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CLOSING THE PHOSPHORUS CYCLE BY RECYCLING LAKE SEDIMENTS IN AGRICULTURE

Doctoral Thesis

Mina Kiani

ACADEMIC DISSERTATION

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Contents

List of Publications

This dissertation is based on the following publications:

- **I** **Kiani, M.**, Raave, H., Simojoki, A., Tammeorg, O. and Tammeorg, P., 2021. Recycling lake sediment to agriculture: Effects on plant growth, nutrient availability, and leaching. *Science of the Total Environment*, 753, p.141984.
- **II** **Kiani, M.**, Zrim, J., Simojoki, A., Tammeorg, O., Penttinen, P., Markkanen, T. and Tammeorg, P., 2023. Recycling eutrophic lake sediments into grass production: A four-year field experiment on agronomical and environmental implications. *Science of the Total Environment*, p.161881.
- **III** **Kiani, M.**, Tammeorg, P., Niemistö, J., Simojoki, A. and Tammeorg, O., 2020. Internal phosphorus loading in a small shallow Lake: Response after sediment removal. *Science of The Total Environment*, p.138279.

The publications are referred to in the text by their Roman numbers.

Contributions

The following table presents the contributions of the authors to the original articles of the dissertation:

Publications	**Planning of the experiments**	**Conducting experiments and data collection**	**Data Analysis**	**Manuscript preparation**
I	PT, **MK**, HR, AS	**MK**, PT, HR	**MK**	**MK**, PT, HR, AS, OT
II	PT, **MK**, OT, AS	**MK**, PT, JZ, TM	**MK,** JZ, TM	**MK**, PT, JZ, AS, OT, PP, TM
III	OT, **MK**, PT	**MK**, OT, PT	**MK**, OT	**MK**, OT, PT, AS, JN

MK = Mina Kiani, PT = Priit Tammeorg, AS = Asko Simojoki, OT = Olga Tammeorg, PP = Petri Penttinen, JZ = Jure Zrim, HR = Henn Raave, JN = Juha Niemistö, TM = Tuuli Markkanen

Abstract

Phosphorus (P) is an essential plant macronutrient, originating mainly from non-renewable phosphate rocks. Rock phosphate (phosphorite) is one of the critical raw materials listed by the European Commission. Unsustainable P use affects food and water security and causes serious environmental problems such as the eutrophication (i.e., nutrient enrichment) of lakes. Given the essential role of P recycled from sediment to overlaying water in sustaining eutrophication, the removal of nutrient-rich sediments from lakes is one of the most effective restoration techniques, especially for small shallow lakes. Sediment removal removes P accumulated in lake sediments and offers the opportunity to close the P cycle by using the removed P in a sustainable manner.

In the three studies of this thesis, I examined the potential of closing the agricultural P cycle by using lake sediments for crop production. The studies focused on the best practice for sediment recycling based on pot (**I**) and field experiments (**II**) along with investigating the changes to lake P dynamics after sediment removal (**III**). Excavating all the 7500 m^3 of sediment from a 1-ha shallow eutrophic Lake Mustijärv (Viljandi, Estonia) was the starting point of this work. During the lake restoration, 6.4 Mg of P was removed.

In the next step, various application methods were examined to use large quantities of lake sediments for grass production during a nine-month lysimeter experiment. The heavy metal and organic contaminant contents in the sediment were below the levels that would pose ecological or health risks according to the threshold values set by Ministry of the Environment, Finland. Using the excavated sediment with a low Fe:P mass ratio (6) resulted in a greater plant-availability of P and other nutrients compared with the sandy loam soil from the lake shore. The fertilization effect and the substantial increase in the growth and P uptake of ryegrass in the sediment-based treatments made the sediment application advisable for crop production. Of all the P fractions, the Fe-P fraction contributed most to plant P uptake. From an environmental impact perspective, even if a relatively thick layer of sediment was applied on top of the soil, it did not increase the risk of phosphate and mineral nitrogen (N) leaching. In addition, a biochar layer slightly reduced P and N leaching from the sediment.

Furthermore, the environmental effects of various sediment application methods for grass production were studied in a four-year field experiment on the shore of the restored lake. The treatment effects on greenhouse gas (GHG) emissions, N and P leaching, aggregate stability, and soil biota were analyzed. The excavated sediment sustained grass biomass yield of 12 Mg ha^{-1} in the field, even though yield enhancement was less obvious compared with the lysimeter experiment. In addition to 75 g m^{-3} of easily soluble P, the sediment had also high contents of other soluble essential plant nutrients, including sulfur (S), calcium (Ca), magnesium (Mg), boron (B), zinc (Zn), and a fair supply of copper (Cu). Also, the sediment continuously provided a moderate supply of N to the plants over the four-year field experiment,

which was likely due to mineralization of the organic reserves of the sediment. Considering the environmental impacts, the sediment-based growing media were observed to have higher carbon dioxide (CO_2) emissions (579 vs. 400 mg CO_2−C m^{-2} h^{-1}) yet broadly similar nitrous oxide (N_2O) emissions compared with the soil surrounding the lake. Also, applying a thick layer of excavated sediment increased the risk of phosphate and mineral N leaching. Treating the sediment with biochar or adding a mixture of soil and biochar did not affect GHG emissions or nutrient leaching from the sediment material. In addition, sediment treatments had different bacterial and fungal community compositions compared with soil. This could result in different mineralization pathways in soil and sediment-based treatments.

During a two-year follow-up period, internal P loading formation in a recently restored lake was examined using sediment and lake water chemistry data. Soon after sediment removal, a high pool of releasable P was rebuilt in the lake due to exceptionally high external P loading. In addition, extensive anoxia of the surface sediments and in the water overlying the lake bottom was revealed. This resulted in high internal P loading, which sustained the eutrophication. Variations in the release rate of P from the newly formed sediments were explained by changes in sediment NaOH-extracted P and labile P fractions. Moreover, the gradual decrease in different P forms and organic matter from lake inflow towards outflow revealed the important role of sediment accumulation basins in lake restoration.

Sediment from Lake Mustijärv was rich in organic matter and was a good source of several essential nutrients, including P, in plant-available form. I suggest that, instead of using a thick layer of sediments alone as a growing medium, sediment would be applied to soils based on crop nutrient requirements. Such an application, similar to the application of organic fertilizers, may reduce nutrient losses through leaching. With the aim of closing the agricultural P cycle, this case study can be upscaled to other shallow lakes with similar sediment properties.

Abbreviations

AGB	Aboveground plant biomass
CEC	Cation exchange capacity
C_{org}	Organic carbon
DM	Dry matter
DO	Dissolved oxygen
EC	Electrical conductivity
Fe:P	Total iron to total phosphorus ratio
GHG	Greenhouse gases
LCA	Life cycle analysis
ICP-OES	Inductively coupled plasma–optical emission spectrometry
LOI	Loss on ignition
N_2O	Nitrous oxide
NH_4^+	Ammonium
NO_3^-	Nitrate
MBM	Meat bone meal
OM	Organic matter
OTUs	Operational taxonomic units
PAH	Polyaromatic hydrocarbons
PO_4^{3-}	Phosphate
SPC	Electrical specific conductance
SRP	Soluble reactive P
SSA	Specific surface area
VM	Volatile matter
WSA	Water stable aggregate
w/w	weight to weight
16S rDNA	16S ribosomal DNA

1 Introduction

1.1 The leaking cycle of agricultural phosphorus and the need for recycling

Among the nutrient elements, phosphorus (P) is second after nitrogen (N) in its impact on the productivity and health of terrestrial ecosystems (Weil and Brady, 2017). It is required for multiple important plant functions, including the structure and function of nucleic acids, photosynthesis, flowering, seed production, and maturation, as well as for legume nodule development and associated N fixation (Weil and Brady, 2017; Schneider et al., 2019).

The total P content in the upper 50 cm of soils ranges from 500 to 10 000 kg P ha^{-1} (Figure 1), of which only a small percentage is available to plants and microorganisms as orthophosphate (HPO_4^{2-} and $H_2PO_4^-$; Schneider et al., 2019). To pursue high yields and quality in crop production, a large quantity of P fertilizer is usually applied to the soil (on average, 8.8 kg P ha^{-1} y^{-1} worldwide, Alewell et al., 2020). The use of mineral P fertilizers has significantly altered the global P cycle, leading to 47% of global agricultural soil P fertility being derived from anthropogenic sources (Demay et al., 2023). However, the global utilization rate of phosphate fertilizer is only 10–20% due to fixation reactions in soil (Weil and Brady, 2017; Zhao et al., 2021). In general, most soil P is tightly sorbed to mineral particle surfaces, bound within organic matter (OM), or precipitated as poorly soluble salts, which leads to its export from soils to waterbodies mainly via water erosion (Alewell et al., 2020). The average P loss rate from arable soils due to erosion by water is globally estimated to be in the range of 5.9 kg P ha^{-1} y^{-1} (Alewell et al., 2020) to 10 kg P ha^{-1} y^{-1} (Smil, 2000). Phosphorus loss leads to inefficient nutrient use and serious environmental problems such as the eutrophication of waterbodies (Djodjic et al., 2004; Räty et al., 2020). Phosphorus that has accumulated at lake bottoms can be recycled back into the overlying water column (i.e. internal nutrient loading) and can sustain eutrophication (Søndergaard et al., 2003; Tammeorg et al., 2017). Furthermore, the open agricultural P cycle contributes to the alarmingly high biodiversity decline rates, economic losses associated with clean-up, and large-scale human health risks from contaminated drinking water supplies (Brownlie et al., 2022), which emphasize the importance of closing the P cycle.

On the other hand, phosphate rock is a non-renewable resource that is listed as a “critical raw material” with a high supply risk noted by the European Commission (Giese, 2022; Spears et al., 2022). Also, the distribution of world P reserves is globally highly skewed, with over 95% of the world’s P rock reserves found in 10 countries (75% in Morocco and Western Sahara, and 6% in China; Roy et al., 2022). The imbalance between P consumption and P sources is likely to increase P fertilizer prices and induce economic scarcity.

Furthermore, considering that Russia, Belarus, and Ukraine together represent 14% of P exports, the Russo–Ukrainian war has recently spiked P fertilizer prices (Kuokkanen, 2022).

Thus, recycling eutrophic lake sediments and associated nutrients into crop production may offer a sustainable solution to P scarcity, nutrient losses from agricultural soils, and the eutrophication of waterbodies.

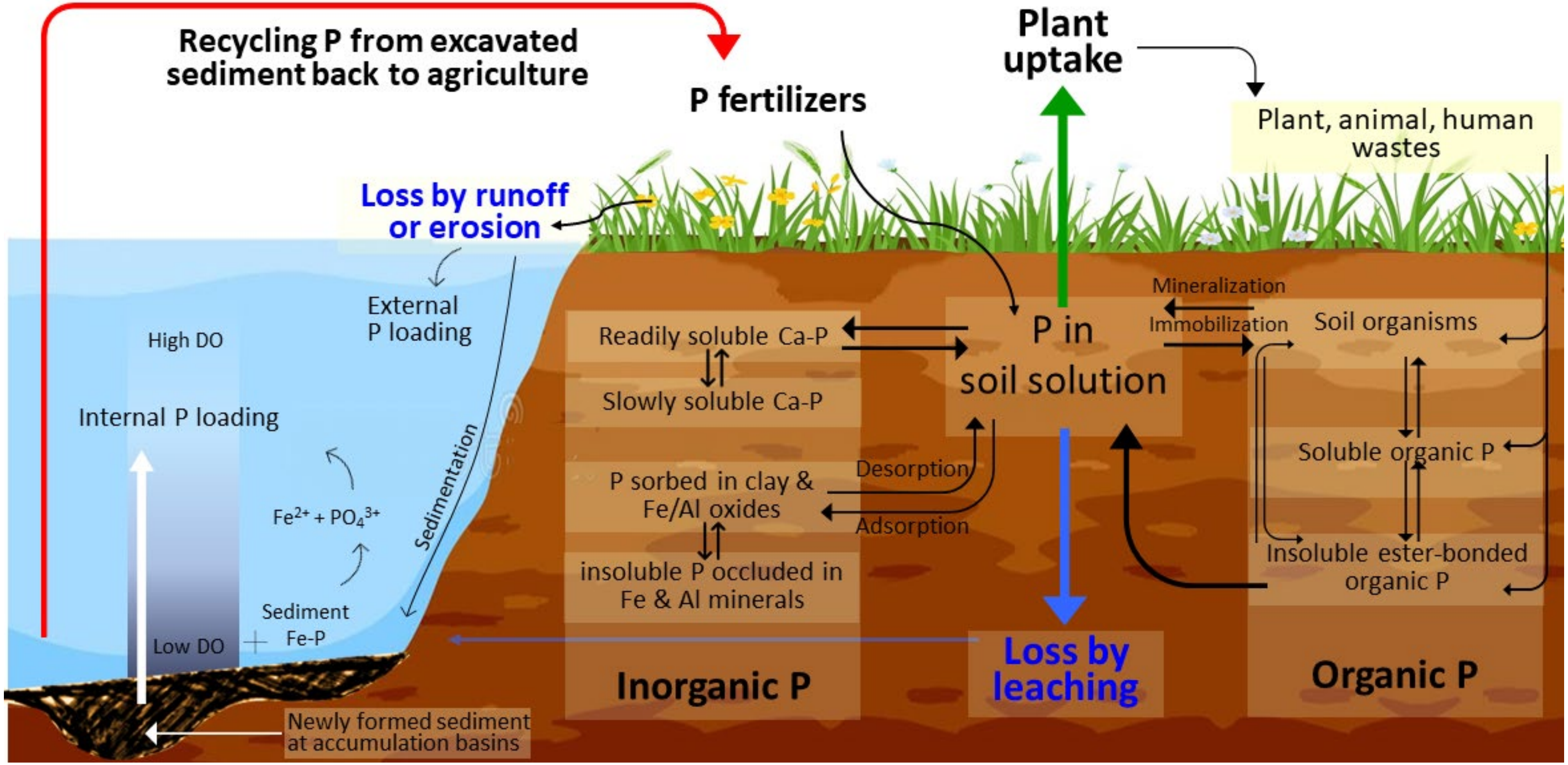

Figure 1. The phosphorus (P) cycle in soil (Adapted from Weil and Brady (2017)) and redox-related P release from sediments after lake restoration by sediment excavation.

1.2 Lake sediments and internal P loading

Lake sediments entrap large nutrient quantities originating from human activities in the catchments (Håkanson and Jansson, 1983). The P pool in lake sediments is often over a hundred-fold greater than the pool present in the water (Søndergaard et al., 2003). Phosphorus availability is one of the most important factors for determining lake water quality (Schindler, 2012; Schindler et al., 2016). Eutrophication and its symptoms, including algal blooms, low water transparency, and anoxia induced by high respiratory activities, are often associated with P release from sediments (i.e., internal P loading; Nürnberg, 2009). Internal P loading often represents the major P load in summer, with pronounced implications for lake water quality due to its bioavailability (Nürnberg, 2009; Nürnberg and LaZerte, 2016; Tammeorg et al., 2016). In shallow lakes, the influence of sediment on lake water P concentrations is particularly pronounced due to the direct contact of sediment with the photic zone, i.e. the uppermost layer of a waterbody that receives sunlight, and the regular mixing regime (Nixdorf and Deneke, 1997; Nürnberg et al., 2013).

Sediment P is mobilized in anoxic conditions when the redox potential decreases to a level approximately below 200 mV. This leads to the breakdown of Fe-P complexes, as the ferric iron of the sediment is reduced to ferrous iron with the subsequent dissolution of P (Mortimer, 1941). Soluble P can be further transported to the overlying lake water by various mechanisms, including diffusion due to the concentration gradient between the sediment pore water and the overlying water column (Håkanson and Jansson, 1983; Tammeorg et al., 2020). However, additional factors also contribute to sediment P release, such as the mineralization of OM and temperature. Additionally, P release was reportedly controlled at a sediment Fe:P mass ratio above 15 (Jensen et al., 1992).

Internal P loading has often delayed the response of lake water quality to the reduced supply of external nutrients (Jeppesen et al., 2005; McCrackin et al., 2017). Hence, tackling the internal P loading is often needed in lake restoration in addition to reducing the external nutrient supply. The most common lake restoration methods that aim to improve water quality by reversing internal nutrient loading focus on inactivating the reactive P. An example of chemical inactivation of P is using iron (Fe) or aluminum (Al) to precipitate the releasable P into the lake bed (Lürling et al., 2020). Artificial aeration and oxygenation have also been performed to speed up the restoration process (Li et al., 2019). Although these techniques can sometimes be helpful if a rapid solution is needed (e.g., in lakes used for drinking water supply), they neither remove the P from the environment nor provide a long-term solution to the problem. They may also alter the sediment characteristics into less suitable forms for agricultural use (Horppila, 2019).

Sediment removal by dredging or excavation is an effective long-term strategy that removes P accumulated in lake sediments and can reduce sediment nutrient release (Pierce, 1970; Bormans et al., 2016; Cooke et al., 2016; Lürling et al., 2020). The drawbacks of the method include a relatively high cost, temporary sediment resuspension in the lake, removal of biota including benthic organisms, and challenges related with relocation and storage of removed sediments.

Sediment removal has reduced the internal P loading immediately and substantially in many cases (Cooke et al., 2016, Lürling et al., 2020), with a few exceptions due to continuously high external nutrient loading (Björk et al., 2010; Bormans et al., 2016; Francescangeli, 2022) or insufficient sediment removal (Liu et al., 2015; Phillips et al., 2015). Successful removal has led to a decrease of nutrient concentrations in the water column, phytoplankton biomass, and cyanobacterial blooms and to an increased coverage of macrophytes (Wichelen et al., 2007; Björk et al., 2010; Hassett and Steinman, 2022). Further benefits include reducing nuisance growth of aquatic macrophytes, lake deepening to provide habitats for fish and other biota, maintaining recreational activities, and removing toxic sediments (Pierce, 1970; Lürling et al., 2020). Moreover, as P rock scarcity is increasing, the extraction processes for handling nutrient-rich sediments are likely to become more relevant in the future (Braga et al., 2019).

Only a few studies are available that explore the effects of partial sediment removal on internal P loading and lake water quality (Churchill and Brashier, 1975; Knaus and Malone, 1984; Pandey and Yaduvanshi, 2005; Björk et al., 2010). The current study focused on the formation of internal P loading in a lake restored by the full removal of sediments.

1.3 Effects of recycled sediment on physicochemical soil properties and plant growth

1.3.1 Lake sediments

Sediment recycling for soil fertilization is in line with the long-promoted circular economy policy and enables the use of micro- and macronutrients accumulated in sediments (Matej-Łukowicz et al., 2021). Recent years have seen increased interest in the effects that sediment collected from waterbodies (seas, lakes, reservoirs, agricultural wetlands, and river basins) may have on plant production. However, most of these studies have been conducted in greenhouse conditions, with adequately replicated field experiments being extremely scarce, including the proper characterization of sediment (Table 1).

