editing, B.R., D.W., B.M. and T.C.; supervision, B.R., D.W., B.M., and T.C.; project administration, B.R. and M.N.-M. All authors have read and agreed to the published version of the manuscript.

Funding: This research was undertaken as part of the Food Transition 2050 partnership (https://www.foodtransitions2050.ac.nz/) initiative where M.N.-M. is pursuing a Doctoral Scholarship from the University of Canterbury.

Data Availability Statement: The data presented in this study are available on request from the corresponding author.

Conflicts of Interest: The authors declare no conflicts of interest.

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