



# Twenty years of afforestation of former agricultural lands with silver birch plantations affects vertical distribution of SOC and macronutrients in the topsoil layer

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## Abstract

**Purpose** Fast-growing tree plantations on abandoned agricultural soils is a promising management system to sequester atmospheric CO<sub>2</sub>. However, the effects of fast-growing trees on the nutritional and organic carbon (SOC) status of soils degraded by agriculture, are poorly understood.

**Methods** We sampled the soil after 20 years in 10 silver birch plantations on former agricultural soils in hemiboreal Estonia to assess changes in soil chemical properties (SOC, N, C:N ratio, pH<sub>KCl</sub>, P, and K) in 10-cm vertical mineral soil layers to a depth of 30 cm and to determine the potential environmental drivers of plant-soil interactions.

**Results** We observed no depletion of SOC or macronutrients in the upper 0–30-cm soil layer, but found some vertical shifts among the sublayers. The SOC

concentration increased by 22% in the upper 0–10-cm soil layer, especially in sites with higher above-ground productivity. Simultaneously, SOC concentration decreased by 17% in the 20–30-cm soil sublayer, which indicating trees' ability to alter decomposition activity in deeper vertical soil layers. In the 20–30-cm sublayer, SOC mineralization was supported by an 11% decrease in the C:N ratio. Similarly, the total N concentration increased in the 0–10-cm soil layer by 13%. The concentration of plant-available P increased by ~30% in the 20–30-cm sublayer.

**Conclusion** Two decades of afforestation of former agricultural soils caused vertical stratification of SOC in the upper mineral soil layer (0–30 cm) where trees can access deeper nutrient pools for active cycling, but caused no loss of SOC or nutrients.

**Keywords** Birch · Afforestation · SOC · Nitrogen · Phosphorus · Soil acidity

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## Introduction

Although fast-growing tree plantations are widely established in some parts of the world, their implementation on former agricultural lands for timber production is still a novel silvicultural practice in Northern Europe. Tree plantations have a great potential to counteract climate change consequences through atmospheric CO<sub>2</sub> uptake in the region (Lutter et al. 2016a, 2021a; Mola-Yudego et al. 2017; Bastin

et al. 2019; Fradette et al. 2021). About 1.8–2.6 Mha of abandoned agricultural land is available for tree plantations in Northern Europe (Rytter et al. 2016), representing an underused land resource to produce additional wood fiber and reduce fellings in native forests (Weih 2004; Tullus et al. 2013). Afforestation is also a tool with which to meet European Union climate targets under the EU Forest Strategy for 2030 of planting an additional 3 billion trees on non-forested land (European Commission 2021) to build carbon (C) stocks in the landscape and provide wood for sustainable bioeconomies. In addition, important C pools are located in soils in whose dynamics and stabilization are also important under climate change after afforestation (Guo and Gifford 2002; Bárcena et al. 2014a).

Theoretically, afforestation will cease both soil disturbance (e.g. ploughing and soil compaction by heavy agricultural machinery) and the annual removal of organic material (i.e., crop harvesting) compared to the previous agricultural land use. Inputs of leaf and root litter into the soil should support shifts toward increasing carbon–nitrogen ratios and accumulating lignin-based organic compounds to promote long-term C stabilization and accumulation in soil (Baum et al. 2009; Mayer et al. 2020). In contrast, it has been posited that intensive forest management such as clearcutting and related soil physical disturbances reduce soil organic carbon (SOC) levels in forestland (Noormets et al. 2015). Few attempts have been made to describe SOC changes after the afforestation of former agricultural soils with fast-growing tree plantations over the longer term in Northern Europe.

So far, existing empirical studies with repeated soil sampling examining the SOC balance have mainly covered the early development period of plantations (< 15 years) (Rytter 2016; Lutter et al. 2016a, b; Rytter and Rytter 2020; Steffens et al. 2021). These studies report no general SOC increase or decrease in the upper 30 cm of mineral soil, which should be followed by recovery and finally accumulation in the later stages of stand development (> 20 years) (Bárcena et al. 2014a; Mayer et al. 2020). The phenomenon of no general SOC increase or even decrease at the establishment phase can be explained by the high demand of trees for nutrients to maintain productivity, while enhanced enzymatic and microbial activities decompose existing organic material pools, releasing nutrients for tree growth (Cebrián and Duarte 1995;

Fontaine et al. 2004; Vesterdal et al. 2012; Alberti et al. 2017). In addition, another driver of the high CO<sub>2</sub> efflux rates in comparison with the litter input could be linked to the higher soil temperature that enhances microbial activity due to low canopy cover and sun-exposed soil in the plantation establishment phase (e.g. Subke et al. 2003). In some cases, for example, in former grasslands, the loss of SOC in fast-growing tree plantations could neutralize the ecosystem C gain due to high soil C efflux and organic material decomposition (Bárcena et al. 2014a; Lutter et al. 2016a).

The key question in a short-rotation forestry system is SOC recovery and stabilization which should occur within the rotation cycle or otherwise the system will reduce the SOC pool existing in the ecosystem before afforestation (Mayer et al. 2020). SOC recovery has been demonstrated by studies examining longer time periods (> 20 years) after the afforestation of agricultural lands in Northern Europe using conventional forestry systems with slower-growing tree species, such as spruces and oaks (Vesterdal et al. 2002; Ritter et al. 2003; Bárcena et al. 2014a, b). However, the SOC accumulation in tree plantations is questionable as the rotation cycle of short-rotation forestry is usually under 30 years (Weih 2004; Tullus et al. 2013; Rytter et al. 2016). Considering the importance of SOC dynamics under climate change and the variability reported in studies from Northern Europe, longer-term observations of various soils and tree species are urgently needed for short-rotation forestry systems, whose rapid expansion is expected in the region (European Commission 2021). Although the main aim of tree plantations is to maximize aboveground biomass production, the trade-off between C uptake in tree biomass and C loss in soil can cancel out the potential climate benefits of afforestation (Noormets et al. 2015).