Lake sediments have been shown to be efficient as soil amendments by increasing nutrient availability, such as P, calcium (Ca), and copper (Cu, Canet et al., 2003; Edesi et al., 2020; Brigham et al., 2021) in soils with poor fertility status; by increasing OM content (Edesi et al., 2020; Brigham et al., 2021;

Szara-Bąk et al., 2023); by increasing the water holding capacity (Darmody and Diaz, 2017) and the cation exchange capacity (CEC; Canet et al., 2003; Darmody and Diaz, 2017; Brigham et al., 2021); by elevating soil pH (Ebbs et al., 2006; Baran et al. 2019); and by decreasing soil bulk density (Woodard et al., 1999; Brigham et al., 2021). This has resulted in crop yield benefits in several studies, especially in nutrient-deficient soils (Canet et al., 2003, Darmody and Diaz, 2017; Edesi et al., 2020).

For instance, in a recent two-year field experiment, the application of 100 Mg ha^{-1} sediment from an unspecified Estonian lake increased the amount of soil organic carbon, and the N, Ca, magnesium (Mg), boron (B), manganese (Mn), and Cu contents compared with an unfertilized clay loam soil, which resulted in higher yields of oats and barley (Edesi et al., 2020). Studies where the tested sediment and control soil were both adequately fertilized resulted in similar growth performance. In a study by Ebbs et al. (2006), soil and sediment had a similar texture, and both were fertilized with 0.3 g kg^{-1} Osmocote 18-6-12 (N-P-K). They suggested that the reclaimed slightly alkaline silty loam sediment from Lake Peoria (30% sand, 51% silt, and 19% clay) can be utilized for vegetable production, as it produced an equal yield of edible plant parts and higher shoot biomass compared with the reference soil (limed for achieving an equal pH with sediment). Moreover, Woodard et al. (1999) reported that the forage crops of alfalfa and big bluestem, grown in silty clay loam sediment from a lake in South Dakota, had similar or slightly higher shoot dry matters and higher relative N and P uptakes compared with those in silty clay loam soil in their pot study. All treatments were fertilized with 200 mg N kg^{-1},100 mg P kg^{-1}, and 100 mg K kg^{-1}. In addition, slightly alkaline lake sediment materials with high OM content increased soil OM and CEC in soils, leading to higher plant yields (Canet et al., 2003; Brigham et al., 2021).

Table 1. Summary of available studies about the effect of recycled sediment on plant production.

Author(s)	**Sediment origin**	**Experiment design**	**Control treatment**	**Plant species**	**Sediment properties**	**Plant biomass or grain yield**	**Environmental effects**
Baksiene and Asakaviciute (2013)	**Lake** Guobstas, Lithuania	**field** rep.: NA 10 years	sandy loam soil TP: 170 mg kg^{-1}	corn, barley, grasses, winter rye, oat+lupin	pH>6.4 clay: NA OM 1.6% in 40t sed ha^{-1} TP: 400 mg kg^{-1}	Averaged +24%	NA
Darmody and Diaz (2017)	Peoria **Lake**, USA	**field** 2 rep. 4 years	Sandy Soil P_{M3}: 87 mg kg^{-1}	corn, soybean	pH 7.4 clay 29% OM 2-4% P_{M3}: 90 mg kg^{-1}	+23−248%	higher Mo levels in soybean grain, above the threshold
Edesi et al. (2020)	**Lake**, Estonia	**field** 6 rep. 2 years	clay loam soil P_{M3}: 44 mg kg^{-1}	oat, spring barley	pH 7.4 clay NA C_{org} 7% P_{M3}: 0.6 kg ha^{-1}	+12−22%	higher dehydrogenase activity
Woodard (1999)	Punished Woman **Lake**, USA	pot 4 rep. up to 10 weeks	silty clay loam soil P_{Olsen}:7.0 mg kg^{-1}	maize, soy, sunflower, alfalfa, bluestem	pH 7.5 clay 37% OM 5.3% P_{Olsen}: 16 mg kg^{-1}	Alfalfa: +45%	no PCBs or heavy metal contamination
Canet et al. (2003)	Albufera **Lake**, Spain	pot 4 rep. up to 2 months	loamy sand soil P_{Olsen}: 17 mg kg^{-1}	lettuce, tomato	pH 7.96 clay 53% OM 5.8% P_{Olsen}: 38 mg kg^{-1}	Lettuce: +99%	higher Zn and Mn levels in lettuce; no risk
Ebbs et al. (2006)	Peoria **Lake**, USA	pot 4 rep. to reproductive maturity	silty loam soil $P^{\ddagger}$: 81 mg kg^{-1}	bean, broccoli, tomato, pepper, carrot	pH 7.7 clay 19% OM 2.1% $P^{\ddagger}$: 60 mg kg^{-1}	equal edible yield	higher Zn, Mo, and Cu level in soil and plant; no risk
Brigham et al. (2021)	**Lake** Erin, USA	pot rep.: NA 123 days	silty clay soil P_B: 110 mg kg^{-1}	soybean	pH 7.9 clay: NA C_{org} 3% P_B: 38 mg kg^{-1}	equal yield	no increase of N and P leaching
Baran et al. (2012)	Besko Reservoir, Poland	pot rep.: NA 70 days	loamy sand soil $P^{\ddagger}$: 79 mg kg^{-1}	maize	pH 7.2 particles <0.02 mm: >84% C_{org} 1.8% $P^{\ddagger}$: 9.0 mg kg^{-1}	-30% shoot mass	Heavy metal content lower than the threshold

Laakso et al. (2017)	Agricultural wetland, Finland	pot 3 rep. 70 days	Silty clay loam soil P_{AC}: 10 mg kg^{-1}	Italian ryegrass	pH 6.8 clay> 60% TC 1.3% P_{AC}: 3.0 mg kg^{-1}	-33% to +11%	NA
Mattei et al. (2018)	Port of Livorno, Italy	pot 8 rep. 7 months	peat-based growing medium P_A: 0.46 mg kg^{-1}	red robin photinia	pH 7.9 clay 16% C_{org} 1.7% P_A: 2.4 mg kg^{-1}	comparable plant growth	higher Zn and Cu levels; no risk. Improve the C footprint
Ugolini et al. (2018)	Navicelli Canal, seawater, Italy	pot 24 rep. 6 months	peat-based nursery substrate TP: 400 mg kg^{-1}	holm oak	pH 7.7; clay 8% C_{org} 1.9% *Fe:P: 44.4 TP: 700 mg kg^{-1}	comparable plant growth	Higher Cd, Cr, Pb, Zn, Cu, Mn, Mo, and Ni levels; no risk
Baran et al. (2019)	Rzeszów reservoir, Poland	pot 4 rep. up to 86 days	sandy soil P_E: 70 mg kg^{-1}	broad bean, maize	pH 7.1 clay 54% C_{org} 2.3% P_E: 144 mg kg^{-1}	+22−73%	higher Mn, Zn, Cu and Ni levels in soil and plant; no risk
Kazberuk et al. (2021)	Fishpond, Poland	pot 3 rep. 2 years	loamy sand soil P_E: 39 mg kg^{-1}	white mustard	pH 6.4 clay 7% C_{org} 1.6% P_E: 33 mg kg^{-1}	+35−55%	higher Zn levels in soil and plant; no risk
Kazberuk et al. (2021)	Vistula River, Poland	pot 3 rep. 2 years	loamy sand soil P_E: 39 mg kg^{-1}	white mustard	pH 7.6 clay 4% C_{org} 0.27% P_E: 2.3 mg kg^{-1}	Not detectable	higher Zn levels in soil and plant; no risk
Tozzi et al. (2021)	Port of Livorno, seawater, Italy	pot 2 rep. 3 years	peat-based growing medium P_M: 2.0 mg kg^{-1}	strawberry	pH>7; clay: ~16% TC 0.15% P_M: 2.3 mg kg^{-1}	+100% fruit production	higher Zn level but no risk. No risk of organic contaminants
Ferrans et al. (2022)	Malmfjärden Bay, seawater, Sweden	pot 3 rep. up to 70 days	Compost TP: 2100 mg kg^{-1}	lettuce	pH 6.6; clay: 10-20% C_{org} 6.8% *Fe:P: 20.8 TP: 1300 mg kg^{-1}	-88%	higher Cd levels. Not good to cultivate edible crops
Koniarz et al. (2022)	Chechlo reservoir, Poland	pot 3 rep. 120 days	sandy clay soil P_E: 91 mg kg^{-1}	grass mixture	pH ~5.0; clay: NA C_{org} up to 1.2% P_E: 12 mg kg^{-1}	-55%	higher Zn and Cd levels. Not good for fodder

Rep.: replicate; NA: not available; Sed: sediment; OM: organic matter; C_{org}: organic carbon; CT: total carbon; TP: total phosphorus; P_{M3}: Mehlich 3 extractable P; P_{Olsen}: Olsen extractable P; P_B: Bray-1 extractable P; P_E: Egnera-Riehma extractable P; P_{AC}: ammonium acetate buffer-extractable P; P_A: Ammonium nitrate extractable P; P_M: Murphy and Riley extractable P; $P^{‡}$: extraction method not available; -: growth reduction compared to the control treatment; +: growth enhancement compared to the control treatment. * The total Fe:P mass ratio was not available for any of the other presented studies.

1.3.2 Sediments from other sources

In contrast with the abovementioned articles, studies focusing on the fertilizer value of sediment from sources other than lakes have generally reported varying responses to crop yield depending on sediment characterizations, plant type, and sediment proportion to the growing medium. Baran et al. (2019) reported that a reservoir sediment material (8% sand, 38% silt, 54% clay) situated on the Wisłok River in southeastern Poland had a higher pH value (7.1) and higher contents of Mg, Cu, Mn, zinc (Zn), organic carbon (C_{org}), and available P and potassium (K) compared with sandy soil (pH 4.7). This resulted in a higher shoot mass of maize and broad beans at the highest sediment rate (50% sediment) in their pot experiment. However, the maize yield was reduced due to compaction (Baran et al., 2012), when 10% or more of fine-textured sediment (84% <0.02 mm) from the Besko Reservoir was added to a loamy sand soil, although the experiment replication was unclear. A similar result, where the growth of a grass mixture was hampered, was reported when over 50% of the growing medium contained sediment from the Chechlo Reservoir (pH ~5) compared with sandy clay soil (Koniarz et al., 2022). This was possibly due to a 2.7-fold lower carbon content and to the poor P and K fertility status of the sediment material.

Also, sediment from constructed wetlands (>60% clay contents) in Finland demonstrated very low P availability for plants (Laakso et al., 2016; Laakso et al., 2017), possibly due to fresh and high amounts of P-binding compounds in sediments (e.g., Al and Fe (hydr) oxides and high clay content). Kazberuk et al. (2021) saw a 30% to 60% increase in white mustard yield using 1% to 20% of fishpond sediment (7% clay, pH 6.5, and C_{org} 1.6%) compared with an unfertilized loamy sand soil (7% clay, pH 5.2, and C_{org} 0.7%) in a two-year pot experiment. However, the yield decreased with river sediment (4% clay, pH 7.6, and C_{org} 0.3%), at a rate of 5% and above, supposedly due to deteriorated soil aeration and moisture conditions.

Phytoremediated sediments from the sea bottom of the Port of Livorno, Italy could potentially replace peat as a growing medium for ornamental plants, due to its significantly higher pH, electrical conductivity (EC), available P, and total P, Ca, Mg, and Zn levels (Mattei et al., 2018). Similar results were reported for a strawberry yield, grown in sediment also originating from the Port of Livorno over three cultivation years, due to the progressive improvement of physical and chemical soil fertility by the sediment matrix (Tozzi et al., 2021). Also, decontaminated dredged sediments from the Navicelli Canal in Italy (58% sand, 34% silt, 8% clay) showed better seed germination and root growth, with excellent results for shoot growth compared with fertilized nursery substrate (Ugolini et al., 2018), suggesting a fertilization effect by the sediments.

1.3.3 Characterization of sediment properties for agricultural use

Based on the current literature, sediment materials with low clay content and neutral pH status provided comparable amounts of available P as control treatments, resulting in comparable plant P contents in both growing media. The high availability of P in sediment materials with a low level of fine clay and associated Al and Fe (hydr) oxides material can be due to the lower P adsorption sites (Sippola, 1974; Laakso et al., 2017). In most studied cases, amendments with uncontaminated sediments were also found to either contribute positively to or at least to not decrease the plant yield, especially when sediment C_{org} was above 1.5%. High soil organic C contents are well known to increase soil fertility, soil stabilization, soil structure, water holding capacity, and crop productivity (Kögel-Knabner and Rumpel, 2018). In addition, a higher clay content necessitates a greater amount of C_{org} to achieve reasonable yields, as evidenced by the negative correlation between N application and the clay:C ratio reported by Soinne et al. (2021). However, despite the overall positive impression, studies showing clear crop responses and providing possible mechanisms behind the results remain scarce. Reporting the sediment's physico-chemical characteristics along with descriptions of the experimental design and data analysis methods is important for a more conclusive discussion.

1.4 Environmental effects of recycling sediments

Lake sediments can have different physico-chemical properties and microbial communities, and may also contain pollutants depending on their origin. For example, sediments from the natural environment may be rich in debris and vegetation, sediments from industrial areas may be contaminated by heavy metals and organic pollutants, sediments from agricultural areas may contain high levels of agrochemicals, and sediments from urban areas may contain boulders, construction debris, and also solid waste (Renella, 2021). Therefore, sediments may need to be examined and treated accordingly before use. Various treatment methods exist, with the most common ones being related to bio- or phytoremediation, composting, washing, and electro-kinetic remediation (Renella, 2021). Considering the ecological risk, it is important to provide a comprehensive analysis of the quality and quantity of sediments before their utilization in agriculture.

1.4.1 Heavy metals in recycled sediments

The pH, organic C content, and silt and clay fraction contents are key factors for the mobility of heavy metals in soil (Kazberuk et al., 2021). Kazberuk et al. (2021) reported that sediment application increased the contents of heavy metals in a loamy sand soil and in the plant tissue of white mustard. Greatest Cu and Pb levels were found in sediment from the pond with 1.6% OM, and the highest Zn levels were observed in sediment from the Vistula River with the lowest OM content (0.27%) and the highest pH (7.6). Regardless of sediment origin, the highest levels of heavy metal contents were recorded for Zn (35 mg kg^{-1} DM sediment), which also had the highest transfer factor from soil to plant. Other studies obtained a similar result; adding sediment to the growing medium may increase the growing medium content of cadmium (Cd) (Ebbs et al., 2006; Ugolini et al., 2018; Ben Chabchoubi et al. 2020; Ferrans et al., 2022), Cu (Woodard et al., 1999; Mattei et al., 2018; Ugolini et al., 2018; Baran et al., 2019;), Mn (Canet et al., 2003; Ugolini et al., 2018; Baran et al., 2019; Ben Chabchoubi et al., 2020), molybdenum (Mo) (Ebbs et al., 2006; Ugolini et al., 2018), nickel (Ni) (Ugolini et al., 2018; Baran et al., 2019;), and Zn (Woodard et al., 1999; Canet et al., 2003; Ebbs et al., 2006; Mattei et al., 2018; Ugolini et al., 2018; Baran et al., 2019; Ben Chabchoubi et al., 2020). In addition, an increase was reported in plant Cd content (grass mixture: Koniarz et al., 2022; lettuce: Ferrans et al., 2022), Cu content (beans, broccoli, pepper, and tomato: Ebbs et al., 2006; red robin photinia: Mattei et al., 2018), Mn content (lettuce: Canet et al., 2003), Mo content (beans, broccoli, and pepper: Ebbs et al., 2006), and Zn content (lettuce: Canet et al., 2003; beans, broccoli, and pepper: Ebbs et al., 2006; red robin photinia: Mattei et al., 2018; strawberry: Tozzi et al., 2021). However, their elemental contents did not exceed locally established threshold values. On the contrary, Darmody and Diaz (2017) reported that, despite normal metal levels in soils and plant tissues, the Mo level (23 mg kg^{-1}) in soybean grain exceeded levels considered safe for livestock fodder. Also, grass biomass grown in a growing medium containing over 10% of sediment from the Chechło Reservoir is not recommended for fodder use due to its high Cd and Zn contents (Koniarz et al., 2022).

1.4.2 Microbial community of recycled sediments

The microbial community of sediment material may play an important role in nutrient recycling to plant production. The sediment-borne microbial community can act as an agent for the bioremediation of contaminated sediments by degrading specific compounds (Renella, 2021). In addition, sediment addition to agricultural soil may change the microbiological community structure due to an increase in soil OM content, water retention, and improvement of soil structure. For instance, soil dehydrogenase activity, which is used as an indicator of the overall activity of microorganisms, was higher when 100 kg ha^{-1} lake sediment was applied to organic farming cultivation compared with unfertile clay loam soil, probably due to higher C_{org} and nutrient contents in this treatment (Edesi et al., 2020). Mattei et al. (2017) used dredged sediment from the Navicelli Canal co-composted with green waste as a plant-growing medium. They reported a lower bacterial biodiversity but higher fungal biodiversity for the dredged sediment compared with the green residues of pruning activities before the composting procedure. In general, the microbial community structure in recycled sediments is poorly investigated.

1.4.3 Nutrient leaching from recycled sediments

Phosphorus application above crop needs is widely known to lead to an accumulation of total soil P, which presents an increased risk of P loss to surface water (Brownlie et al., 2022). Because dredged sediments can be rich in organic and inorganic C and bioavailable nutrients, sediment application as a soil amendment or as a growing medium may increase nutrient losses from agricultural lands, aggravating eutrophication problems in waterbodies. I was able to find only one study investigating nutrient loss from farmland soils amended with lake-dredged sediments, a pot study by Brigham et al. (2021). They reported a higher load of percolated C_{org} from the sediment than from silty clay soil but no differences between the sediment and soil in the loads of percolated $PO_4^{3-}-P$ and $NO_3^{-}-N$ in the presence of soybean plants. Similar N and P leaching can be due to the similar N and P contents in the soil and sediment material. However, they concluded that the nutrients were quickly incorporated into the soybean biomass, and rapidly decreasing loads were observed in the percolated water over the growing season.

It has been reported that the labile C fraction in biochars (biomass pyrolyzed at temperatures above 250 ºC, Lehmann and Joseph, 2015) may cause short-term $NO3^{-}$ immobilization, leading to reduced N leaching from the soil to the environment (Kolb et al., 2009; Tammeorg et al., 2012). A temporary movement into and storage of nitrate within mid-sized pores of biochars may also cause similar results. Biochars can retain P in the soil by sorbing P to oxygen-containing functional groups on the biochar surface or incorporating P into their mineral structures within the biochar (Luo et al., 2022). Both of these mechanisms would likely decrease the leaching losses of P and resulting in more retained P in the soil available for plants. Therefore, combining sediment materials with biochars could be tested to find the best solution for an environmentally friendly approach to sediment applications in agriculture.

Approximately 25% of soil OM consists of carbohydrates derived from plant polysaccharides, and these organic compounds act as a mucilage (glue) in soils, creating a soil that is more resistant to erosion (Oades, 1984). Therefore, OM-rich sediment materials are expected to reduce the risk of erosion. In addition, biochar amendment can control surface runoff by improving aggregate stability (Soinne et al., 2014; Schmidt et al., 2021). However, whether combining organic-rich sediment with biochar would further reduce the runoff remains to be studied.

