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Master Thesis

The Influence of Forest Site Preparation on Soil Functions of an Alluvial Forest in the Upper Rhine Valley, Vorarlberg

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Affidavit

I hereby declare that I have authored this master´s thesis independently, and that I have not used any assistance other than that which is permitted. The work contained herein is my own except where explicitly stated otherwise. All ideas taken in wording or in basic content from unpublished sources or from published literature are duly identified and cited, and the precise references included.

I further declare that this master thesis has not been submitted, in whole or in part, in the same or a similar form, to any other educational institution as part of the requirements for an academic degree.

I hereby confirm that I am familiar with the standards of Scientific Integrity and with the guidelines of Good Scientific Practice, and that this work fully complies with these standards and guidelines.

Vienna, 03.06.2022 Maximilian BEHRINGER

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Abstract

Alluvial forests in the temperate zone of Europe are frequently changing drastically in their hydrological regime, vegetation composition and structure, and disturbance dynamics. Causes are river regulations, historic land use, recent forest management, and introduced species such as *Solidago canadensis agg*. or the pathogenic fungus *Hymenoscyphus fraxineus* causing ash dieback. Climate change increases the scale of these changes. As a result, like in the present case study, premature stands dominated by *Pica abies* or *Fraxinus excelsior* have to be clearcut. In order to achieve a tree species composition which is adapted to the altered site conditions and still economically desirable, tree planting in a larger scale is inevitable. To control competing vegetation (e.g., *Solidago, Clematis, Rubus*…) site preparation (mulching and tilling of planting strips) was deemed to be necessary but is discussed controversially. Effects of site preparation on indicators for soil functions were compared for the two dominating soil types, Fluvisols and Rendzic Leptosols, using a chronosequence approach. The following key results were obtained: (1) Soil type has a significant effect upon most indicators. (2) Areas treated ≥5 years ago have significantly higher SOC stocks and a higher bulk density in the 20 cm topsoil. (3) Tilling strips have significantly lower SOC and N_{tot}-stocks (total N) compared to areas only mulched. (4) Effects of site preparation on C/N-ratio (-), ratio of microbial to organic carbon (+) and hydraulic conductivity estimated from pedotransfer functions (-) were mainly significant for Rendzic Leptosols. This may reflect the mobilization of accumulated forest floor, which was present in mature spruce stands on Rendzic Leptosols but not on Fluvisols. (5) The vegetation shows an expected response to clearing. Mulching effects could not be distinctly separated from clearing effects. (6) Nitrate concentrations in seepage are below drinking water standards and show no clear treatment effect, though highest values were found in declining spruce stands on Leptosols.

Kurzfassung

Auwälder der gemäßigten Zone Europas sind in einem starken Wandel begriffen. Ihr hydrologisches Regime, Artenzusammensetzung und Störungsdynamiken ändern sich. Hauptgründe sind Flussregulierungen, historische Landnutzung und aktuelle Forstwirtschaft sowie Neobiota wie *Solidago canadensis agg.* oder *Hymenoscyphus fraxineus*, der Verursacher des Eschentriebsterbens. Durch den Klimawandel wird das Ausmaß dieser Änderungen noch vergrößert. Das führt, wie in der gegenständlichen Fallstudie im oberen Rheintal, dazu, dass Waldbestände, mit *Picea abies* und *Fraxinus excelsior* als Hauptbaumarten vor Erreichen der Umtriebszeit geschlägert werden müssen. Um eine wirtschaftlich und standörtlich sinnvolle Baumartenzusammensetzung zu erreichen, muss nun im großen Stil aufgeforstet werden. Da konkurrenzstarke Begleitvegetation vorhanden ist (z. B. *Solidago, Clematis, Rubus…*) wurde der Boden vor der Pflanzung bearbeitet; eine Maßnahme die kritisch diskutiert wird. Der Effekt dieser Bodenbearbeitung auf Indikatoren für Bodenfunktionen wird für die beiden Hauptbodentypen, skelettreiche und skelettarme carbonathaltige Auböden, mittels Chronosequenz untersucht. Die Ergebnisse sind wie folgt: (1) Der Bodentyp beeinflusst die meisten Indikatoren signifikant. (2) Flächen die vor ≥5 Jahren aufgeforstet wurden haben höhere SOC und N Vorräte sowie eine höhere Lagerungsdichte. (3) Gefräste Pflanzstreifen haben niedrigere SOC und N Vorräte als nur gemulchte Flächen. (4) Bodenbearbeitung beeinflusste das C/N Verhältnis (-), das Verhältnis von mikrobiellem zu organischem Kohlenstoff (+) und die gesättigte hydraulische Leitfähigkeit (-), geschätzt mittels Pedotransferfunktion, hauptsächlich auf skelettreichen carbonathaltigen Auböden. Das könnte durch die geänderte Humusdynamik begründet sein, die auf diesen Standorten eintrat. (5) Die Vegetation zeigt den erwartbaren Kahlschlageffekt. Bodenbearbeitungseffekte konnten nicht eindeutig davon getrennt werden. (6) Nitratkonzentrationen im Bodenwasser bleiben deutlich unter Trinkwassergrenzwerten. Sie zeigen nach 5 Jahren keinen Einfluss der Bodenbehandlung mehr, wobei die höchsten Werte in einem gestörten Fichtenbestand gemessen wurden.

1. Introduction

1.1. Background and definitions

The initial goal of this work is to provide a brief overview on definitions of soil functions and site preparation. This shall prevent misconceptions and misunderstandings in the further chapters. It also includes a short description of the type of site preparation that was performed in the sampling area. Further information on indicators for soil functions will be provided in section 1.3.

Soil formation is a slow process and soil can be considered non-renewable. Thus, a sustainable use of soil and the adequate protection of soil functions are crucial (European Commission, 2006; Schjønning, et al., 2015). The soil function concept started emerging in the European soil science community during the 1970´s (Glenk et al., 2012; Schjønning, et al., 2015). Like the ecosystem services concept, soil functions are defined by an anthropocentric point of view (Glenk et al., 2012). Soil functions are linked to ecosystem services (Glenk et al., 2012; Schjønning, et al., 2015). Glenk et al. (2012, p. 35) summarizes that "soil functions should be viewed as (bundles of) soil processes that are providing input into the delivery of (valued) final ecosystem services". In literature many lists of soil functions exist and often soil functions, soil ecosystem services and soil roles are used interchangeably (Schjønning, et al., 2015). Another closely related term is soil quality (Drobnik et al., 2018) which often has an emphasis on productivity (e.g., in Burger & Kelting, 1999). This can be problematic as different ideas of soil function can lead to misunderstandings. Soil scientists and ecologists often associate processes with the term function, while social scientists, and policy makers rather think of services (Glenk et al., 2012). Hence, clear definitions are essential for communication, especially in transdisciplinary constellations. In this work the soil function definition coined by Blum (1993) will be used. He distinguishes between three ecological soil functions and three non-ecological soil functions. Ecological soil functions are (1) biomass production, (2) filtering, buffering and transforming capacity (regulatory function) and (3) biological habitat and gene reserve. Non-ecological soil functions include (1) physical medium (e.g., for housing construction), (2) source of raw materials and (3) cultural heritage (Blum, 1993). Nevertheless, ecological and non-ecological functions cannot be viewed completely isolated as non-ecological functions depend on processes related to ecological functions (Glenk et al., 2012). In this work a focus on ecological functions is kept. In a later version Blum slightly adapts some of the functions and explores them in greater detail. Especially in the second ecological soil function, the role of soil for protection of humans and environment is stressed and the function of soil in the carbon cycle is explicitly mentioned (Blum, 2005). These ideas also transformed into the proposal of the EU Soil Framework Directive 2006 (Glenk et al., 2012) and is reflected by the definition of soil functions in the Austrian Advisory Committee for Soil Fertility and Conservation (Fachbeirat für Bodenfruchtbarkeit und Bodenschutz, 2013).

Site preparation includes a variety of well-established physical and chemical methods (Williams & Harrington, 2012). More recently, biological control methods using invertebrate or pathogenic organisms to limit the growth of competing vegetation are being researched (McCarthy et al., 2011). Despite their easy application, chemical methods are rarely used in Europe (Willoughby et al., 2009). This is due to low public acceptance of chemical herbicides (Löf et al., 2012) which is manifested in regulations and in independent forest certification schemes such as FSC or PEFC (Willoughby et al., 2009). Mechanical site preparation has a long tradition and is once again becoming increasingly popular, as it presents a viable alternative to chemical methods (Löf et al., 2012). Thus, this work will focus on mechanical site preparation (if not stated otherwise *site preparation* refers to mechanical site preparation). Like other methods, mechanical site preparation facilitates the establishment of the desired tree species (Williams & Harrington, 2012). The growth of natural regeneration, planted seedlings and direct seeding is improved as competing species are removed and site conditions can

be ameliorated (Löf et al., 2012, 2016; Mayer et al., 2020; Williams & Harrington, 2012). For mechanical site preparation, the soil is treated in varying intensity (Löf et al., 2012, 2016; Williams & Harrington, 2012). Often mechanical site preparation is applied in large scales e.g., for reclamation of mine sites (Löf et al., 2012) or natural disasters (Williams & Harrington, 2012). It is also used in clear-cuts (Löf et al., 2012) or in small scale for other silvicultural purposes such as seeding fir (*Abies alba*) underneath a spruce (*Picea abies*) canopy to promote more climate resilient regrowth (Engeßer et al., 2011). It can be applied extensively over the whole area, systematic (e.g., in strips) or selective on suitable spots (Löf et al., 2012). At the sampling area in Vorarlberg, the whole area was mulched to remove competing vegetation and crush harvest residuals. After that, planting strips were tilled with a rotary tiller. Strips were tilled every 2 m and had a width of 30 cm. The depth varied depending on site conditions between 10 and 30 cm. This is referred to as *treatment* in this work. Sites were mowed or mulched in the first years to control competing vegetation when necessary. This way of site preparation was started in 2015. Before that logging residues and remaining, unwanted vegetation were concentrated in strips approximately every 20 m.

1.2. Rational of the study

Alluvial forests in the temperate zone of Europe are frequently changing drastically in their hydrological regime, vegetation composition and structure, and disturbance dynamics (Klimo et al., 2008). Causes are river regulations, historic land use, recent forest management (Klimo et al., 2008), and introduced species such as *Solidago canadensis agg*. (Schnitzler et al., 2007) or the pathogenic fungus *Hymenoscyphus fraxineus* causing ash dieback (Needham et al., 2016). Climate change increases the scale of these changes. Following legal definitions, forests in Austria have to fulfill a production, recreational and conservation function (Forstgesetz, 1975, §1). The closeness of the sampling area to Feldkirch, the existence of two Natura2000 protection areas and the good productivity of the soil show the importance of all three legally defined forest functions in the area. Management is challenging, and one must take these aspects into account and find balanced solutions. In the wake of calamities in spruce (*Picea abies*) and ash (*Fraxinus excelsior*), large scale planting and site preparation was deemed necessary. A conflict around different interpretations of the Nature Conservation Act of Vorarlberg (LGBl.Nr. 22/1997, 1997, version LGBl.Nr. 67/2019) emerged from this. §25(1) states that long-lasting inferences into the soil structure in alluvial forests require authorisation. Yet, §25(5) also states that no authorisation is required if the inference is part of regular agricultural or forestry use. Additionally, the regulation on soil quality of Vorarlberg requires that management should maintain or improve the soil functions typical for the site (Bodenqualitätsverordnung, 2018, §4). In regard to Landesforstdirektor DI Amann the department for environment in Vorarlberg or rather subordinate departments (Amt der Vorarlberger Landesregierung Abteilung Umwelt), consider the site preparation as long-lasting interference that does not recover within 5 years. The department of forestry doesn't agree with this interpretation. They consider the forestry law (Forstgesetz, 1975) which regarding §13, demands a reafforestation of fallow areas within 5 years (planting or seeding) or 10 years (natural regeneration). One goal of this master´s thesis is to provide a scientific basis for the conflict revolving around these different interpretations. For this quantitative data for an impact assessment of site preparation on soil functions was collected.

Soils are often excluded or reduced to a two-dimensional surface when it comes to land-use management despite their importance (Drobnik et al., 2018). Integrating ecosystem services and in this regard, soil functions into decision-making processes is a challenging task (de Groot et al., 2010; Rozas-Vásquez et al., 2019). Knowledge about soil functions is fragmented among specialised sub-disciplines in soil science making it hard for managers to apply the knowledge (Drobnik et al., 2018; H.-J. Vogel et al., 2018). The needed systemic perspective to capture the impact of management on all soil functions is still in its infancy (Schjønning, et al., 2015; H.-J. Vogel et al., 2018). Yet, this is what is required to present results to stakeholders and policy makers (H.-J. Vogel et al., 2018) which is

crucial for the adaptation of more sustainable management practices. Especially in the context of the climate and the biodiversity crisis we should do everything within the realms of possibility to mitigate our impact on the planet. The base of this is knowledge. A further challenge lies in the scale of the assessment of soil functions. While mapping of soil functions and ecosystem services has been performed in recent years (e.g., Drobnik et al., 2018; Rozas-Vásquez et al., 2019; Taghipour et al., 2022), this might not be the right approach for a small-scale evaluation (Schjønning, et al., 2015) as performed in this work. More studies assessing soil quality, soil ecosystem services and soil functions have been performed on agricultural soils (examples e.g. in Bastida et al., 2008; Drobnik et al., 2018). Some studies assess soil functions from a spatial planning perspective (e.g., Drobnik et al., 2018; Haslmayr et al., 2016). There have also been studies assessing soil functions in forestry. Burger & Kelting (1999) studied soil functions in intensively managed forests in the USA with a main focus on productivity. Zhijun et al. (2018) performed studies on plantation forests with a clear management application of the results. Other studies performed on forests either aim to explore reference values of relatively little disturbed soils (Zornoza et al., 2007) or to show soil degradation processes on different sites in a larger scale (Pang et al., 2006; Taghipour et al., 2022). However, it is still not clear how to specifically measure all soil functions (H.-J. Vogel et al., 2018). To the best of the author's knowledge a versatile method for evaluating soil functions on a scientific level for an application on the spatial scale of forest management units (site preparation, smaller clear cuts) seems to be missing. Thus, the second goal of this work is to develop feasible tools to assess the influence of management on soil functions in a way that they can provide the basis for real-world decision-making. Therefore, indicators for describing soil functions shall be selected.

1.3. Feasible parameters for describing soil functions

Using indicators as proxies for soil functions is justified and supported by empirical evidence. Indicators have to be observable soil properties that contain sufficient information to quantify soil functions (H.-J. Vogel et al., 2018). Thereby, they should be simple and scientifically correct (Schjønning, et al., 2015). The selection of suitable parameters can be challenging. Parameters have to be chosen site specific (H.-J. Vogel et al., 2018) and appropriate for the particular situation (Burger & Kelting, 1999). Different parameters are often in close interaction e.g., soil structure influences water dynamics, structure is formed by soil biota which also depend on structure, this has feedbacks on vegetation etc. (H.-J. Vogel et al., 2018). So a clear separation of indicators for one specific soil function is not reasonable as different processes interact and overlap (Schoenholtz et al., 2000). After compiling the ideas from above with feasible methods available to us, we then selected the following proxies for the assessment of soil functions.

1.3.1. Biological parameters

To draw conclusions on soil functions, biological parameters were assessed. The main parameter sampled for this section is vegetation. Earthworm sampling was planned as well but not conducted as it would have exceeded the scope of this work. Microbial biomass was sampled too, but this is included in the chemical parameters. This was done as these parameters are tightly linked to SOC and N concentrations via eco-physiological ratios. Additionally, the data shows matter concentrations rather than information on communities. Thematically, it seems to fit better with chemical parameters.

Using plant communities to assess the state of a site has a long tradition (Ellenberg et al., 1992). The assessment of vegetation allows direct conclusions on the habitat function of soil. Additionally, the litter inputs of different plant species are important controlling factors for soil parameters such as the C/N ratio (Cools et al., 2014). Vegetation dynamics can reveal valuable information on condition of sites and potential further developments (e.g., Ruskule et al., 2016). Forest management is known to

affect vegetation communities (Aubert et al., 2003; Decocq et al., 2005; A. Fischer et al., 2002; Šebesta et al., 2021; Vanha-Majamaa et al., 2017). On the one hand, disturbances in the canopy via harvesting influence the vegetation. Different intensities of harvesting have distinguishable effects on plant communities (Decocq et al., 2005; Vanha-Majamaa et al., 2017) with clearcutting inducing the biggest changes (Vanha-Majamaa et al., 2017). On the other hand, mechanical site preparation which is frequently combined with clearcutting could have an additional effect on species composition. Šebesta et al. (2021) showed that plant species composition was over decades significantly influenced by mechanical site preparation. The intensity of site preparation seems to be an important property with removal of organic layers via share blading having the largest effect in promoting ruderal species (Haeussler et al., 2002). Generally, species composition often seems to follow a trend. The diversity of herbaceous species increases to a peak before canopy closure, then declines under a closed canopy and increases again when the canopy reopens (Aubert et al., 2003; Haeussler et al., 2002; Šebesta et al., 2021). This follows the trend of the intermediate disturbance hypothesis described by Connell (1978). Disturbances do not only alter the native species composition, but there also seems to be evidence that neophytes frequently profit from disturbances (Haeussler et al., 2002; Schnitzler et al., 2007). Haeussler et al. (2002) found that harvesting in combination with severe site preparation increases the share of non-native species. Šebesta et al. (2021) found that mechanical site preparation significantly increased the share of neophytes in the Thaya flood plains. A species commonly found in European flood plain forests that can be considered to be invasive is *Solidago gigantea* (Schnitzler et al., 2007). As suggested by Haeussler et al. (2002), non-native species can be used to monitor the integrity of ecosystems. *Solidago gigantea* can form dense monospecific stands disrupting the natural species composition (Petrášová et al., 2013) which directly affects the habitat function. Thus, the density of *Solidago gigantea* as vegetation-based indicator for soil functions was used. Species composition was assessed as well. This is an important feature as species are not only part of the habitat function, but they can also have strong effects on ecosystem processes directly influencing energy and material fluxes. Consequently, altering the species composition can affect these processes (Chapin III et al., 2000). Commonly assessed parameters such as species richness, the Shannon index and Evenness were assessed too. Ellenberg indicator values provide a valuable tool to show the ecological behaviour of plants in a community (Ellenberg et al., 1992) and document habitat functions (Schaffers & Sýkora, 2000). They can also show effects of land use changes within a few years (Nitsche & Nitsche, 1994). We expect to show changes in the vegetation composition resulting from harvesting and site preparation as frequently documented in literature (e.g., Haeussler et al., 2002; Šebesta et al., 2021). This effect is assumed to be detectable well above 5 years as the vegetation will need some time after canopy closure to return to the composition of a mature forest stand (e.g., Aubert et al., 2003; Haeussler et al., 2002). Šebesta et al., 2021 could still detect significant differences between plots with and without mechanical site preparation after 40 years. Biodiversity indices are anticipated to follow the trend indicated by the intermediate disturbance hypothesis as documented by Aubert et al. (2003). Species richness will rise until canopy closure and the Shannon index and Evenness will decrease with time as communities get older and thus more organized and homogenous (Aubert et al., 2003). Ellenberg indicator values are assumed to depict the changes in site conditions triggered by the disturbance, i.e., a raise in light and nitrogen values. *Solidago gigantea* covers are expected to increase compared to mature forests after the disturbance and the increased light availability as it is frequently documented in literature (Hall et al., 2022; Schnitzler et al., 2007; Šebesta et al., 2021). We also assume that there will be a mitigation effect caused by the management of the invasive species including initial seeding of a grassland seed mixture and mowing two times a year before flowering. This management has proven to be relatively effective against *Solidago gigantea* (Hall et al., 2022; Info Flora, 2020).

Earthworms play an important role in many soil functions such as organic matter dynamics, nutrient availability and soil structure (Crittenden et al., 2014). They are an important part of forest ecosystems (De Wandeler et al., 2016) and they are expected to be affected by forestry operations. Especially site preparations that includes tilling affects earthworms (Crittenden et al., 2014; Ehrmann, 2015). Soil

compaction affects earthworms too (Ehrmann, 2015). Therefore, it would be interesting to see how strongly and how long-lasting the earthworm population is affected by the tilled strips in the sampling area. Additionally, it can be expected that the change of tree species to more broad leave species will affect earthworms positively (De Wandeler et al., 2018). An evaluation of these impacts on earthworms would have been interesting. Unfortunately, the weather was dry during the sampling periods in spring and fall of 2021 causing dormancy of the earthworm population (Ehrmann, 2015). During dormancy earthworms can be in deep soil layers (Ehrmann, 2015) and the planned extraction method using mustard solution did not work. The mustard extraction method is a cheap, effective and harmless method to extract earthworms. It can also be used on rocky soils where hand sorting is not practical (Valckx et al., 2011). Thus, this method would have been ideal for our site, where half the plots are shallow and rocky. A small experiment after a long rainy period in early November 2021 showed promising results whereas all earlier experiments didn't lead to the extraction of any earthworms. But it was already too late to conduct a full sampling of the area as we already proceeded with the evaluation of the data at that time.

1.3.2. Chemical and microbiological parameters

This paragraph aims to give an overview of the chemical parameters included in this study. This includes pH, microbial biomass and related eco-physiological ratios, soil organic carbon (SOC), total Nitrogen (N_{tot}), C/N-ratio and Nitrate leaching.

Soil pH is a very important soil property affecting soil functions by directly influencing soil chemical reactions and nutrients (Bastida et al., 2008; Schoenholtz et al., 2000). As an example, Bååth & Anderson (2003) suggest that microbial biomass is positively correlated with pH. Microbial respiration increases with the pH value too (Bárcenas-Moreno et al., 2016), impacting the nutrient and carbon cycle etc.. But as pH influences so many biological and chemical reactions simultaneously, it provides little direct information and should be viewed in context with other indicators (Schoenholtz et al., 2000). As a result of this, pH was included as an additional parameter that is not independently seen as an indicator for the effect of treatment. No soil amendments, fertilizers etc. were used so we don't expect the pH to change significantly due to the treatment. This was also shown by Zhijun et al. (2018) who didn't see an influence of rotation cycle on the pH in an intensively managed timber plantation.

Soil microorganisms influence a wide range of soil functions (Blume et al., 2010; Costa et al., 2018). Microbial biomass helps understanding microbial transformations in soil (Bååth & Anderson, 2003) and they present a sink and source of nutrients (Bååth & Anderson, 2003; Blume et al., 2010; Li et al., 2016). Microbial extracellular substances affect aggregation, nutrient availability, water storage and habitat function (Costa et al., 2018). Most importantly (apart from e.g., N fixing and nitrifying bacteria or mycorrhizal fungi), microbes act as decomposers of organic matter (Bååth & Anderson, 2003; Blume et al., 2010). Thus, information about the microbial biomass enables one to draw conclusions on the three ecological soil functions (after Blum (1993), details in chapter 1.1). Microbial biomass is a more sensitive indicator to changes in soil quality than SOC (Bastida et al., 2008; Li et al., 2016) and it is also an indicator for the size of the microbial population (Bastida et al., 2008). Holden & Treseder, (2013) summarized in their meta-analysis that harvesting in forests reduces the microbial biomass. Recovery takes approximately 20 years in boreal forests (Holden & Treseder, 2013). Site preparation affects the microbial community as well (Wang et al., 2018). Tilling in general is known to reduce microbial biomass (Zuber & Villamil, 2016). Based on this information, we expect reduced microbial biomass for treated plots and tilling strips. Additional information about decomposition and carbon storage which is not evident from primary data, can be revealed using eco-physiological ratios (Zechmeister-Boltenstern et al., 2005). The quotient of microbial carbon to organic carbon (Cmic/Corg) is reliable for describing changes in soil (Insam & Domsch, 1988; Malý et al., 2002). If this ratio is high, then the SOC can sustain a large microbial community. Low values indicate a lower decomposability of the SOC for microbes (Zechmeister-Boltenstern et al., 2005). High Cmic/Corg values indicate the accumulation

of carbon at the site (Insam & Domsch, 1988; Malý et al., 2002). This is because recalcitrant carbon remains in more matured soils which causes a lowering of the Cmic/Corg ratio, while easier digestible parts already get decomposed in the O-horizon (Insam & Domsch, 1988). Additionally, Malý et al. (2002) could show a negative correlation of Cmic/Corg (and Cmic) with the metabolic quotient (qCO_2 = CO_2 respiration per g microbial biomass). High qCO_2 values are thought to be indicators for stress and less favourable conditions for microbes (Malý et al., 2002; Zechmeister-Boltenstern et al., 2005). Similar trends can be observed for the ratio of microbial nitrogen and total nitrogen ($Nmic/N_{tot}$) (Malý et al., 2002). As mature soils have lower Cmic/Corg ratios (and $Nmic/N_{tot}$) we can assume higher values on treated sites indicating recovery and carbon accumulation after the disturbance will be found. Another ratio used in literature to assess e.g., the productivity function is Cmic/Nmic (Li et al., 2016). Usually the microbial C/N ratio averages between 4:1 and 8:1 with bacteria having a ratio between 3.5:1 and 7:1 and fungi between 10:1 and 15:1 (Paul, 2007). A shift in this parameter would suggest that the treatment alters the composition of the microbial community on the site. This would also have implications on the mineralization of nitrogen. De Ruiter et al. (1994) showed that feeding of bacteria results in net immobilisation of nitrogen, while fungi cause a net mineralization. Additionally, we tested Nmic/Corg as quotient to further explore potential soil-microbe-interactions.

SOM is a key chemical parameter for the assessment of soil functions (Bastida et al., 2008; Morari et al., 2015; Schoenholtz et al., 2000). It influences all ecological soil functions (Blume et al., 2010; Morari et al., 2015; Schoenholtz et al., 2000). Critical loss of SOM reduces the microbial activity in nutrient cycling (Morari et al., 2015; Schoenholtz et al., 2000; H.-J. Vogel et al., 2018) and the pool and the availability of nutrients (Morari et al., 2015; Schoenholtz et al., 2000). In uncultivated soils SOM represents the largest pool of nitrogen holding approximately 95% of them (Morari et al., 2015; Schulten & Schnitzer, 1998). SOM losses can reduce site productivity and thus litter inputs from plants which causes a further depletion of SOM (Morari et al., 2015; Schoenholtz et al., 2000). It is believed that threshold values for SOM content exist. A value of 2% is suggested in literature (Loveland & Webb, 2003; Romig et al., 1997; Schoenholtz et al., 2000). Yet scientific evidence for this is missing (Loveland & Webb, 2003). Values are believed to differ between soil types (Schoenholtz et al., 2000) and most research in this field focuses on agricultural sites and productivity. Oldfield et al. (2020) showed for example increases in wheat productivity until a SOM concentration of 5% was reached. Losses in SOM also reduce the available water storage capacity (AWC) (Morari et al., 2015; Oldfield et al., 2020) as SOC content is positively correlated with AWC (e.g., Puhlmann & von Wilpert, 2011). Reductions in SOC are directly linked to aggregate stability (Loveland & Webb, 2003; Morari et al., 2015; Schoenholtz et al., 2000) and thus porosity (Morari et al., 2015; Schoenholtz et al., 2000), which is again connected to infiltration and surface run off. SOM depleted soils therefore have higher surface runoff and less infiltration. The depletion process of SOM also negatively affects the green house gas emissions of soils (Morari et al., 2015). SOM depletion impairs the filter and buffer function of soils (Morari et al., 2015), leading to more leaching of e.g., nitrate below the main rooting area (Gundersen et al., 2006). Most often, SOC is used to assess the SOM status or to calculate SOM using a conversion factor. In this work no conversion factors were used as they don´t add further information and the use of general conversion factors from literature might not lead to correct assumptions (Pribyl, 2010). SOC stocks were used as the main indicator for SOM, as this is among SOC content (%) the most appropriate indicator for the evaluation (Morari et al., 2015). This allows a direct assessment of the carbon storage. Carbon storage is part of the regulatory function. Although, due to its relevance in context of the climate crisis, it is often pointed out as a specific soil function in itself (i.e., European Commission, 2006). Additionally, the SOC contents were included for further discussions of the underlying mechanisms impacting the SOC stock. N_{tot} stock, which is tightly connected to SOM (Schulten & Schnitzer, 1998) and where most N in temperate forests is bound (Gundersen et al., 2006) was assessed as well. Measuring nitrogen is commonly used to assess the nutritional status of sites (Schoenholtz et al., 2000) and thus allows drawing conclusions on the productivity function. The assessment of N_{tot} also allows one to draw conclusions on the N-cycle. In temperate forest ecosystems the N-cycle is almost closed between primary producers and the large pool of N in SOM (Gundersen

et al., 2006). Although, alluvial forests with a regular flooding dynamic tend to have less closed cycles (Zechmeister-Boltenstern et al., 2005). The most important processes in the N-cycle are litter production, decomposition, mineralization, immobilization and plant uptake (Gundersen et al., 2006). Disruptions of the N cycle may lead to nitrogen losses (Gundersen et al., 2006; Johnson, 1992). To measure losses, lysimeters were installed below the main rooting area to monitor the nitrate concentration in seepage water. Nitrogen leaching below the rooting area occurs mainly in form of the easily soluble Nitrate (Blume et al., 2010; Gundersen et al., 2006; Schulten & Schnitzer, 1998). These measurements can reveal further information on soil functions. As mentioned above, nitrogen is an essential nutrient for all life forms (Gundersen et al., 2006). Thus, the loss of nitrate via leaching enables one to draw conclusions on the habitat function of soils. Also, measuring nitrate leaching shows parts of the nitrogen matter balance and enables us to investigate post-harvest N losses (Katzensteiner, 2003). Most importantly, it allows us to assess the filter function of soils. This is critical as nitrogen leaching into groundwater and surface water leads to eutrophication (Blume et al., 2010). A further soil parameter assessed in this work is the C/N ratio which is a fundamental indicator of biogeochemical cycles in ecosystems (Li et al., 2016) and a quality indicator for SOM (Morari et al., 2015). The C/N ratio is a determining factor of whether N is mineralized or immobilized by microbes (Schulten & Schnitzer, 1998). Li et al. (2016) summarize that shifts in the C/N ratio could affect soil functions via impacting the nutrient cycling and the structure and composition of plant communities. Research suggests that the C/N stoichiometry affects the structure of the microbial community with the fungal share being positively correlated to the C/N ratio and bacteria vice versa (Wan et al., 2015). Cools et al. (2014) recommend for forests with thin forest floors, the use of the mineral top soil C/N ratio as indicator for the N status of the soil. Changes in land-use (Glenk et al., 2012; Morari et al., 2015), deforestation (Taghipour et al., 2022), timber harvest (Ballard, 2000; Mayer et al., 2020) and site preparation (Mayer et al., 2020; Zuber & Villamil, 2016) affect SOM and connected processes. Therefore, we expect to see an effect of canopy removal and site preparation at our sampling site. On the one hand, SOM is impacted by the change in plant litter input (Morari et al., 2015) and, on the other hand canopy removal increases litter decomposition (Ballard, 2000; Katzensteiner, 2003). Katzensteiner (2003) mentions that this is due to the fact that litter decay is a temperature dependent process. Additionally, tillage as part of site preparation increases the microbial enzymatic activity which accelerates mineralization of SOM (Zuber & Villamil, 2016). There is a large body of literature showing that harvesting generally has negative impacts on SOM (Ballard, 2000; Mayer et al., 2020). The review by Mayer et al. (2020) reviels that the majority of results show site preparation can cause substantial carbon losses. Yet, there are also studies showing that the integration of organic matter into the mineral soil via site preparation can stabilize it and increase the carbon stock (Smolander et al., 2000; Swain et al., 2010). Given this overall information, we can assume that harvesting will diminish the SOM on our sampling sites. The effect of site preparation will likely be negative as well, but here the agreement in the literature is not as strong. The effects on N_{tot} are expected to be similar. The C/N ratio could change after the reafforestation as more deciduous trees are planted. Cools et al. (2014) state that tree species are one of the most important controlling parameters for the C/N ratio in European forests. Increases in nitrate concentration in soil solution after timber harvesting and disturbances are well documented (Gundersen et al., 2006; Hartmann et al., 2016; Hobara et al., 2011; Katzensteiner, 2003; Nave et al., 2011; Palviainen et al., 2014). SOM mineralization is accelerated and exceeds plant and microbial uptake which leads to leaching (Gundersen et al., 2006; Johnson, 1992). Smolander et al. (2000) state that nitrate leaching recovered in the first 3 years after harvesting and site preparation. Gundersen et al. (2006) agree with the statement saying that nitrate loss usually recovers within 3-5 years to pre-harvest levels. Nevertheless, they believe that “site preparation may have large effects on both, the magnitude and the duration of increased nitrate in seepage water” (Gundersen et al., 2006, p. 30). As the degree of site preparation is relatively small on our site, we assume that a recovery of nitrate seepage within 5 years can be shown.

1.3.3. Physical parameters

In this section important physical parameters that can help monitoring the impact of treatment on soil functions will be assessed. This includes soil texture (as master soil property), bulk density, available water storage capacity (AWC) and saturated hydraulic conductivity (Ksat).

Texture changes little over time spans relevant for the assessment of management effects. Thus, it is not useful to directly monitor the impact of certain treatments on texture (Schoenholtz et al., 2000). Nevertheless, it is a fundamental qualitative soil physical property influencing water, nutrient and oxygen exchange, retention and uptake, temperature and consequently all three ecological soil functions (Blume et al., 2010; Schoenholtz et al., 2000) and soil quality (Zhijun et al., 2018). Because texture is such a master soil property, it is included in this work. It makes it possible to account for differences in parameters used for the monitoring of change, potentially caused or influenced by texture differences.

Soil structure is a crucial soil property that is recognized to control many processes in soil. It is a pathway for water, nutrient and gas transport, water retention, influences SOM and nutrient dynamics and root penetration (Peng et al., 2015; Rabot et al., 2018). Soil structure also provides the habitat for soil organisms (Peng et al., 2015) and therefore controls their activity and diversity (Rabot et al., 2018). Soil organisms and plants on the other hand also influence soil structure (Blume et al., 2010; Peng et al., 2015). Mechanical degradation of soil structure induced by compaction (Peng et al., 2015) is a serious problem linked to land-use involving heavy machinery (Peng et al., 2015; Schjønning, et al., 2015). Soil compaction affects nearly all ecological soil functions (Schjønning, et al., 2015). The effects of compaction reduce biomass production (Ballard, 2000; Schjønning, et al., 2015; Schoenholtz et al., 2000) as it affects root growth (Amaranthus et al., 1996; Ballard, 2000; Schjønning, et al., 2015), ectomycorrhizal hyphal growth (Amaranthus et al., 1996), AWC and water and gas transport in soil (Ballard, 2000; Blume et al., 2010; Schjønning, et al., 2015). This also affects the storage, filter, buffer and transformation function of the soil (Blume et al., 2010; Schjønning, et al., 2015), leading to a higher risk of anaerobic conditions causing greenhouse gas emissions form denitrification (Schjønning, et al., 2015). The primary compaction induced on not yet compacted soil triggers the highest changes in soil functions (Blume et al., 2010) and bulk density (Ballard, 2000). Despite being one of the most prominent indicators, bulk density is not ideal to assess soil structure (Rabot et al., 2018). However, bulk density is ideal when it comes to estimating soil compaction (Rabot et al., 2018; Schjønning, et al., 2015) and thus well fitted to assess the potential negative impact of soil compaction induced by treatment on our sampling site. Compaction effects are frequently detectable for decades (DeArmond et al., 2021; Schjønning, et al., 2015). Within a certain range of compaction, soils react resilient and can recover. Beyond that critical point of loading, internal soil structure forming processes can no longer compensate compaction within time spans considered in land-use management (H.-J. Vogel et al., 2018). Increased bulk density after timber harvest is frequently described in literature (e.g., Amaranthus et al., 1996; Aust et al., 2004; Cambi et al., 2015; Frey et al., 2009) and difficult to avoid as any machine used will have an impact on soil properties (Picchio et al., 2020). Froehlich et al. (1985) showed greater compaction for finer soil textures. Recovery rates were similar among soil types and only partial recovery was documented within 23 years. They suggest tilling as a remediation measure. Wang et al. (2016) and Gent Jr. et al. (1984) show decreases in bulk density after site preparation. Yet, harvested sites still had a higher bulk density after site preparation compared to sites not harvested (Gent Jr. et al., 1984). Settling effects after tilling as e.g., described in Mohammadshirazi et al., (2017) commonly occur, increasing bulk density again after the loose soil settles. Thus, the benefits of site preparation might be smaller when assessed several years after tilling. Nevertheless, Bauman et al. (2013) show higher ectomycorrhizal root colonialization and seedling growth on sites objected to mechanical site preparation. They assume that the main driver for this is the lower bulk density. Considering the current state of knowledge, we expect treated sites to have higher bulk densities compared to mature forest stands and tilled strips to have lower values than not tilled areas.

Like soil structure, soil hydraulic properties provide key information for evaluating soil functions (Puhlmann & von Wilpert, 2011). Water retention, availability, infiltration, drainage and water air balance are important for monitoring soil functions e.g., biomass production or storage function (Puhlmann & von Wilpert, 2011; Schoenholtz et al., 2000). AWC and Ksat are the commonly used parameters for this assessment (Schoenholtz et al., 2000). As direct measurements are often difficult, estimating these parameters via pedotransfer functions as a valuable alternative was chosen (Wösten et al., 1999). AWC and Ksat were calculated with pedotransfer functions developed by Wösten et al. (1999) using texture, bulk density and SOC content (%). We want to test if hydraulic parameters based on the interaction of these three input parameters show any significant effects that are not revealed by the single parameters. Jensen et al. (2020) showed that AWC significantly decreases due to compaction induced by land use. This can be partly reversed by site preparation, using methods reaching in deeper soil layers such as subsoiling or tilling. Site preparation also increases the incorporation of organic matter into mineral soil affecting AWC positively (Morris & Lowery, 1988). Fleming et al. (1994) showed no significant effect of harvesting and site preparation on lower soil layers, while AWC in the upper layer was decreased because of site preparation. In their experiment, scalping was used a site preparation method which causes material losses in the upper layer. A review by Aust et al. (2004) clearly showed a decrease of Ksat from logging. This happens due to computation and changes in the pore continuity. If the effects of harvesting can be reversed via site preparation is less clear and depended on the type of site preparation and the pore continuity (Aust et al., 2004). While Gent Jr. et al. (1984) showed positive effects of site preparation on bulk density, this was not the case for Ksat as the soil structure gets degraded. Based on this, we can anticipate AWC and Ksat to be higher in the mature forest stands. AWC might be higher in tilled strips while the effect of tilling on Ksat is uncertain.

