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Soil carbon sequestration in Switzerland: analysis of potentials and measures (Postulate Bourgeois 19.3639)

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Imprint

This report was prepared under the contract of the Swiss Federal Office for the Environment (FOEN). The contractor bears sole responsibility for the content.

Commissioned by: Federal Office for the Environment (FOEN), Soil and Biotechnology Division, 3003 Bern, Switzerland. The FOEN is an agency of the Federal Department of the Environment, Transport, Energy and Communications (DETEC).

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To cite this report:
Keel, S.G., Johannes, A., Boivin, P., Burgos, S., Charles, R., Hagedorn, F., Kulli, B., Leifeld, J., Saluz, A., Zimmermann, S. (2021) Soil carbon sequestration in Switzerland: analysis of potentials and measures (Postulate Bourgeois 19.3639). Report by Agroscope. Commissioned by the Federal Office for the Environment, Bern.

To cite an individual chapter:
[name of authors] (2021), '[title of chapter]' in Keel et al., *Soil carbon sequestration in Switzerland: analysis of potentials and measures (Postulate Bourgeois 19.3639)*, Chapter [nr]. Report by Agroscope. Commissioned by the Federal Office for the Environment, Bern.

Cover photo by Alice Johannes

Abbreviations used

ARC: Annual rate of change
C: Carbon
CA: Conservation agriculture
CH_4: Methane
CO_2: Carbon dioxide
CT: Conventional tillage
FOAG: Federal Office for Agriculture
FOEN: Federal Office for the Environment
FSO: Federal Statistical Office
FYM: farm yard manure
GHG: Greenhouse gas
LTE: Long-term field experiment
N_2O: Nitrous oxide
NABO: Swiss National Soil Monitoring Network
NPP: Net primary productivity
ÖLN: Ökologischer Leistungsnachweis
OM: Organic matter
PEP: Proof of ecological performance
PER: Prestations ecologiques requises
SOC: Soil organic carbon
SOM: Soil organic matter
STIR: Soil Tillage Intensity Rating

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Summary

To reach net zero greenhouse gas emissions by 2050, Switzerland will depend on domestic negative emissions of about 7 million tons of CO_2-equivalents per year (Mt CO_2-eq yr^{-1}). Soil carbon sequestration is one of the cheapest and technically least demanding technologies. It is defined as a net uptake of atmospheric carbon dioxide (CO_2) that leads to an increase in soil organic carbon storage on the same unit, where CO_2 was taken up by photosynthesis. Compared to other technologies it has the great advantage that it rarely competes with food production and is often associated with environmental benefits. Furthermore, higher soil organic carbon stocks increase soil fertility and improve the resilience of the soil system to climate change. The main disadvantage however, is that soil carbon sequestration does not lead to a permanent storage of carbon and most measures are only effective for a few decades. This report addresses questions 1 and 2 of the postulate Nr. 19.3639 'Kohlenstoffsequestrierung in Böden' by national council Jacques Bourgeois. In part 1, which addresses question 1, we assess the potential to store additional carbon in Swiss soils and show what would be necessary to improve our understanding of the actual soil carbon sequestration potentials. In part 2, which addresses question 2, we discuss advantages and disadvantages of specific measures to enhance soil organic carbon stocks. Because there are great differences between different land use types or soil categories, we discuss the measures separately for organic and mineral soils and distinguish unmanaged, agricultural, forest and settlement soils.

In Switzerland soil carbon sequestration potentials are largest on agricultural mineral soils. As a result of historic land use conversions (mainly deforestation and drainage) and an intensification of agricultural use, these soils have lost significant amounts of carbon and current soil organic carbon stocks are rather low, especially on cropland. Part of the lost carbon could be regained by measures that increase soil organic carbon stocks. Permanent grassland and forest soils have higher soil organic carbon stocks and the potential for sequestration is therefore small. However, these high stocks might be at risk under climate change and efforts should focus on maintaining soil organic carbon stocks. Due the small area, settlement soils offer a limited potential for carbon sequestration. Organic soils store significant amounts of carbon but drainage-induced loss rates are high. Efforts should focus on reducing these emissions before their potential to store additional carbon can be considered. Generally, the potential for additional carbon storage is site specific and depends on current soil organic carbon stocks and management. National-scale estimates of soil carbon sequestration potentials are still highly uncertain. To improve estimates, we rely on soil organic carbon maps and spatially explicit management information.

On agricultural mineral soils the measure with the highest potential is conservation agriculture (0.52–1.05 Mt CO_2-eq yr^{-1}) as it could be applied on a large area. Agroforestry and cropland to grassland conversions lead to a reduction of cropping areas and their application is only recommended for selected areas. The estimated potentials are 0–0.12 Mt CO_2-eq yr^{-1} for agroforestry and 0.05 Mt CO_2-eq yr^{-1} for cropland to grassland conversions. In the case of agroforestry, a significant carbon sink is expected in wood, which is not included here. Which measures would be most effective on grassland soils is not clear yet. Generally, it is important to note that additions of organic fertilizer, which can be an integral part of several measures, only count as a true sequestration measure if the biomass was produced

on-farm (also excluding feed imports). Furthermore, it is important to add that the sequestration potentials presented only refer to topsoils due to a lack of data. For a full carbon accounting, effects on subsoils (below 30 cm depth) would need to be included. Biochar as another option for agricultural soils is not considered in this report, but in an accompanying study.

Measures to enhance soil organic carbon stocks on forest soils include the selection of tree species, liming or wood ash application. However, they are all expected to have small effects on total soil organic carbon stocks. Afforestation on former cropland is the only measure that could lead to significantly higher soil organic carbon stocks, but would conflict with food production. However, afforestation generally leads to additional carbon storage in woody biomass.

In settlements, creating new areas for carbon accumulation such as green roofs offers a potential for soil carbon sequestration of 0.07 Mt CO_2-eq yr^{-1}. This measure can have positive effects on urban climate and local biodiversity. The inclusion of biochar underneath newly built roads, could sequester 0.37 Mt CO_2-eq yr^{-1}. Biochar could also be used in tree substrates and would have positive side-effects on water uptake and retention.

Drained organic soils emit 0.51–0.69 Mt CO_2-eq yr^{-1}. Measures should focus mainly on reducing these losses as soil carbon sequestration is difficult to achieve on degraded peatlands. The most promising measure to reduce emissions is rewetting, but the consequence is a severe impairment of the production function. Most likely soil covering and soil mixing cannot reduce CO_2 losses.

Overall, most measures to sequester carbon in mineral soils and reduce carbon losses from organic soils are relatively well known and several measures are ready to be implemented. However, careful selection of sites and measures is highly recommended as the potential to sequester carbon is strongly-site specific and any potential side-effects such as yield reductions need to be factored in. In summary, soil carbon sequestration in Switzerland could offset an upper maximum of 24% of the domestic negative emissions based on the presented measures.

1. Analysis of the SOC sequestration potential of Swiss soils

1.1. Theoretical concept of organic carbon storage in soils

By Jens Leifeld, Frank Hagedorn and Sonja G. Keel

1.1.1. Concepts and terms

1.1.1.1. Soil organic matter and soil organic carbon

Every soil contains soil organic carbon (SOC) in various forms. Most important is soil organic matter (SOM), i.e. the sum of organic substances derived from organisms (plants, microbes and animals), their decomposition residues and transformation products, and residues from vegetation fires (char). SOM contains approximately 50% carbon (C) by weight, the rest being mostly oxygen, hydrogen and nitrogen. SOM does not include living organisms and organs (roots, animals, microbes), but the analytical distinction between living and dead material in soil is challenging. SOM comprises a large variability of molecules of which many are unknown and cannot be assigned to compound classes. Physically, SOM molecules i) are associated with the surface of various soil minerals, particularly of the clay-sized fraction, ii) are incorporated into smaller and larger soil aggregates, and iii) occur freely in soil pores or on the soil surface as litter. In mineral soils, SOM contents are typically a few percent only, whereas in organic soils (peatlands) and in forest floors on top of forest soils, SOM makes up most of the soil mass.

1.1.1.2. Input–output, turnover, turnover time, SOC stock, pool

The SOC stock is the amount of SOC, by mass, stored in a volume of soil. It is calculated as the product of SOC concentration (measured in the laboratory), soil bulk density (measured in the laboratory) and available soil volume (surface area * soil depths minus volume of stones). Often, the stock is reported on a per area basis, but the information on the respective soil depth it refers to is crucial for any evaluation and comparison of SOC stocks.

Under constant environmental conditions regarding land use, management and climate, SOC stocks are principally assumed to be in dynamic equilibrium, i.e. they oscillate around a site-specific mean and variations occur in the order of years. For example, C stocks in agricultural soils will change among years with climatic conditions, crop rotation and management, but the long-term average may show no trend if management is unchanged. Such a situation is referred to as steady state. In steady state, the size of the stock is the product of two factors, the C input rate, and the C turnover time. The C turnover time is the reciprocal of the C turnover rate (leading to C output mainly due to mineralization, Figure 1).

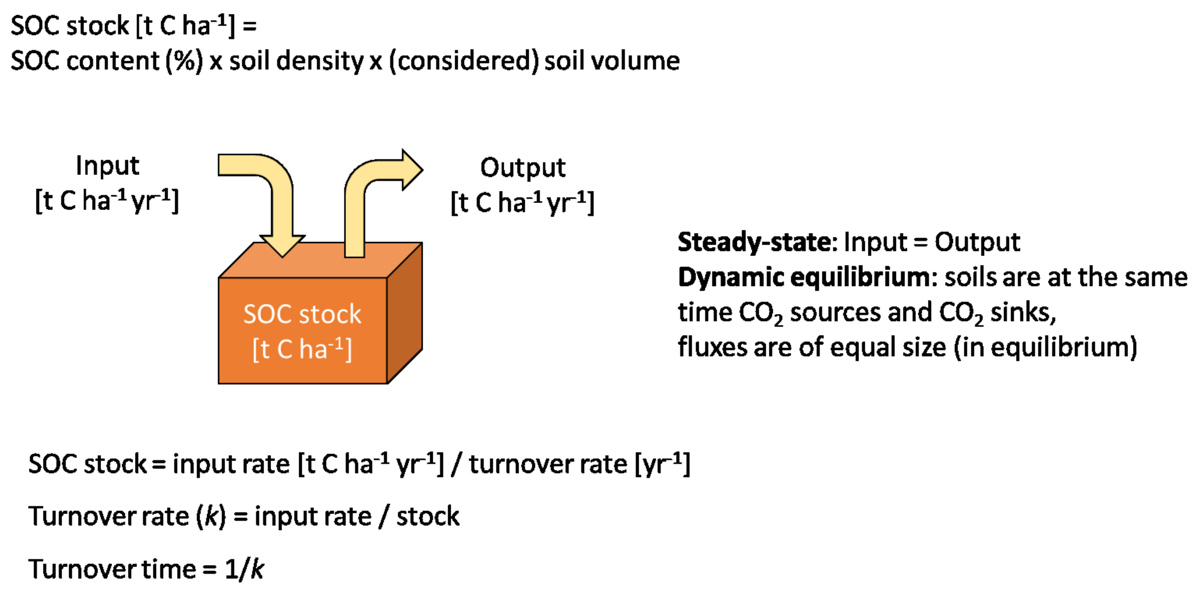

Figure 1: Illustration of soil organic carbon (SOC) stock and its relationship to carbon (C) input and turnover rates.

For example, with an input rate of annually 3 t organic C per hectare to the topsoil and a C turnover time of 20 years in the topsoil, the resulting steady-state stock amounts to 60 t SOC per hectare (0–20 cm). It is important to note that the average turnover time provides no information about the turnover time of single SOM compounds. The variability in turnover times for specific compounds is huge, ranging from minutes to centuries.

One consequence of the steady-state concept is that outputs equal inputs, when averaged over a longer time span of several years or decades. In such a situation, the soil is neither a carbon dioxide (CO_2) source nor a CO_2 sink because the fluxes are of equal size. However, the soil is a SOC store.

The term 'pool' is sometimes used synonymously to 'stock', but more often it refers to a fundamental concept that is frequently used for modelling SOC dynamics. In models, a C pool is operationally defined by its specific decomposition rate constant. Such a mathematically defined pool must not necessarily find its counterpart in the real world, but pool-based modelling has been proven useful to simulate changes in SOC with management (e.g. *Franko et al. 2011*).

1.1.1.3. Dynamic equilibrium versus saturation

The steady-state concept implies that the C stock increases linearly with increasing inputs towards a new steady state when average turnover times remain constant, and, vice versa, the C stock decreases with decreasing inputs towards a new steady state. The concept of equilibrium can be visualized in Figure 2.

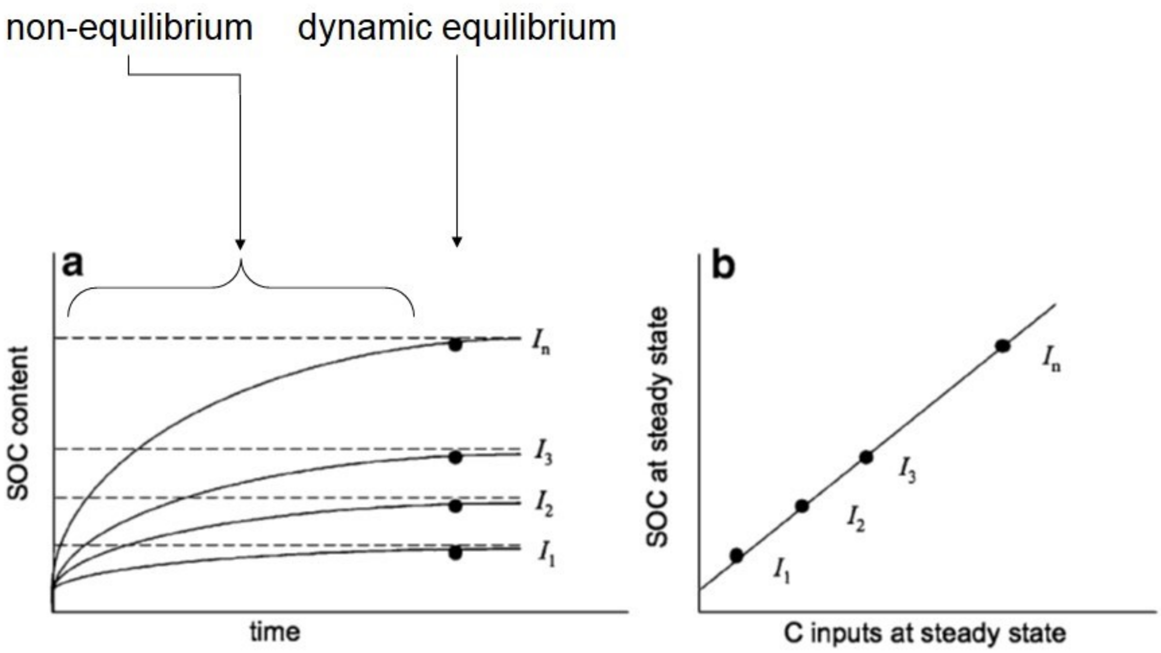

Figure 2: Concept of dynamic equilibrium where steady-state soil organic carbon (SOC) stocks depend linearly on the carbon (C) input rate (from Stewart et al. 2007).

In long-term field experiments, the soil's equilibrium C concentration has empirically been shown to strongly depend on the C input rate (e.g. *Buyanovsky & Wagner 1998; Johnston et al. 2009*). The rate of SOC loss, which is observed in many Swiss long-term experiments (*Keel et al. 2019*), is related to input rates as well (*Leifeld et al. 2009; Oberholzer et al. 2014*). Hence, the concept is supported by evidence from field experiments. It is important to note that the C stock is site specific because the C turnover time depends on factors such as soil properties, hydrology, climate and soil management. Hence, the steady-state stock under the same input rates differs between sites.

An extension of the concept of dynamic equilibrium is the so-called saturation concept. The concept says that the C stock does not linearly increase with increasing inputs but reaches a (site-specific) maximum even with very high inputs. Carbon saturation can be visualized in Figure 3.

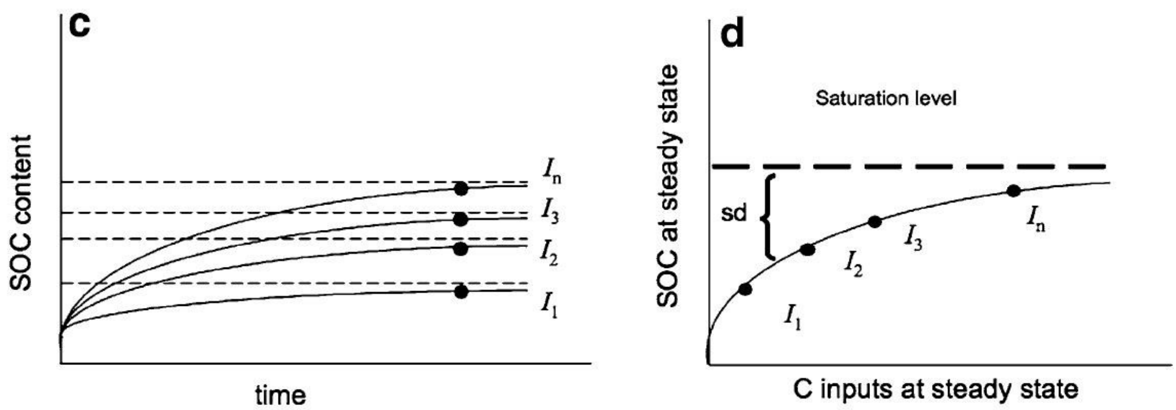

Figure 3: Concept of soil organic carbon(SOC) saturation where steady-state carbon (C) stocks are not linearly related to input rate but reach a site-specific maximum (from Stewart et al. 2007).

In the saturation concept, the difference between the actual stock and the maximum achievable stock is called saturation deficit (sd in Figure 3). The concept implies that soil C turnover rates increase with increasing C stock. Saturation has been described for some long-term field experiments (e.g. *Stewart et al. 2007*), whereas it was not observed for others most likely because organic matter (OM) inputs were too small (*Feng et al. 2014*).

It is important to note that the saturation concept is conceptual in the first place and does not intrinsically refer to specific processes. There are two mechanisms that are discussed with respect to saturation:

A: Limited protective capacity of soil minerals

It has long been recognized that the amount and reactivity of soil particles in the clay and fine silt size fractions is a major factor for the amount of SOM a soil stores (*Hassink 1997*), and it has been shown that clay and silt fractions harbour the greatest part of SOM. In very sandy soils, up to 90% of SOM are associated with these fine fractions (*Christensen 1996*). With an increasing percentage of clay and fine silt, the amount of SOM associated with these fractions increases linearly *(Hassink 1997)*. *Six et al. (2002)* showed that the overall level of SOC associated with the fine fraction is on average higher in grassland or forest than in cropland soils. Under the assumption that soils with permanent vegetation are at their upper end of storing SOC (i.e. they are saturated), the difference in C stock between the lower level in cropland soils and the higher level in permanently vegetated soils might be considered as the above-mentioned saturation deficit. Various ongoing developments of the concept exist, and the supposed saturation deficits are calculated including more explanatory variables. However, a fundamental weakness of the concept is that soil mineral surfaces are associated with SOM to only a small extent as revealed by new techniques *(Vogel et al. 2014)*. Hitherto, it remains unclear as to why a high fraction of mineral surfaces even in biologically active topsoils is bare. Furthermore, the relationship between silt + clay content of soil and the amount of silt- and clay-protected soil C varies with different types of land use activities and clay type. From the analysis of soil fractions from long-term experiments in Canada, *Yang et al. (2016)* concluded that the C storage capacity of a soil can be determined by both the amount of fine particles and the rate of C input, rather than solely by the amount of fine soil particles. In summary, the concept of protective capacity might be useful at an operational level, but it cannot be regarded as a complete mechanistic explanation of SOC storage in fine soil fractions.

B: Reaction kinetics of the substrate–enzyme complex

The kinetics of substrate–enzyme complexes as described by the Michaelis–Menten equation is fundamental to the understanding of OM processing. This enzyme kinetics considers a higher decomposition rate at higher substrate concentration, i.e. a low decomposition rate at low substrate concentrations. Consequently, the carbon turnover is faster at high SOC concentrations, as has been shown also experimentally *(Don et al. 2013).* The saturation of the enzyme–substrate complex as described by these kinetics is of little relevance for SOM, because the amount of enzymes is rather limiting for turnover *(Ekschmitt et al. 2005)*, but unfortunately a reason for an ambiguity of the term. Rather, the coincidence of increasing decomposition rates with increasing substrate concentrations over a wide range of concentrations implies that the relationship of input rate to steady-state stock cannot be linear but should be curved towards an upper limit of SOC storage. This behaviour is independent of any effect of soil texture or mineralogy.

1.1.1.4. Stabilization

Stabilization is a collective term for processes that reduce the intrinsic decomposition rate of a substrate as it would occur under ideal conditions. Whereas the intrinsic stability, i.e. decomposition rate, is largely determined by the chemical structure of the compound and the enzymatic portfolio and activity of the decomposers, this can change in a soil environment. The most important stabilization process works via organo-mineral interactions, in which organic molecules attach to soil minerals in various ways, depending on the nature of both the organic and the mineral phase (e.g. *Kleber et al. 2007*). Associations achieved this way are difficult to break (e.g. *Kaiser & Guggenberger 2007; Hemingway et al. 2019*) and hence prevent molecules from fast decomposition. The longer turnover times of SOM in subsoil can partly be explained by the high ratio of soil minerals to organic matter, i.e. a higher capacity of reducing the biological decomposition rate. This relationship also explains why soils with a high clay content tend to show higher SOC stocks. Organic matter associated with soil minerals typically has a strong microbial imprint, i.e. microbial residues and their transformation products make up a large share of the material stabilized this way (*Guggenberger et al. 1994; Gies et al. 2021*). A second important stabilization mechanism occurs via soil aggregation. Aggregates are higher-level structural units of various sizes that can protect OM from decomposition because they reduce accessibility of the substrate to decomposers (*von Lützow et al. 2006*). Large aggregates are even able to reduce decomposition of macro-organic matter such as fresh and thus labile plant residues.

The intrinsic stability of a substrate mentioned above is also referred to as recalcitrance. From the myriads of organic molecules in soil, pyrogenic organic matter is considered highly recalcitrant (*Leng et al. 2019*). It occurs naturally in many soils as a residue from vegetation fires (*Eckmeier et al. 2010; Reisser et al. 2016; Leifeld et al. 2018*) or is artificially added as biochar (a pyrolysis product of a technical process). The decomposition rates of most other identifiable molecules in soil are in the same order of magnitude (*Amelung et al. 2008*), suggesting that their interaction with the mineral soil matrix rather than their chemical composition drives their decay.

1.1.1.5. Sequestration, sequestration potential and carbon storage

The term soil carbon sequestration is tightly linked to the soil's function as a carbon store and its function as a buffer in the global CO_2 cycle. According to *Olson et al. (2014)* 'soil carbon sequestration is defined as the process of transferring CO_2 from the atmosphere into the soil of a land unit, through plants, plant residues and other organic solids which are stored or retained in the unit as part of the soil organic matter'. Furthermore, the process of SOC sequestration 'should increase the net SOC storage during and at the end of a study to above the previous pre-treatment baseline'. This is often confused with soil carbon storage, which describes either just the soil's carbon stock or the increase in SOC stocks over time in the soils of a given land unit, which must not necessarily be associated with a net removal of CO_2 from the atmosphere. As *Chenu et al.* (2019) exemplified, 'adding the available manure resources on a given agricultural field rather than spreading it homogeneously over the landscape may locally increase SOC stocks (where manure has been added), but not increase the associated CO_2 removal from the atmosphere at the landscape scale'. This does not exclude lateral transport of OM from contributing to sequestration. However, it implies that if OM is moved laterally in the landscape, effects at both spots, i.e. site of origin of the OM and site of its application, must be quantified to get the net effect. This distinction is very important when evaluating the role of soil for mitigating climate change, because only soil carbon sequestration as defined above is a valid approach. What makes the situation even more complicated is the fact that organic fertilizer such as manure can be produced from feed that is imported. The atmospheric CO_2 uptake has therefore taken place in the country of origin and not in Switzerland.

The sequestration potential is thus the difference between the actual SOC stock and the maximum SOC stock that can be achieved via a net removal of CO_2 from the atmosphere under a given climate and for a specified time span. The potential is always site specific. As explained earlier, the SOC stock depends on the input and the turnover rate. Maximum input rates are constrained by the maximum net primary productivity of a site and the fate of the produced biomass (i.e. whether harvested biomass is exported or left on site). For agricultural systems, these input rates can be estimated, though with substantial uncertainty especially for belowground inputs (e.g. *Hirte et al. 2018*). Turnover rates, on the other hand, are much more difficult to estimate as well as to manipulate. They vary not only between differently bound organic residues of varying chemistry but also with soil depth and with biotic and abiotic soil conditions such as oxygen availability, soil pH and soil mineralogy; just to name a few. It must also be recalled that there is no single OM turnover rate for a unit of soil, but turnover rates form a continuum with hitherto unknown statistical distribution. It is therefore not yet possible to assign a general sequestration potential to a soil. Rather the stock's increase (or decrease) can be estimated for specific measures with an associated uncertainty. The concept of soil carbon sequestration is illustrated in Figure 4.

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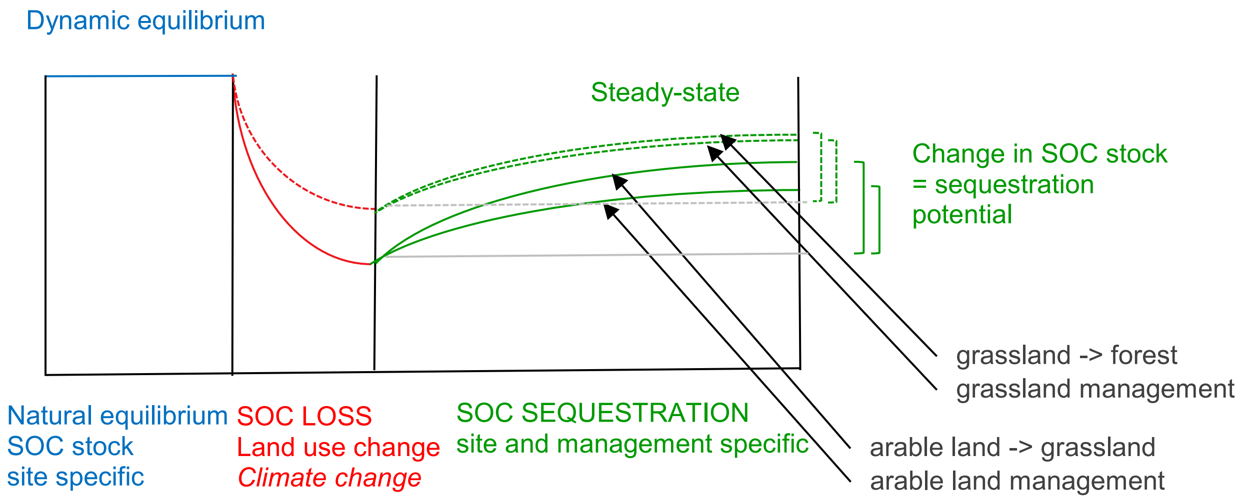

Figure 4 The natural soil organic carbon (SOC) stock is a dynamic equilibrium between carbon inputs and outputs and is site specific (blue line). Losses in SOC stocks can occur in response to e.g. land use change or climate change (red lines). The amount of SOC that can be sequestered depends on the initial SOC stock (end of red lines) and the management applied (green lines). Increases in SOC stocks usually reach a new steady state within about 20 years. The difference in SOC stock for a given time is the sequestration potential.

1.1.2. Factors that control SOC storage

1.1.2.1. Factors that change organic carbon input

The upper limit of carbon input is determined by the net primary productivity (NPP) of the system under study. In managed ecosystems such as agricultural land and many forests, biomass export for food, fibre and wood is the goal, which consequently reduces the available C input to some extent. Earlier estimates for European ecosystems indicated that, on average and over whole management cycles (which can make up decades in the case of forests), the share of NPP export is 47%, 29% and 12% for croplands, grasslands and forests, respectively (*Schulze et al. 2009*). These numbers are subject to large variability, and the share of NPP export varies between 22% and 74% for agriculture, depending on crop type and management practice. A larger harvest export, accompanied with only partial return of OM by organic fertilizers, is considered the main driver for the (on average) smaller SOC stocks in croplands. Management that leads to increasing C inputs must therefore seek measures to increase the NPP of the system and to leave more of the plant biomass on site. Planting cover crops is an option to increase the NPP as discussed in section 2.2.2.7.3.

1.1.2.2. Types of input materials and their properties

The effect of different input materials has mostly been studied in agricultural systems. Experimental trials have shown that crop residues differ regarding their fraction of material that contributes to more stabilized organic matter, a term defined as humification coefficient (or amendment-C retention coefficient). In a Swedish long-term experiment, the humification coefficient was estimated based on the fraction of C input which remained in the soil after about 50 years (*Kätterer et al. 2011*). Root-derived carbon contributed more to relatively stable soil C pools than aboveground crop residues. Based on the same study, the humification coefficient was also assessed for organic amendments and was found to be highest for peat, followed by sewage sludge, sawdust and farmyard manure. The coefficient was lower for green manure than for average aboveground crop residues. In line with these results, amendment-C retention coefficients were lowest for green manure (1.8%) and straw (5.4%) and higher for slurry (12.5%) and fresh cattle manure (13.6%) in a Swiss long-term experiment (*Maltas et al. 2018*). Based on a meta-analysis, FYM-C retention was 12% for average study durations of 18 years (*Maillard & Angers 2014*). It is important to note that organic amendments other than fresh crop residues (e.g. manure or compost) have a higher stability because the more labile forms of carbon in organic material are broken down during storage and lost through respiration (as CO_2). Hence, these losses need to be factored in for a full CO_2 budget.

Compared with all forms of C inputs mentioned above, biochar is a very stable product. Only a few percent are bioavailable and, for very stable biochars produced at higher temperatures, the remaining 97% persist on centennial time scales based on results of a meta-analysis (*Wang et al. 2015*).

1.1.2.3. Factors that change organic carbon turnover (stabilization) and loss

The rate by which OM cycles through soil before it eventually leaves it as CO_2 and other products of metabolism is influenced by many factors, of which some (e.g. soil texture, soil mineralogy, site climate) cannot be changed directly or easily whereas others (e.g. soil water and oxygen content, soil pH, nutrient availability, substrate concentration, soil structure) are influenced by management. However, the resulting change in turnover rate can be inconsistent and therefore difficult to predict. For example, a higher soil pH tends to increase microbial activity and hence C turnover, but it also increases the amount of microbial (by-) products that are eventually stabilized in soil. Also, aggregation is improved, supporting protection from decomposition. Furthermore, pH increases are induced by liming, and the calcium ion is important for specific stabilization mechanisms between negatively charged clay minerals and OM with negative functionality. While these mechanisms tend to move the system towards higher C stocks, declining soil pH may induce the same but for different reasons. Acidic soils have low microbial activity, leading to the accumulation of poorly decomposed plant litter in and on the soil. Furthermore, organo-mineral interactions with specific, poorly crystalline iron and aluminium (hydr)oxides are particularly strong and provide an important stabilization mechanism in acidic forest soils.

