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Recycling eutrophic lake sediments into grass production: A four-year field experiment on agronomical and environmental implications

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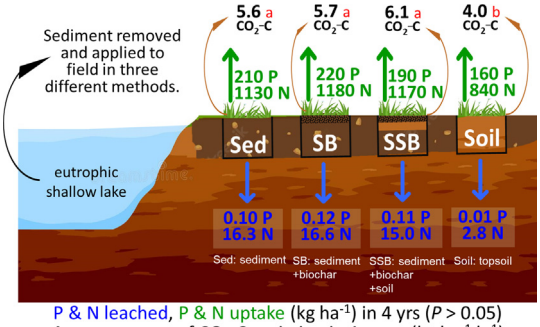
HIGHLIGHTS

- Plant growth in excavated lake sediment was comparable to that in agricultural soil.
- The Sed had higher CO_2 and N_2O emissions than the *Soil* in the first year.
- N and P uptakes were higher in biochar-treated sediment than in the *Soil*.
- More N and P leached from the lake sediment than from the *Soil*.
- Sediment and soil have contrasting bacterial and fungal community structures.

GRAPHICAL ABSTRACT

Cumulative P and N uptake, cumulative leached P and N, and the four-year average of CO2-C emission from the growing medium treatments in Viljandi field experiment in 2017 – 2020.



P & N leached, P & N uptake (kg ha^{-1}) in 4 yrs ($P > 0.05$)
4-year average of CO_2-C emission in August (kg ha^{-1} h^{-1})

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ABSTRACT

Inefficient use of phosphorus (P) fertilizers leads to the transfer of P into water bodies, causing their eutrophication. Sediment removal is a promising lake restoration strategy that removes nutrients including P accumulated in lake sediments, and opens the opportunity to use removed nutrients in agriculture. In the present study, we investigated the effects of using a thick layer of sediment from the eutrophic Lake Mustijärv on plant growth, and estimated the environmental impacts of different sediment application methods by analyzing greenhouse gas emissions, N and P leaching, aggregate stability, and soil biota. The field experiment (2017–2020) was established on the lake shore with the following treatments: the agricultural control soil (*Soil*) surrounding the lake, pure sediment (*Sed*), biochar-treated sediment (*SB*), and biochar and soil mixed with sediment (*SSB*). The sediment-based treatments resulted in a similar grass growth performance to the *Soil*. The availability of most macro- and micronutrients including P (75 vs. 21 g m^{-3}) were far greater in the *Sed* compared to the *Soil*. The sediment-based growing media emitted more CO_2 than the *Soil* (579 vs. 400 mg CO_2 − C m^{-2} h^{-1}) presumably due to the high rate of organic matter decomposition. The bacterial and fungal community structures of the *Sed* were strongly differentiated from those of *Soil*. Also, *Sed* had lower bacterial diversity and a higher abundance of the bacterial phyla associated with solubilizing P including Proteobacteria and Chloroflexi. Sediment-based growing media increased more than seven times the risk of mineral N

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and P leaching, and the biochar treatment only had a short-lived beneficial effect on reduction of the sediment's leached P concentration. The sediment application rate should be adjusted to match the crop requirements to minimize greenhouse gas emissions and nutrient leaching when upscaling the case study to larger lakes with similar sediment properties.

1. Introduction

Phosphorus (P) is an essential plant macronutrient originating mainly from non-renewable phosphate rocks. Considering the finiteness of rock phosphates and the high P fertilizer consumption in the agricultural sector (43 M tonnes in 2019, FAO, 2019), it is becoming increasingly important to close the agricultural P cycle. Lake sediments are usually rich in organic carbon and nutrients, including phosphorus (P), which typically originates from eroded agricultural soils. Unsustainable agricultural P use leads to the P loss which causes serious environmental problems, such as the eutrophication of lakes. This further emphasizes the importance of closing the P cycle.

Sediment removal by dredging or excavation is widely applied in restoration of eutrophic lakes (Cooke et al., 2016; Lürling et al., 2020). It removes P accumulated in lake sediments and opens the opportunity to close the P cycle by using the removed P in crop production. Based on their properties, sediments excavated from waterbodies can act as sources or sinks of P (Harrington and McInnes, 2009), thus the feasibility of recycling nutrients from waterbodies to agriculture depends on the nature of the sediments and soils in question (Braga et al., 2019). Lake sediments have been shown to be efficient as soil amendments: by increasing soil nutrient availability such as P, calcium (Ca), and copper (Cu, Canet et al., 2003; Edesi et al., 2020; Brigham et al., 2021; Kiani et al., 2021), organic matter content (Edesi et al., 2020; Brigham et al., 2021), water holding capacity (Darmody and Diaz, 2017) and cation exchange capacity (Canet et al., 2003; Darmody and Diaz, 2017; Brigham et al., 2021), elevating soil pH (Ebbs et al., 2006; Mattei et al., 2018), and decreasing soil bulk density (Woodard, 1999; Brigham et al., 2021). This has resulted in crop yield benefits, especially in nutrient-deficient soils, in several studies (Canet et al., 2003; Darmody and Diaz, 2017; Edesi et al., 2020; Kiani et al., 2021). However, no adequately replicated multi-year field studies focusing on lake sediment recycling are available (that would have included also characterization of the sediment).

Considering recycling of sediment to agricultural soil, all soil functions including biodiversity and microbial activity, must be maintained. In addition, cultivation of organic-rich sediments increases the aeration of the sediment materials, which may result in higher fluxes of greenhouse gas (GHG) emissions. However, to our knowledge, these aspects have not been studied earlier. Also, applying nutrient-rich sediment to soils may increase the risk of nutrient loss by surface runoff or leaching into groundwater. As biochar amendment can reduce soil N and P leaching by controlling surface runoff through improving aggregate stability (Soinne et al., 2014; Schmidt et al., 2021), combining sediment materials with biochars, porous carbonaceous materials, needs to be tested. Moreover, according to several meta-analyses, biochars can reduce soil N_2O emissions (Cayuela et al., 2014; Borchard et al., 2019; Schmidt et al., 2021). However, it is also currently unknown whether addition of biochars can reduce the GHG emissions from carbon-rich sediments.

Recycling sediments 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; Renella, 2021). With the aim of closing the agricultural P cycle, we investigated the agronomical and environmental implications of different sediment application methods that were hypothesized to reduce loss of nutrients from recycled sediment via leaching, erosion, or gaseous losses over a four-year field experiment. Our specific objectives were to determine the effects of different sediment application methods on i) growth, yield, and nutrient uptake of mixed grasses, and ii) nutrient availability in soil, and iii) GHG emissions, leaching of N and P, aggregate stability, and soil biota. The results are expected to provide guidelines on the efficient and environmentally safe use of excavated sediments, which may lead to more sustainable agroecological systems.

2. Materials and methods

2.1. Study site and weather conditions

The study was conducted during 2017–2020 on the shore of a 1- ha eutrophicated Lake Mustijärv located 1 km west of Viljandi, Estonia (58°21′55.8"N 25°32′32.6"E, 65 m above sea level; Fig. 1). The lake was established by the expansion of the Kurika stream in 1984–1986. The lakeshores are covered by sandy loam soil from the north and by deep peat from the south. The upstream watershed area is mostly used as cattle pastures and annual crop production with conventional tillage. The agriculture for the first 1.5 km upstream is mostly certified organic type. Additionally, the lake has received effluents from local dairy and urban stormwater. By 2015, the lake was heavily eutrophicated and the majority of nutrients were carried into the lake by the upstream. In 2016–2017, the lake was completely desilted to the lakebed by excavating the whole sediment. Details about the sediment excavation process from the lake are described in Kiani et al. (2020).

The daily precipitation, along with the minimum and maximum temperature of the site were recorded by the Viljandi Meteorological Station located 3.6 km northeast of the field experiment (Fig. 1d). The mean air temperature of the growing period (May to October) was 12.2, 15.0, 13.3, and 13.8 °C in 2017, 2018, 2019, and 2020, respectively. The mean precipitation was 80.4, 67.6, 75.1, and 67.4 mm in 2017, 2018, 2019, and 2020, respectively. The growing period was warmer and drier in 2018 to 2020 compared to the long-term means (1981–2010; 12.8 °C and 76.2 mm) in Viljandi (EWS, 2021).

2.2. Experimental design and field establishment

The experiment was set up with four treatments as a randomised complete block design with four replicates of 2 m × 2 m plots along the shore of the lake (Fig. 1c). The treatments included three different sediment application methods that were hypothesized to reduce loss of nutrients from recycled sediment via leaching, erosion or gaseous losses, inspired by the earlier pot experiment (Kiani et al., 2021), and a control soil from the shore of the lake. All the four replicates in the experiment were on different level of the height gradient, but so that within a replicate, all experimental treatments were on the same level of the gradient. For instance, for replicate 1, this height was 75 cm (Fig. 1c). Treatments were tested in the field included *Soil*: column of pure topsoil; and sediment-based treatments (*Sed*, *SB*, and *SSB*) with a 75–100 cm column (1650–2970 t ha^{-1}) of sediment (*Sed*), was top-dressed with a 2.5 cm layer of biochar (*SB*) or a mixture of 2.5 cm of soil and 2.5 cm of biochar layers (*SSB*) mixed with the sediment layer.

Before excavating the sediment, a storage site was prepared close to the lake shore (Fig. 1a). Excavators were used to peel and pile the top 30 cm of soil from the sediment storage site. The soil type was classified as an Endogleyic Lamellic Luvisol (IUSS, 2015) with a sandy loam texture (60 % sand, 27 % silt, 13 % clay), according to the pipette method (Elonen, 1971). Sediment used in the field experiment were removed from open water areas of the eutrophic lake in summer 2016. All sediment (7500 m^3 sediment including six Mg of total P) with a loamy texture (40 %