1.4. Research questions and hypotheses

Based on the current state of knowledge described above, we condensed the information into three research questions:

a) Which indicators are suitable for describing forest soil functions?
b) Do soil functions recover within five years after harvesting and site preparation?
c) Is there a long lasting (≥ 5 years) difference between tilled planting lines and the area in between?

Derived from the research questions we specifically tested the following hypotheses:

I. Site preparation causes significant differences in soil function indicators between tilled and non-tilled areas in the early phase of forest regeneration
II. Site preparation causes significant differences in soil function indicators between mature forest stands and reafforested sites.

The parameters used to assess this are vegetation data, microbial biomass data, SOC stock, N_{tot} stock, C/N ratio, bulk density, available water storage capacity and saturated hydraulic conductivity derived form pedotransfer functions. NO3-concentration in seepage water below the rooting zone was used as a proxy for the filtering function of the soil. Details on the expected outcome can be found above (chapter 1.3. Feasible parameters for describing soil functions).

2. Methods

2.1. Study area

The study area is located at the confluence of Rhine and Ill close to Feldkirch, Vorarlberg, Austria (fig. 1). It is approximately 430 m.a.s.l., has an average temperature of 9.5°C and an annual precipitation mean of 1360 mm (ZAMG, 2012). The study area extends over 800 ha (G. Fulterer, personal communication, March 7, 2022) and is property of the Agrargemeinschaft Altgemeinde Altenstadt, a group of many small-scale local owners that manage their property similar to one large landowner. The flood dynamics of Rhine and Ill lead to great heterogeneity in soil. Nevertheless, two major soil types accounting for the majority for the study area can be identified. On the one hand we have deep soils with little coarse material, Fluvisols. On the other hand, we have shallow soils with high amounts of coarse material, classified as Rendzic Leptosols (due to missing flooding dynamics and a well-developed forest floor). Due to the proximity to Feldkirch, recreational use is an important factor for the local people. The triangle shaped by the confluence of Ill and Rhine is in large parts protected as part of the Natura 2000 network (fig. 1). The focus of management in the area is to protect open wetlands which are not part of this study. An additional goal is set for the forest to become a semi-natural broadleaved mixed forest with native trees suitable for the site (Amt der Vorarlberger Landesregierung Abt. Umwelt- und Klimaschutz, 2021). The measures for the protected area don't interfere with regular agricultural and forestry use (LGBI.Nr.48/2007; LGBI.Nr.49/2007). Before the river regulations in the late 19th century and in the early 30s of the 20th century the forest was flooded regularly (Fulterer, 2010). Since then, no longer does the forest have the typical flooding dynamic of an alluvial forest. Additionally, the riverbed was deepened via erosion for 3.0-3.5 m at the confluence of Rhine and Ill since 1950. As a result, the ground water levels dropped accordingly (Zoderer, 2010). The history of use and over-use of the forest for fuel wood, timber and as pasture for cattle, goats and sheep dates back for centuries (Fiel, 2010). After the river became regulated the alluvial forest was changed into a spruce (*Picea abies*) forest for timber production. Grazing was pushed out of the spruce forest into the remains of the alluvial forests. This drove back the relicts of the original alluvial forests even further (Fulterer, 2010). Livestock grazing in the forest was ended in 1951 (Fiel, 2010) and in the last decades the share of spruce was reduced in favour of broadleaved species more suitable for the site (Fulterer, 2010). Starting in the 1990s more damage via bark beetles (especially *Ips typographus* and *Pityogenes chalcographus*) in combination with heat waves and wind throws reduced the share of spruce strongly (Fiel, 2010). Starting in 2010 the ash dieback caused by *Hymenoscyphus fraxineus* hit the area. This led to a gradual removal of ash trees (*Fraxinus excelsior*) on the whole sampling area. Ash was mainly growing in mixed stands with spruce and is one of the most important tree species in the area. Between 2002 and 2004 the Dutch elm disease caused the dieback of elm species (*Ulmus glabra* and *Ulmus laevis*) in the sampling area. The extend of this was relatively small, but shows the risk this disease poses to the species (G. Fulterer, personal communication, March 7, 2022) . The extend of the area damaged by calamities extends over a large share of the sampling area. The primary goal is to re-

Figure 1: map of the Natura 2000 protection area Bangs Matschels (retrieved from http://vogis.cnv.at/atlas/init.aspx, 03.03.2022)

establish the forest on fallow areas. To achieve this, site preparation is performed. Plots are first mulched and then planting strips are tilled using a rotary tiller mounted on a tractor. After the damages of the calamities in spruce and ash are overcome, the long-term goal is to establish a *permanent forest* (Dauerwald, a vertically rich structured forest with adapted species like *Quercus robur, Carpinus betulus, Fagus sylvatica, Tilia cordata, Acer pseudoplatanus* etc. that mainly renews itself naturally) (Fulterer, 2010).

Figure 2: Study area in winter

2.2. Field methods

2.2.1. Sampling design

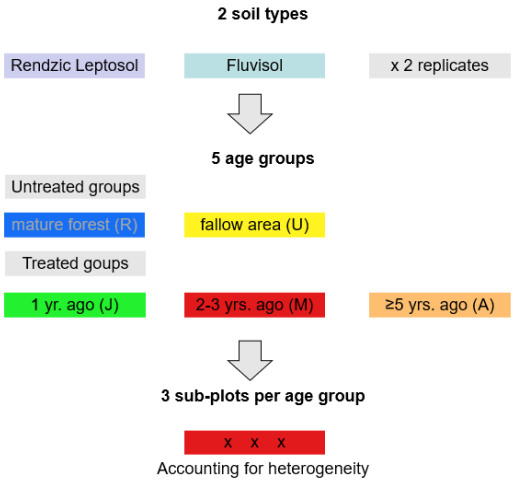

Figure 3: Sampling design

For the sampling design a space-for-time approach was used. The assumption here is that different fragments of a response curve can be found at the same time but at different locations (Wildi, 2017). In this case we are looking at the response curve of soil functions towards the recovery after harvesting, mulching, rotary-tilling and re-planting. Plots that were treated one year ago (J), two to three years ago (M) and larger than or equal to five years ago (A) were sampled. This was compared to mature forest stands (R) and untreated fallow areas (U). The sampling focused on the areas treated ≥5 years ago and the mature forest stands as these areas are the most relevant to answer the research questions. Four clusters containing plots with different age groups were formed. In each of the two soil types two clusters were analysed. Every cluster includes age group A and R and at least two further age groups. To locate two clusters in each of the main soil types in the sampling area, the local forestry operator was interviewed about the area. Potential clusters identified were checked in the field assessing the vegetation and the soil. The soil was evaluated using a drill (so called *Pürckhauer*) to check the lower soil layers and a spade for checking the upper soil and humus forms. After the different aged plots in the clusters were identified, sub-plots were distributed within them. The sampling points for the sub-plots were distributed representatively over the plot and marked with

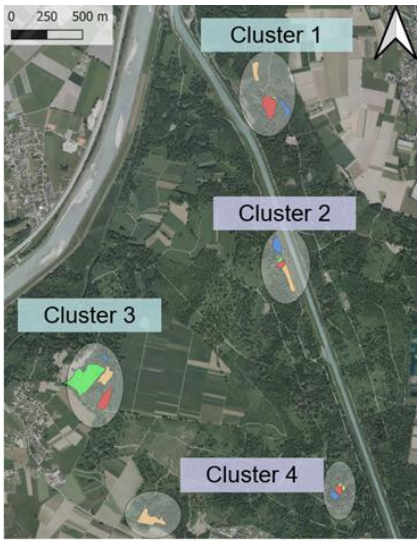

Figure 4: Map of the sampling area (colour coding same as in fig. 3)

poles. A distance of 15-20 m from the edges and between the sub-plots was kept where possible. If the plot was too small, then the distances were minimized accordingly. An overview of the sampling design and a map of the area can be found in figure 3 and 4. For all sub-plots, vegetation data including planted tree seedlings was recorded. Soil profiles were recorded for all sub-plots as well. Only for the subplots of the class A and R soil samples for physical, chemical and biological analysis were collected. These two groups were sampled more intensively as the focus of this study is set on the recovery of soil functions within five years.

Percolating water was sampled with ceramic cup lysimeters (3 bar high flow, Soil Moisture Equ. Corp.) below the main rooting zone at 60 cm depth in order to assess nitrate leaching. Due to the laborious installation of lysimeters as well as sample collection, lysimeters were only installed on plot level for cluster one and two. Per plot two types of lysimeters were installed. Both types can be seen in figure 5 below. For placing the lysimeters in the Fluvisol plots, we used a core sampler to drill holes. In Rendzic Leptosols the holes had to be dug by hand. At all plots a soil suspension of the respective depth was mixed and put on the ceramic suction cup of the lysimeters to ensure a good contact with the soil matrix. The order of soil layers was kept when refilling the holes.

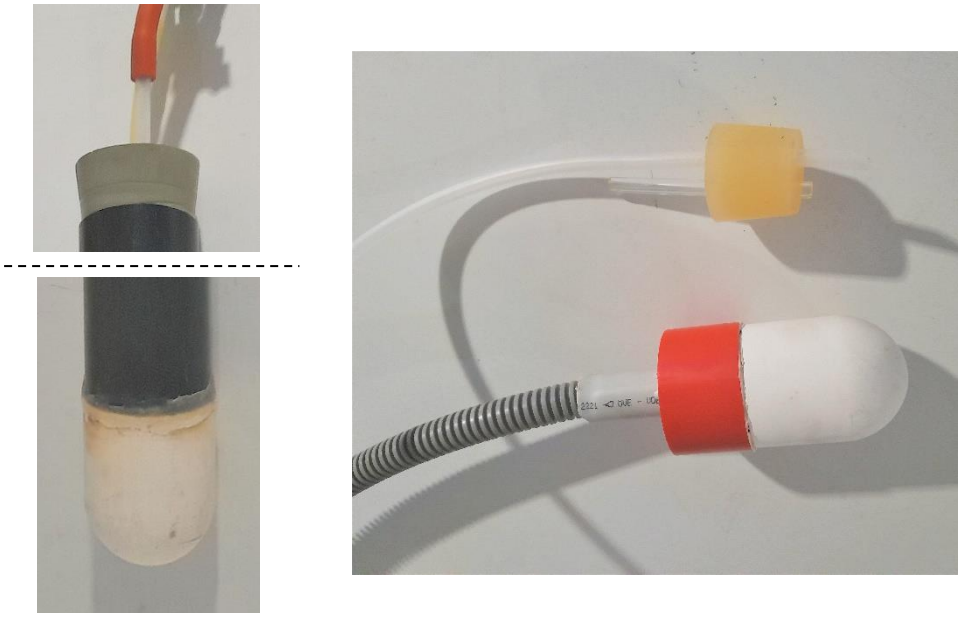

Figure 5: two used lysimeter types

2.2.2. Sample and data collection

Vegetation data was collected for the herbaceous plant cover, the cover of *Solidago canadensis agg.* (*Solidago gigantea* and *Solidago canadensis*) and the planted seedlings.

The sampling of the herbaceous plant cover was conducted between the 19th and 24th of April 2021. Spring was chosen as the sampling time in order to record the spring aspect of vegetation. By doing this the high *Solidago canadensis agg.* covers that can be found in summer were avoided. The plants reach heights above 2 m making systematic sampling of plots difficult. For sampling herbaceous plant cover, a relevé approach was used. Relevés were placed representatively on the sampling area. The decision of placing them representatively was based on the recommendations of Mueller-Dombois & Ellenberg (1974) that the habitat should be uniform and plant cover should be homogeneous without large openings or shifts in dominance of species within the sample area. Vegetation can be recorded better in rectangular shapes due to non-randomness in vegetation distributions (Greig-Smith, 1983). To account for this the size of the relevés was 2x5m (fig. 6). With this size it was possible to cover tilled and non-tilled strips on the replanted plot while still having the benefits of a rectangular shape. Recommendations for sizes of sampling points vary greatly in literature and also depend on the goal of the sampling as well as the sampled community. If a complete inventory of species in one community is the desired outcome, larger areas have to be sampled (Mueller-Dombois & Ellenberg, 1974). Recommended relevé sizes for sampling herbaceous plant vary greatly in literature. In the sampling area both, forest undergrowth vegetation as well as the herbaceous vegetation in the freshly planted areas, which might be closer to open land vegetation, can be found. For these vegetation communities sampling areas from 200 to 10 m² (Mueller-Dombois & Ellenberg, 1974) or even 1 m² (Oosting, 1948) are recommended. It is also important to choose a feasible plot size for the study goal (Mueller-Dombois & Ellenberg, 1974). It is not necessary to measure all species. This does not provide more information or higher significance than an adequate set of samples (Oosting, 1948). As the goal is a comparison of the different treatments a relevé size of 10 m² was considered sufficient. In the relevés the cover of species was recorded in percentage. The procedure was inspired by Dittrich et al. (2013) but adapted to the smaller plot size. For taxa covering more than 15% steps of 5% were used when estimating the cover. Below that the cover was recorded in 1% steps. Species that only occurred once and/or covered less than 1% were put to 0.5%. For the identification of species, the books by Eggenberg & Möhl (2020), Fischer (2008) and Ritz et al. (2017) were used. If there were further uncertainties the vegetation expert Franz Starlinger from the Austrian Research Centre for Forests was asked for advice.

Figure 6: Sampling method herbaceous plant cover

The cover of *Solidago canadensis agg.* was recorded in late June before the sites were mulched or mowed for the first time. The mulching/mowing happens as part of the management of the invasive species. The cover of *Solidago* was estimated in percentage. The separate recording of this species was done because of the dominance it shows later in the year on many plots.

The planted seedlings/saplings were sampled by measuring trees on a 20 m long planting strip per sub-plot. The strip was chosen representatively. For each tree at every sub-plot vigour, damages, height, and diameter (measured at trunk base) were documented. If trees were missing in the planting strips due to death in previous years, they were recorded as missing. Vigour was assessed using a 1 – 5 scale introduced by Carter & Klinka (1992, p. 91):

(1) Seedling not expected to survive more than 2 years and is of very poor vigour with typical symptoms including poor foliage retention, dieback of leading shoots, arrested growth or loss of apical tendency;
(2) Seedling is of poor vigour usually showing many of the symptoms described above but continues to produce new foliage and annual increment;
(3) Seedling is of medium vigour showing poor height and diameter growth but little or none of the symptoms described above;
(4) Seedling is of good vigour showing adequate height and diameter growth and none of the symptoms described above;
(5) Seedling is of very good vigour showing height and diameter growth commensurate with site quality and none of the symptoms described above.

The procedure for sampling soil for chemical and physical analysis differed between soil types. For both soil types, vegetation and organic layer were removed before sampling. In Fluvisols samples were collected using an undisturbed core sampler. Three depth levels were sampled: 0-10 cm, 10-20 cm, and 20-30 cm. The core sampler had a height of 5 cm and a volume of 250 cm^3. Four cores per geometrical horizon were collected. In Rendzic Leptosols this standard procedure was not possible due to the high content of coarse material. Here mini pits were dug, and the extracted soil was separated in three depth levels (10 cm each) as well. Fresh samples were weighted right away. The soil was sieved in field and the content of stones was weighed after sieving for the upper 20 cm. Dead organic material, roots and a portion of the fine soil were packed into plastic bags for further processing in the lab. Then the volume of the geometric horizons using LECA (lightweight expanded clay aggregate) balls was measured. For the lower 10 cm, samples were collected only for chemical parameters and thus without measuring the volume.

Figure 7: Sampling method for obtaining soil samples from Fluvisols (left) and Rendzic Leptosols (right)

Samples for microbial biomass were taken between September 6th and 10th during a period of relatively constant weather conditions. For collecting the samples soil from the upper 10 cm was taken, well homogenized and sieved with a 2 mm sieve. The collected samples were immediately put into a cooler. When returning from fieldwork each day, they were transferred into a freezer where

they were stored at -18°C. The samples were transported from Vorarlberg to Vienna in coolers. They remained frozen during transportation and were then stored at -24°C in the laboratory until analysis.

The lysimeters were placed in the beginning of June at a depth of ca. 60 cm. After that the lysimeters were flushed four times before the sampling started. For sampling a vacuum pressure of 50 kPa was produced with a handpump. Water samples from the lysimeters were collected biweekly by our local partners. The first samples were collected July 13th, 2021, and the last samples October 28th, 2021. An additional set of samples were collected November 22nd, 2021, to collect data from late fall. During the summer one sampling date was missed due to company holidays of our local partners. After collecting, the samples were frozen at -18°C and transported to the laboratory in coolers. The samples remained frozen during transport and were stored at -24°C in the laboratory until analysis. Precipitation data was provided by the water management office of Vorarlberg. Additional weather data was provided by Eduard Walser a local meteorological station owner.

2.3. Laboratory analysis

All laboratory work was carried out at the laboratories of the Institute of Forest Ecology and the Institute of Applied Geology at the University of Natural Resources and Life Sciences, Vienna. The used methods will be described briefly below.

For samples from Fluvisols the first step was weighing the fresh samples for 0-10 cm and 10-20 cm to get the volume reference. Then the samples were sieved with a 2 mm sieve. Dead organic material and roots were sorted out and packed in separate bags. Coarse particles were also weighted. For Rendzic Leptosols, these steps were already carried out in the field. Following this preparation, 20.00g of fine soil was packed and dried to weight constancy at 105°C along with roots and dead organic material. Bulk density of fine soil (in the following referred to as *bulk density*) was calculated using equation 1. The volume of the coarse fraction and organic material was calculated assuming a density of 2.65 g/cm³ and 0.5 g/cm³.

(1) $$bulk\ density = \frac{mass\ soil < 2mm\ dried\ at\ 105°C}{V_{core} - V_{coarse} - V_{organic}}$$

Fresh soil samples were used to measure the pH. pH was measured in both, deionized water (H_2O) and 0.01 M calcium chloride ($CaCl_2$) suspension. Our methods follow the Austrian standard for mineral soil ÖNORM L1083 (Austrian Standards, 2006). For further analysis, calculations were performed using pH in $CaCl_2$ as it is less affected by soil electrolyte concentration and thus provides a more consistent measurement (Minasny et al., 2011).

Total carbon and nitrogen content were measured using the elemental analyser LECO TruSpec (LECO, St. Joseph, MI, USA). For the analysis, samples were combusted and oxygenized at 950°C. After the complete combustion, CO_2 is measured with a infrared detector and N in a thermal conductivity cell (LECO Corporation, 2004). To determine the share of organic carbon the carbonate content was measured, using the gas volumetric method after Scheibler ÖNORM L 1084 (Austrian Standards, 2016). HCl (18%) is used to dissolve the carbonates. The volume of the released CO_2 (*V*) is measured and together with the air pressure (*p*), temperature (*t*), weight of the soil (*m,* 0.5-1 g depending on the carbonate content) and gas constant (*R*) the percentual share of carbonate is calculated (eq. 2).

(2) $$\%CaCO_3 = \frac{p * V}{(273 + t) * m * R}$$

In order to calculate the share of inorganic carbon, the result was multiplied with 0.1265 accounting for the mixture of calcite and dolomite. The inorganic carbon was subtracted from the total carbon to calculate organic carbon. Soil organic carbon stocks and N stocks were calculated linking the percentages with bulk density and the volume of fine soil.

Soil texture analysis was performed combining wet sieving (particles >20 µm) and sedimentation analysis (particles ≤20 µm) using SediGraph III (Micrometrics, Norcross, GA, USA). First 30 g air-dry soil were treated with 10% H_2O_2 to disperse the sample and destroy organic matter. Excess H_2O_2 was removed via heat. Before sieving, the sample was additionally dispersed using ultrasound. The sieved residuals (>630 µm, >200 µm, >63 µm, >20 µm) were dried and weighted. A part of the air-dried soil was oven-dried at 105°C to correct the samples with their water content. Particles ≤20 µm were condensed and sodium polyposphate (0.05% solution) was added. After treatment with ultrasound, samples were measured in the SediGraph III. Soil texture was classified after Sponagel & Ad-hoc-Arbeitsgruppe Boden der Staatlichen Geologischen Dienste und der Bundesanstalt für Geowissenschaften und Rohstoffe (2005).

To measure the microbial biomass the Chloroform Fumigation-Extraction (CFE) technique was used (Jenkinson et al., 2004; Schinner et al., 1996). Two sets of 5g moist soil were put into 50ml centrifuge tubes. One was fumigated with chloroform for 24 h, the reference was left non-fumigated. After fumigation 25 ml of 0.5M K_2SO_4 solution was added and shook in a rotary shaker for 1 hour. As next step the extract is centrifuged for 5 minutes at 4000 rpm and filtered through a Sartorius grade 392 filter paper. A similar extraction procedure was used for the non fumigated sample. The extracts were measured using the Shimadzu TOC-L Analyser (Shimadzu, Kyoto, Japan). To calculate the microbial C and N the organic C or N extracted from the non-fumigated was subtracted from the organic C or N extracted from the fumigated soil. The obtained values were corrected using the weight ratio of dry to fresh soil samples and an extraction efficiency factor of 0.45 for both, C and N (Jenkinson et al., 2004).

For measuring nitrate concentration in the lysimeter samples a spectrophotometer (xMark™ Microplate Absorbance Spectrophotometer, Bio-Rad, Hercules, CA, USA) was used. Incubation time was 30 min at 37°C and a wave length of 540 nm was used. Overall the principles described in Miranda et al. (2001) were followed.

2.4. Data processing and statistical analysis

The data was collected with Excel (Microsoft Corporation, 2020) and processed using R (R Core Team, 2020). Additional R packages used for processing were *here* (Müller, 2020), *rio* (Chan et al., 2021) and *dplyr* (Wickham et al., 2021). Visualization of the data was done with base R (R Core Team, 2020) and the R packages *ggplot2* (Wickham, 2016) and *ggforce* (Pedersen, 2021). To visualize the soil texture, data from the particle size analysis the *R* package *soiltexture* was used (Moeys, 2018).

A key element of the analysis was data exploration. This follows the idea of (Tukey, 1980) who stresses that exploring the data before, during and after analysis is crucial to avoid misinterpretations of structures in the data. Visualization is an important tool for exploration (Tukey, 1980; Zuur et al., 2010). In the initial exploration of the data suggestions by Zuur et al., (2010) were followed. Boxplots were used to detect outliers. Outliers were checked for plausibility and potential measuring or digitalization errors. All values were plausible and thus not excluded from the analysis. Conditional boxplots were used to further explore the structure and homogeneity of the data. Histograms and QQ-plots were used to assess the distribution of the data.

Due to the small number of repetitions, the Nitrate concentrations in leachates will be only assessed visually. Therefore, different plotting methods such as boxplots, line and point plots were used.

Analysis for Vegetation Data

The *Solidago canadensis agg.* cover was evaluated using the *Kruskal-Wallis* test as the data was not normally distributed. To avoid pseudo replications, all calculations were performed with plot-wise mean values. For showing the variation of the data, values on sub-plot level were used for visualisation.

An overview of the growth and vigour of planted seedlings will be provided via graphs and tables.

Ellenberg indicator values (Ellenberg et al., 1992) were assigned to the recorded vegetation. After this, the assessment focused on the herb-layer. The indicator values were obtained using an online tool provided by the University of Natural Resources and Life Sciences, Vienna (https://statedv.boku.ac.at/zeigerwerte/). If the values were not available there, they were looked up in Ritz et al. (2017). Species recorded only on genus level or as *aggregate* were not included. For further assessment the unweighted median values per plot were calculated. The unweighted median was favoured over other commonly used methods for two reasons: (1) the cover percentage follows species traits rather than site conditions (e. g. *Solidago canadensis agg.* covers large percentages without providing very specific site information). (2) the scale of the indicator values is ordinal (or sometimes considered quasi cardinal (Ellenberg et al., 1992)). Thus, the calculation of means is questionable. Additionally, the distribution of the indicator values is often asymmetric. This follows the reasoning by Kowarik & Seidling (1989). In order to provide an overview of the data, all sampling points were kept. Statistical analysis via *Kruskual-Wallis* test (ordinal scale, no variance homogeneity and no normal distribution) were performed on plot level to avoid pseudo replications.

Species richness, Shannon-Index and Evenness were calculated on plot level for the herb layer to give a comparable overview of the different plots. For the calculations the *diversity* function from the *R* package *vegan* (Oksanen et al., 2020) was used.

For the assessment of species composition in the sampling area, a distance-based approach was used. These approaches are commonly used for analysing multivariate data in ecology (Warton et al., 2012). The testing method used was the nonparametric multiple analysis of variance (NP-MANOVA) introduced by Anderson (2001). This method is an advance on previous methods such as ANOSIM introduced by Clarke (1993) (Anderson, 2001). It helps answering the fundamental question in plant ecology about how much of the variance of a full vegetation sample can be explained by one or multiple factors at the same time (Wildi, 2017). NP-MANOVA allows any measure of distance for the distance matrix. Additionally, the results can be easily interpreted because they are provided in the same way as for univariate ANOVA (Anderson, 2001). The NP-MANOVA established in the *adonis2()* function from the *R* package *vegan* was used (Oksanen et al., 2020). Because pseudo replications have to be avoided (Anderson, 2001), it was accounted for the nested design in our data (3 subplots nested in each treatment per plot). The following model-formulation will be used to start:

(3) $$np.mod1 = dist \sim s.type * tr.ment * pl.size * shade * pos$$

With *dist* being the distance matrix of the vegetation sample, *s.type* soil type (Fluvisol or Rendzic Leptosol), *tr.ment* treatment (J, M, A, U, R), *pl.size* plot size (which differs due to the distinction between tilled strips and areas in-between), *shade* the shade casted by the tree or shrub layer and *pos* for the position (tilled, between tilled strips, not tilled). Treatment and position are of interest to answer the research questions. The other parameters were included for logical reasons. Starting from this first model the explanatory variables were reduced stepwise until all input parameters were significant. The final model will also be performed with dummy variables for the different treatments to record their effect (R as reference will be set to 0). Traditional p-values cannot be used for the evaluation of the effects. Due to this, they have to be calculated from permutations of the observations (Anderson, 2001). 999 permutations were used, the default in the *adonis2*-function. To account for this random component in the calculation of the p-value, the models were repeated 25 times to obtain a mean p-value and the standard deviation. The results will be presented in tables. A commonly used distance measure for vegetation analysis is the Bray-Curtis distance (Anderson, 2001; Wildi, 2017). But as the vegetation cover was measured in percentage, the Euclid distance and related distances are also an option (Legendre & Legendre, 2012; Wildi, 2017). The chord distance, which measures the Euclidean distance after normalizing the vectors of the sites to length 1 was chosen (Legendre & Legendre, 2012). This measurement was picked as it provided better results than the

Bray-Curtis distance in this case (see Appendix, NP-MANOVA). For testing different distances and transformations of the vegetation data, *set.seed* was used to get reproducible and comparable results.

Calculation of hydraulic parameters

Hydraulic parameters were calculated for the mature forest sites and sites treated ≥5 years based on the measured physical and chemical parameters. For the calculation of the available water capacity (AWC, or plant available water) and the saturated hydraulic conductivity (ksat) the *R* package *LWFBrook90R* was used (Schmidt-Walter et al., 2021). To obtain the Mualem-van Genuchten (MvG) parameters, two functions of the package were applied. For the upper 30 cm *hydpar_hypres* function with input parameters texture, organic content in percentage and bulk density was used. This function is based on pedotransfer functions derived from the HYPRES data base. It includes 5521 soil horizons from a wide range of soils across Europe (Wösten et al., 1999). Some of the input parameters were not measured in the laboratory. For texture, the estimates from the field analysis were used if no measured values were available. Bulk density was partly estimated using a generalized additive model (details below). In Fluvisols the *hydpar_wessolek,* function based on the work Wessolek et al. (2009) was used for calculating AWC below the depth of 30 cm. This function requires only texture as input. The reduction of input parameters to texture allows the calculation of hydrological parameters for lower depth levels where no laboratory analysis was performed. The data of the upper 30 cm suggests that the influence of SOC below 30 cm is already relatively low, and the bulk density classes estimated in the field were relatively homogeneous in the deeper layer as well. Therefore, it is expected that *hydpar_wessolek* will yield acceptable results for the deeper soil layers. In Rendzic Leptosols, this calculation was not performed as the share of coarse material below 30 cm is very high. Also, the ksat of the deeper soil layers will not be included. It is not expected that this would show more information on soil compaction as bulk density is not used in this formula.

The saturated hydraulic conductivity was directly provided as part of the MvG parameters. As its distribution is logarithmic for further calculations its natural logarithm (ln(ksat)) was used. AWC was calculated using the models of Mualem (1976) and van Genuchten (1980). Additionally, the modifications suggested by (T. Vogel et al., 2001) were included. This approach was applied by Puhlmann & von Wilpert (2012) as well. It leads to the following equation for calculating the water content (θ; in cm³/cm³) dependent on pressure head (h; in cm):

$$\theta(h) = \theta_r + \frac{\theta_m - \theta_r}{(1 + |\alpha h|^n)^m} \quad ;\text{for h} < \text{h}_s \tag{4}$$

With θ_r being the residual water content, α, n and m as empirical parameters of the MvG equation. θ_m is an optimized parameter suggested by T. Vogel et al. (2001). It was calculated using equation (12) in T. Vogel et al. (2001). Equation 4 was used to calculate the water content for the permanent wilting point (θ_{PWP}) and the field capacity (θ_{FC}). The values were chosen following common pressure head values from literature; -150 m for PWP (Blume et al., 2010; Hammel & Kennel, 2001; Wösten et al., 1999) and -0.6 m for FC (Blume et al., 2010; Puhlmann & von Wilpert, 2011). With these values the AWC for each horizon (AWC_H) could be computed:

$$AWC_H = \frac{(\theta_{FC} - \theta_{PWP}) * 100 * T_H * (1 - \text{coarse\_frac})}{1m^2} \tag{5}$$

With T_H being the thickness of the horizon in dm and coarse_frac the fraction of coarse material. As the water content is given in cm³/cm³ this fraction can be transformed into mm/dm by multiplying with 100. With this change the results will be in mm/m². For calculating the AWC per sampling point the sum of all horizons in the main rooting zone was calculated. For a better comparison of the two soil types AWC will be compared for a depth of 0-30 cm. Additionally in Fluvisols AWC will be calculated for lower depth levels to obtain values closer to reality.

Bulk density [*bd*] was only measured for the upper 20 cm (n=60). To estimate the bulk density for the geometrical soil horizon from 20 - 30 cm depth generalized additive model (GAM) was used. Modelling

was performed with the *gam()* function from the *R* package *mgcv* (Wood, 2011). For the variable selection the double penalty approach was used (Marra & Wood, 2011). Marra & Wood, 2011 state, that this approach over all performs significantly better then competing methods. The bulk density classes were excluded beforehand as 54 of the 60 measurements are in the same class. Thus, this doesn't provide any reliable information. The final model includes two explanatory variables, the percentage of SOC [*SOC%*] and the percentage of fine soil [*fs%*], over which smooths [*s()*] were calculated.

$$bd.gam = gam(bd \sim s(fs\%) + s(SOC\%), \quad method = "REML") \tag{6}$$

The method used for smoothness selection was restricted maximum likelihood (*REML*). *REML* was preferred as it yields better results (Marra & Wood, 2011). This is due to a stronger penalizing of overfitting compared to generalized cross-validation (Wood, 2011). Soils with higher SOC concentrations have lower bulk densities (Blume et al., 2010). This is reflected in the model. In the model an increase in fine soil leads to higher bulk density. With this model, an adjusted R^2 of 0.627 could be reached. Checking the diagnostic plots of the model as well as concurvity, did not show any problems. The inclusion of other parameters such as clay content could not further improve the model. Detailed graphs and diagnostic plots of the model can be found in the appendix.

Analysis of chemical, physical and hydraulic parameters

In the specific case of this work classical null hypothesis significance testing (NHST) for modelling was chosen over information theoretic (IT) criteria such as AIC (Akaike´s information criterion). This was done due to the following reasons: (1) we have clear null hypotheses to test, (2) the core assumption of IT-based inference is that candidate models are theoretically and/or empirically well founded (Mundry, 2011). In this case, there are uncertainties about the ability to still detect influence of the site preparation after five years on all of the tested parameters. A mixing of the two approaches (e. g. selecting a model using IT criteria and then applying NHST) is discouraged (Burnham & Anderson, 2002; Mundry, 2011). To avoid the poor strategy of "let[ting] the computer find out" (Burnham & Anderson, 2002, p. 147), a carefully selected set of input parameters was chosen. For testing the effect of treatment and tilling on chemical and physical soil parameters, multiple linear regression and linear mixed-effects models (LMM) were used. If the variation caused by the plot in which the sample was taken did not reach significance the random effect *plot* was excluded. Thus, a simple linear model (LM) was preferred over a mixed-effects model. In case plot reached significance a LMM was used to account for the nested data structure within plot. LMMs were performed and evaluated using the *R* package *lme4* (Bates et al., 2015) and *lmerTest* (Kuznetsova et al., 2017). The global model included the explanatory variable *treatment* to test the hypotheses. Additionally, *soil type* (a major cause of variance on the sampling site) and *plot* were included. Variable interactions were considered as well. Model simplification was done manually with a backward stepwise selection ($p<0.05$). However, the final model selection did not blindly follow p-values. It also considered logical deviations from this approach e.g., caused by the layout of the experiment. In the LMMs, a restricted maximum likelihood estimation was used. Ordinary least squares were used in the LMs. The final models were validated using diagnostic plots. Zuur (2009) calls them a prime tool for model validation. Multicollinearity was checked using the *vif()* function from the *R* package *car* (Fox & Weisberg, 2019) to calculate the variance inflation factors (VIF). The threshold of 3 suggested by Zuur et al. (2010) was used as orientation. Plots used to validate the model assumptions for linearity, homogeneity and normality of the residuals as well as the VIFs can be found in the appendix. Following the recommendations of O'Brien (2007) the goal was an overall balanced model and not simply sticking to rules of thumb. The results of modelling will be presented as p-values as well as point estimates including confidence intervals. Thus, the frequently criticized reporting of "naked" p-values is omitted (Mundry, 2011). Marginal and conditional r-squared for LMMs was calculated using Nakagawa´s R2 for mixed models (Nakagawa et al., 2017; Nakagawa & Schielzeth, 2013) from the *performance R* package (Lüdecke et

al., 2021). The marginal r-squared only assesses the variance of the fixed effects, while conditional r-squared considers both, random and fixed effects (Nakagawa & Schielzeth, 2013).

To test if the parameters were still differed between tilling strip and strip in between after 5 years, a paired sample test was used. Depending on the met requirements either a paired t-test or a paired samples Wilcoxon test were applied.

3. Results

3.1. General Overview

The two different soil types of the sampling area are Rendzic Leptosols and Fluvisols (fig. 8). The description of the soil profile for each soil type can be found in the appendix. For the Fluvisols only mull humus forms were documented. Overall, the soil profiles had little coarse material and the median depth reached with a Pürckhauer drill was 75 cm. At several sites the depth exceeded 100 cm, one plot was relatively shallow with a depth of only 40 cm. Some plots showed (relict) gleying traits and slightly stagnic properties/waterlogging like mottling and concretions were recorded. At the sites with Rendzic Leptosols mull and moder humus forms were found. Less active humus forms were found in mature forest stands. Here the median thickness of the forest floor was 4.75 cm compared to 3 cm on afforested plots. The drill depth never exceeded 30 cm due to the high share of coarse material. In the whole sampling region small-scale heterogeneity formed by past flood dynamics of the rivers could be detected. The strong differences between the two main soil types of the region also show in most recorded parameters. In the following sections the results of the analysis will be presented. Diagnostic plots of models and further information regarding the specific parameters or additional models will be provided in the appendix.

Figure 8: The main soil types of the sampling area, Fluvisols left and Rendzic Leptosols right.

3.2. Soil texture

The soil texture data from the particle size analysis show the different textures of the two soil types. There is one outlier in the Rendzic Leptosols (fig. 10). It can most likely be explained with the alluvial dynamics of the past. Overall, the particle size analysis confirms the results of the finger test in the field. Both points on the border between Uls and Lu were classified as Lu in field. Otherwise, the classes were either identified correctly or a neighbouring class was picked. Only the point at Su3 was wrongly identified as Ls2. This happened probably due to contamination from a higher soil horizon richer in clay, heterogeneity or too much moisture or too high humus content for the finger test. An overview table of the texture for areas treated ≥5 years ago and mature forest can be found below (tab. 1). This shows that Rendzic Leptosols are more heterogenous than Fluvisols and that the mature forest stands have an overall higher sand content. It is also worth noting, that the coarse fragment content differed between plots. Heterogeneity caused mature forest stands to have on the average even rockier soils than the treated plots (fig. 9, tab. 1).