The stabilization of SOC is interlinked with the nutritional supply and turnover, and plant-decomposer competition in the ecosystem (Averill et al. 2014; Noormets et al. 2021). The main mitigation effect of tree plantations on climate change is therefore associated with the maintenance of high productivity rates (Weih 2004; Tullus et al. 2013; Lutter et al. 2021a). Higher productivity rates are favored by the higher soil fertility and nutrient supply of former agricultural soils compared to native forestland soils

(Wall and Hytönen 2005; Nikodemus et al. 2013). However, the high production rates of tree plantations also mean higher demand for soil resources; in the long term, such intensive forest management systems could deplete soil nutrient pools, based on experience from regions where forest plantations have long been intensively managed (Fox et al. 2007; Berthrong et al. 2009). Although nutrient deficit can be compensated for with fertilization (Dimitriou and Mola-Yudego 2014; Sevel et al. 2014), before large-scale fertilization, a mechanistic understanding of organic material stabilization, nutrient longevity, and their internal cycling in tree plantations needs to be clarified. Fertilization is considered an environmental risk and an economic cost (Aronsson et al. 2014; Dimitriou and Mola-Yudego 2016), and according to current practices in Estonia, growth should be supported only by the natural supply and turnover of nutrients.

One of the first responses after the afforestation of agricultural soils is that trees shift the soil pH to an optimal range for nutrient acquisition (Lutter et al. 2016b; Rytter and Rytter 2020). With liming, the pH of former agricultural soils is usually kept in a range ensuring the optimal nutrition of agricultural crops, but afforestation will induce organic material (i.e., leaf and root) inputs and  $H^+$  root excretions that will decrease pH (Hinsinger et al. 2003; De Schrijver et al. 2012). Several studies have shown a significant decrease in soil pH after afforestation just 10–15 years after the planting (Jug et al. 1999; Ritter et al. 2003; Lutter et al. 2016b; Rytter and Rytter 2020). This decrease in pH is usually described in mineral soils in temperate and boreal Europe (Jug et al. 1999; Ritter et al. 2003; Lutter et al. 2016b; Rytter and Rytter 2020), but there is a lack of longer-term observations describing the stabilization and conditions of nutrient uptake in particular for birch plantations.

Nitrogen (N), phosphorus (P), and potassium (K) are considered the most important nutrients for plant growth, regulating the main physiological processes. Especially the availability of N and its decomposition rate (i.e., the C:N ratio), with the supply of available N forms being able to regulate the SOC stabilization through plant-decomposer interactions depending on site N supply (Cools et al. 2014; Dawud et al. 2016; Rahman et al. 2017; Högberg et al. 2021). The early NPK demand of plantations is low due to the small leaf area, but once the stand leaf area has peaked, the demand for available forms of NPK will increase to

maintain production (Landsberg and Waring 1997). Studies in northern Europe have shown that afforestation alters soil macronutrient (i.e., NPK) status in varied ways: Rytter and Rytter (2020) found decreased N and P concentrations and an increased K concentration with different deciduous species in the top 30 cm of soil after the first decade since plantation establishment. Lutter et al. (2016b) found the opposite, i.e., increased N concentration and unchanged P and K concentrations in hybrid aspen plantations after 15 years since plantation establishment.

Former agricultural soils are usually described as possessing homogeneous soil chemical properties due to soil mixing to the depth of ploughing (Vares et al. 2001). Compared with annual plants in agriculture, trees can extend their roots to deeper soil horizons. The roots of boreal trees are mainly located in the upper 30 cm of soil (Kalliokoski et al. 2010), where most available nutrients are located and acquired (Lutter et al. 2016b, 2021b). The ability of trees to access deeper soil layers than can agricultural crops can improve the soil's internal nutrient cycling and translocation to upper soil layers via leaf litter (Stark et al. 2015; Lutter et al. 2016b; Rytter and Rytter 2020). However, internal nutrient cycling dynamics in tree plantations after the afforestation of former agricultural soils have rarely been described in Northern Europe.

The aim of this study was to describe the first 20 years of changes in soil organic carbon (SOC) and nutritional status (i.e., pH, N, P, K, and C:N ratio) based on resampling soil in silver birch (*Betula pendula* Roth) plantations on former agricultural soils of hemiboreal Estonia. Silver birch is one of the most promising tree species for plantation forestry in Northern Europe, with a rotation cycle of about 30–35 years on former agricultural soils (Tullus et al. 2012). Repeated soil sampling after 20 years is rare (Bárcena et al. 2014b; Lutter et al. 2016a, b; Rytter and Rytter 2020) but it helps us describe long-term soil processes without having to meet the assumptions associated with space-for-time substitution used in chronosequence studies (e.g. Uri et al. 2012).

We hypothesize that 20 years after the afforestation of former agricultural land: (1) silver birch plantations have not depleted soil macronutrient (i.e., NPK) and SOC concentrations; (2) plantation productivity (using mean annual increment of stemwood as its proxy) is the main driver describing the dynamics of SOC and nutritional status; and (3) birch plantations

have redistributed SOC and nutrients over time from deeper (10–30-cm) soil layers to the upper 0–10-cm layer and the legacy of homogeneous soil conditions in the vertical soil profile of the former agricultural land use has disappeared.

## Materials and methods

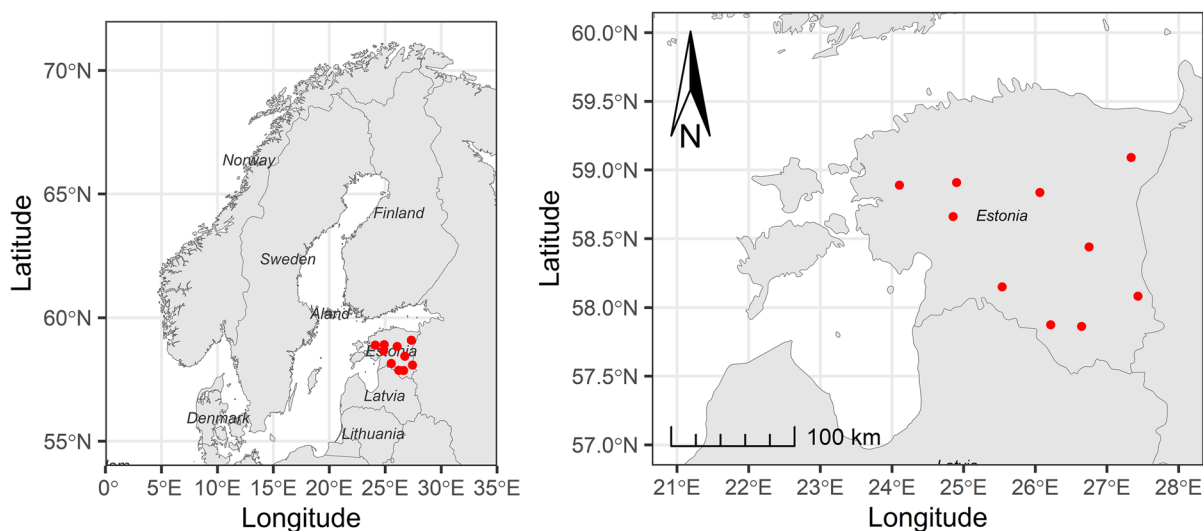
### Study sites

The study was conducted in hemiboreal Estonia in 10 silver birch plantations (Fig. 1). The long-term mean annual temperature (1999–2019) was 6.3°C and the mean annual precipitation was 706 mm according to the weather stations closest to the plantations (The Estonian Weather Service 2022).