The importance of stabilization processes for SOC storage is supported by distribution of SOC stocks in Swiss forest soils which are largely unaffected by management. In forests, SOC stocks are highest in regions with a relatively low forest productivity (Ticino, Jura) and hence low C inputs but with high contents of minerals that can stabilize SOC, such as calcium and clay in the Jura (*Rowley et al. 2018*) and iron and aluminium oxides in Ticino (*Eckmeier et al. 2010*).

Due to the stabilizing effect of SOC–clay binding, agricultural soils with high clay contents show higher SOC stocks (*Leifeld et al. 2005; Johannes et al., 2017; Sauzet et al., submitted*). Therefore, larger SOC stocks can be expected with clayey soils, as well as larger sequestration potentials (section 1.3.1).

A second example refers to nutrients. Building up and maintaining OM in biologically active mineral soils such as Ap-horizons requires nutrients, given the relatively narrow C:nutrient stoichiometry of SOM (*van Groenigen et al. 2017*). Hence, together with C organically bound nutrients (mostly nitrogen, phosphorus and sulphur) are co-sequestered. In 'nutrient-limited' ecosystems such as some wetlands, extensively managed mountain grasslands or some forests, adding nutrients may foster decomposition but may also increase the amount of nutrient-rich microbial necromass. More straightforward are factors that relate to the oxygen availability of soil. Many soils, particularly subsoils and organic soils, are poor in oxygen owing to high groundwater levels or stagnic water. Upon drainage, aeration is improved and the decomposition of SOM is accelerated leading to a decline in SOC stocks. This has been shown not only for organic soils (*Wüst-Galley et al. 2020a*) but for drained mineral soils as well (*Meersmans et al. 2011*). Another, partially manageable factor is the distribution of SOM in the soil profile. SOC turnover declines continuously with depth (*Shi et al. 2020*), suggesting that carbon stored in subsoil will reside for longer times. Any measure that changes the distribution of stocks and inputs towards the deeper soil has the potential to benefit from these longer turnover times, namely deeper incorporation of OM in the soil. For example, even moderate deepening of ploughing with more powerful machinery ('Krumenvertiefung') since the 1960s in Lower Saxony, Germany, has been considered an important factor for increasing C stocks (*Nieder & Richter 2000*). More extreme measures such as deep ploughing may increase C stocks even more as discussed in section 2.1.2.4 (e.g. *Alcántara et al. 2016, 2017; Schiedung et al. 2019*). Whether subsoil inputs via mechanical disturbance and establishment of deep-rooting species have similar effects is still unknown, because fresh inputs may trigger specific biological effects that are difficult to predict.

1.1.2.4. Environmental factors that control soil organic carbon stocks

Temperature exerts a strong influence on the SOC concentration. With cooler conditions, C turnover rates decrease, and therefore SOC concentrations strongly increase with altitude (*Leifeld et al. 2005*). Because soils at higher altitude tend to be shallower and stonier compared with soils in the lowland, SOC stocks do not differ as much with altitude (*Leifeld et al. 2005*). Precipitation has mainly indirect effects on SOC concentration, through NPP.

1.1.2.5. How will climate change impact SOC stocks?

Mean annual and maximal temperatures are continuously rising along with an increased severity and frequency of drought (*Brennpunkt Klima Schweiz 2016*). Changes in temperature and moisture have both direct and indirect effects on SOC stocks by affecting C outputs from the soil through SOC decomposition as well as C inputs from plants into the soil (*Prietzel et al. 2016*). Because SOC stocks are the net result of these fluxes, the overall effect of climatic changes on soil C storage is usually smaller than those on the individual fluxes. Several studies suggest that SOC stocks are decreasing in response to increasing temperatures (e.g. *Bellamy et al. 2005; Crowther et al. 2016*), but reliable predictions remain challenging due to effects on plants and thermal adaptions of microbial communities and their activity. In the longer term, the increased SOC processing at higher temperatures may deplete soils in labile C which in turn dampens SOC responses to climate warming (*Melillo et al. 2017*). Furthermore, interactions of temperature with other factors such as changing precipitation might occur (*Smith et al. 2008*).

The spatial pattern of SOC stocks for Swiss forests indicates decreasing SOC stocks associated with expected climatic changes. Along a natural climatic gradient with 4 °C warmer temperatures and 33% less precipitation, SOC stocks in Swiss forest soils decrease by 27.7 t C ha^{-1} (or 19% of their current stocks) (*Hagedorn et al. 2018*). Modelling SOC stocks in a changing climate suggests similar losses for forest soils (*Manusch et al. 2014*). However, the rate of these changes remains unknown. The conclusion that SOC stocks will decrease with ongoing climatic changes is supported by a repeated soil survey in the Bavarian Alps with similar climatic conditions as in Switzerland. There, *Prietzel et al. (2016)* observed SOC losses of 0.4 to 0.9 t C ha^{-1} since the 1980s, which was largely attributed to higher temperatures in the last decades, which likely stimulated SOC decomposition but did not enhance the growth of dominating spruce trees and hence C-inputs into soils. These SOC losses apparently conflict with unchanged SOC stocks in a repeated survey of 27 Swiss forest soils (conducted by the Swiss National Soil Monitoring Network; *Gubler et al. 2015*), which could partly be attributed to differences in site conditions because SOC losses in Bavaria have primarily occurred on shallow forest soils on dolomitic bedrock (though these soil types also exist in the Jura and Pre-Alps). A repeated spatially resolved soil inventory would provide reliable estimates but is so far lacking.

In the case of agricultural mineral soils, several studies have shown that soil management has had a more pronounced effect on the evolution of SOC stocks than increasing temperatures have had over the last decades (*Smith et al. 2007; Leifeld et al. 2009; Keel et al. 2019*). However, in some regions of Switzerland, precipitation has also changed. In most parts of the country, the average winter precipitation has increased and there are clear indications that the intensity and frequency of heavy precipitation events have increased. These can lead to enhanced erosion. Regional differences in climate change effects are expected (www.nccs.admin.ch). Summer months in the Jura will become drier in the future. This could have a negative impact on SOC stocks through reduced C inputs and faster SOC turnover. In the Central Plateau, longer growth periods are expected, which offer the prospect of higher agricultural yields. These can have a positive effect on SOC stocks. Similarly, productivity might increase in the foothills of the Alps thanks to longer summers. The region south of the Alps will strongly be affected by droughts. Although drought reduces plant productivity and thus C inputs to the soil, the direct effects on SOC stocks are not well documented.

In the case of organic soils, reductions in precipitation can reduce the water table, thereby drying out the peat and eventually accelerating C decomposition (*Fenner & Freeman 2011*), although the net response of peatland carbon storage to future climate conditions may also tend towards a larger sink (*Gallego-Sala et al. 2018*). Heavy rainfalls could lead to erosion that will especially affect already damaged peat soils. In general, partly degraded organic soils are probably more vulnerable to climate extremes than are undisturbed natural peatlands. Additional effects of climate change on peatlands are discussed in more detail in chapter 2.1.

1.1.2.6. Loss of SOC stocks through erosion

Erosion is the loss of the upper soil layer, including SOM contained in this layer. It is caused by wind or water surface runoff in hilly land and generally occurs on uncovered and degraded soils. It is a vicious circle, because the less SOC there is to protect the soil surface, the more soil and C will be lost through erosion. While erosion can lead to C emission into the atmosphere, C deposition can also be buried and sequestered (*Sanderman et al. 2017*). Erosion is therefore not necessarily a complete loss of C to the atmosphere because it is often only a transfer of C (partly also to rivers). However, on agricultural fields, erosion clearly leads to a loss of soil C (for arable fields in Switzerland: 0.75 t soil ha^{-1} yr^{-1}; *Prasuhn 2011*) and is an important soil degradation problem (*Lal. 2001*).

1.1.2.7. Brief summary of factors that control SOC storage

- Input materials (quality and quantity)
- Soil minerals
- Clay content
- pH of soil
- Soil temperature and moisture (climate change feedback effects)
- Land use

1.1.3. Soil types: What is the difference in organic carbon storage and turnover between mineral and organic soils?

Under frequent water logging, oxygen diffusion into soil is slow, leading to anoxic conditions in deeper layers, which in turn induces slow decomposition rates and (owing to different metabolic pathways and lower energy gains) a generally lower microbial activity. In such environments, partially transformed plant residues accumulate, forming peat which is produced either from sedges or other grasses and wood (fens) or primarily from sphagnum mosses (bogs). In bogs, the inherent recalcitrance of the sphagnum material and high soil acidity foster residue accumulation even further. Soils with peat layers of minimum thickness and OM content are classified as Histosols or organic soils. In these soils, organo-mineral interaction or aggregation plays little or no role for reducing decomposition, but the availability of electron acceptors is the major constraint. The resulting soils steadily accumulated OM over millennia with peat thicknesses of up to several meters (*Loisel et al. 2014*). Intact peatlands are therefore not in steady state with respect to C storage, but they continuously accumulate C (Figure 5) at average rates of 0.2 t C ha^{-1} yr^{-1} for northern peatlands (*Loisel et al. 2014*). Despite their low bulk density, C stocks of organic soils exceed those of mineral soils considerably, reaching amounts of about 1000–2000 t SOC ha^{-1} (*Leifeld & Menichetti 2018*), which is in stark contrast to mineral soils, where the estimated average for Swiss agricultural soils is 82 t per ha^{-1} for 0–100 cm *(Leifeld et al. 2005*). The dependency of C sequestration in organic soils on the single factor water makes them also very vulnerable to C loss. Upon drainage for agriculture, these soils tend to lose C (Figure 5) at rates of about 3–10 t C ha^{-1} yr^{-1} (*FOEN 2020*), i.e. 15–50 times faster than they accumulated it. The amount of C stored, the fact that accumulation occurs over very long time scales, and the high vulnerability of the C stock to disturbance are reasons to treat organic soils as an individual soil class not only in soil classifications but also for national greenhouse gas reporting and in this report. Although different measures to protect peat in managed organic soils from further decomposition are currently studied and sought for, raising the water table close to the surface is currently seen as the only reliable option for achieving this aim.

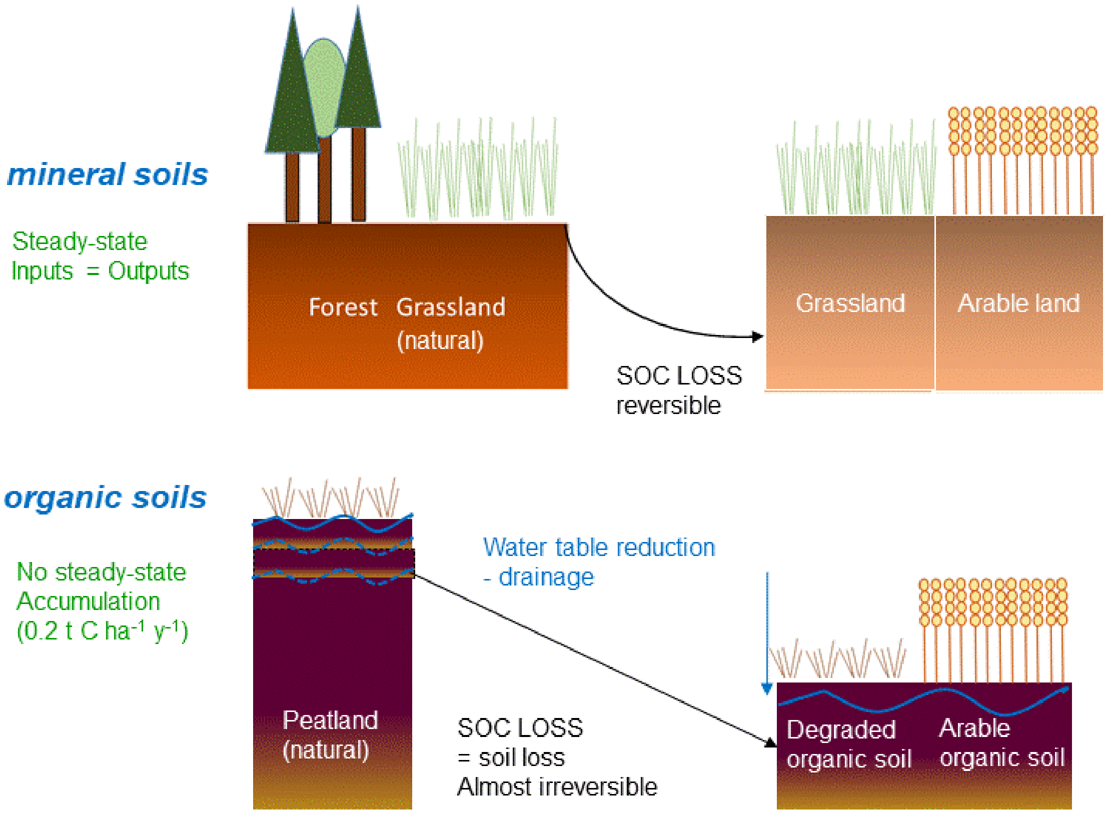

Figure 5: Main differences between land use conversion of mineral and organic soils. Undisturbed organic soils are characterized by large soil organic carbon (SOC) stocks and are sinks for carbon dioxide (net uptake), whereas minerals soils have an even carbon dioxide balance. Both soil types lose carbon (C) after conversion to arable land. Whereas SOC can partly be regained on mineral soil (but at a slower rate than it is lost), losses are mostly irreversible on organic soils.

1.2. Evaluating the knowledge about the current and historical state of SOC stocks

By Jens Leifeld, Pascal Boivin, Stéphane Burgos, Raphaël Charles, Frank Hagedorn, Alice Johannes, Sonja G. Keel, Beatrice Kulli, Andrea Saluz

Soil organic carbon (SOC) stocks differ largely between different land use types. They are generally highest in organic soils (natural peatlands) and decrease in the order presented in Figure 6. In this chapter, the SOC stocks of each land use type are discussed in more detail.

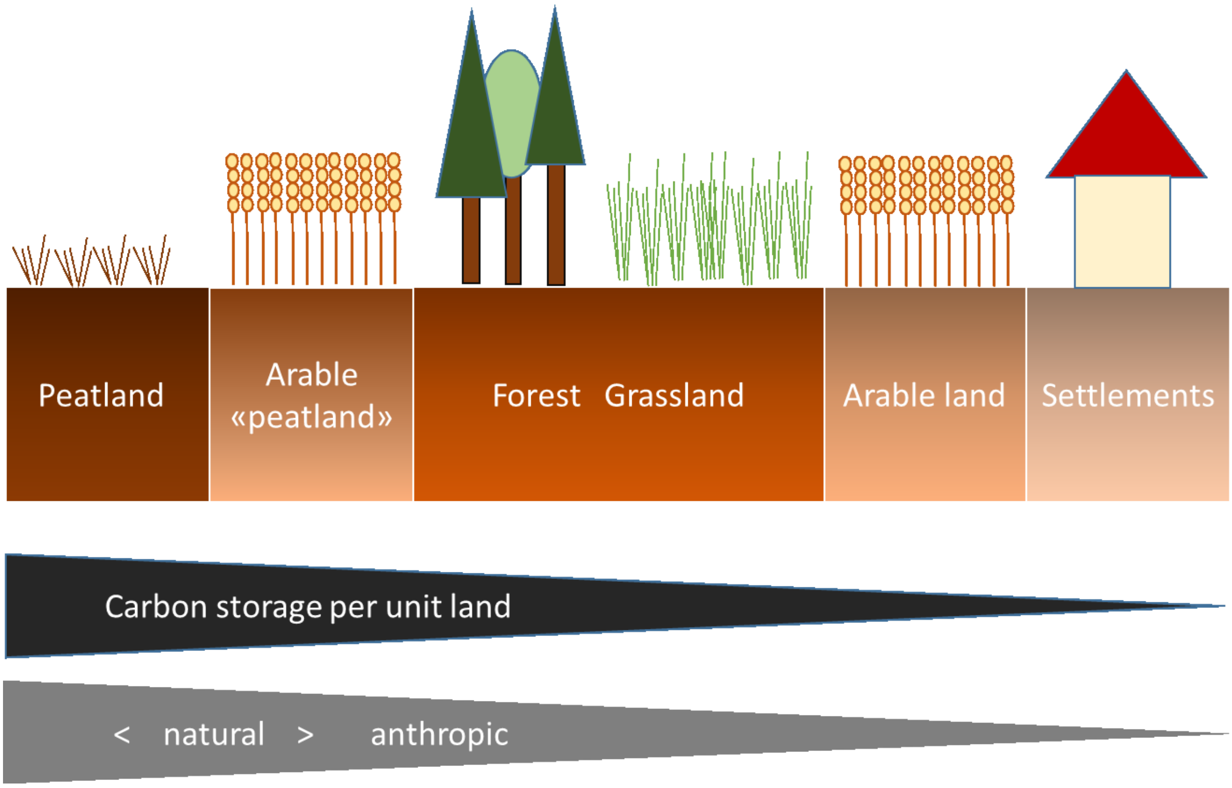

Figure 6: Peatlands under agricultural use start to degrade quickly and strictly speaking are no longer peatlands but rather degraded organic soils. Adapted from Amelung et al. (2020).

1.2.1. Soil carbon storage in organic soils

The amount of SOC stored in organic soils has neither been measured adequately, nor has the corresponding area been mapped to a significant extent. Another shortcoming of any estimate is the thickness of some peat deposits, which can be up to 20 m, for example in some areas of the Gürbetal and the Rheintal, but we are lacking sufficient information on the spatial extent of these deposits. The estimations below, as published by *Wüst-Galley et al. (2020a)* are based on a small and sketchy number of measurements of SOC stocks on site in different periods and an estimate on the corresponding area, both current and historical, on incomplete spatial data of varying quality, as well as on historical records and descriptions and toponyms. More reliable estimates require i) mapping of the remaining area by means of combined remote sensing and ground truthing, including assignment of soil types, and ii) mapping of peat thickness in the field.

The pre-industrial, i.e. undisturbed, peatland area of Switzerland in AD 1710 occupied an area of between 98,000 and 149,000 ha. Since then, the area of organic soils declined, owing to oxidative peat losses and, to a smaller extent, peat extraction, to a current area of about 30,000 ha. Between AD 1900 and 1950+, median peat thicknesses declined from 2 to 0.9 m. Between pre-industrial times (84–128 Mt C) and today (32 Mt C), about 52–96 Mt of SOC has been lost through drainage and peat extraction. Over the last 150 years, land use on the remaining organic soils changed towards a much higher share of cropland whereas the share of intact or only slightly disturbed peatlands accounts for only about 11%. Estimated carbon emissions from peat oxidation reach 139,000–187,000 t yr^{-1} (or 0.51–0.69 Mt CO_2-equivalents yr^{-1}). These emissions have increased since AD 1900 owing to the intensification of land use, despite an overall decline in the remaining area of organic soils, by a factor of 1.5–2.7.

1.2.2. Soil carbon storage in agricultural mineral soils

The area of mineral soils under agricultural use is 1,472,260 ha. This is the sum of the agricultural area estimated by the Federal Statistical Office for the year 2019 (1,043,730 ha; STAT-TAB, farm structure survey) and the summer pastures estimated by the land use statistics and the agricultural zones defined by the Federal Office for Agriculture (446,000 ha; *Chloé Wüst-Galley, personal communication*), minus the cropland and grassland areas on organic soils (17,470 ha; *Chloé Wüst-Galley, personal communication; Wüst-Galley et al. 2015; FOEN 2020*).

Arable land is decreasing in favour of settlement areas in lowland regions and due to forest expansion following the abandonment of alpine pastures. The loss was estimated to be about 1 m^2 s^{-1} by the Federal Statistical Office in the last years.

The SOC stock in mineral soil was not quantified *per se*. However, it can be estimated using different data sources, which agree quite well, despite their limitations.

Based on a method developed for national greenhouse gas reporting, *Wüst-Galley et al. (2020b)* estimated that the total SOC stock in mineral soils under agricultural use is about 77 Mt C in the upper 30 cm of the profile. Thereof, 20 Mt C are stored in cropland soils with the following distribution regarding three elevations zones: 14 Mt C (<601 m a.s.l.), 6.1 Mt C (601–1200 m a.s.l.) and 0.019 Mt C (>1200 m a.s.l.). In grassland soils, 57 Mt C are stored in total, with 9.4 Mt C, 22 Mt C, 26 Mt C at elevation zones <601 m a.s.l., 601–1200 m a.s.l. and >1200 m a.s.l., respectively. These numbers show that cropland SOC stocks are concentrated at low to medium elevations, whereas higher altitude areas contribute more to grassland SOC stocks. On a per area basis, SOC stocks are 50.6 t C ha^{-1} on cropland and 62.8 t C ha^{-1} on permanent grassland (0–30 cm, average 1990–2018; *FOEN 2020*).

The SOC concentration of arable soil in the cantons of Geneva, Vaud and Jura was estimated based on 40,000 topsoil (0–20 cm) samples gathered for analyses of the Swiss standard ‘proof of ecological performance’, in French called ‘prestations ecologiques requises (PER)' and in German 'Ökologischer Leistungsnachweis (ÖLN)'. In the canton of Geneva, the data has been stored in a geographic information system since 1993. A database of bulk density measurements (>130 values) allowed developing a pedotransfer function to estimate specific volumes based on soil analyses (particularly SOC concentration; information available in the corresponding cantons' climate plans reports [arable land sections] and in *Dupla et al. 2020*; Figure 7). Based on C concentrations and bulk densities, SOC stocks of minerals soils for the Léman Region (main crops only) are about 40 t ha^{-1} in the 0–20 cm layer. Corresponding main crop areas are 6800 ha (Geneva) and 55,000 ha (Vaud), thus leading to estimated stocks of 274,000 and 2,168,000 t of SOC for Geneva and Vaud cropland areas, respectively. Because these areas do not include special crops (orchards and vineyards), these 0–20 cm layer stocks are underestimations. Applying the average SOC stock of 40 t ha^{-1} to the Swiss cultivated mineral soil surface area leads to 21 Mt of SOC stored in the 0–20 cm cropland topsoil layer at the national scale (without taking into account pastures, orchards and vineyards) (Table 1).

Table 1: Estimated SOC stocks in mineral topsoils under agricultural use and in cropland soils according to the available sources.

Source	**Soil depth (cm)**	**SOC stock (Mt C)**
Agroscope[1] all soil	0–30	77
Agroscope[1] cropland	0–30	20
HEPIA[2] cropland	0–20	21
[1]: See *Wüst-Galley et al. (2020b)*		
[2]: Extrapolated from measurements in Vaud and Geneva.		

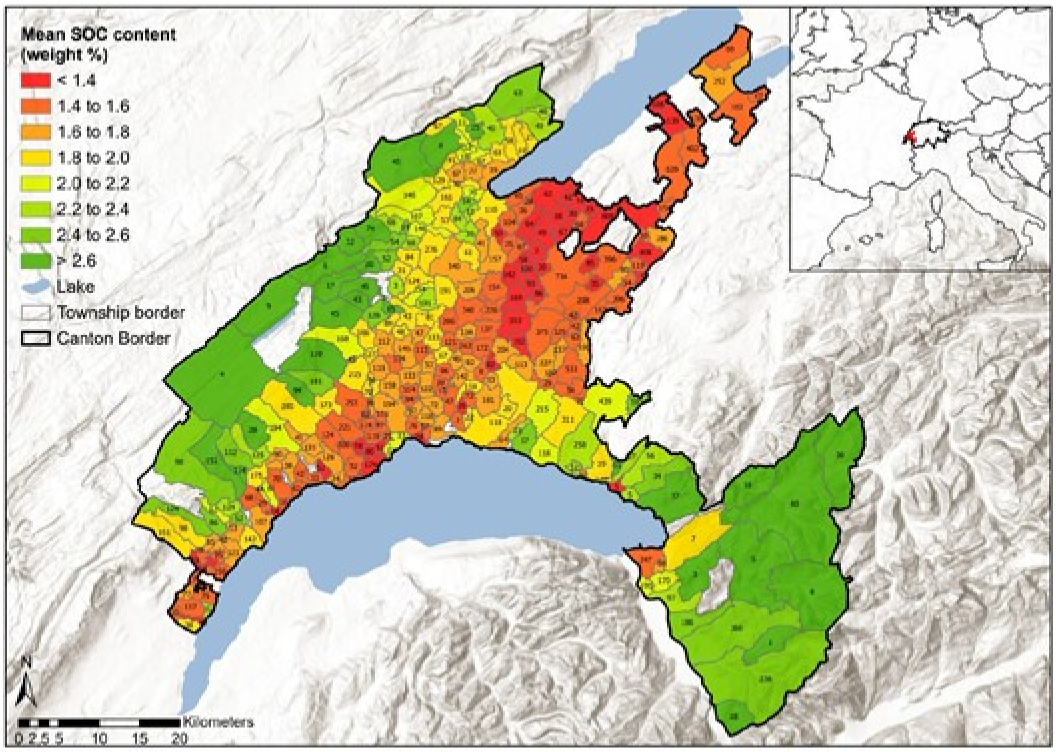

Figure 7: Example of a soil organic carbon (SOC) concentration map per township as obtained from 'proof of ecological performance' analyses for the years 2007 and later (from Dupla 2020). 30,041 soil analyses were included in the study. Figures indicate the number of analyses per township. Transparent zones refer to municipalities without relevant data (modified from swisstopo).

1.2.2.1. History of mineral cropland soils

Globally, mineral soils lost an estimated amount of 133 Pg SOC over the last millennia in the upper 2 m (*Sanderman et al. 2017*), together with an increasing share of land used for agriculture. This equals a loss of about 4.2% of the former stock. The loss rate increased since ca. AD 1800, peaked throughout the 19th century with annual loss rates of >0.3 Pg C and is nowadays at around 0.13 Pg C per year, equivalent to annually 0.004% of the existing stock.
There is no reliable data reaching so far into the past for Switzerland. Most soil monitoring started in the 1980s, and some long-term field experiments have data going back some decades, with the oldest field trial started in 1949 (Zurich Organic Fertilization Experiment). Yet, carbon loss due to conversion to agricultural land (mainly from forests) has happened before the 1950s, meaning that the available data do not show this evolution. However, there is some information on the evolution of SOC since 1900 provided by models. *Aguilera et al. (2018)* studied these aspects for Spain. Figure 8 shows how SOC stocks in cropland first increased between 1900 and 1933 as a result of increasing C inputs and an expansion of cropland (addition of new cropland with higher SOC stocks). After 1933, SOC stocks started to decline due to insufficient C inputs linked to intensified agriculture. The observations by *Aguilera et al. (2018)* depict a historical evolution in Spain that applies to other countries including Switzerland. In Switzerland after WWII, practices that have resulted in lower C- input include a switch from organic to mineral fertilization, intensive mechanization, and faster SOC turnover induced by drainage of mineral soils.

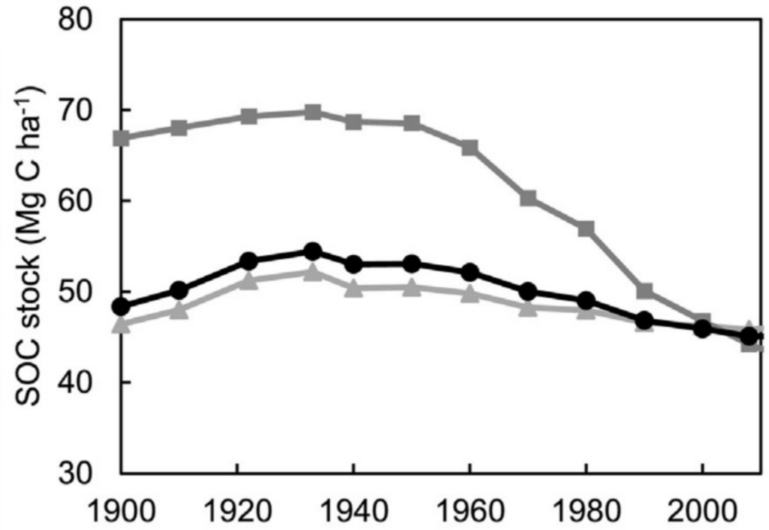

Figure 8: Simulated soil organic carbon (SOC) stocks (at 30 cm depth) per hectare for total cropland in Spain. Cropland in black with dots, herbaceous crops in grey with squares and woody crops in light grey with triangles (from Aguilera et al. 2018).

1.2.2.2. Soil monitoring data in cropland since the 1990s

As previously stated, data from Switzerland only describe the SOC evolution for recent decades – a time when most of the SOC losses from conversion to agriculture already had taken place. Most soil monitoring data started in the 1990s. On average, the different sources of information all show either no significant change in recent decades or losses in SOC stocks with large variation between sites.

- Measurements of changes in SOC stocks done by NABO for 29 cropland and 31 grassland sites indicate no significant overall changes from 1990 to 2014 (*Gubler et al. 2015; FOEN 2020*). However, there are sites with increasing SOC stocks and others that show SOC losses. Cropland sites with a low ratio of SOC:clay (<0.1) generally showed more positive trends than sites with higher ratios (*Gubler et al. 2019*).
- On the other hand, *Stumpf et al. (2018)* found a statistically significant decrease in SOC concentrations of 5.2 g kg^{-1} soil on croplands (without leys; 0–20 cm) between the 1995–1999 and 2011–2014 periods, based on a combination of three data sets from: NABO, the Swiss National Soil Information System (NABODAT) and the Swiss Biodiversity Monitoring System (BDM). For permanent grassland, the SOC decrease was not statistically significant (1.2 g kg^{-1}).
- Eleven Swiss long-term experiments on cropland and permanent grassland covering a large range of management practices showed that SOC stocks have been decreasing at an average rate of 0.29 t C ha^{-1} yr^{-1} since their beginning (mostly in the 1970s and 1980s; *Keel et al. 2019*).
- At the national scale, modelled SOC stocks of agricultural mineral topsoils (0–30 cm) that were not affected by land use change did not significantly change since 1990 (*FOEN 2020*). For cropland, the annual average SOC stock increased slightly by 0.037 ± 0.344 t SOC with high interannual variability mainly associated with climatic conditions. On permanent grasslands, the average change rate was −0.044 ± 0.249 t SOC ha^{-1} yr^{-1} (i.e. net loss of carbon).

- In the Léman Region, a large number of on-farm data were gathered (from 2300 fields) for the years 1995 to present (*Dupla et al. 2020b*). For main crops, the annual rate of change (ARC) of SOC concentrations ranged from −30‰ to +30‰, with a median at zero for the whole 1995–2020 period (Figure 9). On average, change rates of SOC stocks were similar to change rates of SOC concentrations (*Dupla et al. 2020b*). Assuming a SOC stock of 40 t C ha^{-1} for Swiss cropland in the first 20 cm, the mean ARCs of 0.458‰ (slope in Figure 10) translate to an increase of 0.02 t SOC ha^{-1} yr^{-1}. However, in recent years, the ARC distribution shifted to higher values than presented in Figure 9, with a median value significantly larger than zero for the last decade (Figure 11). It is important to note that the values still cover a large range and nearly 50% of the fields still lose carbon.

