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Structure Lime as a Soil Amendment: Impacts on Nutrient Loss Risk and Soil Health

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ABSTRACT

We investigated the impact of structure lime (SL) on soil structural stability and phosphorus (P) loss risk from fine-textured mineral soils, as well as its effects on soil fertility, bacterial and fungal communities, and soil carbon (C) dynamics. Effects on erosion and P loss risks were studied utilising rainfall simulation after laboratory incubation of 14 soils with three SL addition levels and untreated control. In addition, soil samples were collected from six fields that had received a single SL treatment between 1 and 6 years prior to sampling and were compared with adjacent untreated control areas. Soil samples from the plough layer of SL-treated fields were analysed for plant-available nutrient contents and subjected to DNA sequencing. Further, the total C content as well as bulk density (BD) were determined down to 40 cm. Rainfall simulation of the laboratory incubated soils showed that SL effectively reduced turbidity and particle-associated P (PP) concentration of the drainage water, and the reduction was largest in soils with a high risk for colloid dispersion due to low electrical conductivity. Dissolved reactive P (DRP) concentration of the drainage water was unaffected by SL treatment. However, in SL-treated soils, an increase in dissolved organic matter (DOC) concentrations in rain simulation, and higher C content at 30–40 cm depth in field soils were observed. As expected, the microbial communities differed according to soil depth, but they did not exhibit community-level changes due to SL; only a few taxa-specific alterations in bacteria and fungi were observed. Treatment with SL decreases particle dispersion on clay soils with low EC but may increase DOC losses.

1 | Introduction

Structure lime (SL; materials including quicklime and/or slaked lime, CaO or Ca(OH)₂, respectively) has been for long used to stabilise fine-textured soils in highway construction sites (McDowell 1959), but later it has also been found to counteract losses of particle-associate phosphorus (PP) from clayey agricultural soils through enhanced aggregate stabilisation (Blomquist et al. 2018; Svanbäck et al. 2014; Ulén and Etana 2014). The use of SL is promoted as an environmentally beneficial agricultural

management practice, mainly because SL can mitigate erosion and PP transport from fine-textured soils, thereby protecting waters (Ulén and Etana 2014). At high application rates, SL allows for fast boosts of soil pH, which may in favourable soil conditions, result in pozzolanic reactions that produce highly stable soil aggregates (Keiblinger et al. 2016). SL also elevates the electrical conductivity (EC) of the soil solution, which results in a lower repulsion barrier between negatively charged soil particles and the formation of cation bridges upon elevated calcium ion (Ca²⁺) activity. Thus, SL has been found to enhance

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Summary

- Impacts of structure lime on the nutrient loss risk and soil health were studied.
- The structure lime treatment effectively reduced colloid detachment in soils with low EC.
- After the structure lime addition, the increase in pH led to increased mobility of OC.
- At agronomically relevant doses of structure lime, pozzolanic reactions are unlikely to occur.

soil aggregate stability through chemical reactions rather than biological long-term aggregate formation via the activity of soil organisms.

The tendency for stable aggregate formation increases in soils with a clay content above 15% (Horn 1990). However, at high clay contents, the risk of clay dispersion also rises (Kjaergaard et al. 2004). Thus, while high-clay soils have the potential to develop a stable aggregate structure, they are also prone to contributing to surface water eutrophication through losses of PP. According to Keiblinger et al. (2016), quicklime application improves aggregate stability more efficiently in soils with high clay content and cation exchange capacity, indicating that SL could significantly contribute to the reduction of P loss from arable clay soils. In addition to clay content, the long-term stabilisation of clay soil structure with quick lime depends on the soil mineralogy, pH and the amount of lime added (Akula and Little 2020), the temperature (Rao and Shivananda 2005), and on the particle size and the specific surface area of the particles (Walker and Pavia 2011).

Application of SL reduces soil acidity, which affects the availability of plant nutrients and is beneficial for agricultural soils under sub-optimal growth conditions ascribed to low pH. Improved productivity due to liming leads to increased input of plant residuals and may contribute to C inputs in soil (Paradelo et al. 2015). However, reduced plant availability of micronutrients has been reported after liming (Haynes 1984; Kirchmann and Eskilsson 2010), with yield depression associated with their decreased solubility and consequently, lower uptake by plants. Conversely, an increase in pH is known to enhance desorption of P from mineral surfaces and thus increase its concentration in soil water (Hingston et al. 1967) and availability for plants. Concurrently, higher P concentration in soil solution increases the risk of dissolved reactive P (DRP) losses to surface waters (Heckrath et al. 1995; Withers et al. 2019). Similarly, an increase in pH may enhance the solubility of organic matter (OM) (Curtin et al. 2016), leading to increased dissolved organic carbon (DOC) losses. Changes in pH are also a main driver for soil biodiversity (Zhou et al. 2020), particularly in structuring bacterial communities, whereas fungi are found to be less sensitive to changes in pH (Rousk et al. 2010). Because microbes affect soil properties such as aggregation (Costa et al. 2018), manipulation of soil pH may have an indirect link to particle aggregation via microbial community adaptation.

Comprehensive understanding of the mechanisms underlying the impacts of different SL doses, and the potential long-term persistence of these effects, is still lacking. In particular, there

is a significant knowledge gap regarding how SL influences microbial community composition over several years (Palmu and Hedlund 2016). Here, we report the results from a laboratory incubation experiment where the effects of different SL application levels on drainage water quality after rainfall simulation were studied with fine-textured surface soils. Further, we studied the impact of SL on soil nutrient status, P mobility, soil structural stability and the soil bacterial and fungal community in structure limed fields 1–6 years after application. In addition, we measured soil C in the 0–40 cm depth profile to gain information on the possible treatment effects. We hypothesised that (i) the higher the clay content in soil, the higher SL dose would be required for reduced erosion susceptibility, (ii) SL application improves soil aggregate stability, (iii) the influence on microbial community is mainly related to SL-induced rise in soil pH and (iv) that the increase in soil pH enhances OM solubility, resulting in a decrease in soil OM content and/or increased mobility in the soil profile.

2 | Materials and Methods

2.1 | Laboratory Incubation With Rainfall Simulation

For the laboratory incubation experiment, composite soil samples from the 0 to 10 cm soil layer were collected from 14 locations known to differ in clay content (Table 1). Soils were derived from agricultural fields in Southern Finland, within ca. 200 km radius. The fields in this area are characterised by a relatively low slope (ca. 2%) and a low estimated mean erosion rate of 560 kg ha⁻¹ yr⁻¹ (Lilja et al. 2017). Samples were passed through a 10 mm sieve in field-moist conditions, carefully mixed to homogenise, after which their water holding capacity (WHC) was determined using the funnel method. In short, ca. 50 g of dry soil was weighed into a funnel lined at the bottom with 0.3 g of glass wool to prevent soil loss. The soil was saturated with 100 mL of deionised water and allowed to stand for 30 min to ensure full saturation. The funnel was then left to drain freely for 24 h. WHC was calculated as the difference between the volume of water added and the volume of water drained, correcting for the water retained by the glass wool (0.3 g). Air-dried subsamples were analysed for texture according to Elonen (1971), and plant available nutrients according to Vuorinen and Mäkitie (1955), involving extraction with pH 4.65 ammonium acetate-acetic acid solution (Ac). Total C and nitrogen (N) were analysed with LECO628 analyzer (Leco Corp., St. Joseph, MI, USA) and pH and electrical conductivity (EC) were measured from soil–water suspension (v:v 1:2.5).