1.4.4 Greenhouse gas emissions from recycled sediments

To the best of my knowledge, the greenhouse gas (GHG) emissions from recycled sediments on agricultural lands have not been studied in the field. Bates et al. (2015) assessed the carbon dioxide (CO_2) equivalent emissions from three placement types for removed sediment (dumped into open water, containment island, and upland placement) through a life cycle assessment. They suggested that upland placement may be the most environmentally burdensome alternative for uncontaminated sediments due to the emissions associated with diesel fuel combustion and electricity. Another life cycle assessment study showed that recycling sediment (12.5 kg CO_2 eq m^{-3}) as an ingredient of growing media for plant nurseries may improve the C footprint of plant production compared with the use of traditional peat-based growing media (70 kg CO_2 eq m^{-3}; Mattei et al., 2018). Due to their extraction and transport, peat-based growing media are one of the GHG emission sources in the plant nursery sector (Lazzerini et al., 2016).

According to several meta-analyses, biochar amendments can reduce soil nitrous oxide (N_2O) emissions (Schmidt et al., 2021). In general, biochars produced at high pyrolysis temperatures (>500 °C) show a relatively high efficacy in reducing GHG emissions. Wood-derived biochars can reportedly reduce CO_2 emission from the soil, as they have less labile C compared with straw-derived biochars (Awad et al., 2018). However, whether biochar additions have any effect on GHG emissions from carbon-rich sediments remains unknown.

2 Aims

I aimed to find efficient ways to recycle sediment from eutrophic shallow lakes as a soil amendment on agricultural lands to close the agricultural P cycle. The agronomical and environmental benefits and drawbacks of recycling sediment for crop production were studied both in the pot and in the field. Further, I examined the formation of internal P loading in a recently restored lake by monitoring the changes in sediment P mobility after sediment removal. The research questions for this dissertation were:

1. What are the effects of various sediment application methods on soil nutrient availability during a nine-month indoor lysimeter experiment (**I**) and do the effects persist during four subsequent growing seasons in the field experiment (**II**)?
2. How does the applied sediment affect the growth and nutrient uptake of perennial ryegrass (*Lolium perenne* L.) in a nine-month indoor lysimeter experiment (**I**) and are the effects consistent with a four-year field experiment of mixed perennial grasses (**II**)?
3. What are the effects of various application methods of lake sediments on the leaching of N and P (**I, II**), GHG emissions, aggregate stability, and soil biota (**II**)?
4. How does hardwood biochar addition affect P and N losses in leaching under both laboratory and field conditions (**I, II**), along with GHG emissions from the growing media, and the aggregate stability of recycled sediments (**II**)?
5. Can internal P loading be a concern in a lake restored by complete sediment removal? What factors affect the P release from newly formed sediment in Lake Mustijärv (**III**)?

3 Materials and methods

3.1 Study site

This dissertation is based on data gathered from a sediment recycling project at Lake Mustijärv. The lake was created around 1985 and has a surface area of 1 ha. It is located on the Kurika Stream, approximately one km west of Viljandi, Estonia (**III**). The sediment from the lake was studied in an indoor lysimeter experiment (**I**) and in a field experiment established on the shore of Lake Mustijärv (**II**).

By 2015, the lake was heavily eutrophicated and silted with remaining 0.3 ha of free water surface (with less than 1.0 m deep). In 2016–2017, the lake was completely desilted up to the lakebed by excavating all sediment (a layer averaging 1.1 m deep) to a mean lake depth of 2 m (Figure 2). An island of rocks, designed as a fish spawning habitat, was introduced at a location near the center of the lake (CE), along with two sediment collection basins at the lake inflow (IF) and CE locations with mean depths of 3.0 and 3.8 m, respectively. Also, the downstream part of the initial lake was expanded so that, after lake restoration, Lake Mustijärv had a surface area of 1 ha, a length of 257 m, and a width of 58 m at its widest point.

The monthly average air temperature during the four-year field experiment was close to regional averages, and no major differences were observed in temperature accumulation between the years. However, 2018 was slightly hotter than the other years and received relatively less rainfall during the warm spring and summer months (EWS, 2021).

3.2 Materials

Before excavation, a sediment storage site was chosen close to the lake shore in which red clover and grasses were grown in 2012–2016. The storage site was prepared by peeling off a 30-cm layer of topsoil and vegetation. The collected topsoil was tentatively classified as an Endogleyic Lamellic Luvisol (IUSS, 2015) with a sandy loam texture (Tables 2 and 3). This topsoil was later used as a treatment component in the experiments (**I**, **II**). The excavated sediment with a loamy texture (Table 2) was heaped on top of the storage site and left to dry for the following nine months (**I**, **II**).

The sediment material can be used as a fertilizer in small amounts compared with the soil volume or as a growing medium. Therefore, two guidelines were chosen to assess the risk levels associated with various contents of organic contaminations and heavy metals in the sediment: standards set in the Finnish legislation for contaminated soil (MEF, 2007) and the limit values used for a high-quality compost suitable for agriculture in Europe (class A compost, Amlinger et al., 2004). The contents of all the measured elements in sediment were below the threshold values of contamination set by MEF, except for arsenic (As, 9 mg kg^{-1}), antimony (Sb, 3.7 mg kg^{-1}), and Zn (473 mg kg^{-1}). However, As and Sb levels were far below the lower guideline values (As <50 mg kg^{-1} and Sb <10 mg kg^{-1}), suggesting no ecological or health risks. The Zn content exceeded the MEF lower guideline limit (250 mg kg^{-1}) but was lower than the threshold limit set for class A compost (500 mg kg^{-1}). In addition, the contents of total polycyclic aromatic hydrocarbons (PAHs) and polychlorinated biphenyls (PCBs) in the sediment were much lower than the MEF threshold limits (Table 3).

Sediment total P content was much higher than that of the topsoil (Table 2). Also, total C and OM contents were much higher in the sediment. Results indicated that the content of soil test P was excessively high in the raw sediment and only approximately one-tenth of that in the topsoil material. The majority of total P was in calcium phosphate form. Soil pH was high and sediment pH even higher. Calcium content in the raw sediment was over three-fold greater than the recommended upper limit (4000 mg/l) according to standard Finnish soil testing methods, and its content was six times higher than that of the topsoil (**I**, **II**). Out of the analyzed nutrients, K content was low in the sediment and required supplementation. Sulfur (S) content was sixty-five times higher in the raw sediment compared with the topsoil. Also, B and Zn levels were distinctly high in the sediment (**I**, **II**).

Biochar (Table 2) was produced onsite in January 2017 by pyrolyzing the hardwood branches and split logs in a Kon-Tiki Garden kiln (Terra Magica GmbH, Grafenrheinfeld, Germany). The raw materials, mostly constituting 80–90% willow (*Salix*) and 5–10% birch (*Betula*), were collected from the lake shore in 2015–2016 (**I**, **II**). After pyrolysis, the biochar was activated by soaking it in a water:cattle slurry (7:3) mixture. In general, when comparing with European Biochar Certificate (EBC, 2022) criteria, the produced biochar showed a high degree of carbonization during the pyrolysis, and the total PAH content of the biochar was below the limit set for AgroOrganic grade biochar (4.0 mg kg^{-1}). The methods used for analyzing the biochar are presented in Table 3.

The Erikois-Viljo 8-4-8 (N-P-K) certified organic meat bone meal-based fertilizer (MBM, Honkajoki Oy, Honkajoki, Finland) was used in the experiments (**I**, **II**). An application rate of 1250 kg MBM ha^{-1} was calculated based on the requirement to deliver a total of 100 kg N ha^{-1}, resulting also in 50 kg P and 100 kg K ha^{-1} (**I**, **II**). According to Ylivainio and Turtola (2009), 18-19% of the P in MBM can be plant-available in the first year and the proportion of plant-available P can be increased over four years from 32% to 100%, depending on various factors such as crop and soil type and soil acidity. Also, 2% of the total N in the product was water-soluble (Elosato n.d.). The MBM application rate for additional nutrients ranged from 24 g ha^{-1} selenium (Se) to 125 kg ha^{-1} Ca (**I**).

Table 2. Selected physiochemical properties of the sediment, soil, and biochar representative samples.

Property	Sediment	Soil	Biochar
Bulk density (g cm^{-3})	0.33	1.18	0.13
pH	7.23	7.36	9.86
EC (dS m^{-1})	1.58	0.16	1.23
C (%)	17.47	2.15	86.7
N (%)	1.34	0.20	0.25
C:N	13	11	343
Total Fe (mg kg^{-1})	15000	11000	406
TP (mg kg^{-1})	2600	490	1296
Labile P (mg kg^{-1})	115	<15	–
Fe-P (mg kg^{-1})	830	220	–
Ca-P (mg kg^{-1})	1450	135	–
OP (mg kg^{-1})	405	245	–
Fe:P ratio	6	22	–
Sand (%)	40	60	–
Silt (%)	42	27	–
Clay (%)	18	13	–
BET SSA (m^2 g^{-1})	–	–	204
Ash (g kg^{-1})	–	–	92
VM (g kg^{-1})	–	–	165
Liming equivalence (g kg^{-1})	–	–	20
Carbonate–C (g kg^{-1})	–	–	43
H:C$_{org}$	–	–	0.02
Water extracted P (g kg^{-1})	–	–	0.001
Total K (g kg^{-1})	–	–	7.2
Water extracted K (g kg^{-1})	–	–	6.0
Hg (mg kg^{-1})	0.17	–	–
Total PAH (mg kg^{-1})	2.28	–	2.98
Total PCBs (mg kg^{-1})	< 0.1	–	–

BET SSA: the Brunauer–Emmett–Teller specific surface area, VM: volatile matter content, PAH: the content of total polycyclic aromatic hydrocarbons, H: hydrogen, Hg: mercury

3.2 Experimental design, setup, and samplings

3.2.1 Lysimeter experiment (I)

A nine-month lysimeter experiment was conducted to explore the effects of various sediment application methods on the soil nutrient status, nutrient leaching, and growth and nutrient uptake of ryegrass in Tartu, Estonia from February to October 2017 (Figure 2). The sediment and soil materials were collected in February 2017 from the storage site of excavated lake sediment. Moist soils and sediments were sieved and homogenized before packing the lysimeters. The mean gravimetric water contents of the soil and sediment were 0.27 and 2.48 m^3 m^{-3}, respectively.

Forty-eight PVC tube lysimeters (height 100 cm and inner diameter 10.78 cm) each with a capacity of ~8700 cm^3 were set up in a temperature-controlled room as a randomized complete block design with four replicates. The treatments were five sediment application methods and a control soil (*TS*), either in combination with MBM fertilizer or without (Figure 2): *TS*: 100 cm topsoil; *TS10*: 10 cm of soil on 90 cm of sediment; *TS15*: 15 cm of soil on 85 cm of sediment; *TS25*: 25 cm of soil on 75 cm of sediment; *TSB25*: 23 cm of soil plus 2 cm of biochar on 75 cm of sediment; *Sed*: 75 cm of sediment on top of 25 cm of topsoil.

For each 10-cm layer, the quantity of soil or sediment was weighed and added to the lysimeter and compacted to the target bulk density. The bulk density of soil, sediment, and biochar were 1.20, 0.33, and 0.147 g cm^{-3}, respectively, except for the first replicate where sediment was packed to 0.45 g cm^{-3}. At a 3–4-cm depth from the lysimeter's surface, 1250 kg ha^{-1} MBM fertilizer (100 kg N ha^{-1}) was applied and 100 seeds of ryegrass were planted. Tap water was used to irrigate the lysimeters, and the resulting leachate that percolated through the lysimeter columns was collected in containers located beneath the lysimeters.

Aboveground biomass (AGB) was sampled at six cutting times during 2017 (Figure 2). The amount of leachate was recorded at each harvesting time. A subsample of leachate was collected two weeks after planting, at the 1st cut, from the 1st cut to the 3rd cut, from the 3rd cut to the 5th cut, and at the end of the experiment (Figure 2).

After the end of the nine-month lysimeter study, the whole soil–sediment columns were removed from the lysimeter pipes and the soil column was cut into 10 equal sections, each 10 cm in height. The soil samples were taken from the top 30 cm of the lysimeter including 0–10-, 10–20-, and 20–30-cm sections, and mixed to form a composite sample. The soil cores with roots were placed into plastic bags and stored at +5 °C.

3.2.2 Viljandi field experiment (II)

A four-year field experiment with a randomized complete block design was established on the shore of Lake Mustijärv in May 2017 to determine the long-term (2017–2020) effects of recycled sediment application methods on plant growth, soil nutrient availability, and environmental parameters such as GHG emissions, N and P leaching, aggregate stability, and soil biota. The experiment was conducted in 2 × 2-m^2 individual plots with four replicates.

The treatments included three sediment application methods that were hypothesized to reduce nutrient losses from the recycled sediment via leaching, erosion, or gaseous losses, inspired by earlier results of the lysimeter experiment (**I**). Treatments also included a control soil from the lake shore: *Soil*: 100 cm pure topsoil; *Sed*: A 75–100-cm thick sediment layer (1650–2970 Mg ha^{-1}) varying according to the sloping bottom soil; *SB*: a 2.5-cm biochar layer (37.5 Mg ha^{-1}, delivered a total of 32.2 Mg C ha^{-1}) above the sediment; and *SSB*: a mixture of a 2.5-cm soil layer and a 2.5-cm biochar layer above the sediment.

The *Soil* treatment plots were prepared by removing the entire sediment layer from the 4-m^2 plot area with an excavator scoop to the depth of the original subsoil layer and filling it with the same topsoil peeled off from the sediment storage site. For the *SB* and *SSB* treatments, a 2.5-cm layer of biochar (SB) and 2.5-cm layers of both soil and biochar (SSB) were placed on the sediment and hand-mixed with rakes. All experimental treatments were tilled in two directions with a horizontal hand rotary tiller (Husqvarna FT900, USA) to a depth of 20 cm, resulting in an even mixing of all the materials applied to the top 20 cm.

All plots were fertilized with 1250 kg ha^{-1} MBM organic fertilizer at a 3–4-cm depth from the surface and mixed to a depth of 5 cm. Finally, a mixture of grasses with 45% red fescue (*Festuca rubra* L.), 35% Kentucky bluegrass (*Poa pratensis* L.), 15% ryegrass (*Lolium perenne* L.), and 5% white clover (*Trifolium repens* L.) were planted by hand-sowing on 1 June 2017. This plant mixture is a classic combination of grasses used for landscaping and agricultural purposes. The fast-growing perennial ryegrass rapidly establishes ground cover and is complemented with slower-growing but more stress-tolerant and longer-living red fescue and meadow grass. All three species are known to tolerate frequent mowing.

Aboveground biomass was sampled nine times (Figure 2) in the field experiment in 2017 (21.08.), 2018 (21.06., 21.08., 25.09.), 2019 (25.06., 8.08., 15.10.), and 2020 (7.07., 11.08.). All vegetation from a 30 × 30-cm^2 area at the center of the plot was cut at a 2-cm height with scissors. In June 2018, bespoke zero-tension lysimeters (Voll and Roots, 1999) with a 30 × 30-cm^2 metal sheet and a 2.5-L plastic container were installed at a depth of 30 cm in each plot. The amount of leachate was recorded, and a subsample was kept at −20 °C.

Soil samples were collected annually in August from 2017 to 2020: nine subsamples were taken from each plot from depths of 0 to 20 cm by a push probe with a 2.5-cm inner diameter and compiled. Directly after sample collection in 2017 and 2018, subsamples for microbial community measurement were taken from plastic bags using a sterilized spoon and transferred to separate plastic bags. All sampling tools were sterilized with ethanol (70%) between each plot sampling. The samples for microbial community analyses were transported in an icebox to the laboratory and kept frozen at −80 °C. For soil aggregate stability tests, four soil samples were taken from depths of 0 to 5 cm with the same push probe. Also, the initial samples were collected from each plot in June 2017 using the same procedure.

3.2.3 Lake studies (III)

To examine the formation of P internal loading in the restored Lake Mustijärv, sediment P dynamic studies were carried out in August 2017 and in June, August, and October 2018 (i.e., over two subsequent years following sediment removal). The studies included measuring the redox potential and pore water concentrations of soluble reactive P (SRP) and dissolved Fe in surficial sediments (uppermost 3-cm layer), calculating the SRP diffusive flux, and analyzing the surficial sediment compositions (P fractions, OM, and Fe content). For this purpose, three sampling sites in the lake's inflow (IF), center (CE), and outflow (OF) areas were chosen.

Sampling was conducted at each sampling site in August 2017 as well as in June, August, and October 2018 (Figure 2). Environmental factors with potential impacts on the biological, chemical, and physical changes occurring at the sediment–water interface were measured *in situ*. Using a Limnos sampler, water samples were collected from the surface water layer and the water layer overlying the lake bottom. The concentrations of variables potentially crucial for the recycling of P, such as SRP, TP, total Fe, and dissolved Fe, were determined from the lake water samples. Sediment samples were collected using an HTH gravity corer (Renberg and Hansson, 2008).

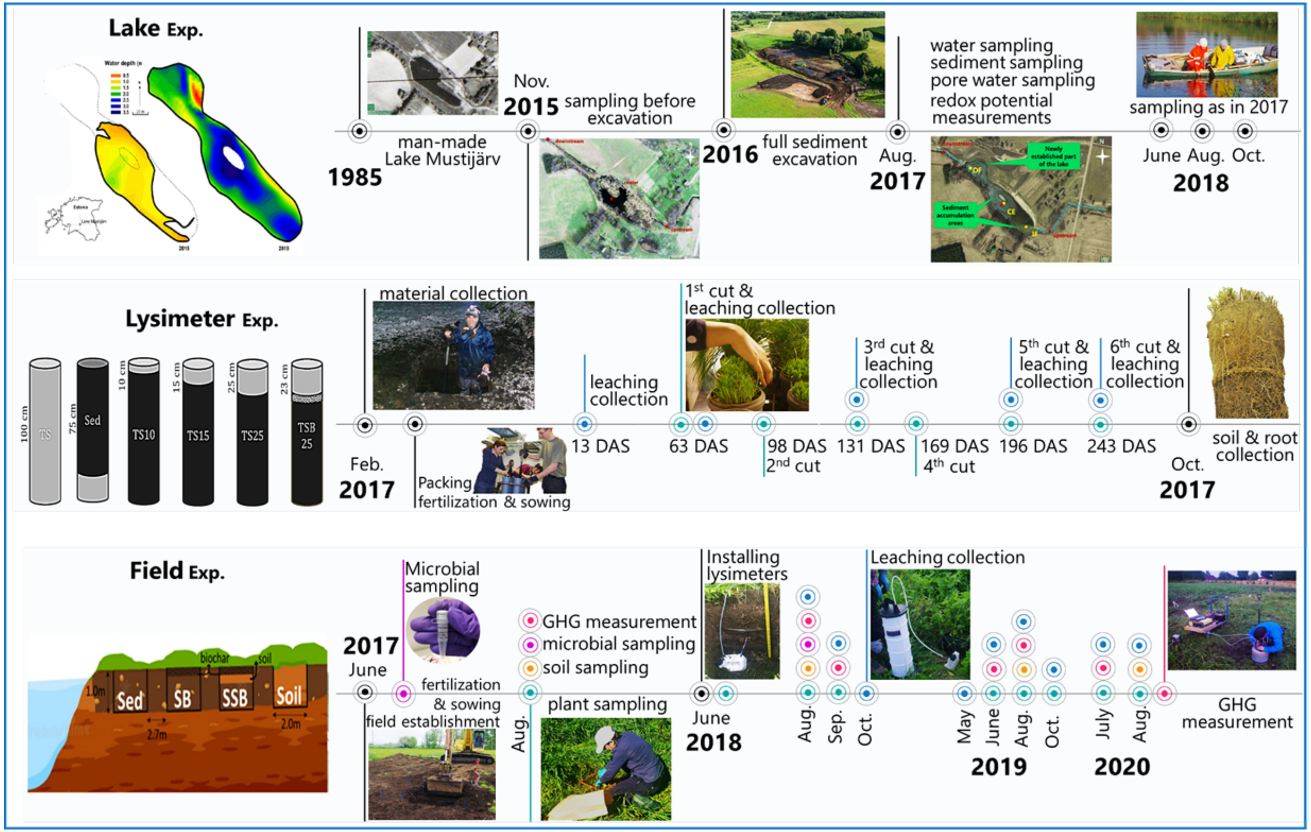

Figure 2. Timeline of the three experiments. Circles in green, blue, yellow, red, and pink colors represent plant sampling, leaching collection, soil sampling, greenhouse gas (GHG) measurements, and microbial sampling, respectively. DAS: days after sowing. Aerial photo for 2016: Kristjan Lust. Photos for 1988, 2015, and 2019: Estonian Land Board.