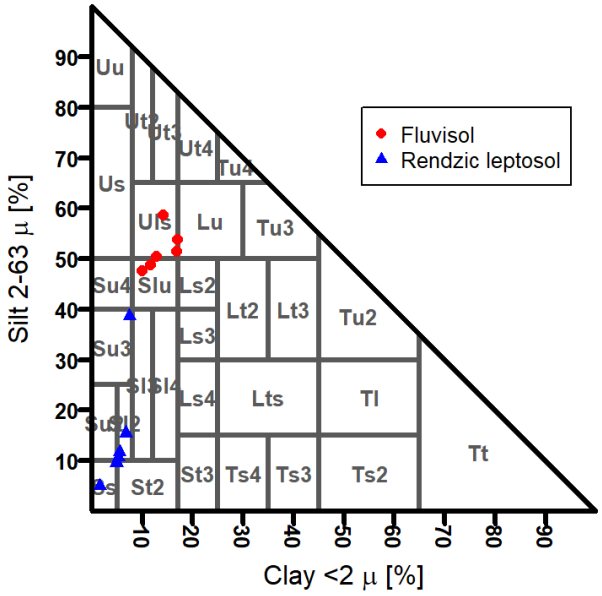

Figure 10: Soil texture triangle with the measured soil texture for 12 soil samples, depth 20-30 cm; L = loam, l = loamy, S = sand, s = sandy, T = clay, t = clayey, U = silt, u = silty

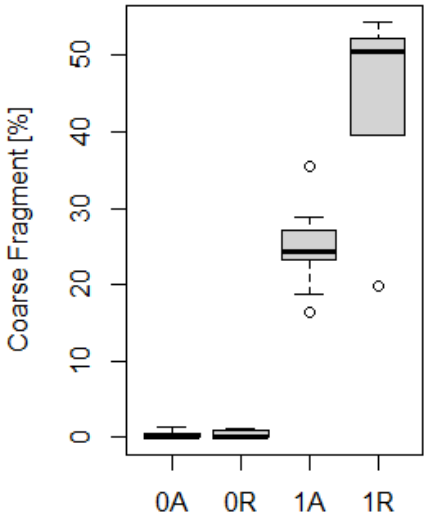

Figure 9: Coarse fragment in percent from 0-20 cm. 0 – Fluvisol; 1 – Rendzic Leptosol; A – area treated ≥5 years ago; R – mature forest

Table 1: Overview texture 0 = Fluvisol, soil type 1 = Rendzic Leptosol, A – area treated ≥5 years ago, R – mature forest, cf = coarse fragment, n = sampling points

		Clay			Silt			cf	n
Soil type	Treatment	0-10cm	10-20cm	20-30cm	0-10cm	10-20cm	20-30cm	0-20 cm	
0	A	21.4±4.5	21.4±4.5	20.2±3.9	63.0±8.4	63.0±8.4	58.9±8.0	0.38±0.51	9
0	R	20.5±7.8	20.5±7.8	18.8±8.2	66.9±8.0	66.9±8.0	60.7±10.6	0.35±0.55	6
1	A	23.9±7.8	23.9±7.8	9.73±7.1	63.2±8.7	63.2±8.7	28.7±22.4	24.80±5.52	9
1	R	20.2±5.2	16.7±8.5	5.6±2.0	50±13.2	43.3±17.9	14.1±5.0	44.52±13.15	6

3.3. Vegetation data

The cover of *Solidago canadensis agg.* did not differ between tilled and non-tilled strips. Thus, the two positions were not distinguished in the analysis either. No significant difference for the two soil types could be measured. The different age groups though had significant effects on the *Solidago* cover (Kruskal Wallis rank sum test p = 0.01484). Significant differences between groups were identified using Dunn´s test (Dino, 2017). In mature forest stands the cover was consistently very low. The untreated fallow areas are varying greatly mainly dependent on shrub cover. From the areas treated one year ago to the areas treated ≥ 5 years ago the *Solidago* cover increases. Overall, the standard deviation, especially for the fallow areas, is very high. Details can be found in figure 11.

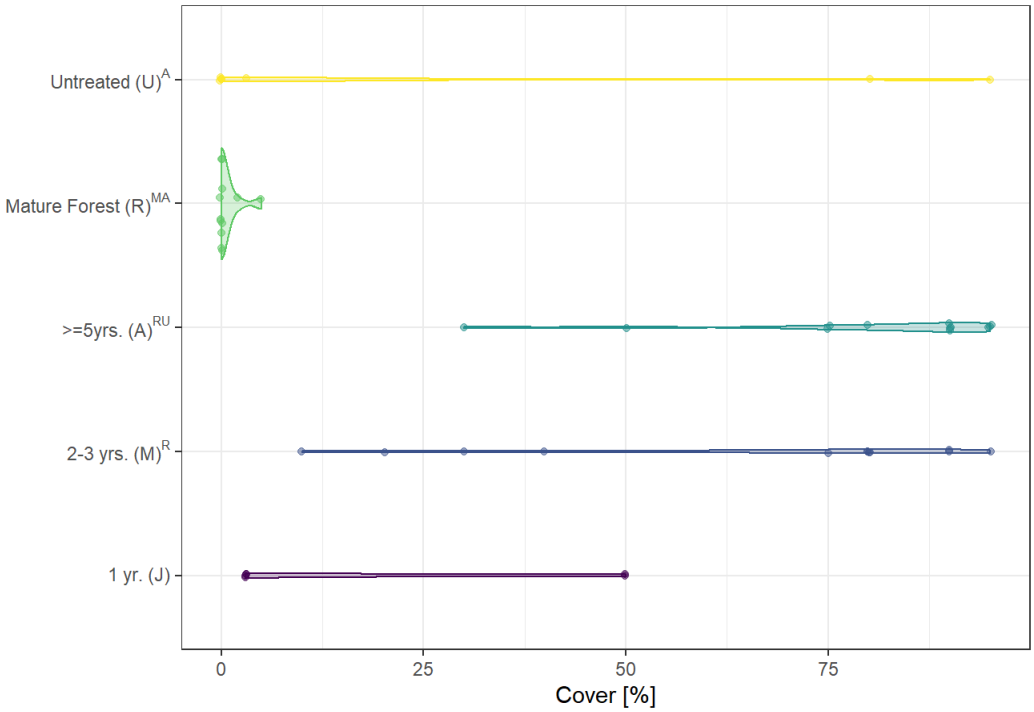

Figure 11: *Solidago canadensis agg.* cover, showing age group on the y-axis and cover percentage on the x-axis. Superscript letters showing significant differences to other age groups. (n for points in graph (sub-plots): n=12 for R, A and M; n=7 for U; n=6 for J; n for significance testing: plot-wise means were calculated thus, n=4 for R, A and M, n=3 for U and n=2 for J)

The planted seedlings perform well. Overall, they are of good vigour (median = 4), and they grow mainly undamaged. The most common damages are mechanical damages by roe deer and mowing equipment. Of the 664 seedlings recorded 88 were dead or missing. The share of missing seedlings was the highest for the untilled plots planted ≥5 years ago (tab. 2).

Table 2: Overview vigour planted seedlings (MN = Median, n = sampling point, vigour class 4 = good vigour with adequate height and diameter growth, further details can be found in chapter 2.2.2.)

Age group	Height (cm)	Diameter (cm)	Vigour (MN)	missing	n
1 yr.	133.0±44.0	1.50±0.47	4	7	6
2-3 yrs.	163.5±61.6	2.30±1.11	4	21	12
≥5 yrs. tilled	242.5±63.5	3.40±2.28	4	18	6
≥5 yrs. untilled	274.0±96.2	3.00±1.15	4	42	6

The most frequently planted seedlings were *Quercus robur, Pinus sylvestris* and *Carpinus betulus* (fig. 12). More drought resistant species were planted at shallower soils. Further species such as *Prunus avium* are planted in the area but were not present on the sampled plots.

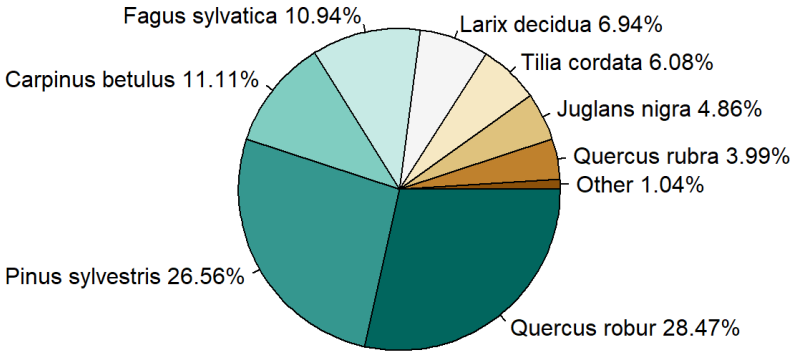

Figure 12: Measured seedlings (n=576); the category *Other* consists of *Acer pseudoplatanus*, *Picea abies* and *Sorbus torminalis* (n<5).

A total of 97 herbaceous species was recorded for the sampling area. 78 in Rendzic Leptosols, 66 in Fluvisols and 47 species were found on both soil types (a full list can be found in the appendix). Typical species for alluvial forests were found in all plots. A clear trend of soil type and light availability could be observed in field. The evaluation of the Ellenberg indicator values shows significant differences between the two soil types: L-values (p=0.038) are higher for Rendzic Leptosols, N (p=0.002) and F-values (p=0.038) are higher in Fluvisols (tab. 3). Comparing the different treatments only L reached marginal significance (p-value = 0.081). But an interesting difference between the N values can be noted. The median for most treatments is 6.5. Only for the mature forest and for the area treated ≥5 years ago the value is 6.

Species richness, Shannon-index and Evenness were normally distributed. No significant differences on plot level could be detected. An overview can be found in table 4. A tendency of a higher species richness for plots outside of mature forests can be seen. Interestingly, the Evenness is the highest for mature forest and areas treated 1 year ago.

The NPMANOVA showed a significant effect of the treatment. Position on tilled strips, in between strips or on non-tilled areas didn't reach significance (p = 0.634±0.017). Transforming abundance data can be seen critically (Warton et al., 2012). With this on mind and given the fact that transformation didn't lead to strong improvements the data was processed untransformed. The initial model (eq. 7) showed that the position was the least significant input parameter. Thus, it wasn't used in the following models. The next reduced model showed that interactions except for soil type and treatment are not significant. This led to the final model:

(7) $$np.mod2 = dist \sim s.type * tr.ment + pl.size + shade$$

Additionally, a model with dummy-variables for the treatment was calculated:

(8) $$np.mod3 = dist \sim s.type * d1 * d2 * d3 * d4 + pl.size + shade$$

With *d1* being a dummy for treatment J, *d2* for M, *d3* for A and *d4* for U. Treatment R is the reference and thus 0. The final NP-MANOVA could explain 65.7% of the variation in the data. The column R2 of table 5 decomposes the total variation of vegetation. Treatment accounts for 38.8 % of the variation, soil type for 7.6%, plot size for 1%, shade adds another 4.4% and the

Table 3: Ellenberg indicator values; minimum, median and maximum values for soil types and treatments. L: light availability; T: temperature; K: climatic continentality; F: moisture; R: reaction (soil acidity); N: nitrogen (soil fertility); n: sampling points

	L			T			K			F			R			N			n
Soil type	min	med	max	min	med	max	min	med	max	min	med	max	min	med	max	min	med	max	
both	3	6	8	5	5	6	3	3	5	4	5	6.5	4	7	8	4	6.5	8	73
Fluvisol	3	5	8	5	5	6	3	3	5	5	5.5	6.5	6	7	8	5.5	7	8	37
Rendzic Leptosol	4	7	8	5	5.5	6	3	3.5	5	4	5	6	4	7	8	4	6	7	36
Treatment																			
1 yr.	6	7	7	5	5.25	6	3	3	5	5	5	6	6	7	8	6	6.5	7	12
2-3 yrs.	3	7	8	5	5.5	6	3	3	5	4.5	5.5	6.5	4	7	8	5.5	6.5	8	24
≥5 yrs.	5	5.5	7	5	5	6	3	3	5	5	5	6	6.5	7	7.5	5	6	7	18
Fallow area	3.5	5	6.5	5	5	5.5	3	3	4	5	5	6	7	7	8	4	6.5	8	7
Mature forest	3	4	5	5	5	5.5	3	3	4	4	5	5.5	7	7	7	4	6	7	12

Table 4: Species richness (S), Shannon index (H) and Evenness (E) on plot-level; n: plots

	S			H			E			n
Soil type	min	mean	max	min	mean	max	min	mean	max	
both	10	22.64±6.20	36	1.52	1.93±0.22	2.34	0.51	0.63±0.06	0.73	17
Fluvisol	10	21.44±7.65	36	1.61	1.89±0.24	2.34	0.53	0.63±0.07	0.73	9
Rendzic Leptosol	19	24.00±4.14	30	1.52	1.96±0.21	2.18	0.51	0.62±0.06	0.69	8
Treatment										
1 yr.	25	27.50±3.53	28	2.11	2.22±0.17	2.34	0.62	0.67±0.08	0.73	2
2-3 yrs.	19	25.50±7.33	30	1.51	1.82±0.23	2.07	0.51	0.57±0.04	0.61	4
≥5 yrs.	15	21.50±5.32	36	1.63	1.87±0.19	2.07	0.53	0.61±0.08	0.69	4
Fallow area	15	21.00±6.56	26	1.76	1.91±0.23	2.18	0.59	0.64±0.04	0.97	3
Matrue forest	10	19.75±6.95	28	1.61	1.94±0.24	2.14	0.62	0.67±0.04	0.70	4

interaction of soil type and treatment 13.9%. Table 6 breaks down the variation of the single treatments from R. Treatment J contributes 6.9% to the 38.8% variation caused by treatment. M accounts for 14.8%, A for 15.9% and U adds another 1.2%. All are highly significant. The interactions of M and A with soil type are not significant. By far the strongest interaction of soil type and treatment is reported for J. This also reaches the highest significance.

Table 5: Output table of *np.mod2* including the used *R*-formula. The p-value is given as mean ± standard deviation of 25 model runs. Soil type:treatment shows the interaction of both.

adonis2(formula = vegdat ~ soil type*treatment+plotsize+shade, method = "chord", permutations = perm)

	Df	Sum of Sqs	R2	F	Pr(>F)
soil type	1	3.762	0.076	13.483	0.001±0.000
treatment	4	19.273	0.388	17.268	0.001±0.000
plotsize	1	0.503	0.010	1.804	0.025±0.004
shade	1	2.192	0.044	7.856	0.002±0.001
soil type:treatment	4	6.909	0.139	6.190	0.001±0.000
Residual	61	17.021	0.343		
Total	72	49.66	1.000		

Table 6: Output table of *np.mod3* including the used *R*-formula. The p-value is given as mean ± standard deviation of 25 model runs. *d1* is the dummy variable for treatment 1 yr. ago, *d2* for 2-3 yrs. ago, *d3* for ≥5 yrs. ago and *d4* for fallow area. Soil type:d1-4 shows the interaction of soil type and treatment.

adonis2(formula =vegdat ~ soil type*d1*d2*d3*d4+plotsize+shade, method="chord", permutations = perm)

	Df	Sum of Sqs	R2	F	Pr(>F)
soil type	1	3.762	0.076	13.483	0.001±0.000
d1	1	3.439	0.069	12.325	0.001±0.000
d2	1	7.338	0.148	26.299	0.001±0.000
d3	1	7.896	0.159	28.297	0.001±0.000
d4	1	0.601	0.012	2.153	0.001±0.000
plotsize	1	0.503	0.010	1.804	0.027±0.005
shade	1	2.192	0.044	7.856	0.003±0.001
soil type:d1	1	4.076	0.082	14.607	0.001±0.001
soil type:d2	1	1.084	0.022	3.884	0.195±0.012
soil type:d3	1	1.132	0.023	4.0578	0.096±0.010
soil type:d4	1	0.617	0.012	2.212	0.021±0.004
Residuals	61	17.021	0.343		
Total	72	49.660	1.000		

3.4. Chemical and microbiological parameters

3.4.1. Microbial biomass

As mentioned above, several eco-physiological quotients additional to the single parameters Cmic and Nmic were calculated. Cmic/Corg and Nmic/Norg showed very similar trends. The same was described by Malý et al. (2002). Thus, Nmic/Norg will be only included in the appendix. Same goes for Nmic/Corg, a quotient tested in this work which also showed very similar results.

Cmic had a mean of 1.90±0.33 mg/g soil in the sampling area (tab. 7). Tilling did not cause significant differences (p=0.219). Areas between tilling strips have a larger variance than the tilled strips (fig. 13, right). Overall, no significant effects were detected, neither from *soil type*, nor from *plot* or *treatment*. Only the intercept model reached statistical significance. Table 7 shows slightly higher values for Rendzic Leptosols compared to Fluvisols, which only reached marginal significance (p=0.081). The boxplot below (fig. 13, left) illustrates this.

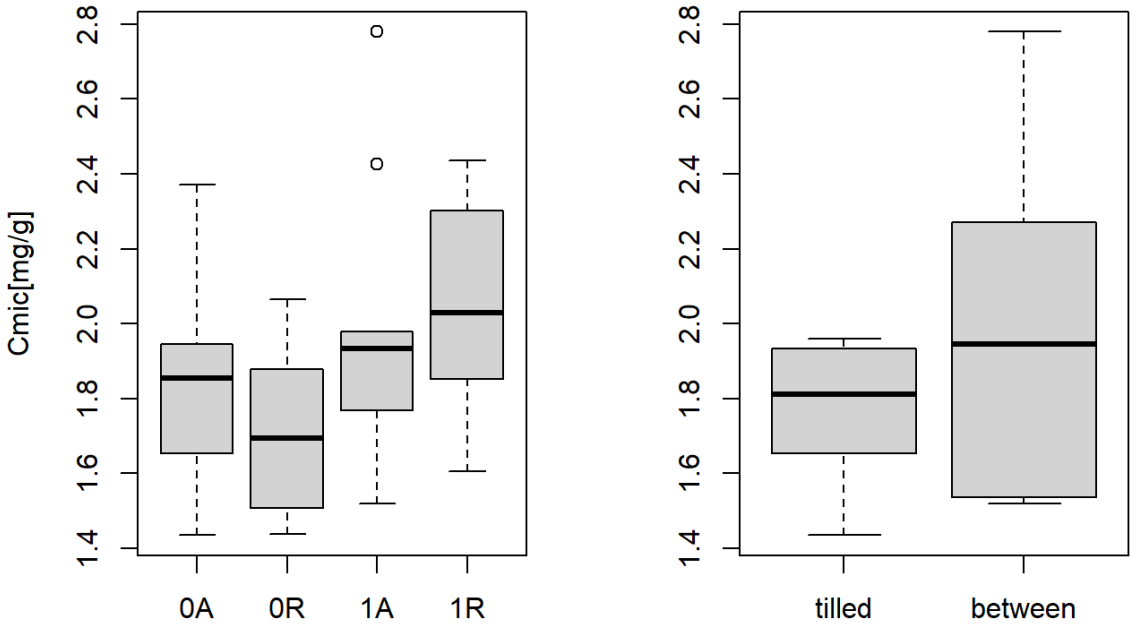

Figure 13: Cmic measured in depth 0-10 cm mineral soil; 0 – Fluvisol; 1 – Rendzic Leptosol; A – area treated ≥5 years ago; R – mature forest

For Nmic, the average on the sampling area was 0.41±0.08 mg/kg (tab. 7). Significant differences between tilled strips and the area in between were not recorded (p=0.438). The statistical test results for Nmic were similar to those for Cmic and the best model only contained the intercept. Model and diagnostic plots can be found in the appendix.

Table 7: Overview of soil microbial biomass (0-10 cm mineral soil depth), soil type 0 = Fluvisol, soil type 1 = Rendzic Leptosol, Cmic, Nmic and Cmic/Corg in mg/g, Cmic/Nmic as ratio.

Soil type	Treatment	Cmic	Nmic	Cmic/Nmic	Cmic/Corg	n
Both	Both	1.90±0.33	0.41±0.08	4.67±0.49	27.18±6.69	30
0	Both	1.80±0.29	0.40±0.07	4.46±0.27	28.48±4.62	15
1	Both	2.01±0.35	0.42±0.09	4.87±0.57	25.89±8.58	15
0	A	1.85±0.32	0.41±0.08	4.55±0.26	28.62±5.17	9
0	R	1.71±0.23	0.40±0.07	4.34±0.26	28.26±4.10	6
1	A	1.98±0.39	0.43±0.10	4.65±0.31	31.54±4.32	9
1	R	2.04±0.31	0.40±0.08	5.21±0.74	17.41±5.69	6

The microbial C/N ratio had a mean of 4.67±0.49 in the sampling area. Tilling strips did not significantly differ from the areas between strips (p=0.971, fig. 14, right). A model with soil type remaining the only significant explanatory variable could be computed (tab. 8). Rendzic Leptosols have a C/N ratio 0.407 higher than Fluvisols. The model reached a R-squared of 0.152 and fulfilled the assumptions. A second model reached a higher R-squared (0.229). This model included the interaction of soil and treatment. While *soil* and *treatment* did not reach statistical significance in this model, their interaction did. It shows clearly that mature forest plots on Rendzic Leptosols have a higher microbial C/N ratio than the intercept (tab. 7, fig. 14, left). The fitted vs. residuals plot (in appendix) shows that the model has a relatively good fit for lower microbial C/N ratios. Higher ratios are not covered as well as lower ones. This is mainly due to one point with a microbial C/N ratio of 6.61. As this presents a realistic value appropriate for the sampling point, it was not excluded. Yet, it means that the effect of the interaction of *soil* and *treatment* could be weaker. Otherwise, model assumptions are met.

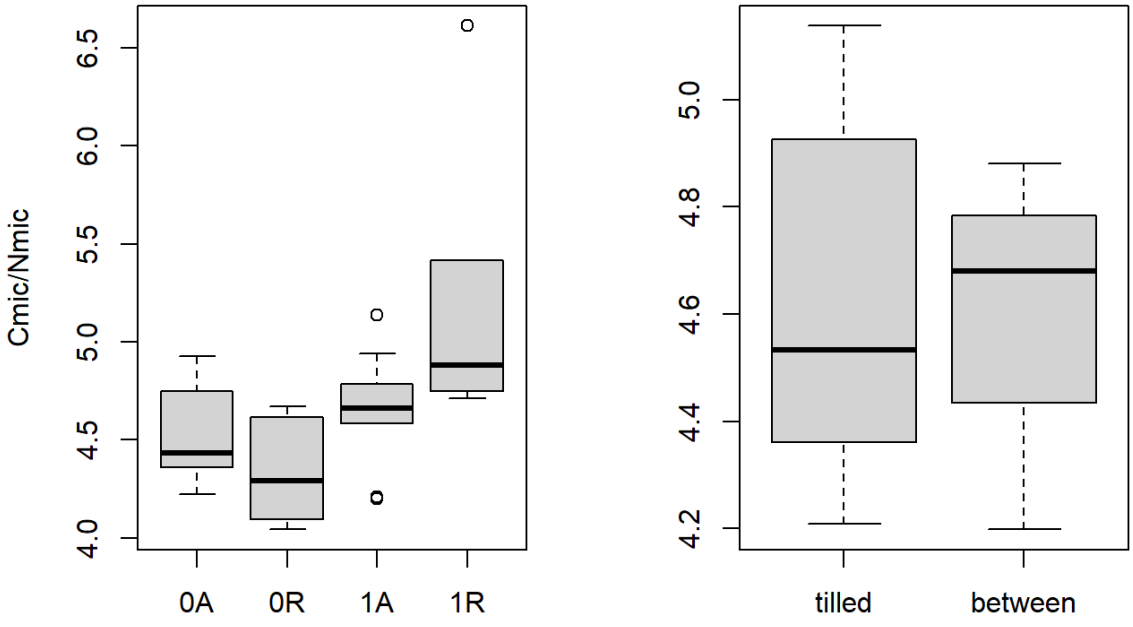

Figure 14: Cmic/Nmic ratio (0-10 cm mineral soil depth); 0 – Fluvisol; 1 – Rendzic Leptosol; A – area treated ≥5 years ago; R – mature forest

Table 8: Output modelling Cmic/Nmic (0-10 cm mineral soil depth); Rendzic Leptosol=*soil1*; mature forest stand=*treatmentR* and their interaction (*treatmentR:soil1*), adj. = adjusted, VIF = Variance Inflation Factor

Formula: CNmic~soil*treatment					
	Estimate	Std. Error	95% CI	t value	p
Intercept	4.551	0.136	[4.271, 4.831]	33.457	<2E-16
soil1	0.098	0.192	[-0.299, 0.492]	0.503	0.619
treatmentR	-0.216	0.215	[-0.658, 0.226]	-1.003	0.325
soil1:treatmentR	0.776	0.304	[0.151, 1.401]	2.553	0.017
R2 (adj.)	0.229	p-value	0.021		
VIF	soil	treatment	soil:treatment		
	1.667	2	2.667		

The Cmic/Corg ratio has a mean of 27.18±6.89 mg/g. Differences between tilled strips and areas in between did not reach significance (p=0.385, fig. 15, right). A first model with only significant parameters was calculated with *treatment* as explanatory variable (tab. 9). It shows that mature forest stands have a Cmic/Corg ratio that is 7.24 lower than reafforested sites. However, the confidence interval is large, and the effect could be weaker or stronger. Model assumptions are met. Nevertheless, the model only explains 24.8% of the variation in the data.

Table 9: Output modelling Cmic/Corg (0-10 cm mineral soil depth); mature forest stand=*treatmentR*, adj. = adjusted

CmicC~treatment					
	Estimate	Std. Error	95% CI	t value	p
Intercept	30.079	1.409	[27.191, 32.965]	21.34	<2E-16
treatmentR	-7.242	2.228	[-11.806, -2.678]	-3.25	0.003
R2 (adj.)	0.248	p-value	0.003		

A significantly better model can be calculated including the interaction of *soil type* and *treatment* (tab. 10). With this model 50.7% of the variance can be explained. Fluvisols tend to have a slightly lower Cmic/Corg ratio. But as the confidence interval includes zero, this effect could also be reversed. The effect of treatment on its own is even more uncertain. But mature forest stands on Rendzic Leptosols have a significantly lower Cmic/Corg ratio. It is 13.77 lower than the intercept and the confidence interval underlines the clear direction of the effect. Figure 15 (left) illustrates this result. Neither diagnostic plots nor the variance inflation factor led to a rejection of the model assumptions.

Table 10: Output modelling Cmic/Corg (0-10 cm mineral soil depth); Rendzic Leptosol=*soil1*; mature forest stand=*treatmentR* and their interaction (*treatmentR:soil1*), adj. = adjusted, VIF = Variance Inflation Factor

Formula: CmicC~soil*treatment					
	Estimate	Std. Error	95% CI	t value	p
Intercept	28.622	1.613	[25.306, 31.938]	17.74	4.78E-16
soil1	2.914	2.282	[-1.776, 7.604]	1.28	0.088
treatmentR	-0.358	2.551	[-5.602, 4.886]	-0.14	0.281
soil1:treatmentR	-13.768	3.608	[-21.184, -6.353]	-3.82	7.53E-04
R2 (adj.)	0.507	p-value	7.87E-05		
VIF	soil	treatment	soil:treatment		
	1.667	2	2.667		

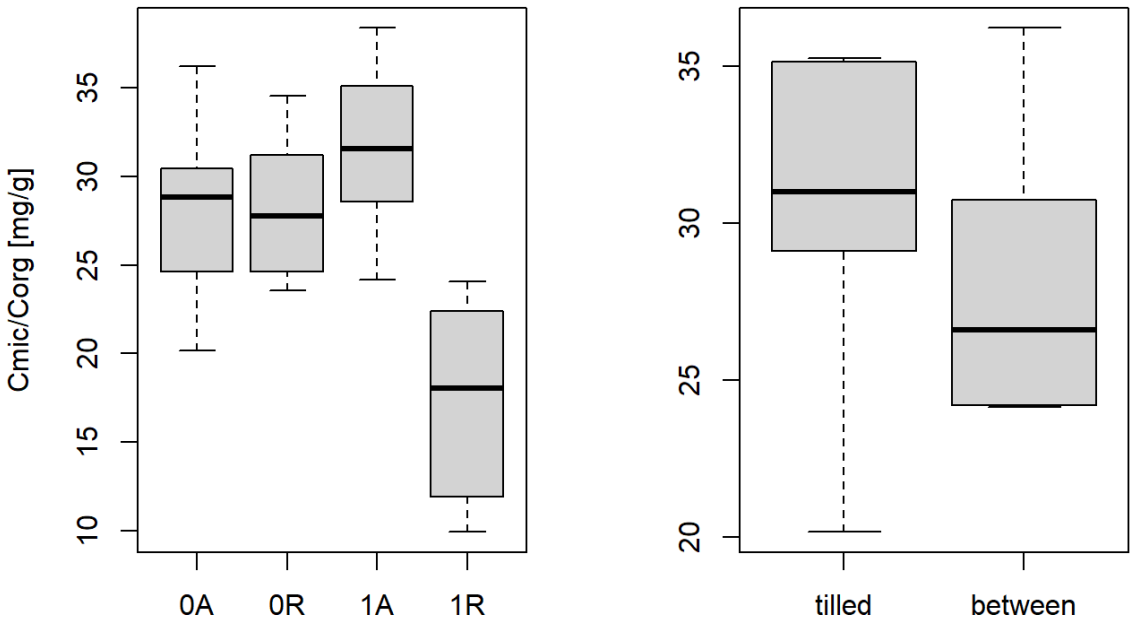

Figure 15: Cmic/Corg ratio, 0 – Fluvisol; 1 – Rendzic Leptosol; A – area treated ≥5 years ago; R – mature forest (0-10 cm mineral soil depth)

3.4.2. pH, SOC and N

The pH values of the area are overall high with a mean of 7.12±0.43 pH [$CaCl_2$] in the upper 20 cm. Fluvisols showed higher pH values (7.19±0.54 [$CaCl_2$]) than Rendzic Leptosols (7.05±0.29 [$CaCl_2$]) (tab. 12). This effect is most pronounced in the upper 10 cm and stronger for pH values measured in water (fig. 16). Treated areas showed slightly higher pH values for the upper 10 cm compared to mature forest stands in Rendzic Leptosols (fig. 16). In the mature forest in Fluvisols, an outlier can be found in the boxplot below. The specific plot was a dense spruce forest with vegetation indicating lower pH values. In the depth from 20-30 cm, pH values in $CaCl_2$ are relatively consistent over soil types and treatments. Differences in pH values between tilled strips and areas in between were not significant ($p = 0.618$) for the depth 0-20 cm. Yet, for the depth 20-30 cm the differences were larger. 7.35±0.26 (in $CaCl_2$) was measured for tilled strips compared to 7.52±0.15 (in $CaCl_2$) between tilling strips. The best model for pH in $CaCl_2$ was achieved with a LMM only containing the intercept and the random nested effect of plot (conditional R2 0.419). The plot effect also covers the effect of soil type. Thus, soil type didn´t reach significance in the model. The outlier mentioned above was not excluded from the model as it represents the heterogeneity of the area caused by vegetation and alluvial dynamics. Diagnostic plots for homogeneity and linearity seemed ok given the small amount of data points. Normality looks acceptable for the center of the data.

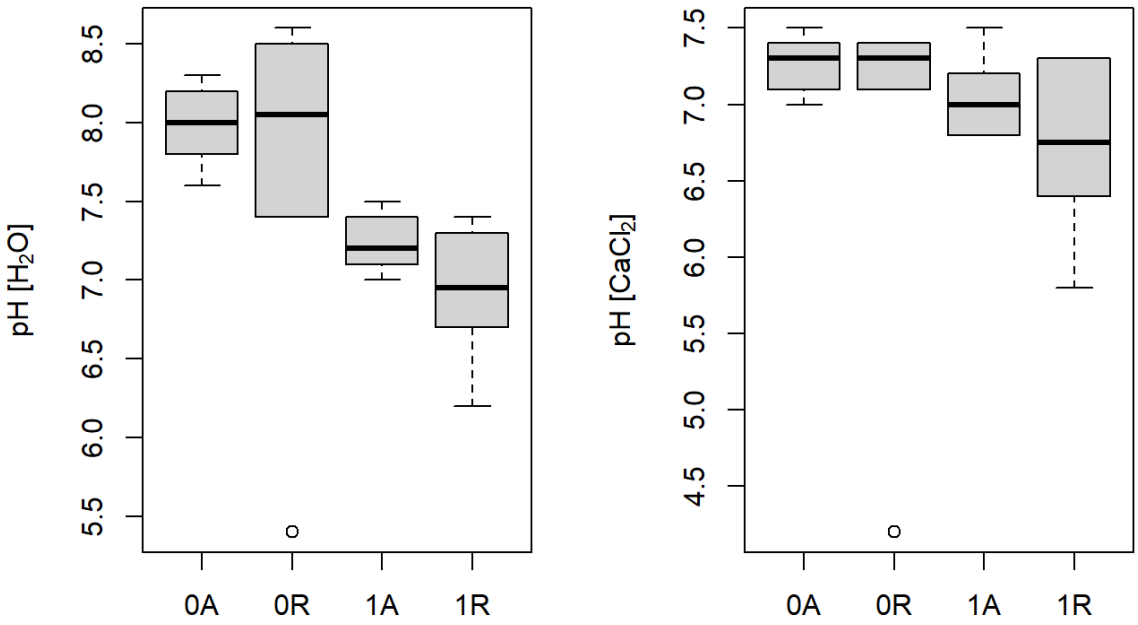

Figure 16: pH values measured in 0-10 cm mineral soil depth; 0 – Fluvisol; 1 – Rendzic Leptosol; A – area treated ≥5 years ago; R – mature forest

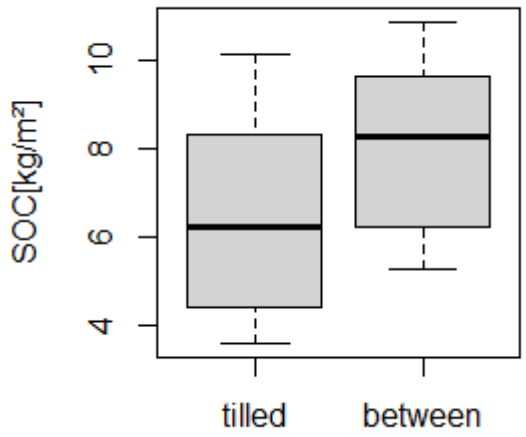

Figure 17: Differences in SOC stocks (0-20 cm mineral soil depth) for tilled strips and areas in between; p = 0.009

SOC stocks from 0 to 20 cm averaged at 6.77±2.12 kg/m² in the area. Again, substantial differences between Fluvisols (8.48±1.03 kg/m²) and Rendzic Leptosols (5.05±1.39) can be reported (tab. 12). Differences between tilled strips and areas in between were highly significant (p = 0.009, fig. 17). Tilled strips showed lower SOC stocks than areas in between. For modelling the SOC stocks, a LM was chosen as the variable plot did not reach significance and was explained well by the input parameters *soil type* and *treatment*. Both were highly significant and showed clear trends. If the soil is a Rendzic Leptosol then the SOC stock is 3.43 kg/m² lower than the intercept. Mature forests have a SOC stock 1.307 kg/m² lower than reafforested plots. Both trends are very clear with confidence intervals staying well below zero (tab. 11). The linear model reached an r-squared of 0.756. All model assumptions were fulfilled. The boxplot (fig. 18) shows the differences in SOC stocks for treatment and soil type for 0-10 cm and 10-20 cm. It shows that the differences between treatments is higher for the depth level 10-20 cm compared to 0-10 cm. In the upper 10 cm, the difference between treatments is larger for Fluvisols than for Rendzic Leptosols. Interestingly, the SOC content (%) shows similar trends in Fluvisols but in Rendzic Leptosols the trend is reversed (tab. 12). Mature forest stands have a lower SOC stock while

having a higher SOC content. Also, when comparing tilling strips with areas in between the differences were no longer significant for the SOC content (p=0.136, boxplot in appendix). Nevertheless, the trend of tilling strips having less SOC content than areas in between was still visible.