The SL used contained 26% of CaO/Ca(OH)₂ and 74% of CaCO₃. It was mixed into soils at equal SL-to-soil volume ratios after measuring the bulk density (BD) of moist, sieved soil. A 900-mL volume of each soil was measured into four pots assembling the following treatments: (i) control without SL (control), and three treatments with increasing SL application rates of (ii) 3.8 Mg ha⁻¹ (SL1), (iii) 7.7 Mg ha⁻¹ (SL2) and (iv) 11.5 Mg ha⁻¹ (SL3) assuming a mixing depth of 10 cm in the field. These applications concurred 1, 2 and 3 Mg ha⁻¹ CaO/Ca(OH)₂, respectively. SL was carefully mixed with the whole sampled volume of soil, and unamended control soils were mixed similarly without lime additions. Based on the WHC measurement done for each soil, soil moisture was adjusted to 60% WHC and

TABLE 1 | Selected properties of soils included in incubation study.

Soil	%			N	C	pH	EC	P-Ac
	Clay	Silt	Sand				mScm ⁻¹	mgkg ⁻¹
A	16	12	72	0.23	2.8	7.1	0.17	61
B	20	12	68	0.14	2.1	6.4	0.94	11
C	37	16	48	0.18	2.5	7.1	0.13	15
D	49	19	32	0.25	3.1	6.3	0.98	19
E	54	20	26	0.24	3.0	6.5	0.10	8
F	61	19	20	0.36	4.0	5.9	0.17	8
G	63	16	21	0.29	3.8	6.2	0.16	9
H	67	18	15	0.25	3.1	6.2	0.83	4
I	73	12	15	0.33	2.6	6.0	0.06	5
J	74	15	11	0.78	10.8	4.5	0.13	17
K	75	8	17	0.34	3.6	6.2	0.12	8
L	77	9	14	0.33	4.1	5.9	0.09	3
M	77	15	8	0.3	3.9	5.5	0.19	8
N	81	8	11	0.36	4.2	6.3	0.07	4

maintained constant during the incubation. After 5 weeks of incubation, a composite sample of 50 mL was carefully taken from the pots for pH and EC measurements and for aggregate stability analysis. The rest of the incubated soil (850 mL) was reserved for rainfall simulation (described below).

To measure the share of water-stable aggregates (WSA%), 4g (dry matter) from each sample was subjected to wet sieving (Heikkinen et al. 2019), using an Eijkelkamp 08.13 apparatus equipped with 0.25-mm mesh size sieves. The aggregates placed on the sieves were pre-moistened for 15 min in cans containing 100mL of deionised water. The pre-moistening was followed by a 3-min up-and-down run with the wet sieving apparatus. Water-stable aggregates remaining on the sieves were oven-dried (110°C) and weighed. The proportion of WSA% was calculated by dividing the weight of the material on the sieves by the total weight of the initial material.

Approximately 850mL of the incubated soils was packed into 15cm diameter cylinders, on top of nylon netting and a layer of quartz sand, from which water percolating through the soil was collected. A stationary drop-former was used in the rainfall simulation tests (Uusitalo and Aura 2005). The rainfall intensity was adjusted to 5 mm h⁻¹ (deionised water with kinetic energy about 80Jm⁻²h⁻¹), which is an intensity that occurs annually during storm events in SW Finland. The samples were subjected to rainfall until a volume of 100 mL of drainage water was obtained. Small subsamples of drainage water were used for immediate analysis of turbidity (NTU; 2100AN IS Turbidimeter, Hach Company). The rest of the samples were then subdivided into two fractions, and a part of the water sample was filtered through a 0.2-µm Nuclepore (Whatman) membrane. Both filtered and unfiltered fractions were frozen and stored at -18°C until further analyses.

The filtered samples were analysed for dissolved molybdate-reactive phosphorus (DRP). Following autoclave digestion with peroxydisulfate and sulfuric acid, the unfiltered samples were analysed for total P (TP) concentration. The P concentrations were determined using a continuous flow injection analyser (LaChat 8000 Quickchem), and particle-associated P (PP) concentration was calculated as a difference between TP and DRP. Any dissolved unreactive P, if present, was thus included in the PP fraction. Before analyses for DOC with a Shimadzu TOC analyzer, unfiltered subsamples were passed through Whatman GF/C glass filters.

2.2 | Properties and Structural Stability of Structure Limed Fields

In 2019, we collected samples from six fields located in Southern Finland that were treated with SL once between the Years 2013 and 2018 (Table 2). SL had been applied on the soil surface in the autumn and, within a couple of days, mixed into surface soil (10 cm). Each of the fields had a part with similar field characteristics that was left untreated and served as a control area. The SL and control areas of the fields had similar farming histories and were cultivated alike after the SL treatment. The clay content in the plough layer (0–20 cm) varied between 29% and 85% (Table 2). In Table 2 are given the years passed since the SL treatment, with sand (> 0.02 mm), silt (0.002–0.02 mm) and clay (< 0.002 mm) percentages (%) in 0–20 and 20–40 cm soil layers. The composition of the SL differed between the fields. One field (number III) was treated with SL originating from side streams of pulp and paper industries containing ca. 45% of CaO/Ca(OH)₂, whereas the other fields were amended with SL containing 15%–25% of CaO/Ca(OH)₂. The total amount of SL applied was 6000 kg ha⁻¹

TABLE 2 | Number of years since structure lime (SL) treatment and soil particle size distributions in control and SL-treated areas of the six studied field parcels.

Field No.	Years since SL treatment	Soil layer	Control			Structure lime–treated		
			%			%		
			Sand	Silt	Clay	Sand	Silt	Clay
I	6	0–20 cm	14	34	52	15	32	53
		20–40 cm	9	25	66	6	15	79
II	6	0–20 cm	20	29	51	21	23	56
		20–40 cm	8	20	72	15	21	64
III	4	0–20 cm	30	35	35	37	34	29
		20–40 cm	25	38	37	30	34	36
IV	3	0–20 cm	22	31	47	22	30	48
		20–40 cm	9	21	70	16	21	63
V	1	0–20 cm	15	18	67	20	15	65
		20–40 cm	12	24	64	10	5	85
VI	1	0–20 cm	6	29	65	10	30	60
		20–40 cm	15	26	59	6	35	59

for the SL originating from the side stream of the pulp and paper industry and 7000–10,000 kg ha⁻¹ for the other types of SL. The amount of CaO/Ca(OH)₂ applied in the different fields thus varied between 1.8 and 2.5 Mg ha⁻¹.