3.3 Measurements and analyses

3.3.1 Soil analyses

Standard Finnish soil testing methods were used to analyze the content of easily soluble P, K, S, Ca, Mg, sodium (Na), B, Cu, Mn, and Zn (Table 3). The EC and pH of the samples were measured from a 1:2.5 (w/w) soil:water mixture (**I, II**). The total C and N contents of ground samples were determined with a Dumas dry combustion method using a Leco CN analyzer (CN828, model 622-200-100, Leco Corporation, St. Joseph, MI, USA) from 2017 to 2020 (**II**).

Phosphorus fractionation was conducted for all samples of the lysimeter experiment and for the initial samples and samples from 2018 in the field experiment following a modified Williams protocol by Ruban et al. (2001). This method is proven to be one of the most appropriate for achieving comparability in an interlaboratory study (Ruban et al., 2001). The analysis resulted in the following fractions: total P (TP), organic P (OP), inorganic P (In−P), P bound to Al and Fe (hydr)oxides (Fe−P), and P bound to Ca (Ca−P). Additionally, labile P (P_{lab}) was extracted with 1 mol L^{-1} NH_4Cl as a part of the Hieltjes–Lijklema (1980) protocol. Also, samples were digested with HNO_3 (2.0 mol L^{-1}) to obtain the total Fe content. The concentrations of P fractions and total Fe were determined by inductively coupled plasma-optical emission spectroscopy (ICP-OES; Thermo-Fisher iCAP3600 MFCDuo, Thermo Fisher Scientific, Cambridge, UK) (**I, II**).

The carbon stocks of the sediment and soil growing media were calculated as follows:

$$C_{stock} = C \times BD \times Z,$$

where C_{stock} is the carbon stock (g m^{-2}), C is the carbon content (g kg^{-1}), BD is the dry bulk density (kg m^{-3}), and Z is the thickness of the sampled layer (m). The loss of C was calculated from changes in the C stock between 2017 and 2020 (**II**).

The aggregate stability of each growing medium was tested with a wet-sieving method as follows (**II**): 20 g of air-dried samples were sieved (0.63−2 mm) for three minutes and 4 g of aggregates from the 0.63−2-mm size class were placed on a 0.25-mm sieve. Then, samples were left to stand in 100 mL of deionized water for 15 minutes. After a 15-min saturation time, the wet-sieving apparatus (Eijkelkamp Agrisearch Equipment, Giesbeek, Netherlands) was turned on, and the aggregates were moved up and down in the water ca. 95 times for 3 min. The water and detached material were then transferred into plastic tubes and the suspension was let to settle for 21 h. A 25-mL sample was pipetted from the surface of the settled suspension into a turbidity meter cuvette, and turbidity was measured using a HACH 2100N turbidity meter

(Hach Co. Loveland, CO, USA). The water-stable material remaining on the sieve was collected and oven-dried (105 °C) for 21 h. The mass percentage of water-stable aggregates was calculated as the ratio of the dry mass remaining on the sieve and the dry mass originally collected for the wet-sieving (Soinne et al., 2014).

Soil penetration resistance as an indicator of soil compaction, was measured in the field in August and June 2017 and August 2018 (**II**). A hand-pushing penetrologger (Eijkelkamp, Giesbeek Equipment, Netherlands) was used for the measurements with a cone base area of 2 cm^2 and an apex angle of 60°. The penetration resistance measurements were made by pushing the penetrometer vertically to a soil depth of 40 cm at an approximated speed of 3 cm s^{-1}.

3.3.2 Plant analyses

After drying the plant samples at 60 °C for 72 h, the AGB dry mass was recorded (**I, II**). The species composition of the collected AGB in the field experiment was also determined (**II**). Later, the dry biomass of the 1st cut, a mixture of the 2nd and 3rd cuts, and a mixture of the 4th, 5th, and 6th cuts in the lysimeter experiment (**I**) and dry biomasses for each year in the field experiment (**II**) were ground. In the field experiment, representative ground biomass samples for each year were formed by combining all cut repeats within a year into a single sample for further analyses.

To determine the nutrient contents, 1 g of ground plant samples were dry-ashed in a muffle furnace adjusted to elevate the temperature to 500 °C for 2 hours and maintained for 1 hour (Table 3). The cooled samples were extracted in Erlenmeyer flasks that were kept on an electric stove with 50 mL of simmering 0.2 M HCl until the liquid volume had halved. The contents were poured into volumetric bottles and topped with deionized water to a final 50-mL volume. The solution was mixed and filtered through a slow filter paper (Whatman 589/3 ashless/Blue Ribbon, GE Healthcare UK Limited, United Kingdom) before being stored at 5 °C. The filtrates were analyzed for determining the concentrations of Al, B, Ca, Cd, cobalt (Co), chromium (Cr), Cu, Fe, K, Mg, Mn, Na, Ni, P, S, and Zn by ICP-OES (**I, II**). The total C and N contents of the plant samples were determined by the Leco CN analyzer (**II**). The plant nutrient uptake was calculated by multiplying the nutrient content by AGB (**I, II**).

In the lysimeter experiment (**I**), the root biomass, soil bulk density, and moisture content were determined from all soil cores of the four most contrasting treatments of Sed, *TS*, *TS25*, and *TSB25*. Before root washing, the bulk density and moisture content were determined by weighing the soil section and collecting a subsample into a metal tin for determining the dry-matter content (105 °C for 24 h). As for root biomass, the visible roots were collected into a container. Next, the mixture of roots and soil was manually washed using a 0.71-mm sieve with running water. Finally, the remaining material was washed with a hydropneumatic elutriation root washer (Gillison's Variety Fabrication, Benzonia, MI, USA).

3.3.3 Leachate analysis

The leachate samples from the lysimeter and field experiments were passed through Whatman blue ribbon filters. Prior to filtering, the filter papers were rinsed thrice with 2 M KCl and twice with MQ water. The samples were analyzed for $PO_4^{3-}-P$, $NO_3^{-}-N$, and $NH_4^{+}-N$ concentrations by spectrophotometry with an automated discrete analyzer (Gallery Plus ECM, Thermo Fisher Scientific, CA, USA). The amounts of nutrients in the leachate were calculated by multiplying the concentration of each nutrient (mg L^{-1}) by the amount of leachate (L m^{-2}) in the lysimeter experiment (**I**) and further divided by the number of days since the last leachate collection campaign (mg m^{-2} d^{-1}) in the field experiment (**II**).

3.3.4 Gas flux measurements

The fluxes of GHGs (CO_2, CH_4, N_2O) and NH_3 from the growing media were measured *in situ* in the field experiment (**II**) at several sampling campaigns through 2017–2020 (Figure 2). Measurements were performed where the AGB was collected in each plot using an automated Fourier Transform Infrared Trace Gas Analyser (Gasmet DX4015, Gasmet, Helsinki, Finland).

Gas concentrations were measured for 10 min after deploying the aluminum chamber in the field. The chamber was cylindrical (27 cm high, 31.5 cm in diameter; total volume 0.0196 m^3), with an electric fan attached inside to circulate the air during measurement. The gas sampling probe was air-tightly inserted inside the chamber, from which the air was continuously pumped to the analyzer and returned to the chamber via an outlet after the analysis. The first two minutes of measurements were discarded to avoid the probable immediate effects of chamber deployment on gas concentration and carry-over effects from previous measurements. The flux calculation was made by fitting a linear regression of gas concentration vs. time of measurement (Kalu et al., 2021). The r values of the regression that were lower than 0.4 were taken to indicate that no detectable gas flux was present.

3.3.5 Soil DNA analyses

Soil DNA was extracted from 0.25-g soil samples using a PowerSoil DNA isolation kit (MoBio, Carlsbad, CA, USA) for the initial samples from June 2017 and samples from August 2017 and 2018 (**II**). The quality of extracted DNA was checked with electrophoresis, and the concentration and purity of DNA were assessed using NanoDrop Spectrophotometer (NanoDrop Technologies, USA). The DNA extracts were stored at -80 °C until further analysis. The V4 region of bacterial 16S rDNA was amplified using primers 515F and 806R and the ITS2 region using primers fITS7 and ITS4 (Gilbert et al., 2014). The amplicons were 2 x 250-bp paired-end sequenced on an Illumina® MiSeq v2 platform at the Institute of Genomics, Tartu, Estonia. The raw nucleotide sequence data are available in the NCBI database under Bioproject PRJNA848979.

The sequences were filtered, de-noised, and clustered into operational taxonomic units (OTUs) with Mothur v1.46.1 (Schloss et al., 2009), as described in the standard operating procedure (Kozich et al., 2013). Briefly, paired-end reads were merged, and sequences containing ambiguous bases and more than eight homopolymers were discarded. The 16S rDNA sequences were aligned against a Mothur-formatted SILVA database (release 132, Quast et al., 2012) and the ITS2 sequences against the UNITE database (UNITE+INSD version 8.3, Abarenkov et al., 2021). Chimeras were removed using the "chimera.vsearch" command (Rognes et al., 2016). The sequences were assigned to OTUs at a 97% similarity threshold, and taxonomy was assigned using the SILVA database for bacterial sequences and the UNITE databases for fungal sequences. Singletons were removed using the "filter_taxa" function and alpha diversity indices Chao1, Shannon, Simpson, and Fisher were calculated with the "estimate_richness" function in the "phyloseq" package (McMurdie and Holmes, 2013) in R (Version 4.1.2, R Core Team, 2020).

3.3.6 Sediment P mobility analyses and calculations in the lake study

For redox potential measurements (**III**), two subsamples of the sediments extracted with the HTH gravity corer were collected into plastic tubes, where 2/3 of the tube volume was filled with sediments and covered by water. Using a redox microelectrode (Unisense RD100 microsensor, Aarhus, Denmark), the redox potential of the sediment was measured directly in the tube down to 3 cm below the sediment surface immediately after sampling. The measurements were repeated three times for each tube.

At each sampling location (section 3.2.3), three sediment cores for pore water extraction were concurrently obtained by the HTH gravity corer (**III**). Rhizon Soil Moisture Sampler (Rhizosphere Research Products, Wageningen, the Netherlands) were inserted horizontally at depths of 1, 2, and 3 cm to extract pore water of the sediment. The pore water samples were analyzed for SRP and Fe concentrations (Table 3). The diffusive SRP flux (mg P m^{-2} day^{-1}) was calculated according to Fick's first law of diffusion (Berner, 1980):

$$J = \Phi \times D_s \times \frac{\Delta C_{SRP}}{\Delta Z},$$

where ϕ is sediment porosity, D_s (cm^2 s^{-1}) is the diffusion coefficient of phosphates, which is $6.12 * 10^{-6}$ cm^2 s^{-1} at 25 °C (D25 °C) in sediment–water systems (Yuan-Hui and Gregory, 1974), and $\Delta C_{SRP}/\Delta Z$ is the phosphate concentration gradient between the pore water of sediment and the overlying water column. The SRP concentration of pore water in the 1-cm surface sediments ($\Delta Z = 0.5$ cm) and the SRP concentration of the overlying lake water were used for calculating the concentration gradient. The temperature dependence of the D25 °C was considered according to the Stokes-Einstein relation (Lewandowski and Hupfer, 2005):

$$D_i = D_{i,25^\circ\mathrm{C}} \times \frac{\nu_{25^\circ\mathrm{C}} \times T}{\nu_T \times T_{25^\circ\mathrm{C}}},$$

where T is the temperature during sampling in Kelvins, $T_{25^\circ C}$ is the absolute temperature at 25 °C (298.15 K), $\nu_{25^\circ C}$ is the dynamic viscosity of water at 25 °C (0.8903 g m^{-1} s^{-1}), and ν_T is the dynamic viscosity of water at a given temperature T (g m^{-1} s^{-1}).

Additionally, 3 cm of the uppermost surface sediments were sampled to analyze the P fractions, Fe content, and OM content of the sediments (Table 3, **III**). The P fractionation and Fe content in the sediment samples were conducted as described in section 3.3.2. The OM of the sediments was determined by loss on ignition (LOI) at 550 °C for 2 h (**III**). The surface and bottom-close water samples were analyzed for TP, SRP, total Fe, and dissolved Fe concentrations (Table 3, **III**). Finally, internal P loading from the anoxic surface in the summer of 2018 was calculated as a product of the anoxic factor (AF) multiplied by the release rate (RR) of P from the sediments (Nürnberg, 2009). The model has been proved to provide adequate estimates of internal P loading in polymictic lakes (Nürnberg, 2005).

The mean P diffusive flux (mg m^{-2} day^{-1}) from June to August was used as a measure of the P release rate (RR). The AF factor, which expresses the extent and duration of anoxia, was calculated from a well-established empirical model (Nürnberg, 2009):

$$AF_{pred} = -36.2 + 50.2 \log(TP_{summer}) + 0.762 \frac{Z}{A^{0.5}},$$

where TP_{summer} is the average TP in lake water during summer (µg L^{-1}), Z is lake mean depth (m), and A is lake surface area (km^2). Epilimnetic TP values from June to August were averaged as TP_{summer}. AF represents a number of days in a summer that a sediment area equal to the lake surface area is anoxic (Nürnberg, 2002). To roughly estimate external nutrient loading to Lake Mustijärv in June-October, the nutrient concentrations in the inflowing water were multiplied by the water discharge measured at the same time as sediment sampling in June, August, and October. These values were then averaged and multiplied by the duration of the summer period, which was approximately 150 days.

Table 3. Measurements and methods used in the experiments.

	Variables	Methods	Reference	Publication
Sediment and soil	pH	Soil-water-suspension 1:2.5 (w/w)	Vuorinen and Mäkitie (1955)	**I, II**
	EC	Soil-water-suspension 1:2.5 (w/w)	Vuorinen and Mäkitie (1955)	**I, II**
	Easily soluble P, K, S, Ca, Mg, Na	Acid ammonium acetate extraction, ICP-OES	Vuorinen and Mäkitie (1955)	**I, II**
	Easily soluble B	Extraction with hot water, ICP-OES	Berger and Truog (1939)	**I, II**
	Easily soluble Cu, Mn, Zn	Acid ammonium acetate-EDTA extraction, ICP-OES	Lakanen and Erviö (1971)	**I, II**
	P fractionation	Sequential extraction with 1 mol L^{-1} NaOH for Fe-P and 1 mol L^{-1} HCl for Ca-P. With another aliquot, 3.5 mol L^{-1} HCl for TP. Using the third aliquot, 1 mol L^{-1} HCl for In-P, and the residual was treated at 450 °C for OP.	Ruban et al. (2001)	**I, II, III**
	Labile P fraction	1 mol L^{-1} NH_4Cl extraction	Hieltjes–Lijklema (1980)	**I, II, III**
	Total Fe	HNO_3 digestion, ICP-OES		**I, II, III**
	Total C and N	Dumas dry combustion		**I, II**
	Organic matter content	Loss on ignition	Nelson and Sommers (1983)	**I, II, III**
	Moisture content	Delta-T WET2 sensor		**II**
	Temperature	Delta-T WET2 sensor		**II**
	Particle size distribution	Pipette method	Elonen (1971)	**I, II**
	Aggregate stability	Wet sieving	Soinne et al. (2014)	**II**
	Turbidity	Wet sieving, HACH turbidity meter	Soinne et al. (2014)	**II**
	Penetration resistance	Penetrologger		**II**
	DNA extracts	PowerSoil DNA isolation kit (MoBio)		**II**
	PAH concentrations	Reflux method	DIN 15527	**I, II**
	PCB concentrations	Extracted with acetone-heptane mixture (1:1, v/v) for 1 hour		**I, II**
	Hg concentration	Amalgamation technique (method 822L)		**I, II**
Plants	Aboveground biomass	Oven-drying at 60 °C for 48–96 hours		**I, II**
	Elemental contents	Dry-ashing at 500 °C for 2 hours, extraction with 0.2 M HCl, ICP-OES	Miller (1998)	**I, II**
	Total C and N	Dumas dry combustion		**I, II**

Leachates	PO_4^{3-}, NH_4^+, NO_3^- contents	Analysis of filtrates with a discrete analyser		**I, II**
Gases	CO_2, N_2O, CH_4, NH_3	Static chamber method and analysis with an automated portable FTIR analyser		**II**
Lake water	TP, SRP		ISO 15681-2	**III**
	Total Fe, dissolved Fe		ISO 17294-2	**III**
	Temperature, pH, DO, SPC	Multiparameter water quality analyser (YSI Professional Plus)		**III**
Lake sediment	Redox potential	Redox microelectrode		**III**
	Pore water SRP	Rhizon Soil Moisture Sampler	ISO 15681-2	**III**
	Pore water dissolved Fe	Rhizon Soil Moisture Sampler	ISO 17294-2	**III**
Biochar	pH	Biochar-water-suspension 1:5 (w/w)		**I, II**
	EC	Biochar-water-suspension 1:5 (w/w)		**I, II**
	Specific surface area	N_2 adsorption	ISO 9277	**I, II**
	Ash content	Gravimetric method (ashed at 500 °C for 3 hours)	DIN 51719	**I, II**
	Total C, H, N	Dumas dry combustion	DIN 51732	**I, II**
	VM content	Weight loss after heating at 910 ± 30 °C for 7 minutes	DIN 51720	**I, II**
	Total PAH		DIN 15527	**I, II**
	$CaCO_3$ equivalence	Reaction with 1 M HCl and titration with 0.1 M NaOH		**I, II**

EC: electrical conductivity; PAH: polycyclic aromatic hydrocarbons; PCB: polychlorinated biphenyls; SRP: soluble reactive P; DO: dissolved oxygen; SPC: electrical specific conductance; VM: volatile matter.