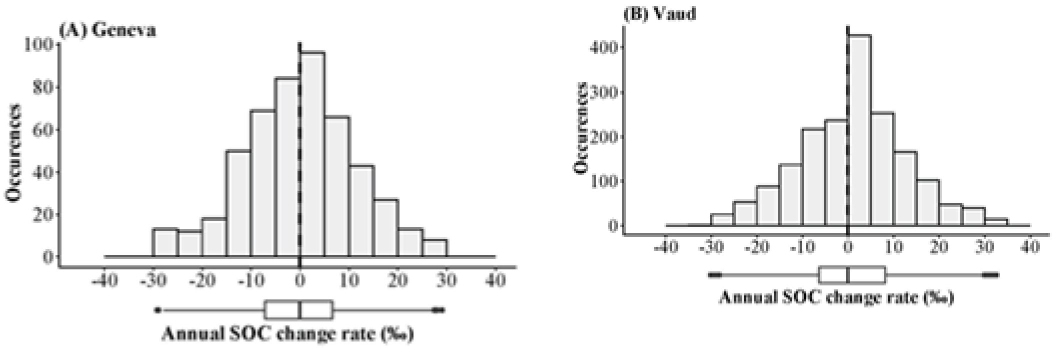

Figure 9: Annual soil organic carbon (SOC) change rates in the 0–20 cm topsoil of cropland fields from the cantons of (A) Geneva (496 fields) and (B) Vaud (1793 fields) over the 1993–2020 period. Dashed vertical line: median value (from Dupla et al. 2020b).

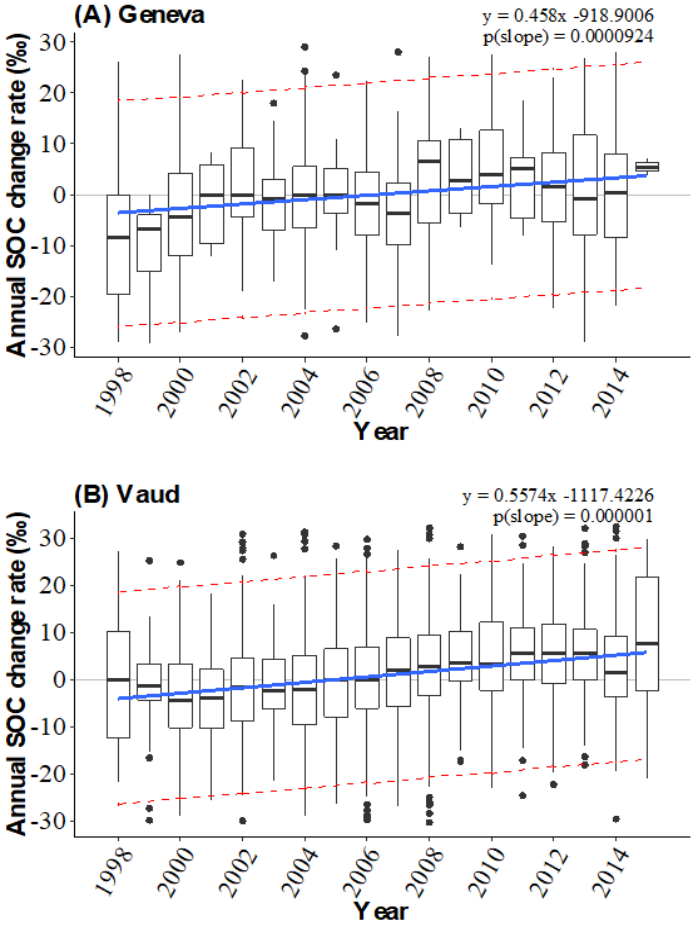

Figure 10: Annual soil organic carbon (SOC) change rates (0–20 cm) of cropland fields over the 1993–2020 period for the cantons of (A) Geneva (496 fields) and (B) Vaud (1793 fields) as a function of the average year between two analyses. Solid line: linear regression. Dashed line: 95% local regression prediction interval (from Dupla et al. 2021).

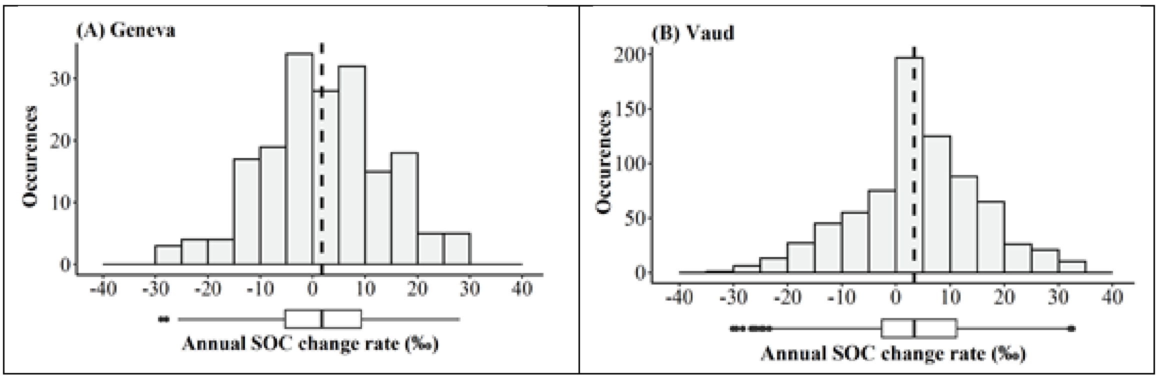

Figure 11: Annual soil organic carbon (SOC) change rates in the topsoil (0–20 cm) of cropland fields from the cantons of (A) Geneva (496 fields) and (B) Vaud (1793 fields) for more recent years than in Figure 9 (2007–2020). Dashed vertical line: median value (from Dupla et al. 2021).

1.2.3. Soil carbon storage in forest soils

Forests cover 1,317,000 ha, an area which corresponds to about one third of the Swiss territory (*Brändli et al. 2020*). The forested area has increased by 22% throughout the last century (*Ginzler et al. 2011*) particularly in the Alps and in southern Switzerland as a result of land abandonment. In the last decade, the forest cover has increased by approx. 2.4%, which is slightly less than in the two decades before (*Brändli et al. 2020*). In Switzerland, forests have primarily been managed as plenter forests with single-tree harvest and promotion of adapted tree species, leading to a near-natural structure of forests (*Angst 2012*). Due to high costs of wood harvesting, the management intensity has even declined during the last decades (*Brändli 2010*). Consequently, Swiss forests are characterized by a high mean tree age (more than a fourth of Swiss accessible forest area is older than 120 years) and by the greatest stand biomass in Europe (*Liski et al. 2002*).

SOC stocks. Swiss forest soils down to the bedrock store on average 143 t C ha^{-1} out of which 17 t C ha^{-1} are in the forest floor (*Nussbaum et al. 2014; Hagedorn et al. 2018*). These SOC stocks are approximately 20% higher than in the living biomass in forests, and they are the highest ones of all European countries (*Rogiers et al. 2015*). German forest soils store 117 t C ha^{-1} (*Grüneberg et al. 2014*). The total SOC stock in Swiss forest soils amounts to roughly 188 Mt C.

Among Swiss biogeographic regions, the southern Swiss Alps have the highest SOC stocks per hectare and the Swiss Plateau has the smallest ones (Figure 12; *Nussbaum et al. 2014*). This pattern is opposed to tree biomass and forest productivity, which have the highest values in the Swiss Plateau (*Brändli et al. 2020*). Disentangling controlling factors of SOC stocks shows that forest biomass and litter inputs are not significantly related to SOC stocks, very likely because C inputs are balanced by losses through SOC mineralization (*Gosheva 2017*). Soil physicochemical properties (texture, pH and mineralogy) exert a dominant influence, explaining about 20% of the variance of 1000 soil profiles across Swiss forests (*Gosheva 2017*). This effect can be explained by their impact on SOC stabilization. In comparison, climate is less important but also exerts a significant influence on SOC stocks. For the forest floor, mean annual temperature seems important (8% of variance), but not mean annual precipitation. For the mineral soil, the opposite pattern is found, with mean annual precipitation being more important than mean annual temperature. Forest type affects C stocks in the forest floor with more SOC under coniferous than under broadleaf trees (38 vs. 10 t C ha^{-1}). The interplay of factors is extremely difficult to implement in soil C models (*Sulman et al. 2018*) such as Yasso07 which are used to quantify the sink strength in forest soils (*FOEN 2020*). The Yasso07 model is based on litter and climate data from Swiss forests and has not implemented stabilization processes due to a limited quantitative process understanding. Modelled SOC stocks differ strongly from measured ones in the Jura with high contents of carbonates and in Ticino with high contents of aluminium and iron oxides that bind SOC (*Gosheva 2017*). In southern Switzerland, high contents of pyrogenic C (*Eckmeier et al. 2010*) also contribute to underestimations of SOC stocks in the model.

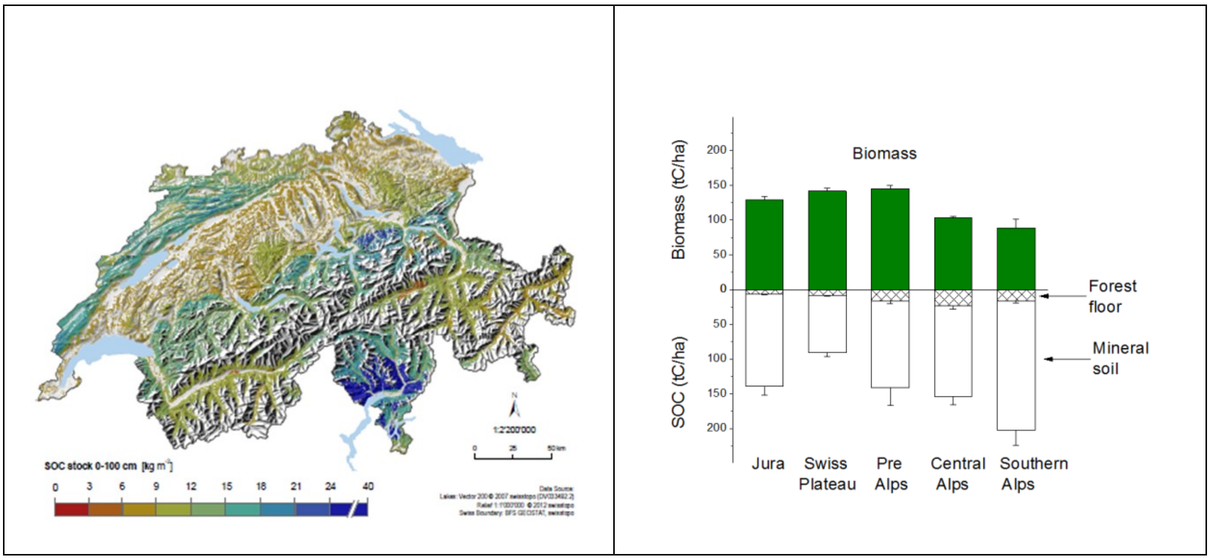

Figure 12 : Soil organic carbon (SOC) stocks in Swiss forest soils estimated by robust kriging show highest SOC stocks in southern Switzerland (left, from Nussbaum et al. 2014). Carbon (C) stocks in forest soils (organic layer or forest floor and mineral soil) exceed C stocks in biomass of the five biogeographical regions in Switzerland (right). Across these regions, there is a negative relation between biomass and SOC stocks.

SOC stock changes. In contrast to SOC stocks, SOC stock changes in Swiss forest soils remain highly uncertain because a repeated inventory of whole soil profiles is so far lacking. The repeated analyses of topsoils (0–20 cm) by NABO at 29 forest sites indicate negligible SOC stock changes in the last decades in these soils. In agreement, modelling of SOC dynamics using litter inputs derived from national forest inventory and climate data suggests minor SOC stock changes of +0.001 t C ha^{-1} yr^{-1} (or 1.3 kt C yr^{-1} in Swiss forest soils; *FOEN 2020*). The forest inventory in Germany based on 1800 soil profiles that have repeatedly been measured in 1990 and 2007 reported SOC stocks to increase by 0.41 t C ha^{-1} yr^{-1} (*Grüneberg et al. 2014*). This increase occurred in the mineral soil, whereas SOC stocks in the forest floor showed a slight decrease. The discrepancy between presumably negligible changes in Swiss forests and increasing stocks in German forests could be explained by the higher forest age in Switzerland than in Germany. SOC stocks (in particular those in the labile pool such as the forest floor) are assumed to increase with forest development but also to decrease rapidly following harvest (*Jandl et al. 2007*).

1.2.4. Soil carbon storage in settlement soils

Currently, there are no soil maps for the Swiss settlement area. Measured values for individual sites, which originate from research projects or the cantonal or national soil monitoring, are only available for isolated cases. Our estimates of the C concentration are therefore additionally based on literature data and on data from agricultural and forest areas, provided that these have a similar soil structure to the settlement areas under consideration.

However, the estimates are subject to considerable uncertainty. One reason is the allocation of the land cover to the categories by the area statistics. It is, for example, difficult to distinguish between lawn and grass–herb vegetation when assigning categories for land cover based on aerial photographs. In the settlement area, it is assumed that grass on open surfaces around buildings represents mostly artificial areas covered by lawn. However, in the vicinity of roads and railroad lines, grassy areas are usually classified as grass and herb vegetation.

Furthermore, the assignment to a category does not necessarily say anything about the structure and the depth of the soil. Especially in older gardens around one- or two-family houses, there are also deep, natural soils, whereas in areas with blocks of flats, shallow soils, for example on top of underground garages, are much more likely.

Pouyat et al. (2006) found that the types of land use and land cover discussed in the literature were often not consistent across cities or countries due to availability of data. They noticed big differences between soils of the same coverage. In their study, they found the highest density of SOC of all settlement soils in residential lawns (measured in Baltimore, MD, USA). The authors concluded that this is most likely a result of lawn management, which typically includes supplements of water and nutrients to maximize grass productivity. *Peach et al. (2019)* analysed yard soils mostly covered by lawn in New England (USA). Their estimate for SOC in the upper 60 cm of the soil was 90 t C ha^{-1}, which was less than they estimated for field or forest soils outside urban areas.

Table 2 shows the values chosen based on the study by *Pouyat et al. (2006)* and their assessments of SOC densities in settlement soils of different land cover. Some adaptations have been made to their data: the maximum value for SOC of tree and shrub areas was reduced to 100 t C ha^{-1} to fit the lower values for forest soils on the Swiss Plateau in the section on forest soils in this chapter (section 1.2.3). Because falling leaves from trees are likely to be taken away under trees in urban areas, we consider it as unlikely that areas with trees in settlements have higher SOC concentrations than forest soils on the Swiss Plateau. SOC stocks in the range of 70–100 t C ha^{-1} have been found by *Bae & Ryu (2015)* in urban soils covered with different forest types in the Seoul Forest Park in South Korea. Their estimation is based on measurements in the upper 1 m of the soil profile.

The values for lawns in the surroundings of buildings bigger than one- and two-family homes have been reduced to 66% of the original value in order to take into account that these soils are likely to be shallow and consist only of topsoil. The lawns in other categories, such as parks or cemeteries, including surroundings of agricultural buildings, have been chosen based on data from NABO.

Table 2: Minimum and maximum values of soil organic carbon (SOC) stocks over the whole soil profile for different types of land cover on settlement soils.

Land cover	min. SOC stock [t ha^{-1}]	max. SOC stock [t ha^{-1}]
Tree and shrub areas	38	100*
Trees on artificial areas	38	71
Grass and herb vegetation	34	83
Gardens with border and patch structures on artificial areas	24	60
Mix of small structures on artificial areas	34	83
Lawn in the surroundings of one- and two-family homes	34	144
Lawns in the surroundings of other buildings**	22	92
Lawns in other categories***	72	87
Bare land	33	33

data generally taken from *Pouyat et al. (2006)*, except for:
* reduced to the value for forest soils on the Swiss Plateau
** reduced to 66% of the values indicated by the study, in order to take into account that these soils are likely to be shallow and consist only of topsoil
*** increased to values measured by NABO (*personal communication*)

Calculating the stocks of settlement soils based on these assumptions and the data of the areal statistics results in a mean value of 7.08 Mt C (min. assumptions: 3.89 Mt C; max. assumptions: 10.26 Mt C). Soils with trees or shrubs contribute about one fourth to the total carbon stored in settlement soils (Figure 13), whereas surfaces covered with lawns or grass and herb vegetation make up more than half of it. Compared with SOC stocks in agricultural or forest soils, the total amounts stored in settlement soils are small. However, they may be of growing interest because settlements are still expanding.

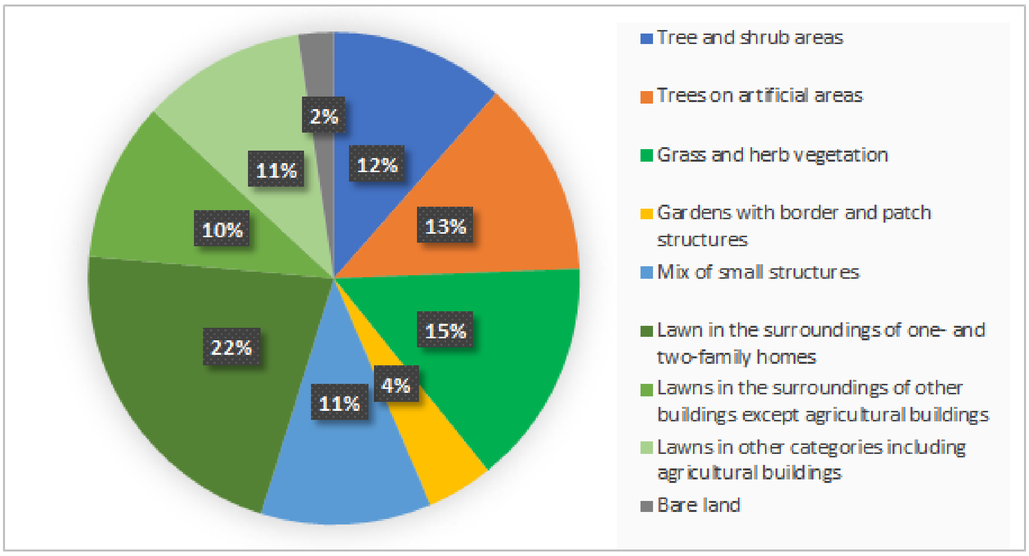

Figure 13: Estimated contribution of the different types of open surfaces to the total soil organic carbon (SOC) stock of the Swiss settlement soils.

In the context of growing settlements, the question of the effect of sealing is also an important topic. We assume that the C concentration beneath streets and buildings is in the range of bare soil. For construction work, usually the topsoil is removed before sealing and replaced with gravel and concrete. Ideally, the topsoil is used somewhere else for restoring a topsoil or improving the quality of agricultural land. Estimating the effect of sealing, we consider the removed topsoil as a removal from the SOC stock of the settlements. The calculation of SOC below sealed surfaces and buildings might be in the range of 3 Mt C for the Swiss settlement areas. The soil loses its C sequestration capacity through progressive sealing (*Romzaykina et al. 2020*).

1.2.5. Soil carbon storage in unmanaged mineral soils

The area of mostly unmanaged mineral soils amounts to 213,430 ha in Switzerland. This is the sum of the two categories of the areal statistics ‘unproductive grassland/shrub vegetation’ and ‘wetlands’ minus the area on organic soils (*Chloé Wüst-Galley, personal communication*). This category includes unproductive grassland/shrub vegetation (transition areas between summer pastures and vegetation-free areas), shrub vegetation, wetlands, riparian vegetation, avalanche barriers and alpine sport infrastructures.

Generally, little information exists regarding this category. The NABO includes a single site on an alpine meadow in the Swiss National Park (2400 m a.s.l.), which has been unmanaged since 1914 (Site No. 75, Zernez). Before 1914, it was used as a pasture, most likely at low intensity. In 2017, the topsoil SOC stock (0–20 cm) was 63 t C ha^{-1}. Since 1988, C concentrations have increased slightly but with a high uncertainty (*Andreas Gubler, personal communication*).

In a study by *Zollinger et al. (2013)*, two sites with alpine tundra (at about 2600 m a.s.l., some of them on permafrost) had SOC stocks ranging between 100 and 150 t ha^{-1} down to the C-horizon or rock surface. For topsoil, C concentrations ranged between 17 and 54 g kg^{-1} at permafrost sites and between 47 and 110 g kg^{-1} at non-permafrost sites. For one non-permafrost site (Bever), C concentrations were significantly higher than at permafrost sites. For five unmanaged sites along an elevation gradient at Furka between 2285 and 2653 m a.s.l., i.e. at or above the treeline, *Budge et al. (2011)* reported SOC stocks of between 55 and 102 t C ha^{-1} (0–30 cm).

As part of the Biodiversity–Soil Monitoring project (BDM-NABO), soil samples of all land use types are collected and SOC stocks will be calculated (*Reto Meuli, personal communication*). Most likely, data will become available in the year 2021.

1.3. Sequestration potentials, conclusions and recommendations

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Estimating realistic soil carbon sequestration potentials is challenging, and different approaches exist. In the following sections, we first discuss estimates based on the concept of saturation, which gives a theoretical potential based on soil texture. This approach assumes for example that biomass inputs are unlimited, and it does not provide information regarding the rate of soil organic carbon (SOC) increase. In section I.3.3, we present an approach based on a more quantitative assessment.

1.3.1. Using the SOC:clay ratio as a potential for mineral soils to store carbon

How to estimate the final capacity of soils to store SOC cannot explicitly be found, mainly because a 'maximum' value is not possible to state (there is always the possibility to store more C) and current stocks are likely limited by inputs rather than storage potential, particularly in subsoils. Yet, there is the possibility to estimate a potential for mineral soils to store C, using the concept of complexation between clay minerals and organic matter. This translates into the SOC:clay ratio, for which guide values linked to soil quality properties are available and presented below. The concept based on the SOC:clay ratio is regarded as an operational tool rather than a scientifically sound concept for C sequestration potentials. This calculation of a SOC storage potential is rather based on a 'minimum' requirement for soil quality, not on a 'maximum' capacity. Attempts are underway to address these topics operationally in order to make them workable for agricultural policies or management recommendations, and these are outlined in chapter 2.2.

For mineral soils, differing methods are proposed in the literature, in particular based on the C saturation concept or on databases allowing the defining of maximum values (*Merante et al. 2017; Chen et al. 2019*). The C saturation concept has first been proposed by *Six et al.*

(2002) and has further been developed, for example by *Dexter et al. (2008)* with special emphasis on physical soil properties. *Dexter et al. (2008)* estimated the SOC complexation potential of the soil to 0.1 × clay content, thus hypothesizing an upper limit for C storage. This concept of clay-saturation was reassessed by *Johannes et al. (2017)* based on field sampling of topsoils on the Swiss Plateau (Figure 14). They showed that the same 0.10 SOC:clay ratio was the limit between good and poor soil structure. This ratio can be used as soil structure vulnerability threshold. They also showed that a soil with a 0.13 SOC:clay ratio had very good dispositions for a good soil structure quality. However, a 0.08 SOC:clay ratio corresponded to extremely vulnerable structure, thus on average leading to degraded structure on cropland. These thresholds were later confirmed for UK soils (*Prout et al. 2020*), thus indicating that their scope of application is probably very large in terms of soil types. It must be underlined that the corresponding SOC:clay ratio values were actually observed on-farm, thus providing a direct potential of application and communication with farmers. The SOC:clay ratio values of 0.10 or even 0.13 do not represent a 'final' capacity of soils to store C but are rather an observation of an achievable SOC value directly linked to soil quality properties. This approach with a conservative value of 0.10 (if 0.13 is used as a goal, the calculation of the potential would be even higher) is used in the next section and applied on a large territory in western Switzerland. *Merante et al. (2017)* used a similar approach (with n: the amount of clay that can be complexed by C, i.e. they displayed a clay:SOC ratio, which is the inverse of the SOC:clay ratio) do draw a European map (Figure 15).

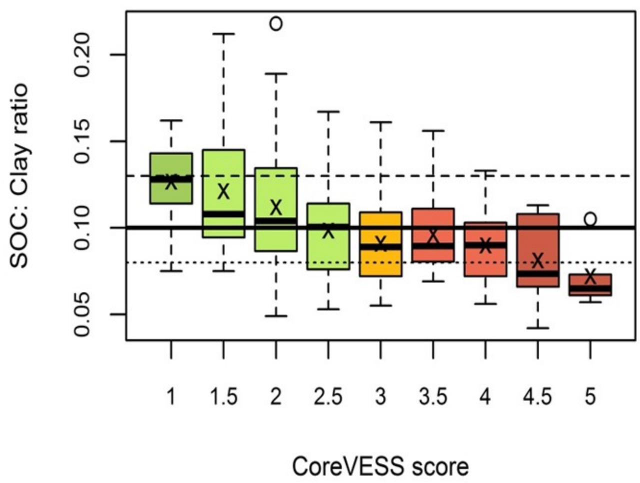

Figure 14: Boxplots showing the ratio of soil organic carbon (SOC) to clay for different CoreVESS scores (visual evaluation of soil structure quality on soil cores). Boxplots show mean values (cross), median values (solid horizontal line), 50th percentile values (box outline), minimum and maximum values (whiskers) and outliers (open circles). The dotted line indicates a SOC:clay ratio of 0.08, the full line a SOC:clay ratio of 0.10, and the dashed line a SOC:clay ratio of 0.13 (from Johannes et al. 2017).

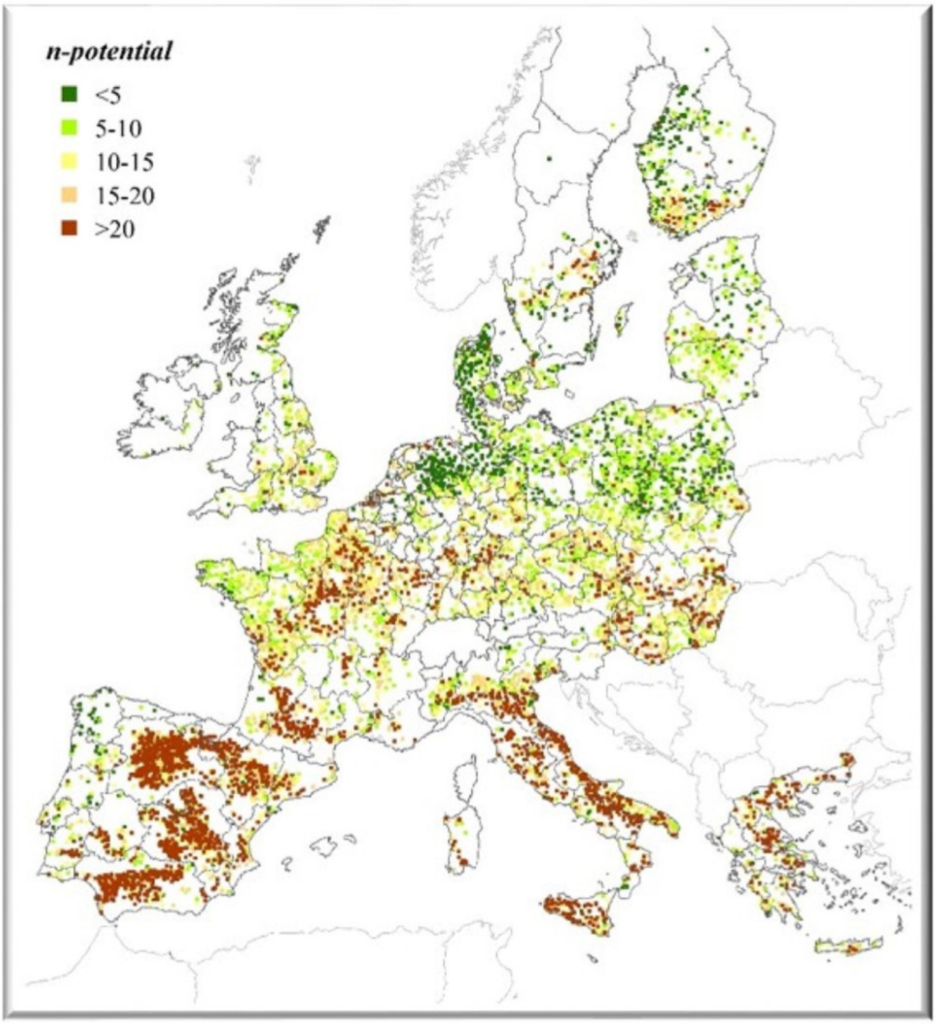

Figure 15: The indicator n-potential in soils (clay:SOC ratio, which is the inverse of the SOC:clay ratio) of the European regions (from Merante et al. 2017).

1.3.2. Evaluation of soil carbon storage potential for the cantons of Geneva, Vaud and Jura

These findings led us to develop an original approach to estimate the deficit in SOC stocks based on SOC:clay ratio threshold values. In this concept, the additional SOC storage potential is the amount of SOC necessary to increase the observed SOC:clay ratio to a value of 0.10. The maximum amount of additional SOC storage would be represented by the same calculation, but applying the 0.13 SOC:clay ratio target.

The SOC:clay ratio of 0.10 was applied to the cantons of Vaud and Geneva to estimate the amount of SOC that could additionally be stored in cropland soils. It is an underestimation because i) it is limited to 0–20 cm depth and ii) many fields show increasing topsoil SOC concentration though presenting a SOC:clay ratio larger than 0.10 (*Dupla et al. 2020*). It is also a realistic target because in terms of soil physical quality, the 0.10 SOC:clay ratio must be reached for sustainable soil management. The difference between the average SOC:clay content and 0.10 at regional scale can therefore be considered as the minimum amount of additional SOC storage for the corresponding area.

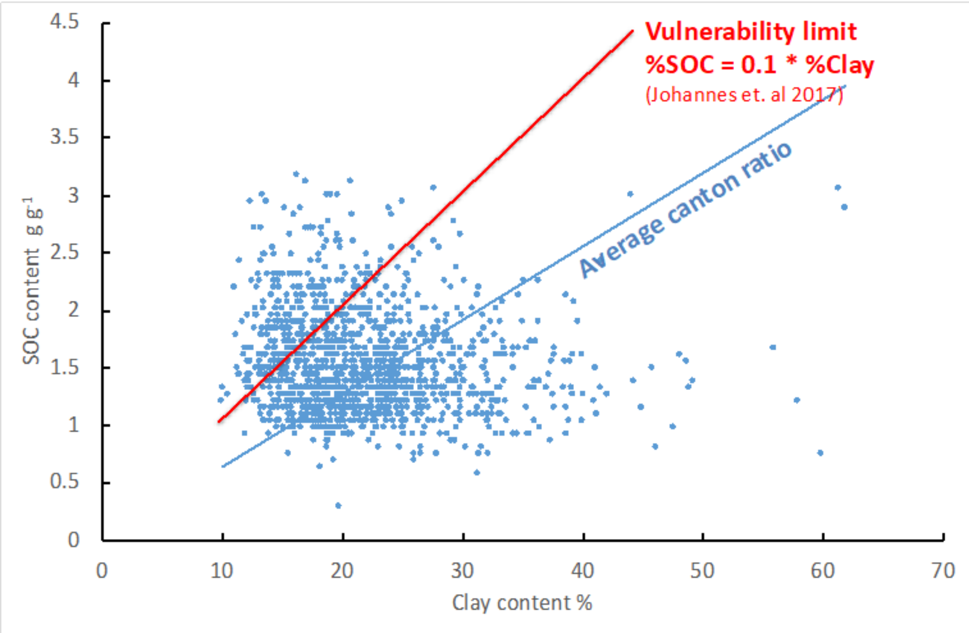

Figure 16: Distribution of the soild organic carbon (SOC) and clay contents for 2200 fields of arable land, canton of Vaud, with average canton SOC:clay ratio and minimum SOC:clay target ratio (from Boivin et al. 2020).