The studied silver birch plantations were planted in former agricultural lands in the spring 1999. The previous land use was crop field ( $n=5$ ) or grassland ( $n=5$ ) (Lutter et al. 2015). The agricultural activity had ceased at least five years before the tree planting and the lands were in fallow. All the soils of the studied plantations are characterized as mineral soils of normal development (i.e., no organic soils or intensive drainage). According to the site type classification of forest soils, the plantations belong to fertile sites of *Hepatica*, *Oxalis* and *Oxalis-Myrtillus*, i.e. the most common forest site types for

former agricultural soils after afforestation (Löhmus 1974). Those forest site types are characterized as mineral soils with a well-drained automorphic water regime and C:N ratio between 10 and 20 that supports a fast decomposition of soil organic matter for nutrient release (Löhmus 1974; Tullus et al. 2010). Each plantation was 1 ha in size. Before planting, the soil was prepared by whole-area ploughing in all study sites, except in the Läänemaa, Tartumaa and Valgamaa plantations, where ploughing was done only for the tree rows (Vares et al. 2001). All plantations were established using 1-year-old bare-root seedlings with a planting density of 2500–3300 trees per ha (Vares et al. 2001; Kund et al. 2010). After the first years of planting, competing ground vegetation was removed to improve the survival and early growth in all plantations. All the plantation sites were fenced to avoid game browsing.

The last productivity estimations were conducted at the end of the 2019 growing season. The total stand production (Table 1) as mean annual increment (MAI,  $\text{m}^3 \text{ha}^{-1} \text{yr}^{-1}$ ) was estimated by summing the volume of retained trees using their individual tree parameters (height and diameter at breast height) from a 0.1-ha sample plot in the center of each plantation according to methods described by Lutter et al. (2015). Thinning from below was done at around the age of 15 years to remove suppressed and dead trees. The combination of artificial thinning and self-thinning resulted in the



**Fig. 1** Locations of the studied silver birch plantations

**Table 1** Soil (0–30-cm layer) and stand characteristics (mean ± SE) of the studied silver birch plantations ( $n = 10$ ) after the 21<sup>st</sup> growing season

Plantation	Soil type (WRB 2014)	Forest site type	Soil texture	Density, trees ha <sup>-1</sup>	<sup>b</sup> MAI, m <sup>3</sup> ha <sup>-1</sup> yr <sup>-1</sup>	<sup>c</sup> AWC <sub>30cm</sub> , mm	pH <sub>KCl</sub>	N, %	P, mg kg <sup>-1</sup>	K, mg kg <sup>-1</sup>	SOM, %	C:N ratio
Ida-Virumaa	<i>Endogleyic Umbri-sol</i>	<i>Oxalis-Myrtilus</i>	Silt loam	1580	9.4	63	4.5 ± 0.07	0.11 ± 0.018	23 ± 1.0	81 ± 20.1	2.5 ± 0.47	13.0 ± 0.30
Järvamaa	<i>Endocalcaric Luvisol</i>	<i>Oxalis</i>	Sandy loam	910	7.2	80	5.5 ± 0.15	0.15 ± 0.001	18 ± 1.6	76 ± 16.6	2.9 ± 0.10	11.3 ± 0.26
Läänemaa	<i>Gleyic Albic Podzol</i>	<i>Oxalis-Myrtilus</i>	Sand	950	13.6	77	4.0 ± 0.15	0.12 ± 0.037	21 ± 5.4	35 ± 10.8	4.3 ± 1.60	20.3 ± 2.59
Põlvamaa	<i>Stagni Fragic Retisol</i>	<i>Oxalis</i>	Sandy loam	1250	10.6	74	5.2 ± 0.12	0.10 ± 0.014	47 ± 4.4	85 ± 18.5	2.4 ± 0.34	13.8 ± 0.06
Pärnumaa	<i>Mollic Gleysol</i>	<i>Oxalis-Myrtilus</i>	Sandy loam	1860	11.0	86	6.8 ± 0.10	0.27 ± 0.014	324 ± 8.3	186 ± 63.7	5.1 ± 0.62	11.1 ± 0.76
Raplamaa	<i>Endocalcaric Cambisol</i>	<i>Oxalis</i>	Sandy loam	1760	11.5	86	6.4 ± 0.06	0.18 ± 0.026	80 ± 3.0	189 ± 21.9	3.5 ± 0.59	11.1 ± 0.43
Tartumaa	<i>Dysstri Glossi Fragic Retisol</i>	<i>Oxalis</i>	Sandy loam	1090	8.1	67	4.3 ± 0.03	0.12 ± 0.016	35 ± 1.3	96 ± 11.4	2.6 ± 0.48	12.2 ± 0.79
Valgamaa	<i>Dysstri Glossi Fragic Retisol</i>	<i>Oxalis</i>	Loam	1440	8.1	66	5.4 ± 0.09	0.14 ± 0.029	20 ± 0.9	55 ± 5.4	3.0 ± 0.66	12.2 ± 0.44
Viljandimaa	<i>Dysstri Glossic Retisol</i>	<i>Oxalis-Myrtilus</i>	Loam	1260	10.7	91	5.4 ± 0.09	0.14 ± 0.012	22 ± 2.7	100 ± 15.7	3.4 ± 0.44	14.2 ± 0.57
Võrumaa	<i>Eutric Regosol</i>	<i>Hepatica</i>	Sandy loam	1590	9.7	86	5.9 ± 0.09	0.11 ± 0.013	52 ± 6.8	163 ± 27.8	2.2 ± 0.34	12.2 ± 0.52

<sup>a</sup>Plantation name refers to the county in Estonia where the plantation is situated

<sup>b</sup>Mean annual increment based on the standing stem volume at the age of 20 years

<sup>c</sup>Available water content in 0–30-cm soil layer

removal of 0.1–2.6 m<sup>3</sup> ha<sup>-1</sup> yr<sup>-1</sup> of stemwood by the end of the 21st growing season.