3.4 Statistical analyses

Statistical analyses were carried out in Rstudio Server 2022.02.2 Build 485. The data were checked for normality (Shapiro-Wilk test or Q-Q plot) and homogeneity of variances (Levene's test or residuals vs. fitted plot). Statistical significance was tested with either the analysis of variance (ANOVA) (**I, III**) or the linear mixed-effects model (**II, III**) using the "lme" function under the "nlme″ library. When linear mixed-effects models were carried out, post hoc tests were computed using the "emmeans" function ("emmeans" package) with the Tukey method for p-value adjustments and a significance level of $P < 0.05$ specified in the "cld" function ("multcompView" package), whereas Tukey HSD was carried out following ANOVA. Variables that did not meet the assumptions of normal distribution and homogeneity of variances were tested by various variance structures in the "nlme" package (Zuur et al., 2009) and the Box-Cox transformation (Box and Cox, 1964). The four-year averaged data were analyzed as repeated measures with a linear mixed-effects model using the "lme" function, where "growing medium" was the fixed factor and "block" and "year" were the random factors (**II**). The rank and linear correlations among variables were studied using Spearman (**I**) or Pearson analysis, respectively (**II, III**).

The relationships between environmental variables and Hellinger-transformed bacterial and fungal community data were tested using distance-based redundancy analysis (db-RDA) in the "vegan" package in R (**II**). The variables were chosen by forward selection with the "ordistep" function. Differential abundance analysis was performed in R package "DESeq2" (Love et al. 2014), and sparsely represented OTUs across samples were removed when the DESeq2-normalized count ("baseMean") was less than 0.6. Operational taxonomic units with a Benjamini–Hochberg corrected significance level of $P < 0.05$ were considered differentially abundant (**II**).

4 Results and discussion

4.1 Nutrients availability and phosphorus fractions in sediment-based growing media

4.1.1 EC and pH

In both the lysimeter experiment (**I**) and the Viljandi field experiment (**II**), *Sed* had a significantly higher EC value than the soil treatment (*TS* and *Soil* in the lysimeter and field experiments, respectively). The difference was most noticeable in the lysimeter experiment and during the first year of the field experiment, when the value was 0.3 dS m^{-1} for the soil treatment and in the range of 1.5−1.7 dS m^{-1} for *Sed*. An EC value of *Sed* exceeding 1 dS m^{-1} is considered high enough to raise concerns about hampering plant germination and water uptake (Viljavuuspalvelu, 2008). However, *Sed* produced a significantly higher AGB yield than the soil treatment on these occasions, which suggests little reason for concern. The difference in EC values between *Soil* and *Sed* persisted throughout the field experiment (**II**) despite the EC of the *Sed* decreasing from 1.47 in 2017 to 0.34 dS m^{-1} in 2020 (Table 4). In the field experiment, EC correlated best with easily available S ($r = 0.96$, $P < 0.01$). The high initial EC values were likely due to the high contents of sulfate in the sediment and S oxidation (Cifuentes and Lindemann, 1993).

The pH was neutral to slightly alkaline in all lysimeter and field experiment treatments (**I, II**). The pH in the lysimeter experiment was slightly lower in *Sed* than in *TS* (7.19 vs. 7.67, $P < 0.05$), while *Sed* had a higher pH than *Soil* in the field experiment, especially during the 3rd and 4th years ($P < 0.01$). Generally, the pH of lake sediments tends to be slightly alkaline or close to neutral (Håkanson and Jansson, 1983). As an example, the pH value of surface sediments from five Finnish and Swedish lakes ranged from 6.6–7.9 (Laamanen et al., 2019). Some studies have reported no effect of added sediments on soil pH (Baksiene and Asakaviciute, 2013; Edesi et al., 2020), whereas some have reported increased pH in sediment-amended soils (Mattei et al., 2018; Baran et al., 2019). The potential liming effect of sediment on acidic soils is important due to the effect on increasing nutrient availability and decreasing solubility of some heavy metals (Baran et al., 2019). Heavy metals are sparingly soluble under alkaline conditions (pH = 8.0) while their solubility increases under acidic conditions (pH = 5.0; Chuan et al., 1996). Hence, the neutralizing capacity of the sediment should be tested in the laboratory. Also, an experiment incorporating sediments to more acidic soils could be used to test any liming effect of the sediment material.

4.1.2 Total nitrogen and carbon

The total C content of all sediment treatments with an average value of 13% was very high in comparison to boreal soils while the C content in *Soil* treatment (2%) was much closer to the range of mineral cropland soils reported in the literature (Heikkinen et al., 2013, Kalu et al., 2021) (**I, II**). Soil C content is proportional to the amount of OM (Pribyl, 2010). Generally, increased soil C levels are associated with improved soil quality. The high LOI of the sediment material further indicated that it was rich in OM (**I**). Also, in the four-year field experiment (**II**), the average C stock of 8.3 kg m^{-2} in the sediment was 64% higher than that in the top 20-cm surface layer of *Soil* (**II**). The *Sed* treatment lost 1.2 kg m^{-2} C during the four-year experiment while no considerable changes were observed in *Soil* C stock. Simola et al. (2012) reported an average net loss of 150 g C m^{-2} $year^{-1}$ from the soil of drained forestry peatlands in central Finland. This loss is approximately half of that from the recycled sediments.

The average N content in all the sediment-containing growing media was 0.89%, which was six times higher than that in *Soil* over the four years of the experiment (Table 4). Also, the N content of the growing medium remained constant in treatments containing sediment (**II**), while the *Soil* N content decreased by 44% ($P < 0.05$) from the first to the second year (Table 4). This implied a constant supply of N from the OM decomposition of the sediment materials. According to the correlation network, as the years passed, the N content in *Soil* decreased, and the C:N ratio increased (**II**). In *Soil*, the increase of the C:N ratio is due to a decrease in *Soil* N ($r = -0.932$, $P < 0.01$) and is not related to C, suggesting that the N pool is more dynamic than the C pool in *Soil,* while a similar relation was not observed in the *Sed* treatment. When an organic substrate has a C:N ratio lower than 15, rapid mineralization releases plant-available N (Brust, 2019). Hence, a C:N ratio ranging from 10 to 18 in the growing media of the current study points to conditions that favored the N supply from the OM decomposition. However, the five-fold lower clay:C ratio in the sediment material compared with the soil suggested that the rate of N mineralization in the sediment was higher than that in the soil. This notion is based on the study of Soinne et al. (2021), who reported that net N mineralization correlates negatively with the clay:C ratio.

4.1.3 Phosphorus fractions

The sediment contained more easily soluble P than the soil treatment (**I, II**). Moreover, the potentially bioavailable P fractions (i.e. labile P and Fe-P) and total P were more abundant in the sediment-based treatments than in the soil treatment (**I, II**). Similarly, Canet et al. (2003) reported a higher content of available P (38 mg kg^{-1}) in the dredged sediment of Lake Albufera in Spain compared with a sandy soil. In our study, the sediment clay content was rather low and comparable to the clay content of soil (18 vs. 13%). In the sediment material, the occurrence of P adsorption by metal oxides may reduce due to the low clay content (Laakso, 2017) rich in Al and Fe (hydr)oxides (Sippola, 1974). This idea is consistent with previous studies reporting higher contents of available P in sediment materials with low clay contents (<20%) compared with agronomic soil (Mattei et al., 2018; Ugolini et al., 2018; Tozzi et al., 2020). Also, Laakso et al. (2017) reported a very low content of easily soluble P (3 mg L^{-1}) in constructed wetland sediments with a high clay content (>60%) compared with silty clay loam soil. Moreover, a sediment Fe:P mass ratio lower than 15 suggests a low availability of Fe sorption sites (Jensen et al., 1992), which may result in available P in sediments. An example of such sediments can be found in the shallow Lake Harku in Estonia, with a low Fe:P ratio (Heinsalu, 1994). An Fe:P mass ratio of 6 in the excavated sediment of Lake Mustijärv could further explain why the content of bioavailable P in the sediment was higher than that in the soil.

The OP fraction (40%) was the largest P pool in the soil material, which was within the common range in surface soil horizons (Weil and Brady, 2017). However, the Ca-P fraction (52%) in the sediment was the largest pool (**I, II**), which is considered a main fraction in the catchments dominated by alkaline soils (Delgado et al., 2000). The Fe-P pool was significantly depleted in the soil treatment at the end of the lysimeter experiment (**I**) and within two years of the field experiment (**II**). This is assumed to reflect the more pronounced contribution of Fe-P to P uptake compared with the other fractions. This notion was further supported by the robust positive correlation ($r_s = 0.71$; $P < 0.01$) of Fe-P with P uptake (**I**), while this correlation was not observed for other fractions of P in *Soil* (P_{lab}, Ca-P, and OP).

4.1.4 Other nutrients

According to the soil test results, the sediment material was considerably abundant in easily soluble S, Ca, Mg, B, Cu, and Zn (**I, II**), to the extent that the fertility level of these elements remained in the range of good to quite high in the fourth year of the field experiment (Table 4). The nutrients in sediment can be assumed to have partially originated from upstream agricultural soils, from where they were washed out by erosion (Kisic et al., 2002). The majority of easily soluble macro- and micronutrients therefore had significantly lower contents in the soil on the lake shore compared with sediment materials, except for K and Mn (Table 4). Thus, compared with the sediment-based treatments, the soil treatment had lower contents of Mg, B, Na, Cu, and Zn in plant tissues in the lysimeter experiment (**I**) and lower plant contents of Mg, Cu, and Zn in the first years of the field experiments (**II**). In addition, sediment had seven-fold more OM and a higher clay content compared with the soil. These likely increased the CEC of the sediment, which increases the retention of nutrient cations and prevents micronutrient deficiencies. Canet et al. (2003) also reported an increase in the CEC by sediment application and a significant increase in lettuce yield due to the nutrient contents of the added sediments.

Both sediment and soil were K-deficient, but adding a 2- to 2.5-cm layer of biochar (~200 kg K ha^{-1}) significantly increased the K status of the growing media (**I, II**). The higher K content was also reflected in plant tissues of biochar-treated growing media, particularly in the first year of application. However, these differences balanced out over the years (Table 5). Our results agree with earlier reports showing that adding hardwood biochar increased the K content of the growing media (Tammeorg et al., 2014, Kalu et al., 2021). Even if no significant differences were observed in the amount of soil test K between *Sed* and *Soil* in the first year, the concentration in plants was higher in *Sed* (Table 5). The difference in plants could be explained in part by the higher clay contents in the sediment, as they can contain potentially available K that may be solubilized from silicate layers (Barré et al., 2008; Paasonen-Kivekäs et al., 2009).

Also, the mineralization of potential K reserves from OM is promoted by high microbial activity, which could have been expected to be higher in the sediment treatments due to the higher C and N contents. Despite the content of easily soluble K ranging from poor to satisfactory levels (according to standard Finnish soil testing methods) in the sediment, a generous amount of K was incorporated in plant tissue (~30 g kg^{-1}), which was above the mean K level in forages (Suttle, 2010; Virkajärvi et al., 2014). Due to the suspected high reserve of K in the sediment, an HCl extraction of total K could have been provided to test for a more accurate prediction of reserve K over the longer term (Virkajärvi et al., 2014).

The Na content was low in all treatments (except sediment-based treatments during the first year). However, this may not have affected plant growth, as Na is usually a non-essential nutrient. Available Na can be helpful when K is deficient because some of the metabolic roles of K in plant cells can be fulfilled by Na (Maathuis, 2014), which was apparently not the case in this study.

The content of easily soluble S was over 20-fold greater in the sediment material than in the soil (**I, II**). However, a higher S content was detected in plants grown in *Soil* than those in the sediment-based treatments in the first year of the field experiment (Table 5). Moreover, the recommended S values in grasses range from 1.5–2 g kg^{-1} (Primary Industries Standing Committee, 2007) to 3–15 g kg^{-1} (Eurofins, 2022). Despite the good and quite high content (>150 g m^{-3}) of S in the sediment material (35–733 g m^{-3}), the range in *Sed*-grown plants was only 1.5–2.6 g kg^{-1} over the four years. The content of easily soluble S showed a significant positive correlation with the EC ($r = 0.96$, $P < 0.01$), with the same temporal trend over the years (**II**). A limitation in S uptake may have occurred due to the high salinity in the sediment treatments in the first year. The *Soil* was depleted of easily soluble S during the last years of the experiment. This was reflected in plant S uptake in the *Soil* treatment, where the S uptake was 75 kg ha^{-1} in 2017 and decreased to 9 kg ha^{-1} in 2020. However, the average plant S uptake from the sediment-based treatments was constant over the four years, with an average value of 24 kg ha^{-1}. This finding indicates the potential benefit of improving the S fertility status by adding sediment to the soil.

The easily soluble S content showed 16 times higher starting values for the sediment-based treatments in comparison to *Soil* (613 vs. 38 g m^{-3} in 2017), and the S contents sharply decreased over time (34 and 9 g m^{-3} in sediment-based treatments and *Soil,* respectively, in 2020). As most soils contain more than 90% of the total S in organic forms (Zhao et al., 2006; Weil and Brady, 2017), the mineralization of organic S is likely to make an important contribution to the plant-available S. Also, previous research documented that carbon-bonded S, or organic S in the reduced and intermediate valences, is the main source for S mineralization (Ghani et al.,1991; Zhao et al., 2006). Therefore, the high content of easily soluble S may have been released during the organic mineralization process of the sediment material. Later, a great proportion of the easily soluble S was likely leached during the four years, as the pH of growing media was above 7 and there is little adsorption capacity for sulfate in soils with a pH above 6 (Curtin and Syers, 1990).

Another reason explaining the low retention of S is the influence of other anions on sulfate adsorption. Phosphate ions can outcompete sulfate ions for adsorption sites on soil particles (Tabatabai, 1988), and the addition of phosphate to soils has been shown to increase sulfur leaching (Chao et al., 1962; Eriksen, 2009). Riley et al. (2002) reported that up 72% of applied S was leached from sandy loam soil fertilized with 50 kg ha^{-1} ammonium sulfate in the first year, which increased to 96% by the end of the third year of the experiment. Also, Sorrenti and Toselli (2016) showed that Ca, S, and Na were the most abundant elements leached through a sandy soil with a pH above 7. Although S leaching was not studied in this thesis, S deposition can cause problems to lakes such as lake acidification in Scandinavian regions (Wright and Henriksen, 1978). Sulfate, like nitrate, is negatively charged and not held tightly by clay particles. Similar management options can therefore be used to minimize the leaching of both nitrate and sulfate. Due to the high concentration and solubility of certain nutrients (i.e., P, S, Ca, Mg, B, Cu, and Zn) in the sediment, sediments should be applied to the soil according to the requirements of crops in a similar manner as fertilizers are applied to avoid environmental risks.

Table 4. Chemical properties of the growing media in the lysimeter and field experiments, 0-30 cm.

	Treatment	EC dS m^{-1}	pH	TC%	TN%	C:N	Acid ammonium acetate extractable (g m^{-3} soil)									
							P	K	S	Ca	Mg	Na	B	Cu	Mn	Zn
Lysimeter	Soil	0.26 c	7.67 a				11.7 d	59.0 b	25 c	2745 e	270 e	56.9 b	0.78 d	2.74 e	160 a	3 e
	Sed	1.74 a	7.19 b				96.6 a	24.1 e	1599 a	17107 a	614 a	97.0 a	3.52 a	9.79 a	99 c	161 a
	TS10	1.39 a	7.34 b				65.1 b	43.4 cd	952 b	11218 b	556 b	88.5 ab	2.27 b	6.44 b	124 bc	79 b
	TS15	0.88 b	7.38 b				56.8 b	41.3 d	361 c	8421 c	463 c	69.1 ab	1.77 c	5.50 c	133 abc	56 b
	TS25	0.40 c	7.62 a				28.1 c	53.3 bc	85 c	4330 de	342 d	76.0 ab	1.03 d	3.48 de	149 ab	17 d
	TSB25	0.36 c	7.69 a				27.5 c	81.6 a	62 c	4482 d	355 d	74.2 ab	0.97 d	3.90 d	151 ab	19 d
Field	**2017**															
	Soil	0.293 b	6.98 b	2.18 b	0.225 b	9.76 c	25.3 b	106 b	38.0 b	2455 b	203 b	9.38 c	1.03 b	2.20 b	53.8 a	4.70 b
	Sed	1.47 a	7.23 ab	12.9 a	0.760 a	13.5 b	73.3 a	52.5 b	733 a	14125 a	498 a	46.5 b	2.60 a	7.60 a	24.0 b	125 a
	SB	1.40 a	7.25 a	16.1 a	0.815 a	16.8 a	86.0 a	268 a	610 a	12025 a	498 a	57.0 a	2.35 a	7.38 a	19.3 b	125 a
	SSB	1.26 a	7.30 a	12.2 a	0.760 a	16.3 a	77.0 a	248 a	495 a	10850 a	475 a	51.0 ab	2.15 a	6.40 a	21.3 b	89.3a
	2018															
	Soil	0.205 b	7.18	2.11 b	0.133 b	16.0	18.3 b	102	12.3 b	2413 b	195 b	7.50	0.975 b	2.13 b	44.3 a	5.23 b
	Sed	0.830 a	7.20	12.3 ab	0.603 a	14.5	73.8 a	84.8	250 a	13875 a	450 a	24.0	2.60 a	8.08 a	22.5 b	121 a
	SB	0.715 a	7.38	15.6 a	0.737 a	16.5	78.5 a	151	190 a	12150 a	440 a	21.5	2.67 a	7.65 a	18.5 b	123 a
	SSB	0.583 ab	7.30	13.6 ab	0.782 a	17.6	74.5 a	158	127 a	9640 a	423 a	17.0	2.25 a	6.65 a	20.3 b	87.5 a
	2019															
	Soil	0.153 b	7.01 b	2.03 b	0.116 b	18.1	15.7 b	114 b	6.26 b	2143 b	170 b	6.53 b	0.884 b	1.83 c	45.2 a	2.80 b
	Sed	0.479 a	7.41 a	13.7 a	0.730 a	14.2	77.6 a	94.4 b	83.9 a	15369 a	427 a	16.4 a	2.61 a	8.18 a	19.0 b	117 a
	SB	0.474 a	7.45 a	13.7 a	0.826 a	16.5	81.7 a	177 a	45.6 a	13055 a	427 a	13.2 ab	2.59 a	7.55 ab	16.7 b	115 a
	SSB	0.392 a	7.45 a	12.6 a	0.813 a	15.7	68.3 a	176 a	56.8 a	11588 a	461 a	15.1 a	2.40 a	6.70 b	19.2 b	91.0 a
	2020															
	Soil	0.205 b	7.18 b	2.17 b	0.126 b	17.7 a	23.6 b	124	8.51 b	2913 b	203 b	6.92 b	0.981 b	2.27 b	42.8 a	9.25 b
	Sed	0.341 a	7.43 a	11.0 ab	0.582 ab	14.6 b	74.2 a	122	35.0 a	17067 a	410 a	14.0 a	2.63 a	8.24 a	19.3 b	131 a
	SB	0.404 a	7.36 a	14.9 a	0.611 ab	18.2 a	79.5 a	160	33.0 a	14048 a	422 a	12.8 a	2.61 a	7.77 a	18.3 b	124 a
	SSB	0.367 a	7.43 a	12.2 ab	0.797 a	15.4 ab	76.2 a	149	34.8 a	13354 a	419 a	11.6 a	2.62 a	7.24 a	19.8 b	105 a
Fertility class[1]																
quite high			> 7.0				> 50	> 500	> 150	> 4000	–	–	> 2	> 20	> 1000	> 50
high			6.6– 7.0				33 –50	350 – 500	50 –150	2600 – 4000	> 400	–	1.3 – 2.0	10 – 20	250–1000	20 – 50
good			6.2 – 6.6				20 –33	200 – 350	15 – 50	2000 – 2600	200 – 400	60 >	0.9 – 1.3	5 – 10	75 – 250	6 – 20
satisfactory			5.8 – 6.2				12 –20	120 – 200	10 – 15	1400 – 2000	120 – 200	45 – 60	0.6 – 0.9	2.7 – 5	25 – 75	2 – 6
passable			5.4 – 5.8				6 – 12	70 – 120	6 – 10	800 – 1400	80 – 120	30 – 60	0.4 – 0.6	1.5 – 2.7	12 – 25	1.5 – 2
poor			5.0 – 5.4				3 – 6	40 – 70	3 – 6	400 – 800	50 – 80	15 – 30	0.2 – 0.4	1.0 – 1.5	6 – 12	1.0 – 1.5
quite poor			< 5.0				< 3	< 40	< 3	< 400	< 50	< 15	< 0.2	< 1.0	< 6	< 1.0

Samples were collected at the end of the growing seasons. Mean values within the treatments followed by a different letter are significantly different, Tukey HSD at $P < 0.05$, n = 4. For interpretation of the treatments abbreviation, the reader is referred to the sections 3.2.1 and 3.2.2 of this thesis.