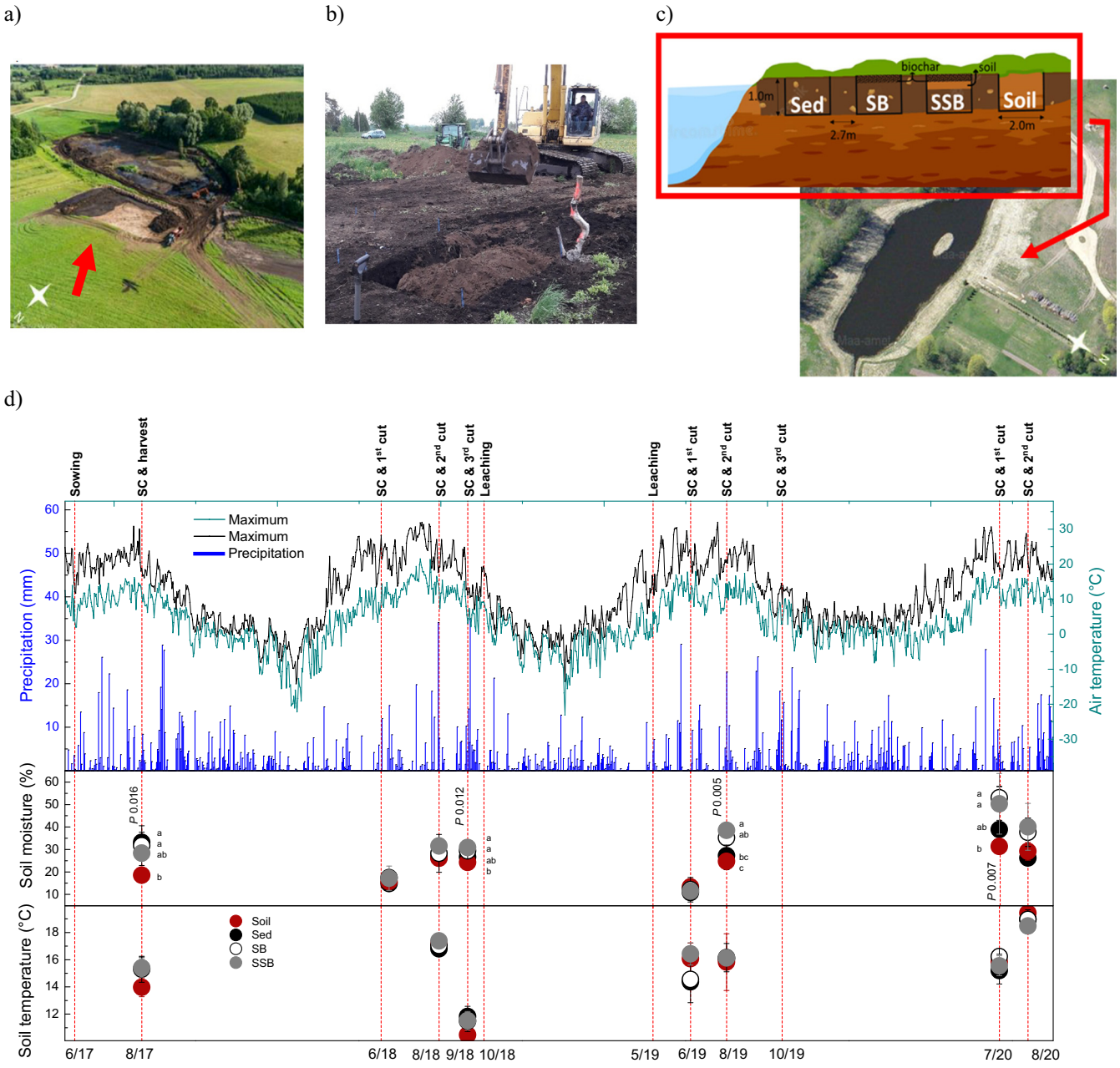

Fig. 1. Aerial photo of the excavation process of the eutrophic Lake Mustijärv in August 2016 by Kristjan Lust. The first piles of sediment stored on the Eastern side of the peeled storage area which were used in the field experiment after a nine-month storage period (a); Sediment removal by excavator and soil replacement in soil plots on May 31, 2017 (b); Aerial photos of the restored lake on 29 April 2019 by Estonian Land Board and a cross-sectional view of the treatments in the field experiment in 2017–2020. The location of the field experiment is shown by the red arrows (c); Variation of daily precipitation, maximum and minimum air temperature, and the average of soil moisture content and soil temperature in different treatments of the field experiment in 2017–2020 (d); SC: sampling campaign which may include cutting the above-ground biomass, collecting soil samples, measuring greenhouse gas emissions, soil moisture, soil temperature, soil penetration resistance, and collecting the leaching samples. Sed: 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; Soil: Pure topsoil. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

sand, 42 % silt, 18 % clay) was removed from the lake and spread onto the storage site. Sediment was left to dry out until the time of field establishment.

For the *Soil* treatment, the whole sediment layer from the 4 m^2 plot area was removed with an excavator scoop to the depth of the original subsoil layer (Fig. 1b); the holes were then filled with the mineral topsoil peeled off from the 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 the depth of 20 cm.

To obtain the activated (soaked in tap water:cattle slurry (7:3) mixture) wood-based biochar, hardwood branches and split logs were collected from the lake shore and then pyrolysed at approximately 680 to 750 °C (Schmidt et al., 2014) in a 0.3 m^3 Kon-Tiki garden kiln in January 2017. The produced biochar had a high degree of carbonization ($H:C_{org} < 0.7$) 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}; EBC, 2022). The

biochar had a specific surface area of 204 $m^2\ g^{-1}$, pH 9.86, and the contents of total N, P and K were 2.5, 1.3 and 7.2 g kg^{-1}, respectively. A 2.5 cm layer of biochar (37.5 t ha^{-1}) delivered a total of 32.2 t C ha^{-1}. More detailed information about the properties of biochar is described in SM Table A and in Kiani et al. (2021).

All treatments were fertilized with a meat bone meal (MBM) organic fertilizer (75 % organic matter) placed at a depth of 3–4 cm below the surface (Erikois-Viljo 8-4-8: Honkajoki Oy, Honkajoki, Finland). The application rate was calculated to deliver 100 kg of N, 50 kg of P and 100 kg of K per hectare. About 500 g MBM per plot was applied by hand, and the top 5 cm of the surface was mixed with a rake in order to distribute the MBM fertilizer into the plant root zone. The content of secondary nutrients in the MBM is presented in Kiani et al. (2021). Finally, a mixture of 45 % red fescue (*Festuca rubra* L.), 35 % Kentucky bluegrass (*Poa pratensis* L.), 15 % ryegrass (*Lolium perenne* L.), and 5 % white clover (*Trifolium repens* L.) with a rate of 80 g seeds per plot was planted by hand-sowing on 1 June 2017. The used plant mixture is a classic combination of different grasses used for landscaping and agricultural purposes. The fast-growing perennial ryegrass establishes quick ground cover and is complemented with slower-growing but more stress-tolerant and longer-living red fescue and meadow grass.

2.3. Soil analyses

Soil samples were collected every August from 2017 to 2020. To determine the soil chemical properties and microbial community characterization, nine sub-samples were taken from a depth of 0–20 cm with a push probe (2.5 cm inner diameter). Sub-samples were then, mixed to form a composite sample (about 800 cm^3 per plot) and transferred into plastic bags. Immediately after sample collection, sub-samples for microbial community measurement were taken from the plastic bags using a sterilised spoon. Between each sampling plot, all sampling tools were sterilised with ethanol (70 %). The microbial samples were transported in an icebox to the laboratory and stored at −80 °C until analysis in October 2018. For aggregate stability tests, four soil surface samples were taken from a depth of 0–5 cm with the same push probe. The initial samples were also collected from each plot in June 2017 with the same procedures.

The content of easily soluble macro- and micronutrients was analysed according to the standard Finnish soil testing methods based on Vuorinen and Mäkitie (1955) for P, K, S, Ca, Mg, and Na, Berger and Truog (1939) for B, and Lakanen and Erviö (1971) for Cu, Mn, and Zn. The electrical conductivity and pH of the samples were measured in a 1:2.5 (w:w) soil-to-water mixture (Vuorinen and Mäkitie, 1955). For initial samples and samples from 2018, the P fractionation was conducted by following the method described in Ruban et al. (2001) based on Williams extraction procedure. 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 (Plab) was extracted with 1 mol L^{-1} NH_4Cl as a part of the Hieltjes and Lijklema (1980) protocol. Also, the total Fe concentration of the samples was determined by inductively coupled plasma-optical emission spectroscopy (ICP-OES; Thermo-Fisher iCAP3600 MFCDuo, Thermo Fisher Scientific, Cambridge, UK) after being digested with nitric acid. The total C and N content of ground (< 2 mm) soil samples were determined using a Leco CN analyser (CN828, model 622–200-100, Leco Corporation, St. Joseph, MI, USA). The carbon stock of each treatment was calculated as follows (Kluge et al., 2008):

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

where C_{stock} is the soil carbon stock (g m^{-2}), C is the carbon content (g kg^{-1}), BD is the bulk density (kg m^{-3}), and Z is the thickness of the sampled layer (0.2 m). The loss of carbon was estimated from the changes in the C stock between 2017 and 2020 considering constant BD values across the years.

Soil penetration resistance was measured in August 2017, June 2017, and August 2018. A hand-pushing Eijkelkamp Penetrologger was used for the measurements, with a 60-degree cone base of 2 cm^2. The penetration resistance measured four times per plot by pushing the penetrometer vertically into the soil at an approximated speed of 3 cm s^{-1}. No penetration resistance was measured in 2019–2020 to reduce the numbers of measurement holes in our small-sized plots and to avoid potential bias to the leaching data due to possible preferential water flow through the holes.

The aggregate stability against slaking in water was tested with the wet-sieving method for two analytical replicates per sample as follows: 20 g of air-dried samples were sieved with a sieving machine through 0.63- and 2- mm sieves for three minutes. From the class of 0.63–2 mm, 4-g of aggregates were placed on a 0.25-mm sieve of wet sieving apparatus (Eijkelkamp Agrisearch Equipment, Giesbeek, Netherlands). Due to hydrophobicity in the aggregates, water was sprayed on the air-dried aggregates and left for 15 min. Then, 100 mL of deionized water was added to the tins of the wet sieving apparatus and the samples were left to stand in water for another 15 min. After the 15-min saturation time, the device was set in motion and the aggregates were dipped into the water about 95 times for three minutes. The water-stable material remaining on the sieve was collected into a glass vessel with the help of a spray bottle, and oven-dried at 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 was originally taken for the wet-sieving (Soinne et al., 2014). The water and detached material in the tins were transferred to centrifuge tubes, and the suspension was left to settle for 21 h. A 25 mL sample was pipetted from the surface of the settled suspension into a turbidity meter cuvette, and the turbidity was measured by a HACH 2100 N turbidity meter (Hach Co. Loveland, CO, USA).

2.4. Plant analyses

Aboveground biomass (AGB) including mixed grasses and weeds was sampled nine times (Fig. 1) 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 was sampled from a 30 cm × 30 cm area at the centre of each plot by cutting the vegetation from the 2 cm height above the soil surface. After drying at 60 °C for 72 h, the dry weight of the plant samples was recorded. The species composition of the collected AGB was also determined and classified for grasses (red fescue, Kentucky bluegrass, ryegrass, and white clover) and weeds (*Bunias orientalis*, *Chenopodium album*, *Polygonum persicaria*, *Taraxacum* sp., *Stellaria* sp., *Silene* sp., and *Urtica* sp.), then the sample was combined and ground through a 1.0 mm sieve. The dry material was dry-ashed in a muffle oven, and the elemental composition (P, K, S, Ca, Mg, Na, Al, B, Ba, Cd, Co, Cr, Cu, Fe, Mn, Ni, Sr, and Zn) in AGB was determined by ICP-OES using a multi-element standard solution Merck IV. The total C and N contents of plant samples were determined with a Leco CN analyser. The plant uptake of N and P was calculated by multiplying the given plant nutrient content by AGB.

2.5. Leaching and GHG emission analyses

On 21 June 2018, bespoke zero-tension lysimeters (Voll and Roots, 1999) with a 30 cm × 30 cm metal sheet and a 2.5 L plastic container were installed at a depth of 30 cm in each plot to collect the leachate (Fig. 5). The leachate was sampled nine times (Fig. 1) over the years in 2018 (21.08., 25.09, 15.10), 2019 (10.05., 25.06., 8.08., 15.10.) and 2020 (7.07., 11.08.). At each sampling time, the full amount of leachate was collected from the containers, the amounts were recorded, and a subsample of leachate was taken (Fig. 1d). The sub-samples were stored at −20 °C until analysis. The leachate sub-samples were passed through Whatman blue ribbon filters which were rinsed three times with 2 M potassium chloride (KCl) and twice with MQ water before filtering. The samples were analysed for PO_4^{3-} − P, NO_3^- − N, and NH_4^+ − N concentrations by spectrophotometry with an automated discrete analyser (Gallery Plus ECM, Thermo Fisher Scientific, CA, USA). The daily leached amounts of

nutrients in the drained water were calculated by multiplying the concentration of each nutrient (mg L^{-1}) by the amount of leachate from the lysimeter (L m^{-2}) divided by the number of days since the last leaching collection (mg m^{-2} d^{-1}).