Based on this method, the deficit in SOC stocks in the 0–20 cm topsoil layer of arable land was estimated for the Jura, Vaud and Geneva cantons. On average, a SOC stock increase of 40%, 20% and 70%, respectively, relative to the current stock would be required for these cantons to reach a minimum soil quality (Figure 16 [Vaud], Figure 17 [Geneva], Figure 18 [Jura]). This corresponds for instance to a minimum of 700,000 t CO_2-equivalents (~190,000 t C) for Geneva and 2,000,000 t CO_2-equivalents (~550,000 t C) for Vaud (*Boivin et al. 2020; Dupla et al. 2020*). In contrast, 30 years with an annual increase of 4‰ would only result in a 13% increase in the total SOC stocks, which is small compared with soil quality requirements. How fast SOC stocks equivalent to minimum soil quality can be reached is thus determined by the rate of SOC increase and could take several decades or even a century. The challenge resides in the successful application of measures to increase SOC, because they are plentiful, site and soil dependent, and therefore need specific adaptations to each case.

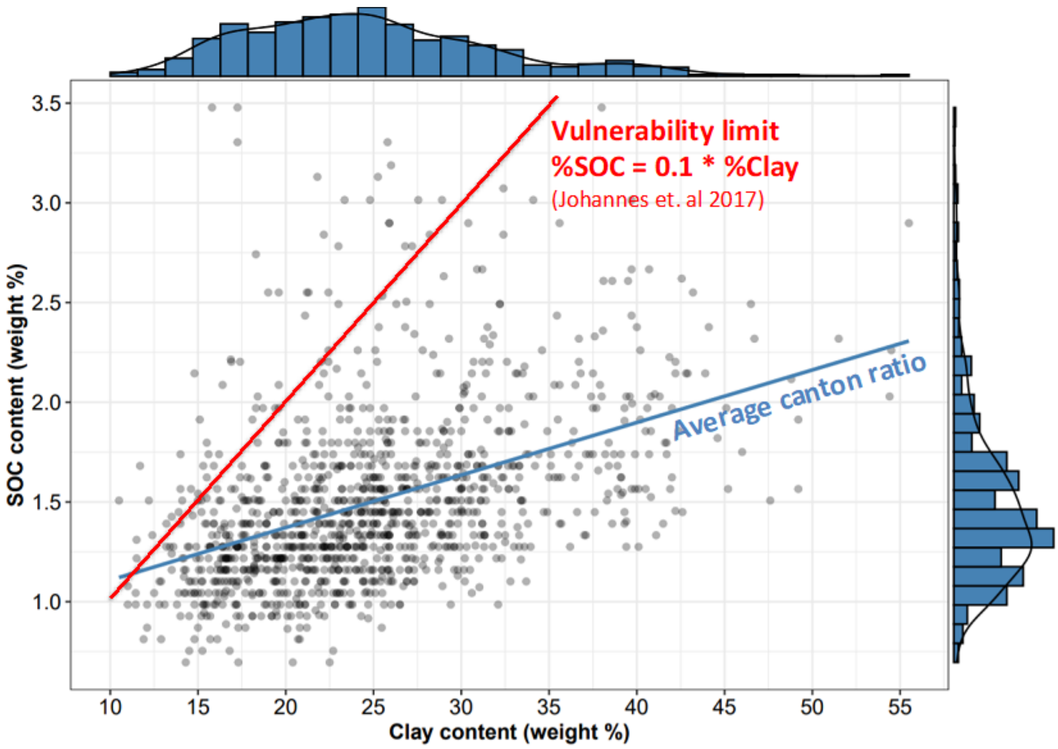

Figure 17: Distribution of the soil organic carbon (SOC) and clay contents for 2700 fields of arable land, canton of Geneva, with average canton SOC:clay ratio and minimum SOC:clay target ratio (from Boivin et al. 2020).

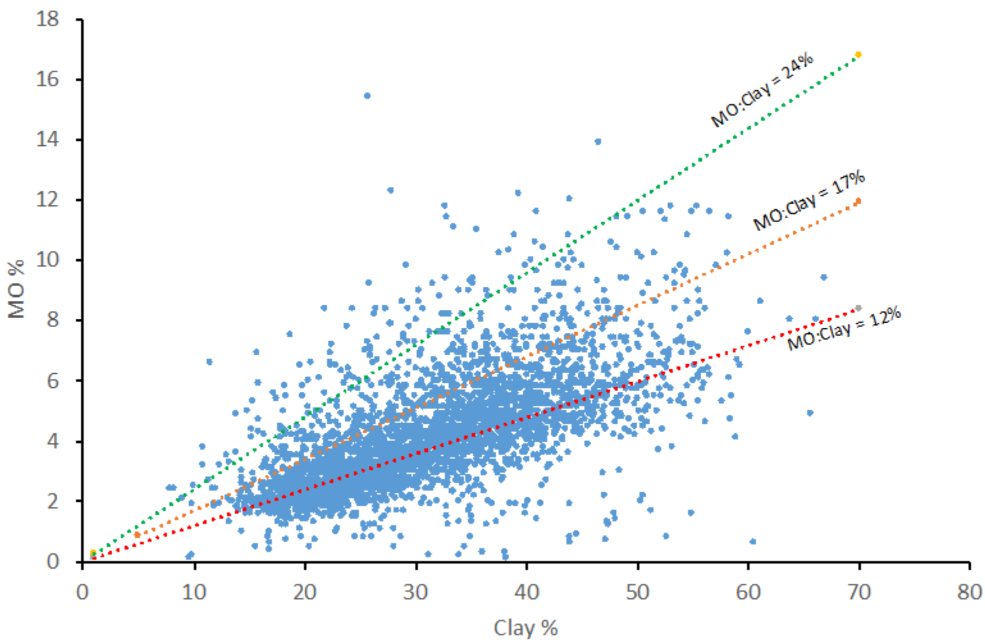

Figure 18: Distribution of the soil organic matter content (denoted MO) and clay content for more than 8000 fields of arable land, canton of Jura, with MO:clay target values corresponding to SOC:clay ratios of 0.13 (green), 0.10 (orange) and 0.08 (red). Canton median value of MO:clay: 0.12, which corresponds to a SOC:clay ratio of 0.07.

It is important to add that the expected state of degradation might be different for other regions of the country due to current and past land use. This means that also the potential for additional storage might differ. Furthermore, carbon storage potentials will be affected by local management (including livestock numbers) and climatic conditions.

1.3.3. Estimating SOC sequestration potentials based on SOC stocks and available measures: a case study from Bavaria

For Bavaria, a region with similar pedoclimatic conditions to Switzerland, *Wiesmeier et al. (2020)* estimated the amount of SOC that could be sequestered based on specific SOC change rates for different measures derived from the literature. They assessed C sequestration potentials for cover crops, improved crop rotations or cropland-to-grassland conversions. Their approach was very precise because they used a high-resolution SOC map as a baseline (i.e. initial SOC stocks) and assessed in a spatially explicit way which measures are already used where and how large the remaining potential is, based on available measures and associated SOC change rates. On average, about 1‰ of the current SOC stock or 0.3–0.4 Mt C could potentially be sequestered per year for several decades (*Wiesmeier et al. 2020*). This sequestration rate could compensate only 1.5% of Bavaria's emissions per year and is much smaller than the rate from a previous study for the same region (*Wiesmeier et al. 2014*). Their previous study assessed the potential saturation on clay and silt minerals based on the concept by *Hassink (1997)*, an approach similar to the one described above (SOC:clay ratio; section 1.3.1). For arable soils, a carbon sequestration potential of 50% relative to current stocks for cropland soils was found, similar to the situation described for the canton of Jura (section 1.3.2). Expressed in CO_2-equivalents, a potential uptake of 395 Mt CO_2-equivalents (107 Mt C) or four times the total emissions of Bavaria would be possible. These two studies demonstrate a huge discrepancy between a sequestration potential based on a soil carbon saturation concept and an upscaling of measured SOC change rates (a total of 400% vs. an annual amount of 1.5% of Bavaria's total emissions). An important difference is that approaches based on soil C saturation concepts describe theoretical limits of storage, i.e. how much additional C a soil can store based on its clay (and in this study also silt) content. This concept describes neither how long it will take until this state is reached nor whether sufficient C inputs are available to reach it. The second approach describes feasible soil carbon sequestration potentials that can be reached by implementing certain measures with known SOC change rates per year, and by accounting for the current management (i.e. if cover crops are already used). However, some important measures were not considered in the later study (*Wiesmeier et al. 2020*), i.e. higher crop residue retention, improved grassland or organic soil management or the addition of biochar. Therefore, the annual sequestration potential for Bavaria is likely to be somewhat higher than 1.5% of total emissions. For Switzerland, numbers in a similar range could be expected.

1.3.4. Soil/land-use types associated with high de- or increases in SOC stocks and soil/land-use types identified as most vulnerable regarding future SOC losses

Overall, there is no doubt that current SOC losses are highest on organic soils. On average, drained organic soils under agricultural use lose 9.52 t C ha^{-1} yr^{-1} or 34.9 t CO_2 per ha^{-1} yr^{-1} (*FOEN 2020*). Based on a review, emissions range from 13 to 29 t CO_2 per ha^{-1} yr^{-1} (Table 4 in chapter 2.1 on organic soils).

On agricultural mineral soils, significant SOC losses have been observed in response to:

- Conversion of permanent grassland to cropland (*Hermle et al. 2008; Oberholzer et al. 2014*).
- Absence of organic amendments (harvest residues, organic manure; *Maltas et al. 2018*)

The state of soil/land-use types for the whole country can briefly be summarized as shown in Table 3 and Figure 19.

Table 3: Summary of current dynamics and sequestration potentials of different soil/land-use types

Soil/land-use type	**Size of total soil organic carbon stock**	**Current dynamic**	**Soil carbon sequestration potential**
Organic soils (mostly drained)	medium	large losses	small
Agricultural mineral soils	medium (low for cropland, rather high for permanent grassland)	rather stable (state ranging from equilibrium to depleted)	medium for cropland, small for grassland
Forest mineral soils	high	rather stable (state ranging from equilibrium to close to saturation)	small
Settlement soils	low	rather stable (at equilibrium)	medium
Unmanaged mineral soils	very uncertain (low to high)	rather stable (at equilibrium)	unknown (probably negligible)

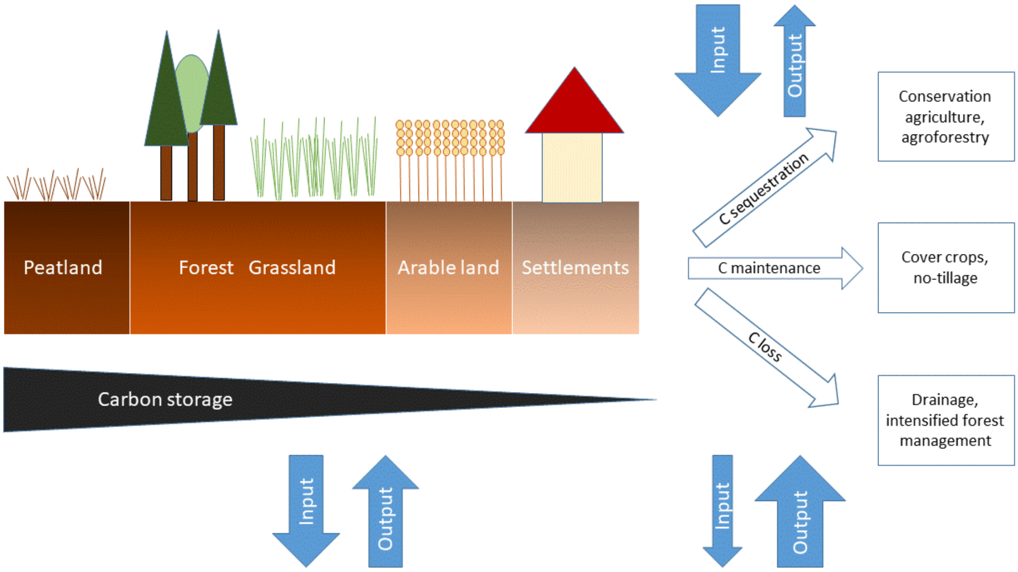

Figure 19: Conceptualization of carbon (C) sequestration. Usually, C is lost after land use conversion from native ecosystems (e.g. peatlands, forests, grasslands) to arable land. Future C storage in agricultural fields then depends on agricultural management practices, with options to regain C by increasing the organic matter input relative to output (ongoing carbon dioxide release) by e.g. conservation agriculture or agroforestry or to maintain C stocks (by e.g. cover crops or no tillage). Intensified management can lead to C losses on forest soils if outputs exceed inputs. Drainage is the main cause of C losses on organic soils. (Modified after Amelung et al. 2020.)

1.3.5. Recommendations for methods and processes to build a national map for SOC sequestration potentials and identification of additional knowledge gaps

Based on the current information available we cannot present any maps for soil carbon sequestration potentials. A first step could be to map national-scale deficits in SOC stocks based on the same approach as described for Geneva and Vaud. For this purpose, data on soil texture and SOC stocks would need to be gathered because there are no national soil maps for Switzerland to date that include this information. There is an ongoing project led by the ‘Kompetenzzentrum Boden’ financed by the Federal Office for the Environment (‘Landesweite digitale Kartierung von Kohlenstoffvorräten in Böden für das Treibhausgasinventar Schweiz’; Felix Stumpf and Armin Keller). Based on digital soil mapping, the project team is currently developing a national map for SOC stocks (for all land use categories except settlement soils) and soil texture. The map is expected to be finished in spring 2021 (*Felix Stumpf, personal communication*).

The data from analyses of the Swiss standard ‘proof of ecological performance’ (PEP) – in French called ‘prestations ecologiques requises’ (PER) and in German ‘Ökologischer Leistungsnachweis’ (ÖLN), hereafter referred to as PEP/PER/ÖLN – might offer an additional possibility. As described above for Geneva and Vaud, numerous SOC concentration analyses of topsoils (0–20 cm) are regularly carried out by some cantons (mainly in the Romandie). For these cantons, SOC stocks could be derived from the average SOC concentration (data mostly available by the measurement for PEP/PER/ÖLN) and additional information on bulk density at the canton level (this data would need to be acquired). *Gondret et al. (2020)* reported no significant bias compared with cumulated field estimations based on thousands of PEP/PER/ÖLN results (climate plans reports: *Dupla et al. 2020; Gondret et al. 2020*). Therefore, it should be possible to roughly estimate at a low cost the SOC stocks of the 0–20 cm layer for some additional regions where measurements exist. For instance, in Jura (Terres Vivantes 77a project) and Vaud (climate plans), more than 40,000 relevant C concentration analyses were available in the soil analysis laboratories (see also above).

To estimate soil carbon sequestration potentials, not only high-quality soil data but also additional information regarding the current management is necessary as shown by the project 'Hubs for Soil Improving Cropping Systems' (*Büchi et al. 2019*), by the above-mentioned regional case studies or by a recent study for Bavaria (*Wiesmeier et al. 2020*; see also section 1.3.3). Because many of the discussed measures either can only be applied where they are not yet in place (e.g. cover crops) or need a transition of the cropping system (e.g. conservation agriculture, agroforestry), we need more detailed and spatially explicit information regarding current practices (this includes the farm type, soil management, cover crops). So far, this information was gathered for only about 25% of farms in the years 2010 and 2016 by the Federal Statistical Office ('Landwirtschaftliche Betriebszählung – Zusatzerhebung, BFS') and does not include the total area and exact location. Combining these management data with current SOC stocks is thus not possible yet but is essential for realistic estimates of sequestrations potentials.

1.4. Summary

Large amounts of carbon (C) are stored in soils in the form of soil organic matter that is derived from organisms, their decomposition and transformation products, and residues from vegetation fires (char). Soil organic carbon (SOC) stocks differ largely between different soil/land-use types. Generally, SOC stocks are highest in organic soils and decrease in the following order: organic soils >> forest, grassland > arable soils > settlement soils (except for organic soils, we always refer to mineral soils). This gradient is partly the result of land use. During the last several hundred years, significant amounts of C were lost mainly due to deforestation, drainage and agricultural intensification. Given their state of C depletion and the large area they cover, agricultural mineral soils on cropland offer the highest soil C sequestration potential. Permanent grassland and forest soils occupy even larger areas, but their higher state of C storage leaves less potential for sequestration. Here, efforts should focus on maintaining SOC stocks. The same holds for unmanaged mineral soils including sites in nature reserves and riparian areas that might store significant amounts of C. Settlement soils offer an interesting possibility to sequester C, but the area concerned is rather small. Organic soils store significant amounts of C but current drainage-induced loss rates are high. Efforts should focus on cutting these emissions before their potential to store additional carbon can be considered. Currently, only small areas of natural organic soils can

sequester C. Generally, it is important to note that SOC is not permanently stored. Any additional C that is gained can be lost again if measures are not maintained. Furthermore, SOC stocks might be prone to losses induced by climate change.
Generally, national-scale estimates of SOC stocks remain highly uncertain because we lack systematic mapping and monitoring programmes at a sufficient spatial and temporal resolution. Ongoing projects are expected to improve our knowledge base in the coming years, but additional efforts are necessary for more realistic estimates of soil C sequestration potentials. Most important knowledge gaps are spatially explicit information on SOC stocks (in the case of organic soils, knowing soil depth is also very important) and texture (or at least the clay content).
Furthermore, we need spatially explicit information of the current management (e.g. whether cover crops are already used). Until more data become available, the amount of additional carbon that could be stored in Swiss soils could roughly be estimated by gathering PEP/PER/ÖLN data and applying the SOC:clay ratio approach. This method offers a good opportunity for assessing the state of mineral soils. The soil carbon deficit is estimated based on a 'minimum' soil quality requirement. For the cantons of western Switzerland, this approach showed that soils are depleted and would be able to sequester significant amounts of carbon. Whether the situation is similar for the rest of the country and whether it is feasible to sequester large amounts of C based on the available measures remains to be shown.

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2. Analysis of measures to improve the soil carbon balance or to sequester SOC

In this part (chapters II.1–II.4), we address the potential measures with which the soil organic carbon (SOC) balance of different soils in Switzerland could be improved. Furthermore, the risks or benefits associated with these measures as well as the challenges are discussed. Finally, we estimate the carbon sequestration potential (or emission reduction potential) of each measure.

Switzerland has an area of 41,285 km^2 and is generally separated into four large land use categories, namely settlement area (7.5%), agricultural area (35.9%), wooded area (31.3%) and unproductive area (25.3%) (https://www.bfs.admin.ch/bfs/fr/home/statistiques/espace-environnement/utilisation-couverture-sol.html). We consider these land use categories and added a particular soil type declination, namely organic soils. Due to their high carbon dioxide emission and different potential measures, they are treated separately. Therefore, the following report about C-sequestration potential and measures is organized as follows:

- The 'organic soils' chapter (II.1) includes agricultural organic soils (land use category: 'agricultural area') and natural peat soils (land use category: humid zones within 'unproductive area').
- The 'mineral soils' chapter (II.2) includes agricultural mineral soils (from land use category 'agricultural area').
- The 'forest soils' chapter (II.3) includes all soils from the land use category 'wooded area', including a majority of natural (i.e. non-agricultural) mineral soils.
- The 'settlement soils' chapter (II.4) includes all soils situated in the land use category 'settlement area'.

In part I, we included **unmanaged mineral soils**. This category mainly includes unproductive grassland/shrub vegetation wetlands, riparian vegetation, avalanche barriers and alpine sports infrastructures. Because these soils are unmanaged or in areas with very difficult access, it is probably difficult to enhance SOC stocks, and we are not aware of any potential measures. However, because their SOC stocks might be large, it is crucial to prevent SOC losses. As already pointed out in section 1.2.5, we currently have very limited information regarding the location and actual SOC stocks of these soils, and information regarding SOC changes are nearly inexistent. Because the total area of unmanaged mineral soils is about half the cropland area, it is not negligible, and efforts should be made to better describe this category of soils. Most likely, a significant fraction of these soils is at high altitude and is thus vulnerable to SOC losses in response to climate change. However, *Zollinger et al. (2013)* (see section 1.2.5) showed that increasing temperatures can also lead to SOC gains through enhanced vegetation growth. This shows that the effect of climate change on SOC stocks is difficult to predict, due to interacting effects on plants and soil.

2.1. Organic soils: measures to improve soil carbon balance

By Stéphane Burgos and Jens Leifeld

2.1.1. Introduction

2.1.1.1. Natural organic soils (fens and raised bogs): definition and generalities

Peat is a solid material formed by the accumulation of more or less decomposed soil organic matter (SOM) under anoxic hydromorphic conditions (*Gobat et al. 2004*). The type of peat can be very variable (*Gobat & Portal 1985*) and needs to be evaluated carefully (*Succow & Joosten 2001; Roßkopf et al. 2015*). The type of plant debris composition influences the peat physicochemical properties. Indeed, woody residues (from coniferous hardwood trees and dwarf shrubs), herbaceous residues from grasses and gramineous plants (Poaceae), from sedges (Cyperaceae, such as *Carex* or reeds), or residues from mosses (Bryophyta such as sphagnum) do not lead to the same fiber content, structure, acidity or trophic level (*Gobat et al. 2004*). The latter characteristics can also be used as classification criteria. The observation of the groundwater level and hydrological regime is also very important to understand the dynamic between the decomposing plant-derived organic matter and the resulting SOM (*Gobat et al. 2004; Gong et al. 2013*) and how it relates to the composition of outflowing water (*Scholz & Trepel 2004; Schwalm & Zeitz 2015*). The water content of saturated peat can reach 95% of fresh weight (*Gobat et al. 2004*).

The main criterion for peat soil classification is the soil organic carbon (SOC) or the SOM content. In Switzerland, peat (T-horizon) is defined as a horizon with a minimal SOM content of 30% (about 18% SOC) (*BGS 2010*). Moreover, peat soils must contain more than 40 cm of peat in the top 80 cm to be classified as such in Switzerland. They are subdivided in two classes: ‘Moor’ and ‘Halbmoor’, without and with interspersed mineral layers, respectively. In other systems, peat soils are named Histosols (*WRB 2015*) or Moor (*Ad-hoc-AG-Boden 2007*). If the peat thickness is less than 40 cm, the soil is no longer considered as peat but as Gleysol, brown soil or Fluvisol anmoorig, depending on the water regime (*BGS 2010*). However, this chemical characterization does not reflect the soil origin and build-up. It is very important to distinguish two types of peat formation:

i) Bogs are peatlands fed by rainwater and are dome shaped, elevated in relation to the neighbouring land, with a surface several meters above the water table. Yet the soil is waterlogged to the surface, like a gigantic sponge absorbing all the water. Bogs are typical for regions in altitudes with high precipitation and are mostly poor in nutrients (*Grünig et al. 1986; Gobat et al. 2004*).

ii) Fens are peatlands characterized throughout the entire soil profile by a water supply from the subsoil, the slopes or from temporary flooding, i.e. by water richer in mineral substances. They typically form in alluvial plains with flood regimes (*Grünig et al. 1986*). The nutrient content is higher than in bogs and the pH is more neutral. The peat formed from alluvial–lacustrine deposition shows a high degree of heterogeneity in terms of spatial arrangement and peat composition (*Lüdi 1935; Wohlfarth-Meyer 1987; Presler 1993; Succow & Joosten 2001*).

Peatlands are providing many ecosystem functions and services, both natural and socio-economical (*Kløve et al. 2017*). However, the services are not always compatible. On the one hand, natural peatlands host a unique biodiversity and play a role as important C sinks and regulators of biogeochemical cycles (*Wüst-Galley et al. 2015*). On the other hand, drained peatlands generate wealth, food and biomass from agriculture, forestry and peat extraction but are subject to soil subsidence (*Kløve et al. 2017*) and, thus, soil degradation.

The greenhouse gas (GHG) balance of natural peatlands, both fens and bogs, is characterized by a steady but small C sink of, on average, 0.2–0.3 t C ha^{-1} yr^{-1} (*Loisel et al. 2014*) and a moderate methane (CH_4) source. Due to the long-lasting effect of the C-sink, as compared with the shorter-lasting CH_4-effect, they are considered natural climate coolers (*Frolking & Roulet 2007*)

2.1.1.2. Organic soils used for agriculture: the impact of drainage on peat properties

Among the various anthropogenic uses, the drainage of peatlands by lowering the water table, which has created nutrient- and oxygen-rich agricultural land, plays a major role in their disappearance throughout the world. The corresponding GHG release from drained peatlands nowadays accounts for about 1.9 Pg CO_2-equivalents globally (*Leifeld & Menichetti 2018*). Artificial drainage systems became widespread during the 20th century, particularly in the Netherlands, Finland, Russia, Ireland, Canada and the United Kingdom (*Holden et al. 2006*).

Drainage of organic soils leads to numerous modifications the impacts of which are now well documented (*Ilnicki & Kuntze 1977; Presler 1993; Holden et al. 2006*). When the water table is lowered, peat soils are exposed to atmospheric oxygen and undergo the combined action of various processes: mainly desiccation, compression and oxidation (*Wösten et al. 1997*), which all lead to soil subsidence.

i) The phenomenon of desiccation or shrinkage (*Wösten et al. 1997*) is a contraction of plant fibres that induces a reduction in soil volume following desaturation.

ii) Compression or subsidence is a mechanical settlement due to the overload exerted by the stresses of the soil's own mass and probably also by heavy machines.

iii) Oxidation represents the loss of peat due to the biochemical mineralization process (*Soutter & Musy 1989*).

This reduction in peat soil thickness can be more or less accentuated by erosion and leaching phenomena (*Soutter & Musy 1989*), climate, the quantity of mineral matter in the peat, the drainage depth and seasonal fluctuations in the water table level (*Presler 1993*) or by the type of agricultural practices (*Wösten et al. 1997*). Globally, the loss of soil volume in drained peatlands ranges from 1 to 2 cm yr^{-1} (Jäggli & Juhasz 1982; Leifeld et al. 2011; Aich et al. 2013; Miettinen et al. 2013; van Mourik & Ligtendag 2015).

Because of drainage, peat physicochemical properties such as SOM content, structure, density and porosity are modified (*Holden et al. 2006; Kechavarzi et al. 2010*). Plant residues tend to be degraded, the structure becomes granular or even powdery, the number of macropores and mesopores decreases in favour of micropores, and apparent density increases (*Okruszko 1993*). *Ilnicki & Zeitz (2003)* describe it as an irreversible process during which peat is gradually transformed into degraded peat (moorsh). These changes in properties strongly influence the soil moisture regime (*Ilnicki & Zeitz 2003*). *Kechavarzi et al. (2010)* demonstrated that the bulk density of peat increases and that the porosity and hydraulic conductivity, i.e. soil permeability, decreases with the degree of decomposition. Finally, it has been shown that the hydrophobicity of organic materials is strongly related to their water content. Therefore, the drier the peat, the more difficult it is for water to infiltrate. Conversely, drying peat can lead to the formation of cracks related to the shrinkage and swelling process and thus create preferential water flows (*Okruszko 1993*). Therefore, the capacity of the soil to store, retain and transmit water decreases with the degree of degradation (*Kechavarzi et al. 2010*). The process of peat degradation with drainage is illustrated in Figure 20.

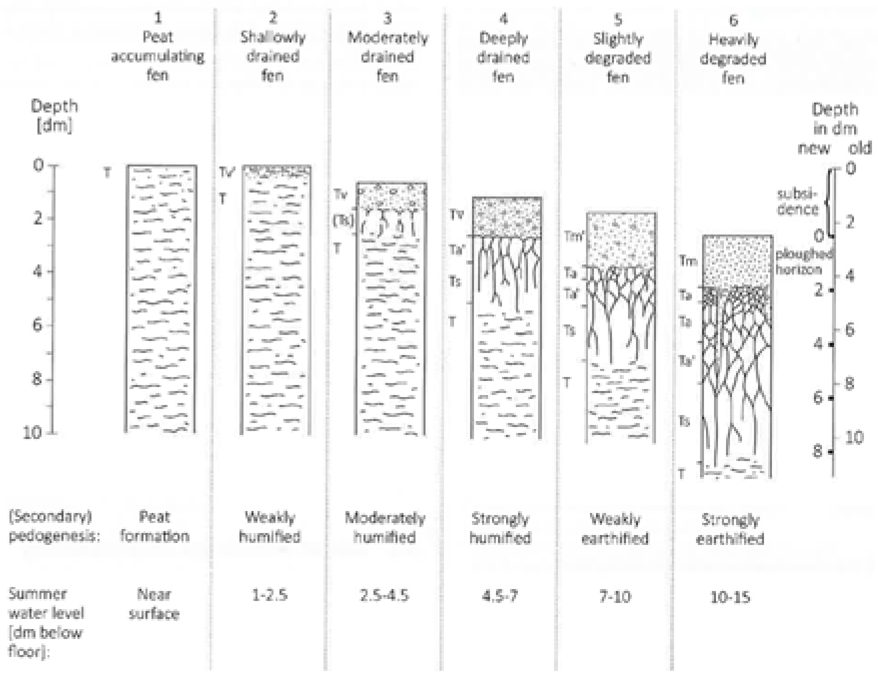

Figure 20: Change in soil properties of a peatland drained for agriculture (from Joosten et al. 2012).

For these reasons, it is very important to assess the actual state of soils in this degradation process. In many areas, degradation is almost complete, and only in the topsoil a higher SOC concentration remains, but not exceeding 18%. With this, the former organic soil has been converted to a mineral soil.

2.1.1.3. Organic soils used for agriculture: the impact of drainage on carbon stocks and GHG emissions

In Switzerland, organic soils (natural and drained) cover only 3% of the total surface but contain 28% of the SOC stocks (*Leifeld et al. 2005*). Cultivation on organic soils has led to a strong reduction in Swiss SOC stocks to nowadays about 32 Mt SOC (*Wüst-Galley et al. 2020*). Indeed, about half of the former SOC stock in organic soils has already been lost, and more than two thirds of the area of organic soils has already disappeared over the last 200 years. Around 90% of the 30,000 ha remaining are mostly intensively managed and degrading (*Wüst-Galley et al. 2020*). If the remaining peat stocks are further depleted, they will potentially emit 100 Mt CO_2-equivalents within the next decades, corresponding to twice Switzerland's total annual GHG emissions. The systematic drainage methods and their effects were analysed between 1960 and 1980 in Swiss cultivated peatlands (*Lüscher 1980, 2004; Walter 1981; Jelmini et al. 1982; Soutter & Musy 1989*).