[1] The Finnish classification for coarse mineral soils (Viljavuuspalvelu Oy, 2008).

4.2 Effects of sediment-based growing media on plant growth and nutrient uptake

4.2.1 Plant biomass

In the last three years of the field experiment (**II**), the average annual yield (12.3 Mg DM ha^{-1}) was close to the average forage crop yield in Estonia (12.2 Mg ha^{-1}) in 2004−2017 (Statistics Estonia, 2021). In 2017, the yield of mixture of grasses in the *Soil* treatment was 83% lower than that in the *Sed* treatment (**II**) due to a significantly higher share of weeds (98%) in comparison to all sediment-based treatments, including *Sed*, *SB*, and *SSB* (81%). However, the overall yield of the grass mixture was consistent across all the treatments in the following years. This result contrasts with the findings of the lysimeter experiment (**I**), where the addition of the same sediment doubled the ryegrass growth in a controlled environment (Figure 3).

In the lysimeter experiment, the ryegrass yield of the first three cuts in the sandy loam soil (*TS*) was within the range of the average yield for common cultivars used in the European region (6.3 Mg ha^{-1}; Sampoux et al., 2011). Furthermore, applying a 75-cm thick top layer of excavated sediment significantly increased the ryegrass yield to 18.1 Mg ha^{-1}. The lack of a growth response in the field experiment may likely be due to the different plant species and their nutrient needs.

The yield of perennial ryegrass, which was the sole crop in the lysimeter experiment, reportedly improves relative to many other pasture species with increasing N and P supply (Waller and Sale, 2001). The easily soluble P in *TS* was below the satisfactory level at the end of the lysimeter experiment (**I**) while the *Soil* treatment had a sufficient amount of easily soluble P (> 15 g m^{-3}) across the four years of the field experiment (**II**; Table 4). In the lysimeter experiment, the significant positive correlation ($r_S > 0.4$; $P < 0.01$) of ryegrass yield with soil P content (particularly Fe-P) implied that ryegrass growth was facilitated by the nutrient-rich sediment (**I**). Previous studies reported the improvement of soil nutrient and OM content due to sediment addition, which has resulted in yield benefits in nutrient-deficient soils (Canet et al., 2003; Edesi et al., 2020; Brigham et al., 2021).

In addition, more diverse plant communities are typically more effective at acquiring more nutrients and water and transforming them more efficiently into biomass than less diverse plant communities (Nyfeler et al., 2011). This is likely due to species-specific interactions between plants and their specific soil biota (Hendriks et al., 2013). Thus, in comparison with the solely cropped ryegrass in the lysimeter experiment, the ability of the grass mixture in the field to acquire their needs may have offset the yield differences due to the different growing media.

Adding biochar to the sediment material did not affect ryegrass biomass in the lysimeter experiment (**I**) or the biomass accumulation by the grass mixture in the four-year field experiment (**II**). These results are in line with the meta-analysis by Schmidt et al. (2021) stating that consistent yield increases as a response to biochar addition have not been detected in alkaline soils.

Lake sediment as a growing medium generally provided similar biomass yields as agricultural soil in the four years of the field experiment (**II**). Similar findings, based mainly on pot experiments, have been reported in several studies; the sediment mixture had a comparable growth performance for lettuce (Canet et al., 2003), holm oak seedlings (Ugolini et al., 2018), ornamental Red Robin photinia (Mattei et al., 2018), and strawberry (Tozzi et al., 2020; Tozzi et al., 2021) compared with soils/peat.

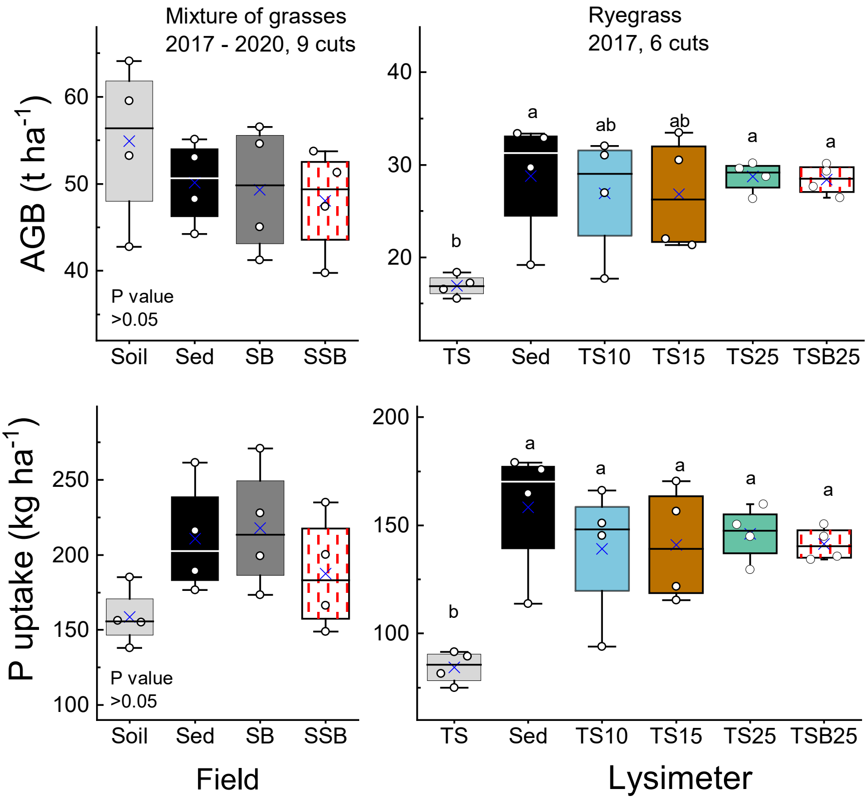

Figure 3. Effect of different sediment application methods on cumulative aboveground biomass (AGB) and cumulative phosphorus (P) uptake in the field and lysimeter experiments. For interpretation of the treatments abbreviation, the reader is referred to the sections 3.2.1 and 3.2.2 of this thesis.

4.2.2 Plant nutrient concentration and nutrient uptake

In the field experiment (**II**), the average N content in plant biomass was 25 g kg^{-1} in the treatments containing sediment, which was significantly (60%)

higher than in the *Soil* treatment for all years except 2017 (Table 5). Consequently, the four-year average for N uptake was 40% more in *SB* and *SSB* than in *Soil* ($P < 0.05$). On average, plants were able to annually take up 306 kg N ha^{-1} from the sediment-based treatments compared with 230 kg N ha^{-1} from *Soil*. Moreover, the soil total N content and plant biomass N content in *Soil* decreased by 44% and 22% ($P < 0.05$), respectively, from the first to the second year after MBM fertilization, while the N contents did not show considerable changes in the sediment-based treatments during the four years of the experiment (Tables 4 and 5).

The major share of N used by plants could not all be derived from the single application of MBM (100 kg N ha^{-1}) but rather from the OM of the growing medium. The 100-cm thick layer of organic-rich sediment (17% C) was likely a good source of bioavailable N during the four-year experiment. Also, the 100-cm profile of the *Soil* treatment was filled with material peeled from the uppermost 30-cm layer of topsoil in the sediment storage site (2% C), which provided a relatively good source of N along with MBM in the first year. The N uptake in *Soil* was declined from 273 kg ha^{-1} in 2017 to 169 kg ha^{-1} in 2020, while it increased by 74% in sediment-based growing media over the four-year period from 2017 to 2020. As in the Lake Mustijärv sediment, the slow release of nutrients from organic-rich materials, such as compost, has been reported previously (Diacono and Montemurro, 2011). Nitrogen mineralization from compost is limited in the short term. However, it maintains the N supply to plants through continuous mineralization several years after application (Diacono and Montemurro, 2011).

In the lysimeter experiment (**I**), the contents of easily soluble P were below the satisfactory level only in the *TS* treatment. As a result, the P content in plant tissue was significantly lower in *TS* than in *Sed* (4.5 vs. 5.5 g kg^{-1}), leading to twice the P uptake from *Sed* than from the *TS* treatment (158 vs. 84 kg P ha$^{-1)}$. In the field experiment (**II**), even though the P content in the growing medium was clearly different between the sediment-based treatments and *Soil*, none of the treatments showed P deficiency. Therefore, compared with *Soil*, the higher P contents in sediment-based treatments resulted in higher plant P content in the first years of the experiment but higher P uptake only in 2018 (78 vs. 46 kg ha^{-1}). As the soil fertility class and the magnitude of soil nutrient contents showed only little change over the years despite significant annual P uptake in all treatments, it could be stated that all growing media (fertilized with MBM) were able to provide a good P supply for four growing seasons.

Adding biochar to the sediment material did not affect the four-year average of N and P uptake, as indicated by insignificant differences between N and P uptakes in the *SB* and *Sed* treatments. However, plants were able to take up significantly more N in biochar-treated *SB* and *SSB* than in *Soil* (294 vs. 210 kg N ha^{-1}) and more P in *SB* than in *Soil* (54 vs. 40 kg P ha^{-1}). Biochar is well known to adsorb initial mineral N from the soil, and the captured N can be released over extended plant growth periods (Schmidt et al., 2021). In addition, meta-analyses suggest that biochars can increase plant P uptake if P is a limiting factor of plant growth (Gao et al., 2019; Glaser and Lehr, 2019), possibly due to the high anion exchange capacity of biochars (DeLuca et al., 2015). These notions may explain the higher N and P uptakes in the biochar-treated sediment than in *Soil* (**II**).

Moreover, adding biochar to the sediment material did not affect plant nutrient contents or the easily soluble nutrients of the growing media, except for K, Mg, and Na (**I**, **II**). The initial higher soil and plant K contents in biochar-treated growing media (**I, II**) were likely due to the high levels of total K in the hardwood biochar (7.2 g K kg^{-1}). Our results agree with earlier reports showing that the application of hardwood biochar increased the K content of the growing medium in boreal field soils (Tammeorg et al., 2014; Kalu et al., 2021).

The harvested yield in the sediment-base treatments fulfilled the nutritional requirements for livestock fodder regarding most nutrients (Bahamonde, 2016), confirming that cultivating animal feed on recycled sediment is feasible. In the fourth year (**II**), the plant N and S contents were insufficient for high-quality animal feed only in the *Soil* treatment (Freer et al., 2007; Suttle, 2010). In addition, only in the first year of the field experiment was the Zn content in both the *Sed* and *SB* treatments more than the maximum allowed quantity in feed (> 150 mg kg^{-1}). However, it dropped below the threshold value after 2017, suggesting a lower rate could be applied to match the fodder requirements. The grass tetany ratio of K:(Ca + Mg) was always lower than the threshold value of 2.2 (Georgievskii, 1982) in all treatments. Overall, there were low concentrations of heavy metals and organic contaminants in the sediment material of Lake Mustijärv (**I**), implying that no ecological or health risks should occur.

Table 5. Average plant nutrient concentrations in the lysimeter and field experiments.

	Treatment	C	N	P	K	S	Ca	Mg	Na	Al	B	Cd	Co	Cr	Cu	Fe	Mn	Ni	Zn
		g kg^{-1} DM								mg kg^{-1} DM									
Lysimeter	TS			4.52 **b**	34.4	2.71	8.65 **b**	4.19 **b**	0.74 **d**	17.5	7.89 **c**	0.096	0.102	0.369	5.64 **b**	66.3	27.9	0.47	23.9 **c**
	Sed			5.46 **a**	30.9	2.73	11.0 **ab**	6.79 **a**	5.99 **a**	30.6	19.8 **a**	0.232	0.176	0.332	9.52 **a**	65.4	16.4	0.33	138 **a**
	TS10			5.11 **ab**	30.3	2.47	11.9 **ab**	6.08 **a**	5.32 **a**	54.9	17.7 **ab**	0.074	0.038	0.280	9.73 **a**	89.0	17.8	0.28	122 **ab**
	TS15			5.18 **ab**	31.6	2.59	11.9 **a**	6.03 **a**	5.17 **ab**	48.5	17.0 **ab**	0.104	0.066	0.344	9.19 **a**	76.3	18.9	0.30	129 **ab**
	TS25			4.79 **ab**	32.0	2.81	10.0 **ab**	5.39**ab**	3.25 **bc**	37.3	14.5 **b**	0.070	0.027	0.225	10.22 **a**	78.6	17.1	0.40	93.2 **ab**
	TSB25			4.83 **ab**	35.9	2.98	8.70 **b**	4.17 **b**	1.64 **cd**	39.6	14.5 **b**	0.042	0.016	0.200	7.26 **ab**	71.8	18.9	0.26	77.6 **b**
Field	**2017**																		
	Soil	409 **a**	20.1	3.54 **b**	17.7 **c**	5.81 **a**	20.6 **a**	2.51 **c**	0.331	414	14.2	0.230 **a**	0.171	1.52	4.95 **b**	415	37.6	0.65	43.7 **b**
	Sed	391 **b**	25.8	5.78 **a**	29.5 **b**	2.62 **b**	14.4 **ab**	6.01 **a**	0.553	527	16.0	0.104 **b**	0.243	2.56	12.30 **a**	515	48.4	0.86	203 **a**
	SB	387 **b**	26.9	5.59 **a**	40.2 **a**	2.85 **b**	13.3 **b**	4.34 **b**	0.384	590	16.5	0.096 **b**	0.227	2.84	10.28 **a**	602	46.6	0.93	186 **a**
	SSB	389 **b**	24.5	5.83 **a**	34.7 **ab**	4.11 **ab**	16.7 **ab**	3.54 **bc**	0.462	800	20.5	0.097 **b**	0.274	2.96	9.78 **a**	724	45.3	1.09	150 **a**
	2018																		
	Soil	408	15.0 **b**	2.83 **b**	22.0 **b**	1.98	5.08	2.00	0.195	93.5	1.78	ND	ND	ND	1.11 **b**	138	38.6 **a**	ND	27.5 **b**
	Sed	398	23.6 **a**	4.51 **a**	33.4 **ab**	1.79	9.08	2.79	0.266	131	2.19	ND	ND	0.707	4.89 **a**	222	30.2 **ab**	ND	120 **a**
	SB	398	26.6 **a**	4.85 **a**	42.6 **a**	2.05	7.68	2.25	0.215	127	0.097	ND	ND	ND	4.65 **a**	165	25.4 **b**	ND	124 **a**
	SSB	404	25.1 **a**	3.93 **ab**	33.2 **ab**	2.07	7.67	2.13	0.203	132	2.84	ND	ND	ND	3.27 **ab**	228	25.7 **b**	ND	94.2 **a**
	2019																		
	Soil	408	16.3 **b**	2.85	26.3	0.968 **b**	5.51	2.07	0.127	66.8	ND	ND	0.027	0.010	1.70	126	50.5 **a**	ND	24.8 **b**
	Sed	407	24.3 **a**	3.40	27.6	1.55 **a**	6.83	2.25	0.130	17.4	ND	ND	0.026	0.069	3.33	88.1	17.4 **b**	ND	79.9 **a**
	SB	406	23.4 **a**	3.48	30.2	1.63 **a**	6.22	2.01	0.100	37.1	ND	ND	ND	0.204	2.99	93.0	16.6 **b**	ND	77.5 **a**
	SSB	410	24.4 **a**	3.06	28.3	1.44 **ab**	8.09	1.95	0.127	59.3	ND	ND	0.040	0.172	2.27	118	13.8 **b**	ND	63.1 **a**
	2020																		
	Soil	418	15.7 **b**	2.64	23.6 **b**	0.811	5.19	1.58	0.128	53.3	ND	ND	ND	ND	2.29	67.5	49.6 **a**	ND	21.2 **b**
	Sed	403	24.2 **a**	3.84	31.0 **ab**	2.00	13.1	2.11	0.094	28.3	ND	ND	ND	0.274	3.29	77.7	14.1 **b**	ND	55.4 **a**
	SB	401	26.4 **a**	4.21	33.0 **ab**	1.99	13.8	1.63	0.122	34.4	ND	ND	ND	1.86	4.14	86.4	11.8 **b**	ND	54.9 **a**
	SSB	402	26.7 **a**	4.00	34.6 **a**	1.83	14.3	2.25	0.094	38.4	ND	ND	0.020	ND	4.28	84.6	12.4 **b**	ND	56.5 **a**

Data show means of four replicates across growing media treatments. Mean values within the growing media treatments followed by a different letter are significantly different at $P < 0.05$. TS: 100 cm topsoil; Sed in lysimeter experiment: 75 cm of sediment on top of 25 cm of topsoil; TS10: 10 cm of soil on 90 cm of sediment; TS15: 15 cm of soil on 85 cm of sediment; TS25: 25 cm of soil on 75 cm of sediment; TSB25: 23 cm of soil plus 2 cm of biochar on 75 cm of sediment; Soil: Pure topsoil; Sed in the field experiment: 75–100 cm layer of sediment directly on topsoil; SB: 2.5 cm of biochar mixed with the top 20 cm of sediment; SSB: Mixture of 2.5 cm of soil and 2.5 cm of biochar on top of the sediment.