From each field, from both structure limed and control areas, a composite sample composed of 10 subsamples of the 0–15 cm layer was collected with a spade for rainfall simulations. Soils were sieved (5 mm), and 900 mL was packed into 15 cm diameter cylinders and subjected to rain simulation, and drainage water samples were analysed as described above.

To examine the effect of SL on aggregate stability, the uppermost surface soil (0–5 cm) was sampled from SL-treated and control areas in each of the six fields. Samples were taken with a 200 cm³ cylinder, and three subsamples from each SL and control area were combined to form a composite sample. Samples were kept at sampling moisture and dry sieved to retain an aggregate fraction of 2–5 mm. The sieved aggregates were air-dried and analysed for WSA as described above.

2.2.1 | Soil C and Fertility

From each field, four separate 0–40 cm profile samples were cored from both the control and SL-treated areas with a 4.8-cm diameter auger. Each soil core was cut into 10-cm length segments, dried at 40°C, and weighed to determine the BD of the given soil layers. Thereafter, the samples were ground to pass a 2-mm sieve, and all profiles and depths were separately analysed for total C and N via dry combustion (Dumas method, Leco TruMac CN).

The soil C stock (kg m⁻²) was calculated using the equivalent soil mass (ESM) methodology as in Heikkinen et al. (2021), which

takes into account variation in soil BD and is less sensitive to soil compaction than volume-based sampling. In the method, soil mass in a square meter area is calculated in 100 kg increments, multiplied by the corresponding C contents and summed up to the desired soil mass. We used 500 kg soil mass in summing the C stock; in the six fields studied, this mass was obtained by summing up soil to average depths between 33 and 36 cm.

For 0–10 and 10–20 cm layers, extractable P, Ca, Mg, K, Na and S were measured in accordance with the Finnish national soil test procedure (acid ammonium acetate, Ac and extraction; Vuorinen and Mäkitie 1955). The soils were also analysed for P with the Mehlich-3 extraction (PMeh, Mehlich 1984). P-Ac was determined using the molybdate colorimetric method (Murphy and Riley 1962) with a Skalar San++ continuous flow analyzer (Skalar Analytical B.V.) and for PMeh, with an inductively coupled plasma emission spectrometer (iCAP 6500 Duo, Thermo Fisher Scientific) (Sims 2000). Micronutrients (Cu, Fe, Mn and Zn) were analysed after Ac-EDTA extraction (Lakanen and Erviö 1971), and B after hot water extraction (Berger and Truog 1944) with ICP-OES (PerkinElmer Optima 8300) measurement of the elements. Results are expressed on a dry matter (oven dry, 105°C) basis.

2.2.2 | DNA Extraction, Soil Microbial Diversity and Bioinformatics

For microbiological analyses, samples were collected from the 0 to 10 and 10 to 20 cm layers of each SL and control area (composite samples of 10 subsamples). After the sampling and prior to the analyses, samples were stored refrigerated (+4°C).

DNA extraction was performed using the DNeasy PowerSoil Pro Kit (Qiagen) from 250 to 280 mg of soil samples. Amplicon

sequencing at Tartu University's Institute of Genomics was done with Illumina MiSeq V3 using the 16S rRNA 515F and 806R primers (Caporaso et al. 2011, 2012) for bacteria, and primers ITS4 (White et al. 1990) and gITS7 (Ihrmark et al. 2012) for the fungal internal transcribed spacer 2 (ITS2) region.

Bioinformatics was done according to Soenne et al. (2020) with truncal 10 for both bacteria and fungi. Annotation against databases silva_138_1_ssuf_nr9 (Quast et al. 2013; Yilmaz et al. 2014) and sh_general_release_04.02.2020 (Nilsson et al. 2019). During secondary quality filtering, OTUs (operational taxonomic units) that had an *e* value higher than e^{-25} and identity < 80% with the database match were removed. Furthermore, OTUs with an affiliation other than bacteria or fungi, as well as singleton OTUs and OTUs with less than 10 reads, were removed from the data.

2.3 | Statistical Analyses

For the laboratory incubation experiment, the statistical analyses on the effects of SL on soil pH, EC and drainage water quality (turbidity, TP, PP, DRP and DOC) after rain simulation were tested with analysis of variance with SL treatment as a fixed effect and soil ($n = 14$) as a random effect. Homogeneity of variance was tested with Levene's test, the normality of residuals was checked using boxplots, and the residuals were plotted against the fitted values. Logarithmic transformation was applied to the turbidity, TP, PP, DRP and DOC results to meet the normality requirements. The Tukey–Kramer test was used for pairwise comparisons of treatment means, except for PP, where the assumption of homogeneity of variances was not met despite logarithmic transformation. For PP, each SL treatment was compared separately to the control soil using Welch's test, assuming unequal variances. The impact of SL on the share of WSA was analysed using the nonparametric Kruskal–Wallis ANOVA.

For the field samplings conducted at six locations, differences between the SL and control areas were tested using analysis of variance with location included as a random effect, based on the means of the four within-plot replicates. For parameters analysed separately by soil layer (TC and BD), statistical tests were performed independently for each layer. For properties measured from the drainage water after the rain simulation, a logarithmic transformation was applied to DRP to meet normality assumptions. Of the properties measured from the soil samples, TC was also log-transformed. For EC measured from soil samples, total C stock and for TP and PP measured from the drainage water, differences between treatments were tested using the nonparametric Mann–Whitney–Wilcoxon test. Analyses were performed using R Statistical Software version 4.3.1 (R Core Team 2023).

Microbial data analyses were conducted in R version 4.4.1 (R Core Team 2024). Microbial community matrices of relative abundance data for bacteria and fungi were subjected to Permutational Multivariate Analysis of Variance Using Distance Matrices by terms with function *adonis2* from the vegan package (Oksanen et al. 2024) with 9999 permutations, with site, depth, SL treatment and time since treatment used as explanatory variables. Observed richness and Shannon diversity indices were estimated from raw OTU counts. The indicative

bacterial and fungal species for SL or control treatments were studied with Multi-level pattern analysis *multipatt* function from the indicpecies library (Cáceres and Legendre 2009) with alpha level $p < 0.05$. To visualise microbial community data, we used the *metaMDS* function from the vegan package (Oksanen et al. 2024).