The fluxes of GHGs (CO_2, N_2O, CH_4, and NH_3) were measured in situ using an automated Fourier Transform Infrared Trace Gas Analyser (FTIR-TGA) (Gasmet DX4015, Gasmet, Helsinki, Finland) at several sampling campaigns through 2017–2020 from the sites where the AGB was collected in each plot. The concentrations of gases were measured for 10 min after deploying the opaque chamber in the field. The aluminium chamber used was cylindrical (27 cm height, 31.5 cm 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 analyser and then returned to the chamber via the outlet after analysis. The first two minutes of measurements were discarded to avoid the probable effects of immediate 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 with the time of measurement. Measurements with r values of regression lower than 0.4 were considered to indicate no detectable flux of gas (Kalu et al., 2021). The CO_2 and N_2O fluxes were presented as CO_2 − C and N_2O − N fluxes considering 3.67 kg CO_2 = 1 kg C and 1.57 kg N_2O = 1 kg N.

2.6. Soil DNA analyses

The microbial community structure of different growing media was determined in June 2017, immediately after applying treatments, and in August 2017 and 2018, after first and second growing season. Soil DNA was extracted from 0.25 g soil samples using a PowerSoil DNA isolation kit (MoBio, Carlsbad, CA, USA), according to the manufacturer's instructions. The quality of extracted DNA was assessed with electrophoresis, and the concentration and purity of DNA were determined using a 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 × 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 from 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 (SOP) (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, Yilmaz et al., 2014) and the ITS2 sequences against the UNITE database (UNITE + INSD version 8.3, Abarenkov et al., 2020). Chimeras were removed using "chimera.vsearch" command (Rognes et al., 2016). The sequences were assigned to OTUs at a 97 % similarity threshold; taxonomy was assigned using the SILVA database for bacterial sequences and the UNITE databases for fungal sequences. The 789,269 16S rRNA gene and 1,950,635 ITS2 sequences were assigned into 430,216 bacterial and archaeal OTUs and 19,589 fungal OTUs, respectively. Singletons were removed using the filter_taxa function, and the alpha diversity indices Chao1, Shannon, Simpson and Fisher were calculated with the estimate_richness function in the package phyloseq (McMurdie and Holmes, 2013) in R (Version 4.1.2, R Core Team, 2021). The bacterial and archaeal OTUs were assigned to 52 phyla and ~ 315 genera, and the fungal OTUs to 18 phyla and ~ 331 species of all taxa found within a given category in representative soil and sediment material.

2.7. Statistical analyses

Statistical analyses were carried out with a linear mixed effects model using the "lme" function under the "nlme" library in the RStudio Server 2022.02.2 Build 485, where "growing medium" was the fixed factor and "block" was the random factor for each year. The variables that did not meet the assumptions of normal distribution and homogeneity of variances were tested by introducing different variance structures in the nlme package, including stratum variance structure (Zuur et al., 2009) and the Box-Cox transformation (Box and Cox, 1964). The four-year averaged data were analysed as a repeated measures ANOVA with a linear mixed effects model using the "lme" function, where "growing medium" was a fixed factor, and "block" and "year" were the random factors. Post hoc tests were computed using the "emmeans" function with the Tukey method and a significance level of $P < 0.05$ was specified in the "cld" package. The correlations between the agronomical and environmental parameters and growing medium properties were tested with Pearson's analysis.

Alpha diversities were tested for significance using ANOVA. The relationships between environmental variables and Hellinger-transformed bacterial

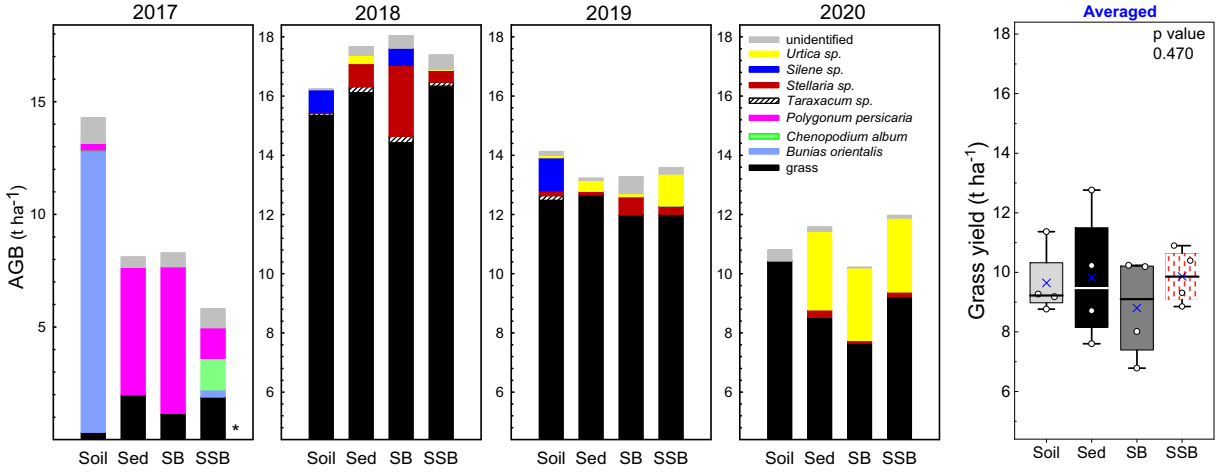

Fig. 2. Effect of growing medium on yearly cumulative aboveground dry biomass (AGB) of grasses and most relevant weed species and the average of grass yield in four years of the field experiment in 2017–2020. The asterisk represents that dry grass yield was significantly lower in *Soil* than in the rest of the treatments ($P < 0.05$) in 2017. The line inside the box represents the median, the blue cross shows the mean, and the top and bottom of the box represent the third (Q3) and first (Q1) quartiles respectively, the top whisker is Q3 + 1.5 IQR and the bottom whisker is Q1–1.5IQR. Sed: 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; Soil: Pure topsoil. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Table 1a

Chemical properties of the growing medium treatments in the 0–20 cm layer, and proportion of water-stable aggregates (WSA) and mean turbidity in air-dried samples of the growing medium treatments in the 0–5 cm layer in the field experiment during the 2017–2020. Samples were collected at the end of the growing seasons in August. Data show means of four replicates across four growing medium treatments. Mean values within the growing medium treatments followed by a different letter are significantly different at $P < 0.05$.

Treatment	EC	pH				Acid ammonium acetate extractable (g m^{-3} soil)										WSA	Turbidity
	mS cm^{-1}		TC%	TN%	C:N	P	K	S	Ca	Mg	Na	B	Cu	Mn	Zn	%	NTU
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	94.5 b	1.85 a
Sed	1.47 a	7.23 ab	12.9 a	0.760 a	13.5 b	73.3 a	52.5 b	733 a	14,125 a	498 a	46.5 b	2.60 a	7.60 a	24.0 b	125 a	97.2 a	1.15 b
SB	1.40 a	7.25 a	16.1 a	0.815 a	16.8 a	86.0 a	268 a	610 a	12,025 a	498 a	57.0 a	2.35 a	7.38 a	19.3 b	125 a	98.2 a	1.29 b
SSB	1.26 a	7.30 a	12.2 a	0.760 a	16.3 a	77.0 a	248 a	495 a	10,850 a	475 a	51.0 ab	2.15 a	6.40 a	21.3 b	89.3a	96.4 ab	1.57 ab
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	89.7 b	0.699
Sed	0.830 a	7.20	12.3 ab	0.603 a	14.5	73.8 a	84.8	250 a	13,875 a	450 a	24.0	2.60 a	8.08 a	22.5 b	121 a	97.1 a	0.854
SB	0.715 a	7.38	15.6 a	0.737 a	16.5	78.5 a	151	190 a	12,150 a	440 a	21.5	2.67 a	7.65 a	18.5 b	123 a	95.1 a	0.726
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	93.3 ab	0.702
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	85.2 b	1.84
Sed	0.479 a	7.41 a	13.7 a	0.730 a	14.2	77.6 a	94.4 b	83.9 a	15,369 a	427 a	16.4 a	2.61 a	8.18 a	19.0 b	117 a	95.4 a	0.982
SB	0.474 a	7.45 a	13.7 a	0.826 a	16.5	81.7 a	177 a	45.6 a	13,055 a	427 a	13.2 ab	2.59 a	7.55 ab	16.7 b	115 a	94.1 a	1.42
SSB	0.392 a	7.45 a	12.6 a	0.813 a	15.7	68.3 a	176 a	56.8 a	11,588 a	461 a	15.1 a	2.40 a	6.70 b	19.2 b	91.0 a	90.3 ab	1.79
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	97.6	0.710
Sed	0.341 a	7.43 a	11.0 ab	0.582 ab	14.6 b	74.2 a	122	35.0 a	17,067 a	410 a	14.0 a	2.63 a	8.24 a	19.3 b	131 a	97.8	0.869
SB	0.404 a	7.36 a	14.9 a	0.611 ab	18.2 a	79.5 a	160	33.0 a	14,048 a	422 a	12.8 a	2.61 a	7.77 a	18.3 b	124 a	97.1	1.06
SSB	0.367 a	7.43 a	12.2 ab	0.797 a	15.4 ab	76.2 a	149	34.8 a	13,354 a	419 a	11.6 a	2.62 a	7.24 a	19.8 b	105 a	97.1	0.790
Fertility class[1]																	
questionably 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		
low		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		
extremely low		< 5.0				< 3	< 40	< 3	< 400	< 50	< 15	< 0.2	< 1.0	< 6	< 1.0		

Sed: 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; Soil: Pure topsoil. The standard deviation values are presented in the SM Table B.

[1] The classification of arable soil (Viljavuuspalvelu Oy, 2008).

Table 1b
Average plant nutrient concentrations in the field experiment during 2017–2020. Data show means of four replicates across four growing medium treatments. Mean values within the growing medium treatments followed by a different letter are significantly different at $P < 0.05$.

Treatment	C	N	P	K	S	Ca	Mg	Na	Al	B	Ba	Cd	Co	Cr	Cu	Fe	Mn	Ni	Sr	Zn
	g kg^{-1}									mg kg^{-1}										
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	18.0 b	0.230 a	0.171	1.52	4.95 b	415	37.6	0.650	57.7 a	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	42.3 a	0.104 b	0.243	2.56	12.30 a	515	48.4	0.860	32.5 b	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	42.5 a	0.096 b	0.227	2.84	10.28 a	602	46.6	0.928	28.1 b	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	39.9 a	0.097 b	0.274	2.96	9.78 a	724	45.3	1.090	28.4 b	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
Ref. values																				
[1]Suttle (2010)		17–32*	1.1–4.1	3	0.9–2.1	1.4–7	0.4–1.4	0.6–1.25	–	–	–	–	–	–	4.3–28.4	30–50	8–20	–	–	8.8–27
[2]Max authorized value in Europe			–	30–40	–	–	–	45	500–8000†	800**	–	1–10*	–	1000‡	25	750	150	100*	2000‡	150

Sed: 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; Soil: Pure topsoil. The standard deviation values are presented in the SM Table C.

[1] Adapted from Suttle (2010). The range represents the concentration in the forage to meet the requirements for sheep of different categories and weights, from growing lambs of 20 kg LW to pregnant ewes carrying twins of 75 kg LW, assuming that the quantity of forage available is not limiting.