4.2.3 Root distribution of ryegrass

In the lysimeter experiment, the root biomass of ryegrass was determined only from the four selected treatments (*Sed*, *TS*, *TS25*, and *TSB25*) due to time and budget constraints. The total root mass of ryegrass grown in the 1-m high lysimeters was significantly higher in *TS* than in *Sed*, which no doubt contributed to the large root:shoot ratio (0.24) in *TS* (**I**). Resource limitations often lead to greater root:shoot ratios, as more biomass is devoted to belowground tissues to increase the root surface area for nutrient uptake (Publicover and Vogt, 1993). The uppermost 25-cm layer of the treatments selected for the root studies always contained some soil material, except for the *Sed* treatment. At an early growth stage, in all treatments, ryegrass had to take up the nutrients from the soil section, except for the *Sed* treatment. As a result, the root mass in the top 30 cm of the column was higher in the treatments where the top layer had soil (355–540 g m^{-2}) compared with *Sed* (134 g m^{-2}). The plant likely invested more growth into its root system in response to any deficiencies in the nutrient supply by the soil material, facilitating the access and uptake of nutrients and water in the growing medium (**I**).

In addition, the root mass in the 20- to 30-cm layer of the column was 72% higher in the *TSB25* treatment, where the sediment section began at 25 cm of the column, than in the *TS* treatment ($P < 0.05$). This may indicate that plants adsorbed nutrients from the sediment section of *TSB25* (**I**). When faced with a resource shortage, plants respond to nutrient-rich hotspots and proliferate roots to increase nutrient and water uptake (Caldwell, 1994; Robinson, 1994). Regarding the plant root mass grown in sediment material, Ferrans et al. (2022) reported a smaller root mass of lettuces in the dredged sediments from Malmfjärden Bay, the Baltic Sea, Sweden, compared with compost. They reported that the poor oxygenation conditions of sediment materials, with a high percentage of silt (60–70 %) and clay (10–20%) was possibly the reason for the smaller root mass in the sediment. However, the reported bulk density of both sediment and compost was as low as 0.20 g cm^{-3}, which should hardly cause problems due to low soil porosity.

4.3 Environmental impacts of recycling lake sediment to agriculture

4.3.1 Risk of N and P leaching

On average, the PO_4^{3-}−P and NO_3^-−N concentrations in the leachate of the soil treatments (*TS* and *Soil*) and *Sed* were below the threshold values for risks to waterbodies in both the lysimeter and field experiments (**I**, **II**; PO_4^{3-}−P < 4.4 mg L^{-1}, NO_3^-−N < 50 mg L^{-1}; European Commission, 2010). At the beginning of the experiments, the EC value of leachate was high (> 2 dS m^{-1})

in the treatments that contained sediment materials, but this decreased during both experiments. In the field experiment, the highest EC value of 2.9 dS m^{-1} was observed in the *Sed* treatment in October 2018, but it declined to 1.0 dS m^{-1} in 2020, which was in the range of freshwater streams (0.05 to 1.5 dS m^{-1}; Behar, 1997).

In general, the amount of leached nitrate, consisting of 96% of the total leached mineral N, was higher from the sediment material than from agricultural soil (**I, II**). The *TS10* and *TS15* treatments, with the highest proportion of sediment material, had the greatest amounts of leached mineral N (daily average 7 mg N m^{-2} d^{-1}) in the lysimeter experiment. In the *Sed* treatment, with a 25-cm soil layer beneath the sediment layer, the amount of N leached from the bottom of the lysimeter was slightly lower than from the *TS* (2.0 vs. 3.4 mg N m^{-2} d^{-1}, $P > 0.05$). However, in the *Sed* treatment of the field experiment, where the leachate was collected from a depth of 30 cm from the sediment surface, the daily average of mineral N leached in *Sed* was significantly higher than that in *Soil* from 2018 to 2020 (2.6 vs. 0.5 mg N m^{-2} d^{-1}).

At the end of the lysimeter experiment, the amount of P leached from the *TS* treatment was 55% lower than those from *TS15*, *TS25*, and *TSB25* ($P >$ 0.05), which contained 75 to 85 cm of sediment materials out of the 100-cm column (Figure 4). However, equally low amounts of P were leached from *Sed* and *TS* in the lysimeter experiment (11 vs. 12 µg P m^{-2} d^{-1} in *Sed* and *TS* respectively, $P > 0.05$), with a far greater content of easily soluble P in Sed. Labile nutrients are both easily leached and readily taken up by plants (Lehmann et al., 2003). The high levels of P uptake from the *Sed* treatment implies that the ryegrass growing in the sediment was abundant enough to take up significant amounts of soluble P and to efficiently prevent the P from being lost through leaching. In addition to the double P uptake in *Sed* compared with *TS*, the low amount of leached P from *Sed* could be due to the 25-cm soil layer beneath the sediment layer, which is much denser than the sediment material (1.18 vs. 0.33 g cm^{-3}). This idea is supported by the 66% lower amount of leached P from *Sed* compared with *TS25* while both had the same proportion of materials (**I**). Similar observation was reported by Yli-Halla et al. (2022) about the lower risk of P leaching in peat soil that has a mineral subsoil compared to the peaty subsoil due to the P sorption capacity of the mineral layer.

In the field experiment, as an average of the four years, the *Sed* treatment had five times more N in drained water and only 35% more N uptake compared with *Soil*, even if the total N contents in the *Sed* material were six-fold greater than in the soil (**II**). Also, the amount of leached P was six times greater in *Sed* than in *Soil* after four years (3 vs. 20 µg P m^{-2} d^{-1}). This is to be expected, as the amount of easily soluble P in *Sed* was over three times higher ($P < 0.05$), but P uptake was only 33% higher ($P = 0.127$) compared with *Soil* (except for 2018). Nitrogen and P leaching were high in the sediment treatments, possibly due to excessive amounts of N and P in relation to the plants' needs. Brigham et al. (2021) reported no differences between lake-dredged sediment and silty clay soil regarding loads of percolated P and N during the 123 days of soybean growth. They indicated that similar N and P leaching may be due to similar N and P contents of the soil and sediment materials. Also, incorporating nutrients into the soybean biomass resulted in rapidly decreasing loads in the percolated water over the growing season.

In addition, the mean soil penetration resistance at depths of 0–10 cm and 0–20 cm from the surface was significantly lower in treatments containing sediment than in *Soil* in 2017 (**II**). This implied that the leaching may be facilitated by pathways of loose and hollow sediment materials. However, one year later in August 2018, the only significant difference observed was between *Soil* and *SB* (0.75 vs 0.61 MPa) at 0–10 cm depth.

When the *TS25* and *TSB25* treatments were compared, I observed that adding 2 cm of biochar between the soil and sediment materials reduced the cumulative leaching of P and N by 47% and 51%, respectively (without a statistically significant difference) in the lysimeter experiment (**I**). In the field experiment, the N and P concentrations in the leachate from the biochar-treated sediment was not statistically different from concentrations in the *Soil* treatment (**II**). A comparison of *Sed* and *SB* treatments showed that occasionally applying 2.5 cm of biochar to the sediment materials significantly reduced the concentration of leached P from the sediment (**II**). A similar short-lived beneficial effect of biochar was observed for the concentration of mineral N in the leachate from *Sed* in July 2020. According to some studies, the labile C fraction in biochar may temporarily immobilize nitrate, reducing the amount of N that leaches from the soil to the environment (Kolb et al., 2009; Tammeorg et al., 2012). However, the total effects of added biochar on the leaching of N and P from the sediment were not significant across all four years of the study.

Yli-Halla et al. (2022) reported that the leaching of N and dissolved P increased in subsurface discharge waters upon increasing peat depth. These observations along with our findings, suggest that instead of using a thick layer of sediments as a growing medium, applying sediment with similar rates as the application of soil amendments may minimize nutrient leaching back into the lake and further help mitigate eutrophication in the lake.

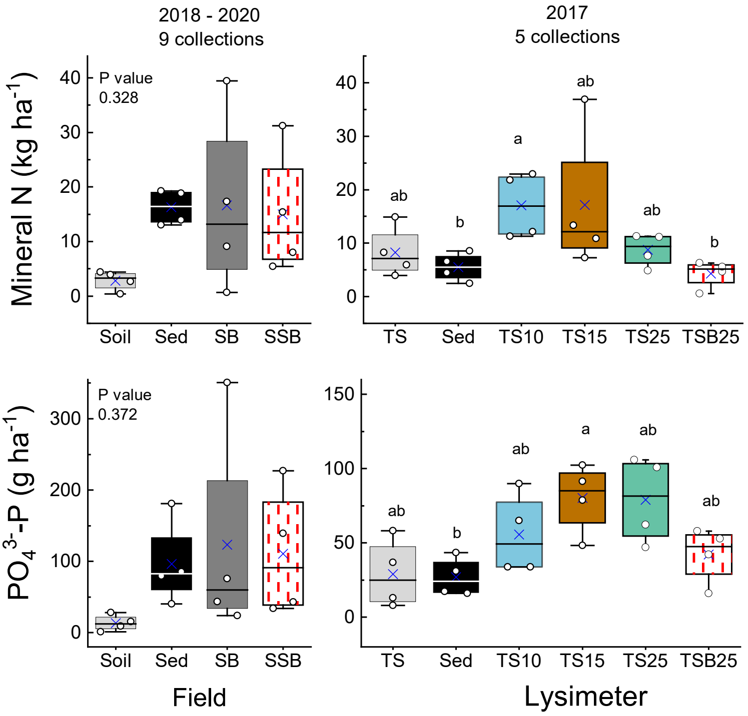

Figure 4. Effects of different sediment application methods on the cumulative leached amounts of mineral N (NO_3^-−N and NH_4^+−N) and PO_4^{3-}−P in the field and lysimeter experiments. For interpretation of the treatments abbreviation, the reader is referred to the sections 3.2.1 and 3.2.2 of this thesis.

4.3.2 GHG emissions by added lake sediment

The CO_2 emissions from *Sed* did not differ statistically from those in *Soil* at any single measurement time, except in August 2017, when the *Sed* treatment emitted 88% more CO_2 than *Soil* (589 vs. 313 mg CO_2−C m^{-2} h^{-1}). However, the four-year average for CO_2 emissions from the sediment-based treatments (*Sed*, *SB*, and *SSB*) was significantly higher than the average for the *Soil* treatment (579 vs. 400 mg CO_2−C m^{-2} h^{-1}). The four-year average for N_2O fluxes in all treatments was broadly similar ($P > 0.05$), although, during the first year after sediment application, the N_2O emissions were higher from *Sed* and *SB* treatments than from *Soil,* which acted as a sink of N_2O (-420 µg N_2O−N m^{-2} h^{-1}).

The higher CO_2 and N_2O emissions from recycled sediment materials during the year of application may be due to OM decomposition. The excavation and cultivation of organic-rich sediment allow oxygen to enter the sediment, which initiates the decomposition of the stored OM, and, in turn, increases the CO_2 and N_2O emissions (Kasimir-Klemedtsson et al., 1997). This notion was further supported by the contents of total C and N in *Sed* being more than four-fold those in *Soil* in 2017. A C:N ratio below 15 in the sediment material further supports the idea of significant microbiological decomposition of OM in the sediment material. The emission of 208 to 1041 mg CO_2−C m^{-2} h^{-1} from cultivated organic soils with a 0.2-0.8 m-thick layer of peat (> 40% C_{org}) during July and August in Finland (Yli-Halla et al., 2022) was in the same range as that of our study. Moreover, soil moisture is the single most important soil factor determining soil gas emissions by controlling microbial activity and all related processes (Oertel et al., 2016). The CO_2 emissions were positively correlated with soil temperature and moisture ($r > 0.595$, $P < 0.05$) in *Soil* but not in *Sed*. This suggests that factors other than temperature and moisture were more critical for determining the CO_2 emissions from *Sed* in this study. The differences between *Sed* and *Soil* regarding the CO_2 and N_2O emissions faded after the first year.

Although CO_2 emissions were greater in *Sed* than in *Soil* in 2017, the biochar-treated sediment (*SB*) and *Soil* did not differ statistically in this regard. The biochar produced at high pyrolysis temperatures (>500 °C) generally showed a high efficacy in reducing GHG emissions (Schmidt et al., 2021). Wood-derived biochars are reportedly associated with lower CO_2 emissions, as they have less labile C compared with straw-derived biochar (Awad et al., 2018). However, as the CO_2 emission from *SB* did not differ statistically from the emissions in *Sed*, even such a slightly beneficial effect of biochar in the reduction of CO_2 emission was not confirmed in this study.

Despite the excavated sediment exhibiting higher CO_2 emissions than *Soil*, it is important to consider that the eutrophic shallow (<20 m) lakes themselves are also important sources of GHGs (Li et al., 2021; Sun et al., 2021; Li and He, 2022). Huttunen et al. (2003) reported methane (CH_4) emissions up to 8 mg m^{-2} h^{-1} from eutrophic lakes with agricultural catchments. Sun et al. (2021) showed that a shallow eutrophic lake (~0.7 m) in a semi-arid region of China had mean CO_2 and CH_4 emission levels of 66 and 4.0 mg m^{-2} h^{-1}, respectively. The low oxygen content, abundant OM and nutrients, dominant primary producers, and harmful algal blooms in eutrophic freshwaters have been found to favor GHG emissions (Li et al., 2021). As there was no detectable flux of CH_4 emissions from the excavated sediment of Lake Mustijärv, recycling lake sediment to agricultural use could be a promising way to reduce GHG emissions on an ecosystem scale, as it has the potential to decrease the CH_4 emissions from both the lake and neighboring fields. Therefore, the magnitude of any GHG emissions from an unexcavated lake in future studies would provide an important larger frame of reference to the GHG emissions resulting from the agricultural use of excavated lake sediment.

4.3.3 Response of aggregate stability to added lake sediment

The proportion of water-stable aggregates (WSA) was high in all treatments, with the highest value of 98% in *SB* in 2017 and the lowest value of 85% in *Soil* in 2019. The *Sed* and *SB* treatments constantly displayed significantly higher aggregate stability than the control soil in the first three years of the experiment. In our study, the higher stability of aggregates in sediment-based treatments implied a higher proportion of labile OM in the sediment material. According to Sarker et al. (2018), stable OM is less capable of improving aggregate structure than labile organic materials when incorporated into the soil.

In addition, a high salt concentration in soil water is known to promote the flocculation of colloids and enable the formation of chemical bonds between surfaces, thus enhancing aggregate formation (Uusitalo et al., 2012; Heikkinen et al., 2019), which is in agreement with the significantly higher EC values in the sediment material. As the breakdown and slaking of soil aggregates produces both runoff-promoting crusting and easily transportable particles, aggregate stability is used as an indicator of soil erodibility (Le Bissonnais, 1996). Although heavy rain can induce runoff even without crusting and wash away aggregates and fragments of larger size, the stable macroaggregate content of the topsoil correlates negatively with both runoff rate and soil losses (Barthes and Roose, 2002).

The turbidity values in our study, indicating the detachment of colloidal particles after wet-sieving, did not differ statistically in any treatments, except for the first year of the experiment, when *Soil* had significantly higher turbidity (1.85 NTU) compared with *Sed* and *SB*. Higher turbidity values indicate that the growing medium is prone to the detachment of colloid-scale particles, thus increasing the transportation of particle-bound nutrients, including P, to surface waters.

In general, using sediment materials as a growing medium did not increase the erodibility risk of the aggregates. This result was more pronounced in the first years of the sediment application. Also, biochar application did not change aggregate stability, possibly because the sediment aggregates already had high stability. According to a meta-analysis, biochar additions increase soil aggregation in neutral and acidic soil but not in alkaline soil (Islam et al., 2021), which is in line with the pH > 7.2 in sediment-based growing media in the field experiment.

4.3.4 Microbial community of added lake sediments

Twelve phyla accounted for 92% of the relative bacterial abundance, including Proteobacteria, Bacteriodetes, Chloroflexi, and Actinobacteria (Figure 5). In this study, 34 bacterial phyla and 274 bacterial genera were differentially abundant in at least one of the treatments. Seven phyla dominated the fungal communities, with 99% of the relative abundance including Ascomycota, Mortierellomycota, and Basidiomycota (Figure 5). Sediment treatments had a lower level of bacterial biodiversity compared with *Soil*. However, sediment and soil materials showed similar fungal species richness and evenness (Figure 5). In addition, a different distribution pattern of dominant genera was observed in the sediment-based growing media in comparison with *Soil*. The differences in bacterial and fungal community compositions were mainly associated with differences in soil Zn, Mn, and TN (29% of variation) and soil Zn, Mn, K, and TC (46% of variation), respectively.

The relative abundance of Proteobacteria was higher in the *SB* and *Sed* treatments than in *Soil* in 2018. Members of this group have diverse functions such as N fixation, the degradation of toxic compounds, and CH_4 oxidation (Aislabie et al., 2013). Also, Proteobacteria include soil P-mobilizing bacteria expressing a large amount of acid phosphatase, which played an important role in the mineralization of refractory organic P (Zhang et al., 2021). In addition, Chloroflexi, with the potential to solubilize biological P fractions (H_2O-P and NaOH-P; Luo et al., 2021), was more abundant in sediment-based treatments in 2018. The higher relative abundance of Proteobacteria and Chloroflexi in the *Sed* treatment is consistent with the higher level of easily soluble P in this treatment. Moreover, Actinobacteria and Planctomycetes, with the potential to accumulate bioavailable P fractions (Luo et al., 2021), had higher abundances in *Soil* than in the sediment-based treatments.

Besides bacteria, also fungi can solubilize and accumulate P (Ye et al., 2015). However, different from bacteria, fungi may solubilize and accumulate more recalcitrant P (Ye et al., 2015). The fungal phyla of Ascomycota and Chytridiomycota can solubilize both bioavailable and refractory P fractions (Luo et al., 2021). In our study, the relative abundance of Ascomycota was higher in *Soil* than in other treatments, while Chytridiomycota had a higher abundance in the sediment-based treatments in 2018. Both Basidiomycota, which are known as litter decomposers able to degrade even complex and recalcitrant OM (Hellequin et al., 2018), and Actinobacteria phylum, as a vital part of the decomposition of OM and the C cycle (Chen et al., 2022), had higher abundances in *Soil* than in Sed.

Calditrichaeota, Fusobacteria, and Atribacteria were the unique phyla in sediment representative material collected in June 2017. Calditrichaeota is widespread in marine sediments and has the capacity to degrade detrital proteins (Marshall et al., 2017). Fusobacteria are anaerobic bacteria commonly found in aquatic environments, especially sediments with available OM (Maintinguer et al., 2015). Atribacteria are most predominant in organic- and hydrocarbon-rich sediments (Lee et al. 2018), and can ferment a variety of organic compounds (amino acids and oligosaccharides) and acetate oxidation (Baker et al., 2021). These unique phyla may express various organic carbon mineralization pathways in sediment-based treatments.