3 | Results

3.1 | Impact of SL on Drainage Water Quality After 5-Week Incubation Experiment

The higher the SL dose, the higher the pH and EC values were measured from the soils (Figure 1). The increase in pH between the addition levels SL2 and SL3 was, however, not statistically significant. Turbidity values were lower in the drainage water from SL-treated soils compared to the unamended control soils, but the SL addition levels did not differ from each other. In contrast, PP concentration was only marginally lower in SL1 and SL2 compared to control soils, and for SL3, no difference from the control was detected (Figure 1). Similarly, the reduction in TP concentration after SL treatments was not statistically significant at the 0.05 level (not shown), but SL1 and SL2 differed marginally ($p = 0.0862$ and 0.0593 , respectively) from the unamended control, whereas SL3 did not. SL did not affect the DRP concentration in drainage water, but DOC concentration increased in concert with increasing SL additions (Figure 1). The share of WSA was on average 84% in untreated soils, while in SL-treated soils, the mean ranged between 87% and 89%; the impact of SL on aggregate stability was not statistically significant.

The highest turbidity values were measured from soils with the lowest EC before any SL additions (Figure 2a). For soils with an EC value lower than 0.1 mS cm^{-1} , the SL-induced reduction in turbidity was pronounced (Figure 2d), but for soils with the highest EC values ($> 150 \mu\text{S cm}^{-1}$ before SL treatment), the reduction was minor and inconsistent. The highest reductions in turbidity due to the SL treatment were measured for high clay soils (clay% > 60; cf. Figure 2b,e), but this reduction was nevertheless variable and negligible turbidity reductions were also measured from high clay soils. Similarly, the highest aggregate stabilities (WSA) were measured in high clay soils (Figure 2c), but the change in WSA for these soils following SL application was mostly small (Figure 2f).

3.2 | Structural Stability, and P and DOC Export Risk in Structure Limed Fields

In field soils, where the SL had been applied 1–6 years prior to the sampling, no consistent reductions in turbidity, DRP or TP concentrations in the drainage water were found in rain simulations (Figure 3). Nevertheless, similarly to the laboratory incubation, the highest turbidity values and PP concentrations were measured in soils with the lowest EC values (Figure 4). The concentration of PP in the drainage water followed a similar pattern to turbidity. The DRP concentrations were linked to soil test P concentrations, and the results did not indicate any SL-induced effect on DRP leaching. There was no consistent

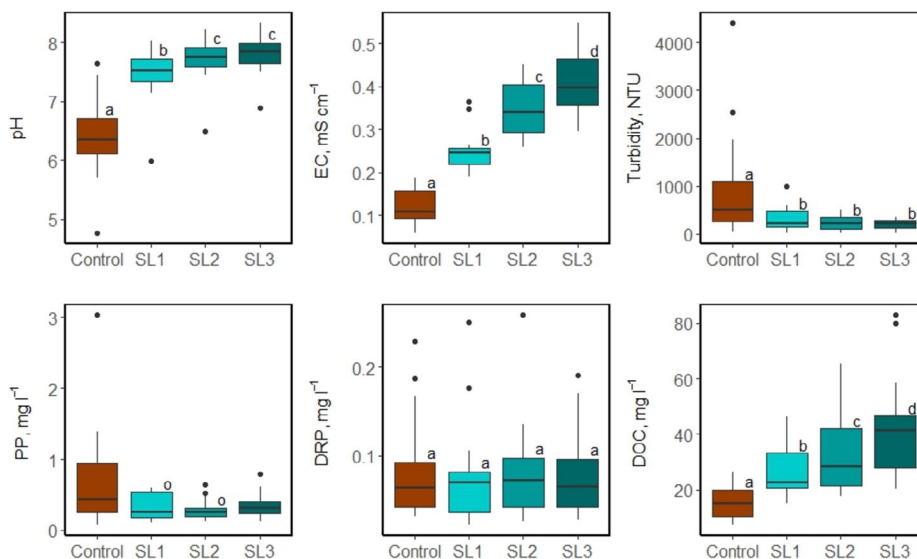


FIGURE 1 | pH and electrical conductivity (EC, mS cm^{-1}) measured from control and structure limed soils (SL1, SL2 and SL3 referring to $\text{CaO}/\text{Ca}(\text{OH})_2$ additions of 1, 2 and 3Mg ha^{-1} , respectively) after 5-week incubation and turbidity (NTU), particle-associated P (PP, mg L^{-1}), dissolved reactive phosphorus (DRP, mg L^{-1}) and dissolved organic carbon (DOC, mg L^{-1}) measured in drainage water after rainfall simulation performed for the incubated soils. Different letters indicate statistically significant differences between the treatments. For PP, the sign o indicates marginally significant differences between the treatment and control.

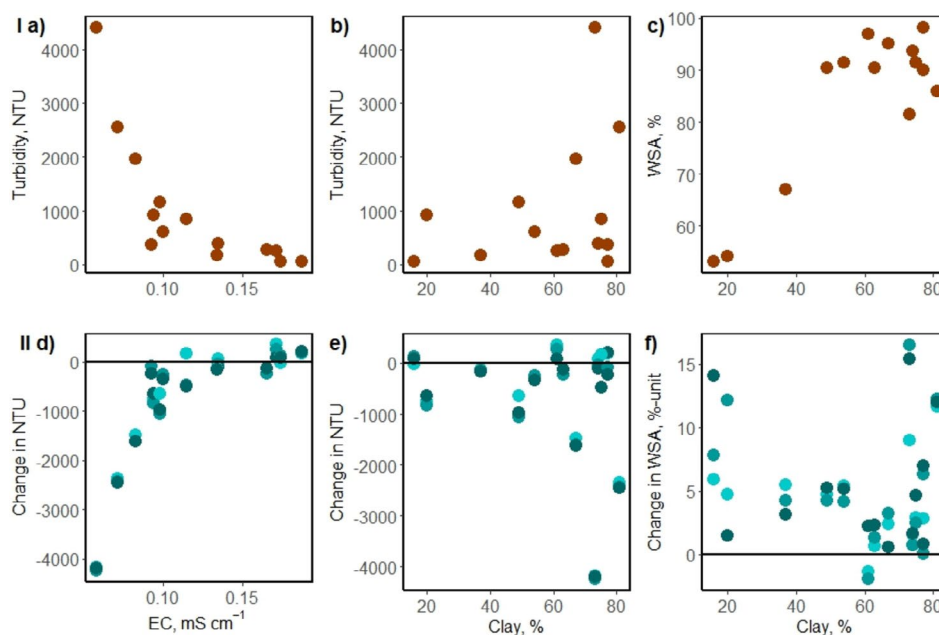


FIGURE 2 | Upper panels: Turbidity (NTU) measured from the drainage water after rainfall simulation of the untreated control soils plotted against (a) electrical conductivity (EC, mS cm^{-1}) of the untreated soils and (b) clay content; (c) water-stable aggregates (WSA, %) plotted against clay content (%). Lower panels: Structure lime induced changes in turbidity (NTU) of the drainage water plotted against (d) EC (mS cm^{-1}) of unamended control soils and (e) clay content (%); (f) structure lime induced change in WSA in percentage (%) units plotted against clay content (%). Treatments SL1, SL2 and SL3 refer to $\text{CaO}/\text{Ca}(\text{OH})_2$ additions of 1, 2 and 3Mg ha^{-1} , respectively.

increase in surface soil WSA in the structure-limed plots compared to the controls. Similarly, pH and soil EC did not differ significantly between the controls and SL-treated areas (for pH, the difference was almost significantly higher in the SL treatment, $p=0.068$; for EC) (Figure 3). Additionally, DOC concentration in the drainage water collected in the rain simulation of the surface layer soils was higher in SL-treated soils than that of the control areas (Figure 3).