Drainage-induced GHG emissions are relatively well studied in the temperate zone although measurements from Switzerland are yet only on their way. An overview of the rates for different GHGs was provided by *Wilson et al. (2016)* (see Table 4) and *Paul & Alewell (2018)*. In Germany, agricultural soils lose between 1 and 15 t C ha^{-1} y^{-1} (or 3.7 and 55 t CO_2 ha^{-1} y^{-1}) when drained, accompanied by substantial contributions from nitrous oxide (N_2O). They are therefore regarded as point sources or hot spots. At the same time, CH_4 emissions cease (*Kandel et al. 2018*). Since the review by *Wilson et al. (2016)*, which represents an update of the IPCC Wetland Supplement released in 2014 (*IPCC 2014*), new GHG flux data from grasslands in Germany indicated that grasslands do not release less CO_2, when drained, than croplands (*Tiemeyer et al. 2016*). Hence, grassland instead of cropland is not an option to reduce the climate warming effect of drained organic soils, as long as water tables are kept the same. For a full assessment of the GHG balance, the emissions from the drainage ditches are to be considered as well, because they can even broaden the difference between drained and restored organic soils if considered as suggested by *IPCC (2014)*.

Tillage and ploughing are considered amplifying factors because they create aerobic conditions and therefore increase decomposition, although their effect on the GHG balance of managed organic soils has not yet been measured. The relatively high export of plant biomass produced on site is another factor that accelerates the overall C loss from managed organic soils, because less organic residues are available for compensating C decomposition. The rate of CO_2 emissions, as well as the overall GHG balance, can be measured by micrometeorological as well as chamber-based methods *in situ*. In Switzerland, this has been done on only very few sites hitherto, because the measurements are resource intensive.

The tabulated data above stress that for the issue of organic soils, a reduction of their GHG source strength and protection of the remaining peat carbon is of highest priority whereas new C sequestration, which can sometimes be achieved via peatland restoration (see below, section II.1.2), has a smaller overall mitigation potential.

Table 4: Greenhouse gas emissions and dissolved organic carbon (DOC) release of drained ('d') and rewetted ('w') organic soils in the temperate zone, summed up as global warming potential (GWP), based on Wilson et al. (2016).

	CO_2-equivalents (t ha^{-1} yr^{-1})									
	CO_2 d/w		DOC d/w		CH_4 d/w		N_2O d/w		∑GWP d/w	
Forest np	9.53	-1.22	1.14	0.88	0.27	4.09	1.31	0.03	12.25	3.78
Forest nr	9.53	0.96	1.14	0.88	0.27	10.7	1.31	0.03	12.25	12.57
Arable land	28.97	0.96	1.14	0.88	1.98	10.7	6.09	0.03	38.18	12.57
Grassland np	19.43	4.09	1.14	0.88	2.04	4.09	2.01	0.03	24.62	3.78
Grassland nr dd	22.37	0.96	1.14	0.88	2.50	10.7	3.84	0.03	29.85	12.57
Grassland nr sd	13.20	0.96	1.14	0.88	2.16	10.7	0.75	0.03	17.25	12.57

d/w = drained/rewetted

np = nutrient-poor

nr = nutrient-rich

dd = deeply drained

sd = shallow drainage

2.1.1.4. Organic soils and climate change

Climate change induces changes in weather patterns such as longer drought periods or more intense rainfalls. Higher temperatures and droughts lead to lowering of the water table, drying out the peat (*Marsden & Ebmeier 2012*), which allows oxygen to enter the soil and increases peat mineralization rates. Besides, dry peat is water repellent leading to soil shrinkage and to the formation of cracks (*Okruszko 1993*). Dried and degraded peat has a more powdery structure and is subject to wind erosion if the soil is not covered (*Marsden & Ebmeier 2012*). On the other hand, heavier rainfalls can cause erosion, especially on damaged peatlands (*Marsden & Ebmeier 2012*). Flood events can lead to increased CH_4 production (*Olefeldt et al. 2017*). Drought periods, on the other hand, reduce the carbon sink function of the peatland ecosystem while at the same time also reducing CH_4 emissions (*Olefeldt et al. 2017*). In general, the soil's organic carbon dynamics is tightly linked to the hydrological cycle, where SOC losses are associated with aerobic heterotrophic peat mineralization that exceeds residue input during drier periods.

Peatland biodiversity might also be affected by climate change. Changes in temperatures and weather patterns can shift the flowering, budding or senescence period of species and impact animals that rely on these plants to feed themselves or their offspring (*Marsden & Ebmeier 2012*). In the long term, severe droughts and floods will have significant effects on vegetation composition, because each plant species has specific requirements regarding water table level, soil temperature and other abiotic variables (*Olefeldt et al. 2017*). *Bragazza et al. (2012)* reported that climate change promotes growth of vascular plants in peat soils (ericaceous shrubs). This induces changes in litter chemistry, root exudates and therefore C and nutrients cycling. Reduced soil water content and increased temperature favour fungi development, which positively feedback on vascular plant growth through symbiosis. Furthermore, roots of these plants release higher amounts of labile C, which was shown to stimulate C-degrading enzymatic activity (*Bragazza et al. 2012*).

As mentioned above (section II.1.1.1), natural and rewetted organic soils are CH_4 sources as well as CO_2 sinks. This has raised the question whether they contribute to global warming. However, it is clear that the long-term C sink function overrides the effect of CH_4 emissions in the long run, i.e. intact peatlands are considered to be climate coolers (*Frolking & Roulet 2007*). An interesting finding in the critical synthesis published by *Lindsay (2010)* indicates that the periods showing a faster peat accumulation were the warmest periods, when the vegetation was richer in specific *Sphagnum* species. Some of these mosses are more resistant to warm and dry weather conditions. Therefore, global warming may induce an increase in peat accumulation and thus SOC stocks.

Consequently, climate change can lead to very complex modifications of the biogeochemical processes and overall equilibrium of organic soils. Impacts and changes for particular regions or peatlands are difficult to predict. Organic soils that are already partly degraded and under pressure are expected to be the most impacted (*Marsden & Ebmeier 2012*). Indeed, climate change might accelerate C decomposition and increase CO_2 emissions, especially by more severe drought events, which might not be compensated by eventual higher plant residue input. Undisturbed natural peatlands are more resilient. Their capacity to sequester C is projected to decrease, but they still should keep their function of C sinks over the next decades

(*Gallego-Sala et al. 2018*). It is therefore of high importance to conserve natural peatlands and restore drained organic soils to maintain C sequestration to a maximum and limit SOC losses.

2.1.2. Assessment of measures for sustainable management of drained organic soils or to restore degraded biotopes

The restoration success of degraded organic soils needs a comprehensive description of the whole ecosystem and suitable observation methods (Schrautzer et al. 2013). Indeed, due to the complexity of the system and the high number of stakeholders, a global understanding is necessary to provide accessible information to bridge science with decision makers, politicians and the broad public (*Blum 2004*).

A precise initial assessment is mandatory for every project to avoid future cultivation problems due to inadequate melioration measures (*Fachstelle Bodenschutz Kanton Zürich 1996*). A preliminary study of the site should evaluate, among others, the water regime, the slopes, the drainage system and the peat degradation state (*Landry & Rochefort 2012*). The land use history is also to be considered. In addition, the potential of water management in relation with climate change has to be evaluated (*Jansen 1988; Hoekstra & Peerboom 2002; Trepel & Kluge 2002; Payne 2012; Curtis et al. 2014*). The effect on the production conditions needs to be analysed carefully to address socio-economic problems early on (*Rawlins & Morris 2010*).

In addition to a good understanding of the site, deciding on specific objectives allows researchers to better target the measures to be implemented (*Landry & Rochefort 2012*). The strategy changes depending on whether the objective is improvement of biodiversity, inhibition of peat mineralization, reduction of suspended solids discharge in the receiving water, C sequestration, as well as whether the site will become a natural reserve or will be managed further. As some measures are irreversible, the potential costs and benefits must be set out clearly.

2.1.2.1. Rewetting and its effects on the GHG balance of organic soils

The most straightforward way to stop peat mineralization is raising the groundwater level. This restoration technique, called rewetting, recreates a waterlogged situation by elevating the water level close to the surface. Consequently, decomposition of SOM is slowed down owing to the lack of oxygen, and the soil eventually returns to being a C sink in the long run (*Maljanen et al. 2010*). This result can be obtained with different methods depending on the objectives and the slope (*Landry & Rochefort 2012*):

i) Backfilling drainage ditches. It is the most effective and most common method. The only drawback is that it requires a large quantity of slightly decomposed peat or other material (e.g. sawdust).

ii) Dams. By building dams, the goal is to prevent water from flowing into the drains and to redistribute water throughout the area. It can be restraining in terms of costs, labour and material resources if many dams need to be installed along the entire drain length (*Lindsay 2010*). On the other hand, water surfaces in the ditches between the dams often appear, which is favourable for biodiversity restoration (*Landry & Rochefort 2012*) but likewise a potential source for CH_4 formation (*IPCC 2014*).

iii) Regulation devices. They regulate the water level and allow it to rise progressively, which prevents sudden flooding of established vegetation (*Landry & Rochefort 2012*). This technique is particularly useful if sphagnum restoration is an objective or for organic soils with former trenches or with part of the relief lower than the rest of the land. Furthermore, submerged drains can be used to raise water levels by pumping the water through the pipes into the subsoil (*van den Akker & Hendriks 2017*).

Rewetting should be considered in all its different aspects. For some organic soils, rewetting is not possible due to their position in the landscape and the associated difficulty to maintain a high stable water table (*Kløve et al. 2017*). It is also uncertain if natural peat vegetation will develop on altered peat. Furthermore, the C balance and GHG emissions are affected by diverse factors and are therefore difficult to control and predict. Water table level management is key, but it can be difficult to use it as a control method for GHG emissions on its own (*Kløve et al. 2017*).

Rewetted organic soils are usually a source of CH_4 but are neutral or even a sink for CO_2. Compared with drained soils, their N_2O source strength is low. *Wen et al. (2019)* showed a significant reduction in CO_2 losses from organic soil when raising the groundwater table to −20 cm. *Landry & Rochefort (2012)* reported a lower total respiration rate and increased photosynthetic capacity of the plants after rewetting, as well as a reduced peat density. *Lindsay (2010)* observed that 50–70% of the measured respiration (leaves, roots and soil microbial communities) was associated with the decomposition of young material, rather than with the ‘old carbon’ released from peat C stocks after rewetting. Although most studies agree that the water table level determines the amount of oxygen present in the soil, thus peat mineralization rate and CO_2 emissions, a few studies seem to observe the opposite (*Berglund & Berglund 2011; Wen et al. 2019*). This inconsistency can be attributed to diverse peat physical properties, dry matter lability, environmental conditions, nutrient availability or the decomposers present (i.e. different biological processes) (*Berglund & Berglund 2011; Wen et al. 2019*). Methane emissions increase from the anaerobic conditions in peat after elevating the water table (*Maljanen et al. 2010; Leiber-Sauheitl et al. 2014*). However, many studies cited in the synthesis of *Lindsay (2010)* agree that CH_4 emissions are not significantly increased. One study found increased CH_4 emissions after rewetting, but still lower emissions than from an undisturbed peatland. *Berglund & Berglund (2011)* measured very small or negative CH_4 fluxes for water table levels of −40 cm and −80 cm. In general, raising the water table to about −25 cm from the surface is considered a good compromise between preventing further peat loss and avoiding high CH_4 emission. Rewetting was reported to both increase and lower N_2O emissions (*Maljanen et al. 2010; Kløve et al. 2017*), but N_2O emissions from rewetted organic soils are generally low. When the water table is close to the surface, complete denitrification to diatomic nitrogen (N_2) occurs, and N_2O production resulting from partial denitrification is avoided.

Although it is generally acknowledged that rewetting of organic soils allows maintenance of C stocks and eventually returns them to being a C sink (*Lindsay 2010; Schrier-Uijl et al. 2013; Mchergui et al. 2014; Kløve et al. 2017*), the initial effects on the GHG balance are variable and GHG fluxes change over time, calling for longer-term GHG measurements on rewetted sites. If the groundwater is much more alkaline than the pH of the peat, it can lead to enhanced biological activity and C mineralization (*van Dijk et al. 2009*). Many studies agree that blocking the drains reduces the concentration of suspended solids and dissolved organic matter in the water (*Landry & Rochefort 2012; Wilson et al. 2016*), which confirms that peat mineralization is inhibited.

In summary, the response of organic soils to rewetting is site specific, and variability between sites is large. Yet, the overall GHG balance of rewetted agricultural sites is on average substantially better than that of managed sites. The GHG benefit of rewetting can partly be counteracted by increased CH_4 emissions. These CH_4 emissions can to some extent be controlled by keeping water table depths close to −20 cm, but not at the surface. A second measure to reduce CH_4 emissions with rewetting is topsoil removal (see below, section II.1.2.2).

The general picture of the GHG balance of intact, drained and rewetted organic soils is summarized in Figure 21.

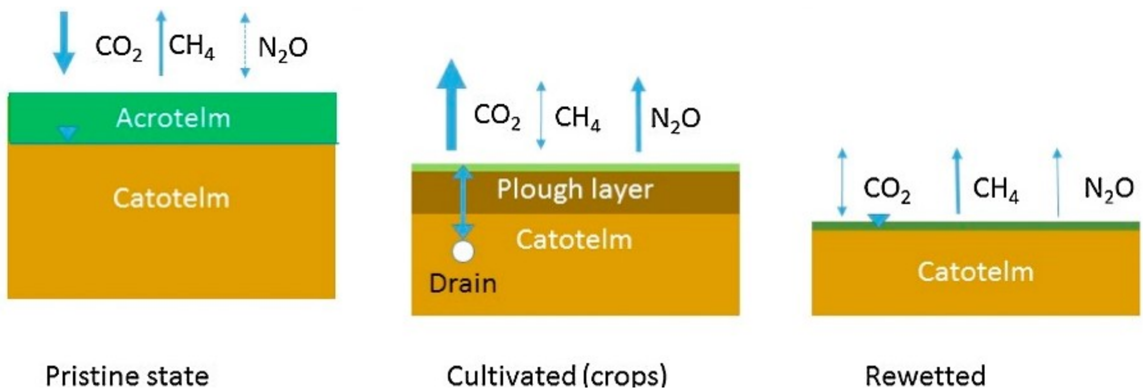

Figure 21: Conceptual presentation of typical greenhouse gas scenarios and water table position in different managed peatlands. Acrotelm is the living and rapidly decomposing plant layer, and catotelm is the more decomposed peat layer (from Kløve et al. 2017).

Advantages/disadvantages

The advantages of rewetting are peat preservation, maintenance of existing SOC stocks, an improvement of the GHG balance, and eventual soil C sequestration. Further benefits might be in improving the local climate as well as improving landscape water retention. For peatland rehabilitation, recovery of biodiversity is another asset. The major disadvantage is the impediment to common intensive land use. There is a direct conflict between the purposes of drainage and rewetting. Consequently, waterlogging will allow GHG mitigation and long-term peat conservation, while it prevents agricultural use of the land in most cases. This presents an issue at the farm level, but also at the political and market levels. Rice production as a possible viable option has not been studied yet with respect to its GHG balance on organic soil. Rewetting of organic soils can also lead to enhanced evapotranspiration by the peatland vegetation, which can reach about 800 to 1600 mm in the temperate zone (*Behrendt et al. 2001*), i.e. successful rewetting of sites is also a matter of water availability.

Our understanding of biogeochemical cycles and effects of peat characteristics on GHG emissions from peat is still limited. Moreover, the changes are not instantaneous. No changes are possibly visible one year after rewetting (*Landry & Rochefort 2012*). Some changes, such as trees which die off when the water table level is raised, are often observed only after 3 to 10 years (*Landry & Rochefort 2012*).

With current knowledge, peat preservation and efficient reduction of GHG emissions can only be achieved by rewetting. It is also the only measure yet established and largely studied within the scientific community. Many methods can be applied to raise the water table such as blocking the drains, building dams or digging out and removing the drains (*Lindsay 2010*). The long-term effects of rewetting on the GHG balance and on the potential of degraded organic soils to eventually return to being C sinks are in general acknowledged; however, these effects and processes still need to be better understood and should always be adapted to the site-specific conditions (*Kløve et al. 2017*).

In most publications, rewetting is presented as a method to bring the soil back to its natural state, encouraging peat-forming plants and C sequestration. In Switzerland, intermediate measures such as partial rewetting are also considered to limit peat degradation and subsidence while keeping on cultivating the land. The impact of such a change is not yet clearly established, especially because the effects depend on many factors and are therefore difficult to generalize. The often-postulated relationship between soil water table and CO_2 emissions indicates that moderate increases in water table may already improve the GHG balance of a site, while allowing for adopted forms of farming, e.g. grazing. A first and immediate compromise to rewetting is the water table variation over the year. In periods outside of sowing and harvesting, the water table can be raised to slow down the degradation rate (*Ferré et al. 2019*).

According to the many unknowns described above and the limited knowledge of and experience with sites that have been rewetted, particularly those managed for agriculture, no general estimate on the potential area for rewetting can be given. A rewetting of all managed organic soils in Switzerland of about 25,000 ha, at the cost of losing the corresponding agricultural productivity, would at maximum save current emissions from these areas in the order of 0.6–0.7 Mt CO_2-equivalents per year.

2.1.2.2. Topsoil removal for restoration of organic soils

Removing topsoil can support restoration of organic soils after agronomic use, because restoration can be very difficult regarding biodiversity aspects due to the presence of very competitive invasive species (*Joyce 2014; Zak et al. 2014*). Furthermore, these soils often show high nutrient contents due to former fertilization (*Behrendt et al. 2001; Kieckbusch & Schrautzer 2007*). Moreover, low species diversity due to high levels of phosphorus was reported (*Geurts et al. 2011*). This situation could be avoided by removing the topsoil and thus allowing the reinstallation of plants with low nutrient requirements (*Klimkowska et al. 2007, 2010*). Topsoil removal will strongly reduce CH_4 emissions, losses of dissolved organic

carbon, and eutrophication problems, and will improve the GHG balance of rewetted former agricultural organic soils (*Harpenslager et al. 2015; Huth et al 2020*).
In short, removing the topsoil allows the restoration of endemic peat species. It is therefore not a suitable measure if the objective is to slow down peat degradation while continuing to cultivate the land.
How large is the area on which this measure could be applied? And how large is the total emission reduction that could be achieved? With the current knowledge, it is not possible to say which area of degrading organic soils would become better suitable for rewetting when topsoil is removed.

2.1.2.3. Soil covering and mixing

Diverse trials on soil covering and mixing with mineral soil have been conducted in the past (*Frei et al. 1972*). These terms imply very different practices (tilling, ploughing) and working depths (30–200 cm). Soil covering, or backfilling, consists of adding a layer of mineral soil on top of the peat. Mixing involves a combination of mineral input and mixing with the peat top layer; this new organo-mineral horizon often extends to the ploughing depth, which is approximately 40 cm. The outcomes of these trials are diverse and depend on the initial site conditions (*König 2015*).
Backfills with high clay and silt contents seem to be less successful than sandy soil covers for improving agricultural suitability, based on the evaluation of a few sites in the Bernese Seeland (*König 2015*). Soil cover with sand did improve the stability for agricultural use (*König 2015*). However, there is little information available on the effect of soil cover on the C or GHG balance of drained organic soils. Studies from Germany (*Höper 2015*) tentatively indicate that coverage with sand (‘Sanddeckkultur’) does not change the overall CO_2 release as compared with a situation without coverage (Table 5). An ongoing study in the Swiss Rhine Valley, conducted by Agroscope, addresses the full GHG balance of an intensively managed grassland on a former fen with backfill since 2019; first results are expected in 2021.

Table 5: Greenhouse gas emissions from drained organic soils covered with sand in Germany (from Höper 2015).

Peatland type	Land use	CO_2-equivalents [t ha^{-1} yr^{-1}]
Bog	Arable farming	26.8 (22.4–31.2)
Fen	Arable farming	34.1 (33.7–34.4)
Fen	Extensive grassland	14.9 (13.2–16.5)

Sandy soil is often mixed with the peat top layer to improve physical characteristics of the peat (*FAO 2021*). Peat mixing with sand does not seem to mitigate GHG emissions but might slow down peat subsidence (*Leiber-Sauheitl et al. 2014*). However, the database on GHG measurements for peat mixing is very small, and more research is needed to quantify the effect. Peat–sand mixing in shallow organic soils did also not reduce concentrations of dissolved organic carbon as compared with pure peat, and therefore mixing does not improve water quality (*Frank et al. 2017*).

Histic Gleysols also lose SOC and have been neglected or underestimated as GHG hotspots, although they may emit as much CO_2 as Histosols (*Leiber-Sauheitl et al. 2014*). *Leiber-Sauheitl et al. (2014)* also found that soils with an organo-mineral horizon (~10% SOC) after mixing emit as much CO_2 as soils with an unmixed peat layer (>30% SOC). In addition, they found that CO_2 emissions from histic Gleysols increase linearly with water depth, even if the water table was below the peat layer.
Soil coverage is practiced by farmers to counteract soil subsidence and hence to have a thicker soil layer above the groundwater table and improve or maintain trafficability. Sand backfills have been shown to prevent the upper peat layer from drying out, to delay degradation processes, to reduce weed pressure, and to improve soil stability and resistance (*König 2015*). However, costs for soil covering are quite high for the following reasons:

i) Lack of appropriate cover material. Sand: lack of structuration, loam: danger of compaction, clay: difficult to cultivate in an appropriate humidity content.
ii) If the mineral layer is not thick enough to ensure that non-degraded peat is below the water table, the mineralization will continue or might be even enhanced. There is a risk of development of small depressions in the field that might induce further water shortage in the relatively higher zones.
iii) If farmers have to pay for the material, the measure will be very expensive. These costs are currently supported by landfill fees that producers of soil materials do not need to pay.

Regarding mixing, if the ratio of peat to mineral soil is less than 1, a certain mechanical stability can be expected. However mixing leads to:

i) Destruction of the original layers, including those which could decrease oxygen transport to deeper peat layers.
ii) Need to renew the drainage system (high costs).
iii) Difficulty to get a stable soil structure depending on the mineral material.

For both soil covering and mixing, the soil will continue to subside over time and CO_2 emissions might not change (*König 2015*), but direct GHG measurements do not yet exist in Switzerland. Moreover, if waterlogging already existed before the backfill, it cannot fully be eliminated (*König 2015*). It is therefore too early to assess the long-term outcomes of soil covering and mixing regarding the soils' GHG balances. In Switzerland, addition of mineral substrate is often used to level the soil rather than protect carbon. In this way, farmers do not need to lower the drainage and can maintain higher soil moisture (*Ferré et al. 2019*).
Sanding and deep ploughing (see following section) can be considered as successful measures to slightly improve soil stability for agricultural use (*König 2015*). Backfills with high clay and silt content are less profitable. With the new 'Revision des Raumplanungsgesetzes' (revision of the land use planning law in English), there is an obligation to valorise the earthy materials of the soil and subsoil and therefore a strong demand for the surface area of degraded soils to be improved. Because drained peat soils are mostly degraded, they are perceived as a good opportunity. Mixing is also increasing, probably more because of the pressure of private companies.
In summary, the GHG saving potential of soil covering or mixing is, according to the currently very limited knowledge, low. Both techniques bear the risk to even worsen subsidence by compressing the underlying peat (*Ferré et al. 2019*). Because soil covering or peat mixing with mineral soil does not change CO_2 emissions of the peat, it is even more unlikely that

these measures can contribute to net C sequestration, although the newly added topsoil material may increase in SOM over time.

Together, the effect of any of the above measures related to adding or mixing mineral material with peat on the soil's GHG balance has not yet been addressed properly. A first experiment of that kind is ongoing in the Rhine Valley (in Rüthi, canton of St. Gallen), but further investigations at other sites are required to evaluate the efficiency of these measures. In addition to direct GHG measurements, which are rather costly, measurements of existing SOC stocks above a radiocarbon-dated reference layer would provide information on relative effects of soil coverage on the soil's C balance (*Krüger et al. 2015, 2016*) and are recommended for future research.

2.1.2.4. Deep ploughing (deep tillage)

Deep ploughing consists of mixing the peat with the underlying mineral soil. Ploughing depths range from 60 to 150 cm (*Alcántara et al. 2017; Ferré et al 2019*). It involves tilting and inclining horizontally buried soil layers by 110–140 degrees. Deep ploughing is followed by shallow tillage (30 cm) to level the soil surface. The SOM content of the newly formed upper layer should not exceed 10–15% (*König 2015*). This technique increases the bearing capacity of the soil and soil stability.

From an agronomic viewpoint, results are best if the underlying mineral soil has a sandy texture, because sand and peat mixture is an excellent medium for plant growth (*FAO 2012*). It is more complicated to obtain a good mixture with clay soils, but they have a greater adsorption capacity for fertilizers (*FAO 2012*). Deep ploughing is often implemented in the Netherlands. This measure is regarded to be most efficient for thin peat layers (0.4 to 1 m) with underlying mineral soil of good quality (sandy sediments) (*Ferré et al. 2019*).

Alcántara et al. (2017) showed that deep ploughing increased SOC stocks in mineral croplands (see section 2.2.2.8) by slowing down SOC mineralization in subsoils. Similarly, for organic soils, deep ploughing shifts peat horizons deeper in the ground, where they are less exposed to oxygen (*König 2015*). Over time, it might be expected that buried peat horizons will be less mineralized and that the topsoil will contain less peat to be exposed to degradation processes. Overall SOC stocks are therefore expected to be better conserved. Without melioration, the top peat layer can be subject to wind erosion due to its lightweight and powdery structure. Deep ploughing or mixing with mineral soil can prevent this risk. Deep ploughing can also improve soil drainage through the new stratification of mineral material (*König 2015*). However, it is likely that subsidence processes will continue over time (*König 2015*). As with other melioration measures, one-off costs can be high. Deep ploughing is expected to help conserve existing SOC stocks, but this assumption still needs experimental validation, and the potential to sequester new C and the effects on CO_2 emissions are still not known.

2.1.2.5. Changing the crop type

Sustainable peatland use requires high water tables. According to current knowledge and experience, this is incompatible with intensive agriculture. Agricultural practices must therefore be reconsidered and adapted. In Switzerland, mineral and organic soils are often identically managed, due to unknown distribution of the different soil types (*Ferré et al. 2019*).

There is no evident relationship between GHG emission rates and crop type, tillage intensity or fertilization rates for cropped organic soils (*Norberg et al. 2016; Kløve et al. 2017*). Therefore, adaptations of cropping and soil tillage cannot mitigate GHG emissions of cultivated peatlands (*Maljanen et al. 2010; Lohila et al. 2004; Norberg et al. 2016*), at least under nordic conditions. In Finland, changing from annual to perennial crops without modifying the water table reduced CO_2 emissions, but the soil remained to be a C source (*Lohila et al. 2004*).

The effect of waterlogging on plant growth depends on their rooting systems and their tolerance to anoxic conditions (*Wen et al. 2019*). Plant growth was reported to be better at high water table levels (−40 cm compared with −60 and −70 cm) on peat soils with similar pore size distribution, but compacted peat soils required deeper drainage in order to avoid aeration problems (*Berglund & Berglund 2011*). It is very likely that optimal conditions for plant growth coincide with optimal conditions for soil respiration.

Farming practices adapted to high water levels, called paludiculture, comprise cropping of perennials on (formerly intensively used) organic soils. It could be a good compromise for peat soils with poor drainage, even if harvesting remains a challenge (*Kløve et al. 2017*). Paludiculture allows land use of organic soils while facilitating peat preservation (*Günther et al. 2015*) and sustaining ecosystem services associated with natural peatlands. This cultivation method is even expected to enable C sequestration (*Joosten et al. 2012*), although this has not yet been studied. Different potential cultures include plants for energy generation and building purposes (reed, sedge, cattail, mosses), for timber production or for pharmaceuticals and cosmetics production (*Wichtmann et al. 2016*). A few edible plants also grow on water-saturated organic soils, for example celery, radish, blueberry or cranberry (*Wichtmann et al. 2016*). Extensive livestock production is another possibility. In the Netherlands, horticultural crops are favoured on organic soils (*FAO 2012*). Paddy rice, as a novel crop in Switzerland, is also highly water tolerant and may become an alternative for the wet use of organic soils in the near future.

Paludiculture is the only currently studied way of cultivating water-saturated organic soils in the temperate zone (*Joosten et al. 2012*). It should be considered with priority for soils with deeper peat, where GHG emissions are highest and large SOC stocks remain (*Ferré et al. 2019*). If drained organic soils cannot be optimally rewetted for technical or socio-economic reasons, it has been recommended to minimize drainage, adopt crops, reduce fertilization and tillage, and favour permanent crops over annual crops (*Joosten et al. 2012*). However, for different crop types, specific soil requirements need to be defined. Whether for example rice can be cultivated on deep peat soils is not clear.

Finally, surfaces with very low opportunity to be further cultivated could eventually be rewetted into lowland peat areas, which are interesting pools for biodiversity. Also, measures such as converting some of the plant biomass of extensively managed wet organic soils *in situ* into biochar and adding this material to the degrading and rewetted organic soils are currently in discussion in other countries and could be envisaged experimentally in the future.

In Europe, arable land use is advised only for shallow (<1 m) or very shallow (<0.5 m) peat soils or for sand-covered peat (*Parent & Ilnicki 2002*). Furthermore, grassland is recommended over cropland because it requires shallower drainage and thus limits peat mineralization, CO_2 and N_2O emissions and nitrate leaching (*Parent & Ilnicki 2002*). General awareness on the importance of peat preservation is increasing, and the interest in finding economically viable productive activities on rewetted soils is growing. This is promising because peat has been estimated to be completely decomposed by 2085 in Switzerland and thus policy design and implementation is required as quickly as possible (*Ferré et al. 2019*). To better estimate the costs of potential measures and changes, more precise information is required in terms of organic soil locations, peat depth and thickness, degradation state and type of mineral subsoil (*Ferré et al. 2019*).

2.1.2.6. Changing the land use type

A recent study on 48 peat sites in Switzerland indicated that peat decomposition is sensitive to land use (*Leifeld et al. 2020*). The SOM content and the C/N ratio were significantly different among four land use types (cropland < grassland < forest < natural peatland) and hence are possible indicators of peat degradation.

Afforestation of drained peatlands has been reported to act as net C sink when rewetted (*Maljanen et al. 2010*). These results refer to specific Nordic conditions with relatively high water tables in boreal forests. In the temperate zone, the results reviewed by Wilson et al. (2016) and presented above (section II.1.1.3), as well as the findings on peat properties in Swiss organic forest soils (*Wüst-Galley et al. 2016; Leifeld et al. 2020*), do not indicate that afforestation is a means to prevent further peat degradation.

2.1.2.7. Potential strategies to implement measures

Various financial aids such as direct payments, payments for environmental services, public agri-environmental payments and carbon credits can all promote sustainable use of organic soils and preservation of peatlands (*Klaus 2007; Ferré et al. 2019*). Agglomeration payments, which are payments for environmental services addressed to collective actions among farmers, could also be an option. They are especially beneficial to incentivize a switch from drainage to rewetting. Besides, ecological areas that provide habitats for many species are much more valuable if they are interconnected. Agglomeration payments could facilitate a collective design of biodiversity areas that support spatial connectivity (*Drechsler et al. 2010*). Ongoing discussion in the EU with respect to the new Common Agricultural Policy addresses the potential of carbon farming also for organic soils.