### Soil sampling and analyses

As part of this study, we resampled soil in experimental plots (one plot per plantation with a size of 0.1 ha, except Läänemaa plantation with a plot size of 0.08 ha) for 20 years after the establishment of silver birch plantations on former agricultural lands. The first soil sampling was conducted in spring 2000, i.e., after the first growing season (Vares et al. 2001). The last sampling was conducted in late fall 2019, i.e., 20 years after the first monitoring using the same field sampling principles and laboratory analysis methods (Vares et al. 2001). A 1-m-deep soil pit about 75 cm wide was dug in the center of the experimental plot in each plantation, where the soil type was determined (Vares et al. 2001) and updated according to the most recent WRB (IUSS Working Group 2014) classification (Table 1). In 2019, the new soil pit was located about 2 m from the center to avoid overlap with the old soil pit from 2000. From the same pit, the top 30 cm (without the litter layer) of mineral soil was divided into 10 cm layers encompassing depths of 0–10-cm, 10–20-cm, and 20–30 cm. As former agricultural soils have previously been influenced by intensive ploughing and fertilization and the variation of soil nutritional status over a plantation is low (Lutter et al. 2016b), at each experimental plot we collected one composite soil sample from each vertical soil layer. Composite samples consisted of 3 subsamples taken across the given soil layer, which was revealed by the soil pit to repeat similar sampling protocol in both sampling occasions.

Soil pH<sub>KCl</sub> was determined from a 1 M KCl suspension at a ratio of 10 g : 25 mL. The concentration of total nitrogen (N, %) was determined using the Kjeldahl method (Tecator ASN 3313 AOAC). The concentration of available ammonium-lactate-extractable phosphorus (P, mg kg<sup>-1</sup>) was determined using the flow injection method (FiaStar<sub>5000</sub>). The concentration of available potassium (K, mg kg<sup>-1</sup>) was determined from the same solution using the flame photometric method. The concentration of soil organic matter (SOM, %) was determined using the loss-on-ignition method at 360°C. The SOM was converted to soil organic carbon (SOC, %) using the standard coefficient of

0.58 (Pribyl 2010). The C:N ratio was calculated according to both elements' concentrations to describe the changes in soil mineralization conditions. All the soil chemical analyses were performed on both sampling occasions (i.e., 2000 and 2019) in the Laboratory of Plant Biochemistry at the Estonian University of Life Sciences.

Sampling for the determination of soil physical properties conducted in 2013 when the plantations were 15-years-old (Lutter et al. 2015). The physical properties of soil were stable and no major changes were expected 5 years later i.e. at the time of this study. Soil physical properties, i.e., bulk density, soil-specific surface area, and soil texture (i.e., the proportions of sand, clay, and silt) were aggregated into the soil-available water content (AWC, mm). AWC is a function of soil bulk density, specific surface area, and texture, from which the proportion of gravel is subtracted to characterize the water-holding capacity of soil (Lutter et al. 2015), i.e., the difference between the soil water content at field capacity and at permanent wilting point. This study uses AWC in the top-30-cm soil layer as a generalized variable (to avoid the potential intercorrelations of soil physical properties in the model) of soil physical properties (see “Statistical analyses”) to describe the 20-year changes in soil nutritional status.

### Statistical analyses

The normality of the soil and stand variables were tested using the Shapiro-Wilk test before analysis. To test the 20-year changes in soil chemical parameters, pairwise comparisons of SOC, N, P, K, pH, and C:N ratio at three sampling depths (i.e., 0–10-cm, 10–20-cm, and 20–30-cm) and the total for the 0–30-cm soil layer were conducted using the paired Student's *t*-test or non-parametric Wilcoxon signed-rank test in the case of non-normal distribution. The effect of vertical soil depth (i.e., 0–10-cm, 10–20-cm, and 20–30-cm) on SOC, N, P, K, pH, and C:N ratio was tested on two sampling occasions using the linear mixed model (plot as random factor; depth and sampling year as fixed factors) of repeated measurements with the R package “*lme4*” (Bates et al. 2015). Comparison of means by depth was conducted using the R package “*emmeans*” post hoc analysis (Lenth 2021).

The effects of stand growth and soil properties (physical and chemical) on nutritional changes (SOC, N, P, K, pH, and C:N ratio) were analyzed using a general linear model (Eq. 1) with the R command “*lm*” and the significance of the factors were tested with Type III ANOVA (R package “*car*”) (Fox and Weisberg 2019). We applied backward stepwise selection of explanatory variables (R package “*MASS*” and command “*stepAIC*”) (Venables and Ripley 2002) to obtain the final models. The explanatory variables for the 20-year change ( $\Delta$ ) were chosen to describe tree layer (i.e., mean annual increment [MAI,  $\text{m}^3 \text{ha}^{-1} \text{yr}^{-1}$ ] of stem wood), soil physical properties (i.e., available water content [AWC, mm] of the 0–30-cm soil layer), initial level of the studied variable in year 2000, and the initial level of soil acidity (soil pH is a key driver determining nutrient mobility and uptake). All factors were standardized to compare the magnitude of the effects (model coefficients). The normality of the model residual distribution was tested using the Shapiro-Wilk test and checked visually from Q-Q plots. To check the potential multicollinearity of the independent variables, the model variance inflation factors (VIFs) were calculated using the R command “*vif*” (R package “*car*”). In all cases, the VIF values were below 2, indicating a low level of multicollinearity.

$$\Delta \text{SCC} = \mu + \beta_1 \times \text{SCC}_i + \beta_2 \times \text{AWC}_{30\text{cm}} + \beta_3 \times \text{pH}_i + \beta_4 \times \text{MAI} + \varepsilon \quad (1)$$

where  $\Delta \text{SCC}$  is the change in the given soil chemical characteristic after 20 growing seasons (monitoring in autumn 2019 – monitoring in spring 2000) at the given soil depth,  $\mu$  is the overall mean,  $\beta_1$  to  $\beta_4$  are the coefficients for the fixed effects,  $\text{SCC}_i$  is the initial value of the given soil chemical characteristic in spring 2000 at the given soil depth,  $\text{AWC}_{30\text{cm}}$  (mm) is the soil-available water content in the 0–30-cm soil layer,  $\text{pH}_i$  is the initial value of the given soil pH in spring 2000 at the given soil depth,  $\text{MAI}$  ( $\text{m}^3 \text{ha}^{-1} \text{yr}^{-1}$ ) is the mean annual increment of stand after the soil monitoring in 2019 autumn), and  $\varepsilon$  is an error term.

The level of significance of  $\alpha = 0.05$  was used to reject the null hypothesis after statistical tests. The mean values are presented with standard errors. All

statistical analyses were conducted using the R Statistics software (R Core Team 2021).