3.3 | Soil C and Fertility in Structure Limed Fields

The SL treatment did not affect the surface soil (0–10 and 10–20 cm) C% but tended to increase C% of deeper soil layers. The difference in C% between the control and SL-treated soils was statistically significant ($p=0.015$) in the 30–40 cm soil layer and nearly significant ($p=0.0597$) in the 20–30 cm soil layer. Similarly, BD was lower in the 30–40 cm soil layer in SL-treated

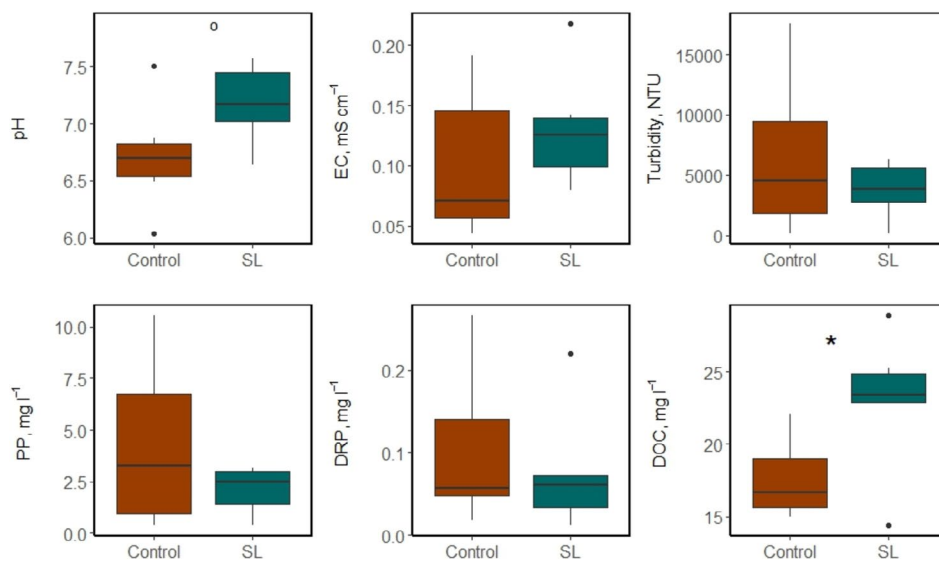


FIGURE 3 | Mean soil pH and soil electrical conductivity (EC, mS cm^{-1}) measured from the soil samples collected from the control and structure limed (SL) areas of the six separate fields (0–15 cm) and turbidity (NTU), particle-associated P (PP, mg L^{-1}), dissolved phosphorus (DP, $\text{PO}_4\text{-P}$; mg L^{-1}) and dissolved organic carbon (DOC mg L^{-1}) measured from the drainage water after rain simulation of the surface (0–15 cm) soil.

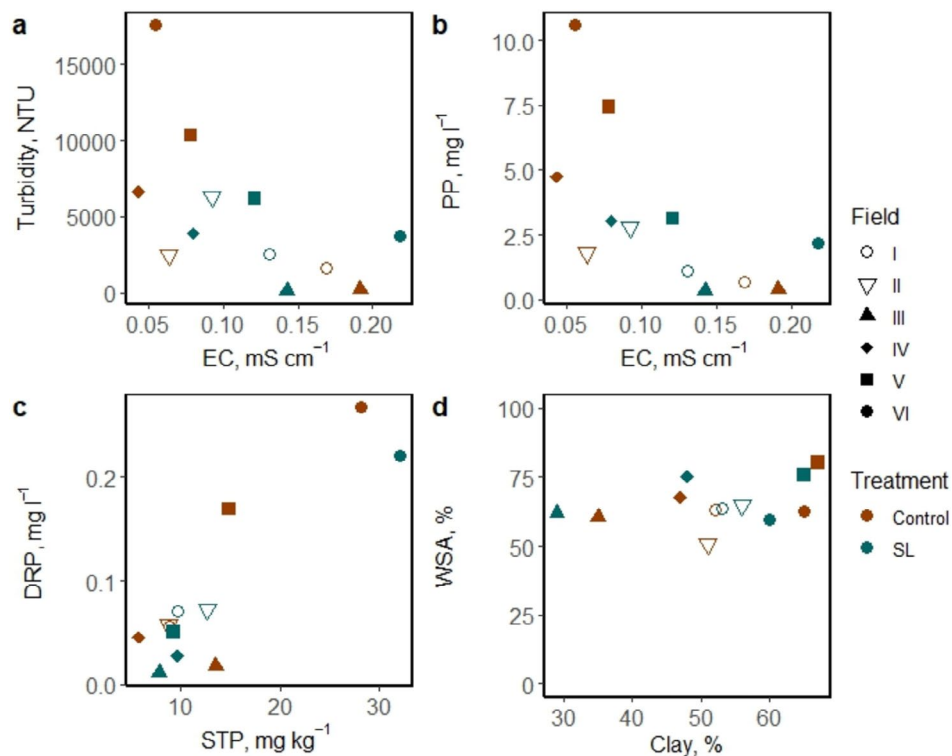


FIGURE 4 | (a) Turbidity (NTU) and (b) concentration of particle-associated P (PP, mg L^{-1}) in drainage water after rainfall simulation of surface soil (0–15 cm) from control and structure lime (SL)-treated areas of the six separate fields plotted against soil electrical conductivity (EC, mS cm^{-1}) and (c) dissolved reactive phosphorus (DRP, mg L^{-1}) in drainage water after rain simulation of control and SL-treated soils plotted against soil test phosphorus (STP, P-Ac, mg kg^{-1}), and (d) water-stable aggregates (WSA, %) in control and SL-treated soils plotted against clay content (%).

soils ($p=0.038$) (Figure 5). Since lower BD would mean that a given soil mass contains less C than the same soil in higher density, we also calculated the C stock using the ESM method. As for the soil C mass incorporated in the 500 kg m^{-2} mass of soil (in these soils extending down to 33–36 cm depth), there were only minor differences between control and SL-treated areas, with

averages of 11.0 and 11.5 kg m^{-2} for control areas and structure limed areas, respectively ($p=0.589$).