Soil subsidence and peat mineralization demand a continuous correction of the water table in order to further cultivate the soil, which also has a direct cost. Moreover, they are associated with indirect costs from GHG emissions, C and N leaching, risks of local flooding and wind erosion. Therefore, it has been suggested to abandon current subsidies for drainage and intensive farming of organic soils (*Ferré et al. 2019*). Besides, the loss of peat soil is irreversible, and depending on the subsoil, farming might be completely impracticable when peat is completely oxidized. Consequently, losses in peat and continuous investment in deeper drainage must be counterbalanced with investments needed to change agricultural practices. Policy measures and financial support are required to incentivize farmers to cultivate their land sustainably.

In Switzerland, a major impediment to changes in agricultural practices is the fact that farmers do not always have individual control over the groundwater table level. Any modification of the water height thus requires unanimous agreement and cooperation among landowners (*Ferré et al. 2019*). More than 10% of the vegetables produced in Switzerland comes from farming on organic soils. If this intensive farming has to stop and production is to be kept at the current level within the country, adverse environmental impacts might arise elsewhere. Importing is not a preferred solution for Switzerland. Intensifying production on other agricultural lands is challenging because most of them are already intensively cultivated, and a full life cycle assessment of these options is required before any implementation. On the other hand, a change in the food system towards less meat products could liberate additional agricultural land (*Ferré et al. 2019*). Paludiculture or afforestation, maybe also for agroforestry, of cultivated peat are both promising solutions if they are accompanied by rewetting measures.

Understanding water table dynamics is a prerequisite for increasing or controlling water tables. A precise soil information is needed to evaluate the costs and benefits of a mitigation measure and to identify the actual problem (groundwater, impermeable layer). This will be addressed in the BOVE project (a FOAG resource project). The objectives of this project are to characterize the plots in five areas (in Seeland, canton of Bern) according to agricultural limitations and soil-related problems, to assess risks, to analyse the hydrology of the area and to propose scenarios to improve and cultivate degraded soils in a sustainable way.

There are several challenges that are holding back a change in agricultural practices. These include the current profitability of the land, the financial constraints that farmers face, the cultural heritage associated with the region, and the lack of information about the soil (peat depth and degradation state, mineral subsoil, etc.) (*Ferré et al. 2019*). Indeed, farmers have several constraints such as pressure from retailers, time, individual (machinery) and collective (drainage) investments. Decisions must include a long-term vision and must balance short-term provisional benefits of peatland cultivation with long-term social and environmental benefits of peatland preservation. Besides, society must recognize the importance of preserving organic soils. Subsidies can help farmers to change. At present, the benefits associated with carbon storage do not offset the profitability of intensive vegetable farming (*Ferré et al. 2019*). Policy makers have recognized the issues associated with the degradation of intensively used peatlands, but still no specific legal regulations have been defined for organic soils conservation on farmland (*Wichtmann et al. 2016*).

2.1.3. Summary

Organic soils occupy a small surface but have a very large SOC stock. There are no exact numbers for total SOC stocks in organic soils due to a lack of precise maps and estimates of peat thickness (which can be up to 20 m, see section 1.2.1 for more details). Nevertheless, it is clear that i) CO_2 emissions from peat oxidation are large, ii) the potential for future losses is high and iii) measures focus mainly on reducing emission rather than on SOC sequestration. The most promising measure is rewetting, but the consequence is a loss of the production function. However, the response of rewetted peatland is hard to predict because it is site specific and depends on local conditions. Yet, the overall GHG balance of rewetted agricultural sites is on average substantially better than that of managed sites.

2.1.4. References

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2.2. Agricultural mineral soils: measures to improve soil carbon balance

By Alice Johannes, Sonja G. Keel, Raphaël Charles, Jens Leifeld, Pascal Boivin

2.2.1. Introduction

In this chapter, mineral soils refer to soils according to the Swiss soil classification, i.e. they encompass all soils except organic soils (‘Moore’, ‘Halbmoore’). They are composed of minerals of all sizes and a small amount (in terms of weight) of soil organic matter (SOM). However, the SOM associated with (or complexed to) clay surfaces is a major driver for all chemical, biological and physical properties in mineral soils.

Agricultural soils may experience very large changes in C concentration and are dependent on human management. Therefore, the potential so sequester C in these soils is particularly large. As stated by *Amelung et al. (2020)*, ‘*the major potential for carbon sequestration is in cropland soils, especially those with large yield gaps and/or large historic soil organic carbon losses’*.

Agricultural land is divided in 4% arboriculture, viticulture and horticulture, 33% arable land, 36% grass and pasture, and 27% alpine pasture. These different subdivisions face different challenges, and the corridor for action (measures) is also quite different. On alpine pastures, the leeway is particularly narrow: not much can be done to enhance carbon storage at this altitude (with the exception of biochar addition, which is not dealt with in this report), but management that prevents soil organic carbon (SOC) losses should be maintained. However, similarly to other land uses, soils from alpine pastures are confronted with climate change, inducing a different SOC accumulation and mineralization dynamic. On the other hand, arable land but also land growing fruits and vegetables are strongly depleted in SOC and offer the most leeway. Meadows and permanent pasture can also be SOC depleted, particularly when drained, but here management options are less well studied and more complicated to improve in that respect.

The state of degradation of certain soils, clearly linked with soil carbon depletion, particularly on arable land suggests that additional carbon accumulation is achievable from a restoration perspective. Therefore, it is worthwhile to distinguish between soil qualities in the implementation of measures, because the most carbon depleted soils have the largest potential to store additional carbon. Conservation agriculture (CA) offers the prospect of regeneration, considering that priority should be given to the implementation of measures that can be posed as follows: i) C inputs (*in situ* biomass production), ii) protection and conservation of C in the soil (adaptation of tillage). Higher carbon inputs through external amendments are important to improve or maintain soil quality, but they are not considered a true soil carbon sequestration measure because this carbon is not exclusively associated with a net uptake of atmospheric carbon dioxide (CO_2) (see definition in section 1.1.1.5). In situations with a high SOM content, additional C storage cannot be excluded, but priority should be given to conserving the SOC stocks, particularly on soils with a low clay content.

As mentioned in section 1.3.1, there are measurable and realistic goals for mineral soils, which can easily be calculated through the SOC:clay ratio. This ratio was developed in relation to soil structure quality, and a ratio of 0.10 is considered a realistic goal for soil management, but it is not a rooftop. Many cropland soils manage to have ratios better than 0.10, indicating that there is potential for even more carbon storage in mineral soils, i.e. more than what we take into account for our calculations for the different measures.

2.2.2. Measures for carbon stock increase on cropland

There are two approaches for carbon stock increase in mineral arable soils, and they are based on completely different paradigms:

1. The first is more straightforward and does not consist in an agricultural management change but rather in a single technological procedure (e.g. deep tillage). The idea is to bring stable forms of carbon in an area where low mineralization rates occur. The main advantage of this approach is that carbon storage is very easy to calculate (if one puts aside some processes such as, for example for deep tillage, the extra mineralization and therefore C loss that occurs in the topsoil). This point is particularly attractive for decision making. However, these approaches are decoupled from soil functions and agronomic practices. Hence, the negative impacts of these procedures on soil functions are insufficiently studied and known.
2. The second approach is to have soil quality improvement as a goal. Evidently, improving soil quality in our SOC-depleted soils implies increasing SOC. In addition to carbon storage, this approach has multiple positive environmental impacts (e.g. decreased erosion, increased biodiversity, fewer flooding events thanks to better water infiltration, climate resilience, etc.) and agronomic advantages (soil fertility improvement). In general, the current state of C depletion of soils is so severe that the needs to increase SOC for soil quality purposes are very high and could exceed the ambitious expectations of an initiative such as the ‘4 per mille’ in France. The means to achieve this goal imply ‘agro-ecological’ methods for farm management. The main disadvantage is the reversibility of this approach. It needs a long-term responsibility towards sustainable agriculture, which can also be seen as a positive development.

As developed in *Baveye et al. (2020)*, focusing on carbon sequestration alone cannot be consistent with other environmental goals. It will not allow the commitment of farmers and may lead to very hazardous technical choices with respect to farmers, agriculture and soil functions. Both for efficiency purposes and for consistency with environmental goals, carbon sequestration in soils should be organized under the umbrella of a more general soil quality restoration framework, because SOC is essential for soil quality. From this perspective, if one considers topsoil quality a goal, carbon storage in soils will naturally be an important co-benefit.

Here, we list and discuss different measures concerning soil management in agricultural soils which are reported to increase soil quality and SOC. They are first introduced as agronomic systems (e.g. CA), which include several pillars and factors and their interactions. After having presented the systems, we discuss the separate factors because this is how the systems are usually studied in the literature, mainly in factorial long-term field experiments (LTEs). As already pointed out above, LTEs address isolated factors, because they are dedicated to identifying mechanisms and quantify them. They are indispensable to this end. However, i) they steadily apply practices with time, which is not the case for the multiple adaptations made yearly by farmers, and ii) they do not integrate the system effects of technical choices (i.e. the interconnection between practices). Therefore, results from LTEs cannot be extrapolated to on-farm C behaviour (*Hall et al. 2005; Govaerts et al. 2009*), though this hyperbole is commonly made (*Baveye 2021*).

2.2.2.1. Conservation agriculture as a system

The methods allowing for soil quality restoration accompanied with C sequestration belong to the CA pillars ('Conservation Agriculture | Food and Agriculture Organization of the United Nations'; http://www.fao.org/conservation-agriculture/en/), namely:

- **minimum mechanical soil disturbance** (no or minimum tillage),
- **permanent soil organic cover** (plants or residues),
- **maximum vegetal intensity** (including main crops and cover crops, rotation length, diversity, plant biomass).

These methods, together with mixed cropland–livestock farming and with agroforestry, were put forward by the ‘4 per mille’ initiative ('Welcome to the "4 per 1000" Initiative | 4p1000'; https://www.4p1000.org/).

Another complementary method could be added to this list, namely:

- **application of organic fertilizer.**

The launching of this initiative was followed by a considerable amount of literature emphasizing the potential of CA, or conversely highlighting its limitations, with respect to C sequestration (e.g. *Powlson et al. 2014; Pittelkow et al. 2015; Autret et al. 2016; Minasny et al. 2017; Mary et al. 2020*). It is the objective of this section to provide a short summary of the knowns and unknowns.

Past research usually studied the separate effects of the methods related to the CA pillars on experimental fields to collect information on the corresponding mechanisms. Therefore, the different agricultural methods reported to increase soil quality and SOC are introduced as single factors in this section.

Although discussed as separate factors in the following section, these pillars cannot act separate one from another. As revealed by LTEs, applying one factor alone (e.g. no-tillage) leads to very contradictory or contrasting results (e.g. *Dimassi et al. 2014; Powlson et al. 2014; Pittelkow et al. 2015; Autret et al. 201*6). CA requires combining the pillars, and the corresponding practices are interacting in the field (see section II.2.2.1.1). The CA pillars can of course also be associated with other C increase measures, e.g. temporary pasture and livestock farming, which may mean increased manure application.

Analysis of the conservation agriculture pillars applied to a large area of field cropping systems in western Switzerland

To quantify C-enhancing and C-releasing cropping systems, a field assessment of the effect of these measures has been performed on a large number of farms in the Léman Region (Vaud and Geneva cantons) (*Dupla et al. 2020*; and corresponding climate plans reports). Note that this assessment uses SOC concentration, not SOC stocks, and had to rely on topsoil data only. Therefore, the transposition to C-sequestration is not trivial and is discussed below.

The annual rates of change (ARCs) in SOC concentrations (ranging from −30‰ to +30‰) were related to the practices over the last 10-year period, for 120 fields corresponding to different cropping systems, ranging from conventional tillage (CT) with short rotations and export of residues to CA with longer rotations and maximum C inputs and plant diversity. Moreover, the overall characteristics of the farm were described (such as cultivated area, livestock, etc.), and the gross margin per hectare was quantified.

The practices that allow discriminating the ARCs over the 10 years are:

- The Soil Tillage Intensity Rating (STIR) index taking into account all mechanical interventions.
- The cover crop diversity and duration, temporary pasture duration.
- Organic manure application (including, in negative value, straw exportation).

The major C-storing systems are CA, with special emphasis on (cover)crop diversity and frequency, and **mixed cropland–livestock farming systems with long temporary pasture duration** in the rotation.

In Figure 22 ARCs of different cropping systems are presented. The role of the initial SOC:clay ratio is also highlighted in Figure 22. The lower the ratio (i.e. the higher the C depletion), the easier it is to improve it and to have a positive ARC. For low initial SOC:clay ratios, all tillage systems (from CT to no-tillage) have a positive ARC, i.e. they increase C storage. On the other hand, high initial SOC:clay ratios have very different ARC values for different tillage systems: only no-tillage and minimum tillage have a positive ARC, whereas irregular tillage and CT have a negative ARC.

This means that no-tillage is mandatory to keep SOC increasing in arable systems when the SOC:clay ratio is larger than 0.08. In other words, the minimal target value for soil quality can only be reached with no-tillage. This observation that ploughed soils hardly reach high SOC:clay ratios might seem surprising. However, it would explain why the average SOC:clay ratios are so low (<0.08) in all cantons for which sufficient data have been gathered (see section 1.3.2) and why, for years, we have considered the SOC stock of cropland soils to be at 'steady state'. With our current practices, i.e. CT, SOC stocks in topsoils are low and stay low.

Moreover, the gross margins of CA fields are equal to or larger than those obtained with the other systems (except organic farming) (see accompanying report by *Baranzini et al.*).

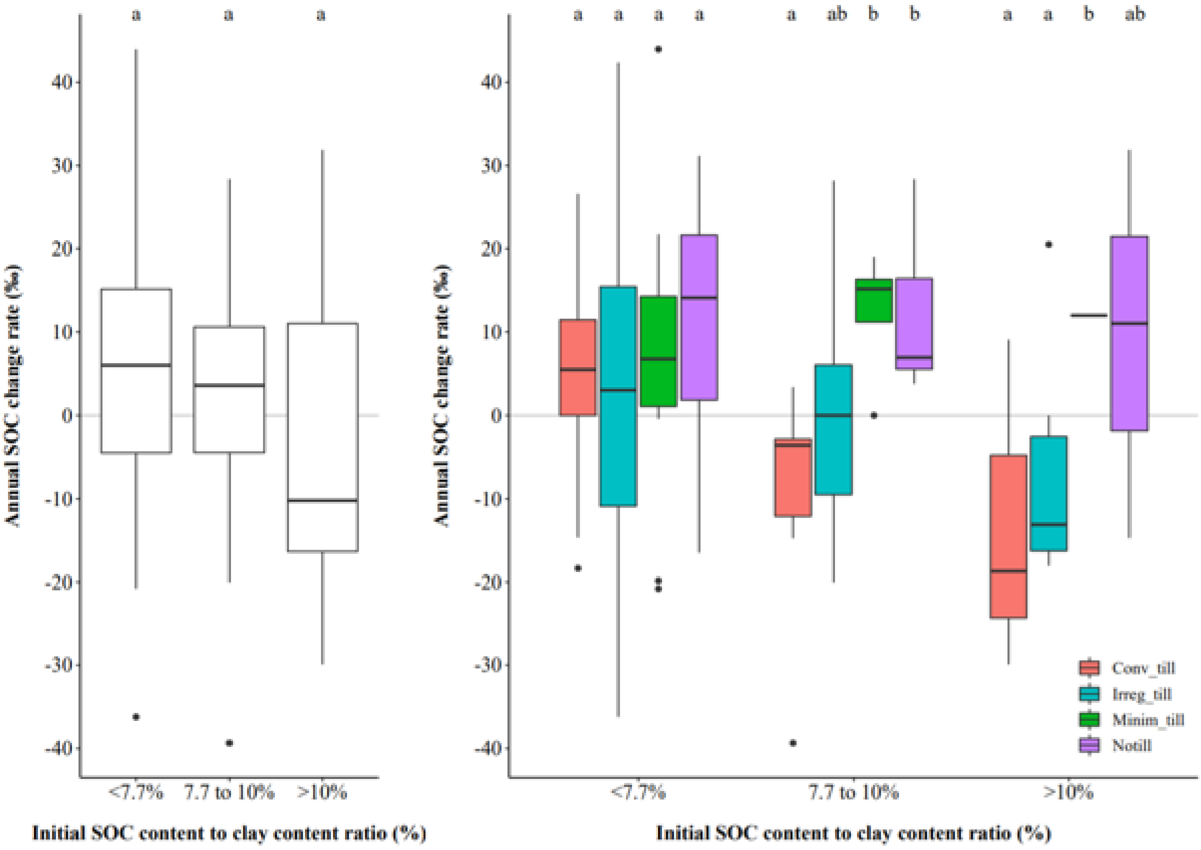

Figure 22 Boxplots of annual soil organic carbon concentration (SOC) change rate (‰) for different initial SOC:clay ratios (left) and for different tillage practices within different initial SOC:clay ratios (right) for the 0-20 cm layer.

These results are obtained on the SOC concentration of the 0–20 cm layer, with sampling performed at constant depth (not 'equivalent soil mass'), and without taking into account the 20–30 cm layer. We do not have the ARCs of the deeper layer; however, on average, **CA fields do not lose SOC in the 20–30 cm layer, contrarily to CT, and their SOC concentrations show positive relationships to the SOC:clay ratio, contrarily to CT**, even for a SOC:clay ratio larger than 0.1. Yet, without information on the subsoil carbon below 30 cm, a quantitative assessment of management practices in terms of SOC sequestration as a climate change mitigation measure remains uncertain.

Table 6 shows the correlation of factors related to an annual increase in organic matter (OM) content (evolution rate or ARC). Cover crops, in particular winter cover crops, have the highest correlation, whereas high mechanization (STIR index), initial SOM:clay ratio and the number of tillage events are negatively correlated to a SOC increase.

In the literature, the increase in SOC concentration is generally related to CA, and a relative loss of carbon under CA is reported below 10–15 cm depth down to the plough layer, compared with CT (*Angers & Eriksen-Hamel 2008*). This relative loss could limit or even balance the increase reported for the 0–10 cm topsoil layer in some papers. Therefore, the results above must be discussed in light of this information. This particular aspect is also discussed in section 2.2.2.5 concerning no-tillage as a single factor.

Table 6: Spearman correlation of variables (including vegetal intensity, cover crops, tillage, etc.) with increasing annual rate of change (ARC) in organic matter content. Boldface indicates statistical significance.

Variable	Correlation with increasing ARC	*P* value
Number of species in crop rotation	−0.216	0.097
Initial SOM:clay ratio (%)	**−0.310**	**0.016**
Proportion of temporary pasture (%)	−0.079	0.550
Proportion of spring crops (%)	−0.018	0.893
Number of potatoes/beet root	0.005	0.970
Number of intercropping cover crops	0.150	0.253
Number of species in intercropping cover cops	0.152	0.247
Number of winter cover crops	0.236	0.070
Number of species in winter cover crops	**0.505**	**0.000**
Total number of cover crops	**0.252**	**0.052**
Number of species in cover crops	**0.488**	**0.000**
Cover crops period (months)	0.,233	0.074
Number of exported straw	−0.182	0.163
Organic matter input (t ha^{-1})	0.188	0.151
Input complex organic matter (t ha^{-1})	0.196	0.132
Tractor power (cv)	0.331	0.010
Number of tillage events	**−0.366**	**0.004**
STIR index	**−0.342**	**0.008**

In conclusion of this section:

- The large ARCs (>10‰) correspond to CA systems with full no-tillage and with highly diversified cover crops included before winter crops (Geneva mostly). These rotations likely provide higher OM inputs to soil. Low cover crop intensity and straw export jeopardize the effects of CA.
- These systems allow equal or larger gross margin per hectare compared with CT.
- OM application is a secondary driver of SOC change. However, high SOC change rates (>20‰) are observed for fields with little organic manure application.
- Large mechanical intensity (STIR index) was correlated to negative SOC change rates. However, if the soil is SOC-depleted (SOC:clay ratio < 0.08), it is possible to observe increasing SOC concentrations with CT.
- The negative ARCs are, in addition to high STIR values, linked to low OM inputs and low cover crop intensity.

Potential calculation for CA:

- SOC per hectare: 0.4–0.8 t ha^{-1} yr^{-1} (using the values mentioned above: 10–20‰ annual increase [median and outer box of 'No-till' boxplot from Figure 22] with a carbon stock for cropland of 40 t ha^{-1}; 0.63 t ha^{-1} yr^{-1} [*Autret et al. 2016*]). These potentials depend on the initial state of the SOC stock, i.e. the initial SOC:clay ratio.
- Could be applied on all cropland areas where not yet applied (approximately 356,600 ha, which is equal to 90% of the total cropland area of 396,200 ha in 2017).
- Timeframe: immediately available but takes several years to become measurable (see accompanying report by *Fliessbach et al.*).

Pros, co-benefits:

- **Improved soil quality**, water quality, water cycle regulation, biodiversity.
- CA can reach a SOC:clay ratio of 0.10 (or even 0.12) (*Autret et al. 2016*).
- Potential to mitigate greenhouse gas (GHG) emissions in Swiss agriculture (*Necpalova et al. 2018*).
- Reduced soil erosion.

Cons, risks:

- Risk of losing SOC quickly if changing back to conventional farming (C not permanently stored).
- Lower yields (*Autret et al. 2016*)
- Depends on successful capacity building of farmers because considerable agronomic knowledge is necessary.

2.2.2.2. Organic agriculture as a field cropping system

The proper functioning of the soil is essential in organic farming to allow working without synthetic products. The maintenance of OM and the management of its cycle within the rotation are essential factors for plant nutrition and health, as well as to compensate for mostly intensive soil tillage. The goal is to obtain soils that are properly stocked with OM after a transition period (5–10 years). According to system comparisons, soils managed in organic agriculture accumulate yearly 0.17–0.45 t C ha^{-1} more than conventional systems, mainly due to temporary meadows and organic fertilizers (*Gattinger et al. 2012*). These differences could also be observed in the DOK LTE (that compares bio-Dynamic, bio-Organic and conventional ('Konventionell') farming systems), and were mainly explained by the mineral fertilization adopted in the conventional system (*Fliessbach 2007*). However, all treatments led to a loss of SOC over time (*Keel et al. 2019*).

The measures implemented in organic farming are sources of references for other systems. This concerns in particular a farm-level approach, the application of organic fertilizers, the cultivation of 20% grassland in field crop rotations, the organization of a rotation taking into account the mineralization of OM to limit the use of external fertilizers, and the maintenance of an intensive soil biological activity. If these measures consist in initially valorising OM for plant growth, without gain or loss of C, they also constitute prerequisites for an accumulation of OM in the long term. In this perspective, minimum soil disturbance is also adopted in organic systems.

The 'HUBS for SICS' network of 60 arable farms in western and eastern Switzerland has also shown that no-tillage fields have a more advantageous C status in the topsoil (0–5 cm) than fields on conventional farms (*Büchi et al. in prep*). However, no-tillage fields showed no difference to organic fields, which nevertheless applied intensive tillage. Even if a stratification of OM was noted in no-tillage, both systems have shown equivalent surface and depth results in terms of SOC:clay ratio, SOC stock, mean aggregate size and proportion of macro-aggregates (with an accumulation of C in the large macro-aggregates). These results showed great variations within the systems, but they also indicated that cross-effects must be considered in relation to the specificities of the cropping techniques and the different biogeochemical cycles between the systems. Across systems, clay content and soil biological variables were major drivers of SOC. In this network, higher gas transport capability was measured on organic fields and was associated with higher SOC inputs and microbial activity, in both topsoil and subsoil, due to improved conditions for root growth provided by tillage compared with the no-tillage system (*Colombi et al. 2019*). Microbial network complexity and the abundance of keystone taxa in roots were significantly higher in the organic system than in the no-tillage system (*Banerjee et al. 2019*). In terms of practices, one should also note the presence of mandatory temporary grasslands at 20% of the organic farm rotation (even without livestock), which is a considerable factor in the regeneration of OM (*Fliessbach et al. 2007*).

<u>Potential calculation for organic agriculture:</u>

- Similar to CA.

<u>Pros, co-benefits:</u>

- Improved soil quality.
- Potentials to mitigate GHG emissions for Swiss agriculture.
- Less pesticides (compared with no-tillage)

<u>Cons, risks:</u>

- Risk of losing SOC quickly if changing back to conventional farming.
- Depends on successful capacity building of farmers because considerable agronomic knowledge is necessary.

2.2.2.3. Mixed farming as a system

Mixed farming systems provide the prerequisites for an interesting agriculture in terms of C inputs: presence of long-lasting grass–clover leys in the rotation, long-straw cereals, management of farmyard manure (form, spreading), reduction of mineral nitrogen applications, catch crops and cover crops in the inter-crop period, and balanced interconnection of crop and animal productions at farm level. A coherent combination of these factors allows C accumulation even more efficiently when CA principles are added: systematic use of living soil cover, reduced tillage. The choice of crops in the rotation is also a factor to be considered in maximizing the use of growing periods with species with high biomass production and an intensification of the crops succession (reduced summer and winter breaks). The utilization of grazing animals in the cover crops may also be a way of accelerating the transformation of green biomass to SOM before the succeeding crop.

Potential calculation for re-developing mixed farms:

- No known study observed the potential associated with this system. The potential of the system is related to the potential of single factors described below such as organic fertilizers (2.2.2.6) or vegetal intensity (2.2.2.7).

Pros, co-benefits:

- Improved soil quality through increased presence of grass–clover ley in the rotation.
- Synergy effects between systems in terms of C balance.

Cons, risks:

- Becoming economically less attractive.
- A problem related to this system is the definition of 'system boundaries' for C sequestration. In Switzerland, fodder for livestock is partly imported, which means that the C sequestration has taken place elsewhere, even if the C is stored in Swiss soils.
- GHG emissions from animals might be much higher than additional C stored in soils.

2.2.2.4. Agroforestry as a system

Agroforestry systems are defined as any combination of woody plants (trees or shrubs) and agriculture (cropland or grassland). Whereas orchards combined with pastures have a long tradition in Switzerland, new combinations have been implemented in recent years and interest is growing. Depending on the tree, not only fruits can be harvested, but also high-quality furniture wood can be produced (e.g. cherry, walnut). Agroforestry systems can deliver many ecosystem services such as increased biodiversity, increased resilience to climate extremes due to two different plant functional types, reduction in nitrate losses and reduction in erosion. Soil carbon sequestration from tree roots can occur not only in topsoil but also in deeper layers, below the horizon of the annual crop. In addition, C can be sequestered in the stem of trees, and part of the wood might be used to produce biochar. On the other hand, any newly implemented agroforestry system reduces the area available for crop production; hence, careful consideration of possible trade-offs is needed. Currently, 140 farms (in the cantons of Geneva, Vaud, Neuchâtel and Jura) participate in a FOAG resource project on large-scale implementation of different agroforestry systems in Switzerland.

Potentials for soil C sequestration show a large range, and only few data exist for Switzerland. In an apple intercropping system, considerable amounts of 0.51 t C ha^{-1} yr^{-1} were sequestered in topsoil and 0.86 t C ha^{-1} yr^{-1} for the whole studied profile (0–60 cm) during the first seven years of an agroforestry system (*Seitz et al. 2017*). Most likely, the accumulation of SOC was due to herbs and not the trees. In a study that compared different European agroforestry systems, C sequestration rates (for soil and tree biomass combined) were highest in Swiss cherry orchards on permanent grassland that are grazed with cattle (*Kay et al. 2018*). In France, lower SOC accumulation rates (0–30 cm depth) were measured that ranged between 0.09 and 0.46 t C ha^{-1} yr^{-1} (mean: 0.24 t C ha^{-1} yr^{-1}) across five different sites with silvoarable systems (*Cardinael et al. 2017*). Based on a meta-analysis, SOC stocks increased by 40% in the 0–30 cm layer (*De Stefano & Jacobson 2017*), but there are also studies showing no increase in SOC for 0–150 cm (*Upson & Burgess 2013*).

<u>Potential calculation for agroforestry:</u>

- SOC per hectare: 0.0–0.86 t C ha^{-1} yr^{-1}
- For the surface: Due to reductions in the cropping area, this measure is only recommended for a fraction of the total cropland area (10% of cropland area: 39620 ha)
- Timeframe: ready to be implemented, but careful consideration of trade-offs is necessary.

<u>Pros, co-benefits:</u>

- Biodiversity.
- Reduced nitrate leaching.
- Higher resilience to climate change because of two different plant functional types.
- Carbon sequestration also in trees.

<u>Cons, risks:</u>

- Reduced cropping area.
- Reduced crop yields (shading).
- Long-term planning/investment necessary.
- Management is more complex (timing of tree and crop harvest, more machinery).

2.2.2.5. No-tillage as single factor

No-tillage alone results in a redistribution of C within the soil profile (higher C concentrations in topsoil and lower C concentrations below), but does not lead to SOC sequestration *(Baker et al. 2007; Luo et al. 2010)*. Nonetheless, it can improve soil quality by reducing mechanical disturbance.

However, there is a gradient between no-tillage and conventional inversion tillage in practice. Many different techniques (associated with different wordings) are used by farmers (Table 7). Soil tillage, and in particular soil inversion (e.g. with ploughing) may cause primary mineralization (transformation of plant residues to mineral and gas species rather than humus) and secondary mineralization of humic substances. The mechanical intensity undergone by soil is summarized by the STIR index. The deeper the tillage, the larger the inversion, the higher is the STIR index.

Table 7: Different expressions associated with tillage, reduced tillage and no-tillage. Adapted from Morris et al. (2010) and Peigné et al. (2007).

Commonly used expressions in the literature	Working depth	Inversion	Mixing horizons	Soil cover			
ploughing, mouldboard ploughing, conventional ploughing	Deep (15–40 cm)	Yes	Yes	Null			
Chisel plough	Deep	No	Yes	Low			No Tillage
Topsoiling	Deep	No	No	Medium		Conservation Tillage	
Subsoiling	Very deep (40–80 cm)	No	No	Medium			
Reduced and minimum tillage	Superficial (0–15 cm)	No	No	Medium	Reduced Tillage		
Direct drilling, no-tillage or no-till, zero tillage	Superficial	No	No	Very high			
Strip tillage	Superficial	No	No	High			

Tillage practices have different, partially counteracting effects on OM mineralization rates. In the Ap-horizon, tillage is usually reported to accelerate OM turnover. The associated reasons are the enhanced mineralization of C through increased oxygen availability for microorganisms and the breakdown of aggregates, making food available to microorganisms that is otherwise physically protected. On the other hand, tillage transports OM to deeper layers of soil, where decomposition rates are lower. Therefore, several reviews show that there is no overall increase in SOC with no-tillage practices but rather a redistribution of SOC over the soil profile (*Baker et al. 2007; Luo et al. 2010*). The regularly observed accumulation of OM in the topsoil is thus counterbalanced by an OM decrease in the plough pan and below, owing to the missing input of agricultural residues by tillage to deeper parts of the soil profile. No-tillage itself can therefore be interpreted as insufficient for increasing C storage. However, the ameliorated soil quality through less disturbance and more SOC at the surface (reduced surface runoff, better conditions for seedlings to develop, etc.) have indirect positive effects on other environmental parameters, including soil fertility, and allow improving the soil adaptation to climate change. Effects on soil-borne nitrous oxide (N_2O) emissions are mixed, site dependent, but in general develop towards the positive the longer the no-tillage system is maintained (*van Kessel et al. 2013*).