## Results

### Twenty-year changes in SOC and nutrients

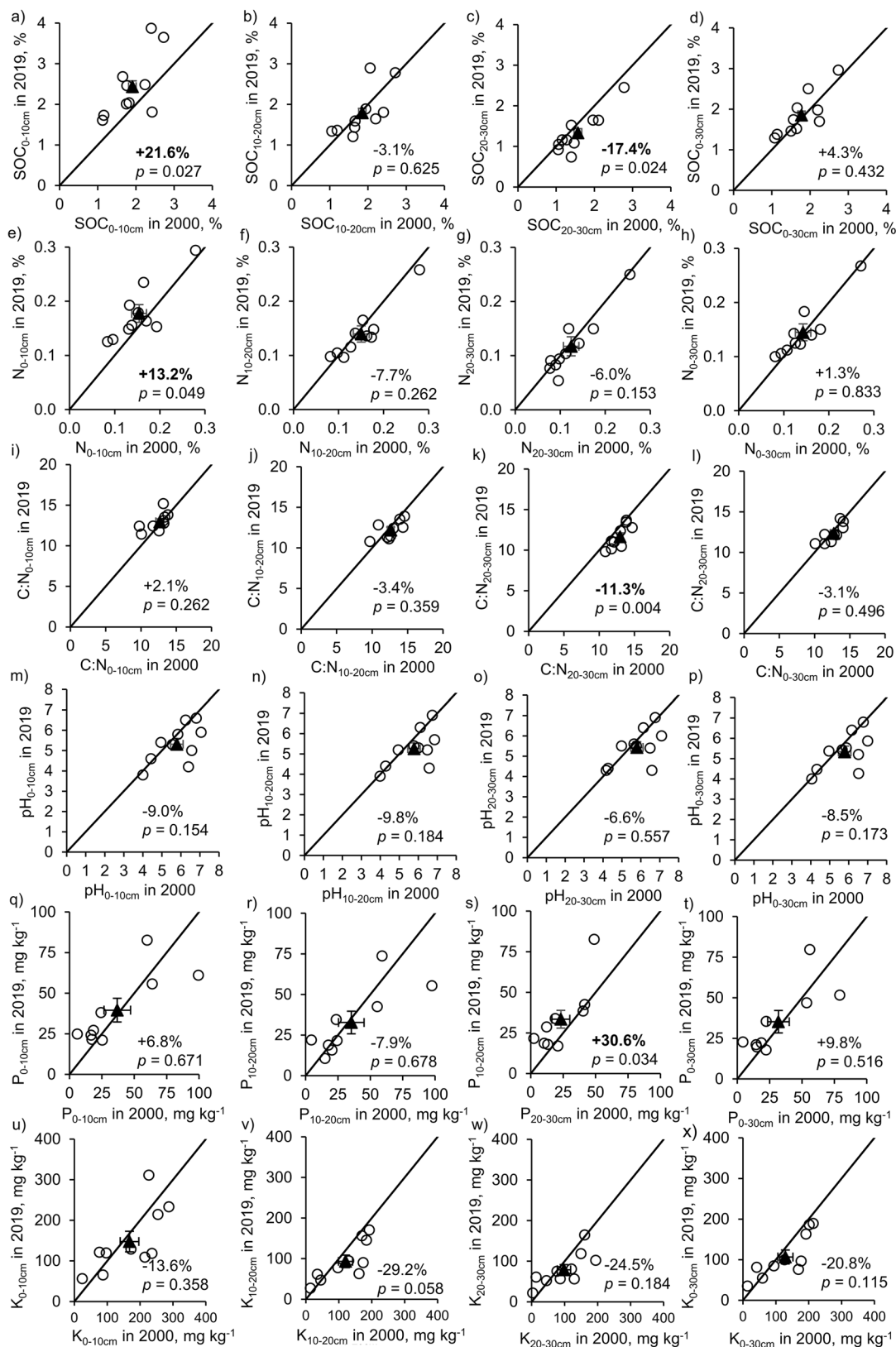
The SOC concentration increased in the 0–10-cm soil layer by 21.6% ( $p=0.027$ ) but simultaneously decreased in the 20–30-cm soil layer by 17.4% ( $p=0.024$ ). These alterations cancelled each other out when describing the whole 0–30-cm soil profile, so the SOC concentration remained unchanged over the 0–30-cm depth (Fig. 2). Twenty years after the establishment of birch plantations, the SOC concentration decreased significantly with soil depth (Fig. 3a), with concentrations of  $2.43 \pm 0.25\%$  in the 0–10-cm layer,  $1.79 \pm 0.19\%$  in the 10–20-cm layer, and  $1.34 \pm 0.16\%$  in the 20–30-cm layer. In 2000, the SOC concentration was homogenous across the vertical soil layers (Fig. 3a).

Similarly to SOC, the total N concentration increased by 13.2% ( $p=0.049$ ) in the 0–10-cm soil layer (Fig. 2e). However, the total N concentration did not change in the 20–30-cm layer as was observed for SOC. In 2000, the total N content was similar in the 0–10-cm and 10–20-cm layers (Fig. 3b), but stratification of N concentration with depth was observed after 20 years, as the 0–10-cm layer had a higher total N concentration that did the deeper layers:  $0.18 \pm 0.02\%$  in the 0–10-cm layer,  $0.14 \pm 0.01\%$  in the 10–20-cm layer, and  $0.12 \pm 0.02\%$  in the 20–30-cm layer (Fig. 3b). No change in total N concentration was apparent over the whole studied 0–30-cm soil profile.

The C:N ratio decreased significantly by 11.3% ( $p=0.004$ ) at a depth of 20–30-cm (Fig. 2k). The repeated-measurement model revealed stratification of the C:N ratio between the 0–10-cm and 20–30-cm layers over 20 years (Fig. 3c).

We found no significant change in soil pH levels in all studied vertical depths layers (Fig. 2m-p), even though the mean pH decrease was about 0.4–0.5 units. The soil depth also insignificantly affected pH level on both the 2000 and 2019 sampling occasions (Fig. 3d).

Plant-available P concentration increased by 30.6% ( $p=0.042$ ) at the depth of 20–30-cm (Fig. 2s). The other studied depths did not show





◀**Fig. 2** Twenty-year changes in SOC (%), N (%), C:N ratio, pH, P ( $\text{mg kg}^{-1}$ ), and K ( $\text{mg kg}^{-1}$ ) at different soil depths in the studied plantations (open circles). The filled triangles show the mean values with standard errors; significant changes are indicated in bold

any change and the overall change of plant-available P in 0–30-cm remained unchanged. The vertical soil depth also insignificantly affected P concentration on both the 2000 and 2019 sampling occasions (Fig. 3e).