[2] The values are the maximum amounts of trace elements authorized in feed (or its correctors) for various domestic species, according to the European Union.

* Adapted from Freer (2007).

** Adapted from Suttle (2010).

† According to Jones (2005).

‡ According to NRC (2000). Reference values specific for cattle tolerance.

and fungal community data were tested using distance-based redundancy analysis (db-RDA) in the R package "vegan". The variables were chosen by forward selection with the ordistep function. Differential abundance analysis was done in the R package "DESeq2" (Love et al., 2014), and sparsely represented OTUs across samples were removed when the DESeq2 normalised count ("baseMean") was <0.6. OTUs with a Benjamini–Hochberg corrected $P < 0.05$ were considered differentially abundant.

3. Results

3.1. Plant growth and nutrient uptake in different growing media

The biomass accumulation (grasses and weeds) did not show any major differences among treatments in any of the years except for the grass yield in 2017 (Fig. 2). In 2017, the grass yield of the *Soil* treatment was 83 % lower than that of the *Sed* treatment ($P = 0.024$). The low grass yield of *Soil* in 2017 was due to a significantly higher share of weeds (98 %) in comparison to all sediment-based treatments including *Sed*, *SB*, and *SSB* (81 %, Fig. 2). The weed-to-grass ratio was similar in all treatments, with the weed percent within 18 to 29 % across all years (Fig. 2). In 2017, the main weed species were perennial *Bunias orientalis* in *Soil*, and the annual *Polygonum persicaria* (redshank) in the sediment-based treatments ($P < 0.05$). In 2018, the *Stellaria* species were abundant in the treatments containing sediment, while they were nearly absent in the *Soil* treatment ($P < 0.05$). From 2019 onwards, the plots containing sediment indicated an increasing trend of *Urtica* ($P = 0.15$) in 2020.

The C content of the growing medium was the highest in the *SB* treatment (16 %, $P < 0.05$) while it was always the lowest in *Soil* (2 %). The average sediment C stock of 8.3 kg m^{-2} was 64 % higher than that in *Soil* in the top 20 cm from the surface. The *Sed* treatment lost 1.2 kg m^{-2} carbon over the four years, while there were no considerable changes in the C stock of the *Soil*. The average N content in all sediment-based growing media was 0.89 %, which was six times higher than that in *Soil* (Table 1a). Correspondingly, the N content in plant biomass was significantly (60 %) higher in sediment-based treatments than in *Soil* (15.6 g kg^{-1}) in all years except 2017 (Table 1b). The total N content of growing medium in *Soil* and its associated N content in plant biomass declined by 44 % and 22 % ($P < 0.05$), respectively, from the first to second year, while they did not considerably change in the sediment-based treatments during the four years of the experiment (Tables 1a and 1b). As a result, the plant N uptake in the sediment-based treatments was higher than that in the *Soil* in 2018 (442 vs. 242 kg N ha^{-1}, Fig. 3). In general, plants were able to take up 40 % more N from *SB* and *SSB* than from the *Soil* in the four years ($P = 0.021$, Fig. 3). The C:N ratio of the growing medium was significantly narrower in the *Soil* (9.8) than in the sediment-based treatments (15) in 2017. Toward the fourth year of the experiment, the *Soil* C:N increased to 18 which differed statistically from that of *Sed* (15).

The contents of total P, Ca − P, Fe − P, and labile P in the control *Soil* treatment were only 15 to 44 % of those in the sediment-based treatments ($P < 0.05$, SM Fig. B). Of all the P fractions, only the Fe—P had significant changes in the P pool. Two years after treatment application, the Fe − P contents in the sediment significantly reduced from 830 to 551 mg kg^{-1} and the *Soil* lost 5 % of the share of Fe − P in the P pool ($P < 0.05$). The amount of easily soluble P was consistently higher in the treatments containing sediment than in the Soil (77 vs. 21 g m^{-3}) during the experiment. A similar observation was found regarding the P content in plant biomass, with up to 47 % more P in the sediment-based treatments than in the *Soil* (5 vs. 3 g kg^{-1}, $P < 0.05$) in 2017 and 2018. As a result, P uptake in the sediment-based treatments (79 kg P ha^{-1} on average) was significantly higher than in the *Soil* in 2018 (46 kg P ha^{-1}, Fig. 3). In addition, on average, plants were able to take up 37 % more P from biochar-treated sediment (*SB)* than *Soil* during the four years ($P < 0.02$, Fig. 3).

The soil test results revealed that the contents of many of the easily soluble macro- and micronutrients were significantly higher in *Sed* than in *Soil*, with the exception of K and Mn, which were more abundant in *Soil* (Table 1a). In agreement with this, the plants grown in the *Sed* treatment had higher contents of S, Mg, Ba, Cu, and Zn than those grown in *Soil*, particularly in the first years of the experiment ($P < 0.05$; Table 1b). The treatments containing biochar had significantly the highest level of easily soluble K, which also resulted in the highest contents of K in plant biomass (Tables 1a and 1b). Comparing *Sed* and *SB* indicated that biochar did not significantly increase the content of other nutrients in the growing medium ($P \geq 0.11$), except for K and Na in 2017. The easily soluble S content sharply decreased from 733 g m^{-3} in 2017 to 35 g m^{-3} in 2020 in Sed growing medium. Similar result was observed in other treatments as well (Table 1a). The sediment-based growing media had higher electrical conductivity (EC) than *Soil* treatment throughout the experiment in 2017–2020 (Table 1a). The effect persisted even though the EC of the sediment-based growing media decreased from 1.38 in 2017 to 0.370 mS cm^{-1} in 2020 (Table 1a). The EC value had the highest positive correlation with the content of easily soluble S ($r = 0.96$, $P < 0.01$, SM Fig. A), which gradually decreased over time. Also, the pH was 0.2–0.4 units higher in treatments containing sediment than in the *Soil* in all experimental years ($P < 0.05$, Table 1a) except 2018.

The nutritional value of plants in all treatments was in the range of the requirements for fodders in during the experiment, except for Na. However, the plant contents of N, S, and Cu in the *Soil* were lower than the sufficient level in 2020 (Table 1b). The contents of most elements in plant tissue declined rapidly after the first year (three- to nine-fold reduction for Ca, Al, B, Cu, and Fe), and the declining trend continued through to 2020. However, K content remained constant in all treatments across the four years of the experiment and exceeded animal requirements by about six-fold.

3.2. Greenhouse gas emissions

The emission of CO_2 from *Sed* did not differ statistically from the emissions from the *Soil* at any measurement time, except August 2017 when the *Sed* treatment emitted 88 % more CO_2 than *Soil* ($P = 0.046$; Fig. 4). Later in the experiment, the *Soil* still had the lowest flux of CO_2: it was significantly lower than *SB* and *SSB* in August 2019, and lower than *SB* in July 2020. As for the emissions of N_2O, in 2017, the *Soil* was a sink of N_2O with an average flux of −420 μg N_2O − N m^{-2} h^{-1} while *Sed* was a source of N_2O emission with an average flux of 195 μg N_2O − N m^{-2} h^{-1} (Fig. 4). Later, however, N_2O fluxes in all treatments were broadly similar except for September 2018, when *Sed* had a small and non-significant negative flux of N_2O, while the *Soil* had a positive flux of N_2O ($P < 0.05$).

In 2017, the biochar-treated sediment (*SB*) had numerically (20 %) lower CO_2 emission compared with the *Sed*, and *Soil* and *SB* did not differ statistically. Similarly, the N_2O emission from the *SB* treatment were 20 % lower compared to the *Sed* in 2017 ($P > 0.05$; Fig. 4). However, in the later years, such trends toward the slightly beneficial effects of biochar in the reduction of gas emissions were not confirmed. The average value of gas fluxes during the four years of measurement showed that sediment-based treatments had significantly higher CO_2 flux compared to the *Soil* (579 vs. 400 mg CO_2 − C m^{-2} h^{-1}).

There were no detectable fluxes of CH_4 and NH_3 at any measurement (r value of linear regressions <0.4). Also, the average concentration of these gases during the 10 min of measurements was not different between *Sed* and *Soil* (SM Fig. C).

3.3. Nutrient leaching and stability of aggregates

The average value of EC in the leachate ranged from 0.52 mS cm^{-1} in the *Soil* to 1.50 mS cm^{-1} in the *SB* treatment. During the experiment, the EC of leachate in the sediment-based treatments decreased from 2.23 mS cm^{-1} in 2018 to 1.09 mS cm^{-1} in 2020 (data not shown). Most leached mineral N was in nitrate form (> 99 %). In general, the four-year average

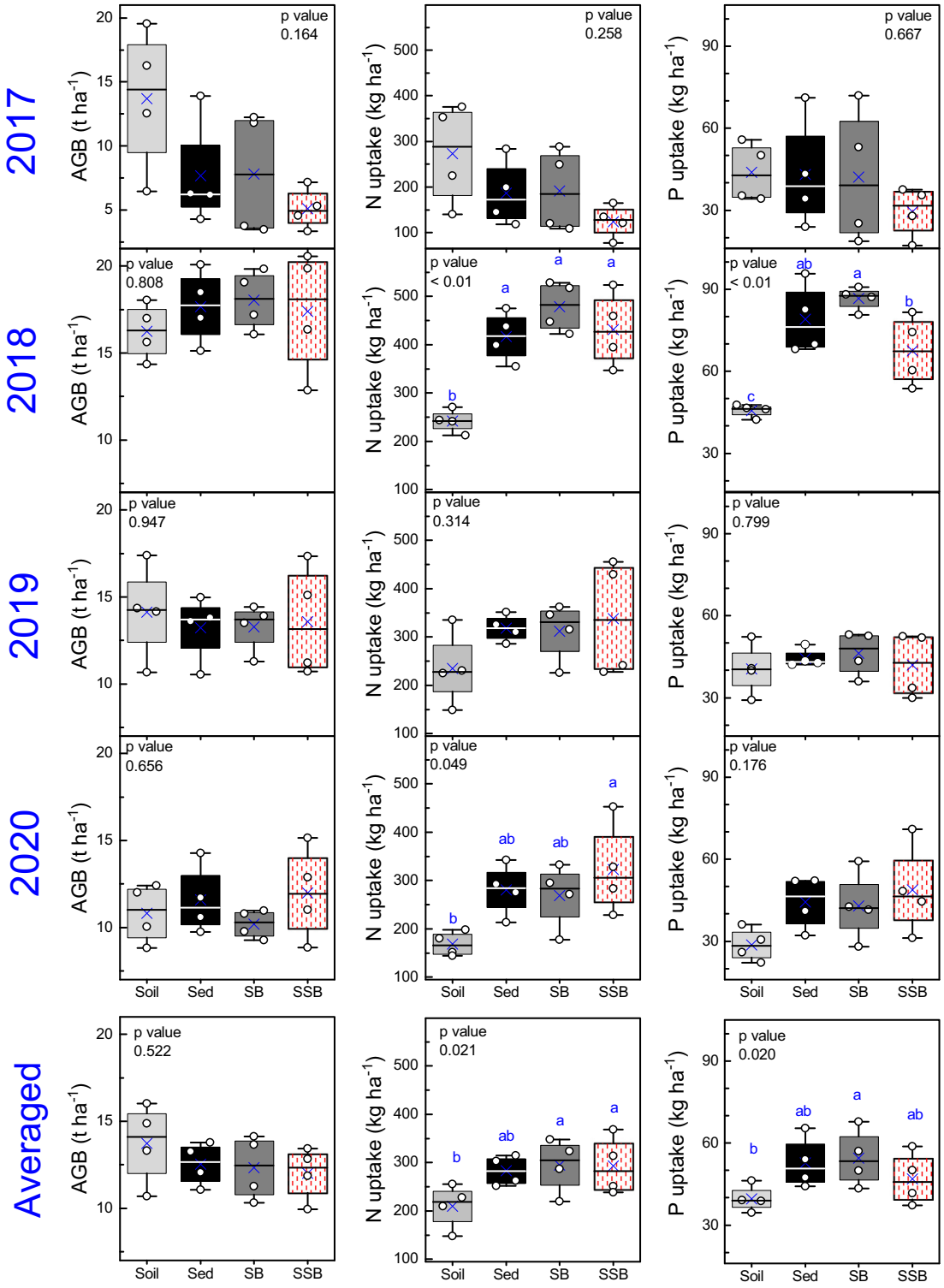

Fig. 3. Effect of growing medium on yearly cumulative plant biomass (AGB, t ha^{-1}), N uptake (kg ha^{-1}), and P uptake (kg ha^{-1}) and the average of these parameters in the four years of the field experiment from 2017 to 2020. The line inside the box represents the median, the blue cross shows the mean, and the top and bottom of the box represent the third (Q3) and first (Q1) quartiles, respectively, the top whisker is Q3 + 1.5 IQR and the bottom whisker is Q1–1.5IQR. Sed: 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; Soil: Pure topsoil. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