However, the second pillar of CA, permanent organic cover, and in particular intense cover crops included before winter crops, require no-tillage. Consequently, the SOC changes on-farm are strongly responsive to the STIR index (see Table 6). The reason is of practical nature and is observed by farm advisors and not in scientific experiments: sowing cover crops in combination with tillage is more difficult than with no-tillage. With tillage, sowing is done later because of the time needed for tillage and seedbed preparation. Furthermore, soils in our regions are often too dry for sowing by the end of the summer, and dry conditions (of tilled soils particularly) may prevent the successful development of cover crops. These elements lead to unproductive cover crops in tilled soils, yielding insufficient biomass to ensure high C input in soils. Hence, no-tillage in combination with more productive cover crops provides ways to increase OM inputs to soil.

The importance of cover crops in alleviating negative effects of reduced soil tillage was highlighted by *Büchi et al. (2018)*. In an LTE of Agrosocope in Changins, the P29C trial, all tillage practices have lost OM since 1969 with a crop rotation with wheat, grain maize, wheat, and rape (*Büchi et al. 2018*). By 2012, the tillage treatment had lost 9 t C ha^{-1} (0–20 cm) whereas minimum tillage had reduced the loss to 1 t C ha^{-1}. However, a full C balance could not be provided given that subsoil C was not measured. Modelling based on data from this trial indicated that the introduction of cover crops resulted in C accumulation in the topsoil, especially as the green manure was cultivated systematically and tillage intensity was reduced. By 2016, only a combination of cover crops and reduced tillage or direct seeding showed an accumulation of C (10 t C ha^{-1} as compared with 1969). Yet cover crops were not able to compensate for C losses due to ploughing. Minimum shallow tillage might offer an intermediate solution to avoid negative constraints of no-tillage.

No-tillage as a stand-alone factor bears little potential to increase C storage. However, it is the combination of no-tillage with cover crops that bears the potential of this measure. As for most measures, it is dependent on initial soil conditions (in particular initial SOC content). As shown in the section about CA (section II.2.2.1), no-tillage becomes a necessary measure to increase C storage when the initial SOC content is already high. In contrast to tillage systems, no-tillage (+ cover crop) still has a high potential for soils with already relatively large SOC content.

<u>Potential calculation for no-tillage:</u>

- Potentials for this measure are not relevant alone and best taken into account in the potential calculation for CA systems (section 2.2.2.1).
- Surface: could be applied on the whole cropland surface where not yet applied (about 356,600 ha).
- Timeframe: immediately available.

Pros, co-benefits:

- For the farmer: reduced costs and working time.
- For environmental protection: reduced erosion risk, better water infiltration in soils.
- In practice, no-tillage management enables farmers to work more efficiently with cover crops (i.e. faster sowing and better surface conditions for germination).

Cons, risks:

- More use of herbicides.
- Lower yields during the transition phase from ploughing to no-tillage (5–10 years).
- Possibly higher N_2O emissions (*Vermue et al. 2016*).

2.2.2.6. Organic amendments as single factor (no true SOC sequestration)

Organic amendments such as farmyard manure, slurry or compost are the traditional way of 'feeding' the soil and ensuring long-term fertility. Most organic amendments are associated with livestock. The current evolution of Swiss farms is rather to abandon cattle, which makes this kind of amendment more difficult to come by.

Another aspect should also be mentioned: In most cases, organic amendments cannot be considered as genuine C sequestration measures (see section 1.1.1.5 for a definition of C sequestration) because they do not provide newly assimilated C to the terrestrial system, but usually just re-allocate material in the landscape. This recycling of nutrients may lead to (wanted) local increases in SOM and improved soil fertility, but it does not lead to a higher overall SOC storage in soils when the system boundaries are properly considered (*Leifeld et al. 2019*). Hence, if an increase in SOC is induced by organic amendments that were not produced by the agroecosystem *in situ*, any corresponding change in SOC at the place(s) of origin needs to be factored in. This is particularly challenging when manure is produced from animals feeding on feed imports. The case is different with biochar or ramial chipped wood, if produced with locally grown biomass.

In addition, it is important to note that significant amounts of C are lost during storage of manure and compost (about 50% in the case of compost). These losses need to be factored in when estimating the C balance of soils.

Because nearly all farmyard manure produced in Switzerland is already returned to fields (except for small amounts that go to biogas plants) and often causes environmental problems (ammonia emission, nitrate losses, N_2O emissions), there is no potential to increase farmyard manure additions within ecological limits. In Switzerland, green waste from households and organic waste from food industry remain largely unused (*Burg et al. 2018*), and part of these could be recycled in the form of compost. Compost has the advantage that it has a high retention in soil.

Potential increase in SOC stocks for organic amendments (no true SOC sequestration):

- SOC per hectare: 0.13–0.41 t ha^{-1} yr^{-1} (*Poulton et al. 2018*).
- SOC per hectare: 0.82–1.13 t ha^{-1} yr^{-1} for high application rates (10 t dry matter ha^{-1} yr^{-1}) of compost.
- Surface: theoretically on entire area that currently receives mineral fertilizer (replace mineral by organic fertilizer). In reality, the amount of compost that could be produced is the limiting factor and the realistic area will be rather small.
- Timeframe: immediately available. Compost production would need to be scaled up.

Pros, co-benefits:

- Local (only if cattle feed is not imported).
- Increased soil biodiversity.
- Compost: GHG emission savings through fertilizer replacement.
- Compost: high retention in the soil.

Cons, risks:

- Increased N_2O and ammonia emissions.
- Increased nitrate losses.
- In most situations not a true C sequestration measure (due to issues with system boundaries and emissions during storage or from cattle).
- Some problems need to be solved for compost (plastic in organic waste).

2.2.2.7. Vegetal intensity and diversity as single factor

2.2.2.7.1. Crop diversity

Longer, diverse rotations are a pillar of CA. However, this measure was recommended while many cropping systems worldwide tended to monoculture. CA recommends three main crops in the rotation, whereas Swiss mandatory practices include four crops in general (including temporary pasture). This condition is always fulfilled in Swiss cropping systems. However, this prerequisite refers mainly to phytosanitary measures to avoid increasing use of pesticides. It does not refer to the way of maximizing photosynthesis and biomass production through the association of perennial and annual plants (agroforestry), the relay cropping, the choice of cultivated species and varieties (winter/spring, earliness), the use of crop mixtures to exploit facilitation interactions between plants (such as legume-based intercropping). Therefore, this measure will not be discussed in more detail in the present report, although it will offer new solutions for cropping systems in transition. However, there is a subtlety to consider: in some cantons, longer rotations mean more sugar beets or potatoes, which tend to impact the SOC content negatively.

2.2.2.7.2. Increased grass–clover leys (temporary grasslands in crop rotations)

The implementation of grass–clover leys in crop rotations leads to higher vegetal intensity, more biomass production and increase in C sequestration and storage. In a study comparing ley rotation and cereal monoculture (*Börjesson et al. 2018*), a significant increase in SOC stocks was found in the ley-dominated rotation, compared with the cereal monoculture, the difference being 0.36 and 0.59 t C ha^{-1} yr^{-1} in the 0–20 cm layer for the two different sites (clay and loam textures). However, the potential in Switzerland is likely to be lower, given that monoculture is not practiced and therefore the difference in vegetal intensity is less pronounced. Furthermore, grass–clover leys are already widespread. Another aspect that should be considered, especially when comparing the use of leys with the practice of CA, is that the positive effect of leys is partly counterbalanced by the negative effect of tillage usually practiced after leys. Lastly, it is important to point out that methan emissions from cattle might be much higher than additional C stored in soils. This is true for any type of fodder crop planted.

2.2.2.7.3. Cover crops

Recent literature mostly emphasizes the positive role of cover crop intensity (high biomass) and diversity (multi-species cover crops) on the C balance (*Poeplau & Don 2015; Wendling et al. 2019*). This is demonstrated at the micro-scale level (*Kravchenko et al. 2019*), as well as at the continental level with meta-analyses (e.g. *O'Connell et al. 2015; Ruis & Blanco-Canqui 2017; Mary et al. 2020*).

Covering the soil in fall is mandatory in Swiss agriculture. Therefore, there is always a cover crop between a soon-harvested winter crop and a summer crop. This practice is therefore correlated with the frequency of summer crops in the rotations, and there is only a limited additional potential for C sequestration related to the use of cover crops with higher biomass. However, in favourable regions, it is also possible to seed a cover crop before a winter crop. In the case of wheat and barley, winter crops contribute 97–99% to the total wheat- and barley-growing areas, and therefore the potential in rotations with winter crops might be large. However, in practice, cover crops before winter crops are only used by no-tillage farmers (section 2.2.2.5), and there are huge differences between farmers in terms of cover crops biomass (e.g. dry biomass ranging from 1 t ha^{-1} to more than 12 t ha^{-1} in Geneva, AgriGenève, *N. Courtois, personal communication*) as well as in the number of species (from 1 to more than 12). Furthermore, cover crop biomass varies largely between years due to meteorological conditions.

Pioneer farmers tend to merge cover crops and main crops in different ways, even with continuous cover crops. This practice is put forward by the concept of vegetal intensity, involving these two conditions: i) soils are always covered with a high biomass of living plants, which ii) present a high diversity.

In summary, cover crops are essential for the restoration of OM in field cropping systems, but also in other production systems (vineyards). If well-managed, cover crops contribute to the regeneration of the OM balance by providing biomass to compensate for the mineralization of OM and crop exports. This regeneration requires an intensive use of the period of intercrop (i.e. the time between the main crops), a high biomass production and a carbon-to-nitrogen ratio favourable for a balanced humification. A minimum dry matter biomass of 3 t ha^{-1} (or roughly 1.35 t C ha^{-1}) is expected to ensure most expected agroecosystem services (i.e. avoidance of erosion and N leaching, promotion of symbiotic fixation, weed control, biodiversity, etc.). Favourable conditions bring 6–8 t dry matter biomass per hectare (or 2.7–3.6 t C ha^{-1}) from July to winter. The association of several complementary species is essential to increase the quantity and stability of biomass produced. *Wendling et al. (2016)* increased the quantity of N acquired (via symbiotic fixation of legumes) but also aimed to control the quality of the biomass (C/N ratio) in relation to the period of destruction, to the N cycle (pre-emptive competition, release), and to the N requirements of the following crop. For a balanced incorporation of cover crop biomass into the soil, a C/N ratio between 15 and 20 is recommended to avoid N starvation. Other C/N ratios can be aimed for according to the objectives (e.g. high N availability). LTEs have shown that an increase in organic C concentration in the surface horizon can be expected by a high frequency of cover crops in the crop rotation (*Büchi et al. 2018*).

<u>Potential sequestration for this measure (cover crops):</u>

- SOC per hectare: 0.32 ± 0.08 t ha^{-1} yr^{-1} (*Poeplau & Don 2015*).
- For the surface: Cover crops could be applied on all cropland where not yet used. Estimates of current use are highly uncertain and mostly depend on main crop types (no cover crops on about 40% of cropland area or 158,480 ha).

<u>Pros, co-benefits:</u>

- Reduction of nitrate losses, increase in biodiversity.

<u>Cons, risks:</u>

- Risk of disseminating diseases.
- Additional work and costs for farmers.

2.2.2.8. Deep tillage (deep ploughing) as single factor

Deep tillage is reported to lead to very large increases in SOC stocks (42% in 45 years, *Alcántara et al. 2016*; 69% or 179 t C ha^{-1} in 20 years, *Schiedung et al. 2019*). This method is sometimes used as a soil improvement measure, e.g. to break up hard pans or to eliminate waterlogged soils, and is also applied on peatland soils (see section 2.1.2.4 on deep ploughing of organic soils). Due to its high potential, it has been proposed as a soil C sequestration measure. The idea is to bury the C so deep that mineralization hardly occurs, resulting in a somewhat stable form of C in the soil. Moving SOC-poor subsoil to the top layer modifies most soil functions significantly and for a long time, because it leads to reduced SOC stocks in topsoil. A mechanistic argument considering only C storage and disregarding soil quality claims that in this new topsoil, long-term accumulation of SOC can occur. To date, this procedure has not been well tested, and ecological as well as economic effects need to be further studied before this measure could be applied. It bears significant risks for soil erosion, especially in hilly areas, because the subsoil (completely deprived of its most important binding agent, SOC) does not have the strength, structure and resilience to cope with surface runoff and may partially disappear before it could successfully accumulate SOC at its surface. Another very important negative effect of putting subsoil at the surface is that this ‘new topsoil’ reduces soil fertility dramatically and increases structure vulnerability, e.g. due to compaction or erosion. Most likely, yields will decrease strongly for several years. Mechanical soil disturbances (the deeper the worse) disrupt the soil structure and therefore the natural habitat of living soil organisms. Avoiding mechanical disturbances as much as possible reduces these risks. Consequently, deep tillage cannot be recommended as a SOC sequestration measure. Experimental research on the issues raised above is required to provide guidance and should consider Swiss soil specificities such as the topsoil being mostly not rich in C and soils being too shallow.

Potential calculation for this measure:

- SOC per hectare: 0.93–8.9 t ha^{-1} yr^{-1}.
- For surface: Only a small fraction of soils are sufficiently deep to be the object of such a measure.
- Timeframe: not ready for implementation. However, effect for C storage would be immediate.

Pros, co-benefits:

- The operation is done only once.
- Large effect on SOC stock.
- Low risk of losing sequestered C in deep soil (low turnover rates in subsoil).

Cons, risks:

- High soil structure degradation.
- Risk of compaction during the operation (very heavy machinery).
- Increased mineralization over a large depth.
- Reduced yields for several years.

2.2.2.9. Biochar amendment as single factor

Here, we do not mention at length the use of biochar to increase C storage because this aspect is covered in the accompanying report by *Schmidt et al. (2021)*. However, some points are mentioned here because biochar amendment is an often-cited solution for C storage in soils. A clear advantage for C storage is that biochar represents a very stable form of C. However, it is only a true sequestration measure on agricultural soils, if biochar is produced with biomass that was grown on-farm (or within settlement areas in the case of settlement soils). Currently, the main disadvantages of this measure are the high cost and limited availability. Given the little amount available, the use of biochar would be most interesting to improve poor-quality substrates in settlement soils (see chapter II.4 on settlement soils). In this case, there would be a real substrate quality improvement, whereas biochar amendment is of minimal impact for soil quality in agricultural soils (given that Swiss agricultural soils are already quite fertile).

2.2.2.10. Land use change

Soils under permanent grassland generally have higher SOC stocks compared with arable soils (50.7 t C ha^{-1} and 40.6 t C ha^{-1} respectively; 0–20 cm; *Leifeld et al. 2005*). Land use conversions from grassland to cropland are thus associated with considerable SOC losses until a new SOC equilibrium is reached about 25 years later, as shown for different sites in the temperate zone (*Poeplau et al. 2011*) and in a Swiss experiment (Hermle et al. 2008). Preventing such conversions is thus a way to circumvent SOC losses.

On the other hand, grassland establishment on arable land causes a long-lasting (more than 100 years) and significant increase in SOC stocks of 128% for many sites in the temperate zone (*Poeplau et al. 2011*). Furthermore, it is a very effective measure to reduce erosion and nitrate leaching (*Prasuhn 2020*). However, suitable crop rotations including e.g. grass–clover leys are also effective in preventing erosion. About 11% of the Swiss cropland area is potentially affected by strong erosion (*Bircher et al. 2019*). Only a part thereof is affected by a real risk associated with unsuitable crop rotations. Conversions of cropland or grassland to forest (i.e. afforestation) are discussed in the chapter on forest soils (chapter II.3).

<u>Potential calculation for cropland-to-grassland conversion:</u>

- SOC per hectare: 0.73 ± 0.17 t ha^{-1} yr^{-1}.
- Surface: a few percent of the cropland area. (5% of cropland area: 19,810 ha)

<u>Pros, co-benefits:</u>

- Permanent soil cover.
- Very effective reduction of erosion.
- Reduction of nitrate leaching.
- Reduced leaching of plant production agents.

<u>Cons, risks:</u>

- Reduction of agricultural production.
- Potentially associated with large structural change of farm.
- Increase in methane emissions if associated with more cattle.

2.2.3. Measures for permanent grassland and alpine grassland

The effect of grassland management on SOC storage has generally been less studied compared with practices on croplands. This is an issue particularly for Switzerland with its large grassland share (about 1 million ha grassland compared with 400,000 ha cropland). Hence, recommendations are more difficult to give, and more research is needed. In general, SOC stocks of permanent grasslands are sensitive to management. In a global review of 126 studies, *Conant et al. (2017)* highlighted that grassland fertilization and adopted grazing intensity are means to increase SOC stocks. Effects were visible down to 1 m soil depth. In a Swiss grassland experiment with mowing but no grazing in Oensingen, a site under extensive grassland management lost C as compared with more intensively managed soil that gained C (high fertilization and frequent cutting; *Leifeld et al. 2011*). This finding is in line with the findings of *Conant et al. (2017)* regarding fertilization effects. Most likely, this difference was mainly driven by OM inputs, which were higher in the intensive treatment (due to higher biomass and organic fertilization). In addition, SOM decomposition rates as measured by soil respiration in the extensive management of Oensingen were higher, which was interpreted as microbial nutrient mining (*Ammann et al. 2007*). Similar fertilization effects on SOC stocks were found for cut grasslands in an experiment that was carried out in Balsthal (*Keel et al. 2019*). For an experiment in Watt, again comparing different fertilization intensities, the results are not as clear. For both sites, Balsthal and Watt, uncertainties of the results were high, due to very low numbers of soil analyses and the experiments being not very representative, because mineral fertilizer was applied. However, results from seven long-term fertilization experiments on meadows in Europe confirmed that fertilization increased SOC stocks (*Poeplau et al. 2018*). A higher microbial carbon use efficiency rather than higher C input was suggested as the most likely explanation for SOC increases.

Results for grazing are particularly variable, and thus possible recommendations are difficult to deduce. Light grazing might be preferable over heavier grazing (*Jiang et al. 2020*), and light grazing was also shown to increase SOC relative to grazing exclosure in Canada (*Hewins et al. 2018*). With increasing stocking density, soils may tend to lose SOC as compared with lighter grazing (e.g. *Mestdagh et al. 2009; Zhou et al. 2017*).

As shown in section 1.2.2, grassland soils generally have higher SOC stocks as compared with arable land. Grassland management should thus focus not only on C sequestration but also on preventing SOC losses. To identify most-vulnerable areas with particularly high SOC stocks, gaining more soil information is crucial. Compared with cropland, grassland covers the whole range of altitudes and the effect of climate change might thus be more variable.

Because grassland management measures, such as amount and type of fertilizers, species composition, and stocking type and density, are highly site and context specific, it is not possible at the moment to draw generalized conclusions for possible sequestration measures on permanent grasslands in Switzerland, but studies cited above indicate that grassland SOC stocks are highly management dependent. To identify meaningful measures, systematic studies on management effects of typical management practices in the various grassland types of Switzerland on SOC storage need to be carried out.

<u>Potential calculation for this measure:</u>

- SOC per hectare: not enough Swiss-specific studies available.
- For the surface: depends on whether measures can be applied on pastures or meadows, on extensively, less intensively or intensively used grasslands, on year-round pastures or summer pastures.
- Idea for time needed: meaningful measures need to be identified.

<u>Pros, co-benefits:</u>

- Large area.
- Possible GHG co-benefit with reduced stocking rates.

<u>Cons, risks:</u>

- N fertilization reduces species richness.
- Higher N fertilization rates lead to higher N_2O and ammonia emissions and nitrate losses.
- Systematic evaluation of management effects on grassland SOC stocks in Switzerland is still outstanding.

2.2.4. Summary of measures on agricultural mineral soils

- Among all soil types, agricultural mineral soils have the largest potential for soil carbon sequestration thanks to a variety of different measures (Table 8), the large area they occupy and – especially in the case of cropland soils – their carbon-depleted state.
- In this chapter, we discussed true soil carbon sequestration measures, which lead to a net uptake of atmospheric CO_2 on the same land unit where it is stored (e.g. cover crops), as well as measures that enhance soil carbon storage but do not comply with the definition of soil carbon sequestration due to e.g. lateral transport or import of carbon (e.g. organic amendments). It is important to note that organic amendments can be an integral part of many measures discussed here (CA, organic agriculture, mixed farming, agroforestry). Only if these organic amendments are produced on-farm without any import of e.g. feed, the measure would count as a true sequestration measure.
- Increasing carbon storage in cropland soil through improved agricultural practices has many important co-benefits in terms of soil quality and improving crucial soil functions such as fertility and environmental protection. Additionally, they improve the resilience of agricultural systems to climate change.
- The measures to increase carbon storage through different agricultural practices are well studied and are ready to be implemented (with the exception of agroforestry and deep tillage). In particular, the different pillars of CA are highlighted as promising measures to increase carbon storage.
- For specific regions with very low carbon stocks, measures to increase carbon storage through improvements in agricultural soil management have the potential to reach the targeted SOC:clay ratio of 0.10 at rates potentially exceeding the ‘4 per mille’ rate suggested by the French initiative. The ratio can be used as an indicator for identifying fields or areas where measures shall be taken. A case study from Bavaria, a region very similar to Switzerland, suggests that potentials are lower. More precise and realistic estimates on soil carbon sequestration depend on a good soil map and detailed information on the current management. This information is currently lacking at the national scale (see section 1.3.5).
- Well-managed soils in terms of their current SOC stock are not likely to contribute much to increasing carbon storage, whereas soils with low carbon concentrations have more potential.
- This is an important aspect for possible support programmes. Farmers who already apply practices favouring carbon storage should be further promoted, while efficient support should take place to motivate farmers who do not yet apply a soil management practice improving carbon storage.
- In the context of carbon accounting, the greatest disadvantage of most soil carbon sequestration measures is that carbon is not permanently stored. Soil carbon stocks can decrease rapidly if measures to maintain or increase stocks are given up. Losses might also occur in response to climate change. Biochar and deep tillage would have a clear advantage in this regard, because either carbon is added in a very stable form (biochar) or carbon is added to the subsoil where decomposition rates are very low. However, this advantage of deep tillage implies a potentially very high cost in terms of losses in soil quality and fertility, and this measure is not ready for

implementation and could only be applied on a small area if at all. Taken together, deep tillage needs sound evaluation before eventual application in the field.

- It is important to note that many measures could imply reductions in crop yields or produced calories (agroforestry, land use change, deep tillage) and their application needs to be carefully planned.
- Permanent grasslands cover about 70% of the agricultural mineral soil area in Switzerland. In general, grassland soils are less carbon depleted than arable soils. Still, carbon storage in those soils could be increased, but there is a knowledge gap in effective measures. Because grasslands cover large ranges in climate conditions and are managed very differently, measures need to be site specific.
- For almost all measures, little or no data are available regarding management effects on subsoils below 30 cm. Because carbon sequestration as a measure to counteract climate change is related to total soil carbon storage and not just changes in topsoil contents, any recommendation that is based on only topsoil measurements shall be treated with caution.

Table 8: Literature-derived mean carbon (C) sequestration rates of different agricultural management (mean ± standard deviation). Most numbers are from Wiesmeier et al. (2020).

Measure	Soil C sequestration rate (t C ha^{-1} yr^{-1})	Area (ha)	Total soil C sequestration rate (Mt CO_2-equivalents yr^{-1})	Soil depth (cm)	Data source
Conservation agriculture	0.63 0.4–0.8	356580	0.52–1.05	0–30 0–20	*Autret et al. (2016)*; section 2.2.2.1
No-tillage as single factor	No change	356580	0	>30 0–40	*Baker et al. (2007); Luo et al. (2010)*
Cover crops as single factor	0.32 ± 0.08	158480	0.19	21 ± 7	*Poeplau & Don (2015)*
Agroforestry	0–0.86	39620	0–0.12	51	*Upson & Burgess (2013); Cardinael et al. (2017); Seitz et al. (2017)*
Land use change (cropland to grassland conversion)	0.73 ± 0.17	19810	0.05	29 ± 5	*Conant et al. (2001); Poeplau et al. (2011); Lugato et al. (2014)*
Deep tillage	0.93[a] 8.95[b]	19810	(0.07–0.65)[c]	50–90 0–150	*Alcántara et al. (2016); Schiedung et al. (2019)*

[a]After an average of 45 years, SOC stocks in deep-ploughed plots were by 42 t C ha^{-1} higher than in reference plots. To estimate the annual change rate, this difference was divided by 45 years.
[b]Total SOC stocks (0–150 cm) increased by 179 ± 40 t C ha^{-1} over 20 years following flipping. The annual change rate was estimated by dividing 179 t C ha^{-1} by 20 years.
[c]Further research is needed before this measure can be implemented.

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2.3. Forest soils: measures to improve soil carbon balance

By Frank Hagedorn and Stephan Zimmermann

2.3.1. Introduction

Forest management optimizes ecosystem services of forests, including timber production, habitat for biodiversity, recreation, natural hazard protection, and C sequestration (e.g. *Schulze et al. 2021*). Management practices can influence soil organic carbon (SOC) storage by i) altering the quantity and quality of C inputs in response to the selection of tree species and rotation times, ii) changing microclimatic conditions through modifying the light and water regime, and iii) physically disturbing soils during harvesting (*Jandl et al. 2007; Mayer et al. 2020*). During a rotation period, SOC stocks are assumed to decrease after harvest by the combined effect of reduced C-inputs and more favourable microclimate (Figure 23; *Jandl et al. 2007*). Thereafter, SOC stocks are increasing through an increasing C input with low quality and a colder microclimate. Management effects are most pronounced in the forest floor, which almost entirely consists of soil organic matter (>20% C), has the highest turnover rate in forest soils and comprises about 17% of the total SOC stock in Swiss forest soils. The quantitative knowledge of SOC dynamics is still poor because historic samples are lacking (*Mayer et al. 2020*). Uncertainty is particularly high regarding the mineral soil, which is already close to saturation and where C stock changes are small as compared with the existing stocks. Here, we focus on the most important forest management practices for Swiss forests (Table 9).

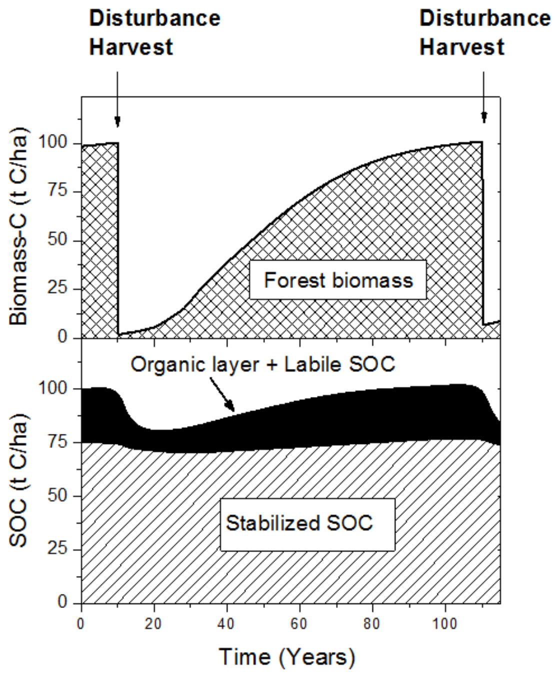

Figure 23: Soil organic carbon (SOC) stocks along a forest rotation show slowly increasing SOC stocks with stand development, very likely through increasing inputs of low-quality litter and a cooling microclimate. Following harvest, SOC stocks are rapidly declining by carbon (C) inputs, physical soil disturbance and a more favourable microclimate. A similar pattern can be assumed under natural disturbances (e.g. windthrow).

2.3.2. Measures

2.3.2.1. Afforestation

Principle: Afforestation of agricultural land is suggested as a measure for mitigation of climate change (*Bastin et al. 2019*) through on-site C sequestration in biomass and soil. Whereas the contribution of new forest biomass to C sequestration is well understood and modelled, C sequestration in soil is more complex. Afforestation alters various processes, including a change in both the quantity and quality of above- and belowground litter as well as a colder and drier microclimate under tree canopies (*Hiltbrunner et al. 2013; Mayer et al. 2020*).

Effects: Globally, reported afforestation effects show increases on former cropland but negligible and even negative effects on former grassland, as it is primarily the case in Switzerland (meta-analysis: *Poeplau et al. 2011; Barcena et al. 2014*). Whereas the forest floor increases with afforestation, soil C storage was found to decrease in the mineral soil and the soil C fractions associated with minerals (*Poeplau & Don 2013; Mobley et al. 2015*). In organic soils, entailing a lowering of the water table through drainage and site preparation leads to C losses from the peat layer, at least for the first decades (Finland: *Simola et al. 2012*; UK: *Vanguleova et al. 2019*). In contrast, afforestation on C-poor cropland has been observed to increase SOC stocks, but at a lower rate than C losses following forest clearing for cropland (*Poeplau et al. 2011*).

An extensive study in Chinese forests clearly shows that afforestation effects depend on SOC stocks of the former land (*Hong et al. 2020*). At low SOC stocks, planting forests leads to C gains in the soil, whereas at high SOC stocks (>100 t C ha^{-1}) such as in grasslands and organic soils, afforestation causes SOC losses.

Switzerland: In agreement with these meta-analyses, afforestation of subalpine pasture at Jaun Pass showed a continuous C accumulation in the forest floor with afforestation but transient C losses in the mineral soil (Figure 24; *Hiltbrunner et al. 2013*). This pattern is supported by the data analysis of historic forest cover for 850 forested soil profiles, showing that SOC stocks decrease slightly with increasing forest cover ages (*Gosheva et al. 2017*). As most of the Swiss forests have been planted on former grasslands, these findings indicate that the continuous forest expansion is unlikely to increase SOC storage. Also for peatlands, the case study by *Bader et al. (2018)* did not observe significant differences in SOC stocks between forests, grasslands and croplands. The overall conclusion on afforestation effects on SOC storage is limited by the lack of data on SOC stocks down to the bedrock in grasslands and the limited number of paired sites differing in land use.

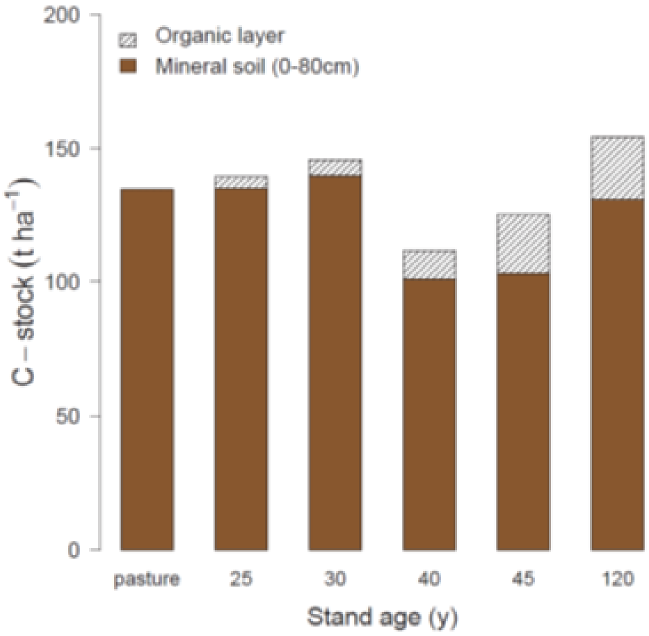

Figure 24: Afforestation on former subalpine grasslands at Jaun Pass leads to a temporary decline in SOC stocks (from Hiltbrunner et al. 2013).