Table 11: Output of linear model describing the SOC stock from 0-20 cm mineral soil depth, Rendzic Leptosol=*soil1*; mature forest stand=*treatment,* adj. = adjusted, VIF = Variance Inflation Factor

Formula: SOCStock ~soil+treatment					
	Estimate	Std. Error	95% CI	t value	p
Intercept	9.005	0.312	[8.365, 9.645]	28.861	<2E-16
soil1	-3.43	0.382	[-4.214, -2.646]	-8.975	1.37E-09
treatmentR	-1.307	0.39	[-2.107, -0.507]	-3.351	0.002
R2 (adj.)	0.756	p-value	2.06E-09		
VIF	treatment	soil			
	1	1			

Table 12: Overview of soil parameters (0-20 cm mineral soil depth), 0 = Fluvisol, soil type 1 = Rendzic Leptosol, A – area treated ≥5 years ago, R – mature forest, SOC Stock [kg/m²], N Stock [kg/m²]

Soil type	Age group	pH [CaCl2]	SOC %	SOC Stock	N Stock	C/N	n
Both	Both	7.12±0.43	5.96±1.75	6.77±2.12	0.49±0.17	14.23±2.30	30
0	Both	7.19±0.54	5.32±1.03	8.48±1.03	0.61±0.10	14.21±2.36	15
1	Both	7.05±0.29	6.59±2.10	5.05±1.39	0.37±0.14	14.25±2.32	15
0	A	7.30±0.20	5.57±1.21	8.94±1.07	0.62±0.10	14.73±2.58	9
0	R	7.03±0.83	4.95±0.57	7.80±0.42	0.59±0.11	13.42±1.92	6
1	A	7.08±0.27	5.50±1.38	5.64±1.29	0.44±0.12	12.83±1.10	9
1	R	6.99±0.34	8.24±1.96	4.17±1.11	0.26±0.07	16.39±2.00	6

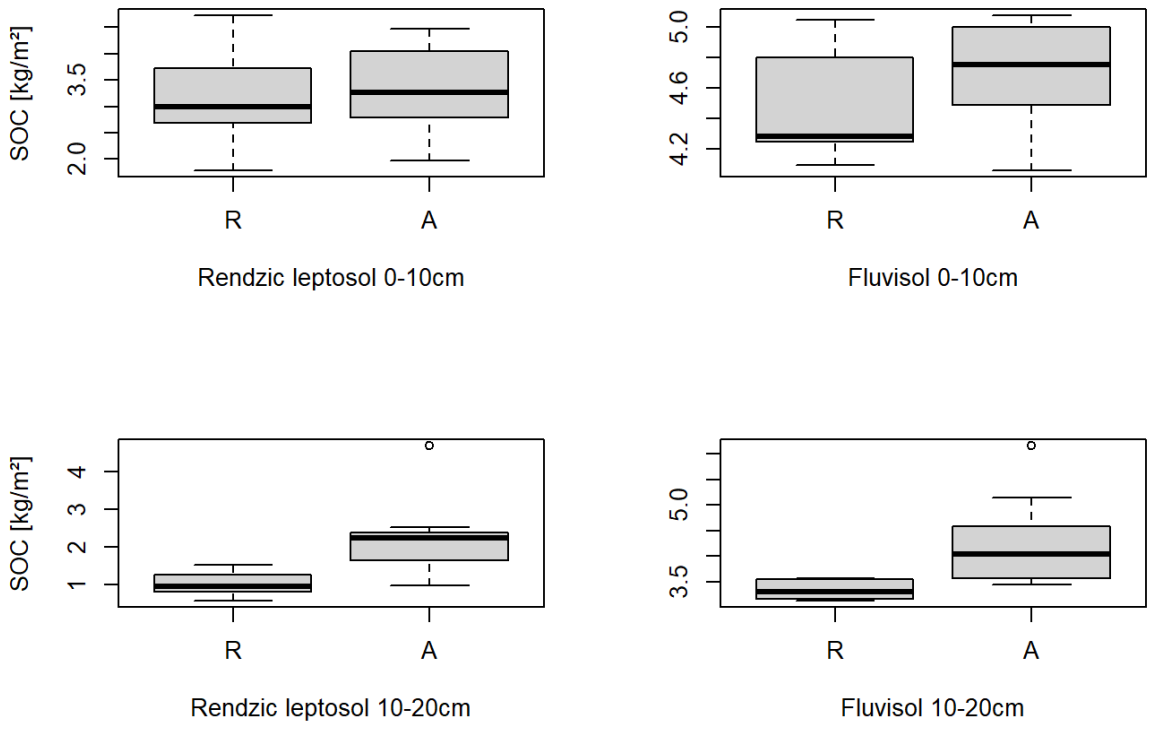

Figure 18: Comparing SOC stocks of mature forest stands (R) and reafforested areas (A) for the two soil types

N stocks again show this clear difference between Fluvisols (0.61±0.10 kg/m²) and Rendzic Leptosols (0.37±0.14 kg/m²) for the upper 20 cm (tab. 12). Differences between tilled strips and areas in between were significant (p=0.025) from 0-20 cm for N stocks. The boxplot (fig. 19, right) shows a similar trend like for SOC stocks with the area in between having higher N stocks. Like for SOC, the N content in percentage doesn't show significance (p=0.146, boxplot in appendix). A boxplot comparing N stocks for 0-10 cm and 10-20 cm for the two soil types and treatments can be found in the appendix as well. It shows very clearly higher N stocks for both depth levels in reafforested plots for Rendzic Leptosols. In Fluvisols this trend is weaker. N stocks are slightly higher for mature forest stands in the upper 10 cm. Reafforested plots show higher N stocks from 10-20 cm. The effect of treatment is reflected in both models performed for N stocks 0-20 cm mineral soil depth. LMMs were used as *plot* reached significance. The first model only consists of the random nested effect of *plot* and the fixed effect of *treatment* (tab. 13). This model includes only significant terms. It reaches a marginal R-squared of only 0.101 and a conditional r-squared of 0.794 with most of the variation being explained by the random effect. If the treatment is *mature forest stand* (*R*) N stocks (0-20 cm mineral soil depth)

Table 13: Output of LMM describing the N Stock (0-20 cm mineral soil depth) with treatment as fixed effect and plot as nested random effect; mature forest stand=treatmentR, R2m = marginal R2, R2c = conditional R2

Formula: Nstock ~treatment+(1|plot/subplot)

Random effects	**Variance**	**sd**				
subplot:plot	9.02E-05	0.009				
plot	2.61E-02	0.162				
residuals	7.81E-03	0.088				
Fixed effects	**Estimate**	**Std. Error**	**95% CI**	**df**	**t value**	**p**
Intercept	0.55	0.084	[0.367, 0.733]	3.145	6.572	0.006
treatmentR	-0.124	0.033	[-0.191, -0.057]	18.737	-3.72	0.001
R2m	0.101					
R2c	0.794					

are 0.124 kg/m² lower than for reafforested plots. Confidence intervals show that this trend is clearly negative, but the magnitude could be substantially stronger or weaker. The residual vs. fitted plot shows some minor irregularities most likely due to the low number of sampling points. Deviations are too small to reject the assumptions of linearity and homogeneity. The normality assumption is fulfilled. A second LMM was calculated containing the interaction of *treatment* and *soil type* (tab. 14). This interaction accounts for the different trends for the soil type visible in figure 19 (left). It shows that the interaction of *soil* and *treatment* is significant. If a Rendzic Leptosol site is a mature forest stand the N stock is 0.139 kg/m² lower than the intercept. Again, the confidence interval is large but shows a clear negative trend. *Soil* and *treatment* by itself do not reach significance. Soil type Rendzic Leptosol and mature forest stands both seem to have lower N stocks than Fluvisols or afforested sites, but the trend is not clear as the confidence interval includes zero. Overall, the model performs better than the previous model with a marginal R-squared of 0.591. The conditional R-squared reaches 0.819. The model assumptions are met, and the diagnostic plots look cleaner for this model compared to the previous model.

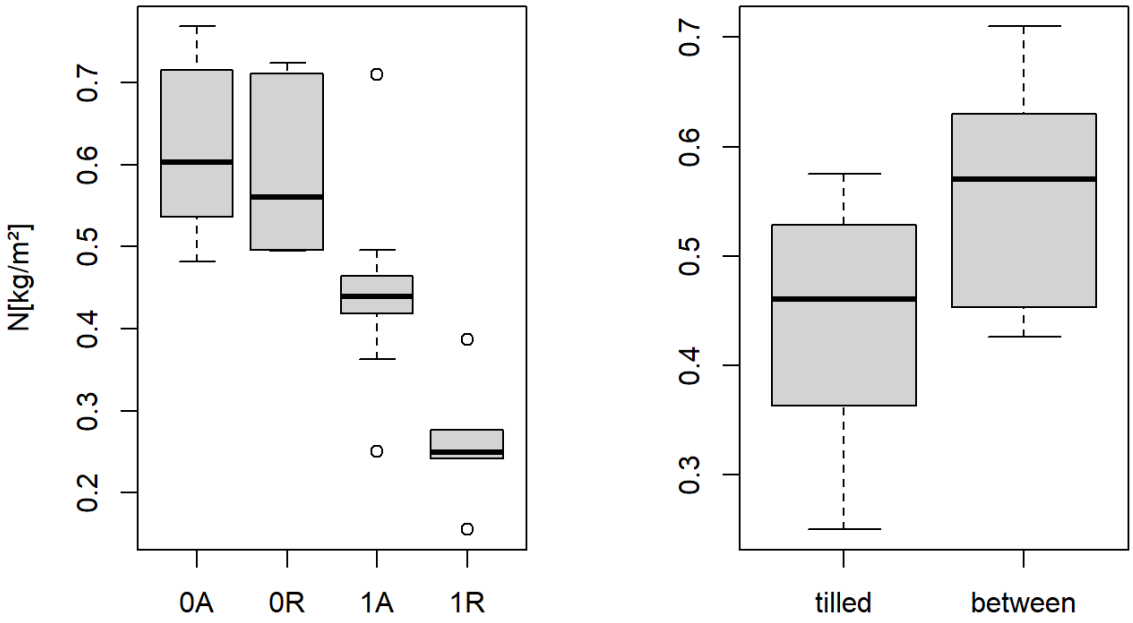

Figure 19: N stocks in kg/m² 0-20 cm mineral soil depth; left: N stocks for soil type and treatment (0 – Fluvisol; 1 – Rendzic Leptosol; A – area treated ≥5 years ago; R – mature forest); right: Differences in N stocks for tilled strips and areas in between (p = 0.025)

Table 14: Output of LMM describing N Stock with *soil, treatment* and their interaction (*treatmentR:soil1*) 0-20 cm mineral soil depth; Rendzic Leptosol=*soil1*; mature forest stand=*treatmentR;* R2m = marginal R2, R2c = conditional R2, VIF = Variance Inflation Factor

Formula: Nstock ~treatment*soil+(1|plot/subplot)

Random effects	**Variance**	**sd**				
subplot:plot	0.0005	0.023				
plot	0.0075	0.0866				
residuals	0.0064	0.08				
Fixed effects	**Estimate**	**Std. Error**	**95% CI**	**df**	**t value**	**p**
Intercept	0.645	0.068	[0.526, 0.766]	2.282	9.513	0.007
treatmentR	-0.054	0.043	[-0.133, 0.034]	17.607	-1.253	0.226
soil1	-0.193	0.096	[-0.363, -0.023]	2.281	-2.008	0.167
treatmentR:soil1	-0.139	0.06	[-0.260, -0.024]	17.607	-2.302	0.034
R2m	0.591		VIF	treatment	soil	treatment:soil
R2c	0.819			2	1.073	2.073

The C/N ratio averaged at 14.23±2.30 for the sampling area. No major differences between soil types were recorded for the upper 20 cm. A paired t-test showed that there were no significant differences between tilled strips and areas in between (p=0.901, fig. 20, right). For testing the influence of *treatment* on the C/N ratio, LMMs were used as *plot* reached significance. A final model with only significant explanatory variables only included *treatment* and the random *plot* variable. Nevertheless, this model was not used as it performs poorly. The marginal r-squared stays below 0.100 reaching 0.082 and the diagnostic plots emphasize that poor performance. A model consisting of the interaction of *soil type* and *treatment* could solve the problems and reached a marginal r-squared of 0.224

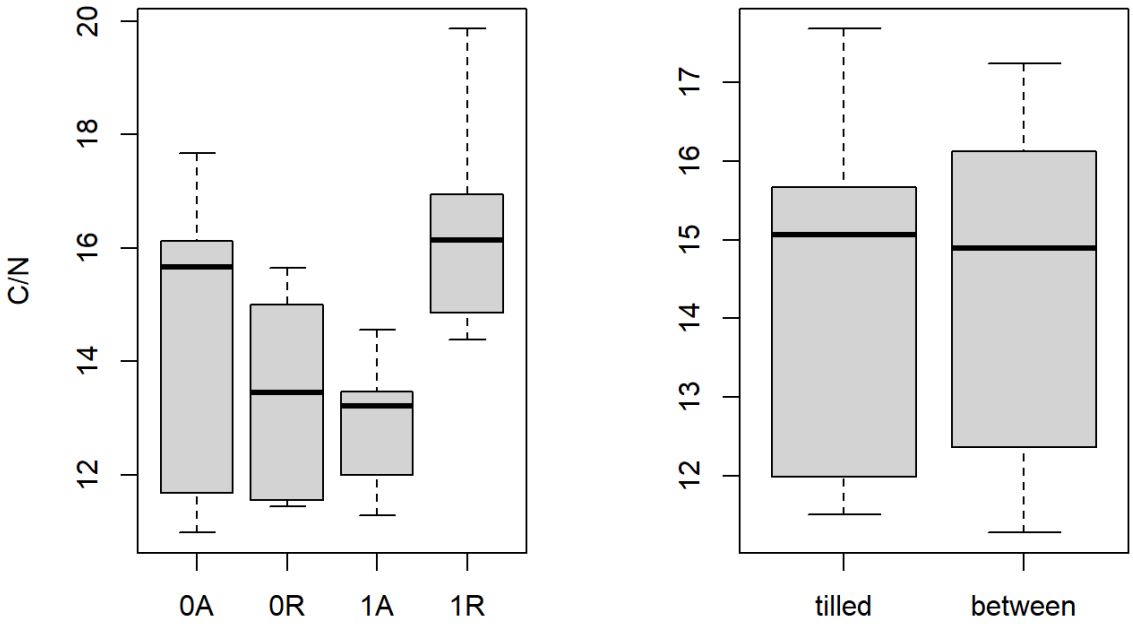

Figure 20: C/N ratios from 0-20 cm mineral soil depth; left: C/N ratios for soil type and treatment (0 – Fluvisol; 1 – Rendzic Leptosol; A – area treated ≥5 years ago; R – mature forest); right: Differences in C/N ratios for tilled strips and areas in between (p = 0.901)

Table 15: Output of LMM describing C/N ratio with soil, treatment and their interaction (treatmentR:soil1) from 0-20 cm mineral soil depth; Rendzic Leptosol=soil1; mature forest stand=treatmentR; R2m = marginal R2, R2c = conditional R2, VIF = Variance Inflation Factor

Formula: CN~treatment*soil+(1|plot/subplot)

Random effect	**Variance**	**sd**				
subplot:plot	0.745	0.863				
plot	4.275	2.068				
residuals	0.823	0.907				
Fixed effects	**Estimate**	**Std. Error**	**95% CI**	**df**	**t value**	**p**
Intercept	14.03	1.536	[11.274, 16.774]	2.068	9.134	0.011
treatmentR	-0.602	0.485	[-1.572, 0.323]	16.784	-1.242	0.231
soil1	-1.22	2.172	[-5.105, 2.672]	2.068	-0.562	0.629
treatmentR:soil1	4.181	0.686	[2.863, 5.540]	16.784	6.097	1.25E-05
R2m	0.224	VIF	treatment	soil	treatment:soil	
R2c	0.891		2	1.018	2.018	

(conditional r-squared 0.891) (tab. 15). Figure 20 (left) shows the interaction of *soil type* and *treatment* well. Especially the reference plots in Rendzic Leptosols stick out, showing that the interaction is highly significant. Confidence intervals shows a clear trend. Mature forest stands in Rendzic Leptosols have a C/N ratio by 4.181 higher than the intercept. Further trends of soil type or treatment are not clear, and their confidence interval includes zero. A tendency of mature forest stands and Rendzic Leptosols having a slightly lower C/N ratio (apart from the combination of Rendzic Leptosol and mature forest site) can be reported too. As the VIF shows, collinearity is no problem in the model. All the other assumptions are met as well. There is a slight deviation from the ideal caused by one outlier. But as the goal is not prediction but explanation, this shouldn't be an issue.

3.4.3. Nitrate in the soil solution

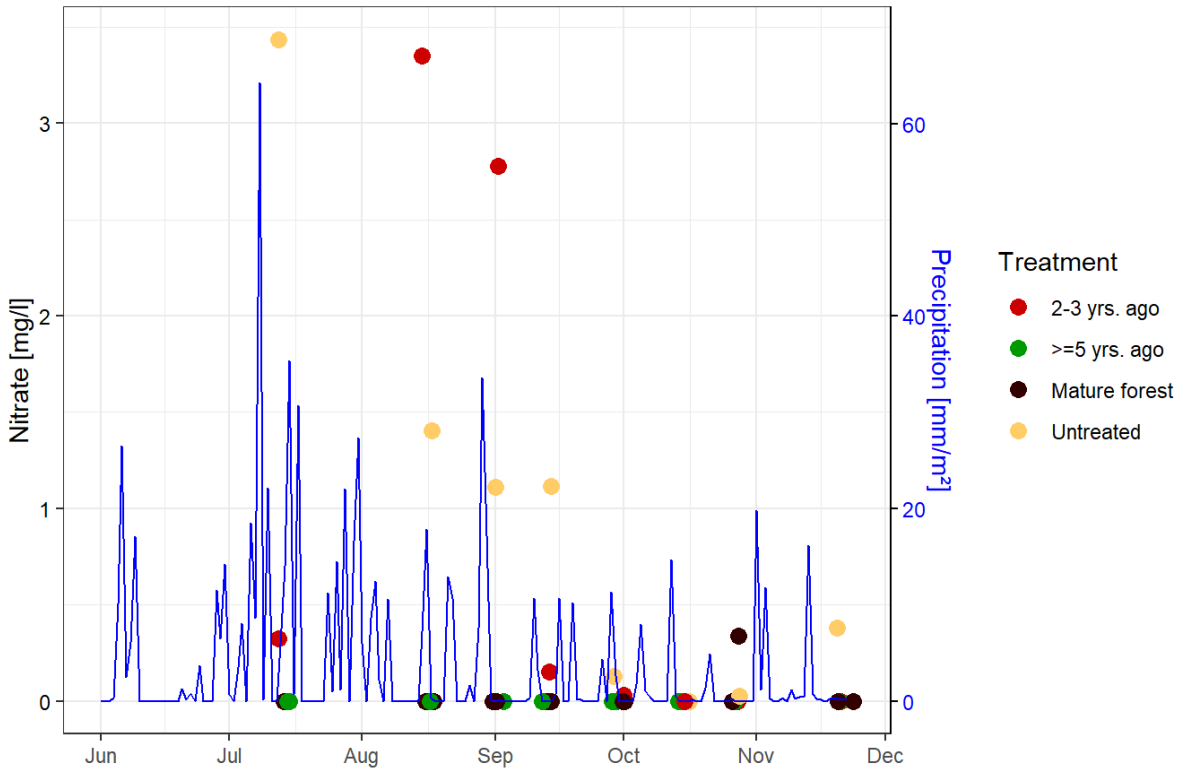

Figure 21: Nitrate concentrations in soil leachate in 60 cm depth by date and treatment for Fluvisols, slight jitter on x-axis improves visibility of data points.

Partly, there are missing points in the graphs due to insufficient water collecting efficiency. The results of the measurement of the lysimeter water samples for nitrate show different trends for the two soil types. It seems like the nitrate leaching recovers for Fluvisols within five years. The nitrate levels at the plot treated ≥5 years ago are on the same level as for the mature forest. Generally, the values are low with a maximum of 3.44 mg/l (fig. 21).

In Rendzic Leptosols this trend of recovery is not visible. The plot treated ≥5 years ago is on a different level than the mature forest. Nitrate values for the mature forest site are often higher than the ones for areas treated 1 year or 2-3 years ago. Striking is that the nitrate values are a lot higher than for Fluvisols. The maximum concentration here is 23.41 mg/l (fig. 22).

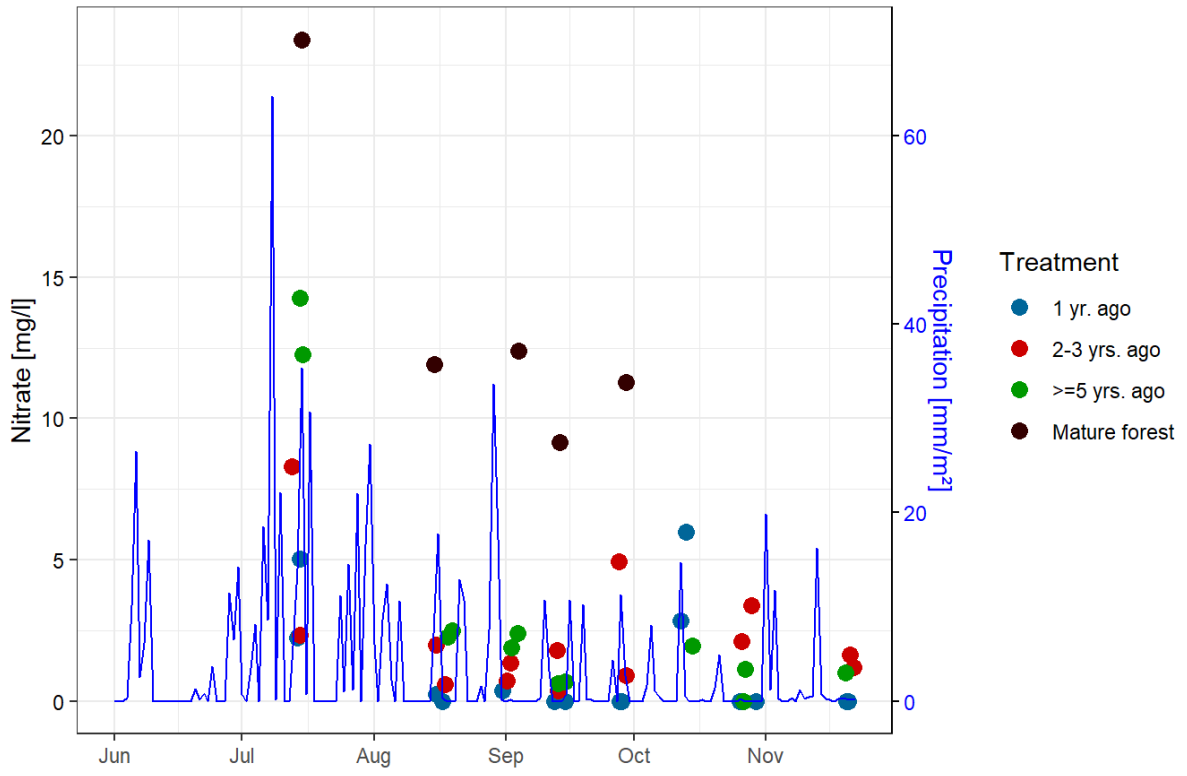

Figure 22: Nitrate concentrations in soil leachate in 60 cm depth by date and treatment for Rendzic Leptosols, slight jitter on x-axis improves visibility of data points.

3.5. Physical parameters

The bulk density in the sampling area is higher for Fluvisols 0.84±0.09 g/cm³ compared to Rendzic Leptosols 0.64±0.17 g/cm³ in the upper 20 cm. No significant differences between tilling strips and areas in between could be measured in the upper 20 cm ($p = 0.401$). Nevertheless, it can be seen that the area between tilling strips has a higher median of bulk density (fig. 23). Reafforested plots show higher bulk densities than mature forest stands. This effect is stronger in Rendzic Leptosols (tab. 17). Nevertheless, this effect is also visible in Fluvisols but only from 10-20 cm. In the upper 10 cm no difference can be detected (fig. 24). As *plot* did not reach statistical significance in the model calculated for bulk density, a simple LM was preferred. The explanatory variables *soil type* and *treatment* reached statistical significance and could explain 43.4% of the variation in our data (tab. 16). Soil type has a strong effect. Rendzic Leptosols have on the average a 0.202 g/cm³ lower bulk density than the intercept. The effect of treatment is not as strong. Mature forest stands have a 0.104 g/cm³ lower bulk density. But here the upper limit of the confidence interval almost reaches 0. Thus, it can not fully be excluded that the effect of treatment is much weaker (or stronger). Collinearity was no problem in this model and residuals were normally distributed. Homogeneity and linearity assumptions were fulfilled. Again, the diagnostic plot shows minor deviations, but they are not strong enough to reject the assumptions.

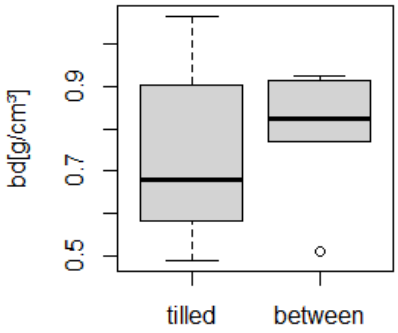

Figure 23: Bulk density (0-20 cm soil depth) for tilling strips compared to areas in between tilled strips. Difference not significant (p=0.401)

Figure 24: Comparing bulk density of mature forest stands (R) and reafforested areas (A) for the two soil types

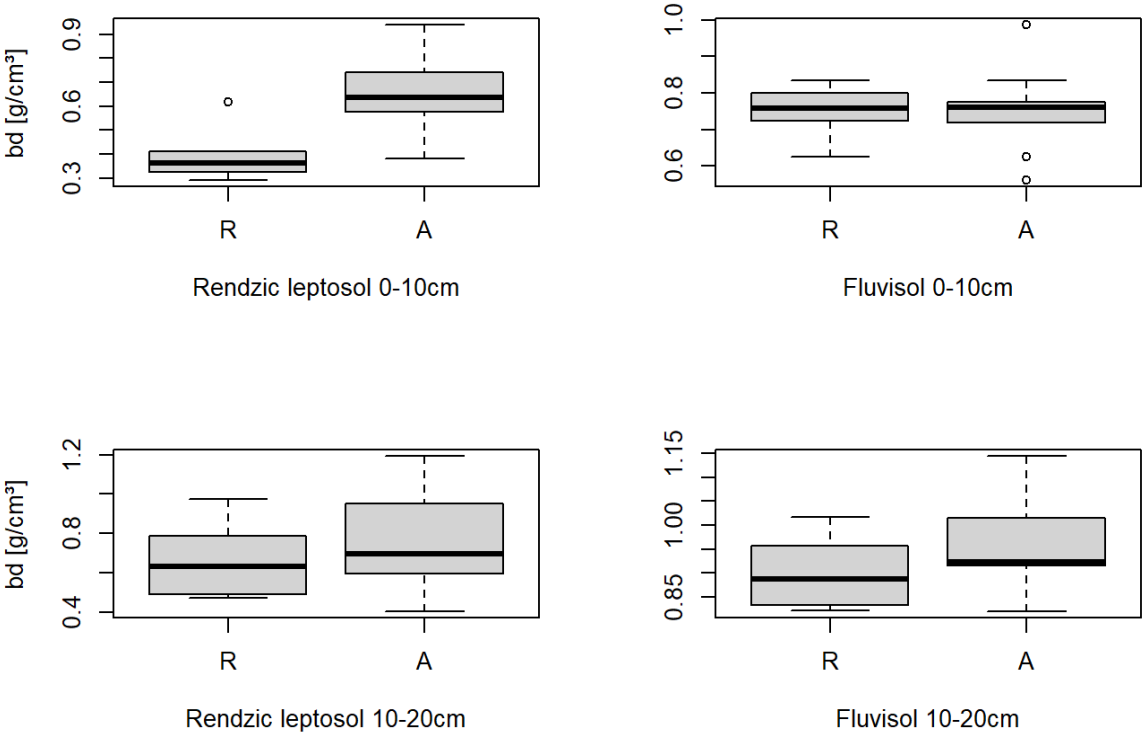

Table 16: Output modelling bulk density (bd) 0-20 cm soil depth, Rendzic Leptosol=*soil1*; mature forest stand=*treatment,* adj. = adjusted, VIF = Variance Inflation Factor

Formula: bd~soil+treatment

	Estimate	Std. Error	95% CI	t value	p
Intercept	0.883	0.038	[0.806, 0.960]	23.474	<2E-16
soil1	-0.202	0.046	[-0.297, -0.108]	-4.391	<0.001
treatmentR	-0.104	0.047	[-0.201, -0.008]	-2.221	0.035
R2 (adj.)	0.434	p-value	0.000177		
VIF	Treatment	Soil			
	1	1			

The differences between Fluvisols and Rendzic Leptosols for AWC and ln(ksat) are apparent. For different treatments it is less obvious. The mean of the total AWC (AWC_{tot}) was 110.18±9.18 mm for Fluvisols and 41.81±9.75 mm for Rendzic Leptosols. Outliers were not removed as they were meaningfully representing the natural variety of the soils. The lowest value for AWC_{tot} in Fluvisols was found at the shallowest plot there. Two outliers, the maximum and minimum, occurred in ln(ksat). The plot with the highest value has a comparably low bulk density and consistently a high sand content while the lowest point has a lower sand content and relatively high bulk density. Possibly this point was located in an old rut that was no longer optically visible. Further details can be found in table 17. Tilling strips did not show significant effects on AWC (p=0.979), but ln(ksat) reached marginal significance with a p-value of 0.067 (fig. 25). For modelling AWC, AWC_{30} (Available Water Capacity in mm for the upper 30 cm) was used to make the two soil types more comparable. As first attempt, a LMM was used to model AWC as *plot* reached significance. Using this approach, the best model consisted of the random nested effect *plot* and the fixed effects *soil* and *treatment* and their interaction. *Soil* and the interaction reached significance and the model had a marginal r-squared of 0.898 and a conditional r-squared of 0.969. Diagnostic plots showed that the assumptions for this model were not fully met. The combination of mature forest and Rendzic Leptosol reached significance. This combination has a higher coarse fragment than the other sites and coarse fragment and AWC are highly correlated. Thus, another approach was tested using the original input parameters and *coarse fragment* to model AWC. With this approach, a LM with only *coarse fragment* as explanatory variable reached significance. It shows a clear negative trend between coarse fragment and AWC reaching a r-squared of 0.975 (fig. 26). This model fulfilled the assumptions. More details on the coarse fragment content can be found above in tab. 1 and fig. 9.

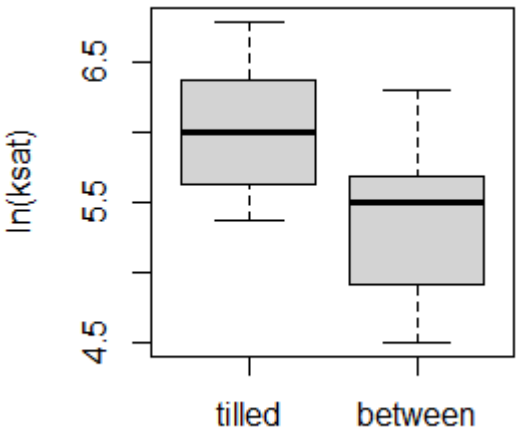

Figure 25: Differences in the natural logarithm of the saturated hydraulic conductivity (ln(ksat)) (0-30 cm soil depth) for tilled strips and areas in between (p = 0.067)

Table 17: Overview physical parameters; Bulk density (bd) in g/cm³ 0-20 cm, Available Water Capacity in mm for the upper 30 cm (AWC_{30}), total Available Water Capacity (AWCtot) in mm, saturated hydraulic conductivity in mm/d (ksat) 0-30 cm, number of sampling points (n), 0 – Fluvisol, 1 – Rendzic Leptosol, A – area treated ≥5 years ago, R – mature forest.

Soil type	Treatment	bd	AWC_{30}	AWC_{tot}	ksat	n
Both	Both	0.74±0.17	58.16±18.05	76.00±35.99	422.82±356.09	30
0	Both	0.84±0.09	74.50±2.68	110.18±9.18	233.85±88.56	15
1	Both	0.64±0.17	41.81±9.75	41.81±9.75	611.80±422.25	15
0	A	0.85±0.11	74.67±2.86	107.25±8.08	226.77±64.87	9
0	R	0.82±0.06	74.25±2.62	114.59±9.64	244.47±122.48	6
1	A	0.71±0.17	46.77±7.36	46.77±7.36	446.78±243.35	9
1	R	0.53±0.10	34.37±8.30	34.37±8.30	859.32±530.97	6

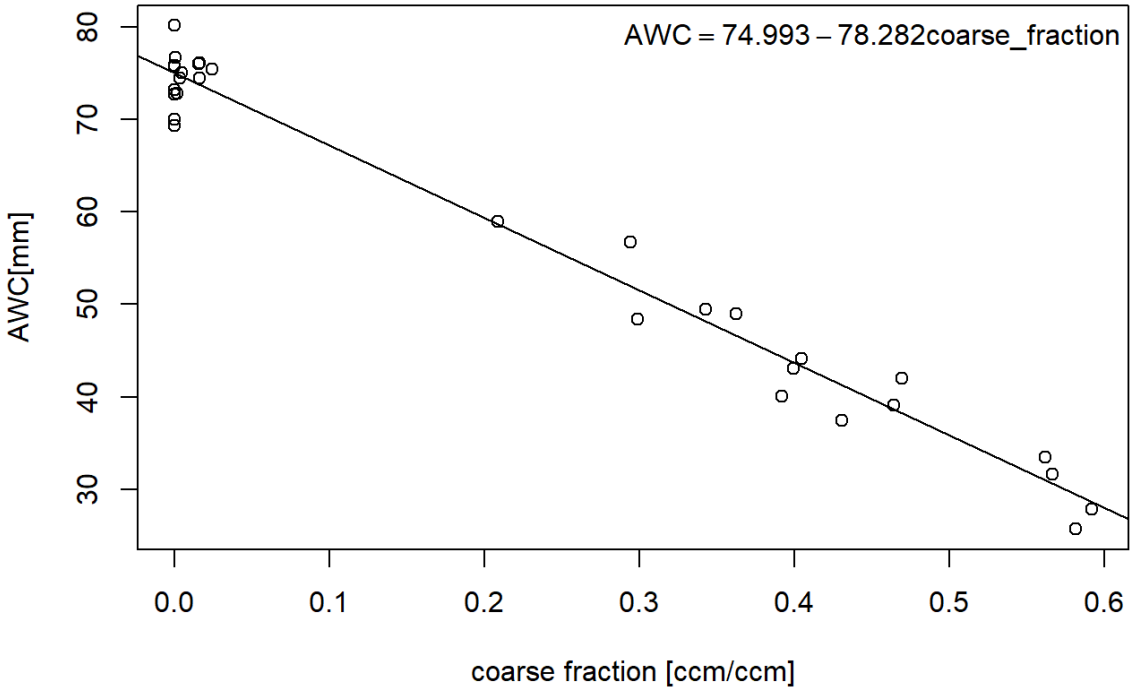

Figure 26: Linear model of Available Water Capacity in the upper 30 cm (AWC) with the explanatory variable coarse fraction, r-squared 0.975

ln(ksat) was modelled using a LM including soil type as explanatory variable (tab. 18). *Treatment* and the interaction of *treatment* and *soil types* both reached marginal significance in two separate models. The strongest effect is the higher ln(ksat) for mature forest stands in Rendzic Leptosols (fig. 27), but in both cases the confidence interval includes zero. The final model only including significant parameters reached a r-squared of 0.34. Rendzic Leptosols have a ln(ksat) by 0.818 higher than Fluvisols. This shows a clear effect of soil type on ksat. Model assumptions were met.

Table 18: Output of linear model describing the natural logarithm of saturated hydraulic conductivity (ln(ksat)) 0-30 cm, Rendzic Leptosol=soil1, adj. = adjusted

Formula: ln(ksat)~soil					
	Estimate	Std. Error	95% CI	t value	p
Intercept	5.391	0.145	[5.094, 5.688]	37.195	<2E-16
soil1	0.818	0.205	[0.398,1.238]	3.989	0.0004
R2 (adj.)	0.340	p-value	0.000432		

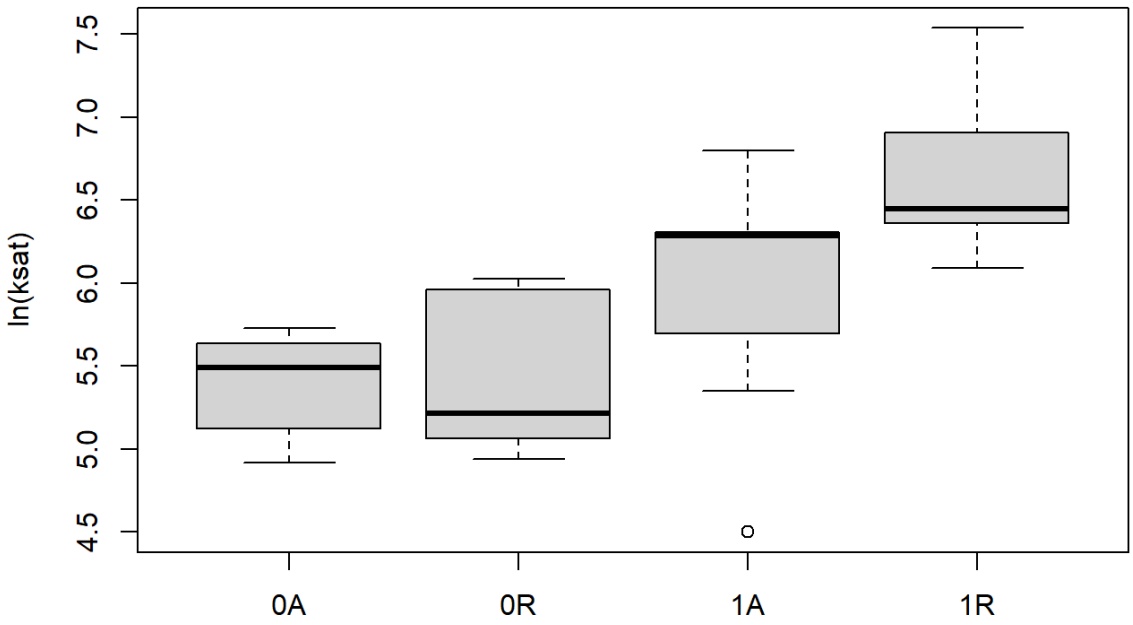

Figure 27: Natural logarithm of saturated hydraulic conductivity (ln(ksat)) for soil type and treatment from 0-30 cm (0 – Fluvisol; 1 – Rendzic Leptosol; A – area treated ≥5 years ago; R – mature forest)

4. Discussion

In the discussion, the aim is to answer the initially asked research questions by linking the findings of this work with existing literature. First, it will be elaborated which indicators are thought to be suitable for describing soil functions and which ones should be additionally integrated or excluded. After that, it will be assessed if soil functions over all recover within five years and finally if differences between tilled strips remain different from the area between strips for more than five years.

4.1. Indicators for soil functions

The use of a distance-based approach for the analysis of multivariate vegetation data yielded good results emphasizing why this approach is commonly used in ecology (Warton et al., 2012). NPMANOVA is mainly focusing on differences in location but also sensitive for dispersion effects. This isn't considered to be problematic as the goal is the detection of environmental impacts of the treatment (Anderson, 2001). In this respect, recognizing strong dispersion effects seems rather desirable. The method was sensitive to the changes in vegetation communities over the five years of our chronosequence. Detection of differences in communities is key in the assessment of impacts of treatment on vegetation and habitat function (Haeussler et al., 2002) and ecosystem processes relevant for soil functions such as energy and material fluxes (Chapin III et al., 2000). Species composition is more sensitive to disturbances than biodiversity indices like species richness (Haeussler et al., 2002). Hurlbert (1971) warns that the importance of a species is not automatically reflected by its relative contribution to the Shannon index. Thus, biodiversity indices give us only impressions of the ongoing dynamics in the ecosystem. They can provide valuable information but leave out the important composition aspect. Hence, it can be concluded that they should not be used as sole parameter to assess the impact of management on ecosystems. This is in agreement with several other authors (Aubert et al., 2003; Haeussler et al., 2002; Šebesta et al., 2021). The same goes for the Ellenberg indicator values. They provide a valuable orientation but they should as Ellenberg et al. (1992) himself stresses, never replace measurements. The indicator values only reflected strong trends in our data, i.e., only the light indicator value showed marginal significant effects of treatment. Ellenberg et al. (1992) warns that especially nitrate indicator values are under pressure as high N emissions in central Europe cause the loss of the extremes making the indicator less meaningful. He also suggests doing a site-specific calibration of indicator values. In this work it was not applicable as this would have exceeded the scope of this work. Assessing specific species of interest provide similar insights in soil functions as described for community composition above. The evaluation of the *Solidago gigantea* cover provided valuable information and allows drawing conclusions on the habitat function as well as other soil functions linked to the plant community. In our work *Solidago* cover provided a parameter reacting to treatment. Other papers reported that *Solidago* covers (Hall et al., 2022) or neophyte covers (Šebesta et al., 2021) respond to management practices as well. An additional sampling of the summer aspect of vegetation could potentially show the suppression effect of *Solidago* covers on other vegetation better than the sampled spring aspect (Petrášová et al., 2013). Nevertheless, the sampling of the spring aspect provided more information on the site as more species could be documented. The *Solidago* cover and the mowing would have made a further sampling of the summer aspect very laborious and methodically challenging as not all areas are mowed. It would have exceeded the scope of this work. Also, simply recording only the *Solidago* cover before mowing in summer provides sound and easily available information on the dominance of *Solidago* in the area. Another parameter one could consider for the assessment of the habitat function in larger works is the presence of rare or endangered species (Haeussler et al., 2002; Šebesta et al., 2021). This would require larger sampling areas in order to gather enough data points to make a statistically rounded

analysis of the data. It is also uncertain how sensitive this parameter is as Šebesta et al. (2021) did not record any relationship between the presence of endangered species and treatment.