concentrations of $PO_4^{3-}-P$ and NO_3^-N in the leachate ranged from 0.11 mg P L^{-1} and 10 mg N L^{-1} in *Soil* to 0.43 mg P L^{-1} in *SB* and 35 mg N L^{-1} in *Sed* (SM Fig. D). The biochar-treated sediment (*SB*) had a significantly lower concentration of P than *Sed* from October 2018 to June 2019, and it was not statistically different from the control *Soil*. Similarly, the concentration of mineral N in the leachate of *SB* was 25 % lower than in the *Sed* in July 2020, and did not differ statistically from that in the *Soil*.

There were no significant differences among the treatments in the daily amount of leached $PO_4^{3-}-P$ or mineral N, or the average volume of

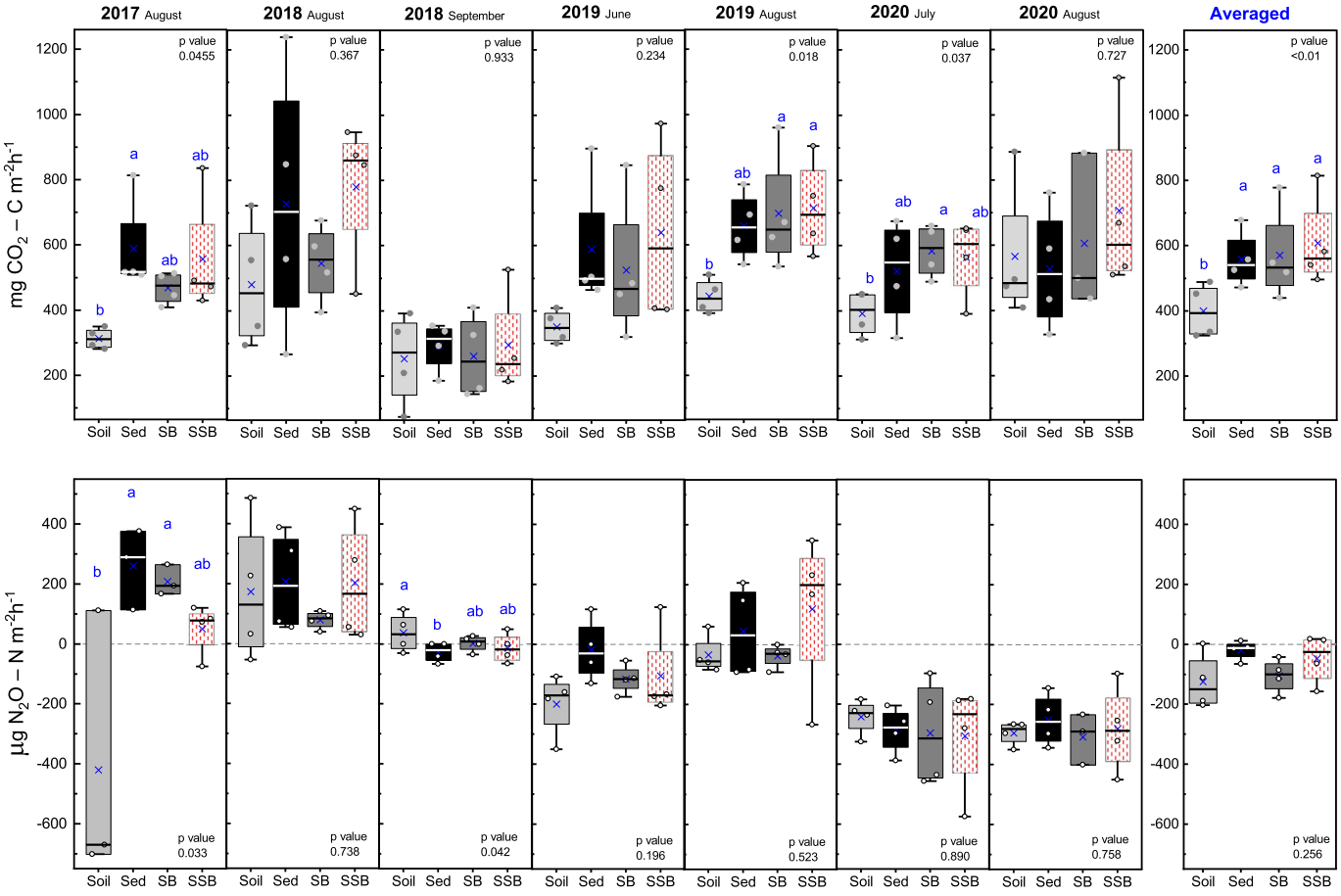

Fig. 4. The average fluxes of CO_2 and N_2O from the different growing medium treatments in the field experiment from 2017 to 2020. The line inside the box represents the median, the blue cross shows the mean, and the top and bottom of the box represent the third (Q3) and first (Q1) quartiles, respectively, the top whisker is Q3 + 1.5 IQR and the bottom whisker is Q1–1.5IQR. Sed: 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; Soil: Pure topsoil. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

leached water, during the measurement periods. The only exception was for the mineral N in 2018 (from June 21 to October 15, 2018), when the highest amount of leached mineral N (6.2 mg m^{-2} d^{-1}) from *SSB* was 62 times more than that from *Soil*, while the leached amounts of mineral N in *Sed*, *SB*, and *Soil* did not differ significantly (Fig. 5). Comparison of *Sed* and *SB* treatments showed that applying 2.5 cm of biochar with sediment materials occasionally but significantly reduced the concentration of leached P from the sediment, even to half the amount from October 2018 to June 2019, and to the degree that it did not differ from that in *Soil* (SM Fig. D). On average, the leached amounts of N and P from the *Soil* were significantly (87 %) lower than those from the sediment-based treatments in the measurements from 2018 to 2020 (Fig. 5).

The mean soil penetration resistance at 0–10 cm and 0–20 cm depths from the surface were significantly lower in treatments containing sediment than in *Soil* in 2017 (SM Fig. E). However, one year later in August 2018, the only significant difference was observed between *Soil* and *SB* (0.75 vs 0.61 MPa) at 0–10 cm depth.

The proportion of water-stable aggregates (WSA) was high in all treatments and ranged between 85 % in *Soil* in 2019 to 98 % in *SB* in 2017 (Table 1a). The *Sed* and *SB* treatments constantly displayed significantly higher aggregate stability than the control *Soil* in the first three years of the experiment. The turbidity values, 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 the *Soil* had significantly higher turbidity (1.85 NTU) compared with *Sed* and *SB* (Table 1a). The Pearson correlation coefficient between WSA and turbidity was −0.451 (significant at $P < 0.01$).

3.4. Microbial community compositions

At the phylum level, the bacterial communities were dominated by Proteobacteria (41.3 % relative abundance across all treatments), Bacteriodetes (13.3 %), Chloroflexi (10.7 %), and Actinobacteria (8.4 %). Twelve phyla accounted for 92 % of the relative abundance of bacteria (Fig. 6). The fungal communities were dominated by Ascomycota (51 %), Mortierellomycota (17.8 %) and Basidiomycota (11.1 %), with seven phyla accounting for 99 % of the relative abundance of fungi (Fig. 6).

In June 2017, the bacterial diversity indices were lower in *Sed* than in *Soil* (SM Fig. F). Also, in August 2017 and 2018, *Sed* and *SB* displayed the least diverse bacterial community compared to the treatments that contained soil material. However, sediment and soil materials showed similar fungal diversity in the three measurement occasions. Also, bacterial and fungal species diversities were not affected by biochar addition to the sediment material. The dbRDA patterns of microbial community composition of bacterial and fungal significantly separated *Soil* and sediment-based treatments (SM Fig. G) in both bacterial and fungal ordination, explaining ~29 and 46 % of the total variation of bacterial and fungal communities, respectively, along the first and second axes.

There were 34 bacterial phyla and 18 fungal phyla that were differentially abundant in at least one of the treatments that contained sediment