Plant-available K was the only element that did not display any significant changes over the 20-year period after afforestation (Fig. 2u and x). The vertical differences in plant-available K concentration remained unchanged over the 20-year period (Fig. 3f), with the concentration in the 0–10-cm layer being higher than in the two deeper layers on both sampling occasions.

#### Effect of environmental variables on soil chemical changes

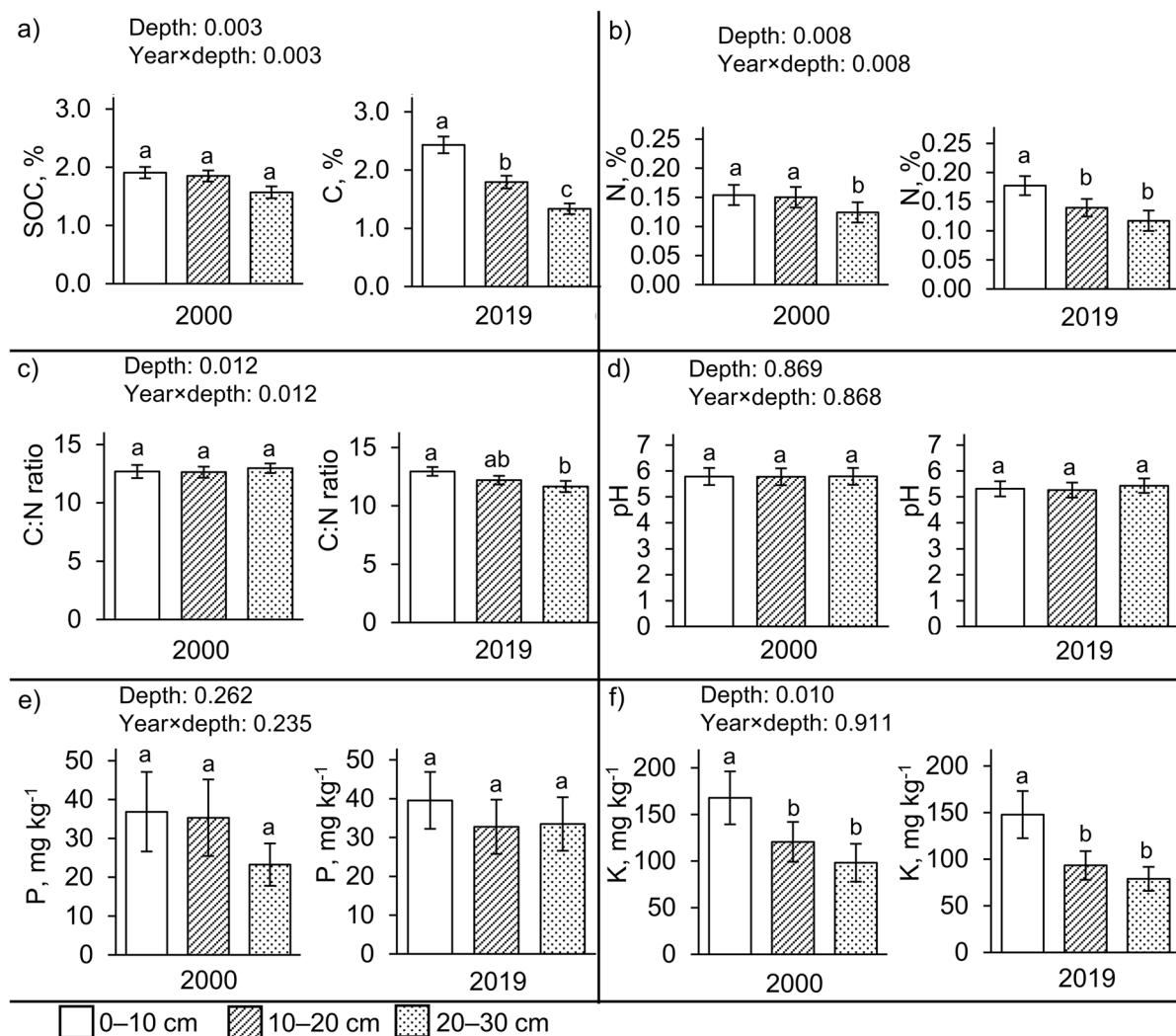
The stepwise regression model describing the changes in soil chemical variables found that the MAI of stemwood was the most important variable describing the SOC concentration increase in the 0–10-cm layer (Table 2). The MAI of stemwood also correlated with the SOC concentration change in the 10–20 cm layer and in the 0–30 cm layer, i.e. SOC increased in sites with a higher productivity of trees. Although, we observed no general unidirectional trend in the change of the mean P concentration in the 0–10-cm layer and the 10–20-cm layer, a significant correlation was found between the change and the initial P concentration (i.e., decreasing where it was initially higher) (Table 2). Even though we observed no changes in the mean K concentration, its change was significantly correlated with its initial level 20 years previously in the 10–20-cm, 20–30-cm, and 0–30-cm layers (Table 2). The mean change in soil pH was insignificant but was correlated with the initial level (i.e., decreasing where it was initially higher) in the 10–20-cm, 20–30-cm, and 0–30-cm layers (Table 2). The C:N ratio change in the 0–30-cm layer was correlated with AWC in 30 cm of soil.

## Discussion

We found that the afforestation of former agricultural soils with silver birch plantations retained the soil primary nutrients and SOC concentrations in the upper 0–30 cm of soil at the same level over 20 years of plantation. However, significant vertical redistributions of SOC and nutrients were observed and the SOC increase was mainly driven by the stand productivity.

We can confirm the first hypothesis, as no changes were found in SOC and N concentrations 20 years after planting in the 0–30-cm soil layer. Such an outcome is in good agreement with previous results for hybrid aspen plantations growing in similar soil types 15 years after afforestation, also studied with repeated soil sampling (Lutter et al. 2016a, b). A similar case study using repeated sampling in Northern Europe showed decreases in SOC and N concentration after agricultural land afforestation over the first 10 years of silver birch growth in the 0–30-cm soil layer over a wide latitude range in Sweden (Rytter and Rytter 2020). Also, Alberti et al. (2017) and Steffens et al. (2021) reported similar outcomes in northern temperate conditions for different tree species around 10 years after afforestation.