As described in the introduction, an assessment of earthworms could have been interesting and might have provided further insights. Unfortunately, it was not feasible to include it in this work. An inclusion of pH, texture and a complete recording of soil profiles proved to be very useful for explaining and discussing trends observed in our data.

Microbial biomass is supposably a more sensitive parameter to indicate changes in soil quality compared to SOC (Bastida et al., 2008; Li et al., 2016). In our study microbial biomass did not react to soil type or management. This is somewhat consistent with literature where varying and indifferent responses of microbial biomass to management are reported (Frey et al., 2009; Holden & Treseder, 2013; Smolander et al., 2000; Zuber & Villamil, 2016). Bacterial community assessments are more sensitive to management impacts such as compaction (Frey et al., 2009). As this is a very resource intensive parameter to measure, it was not included in this work. Eco-physiological ratios are a good option to extract further information from primary parameters (Zechmeister-Boltenstern et al., 2005). Our results agree with this statement as eco-physiological parameter revealed significant results where only microbial biomass didn't. Our results suggest that the Cmic/Corg, Nmic/N_{tot} and Nmic/Corg ratios yield similar results. Due to the tight link of N to organic matter (Morari et al., 2015; Schulten & Schnitzer, 1998), they all represent a ratio of microbial biomass and organic matter. Therefore, the Cmic/Corg ratio was preferred as it is relatively well established in literature allowing conclusions on the SOC pool and ongoing processes (Insam & Domsch, 1988; Malý et al., 2002; Zechmeister-Boltenstern et al., 2005). The Cmic/Nmic ratio also seems to be a good parameter as it presents a simple way of rudimentarily assessing microbial communities. Different C/N ratios in fungi and microbes allow drawing conclusions on the ratio of fungi to microbes (Paul, 2007).

All ecological soil functions are influenced by SOM (Blume et al., 2010; Morari et al., 2015; Schoenholtz et al., 2000). Consequently, SOM has to be included in a soil function assessment. Carbon and Nitrogen are tightly connected to SOM (Morari et al., 2015; Schulten & Schnitzer, 1998). The C/N ratio is a quality indicator of SOM (Morari et al., 2015) determining its mineralisation (Schulten & Schnitzer, 1998). A separate full assessment of nitrogen could be omitted as information on SOM is provided using only SOC and C/N which already represents nitrogen availability. The use of SOC stocks allowed us to directly show the amount of carbon stored in the mineral soil. In this work SOC (and N) stocks were calculated with the commonly applied fixed depth approach. It is affected by bulk density (von Haden et al., 2020). This turns SOC stock into combined parameter reflecting SOM and bulk density changes induced by the treatment. This is accounted for in the discussion. For further analysis, the author would recommend considering the use of the equivalent soil mass method for the determination of SOC stocks (von Haden et al., 2020; Wendt & Hauser, 2013). This method shows SOC stocks without the effect of soil compaction between treatments. If a full assessment of the carbon storage is desired, a quantification of living and dead biomass and forest floor as well as deeper soil layers would be necessary. The focus of this work was not only the carbon storage function. Therefore, the reduced sampling of only the upper mineral soil seems justified. The measuring of nitrate seepage improves the understanding of the ongoing processes in the SOM dynamic. It allowed us to draw conclusions on nitrogen losses linked to fertility of soils and most importantly the filter function of soils.

Soil compaction was identified as one of the major threats to soil functions (Schjønning, et al., 2015). Including bulk density as measurement for soil compaction (Rabot et al., 2018; Schjønning, et al., 2015) is necessary to assess soil functions, especially when the investigated treatment is commonly known to increase bulk density (Amaranthus et al., 1996; Aust et al., 2004; Frey et al., 2009). In this work, changes in bulk density from mature forest stands to reafforested plots could be documented, stressing the importance of this parameter. Including hydrological parameters calculated from pedotransfer functions might reveal impacts not revealed by single parameters. They combine bulk density and SOC content (both influenced by treatment) with texture. Especially in smaller projects

with limited founding, this provides a valuable alternative to directly measuring hydraulic parameters (Wösten et al., 1999). Nevertheless, driving associated with logging and site preparation also affects pore continuity (Schjønning, et al., 2015). Gent Jr. et al., (1984) points out that site preparation degrades soil structure. Site preparation can affect ksat positively or negatively depending on the used method and the pore continuity (Aust et al., 2004). These processes and effects can not be described accurately by pedotransfer functions. Therefore, the use of directly measured hydraulic parameters is suggested if project resources permit it. Alternatively or additionally, an assessment of the pore network could be included due to its great relevance for soil functions (Rabot et al., 2018). The pore network also provides information on hydraulic parameters (Ares et al., 2005; Aust et al., 2004; Gent Jr. et al., 1984; Jensen et al., 2020), making this potentially a superior physical soil function indicator if resources for measurement are available. An overview of tested indicators can be found in tab. 19.

Table 19: Overview evaluation of indicators of soil functions, PTF = pedotransfer function, + = suitable indicator, ~ = indifferent indicator with limited information

Indicator	Soil function	Evaluation
Vegetation composition	Habitat function, regulatory function, productivity function	+
Vegetation cover of taxa of interest	Habitat function, regulatory function, productivity function	+
Biodiversity indicators	Habitat function, productivity function	~
Ellenberg indicator values	Provides information on many soil functions, but high uncertainties, not suitable as single parameter	~
Microbial biomass, eco-physiological ratios	Regulatory function (carbon cycle), habitat function	+
SOM (SOC stock and C/N ratio as parameters)	All ecological soil functions	+
Nitrate in soil solution	Regulatory function (filter function)	+
Bulk density	Indicates compaction as major threat to soil functions	+
Hydraulic parameters from PTFs	Regulatory function (high uncertainties)	~

4.2. Soil function recovery

As expected, the combined treatment of clearcut and site preparation significantly alter the species composition (Bock & Van Rees, 2002; Haeussler et al., 2002; Vanha-Majamaa et al., 2017) in our sampling area as well. The difference between natural disturbance management, e.g., a cleaned up windthrow and natural succession following a wind throw can be detectable for decades (A. Fischer et al., 2002). Thus, it is expectable to find differences between fallow areas and reafforested plots. The relatively small explanatory power of fallow areas can probably be explained by the assorted characteristics of vegetation communities on these plots. Some plots were fully covered in shrubs or dense grass, others had high *Solidago* covers assumably depending on age and formation shaped by the decline of (pre)-mature forest stands due to disturbances. A direct comparison of processes in treated and untreated plots thus seems questionable. The largest distances from the population were recorded for areas treated 2-3 years and ≥ 5 years ago. This confirms the theory that the herbaceous

layers increase in their difference from mature forest stands until canopy closure (Aubert et al., 2003; Haeussler et al., 2002). Modelling by A. Fischer et al. (2002) suggest that species composition takes up to 100 years to recover in Central European forests dominated by spruce. This underlines that a "recovery" of vegetation from disturbances before canopy closure can not be expected (Aubert et al., 2003).

Floodplain forests are often invaded by neophytes (Bergmann & Rak, 2006; Hall et al., 2022; Pfundner et al., 2012; Schnitzler et al., 2007; Šebesta et al., 2021). Our results confirm this. The most important neophyte in the sampling area is *Solidago canadensis agg.* with occurrences in most plots not shaded by a tree or shrub layer. Even some mature forest stands had low covers of *Solidago* showing the plants ability to establish populations in forest habitat without disturbances (Petrášová et al., 2013). As *Solidago canadensis agg.* is a species with high light demand (Info Flora, 2020) it can be expected that densities will increase as soon as the canopy is disturbed. Exotic species seem to be promoted by disturbances in canopy and soil more than native species (Petrášová et al., 2013; Schnitzler et al., 2007). The presence in the seed bank allows *Solidago gigantea* a quick colonialization of new habitats if the canopy clears (Schnitzler et al., 2007). High densities of *Solidago* cover on the plots treated 2-3 years and ≥ 5 years ago underline this statement by Schnitzler et al. 2007. Additionally, *Solidago* covers can be promoted by site preparation (Haeussler et al., 2002; Hall et al., 2022; Petrášová et al., 2013; Šebesta et al., 2021) also via the distribution of rhizome fragments of existing stands (Hall et al., 2022; Info Flora, 2020). The higher densities of *Solidago* in treated areas compared to fallow areas could suggest a similar trend on our site. Nevertheless, *Solidago* cover also reached values up to 95% in fallow area. It seems that shade induced by shrubs on fallow areas and dense initial grass covers halts or slows the spread of *Solidago canadensis agg.* most effectively. But the spread of *Solidago* species is not dependent on intensive forestry, as their presence in the Donau-Auen National Park shows (Bergmann & Rak, 2006; Pfundner et al., 2012). Clonal growth via rhizomes can form dense stands starting from an initially small spot (Jakobs et al., 2004). This appears to be the case at our site as well. Younger afforested areas have a patchier less dense *Solidago* cover which increases with age. Dense *Solidago* covers do not only adversely affect the natural species composition (Jakobs et al., 2004; Petrášová et al., 2013) but also strongly impact survival of planted tree seedlings and saplings (Hall et al., 2022). To mitigate these effects, the forest managers started seeding a grassland mixture and mowing reafforested sites twice a year before the flowering of *Solidago*. Mowing *Solidago* species twice a year is a commonly used practice to control populations (Info Flora, 2020) which proved to be very effective on similar sites as ours in floodplain forests along the Danube (Hall et al., 2022). Hall et al. (2022) state that the establishment of strong competing species can help reducing *Solidago* covers as well. In our sampling area the mitigation measures are only put into effect consistently for the plots treated 1 year ago. Plots treated 2-3 years ago are in large parts managed the same way. Only plots ≥ 5 years are mainly no longer managed for *Solidago* cover. The plots treated ≥ 5 years ago were partly managed differently as the treatment described above was only started in 2015. Also, the saplings of this age group already escaped the competing *Solidago* cover ensuring their survival. However, a potential long-term treatment keeping up the mowing until removal of *Solidago* species or canopy closure could achieve a permanent reduction of *Solidago* cover. Such a treatment would most likely exceed the resources of a small forestry company and require the involvement of further stakeholders. When canopy closes, persisted shade makes less shade tolerant species such as *Solidago canadensis agg.* regress (Aubert et al., 2003). Šebesta et al. (2021) show that the cover density of non-native species decreases after canopy closure. Hence, a decline in *Solidago* covers can be expected with canopy closure.

Biodiversity indicators did not show significant trends. Yet, the phenomenon of the smallest species richness in mature forest stands described in literature (Aubert et al., 2003; Haeussler et al., 2002) can be observed as a trend in our data too. Unlike the observation made by Haeussler et al. (2002) and Šebesta et al. (2021) in areas also affected by neophytes, in this work no increase in species richness until canopy closure could be observed. The youngest area had the highest species richness. Older plots have a decreasing trend. This might suggest that the formation of a "canopy" by dense *Solidago*

covers has a similar effect as the canopy by trees in mature forest stands on other plants. *Solidago gigantea* is known to form dense stands that limit other species (Jakobs et al., 2004; Petrášová et al., 2013). In fallow areas "canopies" are partly formed by *Solidago* or shrubs. The negative correlation trend of species richness and *Solidago* cover might be more pronounced for the summer aspect of vegetation (Petrášová et al., 2013). Potentially this could even overwrite the trend caused by canopy closure in mature forests. In stands with high *Solidago* covers, low numbers for Evenness and Shannon index can be expected. Our data vaguely shows this trend with the lowest Evenness values for areas treated 2-3 years and ≥ 5 years ago. Mature forest stands have relatively high Evenness values, although in older stands a higher degree of organization and thus lower Evenness values can be expected (Aubert et al., 2003). This might be a hint that the disturbance driven decline in the remaining patches of mature forest stands is already showing in the vegetation. Ellenberg indicator values detected significant trends distinguishing the two soil types regarding moisture, nitrogen and light. Regarding the effect of treatment, only light reached marginal significance. The higher light value for younger treatments is only logical. More interesting is the non-significant trend observed regarding the nitrogen value describing the soil fertility. Here a tendency of the same recovery trend as for nitrate leaching can be observed.

Answering the question if the vegetation could recover within five years from site preparation (without considering harvesting and planting) is challenging. In a sampling design where site preparation was only performed on parts of the plots Šebesta et al. (2021) could show a significant impact of site preparation on species composition. Contrary to this, Haeussler et al. (2002) did not find any effect of site preparation concerning species richness and species composition was only altered when using severe site preparation methods. Similar results were found by Bock & Van Rees (2002), who showed that sites with soil preparation did not separate well from clear-cut areas using a canonical correspondence analysis. Only the use of shearblading had a larger impact. Shearblading is similar to the treatment used prior to the current combination of mulching and tilling until 2012. The design of this work unfortunately could not include sites that were afforested but not prepared simply due to the fact that such sites were not available. Consequently, mature forests, reafforestation plots and fallow areas had to be compared. As Haeussler et al. (2002) summarizes the comparison of pre- and post logging areas can only provide insights in ongoing vegetation dynamics. For the separation of the human interference from the natural disturbance, often not available stands in the respective successional state would be needed. In this case, additional plots to separate different human interactions (clearcutting and site preparation) from each other would be needed. Changes in management make a statistical detection of effects even more difficult. This said, no clear evidence that site preparation altered the habitat function for vegetation was found. Isolating site preparation as single factor in this complex situation of natural disturbances, harvesting, site preparation, increasing population of *Solidago* and the interaction of the factors does not seem justified. Moreover, literature suggests that the current method is an improvement regarding its impact on the habitat function (Bock & Van Rees, 2002). Shearblading or the removal of the organic layer promotes ruderal species including neophytes. Generally speaking finding management methods that promote regeneration of native species without promoting neophytes is a challenging task (Haeussler et al., 2002). In addition, forest management has to account for changing site conditions (climate, hydrological regime) and the conditions for natural regeneration of adapted tree species ('mother trees' in sufficient quantity and spacing) are frequently not given. Considering the success of plantation activities (sufficient survival rates, vigor, and height of the saplings), the habitat function for trees is most likely positively affected by site preparation.

Contrary to the assumption that treatment would reduce the microbial biomass, no effect of treatment was found. Despite showing that harvesting (Holden & Treseder, 2013) and tilling (Zuber & Villamil, 2016) mostly lead to a reduction in microbial biomass Holden & Treseder (2013) stress that the variance in their metanalysis was relatively large with some data points even showing increases. This would suggest a relative fast recovery of the microbial biomass compared to values of 20 years reported in other studies (Holden & Treseder, 2013). The values reported are in a similar range as the

ones reported by (Malý et al., 2014). Eco-physiological ratios showed effects. The microbial C/N ratios are generally low suggesting a relatively high share of bacteria in the microbial biomass (Paul, 2007). A first model for microbial C/N ratio showed that Rendzic Leptosols have higher ratios. This suggests a higher share of fungal biomass on Rendzic Leptosols compared to Fluvisols (Paul, 2007). The pH in Rendzic Leptosols is lower than in Fluvisols and as the fungal/bacterial ratio decreases with increasing pH (Bååth & Anderson, 2003). This would support the theory that Rendzic Leptosols have a larger fungal share in their microbial biomass. This is also reflected in the accumulation of a forest floor and the humus form moder in mature spruce stands on Rendzic Leptosols. A second model aiming at treatment effect on different soil types revealed an effect of treatment in Rendzic Leptosols but yielded no other results indicating further effects of soil type or treatment. Higher microbial C/N ratios were found in mature forests in Rendzic Leptosols suggesting fungal/bacterial ratio shift after treatment. Again, this could be related to the slightly lower pH in mature forest stands (by 0.1) (Bååth & Anderson, 2003). More likely though, this is triggered by the changed litter input and an activation of the humus layer by succession and disturbances leading to more active humus layers after treatment. Changes in the humus layer can be expected as soil life is tightly connected to it (Graefe & Beylich, 2006; Ponge, 2003). Mull humus is associated with a higher share of bacterial biomass than moder (Ponge, 2003). This aligns well with the observed trends. As expected, treated areas had higher Cmic/Corg ratios. Despite the model showing that the treatment affects the ratio highly significant the data suggests that a second model is more appropriate as the effect of treatment is concentrated in Rendzic Leptosols. Mature forests in Rendzic Leptosols had lower Cmic/Corg ratios. In Fluvisols no effect was detected. Clay and pH also affect the Cmic/Corg ratio (Insam & Domsch, 1988). While the differences between treatments for pH is relatively low, clay contents are noticeably higher in treated areas in Rendzic Leptosols. In Fluvisols, clay content was more homogeneous. Additionally, forests dominated by deciduous tree species have higher Cmic/Corg ratios than stands with a high percentage of coniferous species (Malý et al., 2014). Consequently, the shift to more deciduous species might also cause higher ratios. Rendzic Leptosols seem to be more sensitive to these changes. The shifts in humus forms in Rendzic Leptosols likely affects the Cmic/Corg ratio as well due to the tight connection of soil microbes with humus forms (Ponge, 2003). Overall, the Cmic/Corg ratios measured (an average of 28mg/g for Fluvisols and 26 mg/g for Rendzic Leptosols) are very similar to those reported for floodplain forests (mean value 25-27 mg/g) (Zechmeister-Boltenstern et al., 2005). These values are relatively high which might indicate carbon accumulation on our sites (Insam & Domsch, 1988; Malý et al., 2002) with the highest rate for reafforested plots on Rendzic Leptosols. In spite of that, high Cmic/Corg ratios do not necessarily imply carbon accumulation in alluvial forests. Large amounts of bacterial biomass might occur as R-strategic organisms quickly decompose litter (Zechmeister-Boltenstern et al., 2005). The high share of bacteria in Cmic is also suggested by the microbial C/N ratio. Also, regular floodings halt succession which also impacts carbon sequestration (Zechmeister-Boltenstern et al., 2005). Nevertheless, there might be carbon accumulation on the plots. The floodplains were cut of from riverine water and the last flooding happened approximately 90 years ago (Fulterer, 2010). Thus, succession is not halted by regular flooding anymore, potentially leading to higher carbon sequestration of soils. Based on the information above it can be concluded that microbial biomass recovers (or less likely is not affected) for both, Fluvisols and Rendzic Leptosols. It is plausible that the observed differences in eco-physiological ratios in Rendzic Leptosols can be mainly explained by the shifts induced by the changes in humus form due to forest regeneration (Ponge, 2003).

The prior expectation that SOM would simply be lower at treated sites could not be shown in our data. Modelling SOC stocks using soil type and treatment as explanatory variables described the situation very well reaching an R2 of 0.756. Overall, the reported SOC stocks of 67.7 t/ha for the upper 20 cm are above average for European forests (De Vos et al., 2015). Higher SOC stocks for Fluvisols compared to Leptosols were reported by De Vos et al. (2015) as well. Interpreting the impact of treatment is less straight forward. The SOC stock in the upper 10 cm of Rendzic Leptosols is very similar among treatments. In the lower 10 cm it is visibly higher for afforested plots. In Fluvisols SOC stocks are higher

throughout the profile. Yet, there is a different trend for SOC content [%] compared to SOC stocks. Sandy soils are especially sensitive towards disturbances (Carlyle, 1993) and react stronger to altered biomass inputs (Oliveira et al., 2018). More finely textured soils can better protect SOM physically (Carlyle, 1993; Oliveira et al., 2018) and are thus more resilient to disturbances. This is reflected in our study, where the sandier Rendzic Leptosols had a reduced SOC content (-33%) after treatment, while the SOC content for Fluvisols increased (+12.5%). Consequently, the results from Rendzic Leptosols are in agreement with our assumption that clearcut harvesting and site preparation will cause SOC losses (Mayer et al., 2020). Shifts in the forest floor to more active humus forms due to succession (Ponge, 2003) might further lower SOC contents in the upper 20 cm (Bonifacio et al., 2011) in reafforested Rendzic Leptosol plots. Unlike Rendzic Leptosols, Fluvisols stay consistent in their humus form. High input of organic matter after harvesting is believed to be integrated into soil via active soil fauna. Anecic earthworms associated with mull humus forms take on a key role for this (Bernier & Ponge, 1994). Carbon inputs from living roots and root litter from herbaceous species might add to the carbon pool in the upper soil layers as well (Sokol et al., 2019). Increases in SOC were documented by other studies too (Bock & Van Rees, 2002; Swain et al., 2010). Integration of organic matter in deeper soil layers via site preparation might stabilize and increase to the SOC stock (Swain et al., 2010). The trend of higher SOC stocks in Fluvisols is further increased by higher bulk densities in reafforested plots underlining the potential pitfalls of the fixed depth approach (von Haden et al., 2020). In Rendzic Leptosols the trend of a decreased SOC concentration is even reversed by a higher content of coarse fragments and lower bulk density in mature forest sites. This underlines that the variable treatment does not only represent the impacts of harvesting, site preparation and planting (with links to bulk density) but also the ongoing humus dynamic and differences in the share of coarse fragments. Nitrogen stocks were significantly lower in mature forest stands in Rendzic Leptosols. In Fluvisols only a small tendency of lower N stocks was measured. These effects were most likely apparent due to the same factors impacting SOC as N is tightly linked to SOM (Schulten & Schnitzer, 1998) and nutrient availability is connected to humus forms (Ponge, 2003). C/N ratios in the sampling area are in a typical range for European forests (Cools et al., 2014). Results from modelling the C/N ratio align with the trends seen in nitrogen and eco-physiological ratios. This is most likely related to the trends described above with a pivotal role of the changing humus form (Ponge, 2003) and altered litter inputs caused by changes in tree cover (Cools et al., 2014). Relating the information on SOM with soil functions shifts in SOM quantities for both, soil types and for Rendzic Leptosols in quality as well, can be seen. Changes in SOM quality and partly quantity in Rendzic Leptosols can be linked to the mobilization of the forest floor. Further changes in SOM quantity are linked to the treatment. Cmic/Corg ratios in reafforested plots of Rendzic Leptosols might indicate the highest rate of carbon accumulation (Insam & Domsch, 1988; Malý et al., 2002) and consequently an ongoing recovery process. Increases in SOC stocks in Fluvisols are caused by harvesting and site preparation and the higher bulk density. SOM is still impacted by the treatment after five years, yet the ongoing recovery and the range of values allow the assumption that no critical thresholds were crossed. The current site preparation method (tilling of planting strips) minimizes the disturbance compared to the prior method (sheareblading) which can mitigate the effects of treatment (Mayer et al., 2020). Additionally, the altered and mixed species composition of the reafforested plots will increase the stability of the future forest reducing the risk of large scale disturbances with high SOM losses (Jandl et al., 2007). We thus conclude that soil functions related to SOM in the investigated flood plain forests are not impaired in the long term due to site preparation.

Fluvisols show the expected trend of recovery in nitrate leaching within 5 years as reported by other authors (Gundersen et al., 2006; Smolander et al., 2000). Rendzic Leptosols do not show this trend. This could be due to the first thinning that happened in the area treated ≥5 years ago. Mostly willow trees and bushes were cut down to create more room and light for the slower growing oaks. In the mature forest a spruce tree in the close vicinity of the lysimeter had to be removed due to bark beetle infestation. This happened shortly after the lysimeters were placed. These events could explain why no recovery trend is shown here because already small disturbances can cause significantly higher

nitrate leaching (Nave et al., 2011). Higher nitrate concentrations in seepage water of mature forest stands could be also linked to forest floor processes. Nitrogen loses from the organic layers are documented in literature (Katzensteiner, 2003). Disturbed and collapsing mature forest sites start moving from moder humus forms to mull again (Bernier & Ponge, 1994). This process could be present at our site leading to accelerated SOM mineralization exceeding plant and microbial uptake and as a result lead to nitrate leaching (Gundersen et al., 2006; Johnson, 1992). Additionally, the highest value was recorded on the first date of measuring. It is possible that the placement of the lysimeters still had a confounding influence on the earliest measurments. Nevertheless, the values are well below the 50 mg/l limit of the EU Directive (Richtlinie 98/83/EG, 1998) implemented in the Austrian drinking water regulation (Trinkwasserverordnung, 2001). It is possible that concentrations were temporarily higher but likely not drastically higher as nitrate concentrations are reported to peak 2-3 years after clearcuts (Gundersen et al., 2006; Katzensteiner, 2003). Water quality standards are rarely exceeded in forest water (Gundersen et al., 2006). A recovery trend in Fluvisols and no effect of treatment in Rendzic Leptosols can be seen. Therefore, soil functions related to nitrate leaching, especially the filter function, don't show a long-term effect of harvesting and site preparation. It seems that enhanced nitrate concentrations in Rendzic Leptosols are related to thinning and succession processes in mature forest sites.

Higher bulk densities were found in Fluvisols. Values for forest soils reported by other authors for Fluvisols (0-20 cm, 1.17 g/cm³) (Gajic et al., 2006) and Rendzic Leptosols (0-30 cm, 0.89 g/cm³) (Homolák et al., 2017) confirm this trend. Due to the different methods used for determining bulk density between soil types this effect could be manipulated. As the method was consistent among treatments this potential effect is irrelevant the comparison of treatments. A medium increase in bulk density of 0.1 g/cm³ could be shown on reafforested plots. The relatively large confidence interval suggests that this effect could be weaker or stronger. Site preparation requires driving over the whole area. Given the fact that the greatest increase in bulk density is associated with the first trips over the soil (Froehlich et al., 1985) it can be assumed that site preparation largely accounts for the higher bulk densities. The differences between treatments is larger for Rendzic Leptosols where dissimilarities in texture might add to the effect (Martín et al., 2017). Higher bulk densities in reafforested Rendzic Leptosols are mainly found in the upper 10 cm. This might be attributed to the higher sensitivity towards compaction of more finely textured soils (Froehlich et al., 1985; Van Haveren, 1983) because this is the layer with the lowest sand concentration in Rendzic Leptosols. In Fluvisols the differences were only detectable in the depth layer from 10-20 cm. Compaction in deeper soil layers is often more persistent as they are less affected by alleviation mechanisms (Schjønning, et al., 2015). In Fluvisols plant roots (Blume et al., 2010) and biota activity (Schjønning, et al., 2015) might have already remediated compaction. Dense plant roots were frequently observed in the upper soil layers throughout the sampling area. Mull humus forms imply a high biota activity (Ponge, 2003). Within a certain range soils can react resilient towards compaction and recover (H.-J. Vogel et al., 2018). Especially the trend from Fluvisols suggests a recovery process. Nevertheless, recovery of compaction is a slow process. Froehlich et al. (1985) showed no full recovery within 23 years after logging and Mohieddinne et al. (2019) predict soil recovery after 50-70 years after logging. Compaction of reafforested sites is still persistent after 5 years. Considering detected processes, vegetation parameters and results from microbial biomass an ongoing recovery process can be expected. It is not assumed that soil compaction exceeded dangerous levels from which a recovery wouldn't be possible. The bulk density stayed below critical values of 1.4 g/cm³ for clay soils and 1.8 g/cm³ for sand and loamy sand soils reported by Picchio et al. (2020). Bulk densities detected in the sampling area were also lower than values reported for the same soil types by other authors (Gajic et al., 2006; Homolák et al., 2017). However, recovery within 5 years was not possible. It is critical to minimize soil compaction to the lowest possible level. Site preparation should be kept at the necessary minimum to limit driving on the soil (Froehlich et al., 1985). Dry conditions for driving should be insured as compaction is very sensitive to soil moisture (H.-J. Vogel et al., 2018).

Trends for hydraulic parameters could not be linked to treatment as clearly as it was the case for bulk density. AWC is predominantly correlated to the coarse fragment in the soils. The LMM showing significantly lower values for mature forest stands in Rendzic Leptosols, the group with the highest rock content, underlines this. The model did not meet model assumptions though. Consequently, another approach was used. AWC was to the largest part (adj. R2=0.975) explained by coarse fragment. Treatment as cause of an increased bulk density did not reach significance in the model. It should be noted that the GAM for predicting bulk densities used the share of fine soil which showed a positive correlation with bulk density. Thus, the model for AWC shows that the negative effect of an increased coarse fragment clearly overrides the negative effect of higher bulk densities in less rocky soils. In literature effects of decreased AWC (Jensen et al., 2020) due to compaction as well of increased AWC (Ares et al., 2005) are shown. Pore size distribution seems to be a controlling factor. If soils are rich in large pores (pore diameter >60 µm), compression can lead to an increased water retention and AWC (Ares et al., 2005). AWC decreases if compaction mainly affects pore sizes of 0.2-30 µm retaining plant available water (Jensen et al., 2020). Using pedotransfer functions, no trends could be detected. Ksat is challenging to measure (Hao et al., 2019; Reynolds et al., 2000) and high uncertainties are related to pedotransfer functions (Wösten et al., 1999). Bearing this in mind, interpretations of this parameter must be done cautiously. On the site only soil type has a significant effect on ln(ksat). Higher ksat values for Rendzic Leptosols are logical due to the impact of soil texture (Wösten et al., 1999). However, the model only explained 34 % of the variance in the data indicating other relevant processes. Treatment and the interaction of treatment and soil reached marginal significance. Both models including treatment indicate higher values for mature forest stands. Yet, only for mature forest stands in Rendzic Leptosols higher values are recorded. For Fluvisols they are very similar between treatments. Again, texture could be the explaining factor for these differences in Rendzic Leptosols as mature forest stands tend to have a larger proportion of sand. No significant effect of harvesting and site preparation on ln(ksat) could be shown, although expected based on previous information (Aust et al., 2004; Gent Jr. et al., 1984; Schjønning, et al., 2015). This could be either due to the absence of an effect or due to high uncertainties related to the parameter. It can be concluded that hydraulic parameters based on pedotransfer functions did not reveal any significant differences in soil functions between mature forest stands and reafforested areas.

4.3. Effect of tilling

None of the measured parameters regarding vegetation showed significant effects of tilling strip. Not even in the first year after the site treatment differences in *Solidago* covers were detectable. This leads to the conclusion that rotary tilling does not promote the regrowth of *Solidago* from rhizome fragments more than mulching which was performed on the whole area. Schultz & Wilhite (1974) showed that site preparation which leads to a concentration of nutrients e.g., via treatments in rows stratifies vegetation and alters the species composition. In this case the tilling strips did not alter the nutrient supply in a magnitude showing in the species composition. The previous treatment in the area, where harvesting slash was concentrated in rows every 20 m was still performed on two of the treatment from ≥5 years. In these rows there was a noticeably different vegetation with higher shares of mosses (no data shown). Nevertheless, tilling strips seem to have a positive impact on seedling survival. On the tilled plots at the age group ≥ 5 years 18 seedlings were missing, whereas on the untilled plots 42 were missing. This is in accordance with the findings of other studies reporting smaller mortality (Bilodeau-Gauthier et al., 2011) and better growth rates (Bauman et al., 2013; Bilodeau-Gauthier et al., 2011) of seedlings after site preparation. Löf et al. (2012) concludes that site preparation often results in improved seedling survival and growth. Regarding the better growth we cannot make any assumptions as the age group ≥ 5 years includes plots with age differences of up to 3 years. Overall, the effects of the tilling strips regarding the vegetation aspect of the habitat function seem to be minimal.

Tilling did not significantly affect microbial biomass or related eco-physiological parameters. As tilling generally reduces microbial biomass (for both Cmic and Nmic) (Zuber & Villamil, 2016) this could suggest a recovery of parameters related to microbial biomass within 5 years. Results from Smolander et al. (2000) show that site preparation in form of mounding did not significantly affect microbial biomass; site preparation was conducted in fall and sampling started in the following spring. Also chisel tillage in agriculture doesn't lead to a reduction in microbial biomass (Zuber & Villamil, 2016). This suggests that some forms of site preparation do not additionally affect the microbial biomass which could be the case here.

The effect of tilling on SOC and nitrate stocks are significant. This is in line with the previous assumption and the majority of literature showing negative effects of site preparation (Mayer et al., 2020). Piirainen et al. (2015) presented similar results showing that soil preparation in strips increases the heterogeneity in C and N stocks on plots. Yet, the differences between tilling strips and areas in between no longer remain significant when only looking at the percentual contents of C and N underlining the pitfalls of calculating stocks von Haden et al. (2020) point out. Tendencies for lower contents in tilled areas continued to exist though. These weaker effects caused by strip wise site preparations are in accordance with Wang et al. (2016). They found that soil preparation only in spots has a much lighter effect on carbon release than preparing the full area. An ongoing recovery of SOC concentrations could be indicated by Cmic/Corg values. They show a non-significant tendency of higher values in tilling strips potentially indicating higher carbon sequestration (Insam & Domsch, 1988; Malý et al., 2002). In this work the calculation of stocks presents an interesting option combining SOC and N with bulk density. This allows us showing differences not apparent using only single parameters. The C/N ratio didn't show any effect of tilling. This might be linked to the non-existent differences between strips in vegetation composition and microbial biomass parameters. Research showing the influence of vegetation on the C/N ratio via litter input (Rowe et al., 2006) and the fact that microbes are the drivers of SOM mineralization (Bååth & Anderson, 2003; Blume et al., 2010) support this assumption. Nitrate leaching could not be linked to tilling. Due to the assessment below the main rooting area sampling distinguishing between tilling strips and area in between was not possible. The Fluvisol plot showed a full recovery of nitrate leaching within 5 years while no clear trend was observed for Rendzic Leptosols. It should be noted that only the Fluvisol plot treated ≥5 years ago was tilled while the plot on Rendzic Leptosols was still treated with the old method. This suggests that small scale soil preparation might not have the long-lasting adverse effect Gundersen et al. (2006) expect site preparation to have on nitrate seepage.

Bulk density seems to be affected by tilling strips. Despite not reaching significance ($p=0.401$) there is a tendency of lower bulk densities in tilling strips. This is also hinted at by the effect of tilling strips on SOC and N stocks where bulk density is combined with C and N concentrations. Lower bulk density after site preparation was documented by Wang et al. (2016) too. Also, the smaller seedling mortality on tilled plots described above suggest a potential positive effect of tilling on bulk density (Bauman et al., 2013; Bilodeau-Gauthier et al., 2011). No effect of tilling was shown for AWC. This agrees with the finding of Fleming et al. (1994) who state that site preparation does not influence the AWC if the upper layer of the mineral soil isn't removed. It contradicts the statement of Morris & Lowery (1988) that AWC can be increased in compacted soil via site preparation. We assume that this might be due to less severe soil compaction on our site. For ln(ksat) we reported higher values in tilling strips with marginal significance. It could be plausible that the tilling improved the ksat (Aust et al., 2004) but due to structure degradation it is also possible that there is no recovery (Aust et al., 2004; Gent Jr. et al., 1984). This underlines the uncertainty of this parameter derived from pedotransfer functions (Wösten et al., 1999). Tilling seems to slightly alleviate the bulk density while no positive effect on hydrological parameters is expected.

5. Summary and Conclusions

Several parameters provide valuable information on soil functions. Vegetation data allows for drawing conclusions on habitat function and the regulatory function of soils as well as on productivity. Species composition and the cover of specific taxa of interest (in our case *Solidago canadensis agg.*) provide the most robust results. Biodiversity indicators and Ellenberg indicator values can present additional insights but shouldn't be used as single parameters for management decisions. Eco-physiological ratios derived from microbial biomass show information on the microbial community and regulatory function of soils, namely the carbon cycle. Assessing soil organic matter is vital for investigating soil functions as it is affected by management and influences all ecological soil functions. If resources allow it, we recommend the use of the equivalent soil mass approach to determine soil organic carbon stocks. To detect a recovery of the filter function of soils the installation of lysimeters and monitoring of nitrate leaching is a good option. Soil compaction is a major threat to soil functions. Thus, bulk density should be included in all assessments of management impact on soil functions. Hydraulic parameters derived from pedotransfer functions might allow further insights but are linked to relatively high uncertainties. Additional parameters of interest for larger studies could be the earthworm population and an assessment of the pore network. Both parameters are quite resource intensive to measure though.

A final judgement of soil function recovery from site preparation is challenging due to all the different aspects impacting them. For vegetation parameters we could not clearly separate site preparation as a single factor from impacts caused by harvesting, calamities, the increasing *Solidago* population and potential interactions of all factors. Additional difficulties are posed by the non-consistent management of *Solidago canadensis agg.* throughout the chronosequence. Generally, the management of invasive species is challenging, and regional solutions would be advisable. No effect of treatment could be measured for microbial biomass. Differences in eco-physiological parameters are most likely linked to changes in humus forms due to forest regeneration. Soil organic matter is impacted by treatment in quantity. Changes in quality (and partly quantity) are linked to the mobilization of the forest floor and altered litter input due to the changed species composition. Treatment causes a lower soil organic carbon concentration in Rendzic Leptosols and a higher concentration in Fluvisols. Despite differences still being measurable, ongoing recovery processes and the range of values allow the assumption that no critical thresholds were crossed. Nitrate seepage did never exceed water quality standards. Fluvisols showed the expected recovery trend of the filter function within 5 years. No effect was detected for Rendzic Leptosols. The higher values recorded here are linked to thinning in treated areas and forest floor dynamics in mature forest sites. Hydraulic parameters calculated from pedotransfer functions did not show effects of treatment.

We could not show any significant effects of tilling on separately assessed parameters. Nevertheless, the non-significant differences of bulk density and soil organic carbon concentrations combined to soil organic carbon stocks reached significance. This shows a negative impact of tilling on soil organic matter. Eco-physiological parameters indicate a recovery though. It also reveals an alleviating effect of tilling on bulk density. Better seedling survival indicates a potential positive effect of tilling on the productivity function via improvement of root penetration.