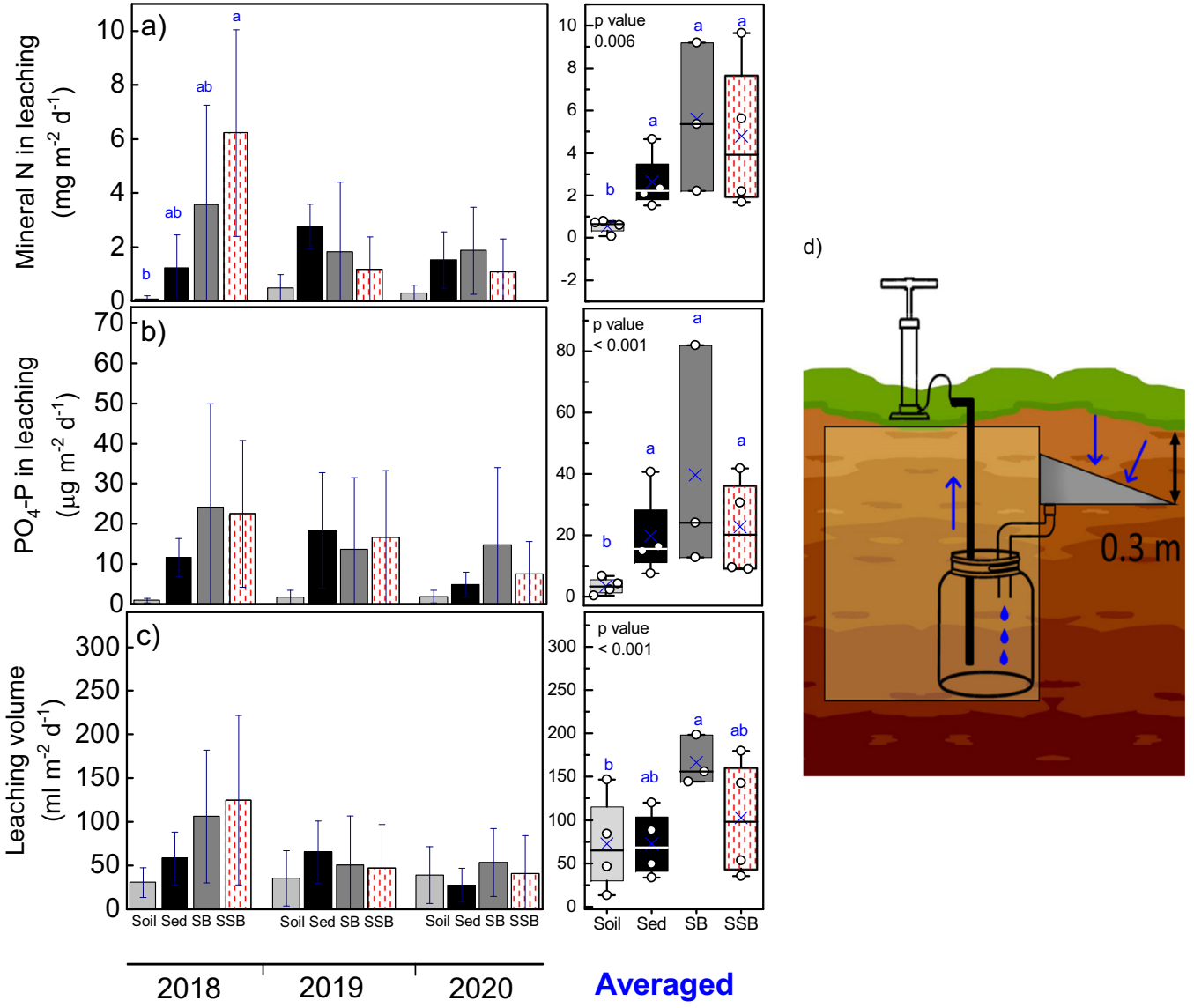

Fig. 5. Effect of growing medium treatments on a yearly average of leached mineral N (a; NO_3^-N and NH_4^+ —N), leached PO_4^{3-} P (b), and leaching volume (c) in the field experiment during 2018–2020 collected by on-site lysimeters (d; Voll and Roots, 1999). The values were calculated as cumulative amounts of leaching in the growing season divided by the total days of leaching collection. (d) the schematic view of the lysimeters installed at depth of 30 cm from the surface. Sed: 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; Soil: Pure topsoil.

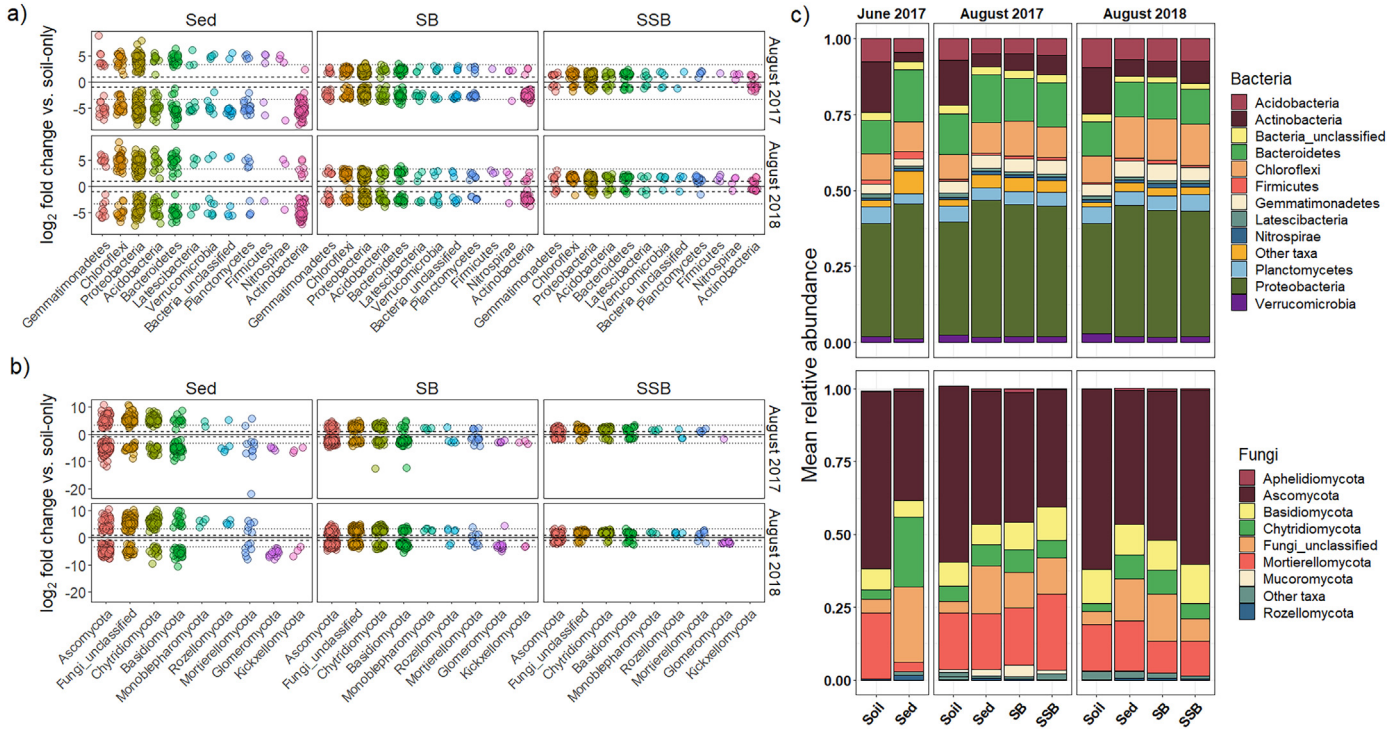

Fig. 6. Log2-fold change in the relative abundance of bacterial OTUs (a) and fungal OTUs (b) as compared with soil treatment in the field experiment during 2017–2018. Each point represents a significant increase or decrease of OTU based on Benjamini and Hochberg adjusted $P > 0.10$. Dashed and dotted lines represent increases or decreases in the relative abundance of 2× and 10×, respectively. c) Relative abundance of most dominant bacterial and fungal taxa at the phylum level across treatments. Values are means of 4 replicates. Sed: 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; Soil: Pure topsoil.

when compared with the control *Soil*. Among these, the relative abundances of phyla Actinobacteria and Kickxellomycota were lower, and those of the phyla Nitrospinae and Monoblepharomycota were higher in the sediment-based treatments than in *Soil* ($P < 0.05$, Table 2). Compared to the control *Soil*, the |log2-fold change| of the phylum Nitrospirae and genera *Nitrospira* and *Anaeromyxobacter* were higher in *SB*, but not in pure sediment in 2018 (Table 2). Also, Firmicutes was the only phylum that had bacterial OTUs whose relative abundances were significantly higher in treatment containing biochar (*SB* and *SSB)* compared with control *Soil* in 2017 and 2018, while there were no bacterial OTUs of this phylum with significantly higher abundance in the *Soil* than *SB* and *SSB* (Fig. 6a). There were 11 unique bacterial phyla found in the sediment representative material, including Calditrichaeota, Fusobacteria, and Atribacteria (data not shown). There were no unique fungi at the phylum level in representative soil and sediment materials.

4. Discussion

4.1. Recycling of lake sediment in agriculture

The grass yields in the last three years of the experiment (average of 12.3 t ha^{-1}) were close to the average multiannual forage crops yield in Estonia (12.2 t ha^{-1}) during 2004–2017 (Statistics Estonia, 2021). The overall grass yield was on the same level among all treatments (Fig. 2). This result contrasts with findings of Kiani et al. (2021) that addition of the same sediment doubled the growth of ryegrass, in comparison with the control soil, in a pot experiment in a controlled environment. We suggest that no increment in plant growth in our field experiment could be due to the different plant species (a mixture of grasses vs. sole ryegrass in the pot experiment) and their nutrient needs. More diverse plant communities are expected to acquire more growth resources and transform 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). Furthermore, the potential benefits of sediment addition and nutrient release could have been reduced due to the abundant weed accumulation such as redshank, which is known to be competitive against crop plants, especially in moist areas (CABI, 2019).

The harvested yield from the sediment-based treatments fulfilled the nutritional requirements for livestock fodder regarding most of the nutrients (Table 1b; Freer, 2007; Suttle, 2010; Bahamonde et al., 2016), showing that the recycled sediment is well-suited for feed production. Overall, Kiani et al. (2021) reported low contents (below the Finnish standards) of heavy metals and organic contaminants in the sediment of Lake Mustijärv, implying that no ecological or health risks should occur.

As years passed the *Soil* N content decreased and C:N ratio increased (SM Fig. A). The increase in the C:N ratio was due to a decrease in *Soil* N ($r = -0.932$, $P < 0.01$), but not related to C. This suggests that the N pool is more dynamic than the C pool in *Soil*, while a similar relation was not observed in *Sed*. The average annual plant N uptake was 306 and 230 kg N ha^{-1} from *Sed* and *Soil*, respectively (Fig. 3). This could not be all derived from a single application of MBM (100 kg N ha^{-1}), but mainly from the mineralised organic matter of the growing medium. When an organic substrate has a C:N ratio below 15, rapid mineralisation and release of plant-available N occurs (Brust, 2019). Hence, the C:N ratio ranging from 10 to 18 in the different growing media of the current study indicates that conditions favoured the supply of N from the decomposition of organic matter in all treatments. However, considering the negative correlation of N mineralisation and clay:C ratio (Soinne et al., 2021), the five times lower clay:C ratio in sediment material than in soil suggested a higher rate of N mineralisation in sediment. This is in line with higher plant N contents in the sediment-based treatments than in the *Soil* in the last three years.

Table 2
Summary of the differential abundance of selected taxa between control soil and the rest of the treatments in the field experiment in 2017–2018. Samples were collected at the end of the growing seasons in August. Analysis was done using the DESeq2 package. P Adj: Adjusted *P* value which accounts for multiple testing and controls the false discovery rate. L2F diff: Log 2-fold difference. The L2F diff values with P Adj < 0.05 are indicated in bold. Positive values indicate a higher abundance in the treatment compared to the *Soil*. Negative values indicate a lower abundance in the treatment compared to the *Soil*.