The SL-treated field areas did not have any notably different concentrations of extractable nutrients in the 0–10 or 10–20 cm soil layers compared to control areas (Table 3). The clearest

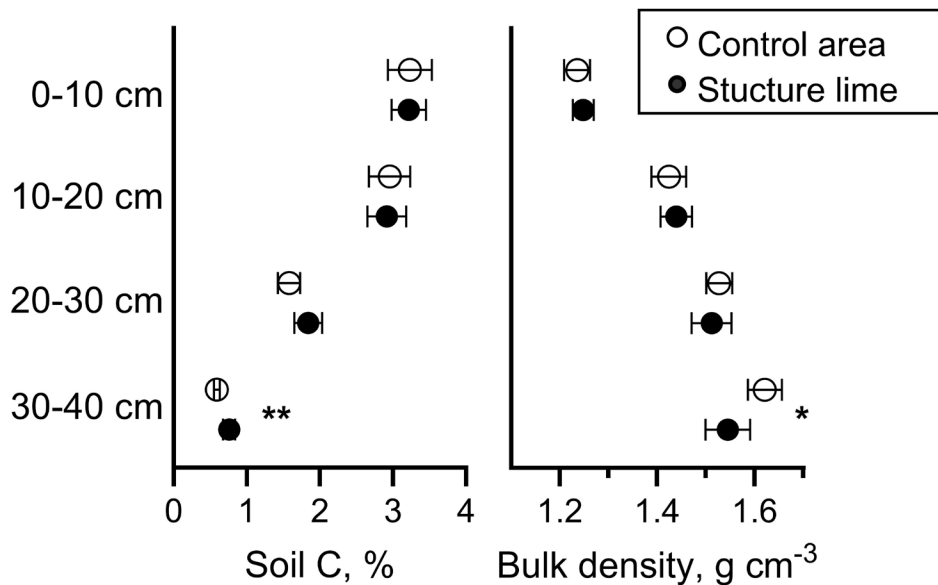


FIGURE 5 | Total carbon (C, %) and bulk density (BD, g cm⁻³) in soil profiles down to 40cm in control and structure lime (SL)-treated areas of the fields. Error bars=SE (*n*=6).

TABLE 3 | Water extractable phosphorus (P_{H2O}) and soil test phosphorus (acid ammonium acetate P, P_{Ac} and Mehlich3, P_{Meh}) and extractable nutrients (acid ammonium acetate extraction) in control and in structure lime (SL)-treated areas of the six separate fields at depths of 0–10 and 10–20cm (SE=standard error, *n*=6).

		Control	SE	SL	SE	<i>p</i>
P _{H2O} mg kg ⁻¹	0–10cm	2.6	0.7	2.1	0.5	0.768
	10–20cm	2.2	0.5	1.9	0.6	0.435
P _{Meh} mg kg ⁻¹	0–10cm	57	11	49	10	0.249
	10–20cm	45	9	43	10	0.721
P _{Ac} mg kg ⁻¹	0–10cm	13	3	13	3	0.948
	10–20cm	11	3	10	4	0.463
Ca mg kg ⁻¹	0–10cm	3314	398	4091	544	0.278
	10–20cm	3025	318	2948	233	0.835
K mg kg ⁻¹	0–10cm	312	64	268	31	0.523
	10–20cm	245	55	216	25	0.668
Mg mg kg ⁻¹	0–10cm	451	79	414	64	0.386
	10–20cm	545	100	528	84	0.763
S mg kg ⁻¹	0–10cm	21.6	7.1	16.0	3.5	0.362
	10–20cm	14.2	2.7	13.4	2.2	0.763
Cu mg kg ⁻¹	0–10cm	5.5	0.9	5.2	0.4	0.688
	10–20cm	6.3	1.1	5.6	0.7	0.406
Fe mg kg ⁻¹	0–10cm	663	98	709	136	0.781
	10–20cm	847	156	812	144	0.822
Mn mg kg ⁻¹	0–10cm	49	7	53	11	0.707
	10–20cm	51	6	51	10	0.944
B mg kg ⁻¹	0–10cm	0.88	0.07	0.69	0.04	0.091
	10–20cm	0.94	0.12	0.72	0.04	0.092

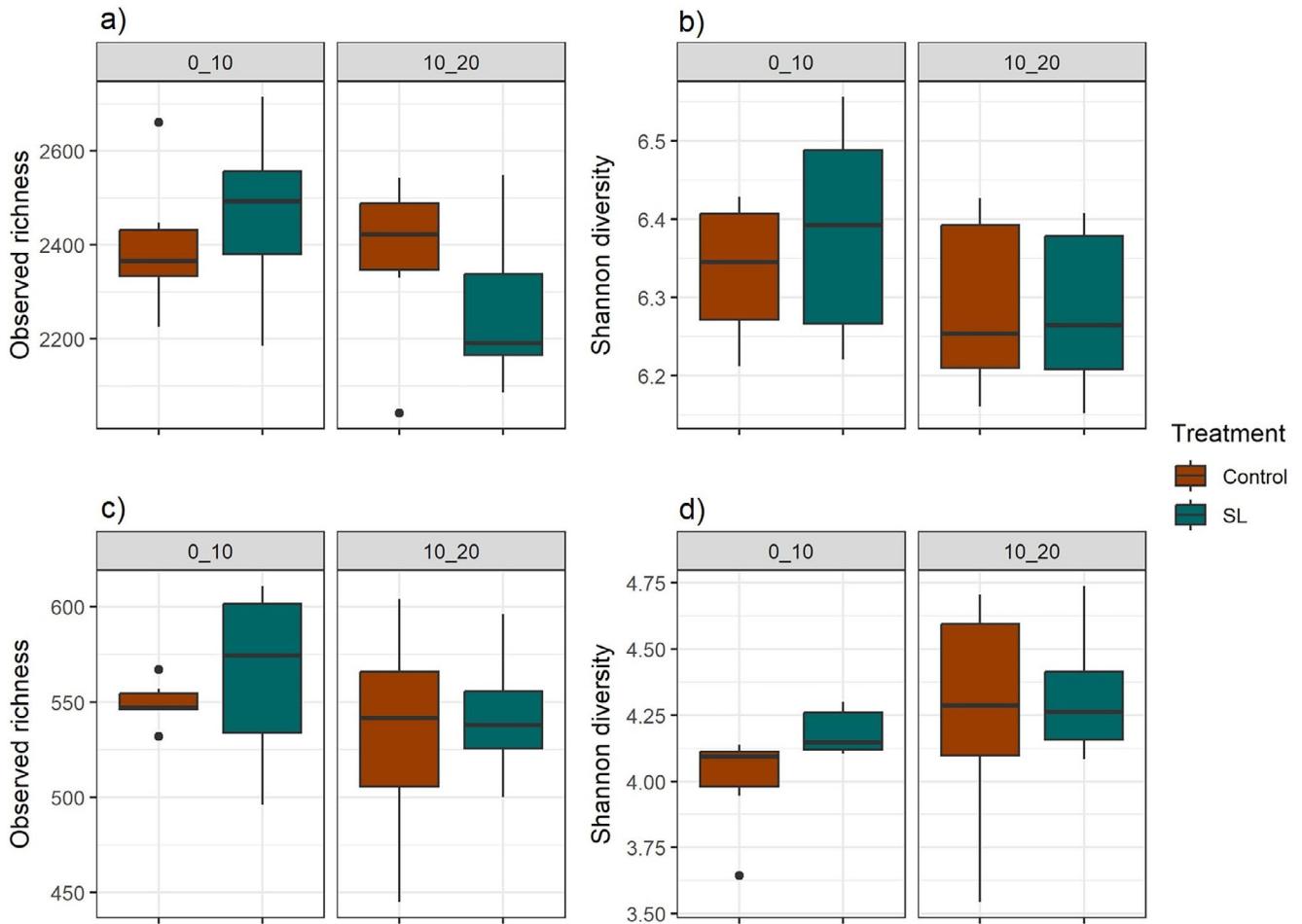


FIGURE 6 | The observed richness and Shannon diversity of bacterial (a and b) and fungal (c and d) communities at the control and structure lime (SL) treated areas of the six separate fields at two depths.

difference between the control and SL-treated areas was the higher mean Ca content in the 0–10 cm layer of the soils treated with SL. There was also a slight indication that B contents were lower in the SL-treated areas of the fields.