Our finding of no change in SOC concentrations in the 0–30-cm layer might indicate that there was no early SOC loss or that SOC had recovered by the stand age of 21 years. If the case is one of recovery after 20 years, then this is earlier than found by some other studies in the region, in which SOC levels showed recovery in soils planted with spruce and oak 30 years after afforestation (Vesterdal et al. 2002; Bárcena et al. 2014b). However, the potential of fast-growing tree plantations to mitigate climate change by increasing soil C stocks seems to be challenging to realize (Bárcena et al. 2014a; Noormets et al. 2015; Lutter et al. 2016a; Rytter and Rytter 2020), as the internal nutrient turnover and fast decomposition of organic material supports C accumulation in biomass rather than in soil as the projected rotation cycle is just 35 years (Tullus et al. 2012). SOC stabilization and accumulation in mineral soil are controlled by different factors, such as soil microbial and enzymatic activities, nutrient supply, soil buffering capacity, and litter quality (i.e., tree species) (Guo and Gifford 2002; Fontaine et al. 2004; Vesterdal et al. 2012; Bárcena et al. 2014a; Noormets et al. 2015; Alberti et al. 2017), but mainly by



**Fig. 3** Effect of vertical soil horizon on the levels of soil SOC (%), N (%), C:N ratio, pH, P (mg kg<sup>-1</sup>), and K (mg kg<sup>-1</sup>). Letters indicate significant differences between vertical depths; error bars indicate standard errors

plant-decomposer competition for nutrients, in particular for nitrogen (Averill et al. 2014).

During the previous agricultural land use, ploughing to a depth of 30 cm had homogenized the soil chemical properties (Vares et al. 2001). The third hypothesis, regarding SOC, was confirmed as we found the translocation of SOC within the 0–30-cm soil profile, as the SOC gains in the 0–10-cm layer were offset by decreases in the deeper layer, 20–30-cm layer. In total, SOC increased by 20% in the 0–10-cm layer and decreased by 17% in the 20–30-cm layer. A similar pattern observed for total N concentration. Such stratification of the vertical soil profile

for SOC and N reflects the ability of trees to access deeper than 20 cm depth soil layers than can annual agricultural crops; trees can change biogeochemical and hydrological plant nutrition characteristics, with enhanced soil microbial activity being able to improve the mobility of nutrients for tree uptake (Mobley et al. 2015; Rytter and Rytter 2020). The decrease in SOC in the 20–30-cm soil layer indicates increased decomposition rates, promoting nutrient mineralization and active uptake (Mobley et al. 2015; Dawud et al. 2016; Rahman et al. 2017), as also confirmed by the decreased C:N ratio found at this depth in the present study. The internal cycling of SOC and

**Table 2** Outcome of the stepwise regression model describing the effects of the main environmental variables on the 20-year changes ( $\Delta$ ) in chemical properties at different vertical soil depths

$\Delta$ Soil variable, 20 years	MAI <sup>a</sup> m <sup>3</sup> ha <sup>-1</sup> yr <sup>-1</sup>		SCCi		AWC <sub>30cm</sub> mm		pH <sub>i</sub>		Model	
	$\beta_1$	<i>p</i>	$\beta_2$	<i>p</i>	$\beta_3$	<i>p</i>	$\beta_4$	<i>p</i>	adj. R <sup>2</sup>	<i>p</i>
$\Delta$ SOC <sub>0–10–cm</sub>	<b>1.48</b>	<b>0.003</b>			-0.31	0.224			0.67	0.008
$\Delta$ SOC <sub>10–20–cm</sub>	<b>1.20</b>	<b>0.018</b>	-0.39	0.150					0.49	0.040
$\Delta$ SOC <sub>20–30–cm</sub>			-0.45	0.195					0.01	0.195
$\Delta$ SOC <sub>0–30–cm</sub>	<b>1.70</b>	<b>0.006</b>			-0.64	0.093	0.39	0.201	0.62	0.032
$\Delta$ N <sub>0–10–cm</sub>	1.16	0.105			-0.84	0.124	0.49	0.259	0.13	0.317
$\Delta$ N <sub>10–20–cm</sub>	0.68	0.274	-0.49	0.164	-0.52	0.308	0.80	0.077	0.37	0.190
$\Delta$ N <sub>20–30–cm</sub>							0.18	0.622	-0.09	0.622
$\Delta$ N <sub>0–30–cm</sub>	0.96	0.197			-0.76	0.190	0.68	0.170	0.02	0.436
$\Delta$ C:N <sub>0–10–cm</sub>	1.18	0.057			0.40	0.181			0.66	0.017
$\Delta$ C:N <sub>10–20–cm</sub>			-0.48	0.097			0.55	0.128	0.58	0.031
$\Delta$ C:N <sub>20–30–cm</sub>					0.45	0.249			0.07	0.249
$\Delta$ C:N <sub>0–30–cm</sub>					<b>0.78</b>	<b>0.022</b>			0.49	0.021
$\Delta$ P <sub>0–10–cm</sub>			<b>-0.71</b>	<b>0.033</b>					0.42	0.033
$\Delta$ P <sub>10–20–cm</sub>			<b>-0.71</b>	<b>0.033</b>					0.43	0.033
$\Delta$ P <sub>20–30–cm</sub>	0.64	0.259							0.06	0.259
$\Delta$ P <sub>0–30–cm</sub>			-0.60	0.117					0.22	0.117
$\Delta$ K <sub>0–10–cm</sub>	-0.75	0.139	-0.38	0.223					0.32	0.110
$\Delta$ K <sub>10–20–cm</sub>	0.61	0.143	<b>-0.62</b>	<b>0.033</b>					0.53	0.029
$\Delta$ K <sub>20–30–cm</sub>			<b>-0.78</b>	<b>0.008</b>					0.56	0.008
$\Delta$ K <sub>0–30–cm</sub>	0.73	0.082	<b>-0.57</b>	<b>0.041</b>					0.56	0.024
$\Delta$ pH <sub>0–10–cm</sub>			-0.73	0.059	0.47	0.229			0.26	0.143
$\Delta$ pH <sub>10–20–cm</sub>			<b>-0.76</b>	<b>0.043</b>	0.53	0.161			0.32	0.105
$\Delta$ pH <sub>20–30–cm</sub>			<b>-0.82</b>	<b>0.022</b>	0.61	0.087			0.45	0.053
$\Delta$ pH <sub>0–30–cm</sub>			<b>-0.78</b>	<b>0.039</b>	0.59	0.126			0.35	0.093

The standardized coefficients ( $\beta$ ) of significant predictor variables are indicated in bold

<sup>a</sup>MAI is the mean annual increment of stand after the soil monitoring in 2019 autumn, SCCi is the initial value of the given soil chemical characteristic in spring 2000 at the given soil depth, AWC<sub>30cm</sub> is the soil-available water content in the 0–30-cm soil layer, pH<sub>i</sub> is the initial value of the given soil pH in spring 2000 at the given soil depth

N shows the saturation of these constituents in the upper at 0–10-cm layer in fresh leaf and root litter (Vesterdal et al. 2002; Bárcena et al. 2014a, b; Mobley et al. 2015). Root estimates show that up to 30% of fine root biomass is located in soil below a depth of 20 cm in middle-aged (i.e., 20–30-year-old) birch stands in fertile *Oxalis* site types (Aun et al. 2021), a share that could be sufficient to alter soil organic material status and the mobilization of nutrients. Moreover, SOC translocation within the vertical profile was more notable in sites with higher productivity, indicating that trees can acquire available nutrients from deeper soil layers and that high production rates provide more litter supply for the upper soil.