Bacterial Taxa		2017			2018		
		Sed	SB	SSB	Sed	SB	SSB
Phylum							
Proteobacteria	P Adj	0.537	0.928	0.116	0.004	0.022	0.241
	L2f diff	0.02	0.00	−0.03	**0.18**	**0.06**	0.04
Bacteroidetes	P Adj	0.582	0.118	0.009	0.706	0.966	0.743
	L2f diff	−0.04	−0.08	**−0.06**	−0.05	0.00	−0.01
Actinobacteria	P Adj	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
	L2f diff	**−2.07**	**−0.83**	**−0.51**	**−1.53**	**−0.78**	**−0.36**
Chloroflexi	P Adj	0.497	0.047	0.708	<0.001	<0.001	<0.001
	L2f diff	0.05	**0.13**	0.01	**0.59**	**0.30**	**0.19**
Planctomycetes	P Adj	<0.001	<0.001	<0.001	0.002	0.010	0.227
	L2f diff	**−0.72**	**−0.26**	**−0.18**	**−0.35**	**−0.14**	−0.03
Verrucomicrobia	P Adj	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
	L2f diff	**−0.785**	**−0.314**	**−0.244**	**−0.698**	**−0.452**	**−0.225**
Nitrospirae	P Adj	0.553	0.036	0.167	0.089	<0.001	0.001
	L2f diff	0.12	**0.20**	0.09	0.30	**0.34**	**0.20**
Firmicutes	P Adj	0.743	0.173	0.415	0.025	<0.001	0.217
	L2f diff	−0.091	0.218	0.067	**0.518**	**0.457**	0.170
Nitrospinae	P Adj	0.002	<0.001	0.012	0.002	0.031	0.031
	L2f diff	**4.49**	**2.37**	**1.32**	**4.61**	**1.89**	**1.20**
Genus							
Thiobacillus (Proteobacteria)	P Adj	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
	L2f diff	**3.75**	**1.90**	**1.13**	**5.23**	**2.64**	**1.67**
Nitrospira (Nitrospirota)	P Adj	0.441	0.394	0.205	0.324	<0.001	0.002
	L2f diff	−0.18	0.10	0.10	0.23	**0.40**	**0.23**
Terrimonas (Bacteroidetes)	P Adj	0.709	0.339	0.704	0.001	<0.001	<0.001
	L2f diff	−0.12	−0.16	0.04	**1.01**	**0.56**	**0.40**
Geobacter (Proteobacteria)	P Adj	<0.001	<0.001	0.007	0.001	0.012	0.003
	L2f diff	**1.36**	**0.65**	**0.28**	**1.25**	**0.52**	**0.32**
Pseudomonas (Proteobacteria)	P Adj	0.606	0.193	0.341	0.396	0.950	0.464
	L2f diff	0.27	−0.24	−0.14	0.29	−0.01	0.08
Sulfuricurvum (Epsilonbacteraeota)	P Adj	<0.001	<0.001	0.001	<0.001	<0.001	<0.001
	L2f diff	**4.36**	**2.04**	**1.03**	**4.65**	**2.49**	**1.39**
Cellvibrio (Proteobacteria)	P Adj	0.005	0.182	0.006	0.407	0.816	0.690
	L2f diff	**1.33**	0.37	**0.42**	−0.82	−0.13	0.23
Rhodanobacter (Proteobacteria)	P Adj	0.022	0.025	0.044	–	–	–
	L2f diff	**−4.57**	**−3.67**	**−1.57**	–	–	–
Nitrosomonas (Proteobacteria)	P Adj	<0.001	<0.001	0.007	0.003	0.002	0.040
	L2f diff	**3.19**	**1.67**	**0.90**	**4.12**	**2.15**	**1.16**
Lysobacter (Proteobacteria)	P Adj	0.000	<0.001	0.004	<0.001	<0.001	0.029
	L2f diff	**−5.62**	**−2.86**	**−1.26**	**−5.97**	**−2.57**	**−0.59**
Hyphomicrobium (Proteobacteria)	P Adj	0.470	0.773	0.645	0.049	0.289	0.079
	L2f diff	−0.78	0.15	0.17	**2.16**	0.67	0.61
Opitutus (Verrucomicrobiota)	P Adj	0.067	0.505	0.620	0.876	0.120	0.510
	L2f diff	−3.64	−0.49	−0.28	−0.22	−1.56	−0.38
Anaeromyxobacter	P Adj	0.641	0.191	–	0.059	0.044	0.245

Fungal Taxa		2017			2018		
		Sed	SB	SSB	Sed	SB	SSB
Phylum							
Ascomycota	P Adj	0.002	0.005	0.003	0.001	0.027	0.403
	L2f diff	**−0.90**	**−0.54**	**−0.30**	**−1.01**	**−0.30**	−0.07
Basidiomycota	P Adj	0.218	0.394	0.706	0.040	0.091	0.820
	L2f diff	−0.75	−0.28	0.07	**−0.69**	−0.24	0.05
Chytridiomycota	P Adj	0.987	0.658	0.804	0.001	<0.001	<0.001
	L2f diff	0.01	−0.11	−0.06	**0.93**	**0.64**	**0.27**
Glomeromycota	P Adj	0.154	<0.001	0.122	<0.001	<0.001	<0.001
	L2f diff	−1.88	**−3.91**	−0.39	**−3.87**	**−1.50**	**−1.20**
Monoblepharomycota	P Adj	<0.001	<0.001	0.002	<0.001	<0.001	<0.001
	L2f diff	**3.72**	**1.59**	**0.94**	**4.54**	**2.62**	**1.43**
Kickxellomycota	P Adj	<0.001	<0.001	0.001	<0.001	<0.001	<0.001
	L2f diff	**−8.38**	**−4.44**	**−0.96**	**−2.17**	**−0.26**	**−5.74**
Genus							
Fusicolla (Ascomycota)	P Adj	<0.001	0.106	0.452	0.468	<0.001	0.416
	L2f diff	**−4.48**	−0.88	0.22	−0.83	**−1.53**	0.26
Species							
Claroideoglomus claroideum (Glomeromycota)	P Adj	<0.001	0.002	<0.001	0.216	0.680	0.338
	L2f diff	**2.89**	**−3.02**	**0.40**	−2.01	−0.45	−0.56

(continued on next page)

Table 2 (*continued*)

Bacterial Taxa		2017			2018			Fungal Taxa	2017			2018		
		Sed	SB	SSB	Sed	SB	SSB		Sed	SB	SSB	Sed	SB	SSB
(Proteobacteria)	L2f diff	0.91	1.02	–	3.09	**1.62**	0.80							
Methylobacterium	P Adj	–	–	0.353	0.590	–	–							
(Proteobacteria)	L2f diff	–	–	0.61	−1.11	–	–							

Also, the constant value of N content in sediment-based treatments versus its sudden reduction in the *Soil* from the second year implied a constant supply of N in sediment treatments.

The content of easily soluble P together with the labile P and Fe − P fractions in the growing medium, and plant P contents were higher in the sediment-based treatments than in the *Soil*, indicating the high availability of P in sediment materials (Table 1 and SM Fig. B). In comparison to *Soil*, the high biomass P contents in all sediment treatments were associated with significantly higher P uptake in all sediment-based treatments compared with *Soil* in 2018, and the effect was less pronounced in the mean P uptake across the whole experiment (Fig. 3). The significant depletion of Fe − P in the 0–20 cm layer of *Sed* may imply a larger contribution of Fe − P to P uptake compared to other fractions (SM Fig. B). The high availability of P in sediment materials was mainly due to the low level of fine clay material (18 % in sediment), as well as associated Al and Fe (hydr) oxides, which may have reduced the P adsorption by metal oxides (Sippola, 1974; Laakso et al., 2017). This idea is consistent with previous studies reporting higher contents of available P in sediment materials with low clay contents (< 20 %) compared to agronomic soil (Mattei et al., 2018; Tozzi et al., 2020). 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 the available P in sediments. Additionally, in the *Sed*, the easily soluble P had a strong positive correlation with TC ($r = 0.898$, $P < 0.01$, SM Fig. A) suggesting that the bioavailable P pool was supplied by mineralisation of sediment organic matter. However, a similar relationship was not observed in the *Soil* treatment.

Over the four years, plants were able to take up significantly more N and P in biochar-treated sediments than in *Soil*, but the uptake did not differ significantly from the pure sediment treatment (Fig. 3). The impacts of biochars are most pronounced in infertile soils (El-Naggar et al., 2019). It is well known that biochars can adsorb initial mineral N from the soil and the captured N can be released over extended periods of plant growth (Kammann et al., 2017; Schmidt et al., 2021). Moreover, biochars can increase plant P uptake due to its higher anion exchange capacity in soil (DeLuca et al., 2015). Meta-analyses have suggested a positive effect of biochars on the plant P uptake, if P is a limiting factor of plant growth (Gao et al., 2019; Glaser and Lehr, 2019). In this study, however, the sediment was apparently fertile enough (rich in N and P) that such effects of added biochar were not detected.

On the other hand, mixing a 2.5 cm layer of biochar with K-poor sediments improved K status in the *SB* and *SSB* treatments, and increased the plant K content of biochar-treated growing media (Tables 1a and 1b). This was likely due to the K-fertilization effect of the hardwood biochar containing 7.2 g K per kg. Our results are consistent with earlier reports showing that the application of biochar made from woody raw material increased the K content of the growing medium (Tammeorg et al., 2014; Kalu et al., 2021). However, plant growth was likely not limited by K in any of the treatments, as its content in the biomasses was constantly higher than 8 g kg^{-1} (De Wit et al., 1963; Lawniczak et al., 2009).

The easily soluble S content sharply decreased over time in all growing media. Since most soils contain >90 % of the total S in organic forms (Weil and Brady, 2017), the easily soluble S may have been released during the organic mineralisation process of 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 that sulfate adsorption is influenced by the presence of other anions. The stronger adsorption of phosphate than sulfate is the basis for the extraction of adsorbed sulfate (Tabatabai et al., 1988) and the addition of phosphate to soils has been shown to increase sulfur leaching (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 the leaching of S was not studied in this experiment, the leached sulfur can cause problems to lakes such as acidification of lakes in Scandinavian regions (Wright and Henriksen, 1978). Sulfate, similar to nitrate, is negatively charged and not held tightly by clay particles. Therefore, similar management options can be used for minimizing the leaching of both nitrate and sulfate. Due to the high content 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 way that fertilizers are applied to avoid environmental risks. Moreover, further research could lead to interesting neutralizing effects in acidic soils that require liming, as the sediment has a high pH above 7.2.

4.2. Risks of greenhouse gas emissions from recycled lake sediment in agriculture

Recycled sediment had much greater CO_2 and N_2O emissions than those from the control *Soil* in the first year (Fig. 4). Excavation and cultivation of organic-rich sediment allow oxygen to enter the sediment, which initiates rapid microbial decomposition of the stored organic material, and in turn, increases CO_2 and N_2O emissions (Kasimir-Klemedtsson et al., 1997). The organic-rich sediment material having a C:N ratio lower than 15, along with the loss of 1.2 kg m^{-2}C in four years, further supports the idea of considerable microbiological decomposition of organic matter in the sediment material. Also, soil moisture is the single-most important soil parameter for 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 the *Soil*, while there was no similar pattern in the *Sed* (SM Fig. A). This suggests that factors other than temperature and moisture might exist that were more critical for determining the CO_2 emissions from sediment in this study. The average of four-year measurements showed that the sediment-based growing media emitted more CO_2 than *Soil*. More research is needed to study whether GHG emissions from recycled sediments can be reduced by using the sediments in lower application rates (i.e. <50 Mg^{-1} ha) similarly to organic fertilizers and soil amendment materials, instead of using excavated lake sediments in large quantities as a growing medium in our study.

On average, the *SSB* emitted 50 % more CO_2 than *Soil* in the measurements from 2017 to 2020 (Fig. 4). In *SSB*, where the top 20 cm was a mixture of soil with low C content plus biochar and organic-rich sediment, labile fractions of biochar and sediment may accelerate microbial activities by providing them with selective substrates (Cross and Sohi, 2011). This may have increased the CO_2 emissions (Yu et al., 2013; Sagrilo et al., 2015).

The wood-based activated biochar slightly reduced CO_2 emissions from the sediment in the year of application, but this effect was not statistically significant. In general, biochar produced at high pyrolysis temperatures

(> 500 °C) shows a relatively high efficacy in reducing GHG emissions (Schmidt et al., 2021). But a significant reduction of GHG emission by the wood-derived biochar pyrolyzed at >500 °C was not observed in the current study.