3.4 | Soil Microbial Communities

Amplicon sequencing of bacterial 16S and fungal ITS2 barcoding regions of the extracted soil DNA did not show significant changes within the microbial communities due to the SL treatment (Figure 6) but rather by the site and sampling depth of the soil (Bacteria [PERMANOVA site_s: $R^2=0.544$, $p<0.001$; depth₁: $R^2=0.110$, $p<0.001$; treatment₁: $R^2=0.030$, $p=0.23$] and fungal [PERMANOVA site_s: $R^2=0.328$, $p=0.058$; depth₁: $R^2=0.089$, $p=0.045$; treatment₁: $R^2=0.023$, $p=0.582$]). Nevertheless, a few microbial OTUs were found to be more abundant in SL-treated sites, such as bacterial OTUs Chloroflexales, Sphingobacteriales and Elsterales (Supporting Information S1).

4 | Discussion

Our results of soil incubation and rain simulation did not support the hypothesis that soils with higher clay content would

require greater SL additions to enhance structural stability. Instead, regardless of clay content, the lowest SL addition (SL1) already significantly reduced colloid dispersion, and increasing the SL dose did not further decrease colloid detachment. All SL treatments increased soil pH, but after the 5-week incubation, pH remained mostly below 8. For the dissolution of siliceous and aluminous compounds from clay minerals to occur, pH needs to exceed 12 (Rao and Shivananda 2005), and according to Di Sante et al. (2014), the pozzolanic reactions start within the first 7 days after the application of quick lime and continue for at least 1 month. As stabilisation reactions progress in soil, the pH gradually decreases, but in the study by Di Sante et al. (2014), pH remained around 9.5 even after 40 days. Achieving such pH levels typically requires reactive lime (quicklime) additions of 4–8 mass% (Sivapullaiah et al. 2000). In our experiment, the highest SL dose corresponded to approximately 0.3 mass% of reactive lime (about 1 mass% of the SL mixture) in the 10 cm soil layer, which is well below the amounts considered sufficient for pozzolanic reactions to occur. According to Metelková et al. (2012), when the quicklime application is lower than 2 mass%, the improvement in structural stability will result from the formation of calcium hydroxides and calcites rather than pozzolanic reactions. Therefore, it is unlikely that the pozzolanic reactions took place in our experiment, but instead, the reduction in colloid dispersal resulted from an increase in EC and Ca^{2+} concentration in SL-treated soils. Similarly, Blomquist (2021) did not find

evidence to support the pozzolanic reactions in Swedish arable soils after SL application, but instead, he suggested the cation exchange reactions to be the dominant mechanism behind reduced erosion risk in SL-treated soils. Then, the long-lasting effects on structural stability over decades seem unlikely; but instead, if other farming practices remain unchanged, the soils will gradually return to their initial state. Contributing processes include acidification from N fertilisation, as well as plant Ca uptake and leaching, which alter the cation composition of the exchange sites, making them less dominated by Ca^{2+} . The effect of higher SL applications could slow down this development, but the immediate effects of increased rates appear small.

In the incubation study, the highest drainage water turbidities were observed in soils with the lowest EC, and the SL treatment reduced colloid dispersion most in soils with inherently low EC. An increase in soil EC results in compression of the diffuse electrical double layer around the clay particles, which enhances flocculation (Kjaergaard et al. 2004; Luckham and Rossi 1999). Further, an increase in Ca^{2+} in soil cation exchange sites supports flocculation by supporting the formation of cationic bridges (Tisdall and Oades 1982). Similarly, among the field sites, the highest turbidities were measured from the soils with the lowest EC, but the effect of SL on soil structural stability was, in general, inconsistent. In the laboratory, the controlled conditions and uniform mixing of SL likely contributed to the significant reduction in turbidity observed in the incubation study.

The results from the most recently treated fields indicated SL induced reduction in drainage water turbidity, whereas the fields treated over 3 years before the sampling did not show a reduction in colloidal dispersion. However, the fields that were most recently treated with SL inherently had the lowest EC, being therefore most likely to respond to SL treatment with reduced turbidities. Thus, based on these results, we cannot conclude anything about the longevity of the SL-induced structural stability. While cation exchange contributes to reduced colloid dispersion, maintaining improved structural stability requires replenishing Ca ions (Blomquist 2021). Moreover, one key mechanism behind the reduced soil detachment observed in the SL treatment appears to be the increase in EC. However, over time, EC is likely to return to baseline levels as soluble constituents leach deeper into the soil profile. Therefore, SL applications at agriculturally reasonable rates, considering soil pH, are unlikely to result in a permanent improvement of clay soil structure.

Contrary to our hypothesis, we did not observe increased aggregate stability following the SL treatments. Instead, the results support the proposed mechanism that, rather than promoting the formation of highly stable bonds between mineral soil particles and stabilising aggregate structure, SL-induced increases in EC reduce colloid detachment and thus lower the risk of PP loss. However, the effect of reduced particle detachment on PP loss depends on the soil's P content. In soils with low P levels, SL may significantly reduce turbidity without causing a notable decrease in PP concentrations in drainage water. Of the 14 soils included in the incubation study, in only four, the agronomic soil test showed good or excessive P status according to the Finnish guidelines (Eurofins Agro 2020). This likely explains why the SL treatment did not lead to a significant reduction in PP following

rain simulation of the laboratory-incubated soils. Similarly, of the six field soils, only one had excessive P status, whereas in the other soils, the P status was adequate.