According to our knowledge, most parameters are in a tolerable variance that can be expected throughout forest dynamics. Only the increases in bulk density are slightly concerning. Despite not exceeding thresholds, recovery times are long. Driving should be limited to a minimum. Potentially the better afforestation success with more diverse tree species can override the negative impacts. The current method of mulching and tilling planting strips is thought to have a lower impact on soil functions than shearblading which was a common practice in the past.

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List of abbreviations

Age groups / treatments:

J	treated 1 year ago
M	treated 2-3 years ago
A	treated ≥5 years ago
R	mature forest site, reference
U	fallow area

AIC	Akaike's Information Criterion
AWC	Available Water storage Capacity
AWC_{30}	AWC for the upper 30 cm
AWC_{tot}	total AWC
C	Carbon
C/N	Carbon to Nitrogen ratio
CFE	Chloroform Fumigation-Extraction
Cmic	Microbial Carbon
Corg	Organic Carbon
FSC	Forest Stewardship Council
GAM	Generalized Additive Model
IT	Information Theoretic
Ksat	Saturated hydraulic conductivity
LECA	Lightweight Expanded Clay Aggregates
LM	Linear Model
LMM	Linear Mixed Model
ln	natural logarithm
M	Mol
m.a.s.l.	meters above sea level
MvG	Mualem-van Genuchten
n	sampling size
N	Nitrogen
NHST	Null Hypothesis Significance Testing
Nmic	microbial Nitrogen
NP-MANOVA	Non-Parametric Multiple Analysis of Variances
N_{tot}	total Nitrogen
PEFC	Programme for the Endorsement of Forest Certification

PTF	Pedotransfer function
qCO_2	metabolic quotient, CO_2 respiration to g microbial biomass ratio
R2	R squared
R2c	Conditional R squared
R2m	Marginal R squared
REML	Restricted Maximum Likelihood
SOC	Soil Organic Carbon
SOM	Soil Organic Matter
VIF	Variance Inflation Factor

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If not stated differently all pictures and graphs were taken/produced by the author.

Appendix

Soil profile examples

Soil profile Fluvisol, depths below 30 cm were assessed using a Pürckhauer drill.

O: 0-1 cm, only litter, clear transition

Ah: 1-36 cm, fine to medium blocky, granular (worm cast), loamy sandy silt, 10YR 3/2, strongly calcareous, intermediate bulk density, roots common, diffuse smooth boundary

C1: 36-54 cm, sandy silt, 10YR 5/2, extremely calcareous, clear smooth boundary

Cg2: 54+ cm, clayey silt, 10YR 5/1, extremely calcareous, very fine to fine mottles and concretions common

Soil profile Rendzic Leptosol

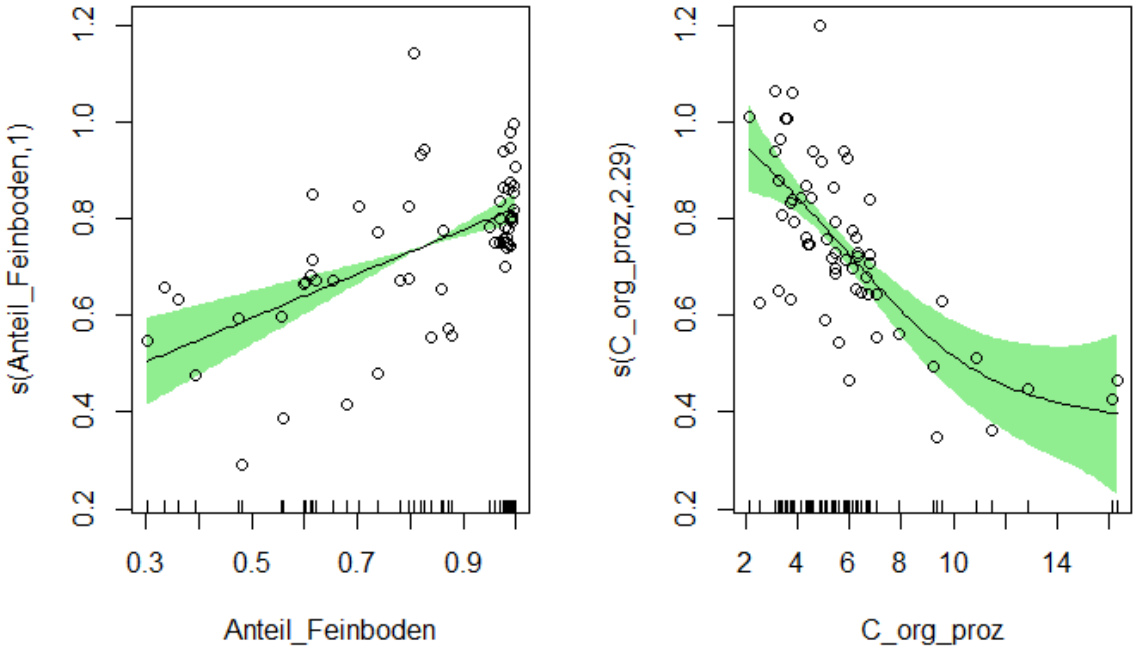
O: 0-2 cm, layered, many roots, clear transition

Ah: 2-20 cm, fine to medium blocky, granular, silty loam, 7% fine to coarse gravel, 10YR 2/1, moderately calcareous, intermediate bulk density, roots common, clear smooth boundary

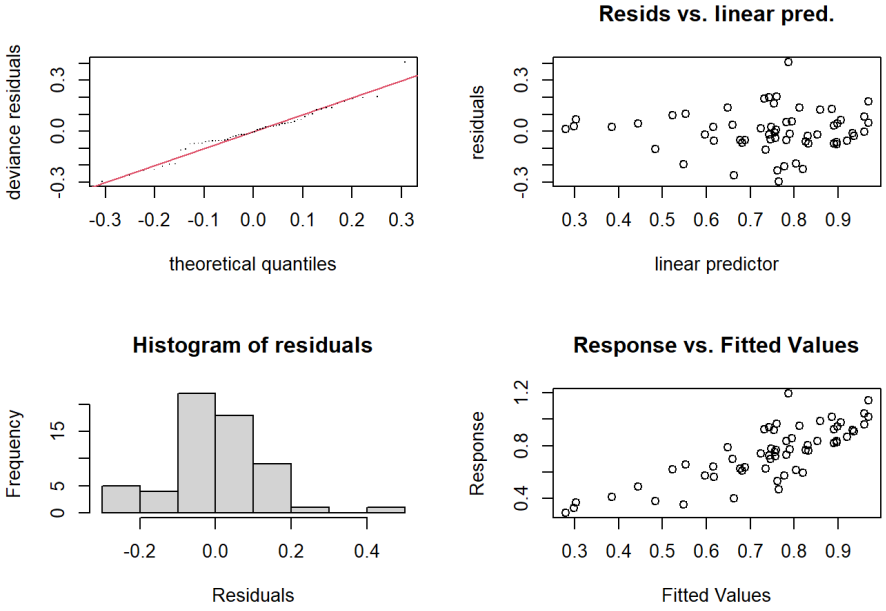
C: 20+ cm, single grain, loamy sand, 80% fine gravel to stones, 10YR 4/1, extremely calcareous, firm bulk density, few roots

GAM for bd

Model plot



The plots show on the left the smooth of the share of fine soil in percent on the y-axis and the share of fine soil in percent on the x-axis. On the right the same is plotted for SOC in %. The y-axis has been shifted by the intercept which improves the interpretability. Given that the other variable is at their average value the y-axis shows the prediction of the output.



Diagnostic plots

Concurvity

```
##                   para s(Anteil_Feinboden) s(C_org_proz)
## worst    4.202762e-24           0.6491132     0.6491132
## observed 4.202762e-24           0.1875006     0.3376641
## estimate 4.202762e-24           0.1640376     0.2858842
```

NP-MANOVA

R-output of the NP-MANOVA with the Bray-Curtis distance compared with the chord distance for:

a) The final model

```
> set.seed(1130)
> adonis2(vegdat~sitdat$Bodentyp*sitdat$Altersklasse+sitdat$plotsize_m2+
+             sitdat$BauStr, method = "chord", permutations = perm)
Permutation test for adonis under reduced model
Terms added sequentially (first to last)
Blocks:  with(sitdat, plot)
Permutation: free
Number of permutations: 999

adonis2(formula = vegdat ~ sitdat$Bodentyp * sitdat$Altersklasse + sitdat$plotsize_m2 +
 sitdat$BauStr, permutations = perm, method = "chord")
                                Df SumOfSqs      R2       F Pr(>F)
sitdat$Bodentyp                  1    3.762 0.07575 13.4825  0.001 ***
sitdat$Altersklasse              4   19.273 0.38810 17.2684  0.001 ***
sitdat$plotsize_m2               1    0.503 0.01014  1.8041  0.033 *
sitdat$BauStr                    1    2.192 0.04414  7.8558  0.002 **
sitdat$Bodentyp:sitdat$Altersklasse  4    6.909 0.13912  6.1902  0.001 ***
Residual                        61   17.021 0.34274
Total                           72   49.660 1.00000
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
> set.seed(1130)
> adonis2(vegdat~sitdat$Bodentyp*sitdat$Altersklasse+sitdat$plotsize_m2+
+             sitdat$BauStr, method = "bray", permutations = perm)
Permutation test for adonis under reduced model
Terms added sequentially (first to last)
Blocks:  with(sitdat, plot)
Permutation: free
Number of permutations: 999

adonis2(formula = vegdat ~ sitdat$Bodentyp * sitdat$Altersklasse + sitdat$plotsize_m2 +
 sitdat$BauStr, permutations = perm, method = "bray")
                                Df SumOfSqs      R2       F Pr(>F)
sitdat$Bodentyp                  1   2.2630 0.09719 17.2378  0.002 **
sitdat$Altersklasse              4   8.6588 0.37188 16.4890  0.002 **
sitdat$plotsize_m2               1   0.2738 0.01176  2.0858  0.026 *
sitdat$BauStr                    1   0.8604 0.03695  6.5542  0.009 **
sitdat$Bodentyp:sitdat$Altersklasse  4   3.2195 0.13827  6.1310  0.002 **
Residual                        61   8.0081 0.34394
Total                           72  23.2837 1.00000
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
```

b) The final model with dummies

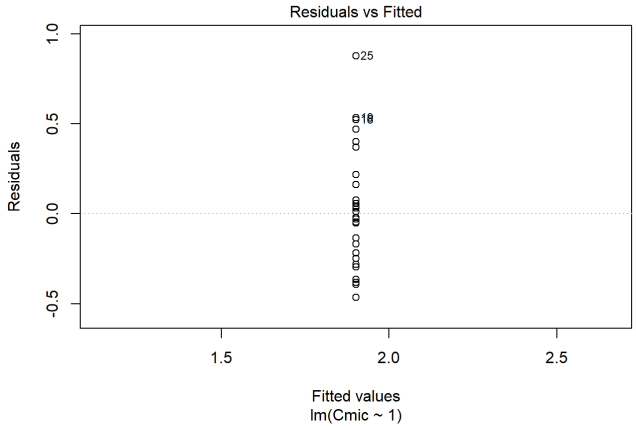
```
> set.seed(1130)
> adonis2(vegdat~sitdat$Bodentyp*sitdat$d1*sitdat$d2*sitdat$d3*sitdat$d4+
+             sitdat$plotsize_m2+sitdat$BauStr, method = "bray",
+         permutations = perm)
Permutation test for adonis under reduced model
Terms added sequentially (first to last)
Blocks:  with(sitdat, plot)
Permutation: free
Number of permutations: 999

adonis2(formula = vegdat ~ sitdat$Bodentyp * sitdat$d1 * sitdat$d2 * sitdat$d3 * sitdat$d4 +
 sitdat$plotsize_m2 + sitdat$BauStr, permutations = perm, method = "bray")
                          Df SumOfSqs      R2       F Pr(>F)
sitdat$Bodentyp            1   2.2630 0.09719 17.2378  0.002 **
sitdat$d1                  1   1.5693 0.06740 11.9541  0.002 **
sitdat$d2                  1   3.3858 0.14541 25.7903  0.002 **
sitdat$d3                  1   3.3576 0.14421 25.5760  0.002 **
sitdat$d4                  1   0.3460 0.01486  2.6356  0.002 **
sitdat$plotsize_m2         1   0.2738 0.01176  2.0858  0.026 *
sitdat$BauStr              1   0.8604 0.03695  6.5542  0.009 **
sitdat$Bodentyp:sitdat$d1  1   1.7081 0.07336 13.0112  0.002 **
sitdat$Bodentyp:sitdat$d2  1   0.3954 0.01698  3.0117  0.423
sitdat$Bodentyp:sitdat$d3  1   0.7652 0.03287  5.8290  0.370
sitdat$Bodentyp:sitdat$d4  1   0.3508 0.01507  2.6722  0.038 *
Residual                  61   8.0081 0.34394
Total                     72  23.2837 1.00000
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
> set.seed(1130)
> adonis2(vegdat~sitdat$Bodentyp*sitdat$d1*sitdat$d2*sitdat$d3*sitdat$d4+
+             sitdat$plotsize_m2+sitdat$BauStr, method = "chord",
+         permutations = perm)
Permutation test for adonis under reduced model
Terms added sequentially (first to last)
Blocks:  with(sitdat, plot)
Permutation: free
Number of permutations: 999

adonis2(formula = vegdat ~ sitdat$Bodentyp * sitdat$d1 * sitdat$d2 * sitdat$d3 * sitdat$d4 +
 sitdat$plotsize_m2 + sitdat$BauStr, permutations = perm, method = "chord")
                          Df SumOfSqs      R2       F Pr(>F)
sitdat$Bodentyp            1    3.762 0.07575 13.4825  0.001 ***
sitdat$d1                  1    3.439 0.06925 12.3246  0.001 ***
sitdat$d2                  1    7.338 0.14777 26.2989  0.001 ***
sitdat$d3                  1    7.896 0.15899 28.2967  0.001 ***
sitdat$d4                  1    0.601 0.01210  2.1534  0.001 ***
sitdat$plotsize_m2         1    0.503 0.01014  1.8041  0.033 *
sitdat$BauStr              1    2.192 0.04414  7.8558  0.002 **
sitdat$Bodentyp:sitdat$d1  1    4.076 0.08207 14.6068  0.001 ***
sitdat$Bodentyp:sitdat$d2  1    1.084 0.02182  3.8843  0.198
sitdat$Bodentyp:sitdat$d3  1    1.132 0.02280  4.0578  0.105
sitdat$Bodentyp:sitdat$d4  1    0.617 0.01243  2.2119  0.023 *
Residual                  61   17.021 0.34274
Total                     72   49.660 1.00000
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
```

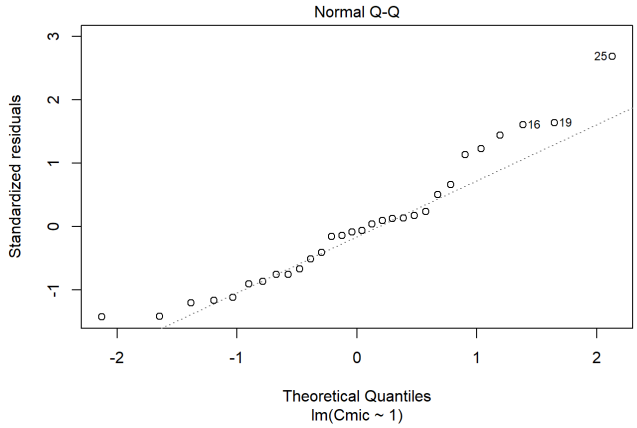
Models for chemical and physical parameters

Microbial Biomass

Cmic

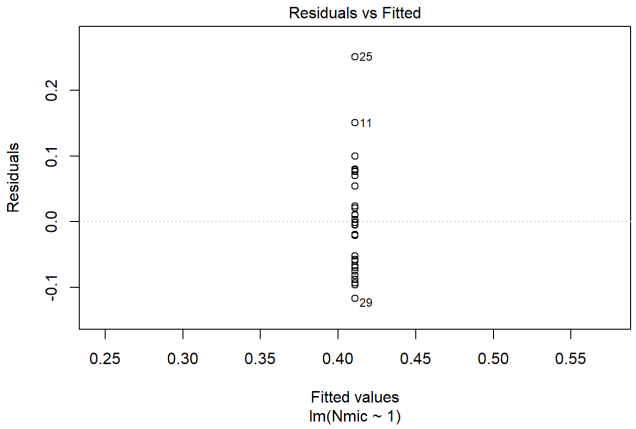
Formula: Cmic~1					
	Estimate	Std. Error	95% CI	t value	p
Intercept	1.902	0.061	[1.778, 2.026]	31.58	<2E-16

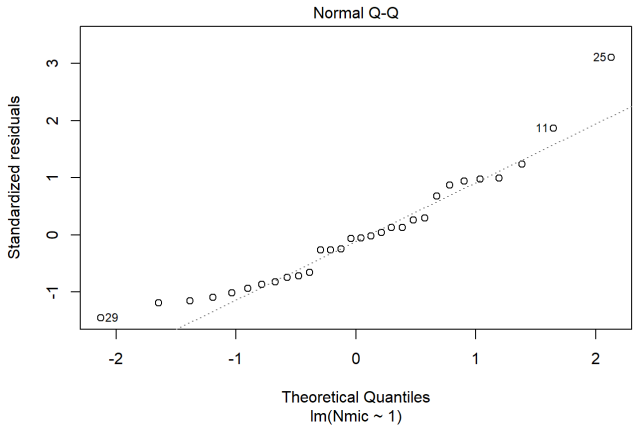




Nmic

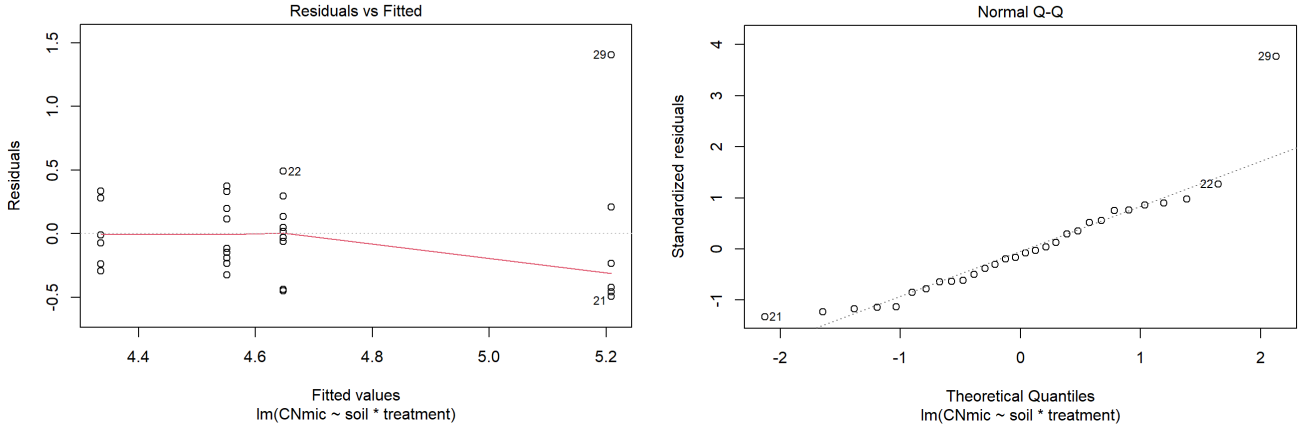
Formula: Nmic~1					
	Estimate	Std. Error	95% CI	t value	p
Intercept	0.411	0.015	[0.380, 0.442]	27.41	<2E-16





Cmic/Nmic

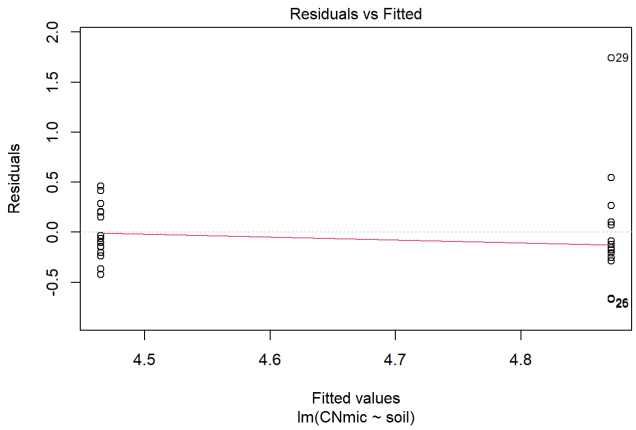
Diagnostic plots of Cmic/Nmic~soil*treatment

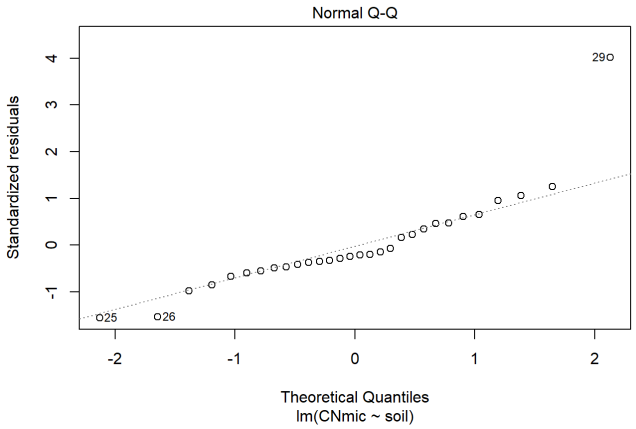


Simpler model only including *soil* as explanatory variable

```
##
## Call:
## lm(formula = CNmic ~ soil, data = nestdat)
##
## Residuals:
##      Min       1Q   Median       3Q      Max
## -0.67361 -0.20721 -0.09592  0.18779  1.74113
##
## Coefficients:
##             Estimate Std. Error t value Pr(>|t|)
## (Intercept)   4.4648     0.1157  38.577   <2e-16 ***
## soil1         0.4072     0.1637   2.488   0.0191 *
## ---
## Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
##
## Residual standard error: 0.4483 on 28 degrees of freedom
## Multiple R-squared:  0.181,  Adjusted R-squared:  0.1518
## F-statistic: 6.189 on 1 and 28 DF,  p-value: 0.01908
```

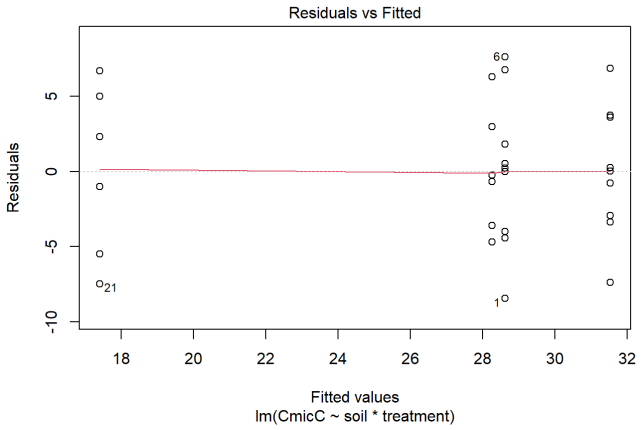
Diagnostic plots of Cmic/Nmic~soil

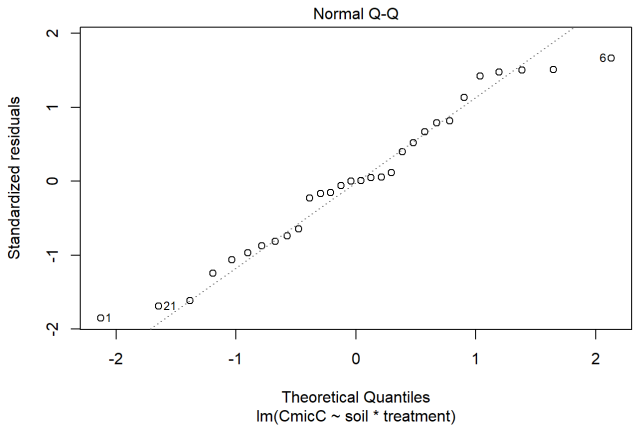




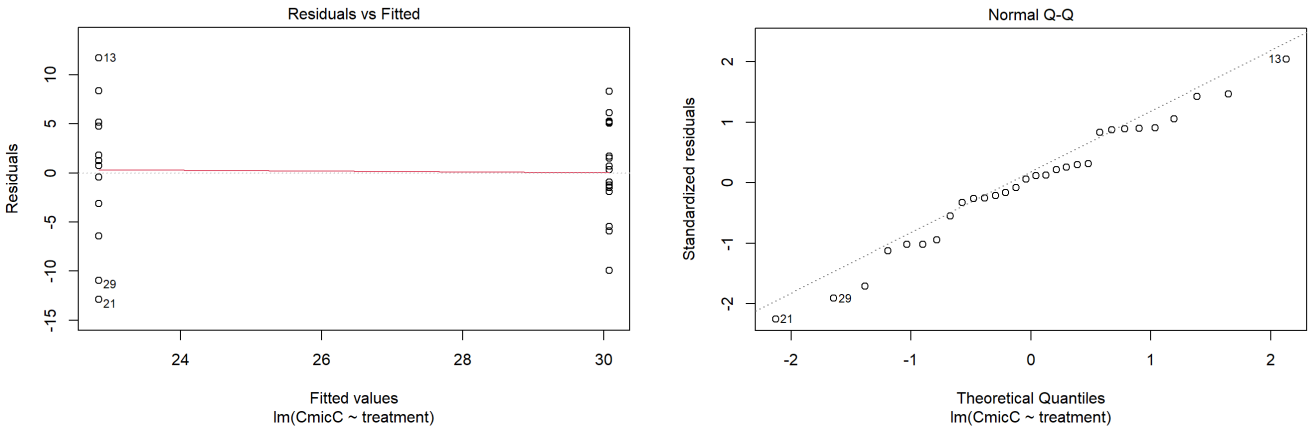
Cmic/Corg

Model containing soil*treatment



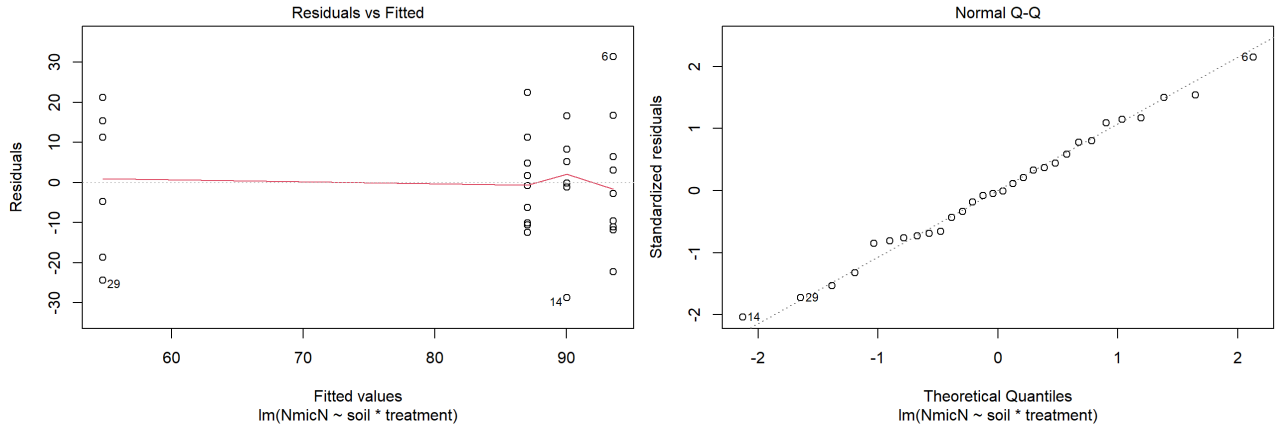


Model including only treatment



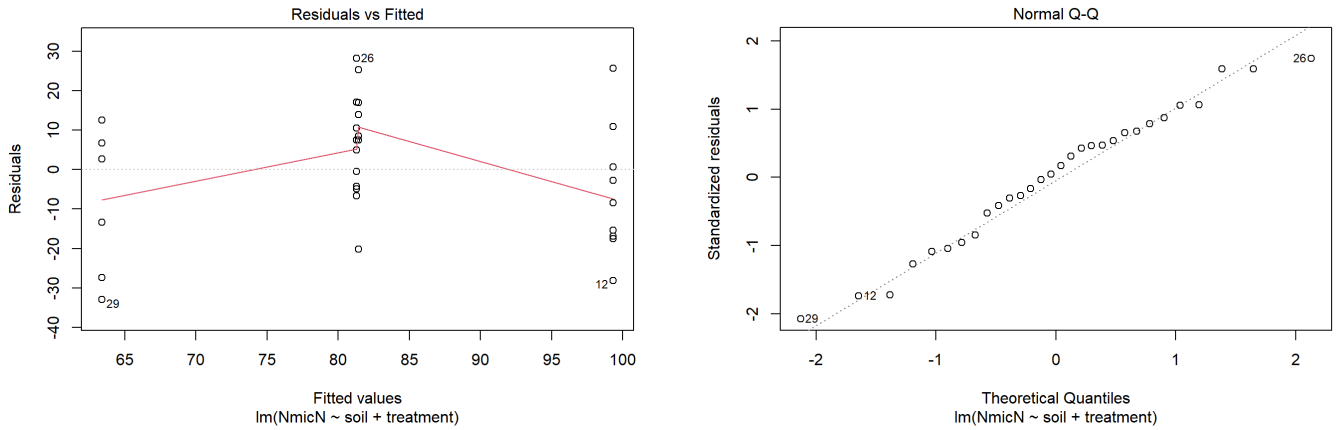
Nmic/N$_{tot}$

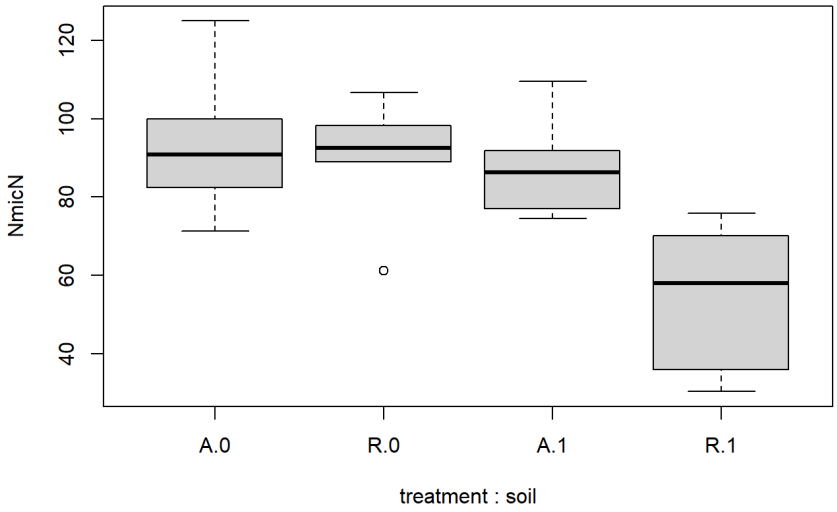
```
## Call:
## lm(formula = NmicN ~ soil * treatment, data = nestdat)
##
## Residuals:
##      Min       1Q   Median       3Q      Max
## -28.7702 -10.4866  -0.4508  10.5493  31.3652
##
## Coefficients:
##                  Estimate Std. Error t value Pr(>|t|)
## (Intercept)        93.561      5.152  18.159 2.72e-16 ***
## soil1              -6.521      7.286  -0.895   0.3790
## treatmentR         -3.503      8.147  -0.430   0.6708
## soil1:treatmentR  -28.779     11.521  -2.498   0.0191 *
## ---
## Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
##
## Residual standard error: 15.46 on 26 degrees of freedom
## Multiple R-squared:  0.5009, Adjusted R-squared:  0.4433
## F-statistic: 8.698 on 3 and 26 DF,  p-value: 0.0003652
```



Simpler model without the interaction reached significance too. Yet this model did not meet the assumptions and had a noticeably lower adjusted R2. Boxplots suggest that the inclusion of the interaction makes sense and is well justified by the data.

```
## Call:
## lm(formula = NmicN ~ soil + treatment, data = nestdat)
##
## Residuals:
##     Min      1Q  Median      3Q     Max
## -32.971 -12.175   1.691  10.821  28.152
##
## Coefficients:
##             Estimate Std. Error t value Pr(>|t|)
## (Intercept)   99.317      5.036  19.722  < 2e-16 ***
## soil1        -18.033      6.167  -2.924  0.00692 **
## treatmentR   -17.892      6.295  -2.842  0.00842 **
## ---
## Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
##
## Residual standard error: 16.89 on 27 degrees of freedom
## Multiple R-squared:  0.3811, Adjusted R-squared:  0.3353
## F-statistic: 8.314 on 2 and 27 DF,  p-value: 0.001536
```





Difference between tilled strip and area in between did not reach significance (p=0.564).

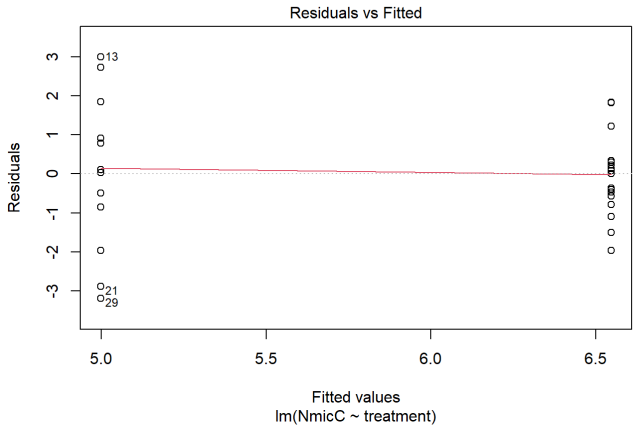
Nmic/Corg

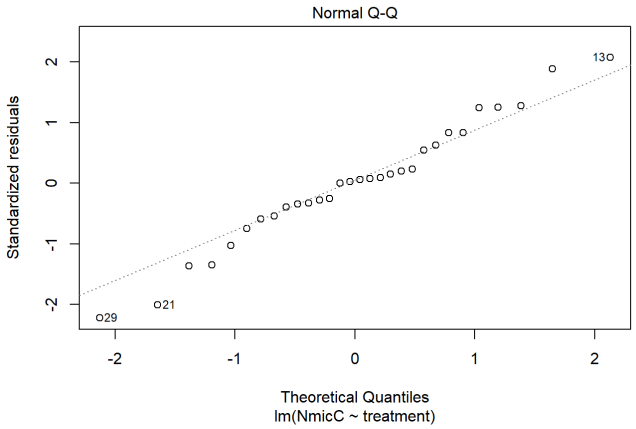
Without interaction treatment reached significance too. Again, the model with interaction performed better. Difference between tilled strip and area in between did not reach significance (p=0.473).