For the other studied soil variables, we found no major changes over 20 years, except in the concentration of available P in 20–30-cm. Similar study examining shorter periods (< 15 years) since agricultural land afforestation, showed no change (Lutter et al. 2016a) and even a decrease (Rytter and Rytter 2020) in available P. The increase of available P in the 20–30-cm layer matches well with the loss of SOC and decreased C:N ratio, and might describe its release from decomposition. Another explanation could be its increased bioavailability from older pools: former agricultural soils were amended with very large amounts of mineral P fertilizers during their previous land use in order to

provide available P for agricultural crops, as added P rapidly becomes unavailable for plant acquisition, resulting in long-term P accumulation (Kõlli et al. 2008; Rubæk et al. 2013; Azeez et al. 2020). The available P pool describes only a minor share of the total organic and inorganic P pool (Azeez et al. 2020), i.e., only 1–3% in the soils of Estonia (Kõlli et al. 2008). This P legacy from agriculture results in very cumulative inactive P pools, and trees along with microbial priming can enhance the supply of available forms of P from this pre-existing pool (Chen et al. 2008; Deng et al. 2017). As P weathering is a slow process occurring at depths > 30 cm, P uptake to recycling is rather unlikely (Jobbagy and Jackson 2001; Deng et al. 2017; Azeez et al. 2020). It has been shown that soils with lower P supply might have more efficient P recycling than sites with a surplus P (Lang et al. 2016). The initial level of available P was relatively low (23 mg kg<sup>-1</sup>, range 2–49 mg kg<sup>-1</sup>) in the 20–30-cm layer and the relative increase of 30% means a concentration of 34 mg kg<sup>-1</sup> (range 17–83 mg kg<sup>-1</sup>), which is still lower than in hybrid aspen plantations in the region (~50 mg kg<sup>-1</sup>) (Lutter et al. 2016b; Deng et al. 2017) meta-analysis identified a general decrease in available P after afforestation in boreal regions in the top 20 cm of soil; however, they found an increase in available P in soils with a pH less than 6, which is fairly comparable to our sites, with an average pH of 5.3 ± 0.3.

The concentration of available K remained unchanged in 0–30-cm and its levels are in the same range (107 mg kg<sup>-1</sup>, range 35–189 mg kg<sup>-1</sup>) as those found in other studies of fast-growing hybrid aspen plantations (Lutter et al. 2016b). Some studies reported a relatively high K accumulation in birch biomass (Alriksson and Eriksson 1998; Uri et al. 2007) and a reduced level in soil (Rosenvald et al. 2012), but our outcome is different from those findings. Rytter and Rytter (2020) found increased K content in plantations of birch and other fast-growing tree species in the upper 0–10-cm soil layer, but a decrease at greater depths, explaining this in terms of the redistribution of K with litter decomposition.

Previous studies of agricultural land afforestation report a rapid decrease in soil pH with deciduous species, as trees tend to improve their conditions for nutrient uptake (Lutter et al. 2016a; Rytter

and Rytter 2020). We found an insignificant 8.5% decline in soil pH; however, the decrease in soil pH in the upper 30-cm soil layer from 5.8 to 5.3 is well aligned with declines reported in other studies after afforestation and is in an optimal range for nutrient cycling (Lutter et al. 2016a; Rytter and Rytter 2020).

Most of the available nutrients are located in the upper 0–10-cm soil layer (Lutter et al. 2016b; Rytter and Rytter 2020), but our results raise the question of why trees are mobilizing deeper nutrient pools. We found a significant correlation with productivity only for SOC change in the 0–10-cm soil layer, partly confirming the second hypothesis, i.e., that higher production also means higher litter inputs to the upper soil. Previous studies in young fast-growing tree plantations have found that productivity is driven mainly by soil-available water content (Tullus et al. 2010; Lutter et al. 2015, 2017). The high transpiration demand of birches in their prime production period could force trees to access deeper soil layers for water uptake (Tullus et al. 2010; Lutter et al. 2015). This means that along with the water uptake stream from deeper soil layers, nutrients will be also transported via the transpiration-driven mass-flow and deeper soil colonization by roots and mycorrhizal fungi (McMurtree and Näsholm 2018; Henriksson et al. 2021).

As macronutrient levels remain unchanged or even increased relative to available P concentrations in the 20–30-cm layer, the supplemental fertilization of tree plantations might not have a significant effect in terms of improved production (Lutter et al. 2016b). Internal nutrient cycling and fast organic material turnover, coupled with the ability to translocate nutrients to upper soil layers, might be sufficient for the first 30-year rotation cycle in former agricultural soils. Some chronosequence studies show that the legacy effect of former agriculture on soil nutritional status can persist even after 80 years in comparison with native forestland (Wall and Heiskanen 2003). However, as forest plantations maintain high production rates through intensive management cycles, at some later growth stage or in the case of multiple repeated cycles, supplemental fertilization might be needed to maintain high productivity.

## Conclusion

Repeated soil monitoring after 20 years revealed no loss of SOC or NPK concentrations in the upper mineral soil layer (0–30 cm) in fast-growing silver birch plantations in former agricultural soils. However, a significant translocation was found within the vertical soil profile, with a decrease in SOC and N in the 20–30-cm layer being compensated for by an increase in the top 0–10-cm layer. The shifts of SOC were more notable in plantations with higher productivity. Silver birch plantations improved the mobility of available P forms and, moreover, did not reduce soil nutrient and SOC status after 20 years. We found that afforestation of former agricultural soils with silver birch plantations has caused significant shifts in soil organic material distribution in the 0–30-cm soil layer during the first 20 years since the establishment and fast-growing trees have activated deeper nutrient pools.

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**Authors' contributions** R.L, A.T, A.V and H.T designed the study. A.V, R.L and R.S performed field work and data collection. R.L and A.T analysed the data. The draft of the manuscript was written by all authors. All authors read and approved the final manuscript.

## Declarations

**Conflicts of interest/Competing interests** 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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