Nitrous oxide is produced as a by-product of the microbiological processes of nitrification and denitrification in soils (Kasimir-Klemedtsson et al., 1997) which can explain the higher N_2O emission in *Sed* than in *Soil* in 2017. However, there was a net consumption of N_2O, i.e. negative fluxes of N_2O, in some of measurement occasions. Net negative N_2O fluxes often connected to low mineral N and large moisture contents (Chapuis-Lardy et al., 2007). Ryden (1983) associated N_2O sink activity of soil with a low NO_3^- level (1 mg N kg^{-1} soil), a soil moisture content above 20 % and relatively low soil temperatures of 5–8 °C.

Even though the excavated sediment had higher CO_2 emissions, it is important to consider that eutrophic shallow (< 20 m) lakes can be important sources of GHGs (Li et al., 2021; Sun et al., 2021). Huttunen et al. (2003) reported CH_4 emissions of up to 12 mmol m^{-2} d^{-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 a mean CO_2 emission of 36 mmol m^{-2} d^{-1} and CH_4 emission of 6 mmol m^{-2} d^{-1}. The low oxygen content, abundant organic matter and nutrients, dominant primary producers, and harmful algal blooms in eutrophic freshwaters have been found to favour GHG emissions (Li et al., 2021). Therefore, the magnitude of any GHG emissions from an unexcavated lake would give a larger frame of reference for the GHG emissions from agricultural use of excavated sediment.

4.3. Risks of N and P leaching from the recycled lake sediment in agriculture

The EC values in the leachate of sediment-based treatments decreased during the experiment to 1.09 mS cm^{-1} in 2020 which was in the range of freshwater streams (0.05 to 1.5 mS cm^{-1}; Behar, 1997). In general, the four-year average concentrations of $PO_4^{3-}-P$ and $NO_3^- - N$ in the leachate of all treatments were below the threshold values for causing risk in groundwaters set by the European Union ($PO_4^{3-}-P < 4.4$ mg L^{-1}, $NO_3^- - N < 50$ mg L^{-1}; European Commission, 2010).

The concentration of easily soluble P in the *Sed* remained far greater than in the *Soil* treatment (Table 1a) during the four years of the experiment. However, plants did not take up significantly more P (33 %, $P = 0.127$) in the *Sed* compared to those in the *Soil* (except for 2018), implying that the availability of P was apparently not limiting plant growth in this experiment. Labile nutrients are both more prone to leaching and more readily taken up by plants (Lehmann et al., 2003). In line with this, six times more P was lost through leaching from the P-rich sediment than from the *Soil* after four years of applying sediment (Fig. 5).

Similar results were obtained on the leached mineral N during the four years of experiments: an average during the four years, the *Sed* treatment had five times more N in drained water and only 35 % more N uptake compared to those in the *Soil* (Figs. 3 and 5), even if the total N concentrations in the *Sed* growing medium were sixfold of those in *Soil* (Table 1a). The possible reason that the N and P leaching was high in sediment treatments could be due to excessive amounts of N and P in relation to the plants' needs. In keeping with this notion, the significantly lower soil penetration resistance in the sediment-based treatments compared to the *Soil*, especially in 2017, implied that the leaching was facilitated by pathways of loose and hollow sediment materials (SM Fig. E).

The concentration of P in the leachate from the biochar-treated sediment was statistically as low as that from the *Soil* treatment. A similar short-lived beneficial effect of biochar on reducing the concentration of mineral N in the leachate from sediment in July 2020 may be attributed to short-term NO_3^- immobilisation. The labile C fraction in biochar may cause such an effect, leading to reduced N leaching from the soil to the environment (Kolb et al., 2009; Tammeorg et al., 2012). Further, the temporary movement into and storage of nitrate and phosphate inside mid-sized biochar pores may also cause similar results. Phosphorus retention by biochar is also described as a sorption mechanism that would often decrease the leaching losses of P, resulting in more retained P in the soil available for plants (Yang et al., 2021). However, the total effects of added biochar on the leaching of N and P from sediment were not significant across all four years of the study.

In our study, the aggregates in sediment-based treatments were more stable than those in *Soil* (Table 1a). Since the breakdown of soil aggregates produces both runoff-promoting crusting and easily transportable particles, aggregate stability is commonly used as an indicator of soil erodibility (Le Bissonnais, 1996). The stable macroaggregate content of topsoil has been shown to correlate inversely with both runoff rate and soil losses (Barthès and Roose, 2002). According to Sarker et al. (2018), labile organic materials are more capable of improving aggregate structure than the stable organic matter. This notion is in line with our study that the *Sed* with a higher rate of labile organic matter compared to the *Soil* had higher stability of aggregates. In addition, the 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). The significantly higher EC values in sediment material agree with this notion (Table 1a).

The main sources of N and P input to Lake Mustijärv are certainly the agricultural fields surrounding the lake as the upstream watershed area is mostly used as cattle pastures and annual crop production with conventional tillage. Therefore, the rate of sediment application should be adjusted to match the requirements of crops to minimize nutrient leaching back into the lake and further help mitigate eutrophication in the lake.

4.4. Diversity of microbial communities in soil and sediment-based growing media

The sediment material had a lower level of bacterial diversity compared with soil (SM Fig. F). Liu et al. (2018) also reported a lower microbial alpha diversity in the electronic-waste contaminated river sediments than in soil. While increased microbial diversity has been reported to correlate with higher soil organic carbon content and fertility (Wolińska et al., 2017), this was not the case in our study. In addition, ordination analysis demonstrated a different distribution pattern of dominant genera in the sediment-based growing media in comparison with the control soil (SM Fig. G). The differences in the bacterial and fungal community compositions were mainly associated with the soil Zn, Mn and TN (29 % of variation), and soil Zn, Mn, K and TC (46 % of variation), respectively, indicating that these factors are likely important determinants of bacterial and fungal community structure and their changes.

At the phylum level, Proteobacteria as one of the main soil P-mobilising bacteria with an important role in the mineralisation of refractory organic P (Zhang et al., 2021), had a higher abundance in *SB* and *Sed* treatments compared with *Soil* in 2018 (Table 2, Fig. 6c). Similar result was observed in 2018 for Chloroflexi which has the potential to solubilise bioavailable P fractions (H_2O-P and NaOH-P; Luo et al., 2021). Actinobacteria and Planctomycetes have the potential to accumulate bioavailable P fractions (Luo et al., 2021). Both had a higher abundance in the *Soil* than the sediment-based treatments. However, the *Lysobacter* genus, which is reported to facilitate mineralisation of labile organic P, was more abundant in the *Soil* than in the sediment-based treatments in 2017 and 2018, despite its low relative abundance. Besides bacteria, fungi can also solubilise and accumulate P (Ye et al., 2019). Fungal phyla Ascomycota and Chytridiomycota can solublize both bioavailable and refractory P fractions (Luo et al., 2021). In our study, the relative abundance of Ascomycota was higher in the *Soil* than in other treatments in both 2017 and 2018 (Table 2). Chytridiomycota had a higher abundance in sediment-containing treatments in 2018, but not in 2017.

Calditrichaeota, Fusobacteria, and Atribacteria were the unique phyla in the sediment representative material. Previous studies reported that these phyla are commonly found in aquatic sediments and involved in organic matter transformations such as degradation of detrital proteins (Calditrichaeota, Marshall et al., 2017), are abundant in organic-rich sediments (Fusobacteria and Atribacteria, Maintinguer et al., 2015; Lee et al.,

2018), and can ferment a variety of organic compounds (Atribacteria, Baker et al., 2021). These unique phyla may confirm the various organic carbon mineralisation pathways in sediment-based treatments. In addition, the relative abundance of some groups of bacteria and fungi associated with the decomposition of organic matter, was lower in at least one of the sediment-based growing media than in the *Soil* (Table 2): Actinobacteria (decomposition of organic matter, Chen et al., 2022), Basidiomycota (degradation of even recalcitrant organic matter, Hellequin et al., 2018), and Verrucomicrobia (positively correlated with recalcitrant C compounds, Fierer et al., 2013).

At the genus level, a group of nitrogen-fixing genera such as *Terrimonas* (Tan et al., 2021) and *Cellvibrio*, and nitrifier genera such as *Nitrosomonas* (oxidise ammonia to nitrite) and *Nitrospira* (oxidise nitrite to nitrate) were more abundant in at least one of the sediment-based treatments than in *Soil*. On the other hand, the relative abundance of denitrifier *Geobacter* was higher in sediment-based treatments than in *Soil* in both years. The *Claroideoglomus claroideum,* a fungal strain involved in improving soil N, P, and K fertility (Rashid et al., 2016), was more abundant in the *Sed* than *Soil* treatment in 2017. Also, *Sulfuricurvum* and *Thiobacillus* were significantly more abundant in the sediment-based treatments than in the *Soil*. They are known to have degradation genes for toxic organic compounds and various mechanisms for heavy metal tolerance (Liu et al., 2018; Tian et al., 2020). The *Sulfuricurvum* is also able to oxidise sulfur and make it available for plants which is in line with the high content of easily soluble S in sediment-based growing media.

5. Conclusions

This study addressed the temporal changes in the effects of different sediment application methods on soil, plants and environment in grass production. Using excavated sediment from the eutrophic Lake Mustijärv with a low Fe:P ratio (6) in a thick layer as a growing medium resulted in the increased availability of phosphorus and other nutrients compared with agricultural sandy loam soil. The grasses in the sediment-based growing media had similar growth performance compared to the agricultural soil, and even significantly better quality, as the nitrogen content of plants grown in the sediment was higher compared with the *Soil*. As for the environmental impacts, the high amount and rate of decomposition of organic matter in the pure sediment no doubt caused initially higher CO_2 and N_2O emissions compared with the *Soil* in the first year. On the average, during the four years of measurements, the sediment-based growing media emitted more CO_2 than the *Soil*. Also, the excavated sediment increased the risk of phosphate and mineral nitrogen leaching in all sediment-based growing media. However, their average concentrations in the leachate were below the threshold values for causing risk in groundwaters set by the European Union. Applying biochar did not significantly reduce the sediment's CO_2 and N_2O emissions and the P and N leaching. However, plants were able to take up more N in biochar-treated SB and SSB than in *Soil*. The bacterial and fungal community structures of sediment-based growing media differ strongly from those in *Soil*, with a higher abundance of P-mobilising Proteobacteria and a lower abundance of P-accumulating Actinobacteria and Planctomycetes in the sediment-based treatments. Biochar did not significantly change microbial communities. Our results indicated that recycling nutrient-rich lake sediment on the agricultural soil close to the lake of origin may be an environmentally friendly alternative for disposing of large amounts of excavated lake materials providing potential co-benefits to lake restoration by sediment removal. More research is needed to study whether GHG emissions and nutrient leaching from recycled sediments can be reduced by using the sediments in lower application rates (i.e. <50 Mg^{-1} ha) similarly to organic fertilizers and soil amendment materials, instead of using excavated lake sediments in large quantities as a growing medium in our study. Our small case study can likely be upscaled to lakes with similar sediment properties and used in the life-cycle assessment of environmental impacts of sediment recycling of eutrophic lakes for agricultural purposes in the future.

CRediT authorship contribution statement

Mina Kiani: Conceptualization, Data collection, Formal analysis, Funding acquisition, Investigation, Methodology, Resources, Writing - Original Draft, Visualization.

Jure Zrim: Data collection, Methodology, Formal analysis, Validation, Writing - Review & Editing.

Asko Simojoki: Conceptualization, Data collection, Methodology, Supervision, Validation, Writing - Review & Editing.

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Data availability

Data will be made available on request.

Declaration of competing interest

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

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Appendix A. Supplementary data

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