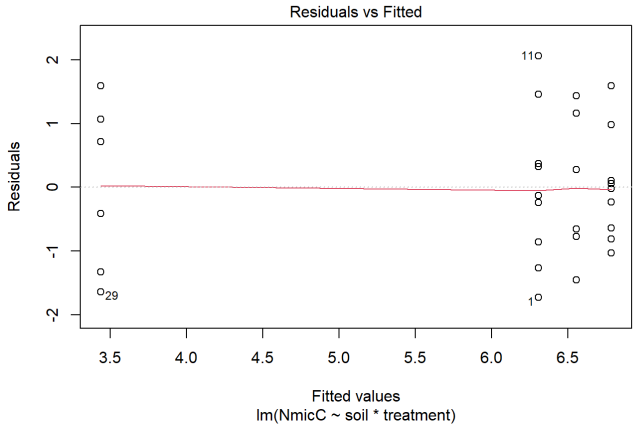
Model including treatment*soil:

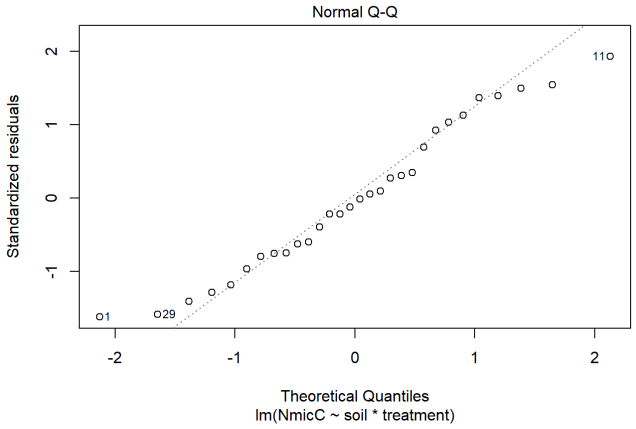
```
## Call:
## lm(formula = NmicC ~ soil * treatment, data = nestdat)
##
## Residuals:
##      Min       1Q   Median       3Q      Max
## -1.72873 -0.80029 -0.07266  0.91615  2.06014
##
## Coefficients:
##                 Estimate Std. Error t value Pr(>|t|)
## (Intercept)        6.3089     0.3763  16.766 1.85e-15 ***
## soil1              0.4764     0.5321   0.895 0.378877
## treatmentR         0.2486     0.5950   0.418 0.679545
## soil1:treatmentR  -3.5960     0.8414  -4.274 0.000228 ***
## ---
## Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
##
## Residual standard error: 1.129 on 26 degrees of freedom
## Multiple R-squared:  0.5891, Adjusted R-squared:  0.5417
## F-statistic: 12.43 on 3 and 26 DF,  p-value: 3.131e-05
```

Model only including treatment:



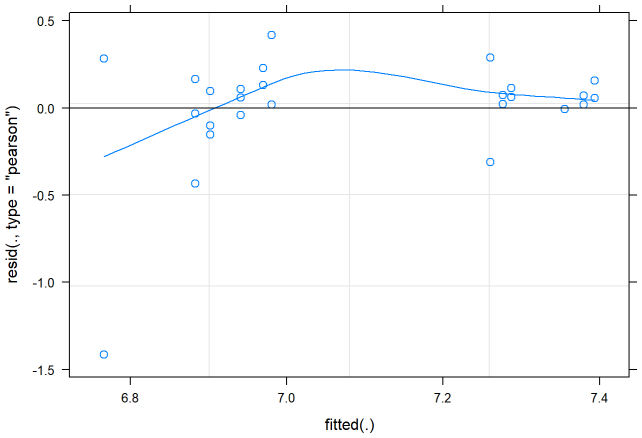


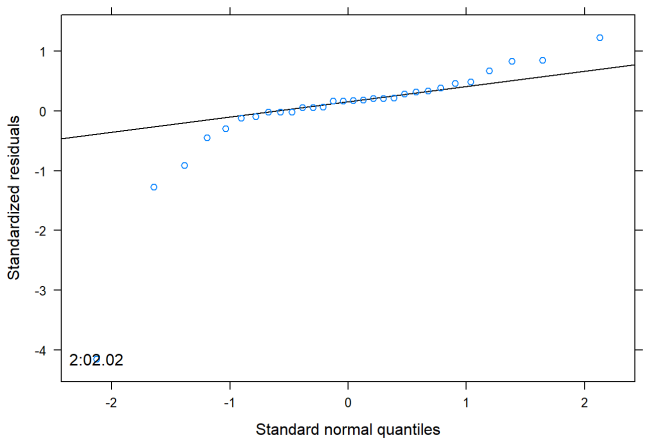





```
## Call:
## lm(formula = NmicC ~ treatment, data = nestdat)
##
## Residuals:
##     Min      1Q  Median      3Q     Max
## -3.1977 -0.7383  0.0589  0.8780  2.9946
##
## Coefficients:
##             Estimate Std. Error t value Pr(>|t|)
## (Intercept)   6.5471     0.3545  18.467  < 2e-16 ***
## treatmentR   -1.5494     0.5606  -2.764  0.00998 **
## ---
## Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
##
## Residual standard error: 1.504 on 28 degrees of freedom
## Multiple R-squared:  0.2144, Adjusted R-squared:  0.1863
## F-statistic:  7.64 on 1 and 28 DF,  p-value: 0.009981
```

pH



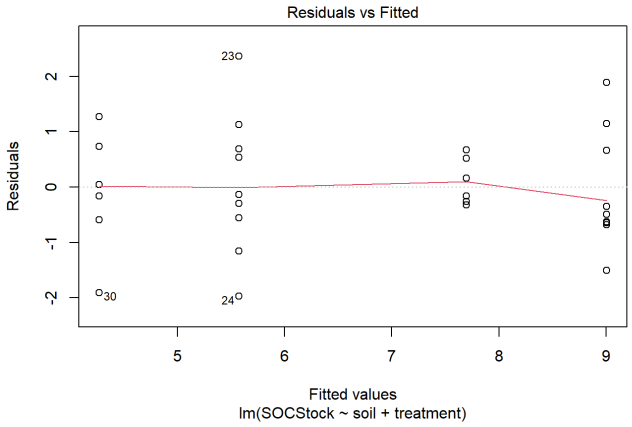


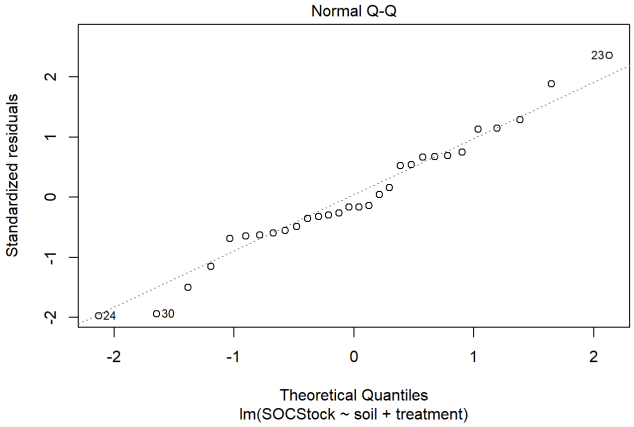
Formula: pH~1+(1|plot/subplot)

Random effect	Variance	sd				
subplot:plot	0.016	0.126				
plot	0.068	0.261				
residuals	0.116	0.341				
Fixed effects	Estimate	Std. Error	95% CI	df	t value	p
Intercept	7.116	0.149	[6.788, 7.444]	3.039	47.64	1.82E-05

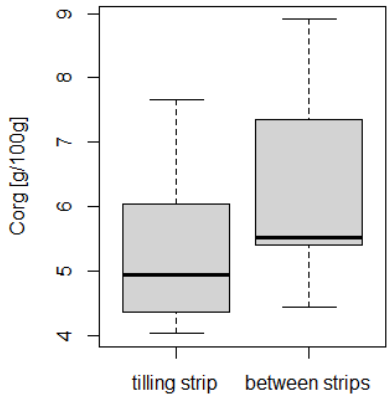
R2m	0
R2c	0.419

SOC Stock



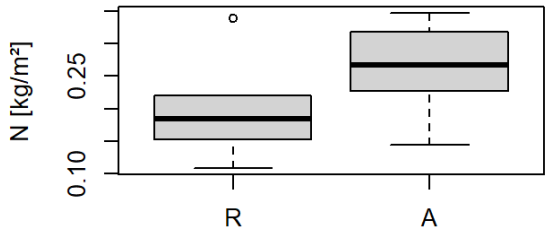


SOC content (%) for tilling strips and area in between

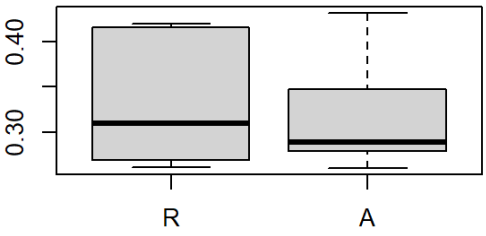


N Stock

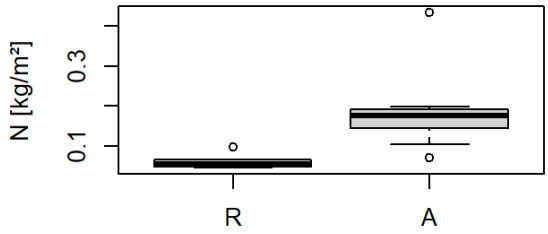
N stocks comparing reafforested plots (A) with mature forest sites (R) for the depth levels from 0-10 cm and 10-20 cm for Fluvisols and Rendzic Leptosols:



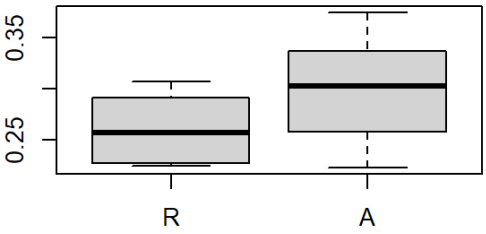
Rendzic leptosol 0-10cm



Fluvisol 0-10cm

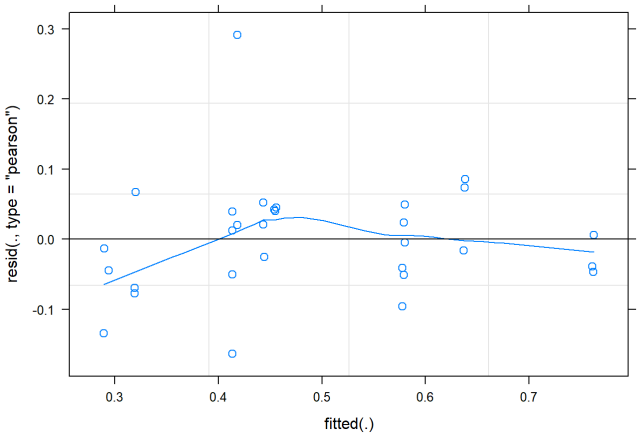


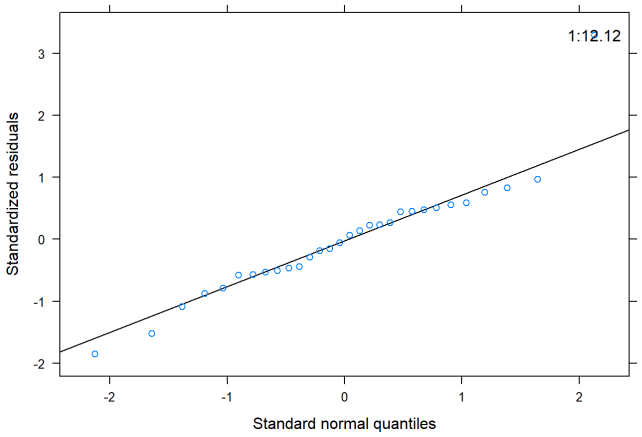
Rendzic leptosol 10-20cm



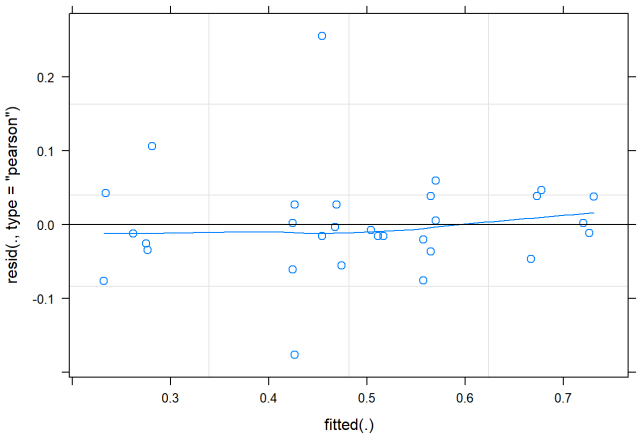
Fluvisol 10-20cm

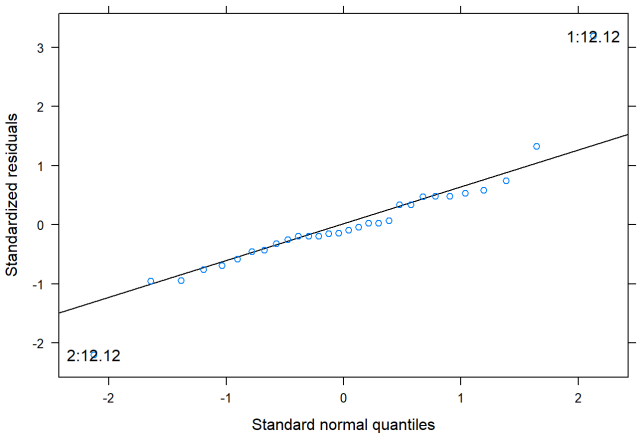
LMM with treatment as fixed effect:



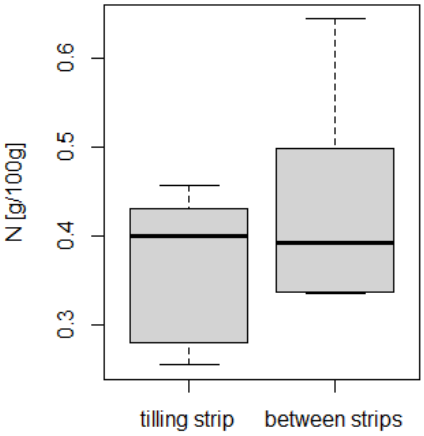


LMM containing soil*treatment as fixed effect:





N content (%) for tilling strips and area in between

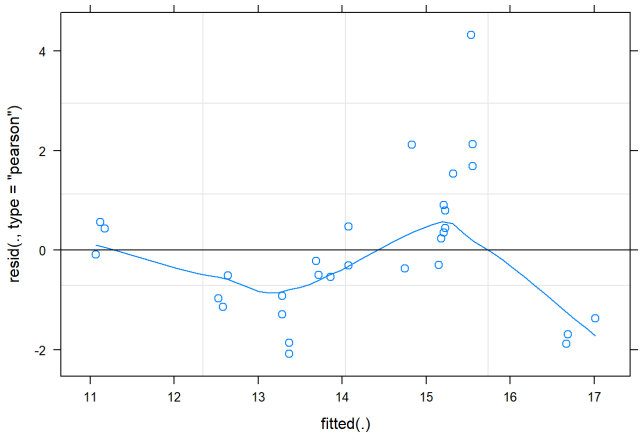


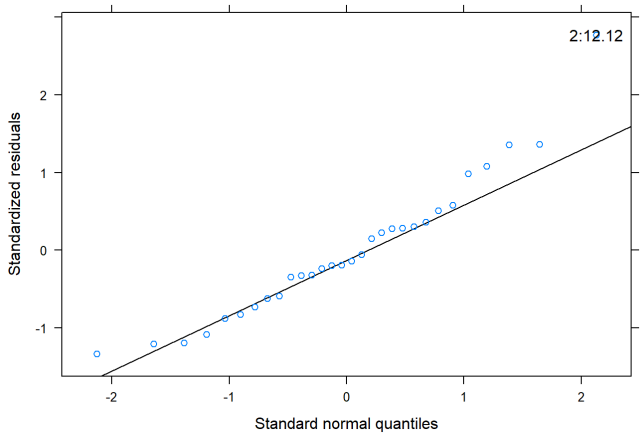
C/N ratio

Model with only significant input:

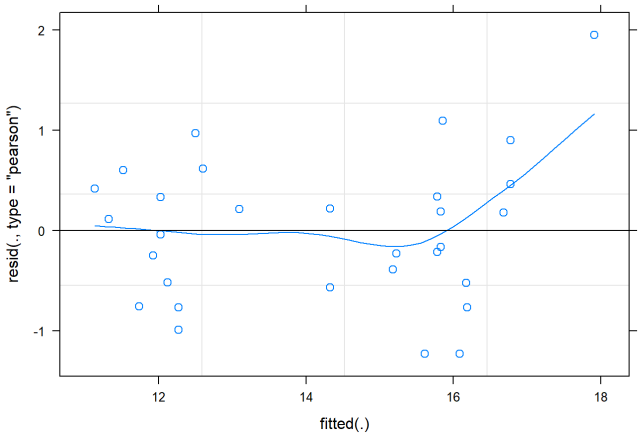
```
Linear mixed model fit by REML. t-tests use Satterthwaite's method ['lmerModLmerTest']
Formula: CN ~ treatment + (1 | plot/subplot)
   Data: nestdat

REML criterion at convergence: 119

Scaled residuals:
    Min      1Q  Median      3Q     Max
-1.3338 -0.6127 -0.1638  0.3463  2.7694

Random effects:
 Groups       Name        Variance Std.Dev.
 subplot:plot (Intercept) 0.2723   0.5219
 plot         (Intercept) 3.2432   1.8009
 Residual                 2.4415   1.5625
Number of obs: 30, groups:  subplot:plot, 12; plot, 4

Fixed effects:
            Estimate Std. Error      df t value Pr(>|t|)
(Intercept)  13.4446     0.9889  3.3597  13.596  0.00047 ***
treatmentR    1.4624     0.5898 18.7270   2.479  0.02285 *
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Correlation of Fixed Effects:
           (Intr)
treatmentR -0.248
```

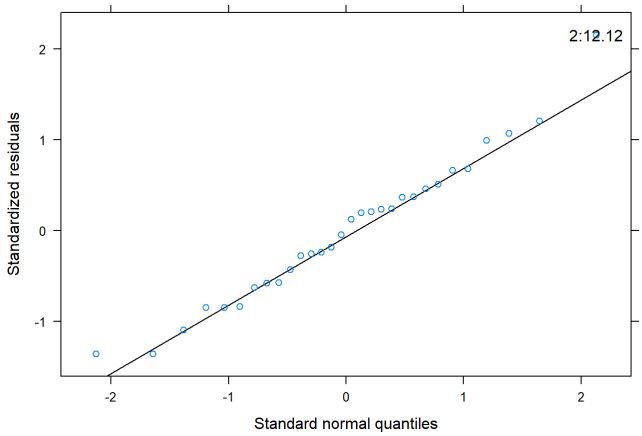
Conditional R2: 0.624; Marginal R2: 0.082



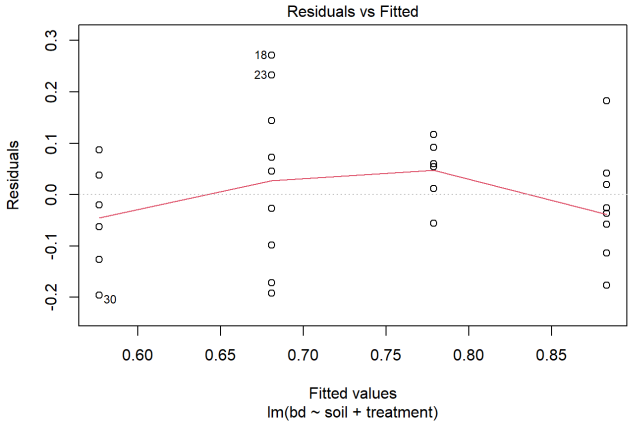


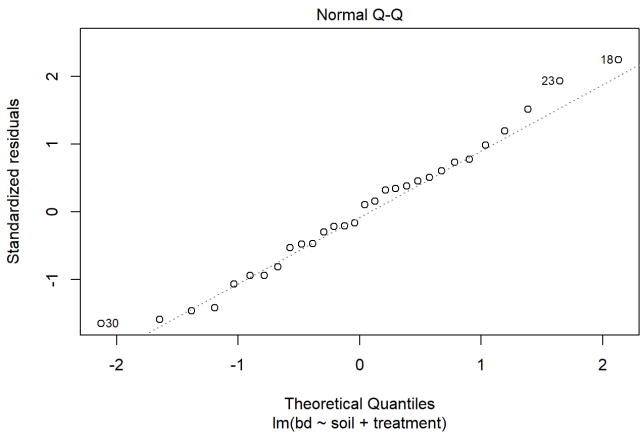
Diagnostic plots of model containing treatment*soil as fixed effect:





Bulk density

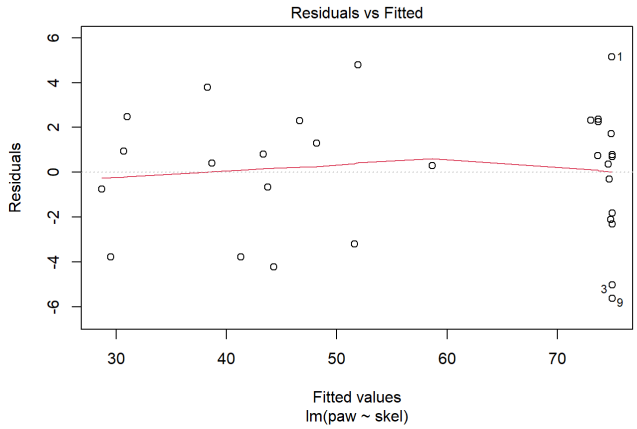


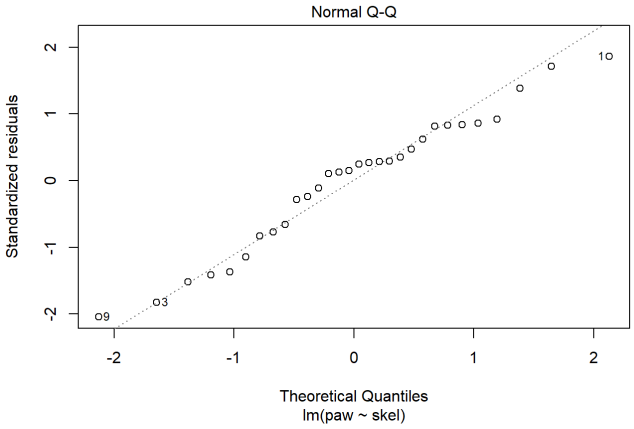


AWC

LM zu AWC

Formula AWC~coarse					
	Estimate	Std. Error	95% CI	t value	p
Intercept	74.993	0.721	[73.517, 76.469]	104.07	<2E-16
coarse	-78.282	2.321	[-83.036, -73.528]	-33.73	<2E-16
R2 (adj.)	0.975	p-value	<2.2E-16		



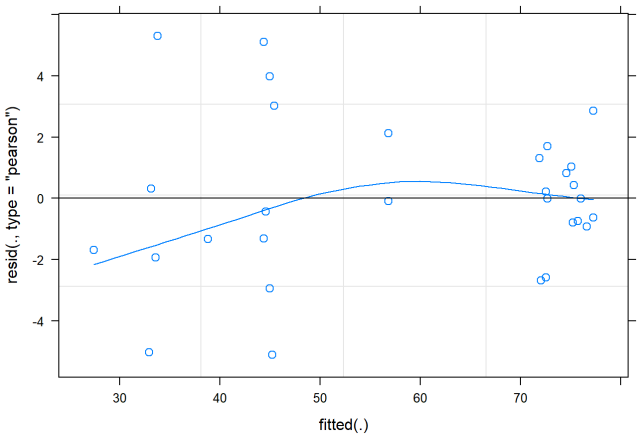


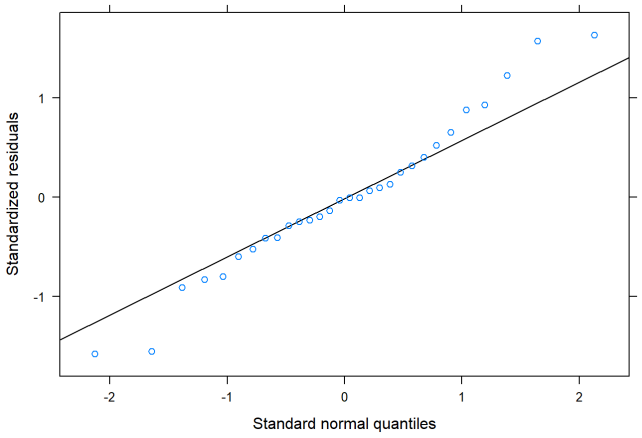
LMM

```
## Linear mixed model fit by REML. t-tests use Satterthwaite's method [
## lmerModLmerTest]
## Formula: paw ~ treatment * soil + (1 | plot/subplot)
##    Data: nestdat
##
## REML criterion at convergence: 161.2
##
## Scaled residuals:
##      Min       1Q   Median       3Q      Max
## -1.57802 -0.40852 -0.01499  0.38207  1.63738
##
## Random effects:
##  Groups       Name        Variance Std.Dev.
##  subplot:plot (Intercept) 20.853   4.566
##  plot         (Intercept)  3.474   1.864
##  Residual                 10.498   3.240
## Number of obs: 30, groups:  subplot:plot, 12; plot, 4
##
## Fixed effects:
##                  Estimate Std. Error       df t value Pr(>|t|)
## (Intercept)       74.9178     2.5400   2.3318  29.496 0.000471 ***
## treatmentR        -0.6685     1.7288  16.4437  -0.387 0.703953
## soil1            -29.1262     3.5921   2.3318  -8.108 0.009298 **
## treatmentR:soil1 -10.7501     2.4450  16.4437  -4.397 0.000424 ***
## ---
## Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
```

Confidence interval

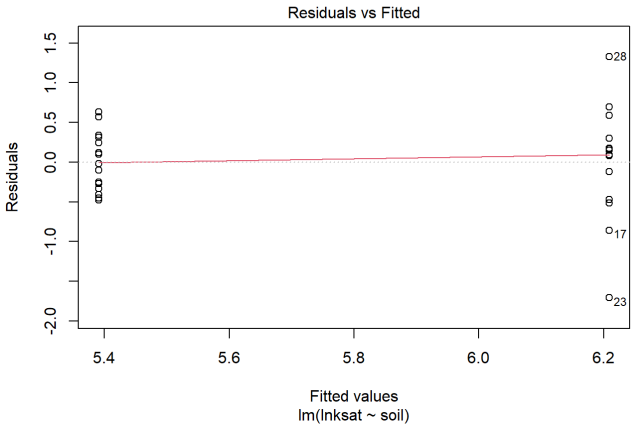
```
## (Intercept)        70.294077  79.563884
## treatmentR         -4.011248   2.724526
## soil1             -35.737569 -22.624525
## treatmentR:soil1  -15.644600  -6.103248
```

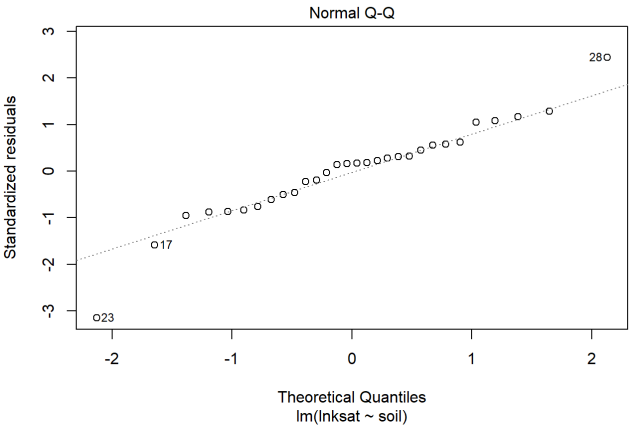




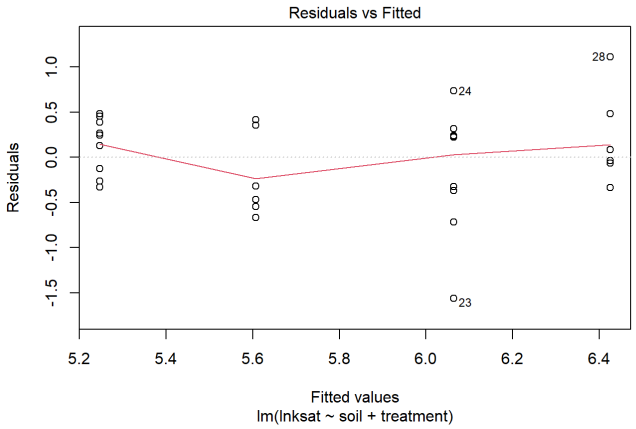
ln(ksat)

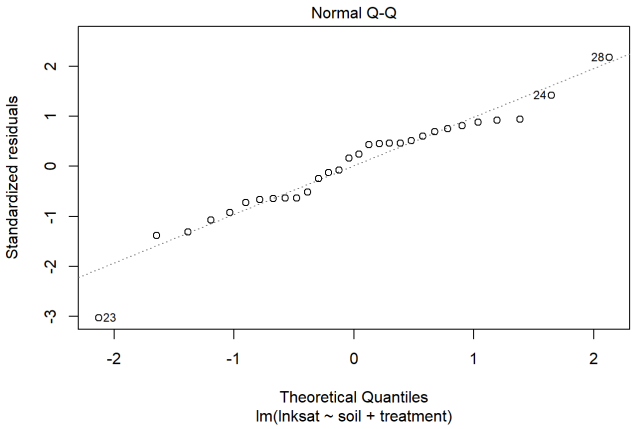
LM including only soil:





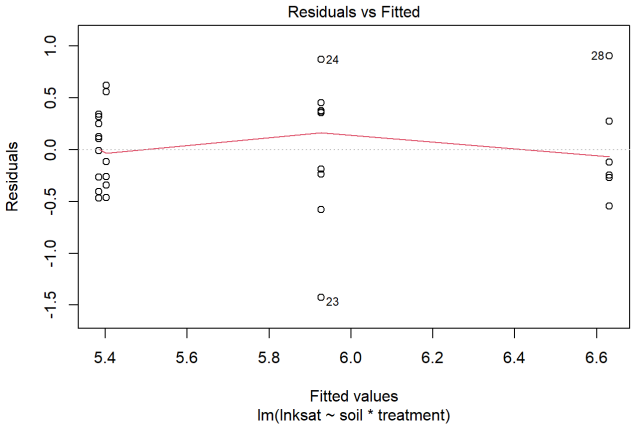
LM including soil + treatment

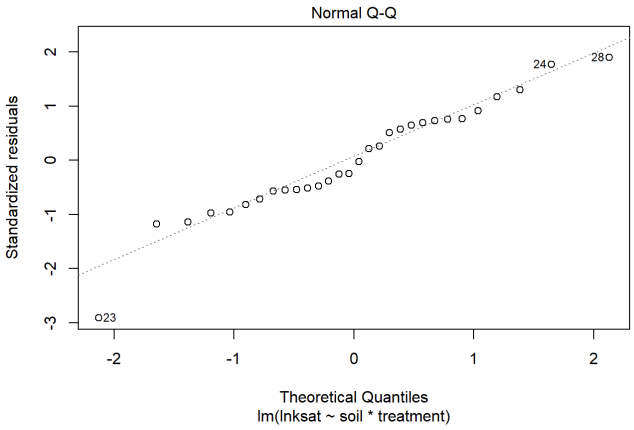





```
## Call:
## lm(formula = lnksat ~ soil + treatment, data = nestdat)
##
## Residuals:
##     Min      1Q  Median      3Q     Max
## -1.5650 -0.3301  0.1054  0.3433  1.1106
##
## Coefficients:
##             Estimate Std. Error t value Pr(>|t|)
## (Intercept)   5.2467     0.1611  32.563  < 2e-16 ***
## soil1         0.8177     0.1973   4.144 0.000302 ***
## treatmentR    0.3609     0.2014   1.792 0.084372 .
## ---
## Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
##
## Residual standard error: 0.5404 on 27 degrees of freedom
## Multiple R-squared:  0.4302, Adjusted R-squared:  0.388
## F-statistic: 10.19 on 2 and 27 DF,  p-value: 0.0005042
```

LM including soil * treatment






```
## Call:
## lm(formula = lnksat ~ soil * treatment, data = nestdat)
##
## Residuals:
##      Min       1Q   Median       3Q      Max
## -1.42793 -0.26899 -0.06252  0.35474  0.90511
##
## Coefficients:
##                  Estimate Std. Error t value Pr(>|t|)
## (Intercept)       5.38374    0.17346  31.037   <2e-16 ***
## soil1             0.54369    0.24531   2.216   0.0356 *
## treatmentR        0.01833    0.27427   0.067   0.9472
## soil1:treatmentR  0.68512    0.38787   1.766   0.0891 .
## ---
## Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
##
## Residual standard error: 0.5204 on 26 degrees of freedom
## Multiple R-squared:  0.4912, Adjusted R-squared:  0.4325
## F-statistic: 8.368 on 3 and 26 DF,  p-value: 0.000465
```

Vegetation list

Plot ID:

Position 1: soil type, 0 = Fluvisol, 1 = Rendzic Leptosol

Position 2: Replicate of soil type cluster, either 1 or 2

Position 3: Treatment, J, M, A, R, U

Position 4: Sub-plot, either 1, 2 or 3

Position 5: 0 if not tilled, F if tilling strip, Z if between tilling strips

Layer: B = tree layer, S = shrub layer, K = herbaceous layer, M = moss layer

Plot ID	Species	Cover [%]	Layer
01A1F	Aegopodium podagraria	7	K
01A1F	Anemone nemorosa	2	K
01A1F	Carex alba	5	K
01A1F	Carex sylvatica	3	K
01A1F	Carpinus betulus	25	S
01A1F	Deschampsia cespitosa	10	K
01A1F	Galium mollugo	1	K
01A1F	Glechoma hederacea	10	K
01A1F	Hypnum cupressiforme	35	M
01A1F	Primula elatior	1	K
01A1F	Rubus fruticosus agg	2	K
01A1F	Solidago gigantea	45	K
01A1F	Taraxacum officinale	1	K
01A1Z	Aegopodium podagraria	7	K
01A1Z	Anemone nemorosa	5	K
01A1Z	Carex alba	1	K
01A1Z	Carex sylvatica	5	K
01A1Z	Carpinus betulus	5	S
01A1Z	Deschampsia cespitosa	10	K
01A1Z	Galium mollugo	2	K
01A1Z	Glechoma hederacea	10	K
01A1Z	Hypnum cupressiforme	20	M
01A1Z	Rubus fruticosus agg	2	K
01A1Z	Solidago gigantea	55	K
01A2F	Aegopodium podagraria	0.5	K
01A2F	Anemone nemorosa	5	K
01A2F	Angelica sylvestris	2	K
01A2F	Carex alba	15	K
01A2F	Carex sylvatica	2	K
01A2F	Glechoma hederacea	5	K
01A2F	Hypnum cupressiforme	10	M
01A2F	Juglans nigra	10	S
01A2F	Rubus fruticosus agg	2	K
01A2F	Solidago gigantea	50	K
01A2Z	Ajuga reptans	0.5	K

01A2Z	Anemone nemorosa	7	K
01A2Z	Angelica sylvestris	2	K
01A2Z	Arum maculatum	1	K
01A2Z	Carex alba	15	K
01A2Z	Carex sylvatica	2	K
01A2Z	Cornus sanguinea	10	S
01A2Z	Glechoma hederacea	3	K
01A2Z	Hedera helix	2	K
01A2Z	Hypnum cupressiforme	7	M
01A2Z	Planthera spp.	0.5	K
01A2Z	Rubus fruticosus agg	2	K
01A2Z	Solidago gigantea	50	K
01A3F	Anemone nemorosa	3	K
01A3F	Angelica sylvestris	2	K
01A3F	Arum maculatum	1	K
01A3F	Carex alba	7	K
01A3F	Clematis vitalba	5	S
01A3F	Cornus sanguinea	5	S
01A3F	Glechoma hederacea	3	K
01A3F	Hypnum cupressiforme	10	M
01A3F	Prunus avium	35	S
01A3F	Prunus padus	5	K
01A3F	Salix spec	5	S
01A3F	Salix spec	2	K
01A3F	Solidago gigantea	50	K
01A3Z	Anemone nemorosa	5	K
01A3Z	Angelica sylvestris	3	K
01A3Z	Arum maculatum	3	K
01A3Z	Carex alba	5	K
01A3Z	Carex sylvatica	1	K
01A3Z	Clematis vitalba	2	K
01A3Z	Euphorbia amygdaloides	0.5	K
01A3Z	Galium mollugo	1	K
01A3Z	Glechoma hederacea	5	K
01A3Z	Hypnum cupressiforme	10	M
01A3Z	Paris quadrifolia	2	K
01A3Z	Polygonatum multiflorum	0.5	K
01A3Z	Prunus avium	25	S
01A3Z	Solidago gigantea	50	K
01M1F	Aegopodium podagraria	3	K
01M1F	Anemone nemorosa	7	K
01M1F	Arum maculatum	1	K
01M1F	Cirsium oleraceum	2	K
01M1F	Festuca ovina agg	5	K
01M1F	Festuca rubra agg	45	K
01M1F	Glechoma hederacea	1	K
01M1F	Hypnum cupressiforme	10	M
01M1F	Mercurialis perennis	1	K

01M1F	Paris quadrifolia	1	K
01M1F	Quercus rubra	5	S
01M1F	Rubus fruticosus agg	1	K
01M1F	Solidago gigantea	40	K
01M1F	Taraxacum officinale	2	K
01M1F	Tilia cordata	10	S
01M1Z	Aegopodium podagraria	2	K
01M1Z	Alium ursinum	3	K
01M1Z	Anemone nemorosa	20	K
01M1Z	Arum maculatum	3	K
01M1Z	Brachypodium sylvaticum	1	K
01M1Z	Cirsium oleraceum	1	K
01M1Z	Colchicum autumnale	2	K
01M1Z	Festuca ovina agg	5	K
01M1Z	Festuca rubra agg	55	K
01M1Z	Glechoma hederacea	2	K
01M1Z	Hypnum cupressiforme	10	M
01M1Z	Lamium maculatum	1	K
01M1Z	Mercurialis perennis	3	K
01M1Z	Paris quadrifolia	2	K
01M1Z	Solidago gigantea	35	K
01M1Z	Taraxacum officinale	1	K
01M2F	Aegopodium podagraria	3	K
01M2F	Anemone nemorosa	7	K
01M2F	Arum maculatum	2	K
01M2F	Brachypodium sylvaticum	1	K
01M2F	Carex sylvatica	1	K
01M2F	Festuca ovina agg	5	K
01M2F	Festuca rubra agg	35	K
01M2F	Glechoma hederacea	2	K
01M2F	Hypericum hirsutum	0.5	K
01M2F	Hypnum cupressiforme	5	M
01M2F	Mercurialis perennis	1	K
01M2F	Paris quadrifolia	2	K
01M2F	Rubus fruticosus agg	2	K
01M2F	Solidago gigantea	40	K
01M2F	Tilia cordata	20	S
01M2Z	Aegopodium podagraria	2	K
01M2Z	Alium ursinum	2	K
01M2Z	Anemone nemorosa	7	K
01M2Z	Arum maculatum	3	K
01M2Z	Brachypodium sylvaticum	0.5	K
01M2Z	Carex sylvatica	2	K
01M2Z	Cirsium arvense	4	K
01M2Z	Festuca ovina agg	5	K
01M2Z	Festuca rubra agg	30	K
01M2Z	Glechoma hederacea	2	K
01M2Z	Hypericum hirsutum	1	K

01M2Z	Hypnum cupressiforme	7	M
01M2Z	Lamium maculatum	0.5	K
01M2Z	Mercurialis perennis	2	K
01M2Z	Primula elatior	1	K
01M2Z	Pulmonaria officinalis	1	K
01M2Z	Rubus fruticosus agg	2	K
01M2Z	Solidago gigantea	40	K
01M2Z	Taraxacum officinale	1	K
01M3F	Aegopodium podagraria	2	K
01M3F	Anemone nemorosa	4	K
01M3F	Arum maculatum	0.5	K
01M3F	Festuca rubra agg	60	K
01M3F	Mercurialis perennis	0.5	K
01M3F	Rubus fruticosus agg	1	K
01M3F	Solidago gigantea	45	K
01M3F	Taraxacum officinale	1	K
01M3Z	Aegopodium podagraria	5	K
01M3Z	Anemone nemorosa	10	K
01M3Z	Arum maculatum	3	K
01M3Z	Brachypodium sylvaticum	1	K
01M3Z	Carex alba	0.5	K
01M3Z	Carex sylvatica	1	K
01M3Z	Dactylis glomerata	3	K
01M3Z	Festuca rubra agg	65	K
01M3Z	Lamium maculatum	0.5	K
01M3Z	Mercurialis perennis	2	K
01M3Z	Pulmonaria officinalis	1	K
01M3Z	Rubus fruticosus agg	1	K
01M3Z	Solidago gigantea	30	K
01M3Z	Taraxacum officinale	2	K
01M3Z	Vicia sepium	1	K
01R10	Acer pseudoplatanus	25	S
01R10	Anemone nemorosa	30	K
01R10	Arum maculatum	1	K
01R10	Brachypodium sylvaticum	2	K
01R10	Carex alba	55	K
01R10	Carex sylvatica	3	K
01R10	Clematis vitalba	10	K
01R10	Crataegus spec	5	K
01R10	Euonymus europaeus	1	K
01R10	Hedera helix	10	B
01R10	Hedera helix	7	K
01R10	Hypnum cupressiforme	40	M
01R10	Lamium maculatum	2	K
01R10	Listera ovata	3	K
01R10	Mercurialis perennis	5	K
01R10	Picea abies	80	B
01R10	Primula elatior	4	K