At the genus level, *Lysobacter,* which reportedly facilitates the mineralization of labile organic P and stable residual-P (Zhang et al., 2021), was more abundant in *Soil* than in the sediment-based treatments, although its relative abundance was low. *Sulfuricurvum* and *Thiobacillus,* with relatively high abundances in the community of the current study, are known to have degradation genes for toxic organic compounds and various mechanisms for heavy metal tolerance (Handley et al., 2014; Liu et al., 2018; Tian et al., 2020). They were more abundant in sediment-based treatments than *Soil*. *Sulfuricurvum* is also able to oxidize S and make it available for plants. The higher abundance of *Sulfuricurvum* in sediment-based treatments is consistent with the higher easily soluble S in these treatments.

The nitrogen-fixing *Terrimonas* (Khan et al., 2015) and *Cellvibrio* were more abundant in *SSB* and *Sed* in 2017 and in sediment-based treatments in 2018, respectively. Also, the nitrifier genus *Nitrosomonas*, which oxidizes ammonia to nitrite, was more abundant in all sediment-based treatments in both years. Also, *Claroideoglomus claroideum,* a fungal strain involved in improving soil N, P, and K fertility (Rashid et al., 2016), was more abundant in *Sed* than in *Soil* in 2017. In general, this research helped to expand the knowledge concerning bacterial and fungal communities associated with excavated sediments from a eutrophic shallow Lake Mustijärv.

Bacterial and fungal species diversities were not affected by biochar addition to the sediment material (**II**). In addition, *Sed* and *SB* had similar community compositions. However, the *Firmicutes,* which is known as a P-mobilizing phylum (Luo et al., 2021), had bacterial OTUs whose relative abundances were significantly higher in biochar-treated *SB* and *SSB* compared with control *Soil*. Also, the nitrifier *Nitrospira* (oxidizing nitrite to nitrate) with a relatively high abundance in taxa was more abundant in the biochar-treated growing media in 2018.

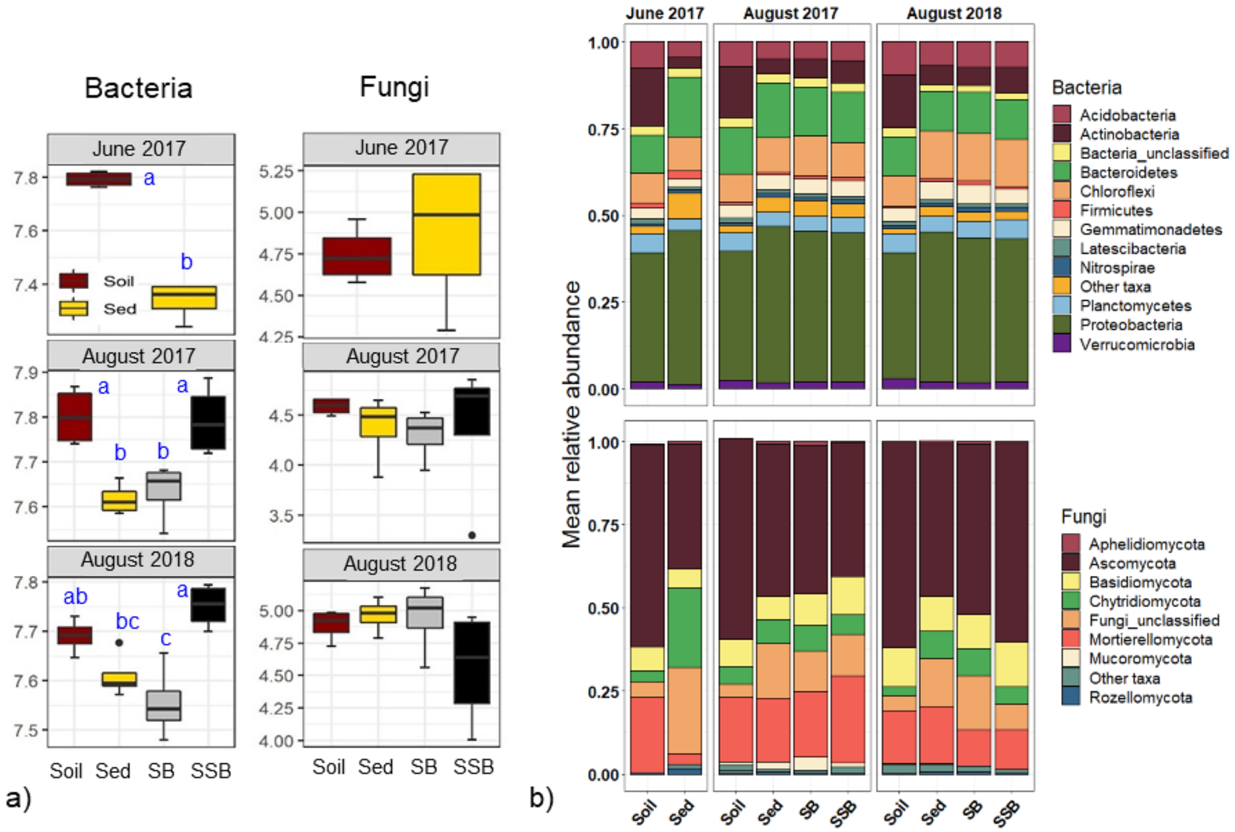

Figure 5. The Shannon index for bacterial and fungal operational taxonomic units (OTUs) (a); Relative abundance of most dominant bacterial and fungal taxa at phylum level in the field experiment (b). For interpretation of the treatments abbreviation, the reader is referred to the sections 3.2.1 and 3.2.2 of this thesis.

4.4 Internal P loading in a lake after sediment excavation

The excavation of approximately 7500 m^3 of sediment from Lake Mustijärv resulted in the removal of around six Mg of total P from the lake sediments (**III**). After sediment removal, the potential for P release from the sediments decreased, as the P accumulations (legacy P) were completely removed with the sediment. Nevertheless, high external nutrient loading (about 6.6 g P m^{-2} $summer^{-1}$ and 47 g N m^{-2} $summer^{-1}$) continued to supply nutrients in the years following sediment excavation. Exceptionally high external nutrient loading was caused largely by upstream riverbed excavation. As a result, large amounts of sediment were found to concentrate in the sediment accumulation basins introduced at lake restoration. The sediments in these basins tended to release P (calculated as the diffusive P flux) at much higher rates than the reference area (a newly established lake area close to the outflow).

Moreover, the diffusive P fluxes in the restored lake were at the level of the release rate reported by Nürnberg (1984) for hypertrophic lakes with anoxic conditions, resulting in calculated internal P loading values as high as 1.4 g m^{-2} $summer^{-1}$. In general, high external nutrient loading is often reported as a reason for unsuccessful lake restoration, including sediment removal (Bormans et al., 2016; Cooke, 2016; Lürling et al., 2020). In the current study, the high external nutrient loading also explained the phenomenon of high internal P loading in a lake restored recently by complete sediment removal. Similarly, Kragh et al. (2022) showed the internal P load to play an important role in the P budget of recently established lakes.

4.4.1 Factors behind sediment P release after lake restoration

During all studied occasions (August 2017, and June, August, and October 2018), the redox potential at the sediment surface was close to the critical value (200 mV), supporting the reduction of ferric iron. The pore water SRP concentration in the surface sediment was 1.7 times greater and the mean diffusive flux of SRP was 1.6 times higher two years after sediment removal than in 2017. Among the parameters for water overlying the lake bottom, dissolved oxygen (DO) was 4.7 mg L^{-1} in 2017 (as a mean of three sampling sites,) while it decreased to 0.7 mg L^{-1} in 2018. The DO value close to zero (0.06 mg/l) was registered only at CE in 2017, while anoxia-indicating values of DO (<2 mg L^{-1}; Nürnberg et al., 2002) were registered at all sampling sites already in 2018 (0.26, 0.47, and 1.28 mg L^{-1}). In addition, the concentrations of total P, SRP, and total Fe in near-bottom water were much higher in 2018 than in 2017.

In comparison with August 2017, the Fe-P fraction constituted a larger amount of TP in 2018. Also, the sampling location in close vicinity to the upstream (IF) had the greatest share of Fe-P in the sediment TP pool over the entire lake. The Fe-P has often been shown to be of anthropogenic origin

(Ruban et al., 2001), and external P loading is one of the main contributors of Fe-P content in sediment (Gao et al., 2005; Zhu et al., 2013). Also, a significant increase in the concentration of sediment labile P was observed between 2017 and 2018 (10 vs. 26 mg kg^{-1}; $P < 0.05$). In addition to the external loading that provided a pool of potentially releasable P forms, the significantly higher temperature during sampling in 2018 than in 2017 could partially explain the higher P flux in 2018. These findings indicate more favorable conditions for P release in 2018 than in 2017 (August data).

In 2018, the seasonal pattern in the diffusive flux of P reflected the seasonal variation in the concentration of P in lake water, suggesting high implications of P release from sediments for lake water quality. As an average of three sampling sites, the highest values of the P flux were observed at highest water temperatures in August 2018 (22 mg P m^{-2} day^{-1}), which was 76% greater than in June and 2.4 times greater than the flux in October ($P < 0.05$; Figure 6). Such variations in P release rates were expected, given that temperature affects P mobility through a number of processes including stimulating desorption, mineralization of newly settled organic matter, decrease in redox potential, and promoting diffusion and transport from the deeper sediment layers (Søndergaard et al., 2003; Søndergaard et al., 2013). Of the sediment TP components (P_{lab}, Fe-P, Ca-P, and OP), the share of Ca-P in TP (45%) was seasonally stable (Figure 6) and the concentration of OP decreased from June (410 mg kg^{-1} DM) to August (337 mg kg^{-1} DM). On the other hand, the concentrations of potentially algal-available P (fractions Fe-P and P_{lab}) in the sediment experienced notable seasonal changes in 2018 (Figure 6). The lowest concentration of Fe-P and P_{lab} was 390 mg kg^{-1} DM in June and the highest in October (488 mg kg^{-1} DM). The lowest share of sediment Fe-P and P_{lab} fractions in the sediment TP was in June (24%) and the highest was in August (33%).

A strong correlation of Fe-P with the diffusive flux (Figure 6) suggested that this form contributed to the P release. Moreover, Fe-P was easily released at reduced conditions of surface sediments and the overlying water column in Lake Mustijärv. Hence, both anoxia and the pool of mobile P supported the high internal P load in a recently restored lake. These findings are consistent with other studies reporting that Fe-P is the primary contributor to internal P loading and the most important algal-available P fraction in highly eutrophic shallow lakes (Petticrew and Arocena, 2001; Zhu et al., 2013). The importance of redox-related P release has been sometimes underestimated in shallow polymictic lakes, which is often associated with the limited sampling frequency (Tammeorg et al., 2020). Nevertheless, contemporary technologies enabling high-frequency measurements have been providing an increasing evidence for the importance of the mechanism. Recently, Søndergaard et al. (2023) has demonstrated frequent developments of stratification and anoxia followed by mixing events with considerable implications for lake water quality in small, shallow Lake Ormstrup (Denmark).

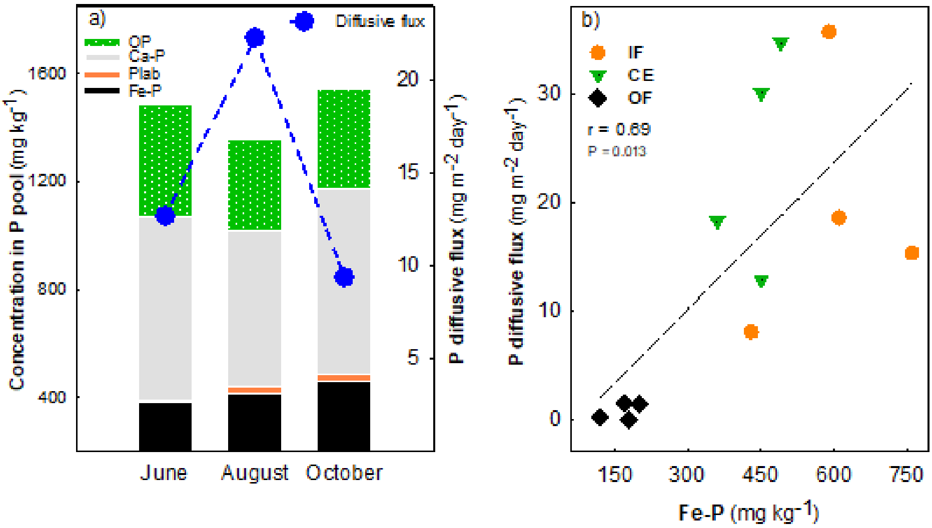

Figure 6. a) The concentrations of organic P (OP), Ca-bound P (Ca-P), labile P (Plab), and P bound to Al and Fe (hydr)oxides (Fe-P) in sediments of Lake Mustijärv and diffusive fluxes of P in 2018. The data consist of the averaged values for the IF, CE, and OF locations. b) The correlation between diffusive P flux and sediment Fe-P in August 2017, and June, August, and October 2018.

4.4.2 Role of sediment accumulation basins introduced during lake excavation

The average depth of the newly formed sediment was 0.19 ± 0.12 m, with highest sediment depths observed in the accumulation basins of the IF and CE locations (**III**). In lakes with a dynamic ratio lower than 0.8 (Lake Mustijärv <0.1), sediments are typically concentrated in central deeper areas (Håkanson, 1982; Bachmann et al., 2000). Sediments tend to move until they reach an undisturbed area where they can accumulate. For example, there was a decreasing trend of LOI from IF to CE and to OF, indicating that the allochthonous organic materials settled on the lake bottom mainly before reaching the OF location.

The total P concentration and all P fractions of sediment showed considerable spatial variation in Lake Mustijärv, with a decreasing trend from IF to CE and then to OF (comparing the mean values of 2017 and 2018 in August). This trend was reversed for the sediment Fe concentration and Fe:P ratio. The OF location in the expanded area of the lake after excavation differed significantly from the other two locations, with its nearly 50% lower P content (and potentially mobile P forms) in the sediment and two-fold higher Fe:P ratio. Such a distribution of P forms within the lake determined further patterns in internal P recycling.

The P diffusive flux in CE (34.6 mg P m^{-2} day^{-1}) was significantly greater than in IF (8.0 mg P m^{-2} day^{-1}), and IF had a significantly higher flux than OF location (0.25 mg P m^{-2} day^{-1}) in 2017. Nevertheless, the TP concentrations in the surface water layer showed a gradual decrease from IF to OF. Hence, these findings direct us to suggest that introducing accumulation basins may help to efficiently entrap point source nutrient inputs. More research is needed to explore to what extent such accumulation basins, if emptied regularly, can help to localize the high P sediment and facilitate improvements in water quality in the majority of the lake area.

5 Conclusions

My observations spanned a nine-month lysimeter experiment and a four-year field experiment along with two years of monitoring the shallow eutrophic Lake Mustijärv after restoration by sediment removal. These observations indicate that recycling excavated sediments back to agriculture can be a solution to closing the leaking agricultural P cycle and reducing its impacts on the environment. Furthermore, recycling sediment P to agriculture can be a co-benefit of lake restoration by sediment removal. The core conclusions were:

1. The high contents of easily soluble P and potentially bioavailable P fractions (e.g., P_{lab} and Fe-P) in the sediment-based growing media were mainly attributed to the low Fe:P mass ratio (<15) and low clay content (18%) in the excavated sediment.
2. In addition to P, the sediment had very high content of easily extractable essential plant nutrients, including S, Ca, Mg, B, Zn, and a fair supply of Cu. Also, the sediment provided a continuous supply of N to the plants over the four-year field experiment, which was likely due to the mineralization of the sediment's OM. Considering the high solubility of essential nutrients, high plant nutrient contents, and low contents of heavy metals and other harmful elements in the sediment, it can be considered safe and potent for use as a soil amendment material in agriculture.
3. The sediment as a growing medium provided similar or higher grass biomass yields compared with the agricultural soil. Additionally, the N and P contents were generally higher in plants grown in the sediment-based growing media than in *Soil*, even if increases in N and P uptake were not consistent.
4. The use of excavated sediment as a growing medium increased the risk of phosphate and nitrate leaching compared with soil. However, their average concentrations were below the maximum threshold values for causing risk in waterbodies. The increased N and P leaching from the sediment was attributed to their excessive levels in relation to plant needs. The sediment application rate should be adjusted to match the requirements of crops, similar to how fertilizers are applied. This can help minimize nutrient leaching back into the lake and further help mitigate eutrophication in the lake.
5. The organic-rich sediment with a C:N ratio below 15 had the highest CO_2 and N_2O emissions in the first year of application. This was due to the high rate of microbiological decomposition of OM. Also, the average over the four-year measurements showed that the sediment-based growing media emitted more CO_2 than *Soil* did. More research is needed to study whether GHG emissions from recycled sediments can be reduced by using the sediments in lower application rates similarly to organic fertilizers, instead

of using excavated lake sediments in large quantities as a growing medium.

6. The sediment material had a lower level of bacterial biodiversity compared with the soil. However, both sediment material and soil showed similar fungal biodiversity. The bacterial phyla associated with solubilizing P, including Proteobacteria and Chloroflexi, were more abundant in the sediment material than in *Soil.* This is while the relative abundances of Actinobacteria and Planctomycetes*,* with the potential to accumulate bioavailable P fractions, were higher in *Soil*. In addition, Calditrichaeota*,* with the capacity to degrade detrital proteins, Fusobacteria*,* commonly found in organic-rich sediments, and Atribacteria, fermenting a wide range of organic compounds, were the unique phyla in the sediment. These unique phyla may confirm various organic C mineralization pathways in sediment-based treatments.
7. Adding biochar to the sediment material did not affect plant biomass accumulation, plant nutrient content, or easily soluble nutrients (except for K) of the growing medium. Biochar did not increase the average N and P uptake by plants in the sediment, but the highest rates were observed in the biochar-treated sediment, which significantly exceeded those in *Soil*. As no negative agronomic or environmental effects were observed for biochar application, my results support the use of biochar-treated sediment for both soil C sequestration and potential nutrient benefits to crop production.
8. Theoretically, no markable internal P load should appear after complete sediment removal (ca. 6 Mg of P removed from the eutrophic, shallow Lake Mustijärv). However, the opposite was indicated by extensive anoxia and high release rates of P by diffusion (of a magnitude reported for hypertrophic lakes) in two years after sediment excavation. P release rates were correlated positively with the sediment Fe-P content, indicating a contribution of this fraction to P release. The pool of releasable P was associated with the external nutrient loading accelerated by the riverbed cleaning activities in the upstream of the lake.
9. An important role of sediment accumulation in entrapping external nutrient loading was revealed by the gradual decrease in sediment TP and P fractions, OM content, and TP concentrations in the surface water layer from IF towards OF. More research is needed to explore whether regularly emptying these accumulation basins could help localize the sediment with high P content and facilitate improvements in water quality across the lake. The frequent removal of sediments from accumulation basins will result in large sediment quantities, these sediments may have potential as fertilizers for agriculture.
10. Future research could focus on conducting cost-efficiency analyses and life-cycle assessments to evaluate the economical and environmental

impacts of sediment recycling from eutrophic lakes for agricultural use. This would provide valuable insights into the feasibility and sustainability of using sediment as a fertilizer source and help inform decision-making for future environmental management practices.

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