In the incubation study, the increasing effect of SL on soil pH was clearly evident, but for these fields, where the SL applications were done 1–6 years earlier, the impact of SL on pH was only marginally significant. Increase in pH is highly dependent on the amount of lime added and on the buffering capacity of the soil (Li et al. 2019) and therefore similar lime addition does not result in equal pH change in different soils. Supporting our hypothesis, the SL treatment increased DOC mobility during rain simulations in both the controlled laboratory incubation study and farmers' fields. This concurs with the studies that report that liming agents and the resulting rise in pH increase the solubility of OM in soil (Curtin et al. 2016; Jokinen et al. 2006). While entering the surface waters, DOC can enhance the primary production and therefore eutrophication of the system (Hoikkala et al. 2015). However, in contrast to our hypothesis, the SL application on field soils appeared not to result in lower soil C stocks, but instead, the higher C content in the subsoil of the SL amended field plots suggests that increased DOC solubility has led to enhanced migration of C in soil which has further deposited in the deeper layers where the pH effect of SL treatment subsides. If the DOC leaching and increase in C concentration in deeper layers is a general quicklime-induced phenomenon, this becomes interesting as a measure of C acquisition. Building C stocks in soils already high in OM is challenging (Chenu et al. 2019; Soenne et al. 2024), but in mineral soils with high OC content in the surface layer, deeper soil layers often have the potential to store more C (Chen et al. 2019; Salonen et al. 2023). According to Cotrufo et al. (2022), inputs of soluble C could be an effective approach to enhance the formation of relatively stable C in soils with low OM content, such as deeper soil layers. Nonetheless, ESM-based calculation in our small data of six soils did not point to any notable increase in soil C stocks.

In a review produced by Palmu and Hedlund (2016) comparing SL and regular agricultural lime, they acknowledged the environmental benefits of SL treatments but mentioned the bottleneck to be the nearly non-existent scientific results of SL application on the microbial soil community structure. Here, we compared the microbial community in the SL-treated field areas to their respective untreated control areas. Neither the fungal nor the bacterial community changed due to the treatments. Thus, no long-lasting harmful effect of SL addition on the soil microbial community could be detected. As soil pH is a major driver of microbial community composition, especially for bacteria (Lauber et al. 2008; Rousk et al. 2010), we expected the impact of SL on soil microbial communities to result from its pH-increasing effect. However, we did not find the microbial communities to change significantly due to the SL treatment, likely due to the fact that soil pH in the field sites was only marginally affected by the SL. Other studies with comparable quicklime application levels have reported an increase in pH, and the treatment has induced changes in the microbial community (Pang et al. 2019; Xun et al. 2016). However, in their studies, the soil pH was well below 5 before the treatment, and the results of Rousk et al. (2010) indicated that the dissimilarity in the bacterial community composition was larger at the lower pH values

than in pH values near neutral, and even insignificant in soils with pH above 6.8.

Even though we did not detect differences in the overall microbial community structure, a few bacterial and fungal species showed significant shifts in abundance between treatments. Representatives from the orders Chloroflexales, Elsterales and Sphingobacteriales were among the bacterial indicator OTUs. In a global meta-analysis on the impact of agriculture on soil biota, *Chloroflexi* was reported to increase in abundance compared to natural soils (Trivedi et al. 2016), and Zhang et al. (2017) found that the relative abundance of *Chloroflexi* increased with rising pH in near-neutral to alkaline soils, reflecting the potential influence of SL on soil pH. Similarly, Elsterales have been reported to be positively correlated with soil pH (Wang et al. 2021), and in our study, their relative abundance appeared higher in SL-treated areas. Sphingobacteriales, on the other hand, have been found to increase in relative abundance with greater P availability (Ling et al. 2017). However, SL-treated areas in our fields did not show higher levels of extractable P, and the SL only marginally increased soil pH, which is known to weaken the sorption of P (Rajan et al. 1974). OTUs found more abundant in control areas included *Penicillium*, which is suggested to contribute to P solubilisation (Richardson and Simpson 2011). The lower abundance of *Penicillium* in SL-treated field areas suggests that the impact of SL on soil pH likely enhanced P desorption, thereby reducing *Penicillium*'s competitive advantage over other microbes. Importantly, we did not observe an increase in plant pathogenic OTUs.

The microbial community change between the two sampling depths 0–10 and 10–20 cm was not related to the SL treatment. In agricultural soils, the aggregates and their composition serve as incubators for the microbial community evolution (Rillig et al. 2017) and affect the microbial performance in all aspects (Wilpiseski et al. 2019, and references therein). As we have not seen SL-induced changes in the aggregate formation, this probably correlates also to the unchanged picture of the soil microbiome. Our study shows that depth influences the composition of the microbial community, and this can be reflected in activity affecting soil C storage.

In the present study, no significant treatment effects were observed in the solubility of macro- and micronutrients in the field soils, except for B, where the SL-induced reduction was marginally significant. Similarly, Gunnarsson et al. (2022) reported lower H₂O-extractable B following either CaCO₃ or SL treatments. However, in contrast to our results, Gunnarsson et al. (2022) also reported lower CAT-extractable (CaCl₂ + DTPA solution) Fe and Mn. Increase in pH weakens the sorption tendency of P (Rajan et al. 1974), but similar to Gunnarsson et al. (2022), we did not detect an increase in P availability extracted with the three different extractants used in agronomic soil testing. The results of long-term liming experiments at Rothamsted showed that liming had a highly significant positive effect on soil pH, but there was large variability in the extractable P (Olsen) (Holland et al. 2018). Application of SL or CaCO₃ has been reported to result in increased availability of Ca (Blomquist et al. 2022; Gunnarsson et al. 2022), but according to Blomquist et al. (2022), the effect was not detectable after 6 years,

in relation to the un-limed control. Also in the present study, when the soil samples were collected 1–6 years after SL application, some short-term effects on plant available nutrient contents might have been passed over due to their time-dependency.

5 | Conclusions

The increase in EC following SL amendment reduced colloid detachment, indicating a lowered risk of erosion. SL was most effective in reducing drainage water turbidity in soils with inherently low EC, suggesting that EC could serve as an indicator of SL's potential to reduce erosion and P losses. The agronomically relevant doses of SL used in this study increased soil pH after incubation, but pH remained well below 9, indicating that pozzolanic reactions are unlikely to occur. At the field sites, the SL-induced pH increase was only marginal, which likely contributed to the absence of significant differences in microbial community structure. Nevertheless, our results suggest that long-lasting harmful effects of SL on the soil microbial community are unlikely. The SL treatment also increased the solubility of OC, whose impact on surface water quality and soil C stocks warrants further investigation.

Author Contributions

Helena Soinne: conceptualization, methodology, visualization, investigation, writing – original draft, writing – review and editing. **Hannu Fritze:** methodology, investigation, writing – original draft, writing – review and editing. **Taina Pennanen:** methodology, investigation, writing – review and editing. **Sannakajsa Velmala:** investigation, visualization, writing – original draft, writing – review and editing. **Mari Rätty:** writing – original draft, writing – review and editing. **Risto Uusitalo:** conceptualization, funding acquisition, supervision, visualization, writing – review and editing.

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Data Availability Statement

The data that support the findings of this study are openly available in Zenodo at <https://zenodo.org/>, reference number <https://doi.org/10.5281/zenodo.15516466>

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Supporting Information

Additional supporting information can be found online in the Supporting Information section. **Data S1:** Supporting Information.