01R10	Quercus robur	2	K
01R10	Rubus fruticosus agg	5	K
01R20	Acer pseudoplatanus	30	B
01R20	Acer pseudoplatanus	20	S
01R20	Anemone nemorosa	5	K
01R20	Arum maculatum	3	K
01R20	Carex alba	90	K
01R20	Clematis vitalba	2	K
01R20	Cornus sanguinea	3	K
01R20	Corylus avellana	2	K
01R20	Euphorbia amygdaloides	3	K
01R20	Hedera helix	10	B
01R20	Hedera helix	3	K
01R20	Listera ovata	1	K
01R20	Mercurialis perennis	2	K
01R20	Picea abies	40	B
01R20	Quercus robur	2	K
01R20	Rubus fruticosus agg	2	K
01R20	Tilia cordata	20	B
01R20	Viola reichenbachiana	0.5	K
01R30	Acer pseudoplatanus	30	S
01R30	Aegopodium podagraria	2	K
01R30	Anemone nemorosa	50	K
01R30	Arum maculatum	7	K
01R30	Asarum europaeum	5	K
01R30	Brachypodium sylvaticum	1	K
01R30	Carex alba	1	K
01R30	Clematis vitalba	10	B
01R30	Euphorbia amygdaloides	3	K
01R30	Glechoma hederacea	5	K
01R30	Hedera helix	25	B
01R30	Hedera helix	3	K
01R30	Hypnum cupressiforme	40	M
01R30	Lamium maculatum	4	K
01R30	Mercurialis perennis	20	K
01R30	Oxalis acetosella	2	K
01R30	Paris quadrifolia	7	K
01R30	Picea abies	65	B
01R30	Polygonatum multiflorum	2	K
01R30	Primula elatior	1	K
01R30	Ulmus glabra	30	B
01R30	Ulmus glabra	3	K
01U10	Anemone nemorosa	7	K
01U10	Arum maculatum	1	K
01U10	Carex alba	4	K
01U10	Cornus sanguinea	7	S
01U10	Euphorbia amygdaloides	0.5	K
01U10	Glechoma hederacea	3	K

01U10	Hedera helix	0.5	K
01U10	Paris quadrifolia	1	K
01U10	Primula elatior	1	K
01U10	Pulmonaria officinalis	10	K
01U10	Rubus fruticosus agg	8	K
01U10	Solidago gigantea	80	K
01U10	Viburnum lantana	3	S
01U10	Viburnum lantana	3	K
01U20	Aegopodium podagraria	1	K
01U20	Anemone nemorosa	2	K
01U20	Arum maculatum	3	K
01U20	Brachypodium sylvaticum	1	K
01U20	Carex alba	70	K
01U20	Cornus sanguinea	5	S
01U20	Euonymus europaeus	4	K
01U20	Euphorbia amygdaloides	2	K
01U20	Fraxinus excelsior	0.5	K
01U20	Hedera helix	7	K
01U20	Hypnum cupressiforme	5	M
01U20	Lamium maculatum	2	K
01U20	Lonicera xylosteum	40	S
01U20	Polygonatum multiflorum	1	K
01U20	Rubus fruticosus agg	30	K
01U20	Ulmus glabra	0.5	K
01U20	Viburnum lantana	5	S
01U30	Aegopodium podagraria	2	K
01U30	Anemone nemorosa	3	K
01U30	Arum maculatum	5	K
01U30	Carex alba	70	K
01U30	Cornus sanguinea	5	S
01U30	Crataegus spec	5	S
01U30	Glechoma hederacea	1	K
01U30	Hedera helix	3	K
01U30	Hypnum cupressiforme	5	M
01U30	Lamium maculatum	0.5	K
01U30	Lonicera xylosteum	2	S
01U30	Mercurialis perennis	0.5	K
01U30	Prunus spinosa	3	S
01U30	Quercus robur	2	S
01U30	Rubus fruticosus agg	10	K
01U30	Solidago gigantea	4	K
01U30	Ulmus glabra	3	S
01U30	Ulmus glabra	1	K
01U30	Viburnum lantana	3	S
02A10	Aegopodium podagraria	1	K
02A10	Anemone nemorosa	20	K
02A10	Arum maculatum	5	K
02A10	Calamagrostis epigejos	5	K

02A10	Cardamine bulbifera	0.5	K
02A10	Carex alba	10	K
02A10	Carex flacca	2	K
02A10	Cornus sanguinea	15	S
02A10	Corylus avellana	10	S
02A10	Euphorbia amygdaloides	0.5	K
02A10	Lonicera xylosteum	2	S
02A10	Quercus robur	15	S
02A10	Rhamnus frangula	5	S
02A10	Rubus fruticosus agg	5	K
02A10	Solidago gigantea	40	K
02A10	Vicia sepium	0.5	K
02A20	Aegopodium podagraria	1	K
02A20	Anemone nemorosa	10	K
02A20	Arum maculatum	10	K
02A20	Carex alba	1	K
02A20	Carex flacca	1	K
02A20	Carex sylvatica	1	K
02A20	Cornus sanguinea	5	S
02A20	Corylus avellana	100	S
02A20	Glechoma hederacea	1	K
02A20	Hedera helix	5	K
02A20	Lamium maculatum	2	K
02A20	Lonicera xylosteum	5	S
02A20	Quercus robur	10	S
02A20	Rubus fruticosus agg	7	K
02A20	Solidago gigantea	4	K
02A30	Aegopodium podagraria	1	K
02A30	Anemone nemorosa	5	K
02A30	Arum maculatum	5	K
02A30	Calamagrostis epigejos	5	K
02A30	Carex alba	5	K
02A30	Carex sylvatica	2	K
02A30	Carpinus betulus	5	S
02A30	Cornus sanguinea	20	S
02A30	Corylus avellana	30	S
02A30	Fraxinus excelsior	10	S
02A30	Hedera helix	1	K
02A30	Quercus robur	30	S
02A30	Rubus fruticosus agg	2	S
02A30	Rubus fruticosus agg	5	K
02A30	Salix spec	2	S
02A30	Solidago gigantea	40	K
02J1F	Aegopodium podagraria	0.5	K
02J1F	Anemone nemorosa	3	K
02J1F	Cirsium arvense	1	K
02J1F	Festuca altissima	5	K
02J1F	Festuca ovina agg	10	K

02J1F	Festuca rubra agg	60	K
02J1F	Lamium album	0.5	K
02J1F	Lolium perenne	5	K
02J1F	Quercus robur	30	S
02J1F	Rubus fruticosus agg	3	K
02J1F	Solidago gigantea	0.5	K
02J1F	Taraxacum officinale	0.5	K
02J1Z	Aegopodium podagraria	5	K
02J1Z	Anemone nemorosa	5	K
02J1Z	Arum maculatum	4	K
02J1Z	Brachypodium sylvaticum	10	K
02J1Z	Cirsium arvense	0.5	K
02J1Z	Cirsium vulgare	1	K
02J1Z	Festuca ovina agg	10	K
02J1Z	Festuca rubra agg	15	K
02J1Z	Lolium perenne	30	K
02J1Z	Mercurialis perennis	3	K
02J1Z	Rubus fruticosus agg	3	K
02J1Z	Solidago gigantea	3	K
02J1Z	Taraxacum officinale	1	K
02J2F	Aegopodium podagraria	3	K
02J2F	Carpinus betulus	30	S
02J2F	Cirsium arvense	7	K
02J2F	Cirsium oleraceum	0.5	K
02J2F	Euphorbia dulcis	0.5	K
02J2F	Festuca ovina agg	15	K
02J2F	Festuca rubra agg	45	K
02J2F	Mercurialis perennis	0.5	K
02J2F	Solidago gigantea	30	K
02J2F	Taraxacum officinale	0.5	K
02J2Z	Aegopodium podagraria	3	K
02J2Z	Anemone nemorosa	4	K
02J2Z	Brachypodium sylvaticum	5	K
02J2Z	Carex flacca	1	K
02J2Z	Cirsium arvense	30	K
02J2Z	Cirsium oleraceum	7	K
02J2Z	Clinopodium vulgare	2	K
02J2Z	Erigoron annuus	0.5	K
02J2Z	Festuca ovina agg	14	K
02J2Z	Festuca rubra agg	45	K
02J2Z	Mercurialis perennis	1	K
02J2Z	Rubus fruticosus agg	2	K
02J2Z	Solidago gigantea	35	K
02J2Z	Taraxacum officinale	1	K
02J3F	Aegopodium podagraria	1	K
02J3F	Anemone nemorosa	15	K
02J3F	Arum maculatum	0.5	K
02J3F	Brachypodium sylvaticum	10	K

02J3F	Carpinus betulus	30	S
02J3F	Cirsium arvense	7	K
02J3F	Cirsium oleraceum	2	K
02J3F	Dactylis glomerata	1	K
02J3F	Euphorbia amygdaloides	1	K
02J3F	Festuca ovina agg	5	K
02J3F	Festuca rubra agg	40	K
02J3F	Lolium perenne	2	K
02J3F	Mercurialis perennis	1	K
02J3F	Rubus fruticosus agg	20	K
02J3F	Solidago canadensis	1	K
02J3F	Solidago gigantea	1	K
02J3F	Vicia sepium	1	K
02J3Z	Aegopodium podagraria	1	K
02J3Z	Anemone nemorosa	20	K
02J3Z	Arum maculatum	2	K
02J3Z	Brachypodium sylvaticum	20	K
02J3Z	Carpinus betulus	5	S
02J3Z	Cirsium arvense	3	K
02J3Z	Cirsium oleraceum	4	K
02J3Z	Cirsium vulgare	3	K
02J3Z	Festuca rubra agg	10	K
02J3Z	Galium mollugo	0.5	K
02J3Z	Lolium perenne	15	K
02J3Z	Mercurialis perennis	1	K
02J3Z	Rubus fruticosus agg	2	K
02J3Z	Solidago gigantea	7	K
02J3Z	Taraxacum officinale	1	K
02J3Z	Vicia sepium	1	K
02M1F	Arum maculatum	1	K
02M1F	Brachypodium sylvaticum	3	K
02M1F	Carpinus betulus	30	S
02M1F	Festuca rubra agg	15	K
02M1F	Rubus fruticosus agg	4	K
02M1F	Solidago gigantea	35	K
02M1Z	Agrostis stolonifera	5	K
02M1Z	Anemone nemorosa	1	K
02M1Z	Angelica sylvestris	0.5	K
02M1Z	Arum maculatum	1	K
02M1Z	Brachypodium sylvaticum	2	K
02M1Z	Calamagrostis epigejos	3	K
02M1Z	Carex sylvatica	5	K
02M1Z	Euphorbia amygdaloides	2	K
02M1Z	Festuca rubra agg	10	K
02M1Z	Rubus fruticosus agg	3	K
02M1Z	Salix spec	5	S
02M1Z	Solidago gigantea	35	K
02M2F	Calamagrostis epigejos	10	K

02M2F	Carpinus betulus	20	S
02M2F	Lamium maculatum	1	K
02M2F	Rubus fruticosus agg	2	K
02M2F	Solidago gigantea	3	K
02M2F	Valeriana officinalis	5	K
02M2Z	Anemone nemorosa	2	K
02M2Z	Arum maculatum	1	K
02M2Z	Calamagrostis epigejos	10	K
02M2Z	Carpinus betulus	5	S
02M2Z	Corylus avellana	10	S
02M2Z	Fraxinus excelsior	5	S
02M2Z	Lamium maculatum	1	K
02M2Z	Rubus fruticosus agg	2	K
02M2Z	Valeriana officinalis	5	K
02M3F	Cirsium arvense	1	K
02M3F	Cirsium oleraceum	7	K
02M3F	Quercus robur	30	S
02M3F	Rubus fruticosus agg	10	K
02M3F	Solidago gigantea	20	K
02M3Z	Anemone nemorosa	3	K
02M3Z	Carex sylvatica	1	K
02M3Z	Cirsium arvense	1	K
02M3Z	Quercus robur	10	S
02M3Z	Rubus fruticosus agg	10	K
02M3Z	Solidago canadensis	3	K
02M3Z	Solidago gigantea	40	K
02R10	Acer spec	0.5	K
02R10	Anemone nemorosa	7	K
02R10	Arum maculatum	7	K
02R10	Asarum europaeum	0.5	K
02R10	Campanula trachelium	0.5	K
02R10	Carex alba	50	K
02R10	Clematis vitalba	1	K
02R10	Corylus avellana	15	S
02R10	Euphorbia amygdaloides	1	K
02R10	Hedera helix	5	K
02R10	Lamium maculatum	2	K
02R10	Mercurialis perennis	60	K
02R10	Oxalis acetosella	2	K
02R10	Picea abies	100	B
02R10	Thuidium tamariscinum	5	M
02R20	Anemone nemorosa	40	K
02R20	Arum maculatum	5	K
02R20	Asarum europaeum	3	K
02R20	Cornus sanguinea	5	S
02R20	Cornus sanguinea	1	K
02R20	Corylus avellana	95	S
02R20	Fagus sylvatica	2	S

02R20	Galium aparine	1	K
02R20	Hedera helix	10	K
02R20	Lamium maculatum	3	K
02R20	Listera ovata	1	K
02R20	Mercurialis perennis	40	K
02R20	Oxalis acetosella	5	K
02R20	Picea abies	50	B
02R20	Quercus robur	50	B
02R20	Rubus idaeus	1	K
02R20	Thuidium tamariscinum	5	M
02R20	Ulmus glabra	1	K
02R20	Viola reichenbachiana	1	K
02R30	Aegopodium podagraria	5	K
02R30	Anemone nemorosa	50	K
02R30	Arum maculatum	10	K
02R30	Asarum europaeum	5	K
02R30	Brachypodium sylvaticum	1	K
02R30	Carex sylvatica	1	K
02R30	Cornus sanguinea	5	S
02R30	Corylus avellana	70	S
02R30	Euphorbia amygdaloides	1	K
02R30	Fraxinus excelsior	1	K
02R30	Galium odoratum	1	K
02R30	Glechoma hederacea	1	K
02R30	Hedera helix	4	K
02R30	Lamium maculatum	5	K
02R30	Lonicera xylosteum	20	S
02R30	Maianthemum bifolium	5	K
02R30	Mercurialis perennis	80	K
02R30	Oxalis acetosella	10	K
02R30	Picea abies	70	B
02R30	Polygonatum multiflorum	1	K
02R30	Primula elatior	1	K
02R30	Quercus robur	10	B
02R30	Rubus fruticosus agg	5	K
02R30	Thuidium tamariscinum	5	M
02R30	Viola reichenbachiana	1	K
02U10	Aegopodium podagraria	5	K
02U10	Anemone nemorosa	75	K
02U10	Arum maculatum	10	K
02U10	Asarum europaeum	5	K
02U10	Cornus sanguinea	30	S
02U10	Corylus avellana	90	S
02U10	Hedera helix	20	K
02U10	Hypnum cupressiforme	5	M
02U10	Lamium maculatum	2	K
02U10	Mercurialis perennis	10	K
02U10	Rubus fruticosus agg	5	K

11A10	Arum maculatum	1	K
11A10	Brachypodium sylvaticum	1	K
11A10	Carex alba	2	K
11A10	Carex sylvatica	2	K
11A10	Clematis vitalba	1	K
11A10	Clinopodium vulgare	1	K
11A10	Cornus sanguinea	1	S
11A10	Fraxinus excelsior	1	S
11A10	Galium mollugo	3	K
11A10	Listera ovata	0.5	K
11A10	Mercurialis perennis	2	K
11A10	Solidago gigantea	30	K
11A10	Ulmus glabra	3	S
11A20	Cardamine pratensis agg	1	K
11A20	Carex alba	4	K
11A20	Carex sylvatica	1	K
11A20	Crataegus spec	3	K
11A20	Hypnum cupressiforme	50	M
11A20	Ligustrum vulgare	2	K
11A20	Lonicera xylosteum	3	K
11A20	Salix spec	3	K
11A20	Solidago gigantea	10	K
11A30	Arum maculatum	2	K
11A30	Brachypodium sylvaticum	0.5	K
11A30	Carex alba	3	K
11A30	Carex sylvatica	4	K
11A30	Colchicum autumnale	0.5	K
11A30	Euonymus europaeus	1	K
11A30	Hedera helix	1	K
11A30	Hypnum cupressiforme	50	M
11A30	Mercurialis perennis	1	K
11A30	Picea abies	0.5	K
11A30	Quercus robur	10	S
11A30	Solidago gigantea	15	K
11A30	Valeriana officinalis	1	K
11J1F	Carex alba	3	K
11J1F	Carex flacca	2	K
11J1F	Dactylis glomerata	15	K
11J1F	Elymus repens	15	K
11J1F	Fagus sylvatica	30	S
11J1F	Festuca ovina agg	3	K
11J1F	Festuca rubra agg	7	K
11J1F	Fragaria vesca	3	K
11J1F	Lathyrus pratensis	2	K
11J1F	Rubus fruticosus agg	3	K
11J1F	Solidago gigantea	0.5	K
11J1F	Taraxacum officinale	1	K
11J1F	Trifolium repens	4	K

11J1Z	Anemone nemorosa	0.5	K
11J1Z	Brachypodium sylvaticum	1	K
11J1Z	Carex alba	1	K
11J1Z	Carex flacca	2	K
11J1Z	Carex sylvatica	2	K
11J1Z	Clinopodium vulgare	2	K
11J1Z	Dactylis glomerata	20	K
11J1Z	Elymus repens	50	K
11J1Z	Euphorbia cyparissias	1	K
11J1Z	Festuca ovina agg	2	K
11J1Z	Festuca rubra agg	5	K
11J1Z	Fragaria vesca	3	K
11J1Z	Rhamnus frangula	0.5	K
11J1Z	Rubus fruticosus agg	2	K
11J1Z	Solidago gigantea	2	K
11J1Z	Trifolium repens	5	K
11J1Z	Viola riviniana	1	K
11J2F	Aegopodium podagraria	5	K
11J2F	Angelica sylvestris	2	K
11J2F	Berberis vulgaris	7	K
11J2F	Dactylis glomerata	20	K
11J2F	Elymus repens	20	K
11J2F	Fagus sylvatica	10	S
11J2F	Festuca rubra agg	10	K
11J2F	Galium mollugo	4	K
11J2F	Larix decidua	20	S
11J2F	Rubus fruticosus agg	30	K
11J2F	Trifolium repens	1	K
11J2F	Vicia sepium	2	K
11J2Z	Aegopodium podagraria	2	K
11J2Z	Agrostis stolonifera	5	K
11J2Z	Angelica sylvestris	3	K
11J2Z	Berberis vulgaris	10	S
11J2Z	Brachypodium sylvaticum	5	K
11J2Z	Carex alba	1	K
11J2Z	Cornus sanguinea	7	S
11J2Z	Dactylis glomerata	10	K
11J2Z	Elymus repens	35	K
11J2Z	Euphorbia amygdaloides	2	K
11J2Z	Festuca rubra agg	5	K
11J2Z	Fragaria vesca	3	K
11J2Z	Galium mollugo	3	K
11J2Z	Larix decidua	5	S
11J2Z	Rosa canina agg	1	K
11J2Z	Rubus fruticosus agg	30	K
11J2Z	Solidago gigantea	3	K
11J2Z	Taraxacum officinale	1	K
11J2Z	Trifolium repens	2	K

11J3F	Dactylis glomerata	15	K
11J3F	Elymus repens	45	K
11J3F	Euonymus europaeus	3	K
11J3F	Euphorbia amygdaloides	2	K
11J3F	Fagus sylvatica	5	S
11J3F	Festuca rubra agg	5	K
11J3F	Larix decidua	5	S
11J3F	Rubus fruticosus agg	7	K
11J3F	Solidago gigantea	0.5	K
11J3F	Taraxacum officinale	0.5	K
11J3F	Trifolium repens	5	K
11J3F	Vicia cracca	3	K
11J3Z	Cirsium arvense	1	K
11J3Z	Dactylis glomerata	10	K
11J3Z	Elymus repens	40	K
11J3Z	Euphorbia amygdaloides	1	K
11J3Z	Festuca rubra agg	5	K
11J3Z	Galium mollugo	0.5	K
11J3Z	Rubus fruticosus agg	5	K
11J3Z	Solidago gigantea	3	K
11J3Z	Trifolium repens	7	K
11J3Z	Vicia cracca	5	K
11M1F	Ajuga reptans	3	K
11M1F	Encalypta vulgaris	2	M
11M1F	Euphorbia amygdaloides	2	K
11M1F	Festuca ovina agg	3	K
11M1F	Festuca rubra agg	50	K
11M1F	Larix decidua	20	S
11M1F	Mercurialis perennis	1	K
11M1F	Solidago gigantea	2	K
11M1F	Taraxacum officinale	2	K
11M1F	Tortella inclinata	3	M
11M1Z	Aegopodium podagraria	2	K
11M1Z	Brachypodium sylvaticum	1	K
11M1Z	Carex sylvatica	1	K
11M1Z	Cirsium arvense	3	K
11M1Z	Encalypta vulgaris	2	M
11M1Z	Euphorbia amygdaloides	1	K
11M1Z	Festuca ovina agg	5	K
11M1Z	Festuca rubra agg	50	K
11M1Z	Fragaria vesca	3	K
11M1Z	Galium mollugo	2	K
11M1Z	Lamium album	1	K
11M1Z	Larix decidua	5	S
11M1Z	Lolium perenne	2	K
11M1Z	Mercurialis perennis	3	K
11M1Z	Rubus fruticosus agg	4	K
11M1Z	Solidago gigantea	5	K

11M1Z	Taraxacum officinale	5	K
11M1Z	Tortella inclinata	3	M
11M1Z	Verbascum thapsus	1	K
11M1Z	Viola riviniana	0.5	K
11M2F	Ajuga reptans	4	K
11M2F	Clinopodium vulgare	2	K
11M2F	Encalypta vulgaris	2	M
11M2F	Euphorbia cyparissias	1	K
11M2F	Festuca rubra agg	50	K
11M2F	Fragaria vesca	10	K
11M2F	Galium mollugo	1	K
11M2F	Larix decidua	30	S
11M2F	Mercurialis perennis	1	K
11M2F	Rubus fruticosus agg	2	K
11M2F	Solidago gigantea	5	K
11M2F	Taraxacum officinale	2	K
11M2F	Tortella inclinata	2	M
11M2F	Vicia sepium	5	K
11M2F	Viola riviniana	2	K
11M2Z	Ajuga reptans	4	K
11M2Z	Anemone nemorosa	2	K
11M2Z	Cirsium arvense	2	K
11M2Z	Clinopodium vulgare	2	K
11M2Z	Encalypta vulgaris	2	M
11M2Z	Euphorbia cyparissias	1	K
11M2Z	Festuca ovina agg	5	K
11M2Z	Festuca rubra agg	50	K
11M2Z	Fragaria vesca	10	K
11M2Z	Galium mollugo	2	K
11M2Z	Larix decidua	10	S
11M2Z	Mercurialis perennis	2	K
11M2Z	Rubus fruticosus agg	3	K
11M2Z	Solidago gigantea	7	K
11M2Z	Taraxacum officinale	4	K
11M2Z	Tortella inclinata	3	M
11M2Z	Vicia sepium	3	K
11M2Z	Viola riviniana	1	K
11M3F	Ajuga reptans	4	K
11M3F	Carex alba	1	K
11M3F	Cirsium arvense	4	K
11M3F	Euphorbia amygdaloides	5	K
11M3F	Festuca rubra agg	50	K
11M3F	Fragaria vesca	2	K
11M3F	Galium mollugo	4	K
11M3F	Larix decidua	30	S
11M3F	Mercurialis perennis	1	K
11M3F	Rubus fruticosus agg	1	K
11M3F	Solidago gigantea	3	K

11M3F	Taraxacum officinale	2	K
11M3F	Vicia sepium	1	K
11M3Z	Ajuga reptans	7	K
11M3Z	Brachypodium sylvaticum	4	K
11M3Z	Carex alba	0.5	K
11M3Z	Cirsium arvense	5	K
11M3Z	Euphorbia amygdaloides	2	K
11M3Z	Festuca rubra agg	50	K
11M3Z	Fragaria vesca	4	K
11M3Z	Galium mollugo	1	K
11M3Z	Larix decidua	5	S
11M3Z	Lolium perenne	3	K
11M3Z	Mercurialis perennis	2	K
11M3Z	Rubus fruticosus agg	2	K
11M3Z	Solidago gigantea	7	K
11M3Z	Taraxacum officinale	3	K
11M3Z	Tortella inclinata	3	M
11R10	Acer pseudoplatanus	5	S
11R10	Betula pubescens	5	S
11R10	Brachypodium sylvaticum	1	K
11R10	Carex alba	5	K
11R10	Fraxinus excelsior	30	S
11R10	Fraxinus excelsior	2	K
11R10	Hedera helix	7	K
11R10	Hylocomium splendens	60	M
11R10	Ligustrum vulgare	20	S
11R10	Lonicera xylosteum	5	S
11R10	Picea abies	40	B
11R10	Picea abies	30	S
11R10	Pinus sylvestris	30	B
11R10	Polygonatum multiflorum	3	K
11R10	Prunus padus	7	S
11R10	Quercus robur	10	S
11R10	Rubus fruticosus agg	3	K
11R10	Solidago gigantea	1	K
11R10	Ulmus glabra	10	B
11R10	Vinca minor	20	K
11R20	Anemone nemorosa	1	K
11R20	Carex alba	30	K
11R20	Euonymus europaeus	2	K
11R20	Euphorbia amygdaloides	1	K
11R20	Fraxinus excelsior	0.5	K
11R20	Hedera helix	10	K
11R20	Ligustrum vulgare	1	K
11R20	Listera ovata	0.5	K
11R20	Lonicera xylosteum	20	S
11R20	Paris quadrifolia	0.5	K
11R20	Picea abies	90	B

11R20	Prunus avium	7	K
11R20	Ulmus glabra	2	K
11R20	Viburnum lantana	0.5	K
11R20	Vinca minor	15	K
11R30	Berberis vulgaris	5	S
11R30	Brachypodium sylvaticum	2	K
11R30	Carex alba	40	K
11R30	Carex digitata	4	K
11R30	Cornus sanguinea	3	K
11R30	Crataegus spec	15	S
11R30	Euphorbia cyparissias	3	K
11R30	Festuca altissima	4	K
11R30	Fraxinus excelsior	2	K
11R30	Galium mollugo	1	K
11R30	Hedera helix	10	K
11R30	Hylocomium splendens	70	M
11R30	Ligustrum vulgare	1	K
11R30	Lonicera xylosteum	1	K
11R30	Melica nutans	1	K
11R30	Picea abies	90	B
11R30	Quercus robur	1	K
11R30	Rubus fruticosus agg	2	K
11R30	Rubus idaeus	1	K
11R30	Taraxacum officinale	1	K
11R30	Ulmus glabra	2	K
11R30	Viburnum lantana	0.5	K
11R30	Viola riviniana	1	K
12A1F	Astragalus glycyphyllos	7	K
12A1F	Cirsium arvense	2	K
12A1F	Cornus sanguinea	5	K
12A1F	Festuca rubra agg	30	K
12A1F	Fragaria vesca	2	K
12A1F	Ligustrum vulgare	1	K
12A1F	Pinus sylvestris	35	S
12A1F	Rubus fruticosus agg	5	K
12A1F	Solidago canadensis	7	K
12A1F	Solidago gigantea	5	K
12A1F	Taraxacum officinale	1	K
12A1Z	Anemone nemorosa	0.5	K
12A1Z	Arum maculatum	1	K
12A1Z	Astragalus glycyphyllos	15	K
12A1Z	Carex flacca	2	K
12A1Z	Cirsium arvense	3	K
12A1Z	Clinopodium vulgare	1	K
12A1Z	Fagus sylvatica	5	S
12A1Z	Festuca rubra agg	30	K
12A1Z	Fragaria vesca	5	K
12A1Z	Galium mollugo	7	K

12A1Z	Hypericum perforatum	2	K
12A1Z	Lathyrus pratensis	2	K
12A1Z	Pinus sylvestris	20	S
12A1Z	Rubus fruticosus agg	5	K
12A1Z	Solidago canadensis	7	K
12A1Z	Solidago gigantea	30	K
12A1Z	Taraxacum officinale	1	K
12A2F	Aquilegia atrata	15	K
12A2F	Arum maculatum	0.5	K
12A2F	Cirsium arvense	1	K
12A2F	Clematis vitalba	1	K
12A2F	Festuca rubra agg	10	K
12A2F	Fragaria vesca	7	K
12A2F	Galium mollugo	3	K
12A2F	Lathyrus pratensis	3	K
12A2F	Pinus sylvestris	80	S
12A2F	Rubus fruticosus agg	7	K
12A2F	Solidago canadensis	2	K
12A2F	Solidago gigantea	10	K
12A2Z	Anemone nemorosa	1	K
12A2Z	Aquilegia atrata	5	K
12A2Z	Arum maculatum	0.5	K
12A2Z	Cirsium arvense	3	K
12A2Z	Festuca rubra agg	20	K
12A2Z	Fragaria vesca	7	K
12A2Z	Galium mollugo	7	K
12A2Z	Lathyrus pratensis	3	K
12A2Z	Pinus sylvestris	70	S
12A2Z	Solidago canadensis	7	K
12A2Z	Solidago gigantea	20	K
12A3F	Festuca rubra agg	20	K
12A3F	Galium mollugo	1	K
12A3F	Glechoma hederacea	1	K
12A3F	Lathyrus pratensis	2	K
12A3F	Pinus sylvestris	30	S
12A3F	Rubus fruticosus agg	7	K
12A3F	Solidago gigantea	25	K
12A3Z	Festuca rubra agg	20	K
12A3Z	Galium mollugo	1	K
12A3Z	Lathyrus pratensis	3	K
12A3Z	Primula elatior	2	K
12A3Z	Rubus fruticosus agg	7	K
12A3Z	Solidago canadensis	5	K
12A3Z	Solidago gigantea	25	K
12M1F	Cirsium arvense	3	K
12M1F	Fagus sylvatica	10	S
12M1F	Festuca rubra agg	60	K
12M1F	Galium mollugo	3	K

12M1F	Lathyrus pratensis	3	K
12M1F	Pinus sylvestris	10	S
12M1F	Rubus fruticosus agg	5	K
12M1F	Solidago gigantea	2	K
12M1Z	Aquilegia atrata	2	K
12M1Z	Brachypodium sylvaticum	2	K
12M1Z	Cirsium arvense	5	K
12M1Z	Dactylis glomerata	5	K
12M1Z	Euphorbia cyparissias	1	K
12M1Z	Festuca rubra agg	55	K
12M1Z	Galium mollugo	2	K
12M1Z	Hypericum perforatum	3	K
12M1Z	Lathyrus pratensis	3	K
12M1Z	Pinus sylvestris	5	S
12M1Z	Rubus fruticosus agg	3	K
12M1Z	Solidago canadensis	3	K
12M1Z	Solidago gigantea	35	K
12M2F	Festuca rubra agg	60	K
12M2F	Fragaria vesca	2	K
12M2F	Pinus sylvestris	50	S
12M2F	Rhamnus frangula	2	K
12M2F	Rubus fruticosus agg	3	K
12M2F	Solidago gigantea	2	K
12M2Z	Ajuga reptans	2	K
12M2Z	Aquilegia atrata	0.5	K
12M2Z	Carex alba	1	K
12M2Z	Carex flacca	1	K
12M2Z	Cirsium arvense	2	K
12M2Z	Dactylis glomerata	1	K
12M2Z	Encalypta vulgaris	5	M
12M2Z	Euphorbia amygdaloides	1	K
12M2Z	Festuca rubra agg	40	K
12M2Z	Fragaria vesca	3	K
12M2Z	Pinus sylvestris	10	S
12M2Z	Rhamnus frangula	2	K
12M2Z	Rubus fruticosus agg	7	K
12M2Z	Solidago canadensis	5	K
12M2Z	Solidago gigantea	7	K
12M3F	Aquilegia atrata	0.5	K
12M3F	Cirsium arvense	1	K
12M3F	Fagus sylvatica	15	S
12M3F	Festuca rubra agg	80	K
12M3F	Fragaria vesca	1	K
12M3F	Pinus sylvestris	25	S
12M3F	Rubus fruticosus agg	2	K
12M3F	Solidago canadensis	2	K
12M3F	Solidago gigantea	4	K
12M3Z	Carex alba	1	K

12M3Z	Cirsium arvense	2	K
12M3Z	Dactylis glomerata	1	K
12M3Z	Encalypta vulgaris	3	M
12M3Z	Fagus sylvatica	3	S
12M3Z	Festuca rubra agg	70	K
12M3Z	Fragaria vesca	3	K
12M3Z	Galium mollugo	1	K
12M3Z	Ligustrum vulgare	1	K
12M3Z	Pinus sylvestris	7	S
12M3Z	Rhamnus frangula	3	K
12M3Z	Rubus fruticosus agg	3	K
12M3Z	Solidago canadensis	2	K
12M3Z	Solidago gigantea	5	K
12R10	Acer platanoides	0.5	K
12R10	Anemone nemorosa	5	K
12R10	Carex alba	40	K
12R10	Carex ornithopoda	3	K
12R10	Euphorbia cyparissias	0.5	K
12R10	Fraxinus excelsior	7	K
12R10	Hedera helix	7	K
12R10	Hepatica nobilis	5	K
12R10	Hylocomium splendens	30	M
12R10	Ligustrum vulgare	4	K
12R10	Lonicera xylosteum	10	K
12R10	Melica nutans	5	K
12R10	Picea abies	80	B
12R10	Rubus fruticosus agg	2	K
12R10	Sesleria caerulea	40	K
12R10	Sorbus aucuparia	2	K
12R10	Taraxacum officinale	0.5	K
12R10	Ulmus glabra	5	K
12R10	Viburnum lantana	12	K
12R10	Vinca minor	15	K
12R20	Acer spec	0.5	K
12R20	Anemone nemorosa	5	K
12R20	Carex alba	50	K
12R20	Carex ornithopoda	20	K
12R20	Clematis vitalba	7	K
12R20	Euphorbia amygdaloides	10	K
12R20	Euphorbia cyparissias	2	K
12R20	Fraxinus excelsior	2	K
12R20	Hedera helix	7	K
12R20	Hepatica nobilis	1	K
12R20	Hylocomium splendens	40	M
12R20	Melica nutans	2	K
12R20	Oxalis acetosella	1	K
12R20	Picea abies	90	B
12R20	Picea abies	0.5	K

12R20	Pinus sylvestris	2	K
12R20	Quercus robur	0.5	K
12R20	Rhamnus frangula	2	K
12R20	Rubus fruticosus agg	5	K
12R20	Viola riviniana	0.5	K
12R30	Berberis vulgaris	25	S
12R30	Berberis vulgaris	10	K
12R30	Carex alba	35	K
12R30	Clematis vitalba	10	K
12R30	Cornus sanguinea	3	K
12R30	Euphorbia amygdaloides	0.5	K
12R30	Fagus sylvatica	1	K
12R30	Fraxinus excelsior	5	K
12R30	Hedera helix	15	K
12R30	Hepatica nobilis	10	K
12R30	Hylocomium splendens	40	M
12R30	Ilex aquifolium	5	K
12R30	Lonicera xylosteum	5	K
12R30	Picea abies	100	B
12R30	Rubus fruticosus agg	70	K
12R30	Vinca minor	20	K
12U10	Angelica sylvestris	2	K
12U10	Berberis vulgaris	3	K
12U10	Carex alba	50	K
12U10	Carex flacca	7	K
12U10	Carex ornithopoda	5	K
12U10	Clematis vitalba	10	K
12U10	Cornus sanguinea	5	K
12U10	Euphorbia amygdaloides	10	K
12U10	Fraxinus excelsior	3	K
12U10	Galium mollugo	7	K
12U10	Lamium maculatum	1	K
12U10	Lonicera xylosteum	20	K
12U10	Melica nutans	10	K
12U10	Rhamnus frangula	10	K
12U10	Rosa canina agg	3	K
12U10	Rubus fruticosus agg	40	K
12U10	Sorbus aucuparia	10	S
12U10	Ulmus glabra	10	S
12U20	Agrostis stolonifera	3	K
12U20	Angelica sylvestris	0.5	K
12U20	Brachypodium sylvaticum	5	K
12U20	Carex alba	45	K
12U20	Carex digitata	5	K
12U20	Carex ornithopoda	5	K
12U20	Cirsium arvense	3	K
12U20	Cornus sanguinea	5	K
12U20	Euphorbia amygdaloides	0.5	K

12U20	Galium mollugo	7	K
12U20	Hedera helix	3	K
12U20	Hepatica nobilis	1	K
12U20	Ilex aquifolium	5	K
12U20	Lathyrus pratensis	3	K
12U20	Ligustrum vulgare	3	K
12U20	Lonicera xylosteum	7	K
12U20	Melica nutans	5	K
12U20	Rubus fruticosus agg	5	K
12U20	Sesleria caerulea	10	K
12U20	Solidago gigantea	1	K
12U20	Taraxacum officinale	0.5	K
12U20	Viburnum lantana	7	K
12U20	Viburnum opulus	3	K
12U30	Angelica sylvestris	5	K
12U30	Brachypodium sylvaticum	5	K
12U30	Carex alba	40	K
12U30	Carex digitata	5	K
12U30	Carex ornithopoda	1	K
12U30	Euphorbia amygdaloides	20	K
12U30	Galium mollugo	2	K
12U30	Hepatica nobilis	3	K
12U30	Hypnum cupressiforme	3	M
12U30	Lamium album	1	K
12U30	Lamium maculatum	3	K
12U30	Melica nutans	15	K
12U30	Rhamnus frangula	4	K
12U30	Rubus fruticosus agg	2	K
12U30	Taraxacum officinale	1	K
12U30	Viola riviniana	2	K