2.3.2.2. Promoting tree species composition

Planting tree species is an active measure to enhance forest productivity, biodiversity and soil fertility. Promoting broadleaf species with a greater rooting depth increases the uptake of base cations from subsoils and stimulates soil biological activity (e.g. *Reich et al. 2005; Berger et al. 2006*). In particular, earthworms are profiting from calcium-rich litter of broadleaf species (*De Wandeler et al. 2018*). As earthworms are transferring litter C into the mineral soil, broadleaf trees species are associated with smaller C stocks in the forest floor but greater C stocks in the mineral soil and, thus, a sustainable C sequestration (*Jandl et al. 2007; Mayer et al. 2020*). In addition, broadleaf trees may provide additional C inputs into mineral soils by their deeper rooting system and their association with arbuscular mycorrhiza (*Craig et al. 2018*).

Common garden experiments indeed observed pronounced tree species effects in the forest floors with greater C stocks under coniferous trees due to a reduced litter decomposition as compared with high-quality litter from broadleaf trees (*Reich et al. 2005; Vesterdal et al. 2013*). Also, in Swiss forests, SOC stocks in the forest floor are greater under coniferous than under broadleaf trees (Figure 25; *Gosheva 2017*). The opposite pattern exists for the mineral soils, where SOC stocks at a given elevation are higher under broadleaf trees. However, total SOC stocks (forest floor + mineral soil) do not differ between the two forest types.

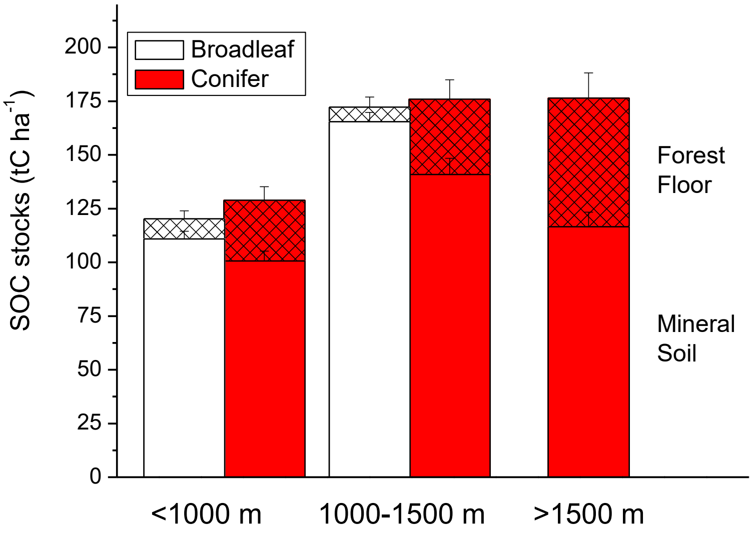

Figure 25: Smaller soil organic carbon (SOC) stocks in the forest floor but greater stocks in the mineral soil in broadleaf compared with coniferous forests of Switzerland. Total SOC stocks do not differ between forest types but depend on elevation (based on approximately 1000 soil profiles).

Swiss forests already have a great share of broadleaf forest in their adequate growth region (e.g. 68% broadleaf forest in the Swiss Plateau), which limits the potential to promote broadleaf trees for increasing SOC storage in mineral soils. In response to the ongoing climatic changes, tree species composition has to be adapted with potential impacts on SOC storage. Because spruce responds very sensitively to drought, promotion of spruce is inappropriate in a future climate. Silver fir and oak, on the contrary, form deeper rooting systems (*Vitasse et al. 2019*), which potentially provide a greater C input into the mineral soil (*Reich et al. 2005*). Planting Douglas fir as another drought-resistant conifer that originates from North America may conflict with the paradigm of forests composed of native tree species.

Overall, these findings indicate that promotion of tree species (e.g. broadleaf tree species) can have positive effects on soil biodiversity and fertility. Although replacing coniferous by broadleaf trees leads to C losses in the forest floor, long-term soil C storage is increased through greater rooting depths and incorporation of litter into mineral soils by an enhanced faunal activity. However, overall tree species effects on total SOC stocks seem small, and in Switzerland, the effect size of promoting tree species is limited by the fact that Swiss forests already have a high contribution of broadleaf trees.

2.3.2.3. Intensification of forestry

During the last decades, forest harvest was smaller than regrowth in Switzerland (*Brändli et al. 2020*). Nonetheless, it could be that forestry will be intensified in the near future, for instance to increase the use of Swiss wood products or in response to increasing wood prices.

Management practices generally induce SOC losses due to soil disturbance during harvest, removing biomass, reducing litter inputs into soils and by creating a more favourable microclimate for decomposition (Table 9; *Mayer et al. 2020*). Chronosequence studies and meta-analyses suggest that soil C stocks in the forest floor and mineral soil strongly decrease after harvesting and start to recover during one to five decades following harvest (*Achat et al. 2015; James & Harrison 2016; Mayer et al. 2020*). In a meta-analysis, forest harvesting and whole tree thinning was found to reduce total soil C stocks by an average of 11% with greatest losses in organic horizons (−24% to −30%; *James & Harrison 2016; Clarke et al. 2021*). However, SOC stocks are recovering from these disturbances, and overall, soils of managed and unmanaged forests show similar C stocks (soil survey in Germany; *Schulze et al. 2021*).

Removal of harvest residues for a better biomass exploitation was observed to lead to significant losses of soil C stocks in the forest floor (10–45%) and even in deeper soil layers >20 cm belowground (–10%) (*Achat et al. 2015*). Removal of harvest residues also leads to a nutrient depletion of soils.

Switzerland. So far, impacts of management intensities and residue removal on soil C have not been studied in Switzerland. However, windthrow (with a presumably similar effect to clear-cutting in a more intense management regime) was found to induce a strong SOC loss in the forest floor (*Thürig et al. 2013*). The reduction in SOC stocks was particularly strong (−25 t C ha^{-1}) in high-elevation soils with thick organic layers, whereas the effects were small and short-lived in the Swiss Plateau.

In addition to the expected SOC losses, an intensification of forestry would lead to nitrate leaching and would have negative consequences for biodiversity by removing dead wood.

2.3.2.4. Fertilization

Fertilization is not allowed in Swiss forests, but it represents a potential measure to overcome nutrient deficiency or to promote forest productivity, e.g. to increase wood production. Potential impacts have most intensively been studied for nitrogen, which was found to increase total soil C stocks (combined forest floor and mineral soil) by 7.7% (review by *Nave et al. 2009* on N fertilization and N deposition effects). The mechanisms involved include increased litter input and reduced decomposition (*Mayer et al. 2020*). However, Swiss forests are already receiving high N loads via deposition exceeding critical limits (e.g. *Braun et al. 2017*). The potential short-term and probably limited benefits of N fertilization for increasing tree growth and soil C stocks must be weighed against the associated environmental costs, such as the production, transport and application of synthetic fertilizers all entailing fossil fuel combustion and emission of carbon dioxide (CO_2).

Table 9: Overview of forest management practices, their potential effect on soil organic carbon (SOC) stocks and additional consequences.

Mangement practicies	Mechanism	Effects	Uncertainty	Data for Switzerland	Additional consequences	Key references
1) Afforestation						
a) on former cropland	greater C input by trees, less soil disturbance	positive	low	no data	no agricultural yield	Poeplau et al. (2011), Barcena et al. (2014), Hong et al. (2020)
b) on former grassland	smaller C-input by tree roots, greater C-input with a lower quality from aboveground, colder and drier microclimate	low (positive in forest floor, temporary negative in mineral soil)	low	1 case study + historic forest ages	no agricultural yield	Hiltbrunner et al. (2013); Gosheva et al. (2017); Poeplau et al. (2011), Barcena et al. (2014), Hong et al. (2020)
c) on organic soil	Lowering of water table for planting leads to peat oxidation	strongly negative	low	Single site case study	no agricultural yield, N and P leaching	Mayer et al. (2020); Vanguelova et al. (2019); Bader et al., (2018)
2) Intensified forest management and harvest	Soil mechanical disturbance, lower C-input by biomass removal, more favorable microclimate	negative (at least SOC loss from forest floor)	low	no data	Greater timber production, N and P leaching, loss in biodiversity (less deadwood)	Nave et al. (2010); James & Harrison (2016)
3) Removal of harvesting residues (as a renewable energy source)	Export of biomass C reduces soil C inputs	negative	low	no data	Soil nutrient depletion, altered biodiversity (less deadwood)	Achat et al. (2015); Mayer et al. (2020);
4) Tree species selection	Coniferous species store more C in the forest floor; broadleaved species may store more stabilized soil C in soil mineral soil.	low	low	Higher SOC stocks under conifer trees (+ 28 tC/ha), but estimates are biased by other factors	Site specific suitability of tree species. Climate change will alter species composition.	Gosheva (2017); Barcena et al. (2014)
5) Promoting tree diversity	Niche and ressource complementary enhances C inputs	tends to be positive	high	no data (frequent practice in Switzerland)	Higher biodiversity, greater resilience towards climate change	Gamfeldt et al. (2013)
6) Increase in stand density	Increased C input in denser stands	tends to be positive	high	no data	Suppression oft understory plants	Mayer et al. (2020)
7) Fertilization (not allowed in Swiss forests)	Increased C input by growth enhancement, suppressed decomposition	positive	medium	No data	Acidification, higher nitrate leaching, biodiversity changes	Nave et al. (2009); Synthesis by Mayer et al. (2020)
8) Liming (not allowed in Swiss forests)	Increased C input by growth enhancement, deeper rooting, but increased decomposition by greater C solubility and stimulated biological activity. Leads to losses of forest floor, effects on mineral soil uncertain	variable (losses in forest floor, gains in mineral soils)	high	no data	altered biodiversity, increases nitrate leaching and P deficiency	Matzner et al. (1985); Kreuzer (1995); Persson et al. (1995); Bauhus et al. (2004)
9) Wood ash application (not allowed in Swiss forests)	Similar as for liming	Similar as for liming	high	Field experiment (not focussing on SOC)	Input of contaminants, nitrate leaching, altered biodiversity	Zimmermann & Frey (2002)

2.3.2.5. Liming and wood ash application

Additions of substances and fertilization of forests are prohibited in Switzerland. Currently, liming is explored as a measure to reverse negative effects of soil acidification mainly due to N deposition. Because lime is listed as a measure to improve soils (‘Bodenverbesserungsmittel’) and not as fertilizer, trials are allowed (see: *Schweizer Eidgenossenschaft, Postulat von Siebenthal 2017*). Adding lime to soils leads to the release of CO_2 from carbonate in the lime (~12% of its mass). This corresponds to 0.036 t C ha^{-1} yr^{-1} at a standard addition of 3 t lime ha^{-1} every 10 years, which is small as compared with C sequestered in increasing forest biomass stocks (~0.4 t C ha^{-1} yr^{-1}; *FOEN 2020*).

The original aim of liming in the 1980s was to reverse acidification and to improve soil fertility (*Hildebrand 1996*) and not to increase SOC storage. However, liming – leading to a more 'active' humus form – also affects a number of soil processes, and thus it may also impact the soil C cycle (*Puhlmann et al. 2021*). In principle, liming influences SOC storage by potentially increasing C inputs into soils through an improved tree growth and a deeper rooting system. However, it also accelerates decomposition by enhancing C solubility and stimulating biological activity associated with an increased soil pH. Liming enhances the abundance and activity of earthworms, which leads to a translocation of litter-derived C into the mineral soil and hence to a C loss in the forest floor but a gain in the mineral soil.

Table 10: Effect of liming on soil organic carbon storage in forest soils. Please note the different units in the last row.

		Mode of liming	Years after liming	Forest floor	Forest floor	Mineral soil (0–20 cm)	Mineral soil (0–20 cm)	Total soil	Comments & limitations
			n	tC ha^{-1}	change rate	tC ha^{-1}	change rate	tC $ha^{-1} yr^{-1}$	
Experiments									
Bauhus et al. (2004)	Beech (Solling) Stand	Dolomite (3 t ha^{-1})	8	−0.5	−3%	−14.4	−22%	−1.86	
	Beech (Solling) Gap	Dolomite (3 t ha^{-1})	8	−9.0	−60%	20.6	38%	1.45	
Kreutzer (1995)	Spruce (Höglwald)	Dolomite (4 t ha^{-1})	7	−7.2	−23%	0.8	4%	−0.91	0–5 cm depth; no data on deeper soil
Court et al. (2015)	Beech (Northern France)	Calcite (2.5 t ha^{-1})	20–40	−2.1	−0.1%	n. sign.	n. sign.		average of 5 sites
Marschner & Wilczynski (1991)	Spruce (Berlin)		3	−6.9	−23%	−2.9	−5%	−3.27	heterogenous sandy site, liming was combined with potassium fertilization
Matzner et al. (1985)	Spruce (Solling)	Calcite (5 t ha^{-1})	10	−7.1	−15%	−12.5	−19%	−1.96	combined with N fertilizer
	Beech (Solling)	Calcite (5 t ha^{-1})	10	4.4	18%	20.6	37%	2.50	combined with N fertilizer
Persson et al. (1995) (values 0–50 cm)	Spruce (Sweden)	Calcite (9 t ha^{-1})	42	−6.0	−40%	3.0	4%	−0.07	includes a stand rotation
	Spruce (Sweden)	Calcite (12 t ha^{-1})	37	−10.3	−30%	−10.7	−12%	−0.57	planted former heathland
	Spruce (Sweden)	Calcite (10 t ha^{-1})	38	−7.5	−94%	−8.0	−8%	−0.41	includes a stand rotation
	Beech (Sweden)	Calcite (10 t ha^{-1})	38	−20.0	−80%	0.0	0%	−0.53	
Soil Survey									large data set
Grüneberg et al. (2017)	German forests	mostly 3 t ha^{-1}	variable	−0.25 t $ha^{-1} yr^{-1}$		0.43 t $ha^{-1} yr^{-1}$		0.18 t $ha^{-1} yr^{-1}$	comparability of sites uncertain

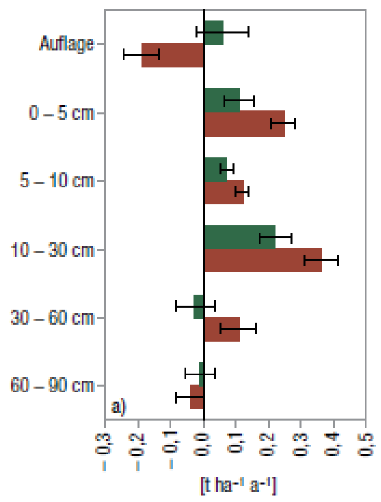

Figure 26: Effects of liming on soil organic carbon (SOC) storage in German forests illustrating a transfer of C from the forest floor ('Auflage') into the mineral soil. The annual rates of change in SOC stocks are based on the repeated soil survey in Germany and compare acidic forest soils which were limed (brown) with unlimed acidic soils (green; from Grüneberg et al. 2017).

Soil surveys and liming experiments including observations of humus forms in up to 40-year-long liming experiments show that liming effectively transforms thick forest floors into more 'active' humus forms (*Immer et al. 1993; Court et al. 2018*). This transformation leads to consistent SOC losses through liming in the forest floor (Table 10). Mechanisms include a stimulated decomposition and mineralization of higher-quality litter at higher pH values (*Kreutzer 1995*) or enhanced C leaching (e.g. *Feger et al. 2000; Puhlmann et al. 2021*). There is also evidence for a deeper rooting system and for an enhanced activity of soil fauna, in particular of earthworms (*Schäffer et al. 2001; Puhlmann et al. 2021*) which are translocating and incorporating litter into the mineral soil. Potentially, a SOC transfer from the forest floor to the mineral soil by liming increases SOC storage (*Bauhus et al. 2004*). However, observed effects on SOC stocks in the mineral soil under the forest floor are inconsistent (Table 10), which could be attributed to the stimulated SOC mineralization by liming, the C stocks in the mineral soil being already close to saturation or the difficulty in detecting changes in the large C reservoir in mineral soils. Although liming experiments tended to result in negative effects on total SOC stocks, SOC changes between two repeats of soil surveys in German forests showed slightly positive liming effects (Table 10). The repeated measurement at 385 limed sites revealed increased SOC stocks in the mineral soils as compared with unlimed but similarly acidic sites (Figure 26). This increase surpassed C losses in the forest floor, leading to an overall increase of 0.18 t C ha^{-1} yr^{-1}. The authors explained their finding by potentially deeper-rooting trees and stabilization of soil organic matter by calcium cations added with the lime.

Taken together, liming effects in the forest floor associated with a transformation towards active humus forms lead to a smaller SOC stock in the forest floor, which is at least partly outbalanced by C gains in the mineral soil (Table 10). Potentially, this leads to a long-term stabilization of soil C but the overall effect is uncertain and seems small due to higher C mineralization rates in limed soils and a limited capacity of C-rich forest soils to sequester additional C. The magnitude of responses appears to depend on site and soil conditions, with greater SOC losses to be expected in soils with a thick forest floor. In addition, effect size will depend on the amount (mostly ranging between 2 and 5 t ha^{-1} and the type of lime added. Most of the liming tests and measures were carried out with calcite ($CaCO_3$) or dolomite [$CaMg(CO_3)_2$] in ground or semi-burned form. Calcite as a more soluble mineral leads to faster responses than dolomite.

Wood ash application aims at returning base cations removed by harvesting back to the soil. Mechanistically, impacts of wood ash on soil processes are similar to those of lime, with an increased soil pH and an improved supply with base cations (review by *Huotari et al. 2015*). However, although several studies have assessed the effect of wood ash on soil fertility and functioning, '*net effects on soil carbon balance have not been thoroughly evaluated'* (see *Huotari et al. 2015; H. Puhlmann, P. Hartmann, FVA Freiburg, personal communication 2021*). In agreement with liming studies, *Rosenberg et al. (2010)* found accelerated C and N mineralization rates following wood ash application. Also, in a field experiment in the Swiss Plateau at Unterehrendingen (CH), *Zimmermann & Frey (2002)* observed long-lasting increases in soil CO_2 effluxes and a decrease in SOC contents in the uppermost 5 cm by approximately 20% during the first three years after wood ash application. Whether the C losses in the forest floor and upper mineral soil are compensated by C gains in the deeper mineral soil through a C transfer mediated by soil fauna or by the development of a deeper rooting system has not been analysed. However, fine root biomass remained unaffected by wood ash additions in the Swiss experiment (*Genenger et al. 2003*). Earthworms showed a decreasing abundance in the first treatment year but similar abundances in the following year (*Hallenbarter 2002*). The abundance of other species (springtails, mites and spiders) decreased during the entire study period.

Additional impacts of liming and wood ash. In principle, the enhanced SOC mineralization in the forest floor and upper mineral soil by a rise in soil pH is associated with an increased N mineralization, an enhanced nitrification (transformation of ammonium cations to nitrate anions) and as a consequence nitrate leaching. Although most liming and wood ash experiments have indeed observed higher nitrate concentration in soil solution in the years following the addition (*Matzner et al. 1983; Kreutzer 1995; Feger et al. 2000; Puhlmann et al. 2021*), there are also reports of unaffected nitrate concentrations in spring waters in Saxonian forests following standard liming of forests (*Franz 2004*). The loss of forest floor material by liming may also induce phosphorus deficiency in phosphorus-poor acidic soils with reactive surfaces in the mineral soil (where the forest floor plays a central role in phosphorus nutrition (*Lang et al. 2017; Brödlin et al. 2019; Puhlmann et al. 2021*). However, liming might also improve phosphorus supply by increasing the biological activity. Some negative effects can be reinforced by the common practice of applying lime directly after harvesting as a preparatory measure for the next tree generation. Logging leads to higher temperatures and reduces the uptake of nutrients by tree roots, which further promotes nitrate leaching and aluminium mobilization and as a consequence pollutes groundwaters (*Neal et al. 1998*). These consequences apply not only to liming after clear-cutting but also to liming after large shelterwood cuttings.

The negative liming impacts can be minimized by a specific management of forest stands. In Solling, for instance, it was shown that nitrate and aluminium concentrations were greater in stand gaps than under tree canopies (*Bauhus & Barthel 1995; Bauhus & Bartsch 1995;*

Brumme 1995; Bauhus et al. 2004). However, when the gaps were limed with 3 t dolomite ha^{-1}, nitrate concentration in the leachate decreased due to the rapid development of a dense herb layer assimilating the mineralized nutrients. This observation leads to the recommendation that liming to improve soil acidification shall be carried out when the canopy has been thinned out slightly, so that light conditions allow for the establishment of a dense herb layer. Moreover, changes in micrometeorological conditions are still minimal, and root functions of the trees do not completely fail (*Godbold 2003*). Liming should therefore be carried out on small areas after shelterwood cutting interventions.

While liming reverses soil acidification, it alters soil biodiversity because soil pH and nitrogen availability are key factors for the abundance and activity of soil organisms (*Nicol et al. 2008; Cho et al. 2016; Wang et al. 2019*). Some of these changes such as a promotion of earthworm abundance are positive (*Puhlmann et al. 2021*), due to the incorporation of organic matter into the mineral soil, deepening the rooting zone, and enhancing water infiltration and the water holding capacity of soils. Mycorrhizal fungi fulfilling key ecosystem functions show distinct community compositions at varying pH values and N status (*De Witte et al. 2017; van der Linde et al. 2018*), which are both influenced by liming. In Swiss forests, diversity and productivity of mycorrhiza were found to decrease with increasing nitrogen deposition. In agreement, the production of mycorrhizal fungi in soils of Swedish coniferous forests has been found to decrease with increasing nitrogen availability and increasing soil pH (*Högberg et al. 2003; Nilsson et al. 2005*), suggesting similar effects for liming.

By analysing universal primers for soil microorganisms, several liming studies revealed an altered structure of the microbial population (*Clivot et al. 2012; Ragot et al. 2013*). The taxonomic diversity is generally lower in limed soils, especially of acidobacteria and gram-positive bacteria, whereas the diversity of proteobacteria increases. Ammonium-oxidizing bacteria are also among the proteobacteria promoted by liming (e.g. *Carnol et al. 2002; Bäckman et al. 2003, 2004; Gray et al. 2003; Hermansson et al. 2004*). The diversity of ammonium-oxidizing bacteria was found to increase with the dose of liming (limestone:dolomite = 1:1; 6 t ha^{-1} compared with 3 t ha^{-1}; *Hermansson et al. 2004*). The enhanced abundance and diversity of ammonium-oxidizing bacteria is in agreement with increased nitrification rates and nitrate leaching in several studies (*Kreutzer 1995; Neal et al. 1998*).

Numerous studies reported a sensitive response of ectomycorrhizal fungi to liming (reviewed by *Kjøller & Clemmensen 2008*). Analysing fungal rDNA, *Jonsson et al. (1999)* found a change in fungal community after dolomite application of 8.8 t ha^{-1}. Also, in Swedish forests, liming with 6 t calcite ha^{-1} led to a change in the ectomycorrhizal population, with increased ammonium uptake via mycorrhizal root tips (*Wallander et al. 1997*). In the Vosges Mountains, liming decreased the number of mycorrhizal root tips in the upper mineral soil horizons but increased it in the forest floor (*Rineau & Garbaye 2009; Rineau et al. 2010*). The review by *Kjøller & Clemmensen (2008)* based on Scandinavian studies indicates that liming did not change species richness but caused species displacements or replacements. In principle, the abundance of acidophilic fungal species is decreasing and replaced in part by universally occurring fungal species. *Kjøller & Clemmensen (2008)* concluded that liming is very unlikely to alter functions of mycorrhiza in forest ecosystem. However, to minimize the risk of species extinction, they recommended restricting the size of limed plots as much as possible and applying lime heterogeneously at the regional scale.

2.3.3. Summary of forest management effects

Forest soils in Switzerland have the highest SOC stocks in Europe and are close to C saturation as a result of a low management intensity and the inherently cold and humid climate. These high SOC stocks are at risk in a warmer climate with more frequent dry periods or when forests are more intensively managed (risk of reversibility of soil carbon sequestration).

Forest management practices such as selection of tree species can have positive effects on soil biodiversity and functioning. However, with respect to SOC storage, tree species effects are largely constrained to the forest floor and uppermost soil. Promotion of broadleaf trees has potentially positive effects on SOC stocks in the mineral soil, but the overall impacts on total SOC stocks appear to be small. Moreover, Swiss forests are already diverse with a high proportion of broadleaf trees.

Other practices (e.g. promotion of high tree diversity) are already common in Swiss forests, whereas intensification of forest management would probably lead to lower SOC stocks. Transferring reported SOC losses associated with intensified management (11%) to the forest area used for wood production (410,000 ha) would induce a SOC loss of 6.4 Mt C. This is 15 times greater than the current C sink in Swiss forests and corresponds to 48% of the annual greenhouse gas emissions in Switzerland (*FOEN 2020*). Intensified management would also reduce the current high biodiversity in Swiss forests due to their large reservoir of deadwood and high plant species diversity.

Afforestation of marginal land, which is already ongoing, appears at the first view as a strong measure to enhance C storage in Swiss ecosystems. However, with respect to C sequestration in the soil, the potential effect sizes are negligible when former grasslands are afforested or even negative on organic soils. Positive afforestation effects are confined to former cropland – a measure that conflicts with food production. While increasing C storage in biomass, an increase in forest cover may have a negative impact on global warming through changes in albedo, with canopies reflecting less radiation than open vegetation. For high-elevation sites, these albedo changes may offset CO_2 sequestration in growing forest biomass (*Schwaab et al. 2015*).

Liming and wood ash application to reverse soil acidification and to promote more favourable soil conditions with ‘active’ humus forms will accelerate the biological activity of soils, thereby inducing SOC losses from the forest floor. Potentially, part of this C is transferred to the mineral soil, where it could be stabilized. However, the quantitative evidence for increased SOC stocks in the mineral soil is elusive, and effects on total SOC stocks seem small. Liming is frequently accompanied by an initially enhanced nitrate leaching and by changes in soil biodiversity. Possible negative effects can be minimized by the application of lime on small areas only and following shelterwood cutting interventions and by application of lime with low solubility.

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2.4. Settlement soils: measures to improve soil carbon balance

By Beatrice Kulli and Andrea Saluz

2.4.1. Introduction

In settlement areas, a wide range of open surfaces can be found, from more or less naturally built and only moderately affected soils in parks and larger gardens to shallow, man-made surfaces in the vicinity of buildings and streets. The latter are particularly strongly influenced by anthropogenic activity (construction, transport, industry, etc.) without being cultivated by humans (agriculture, forestry, etc.) (*Lehmann 2006*). Today, there is no established internationally recognized method for settlement soil mapping in urban soil research (*Sauerwein et al. 2015*). However, a categorization of soils of urbanized areas was proposed by *Morel et al. (2015)*. Most of them belong to the Technosols and Anthrosols groups of the FAO-IUSS soil classification (*FAO 2014*). In some countries, including Germany, the USA, the UK and Russia, there are efforts to adapt national mapping instructions to urban soils (*Rossiter 2007*).

Artificially constructed settlement soils are mostly young soils and thus usually show only minimal soil development (*Lehmann 2006*). These are disturbed systems, which may consist partly or completely of foreign material (e.g. backfill). Therefore, these soils can be compared to a certain extent with alluvial soils, in which the soil material originates from the upper side of the river (*Amossé et al. 2015*).

Settlement surfaces can be modified in different ways according to *Sauerwein (2006)*:

- Sealed
- Spillage (technical substrate is poured onto the natural soil)
- Elevated (application of soil material, e.g. in gardens or parks)
- Dug up (excavated soil)
- Dried up (groundwater table lowering)
- Compacted (e.g. by machines, levelling, treading, etc.)
- Contaminated (contaminated sites, de-icing fluid, etc.)
- Mixed (soil cultivation)

Settlement soils are often fragmented like mosaics and show large differences depending on the degree of naturalness. Besides, they often contain a proportion of organic carbon that is not of geogenic origin but is technogenic (*Makki & Thestorf 2020*). These are called technogenic soil substrates (*Sauerwein 2006*), such as ashes, building rubble, waste, sludge, slag, etc. (*Hiller & Meuser 1998*). According to *Sauerwein (2006)*, the identification of these technical substrates often poses a major problem in mapping. It is difficult to make an accurate estimate of the technogenic proportion, because this substrate is often available in mixed forms. According to *Makowsky & Meuser (2007)*, the determination of the technogenic substrate is relevant because the soil properties depend on it. The total carbon (TC) in urban soils is composed of total inorganic carbon (TIC) and total organic carbon (TOC) originating from humification (TOC_{humic}) or from the incorporation of technogenic substrates (TOC_{tech}) as follows: $TC = TIC + TOC_{humic} + TOC_{tech}$. If no distinction is made between TOC_{humic} and TOC_{tech}, misinterpretations are made regarding the sorption properties of soils. TOC_{tech} has almost no influence on the sorption properties of settlement soils. However, it is estimated that 40% of all soil horizons in settlement soils are free of technogenic substrates. Less than 2% of all horizons consist of complete technical substrates (e.g. dumps) (*Sauerwein 2006*).

2.4.1.1. Area and composition of settlement soils

In Switzerland, as well, there is no systematic mapping of settlement soils. Within the framework of research projects and the cantonal and national soil monitoring, information on urban soils and their properties is available on a punctual basis. In 2013, the Soil Science Society of Switzerland named the Urban Soil the Soil of the Year 2013 (*Amossé et al. 2013*). Figure 27 shows examples of urban soils given by the authors in their information. However, because there are no soil maps of urban areas in Switzerland, we are depending on other data to assess the carbon storage and its potential in settlement soils.

We therefore used the data provided by the Federal Statistical Office (FSO) in the framework of their areal statistics. We are aware that the FSO land use and land cover data do not take account of the fragmented mosaic of settlement soils and that estimates of soil C concentration based on land cover are subject to considerable uncertainty. Nevertheless, we consider this approach to be the best possible as long as there is no comprehensive soil mapping in settlement areas.

According to *BAFU (2017)*, almost 40% of the settlement areas consist of open surfaces. Although many of these areas are strongly influenced by anthropogenic activities, they fulfil important functions. In addition to storing carbon, they allow the infiltration of water, which purifies the water and reduces the risk of flooding; furthermore, they enable the growth of plants, which has a positive effect on the urban climate and provides habitat for organisms and recreational space for people (*BAFU 2017*).

Figure 27: Examples of settlement soils (from Amossé et al., 2013). Deep natural soil on the left, strongly disturbed soil on the right.

Our analysis of the composition of the cover and use of settlement soils is based on the information provided by the FSO. As the results from the latest survey 2013/18 are not yet available for download for all Swiss regions, the data used for our analysis are taken from the survey carried out between 2004 and 2009.

2.4.1.2. Types of settlement soils in Switzerland

Settlements analysed here are not limited to the building zones. We also include roads and railway tracks with their surroundings, airports as well as supply or waste treatment plants. Based on this definition, 7.5% of the area of Switzerland is covered by settlement soils.

While 62% of these soils are covered with buildings or sealed surfaces, 38% of the soils are open according to the survey of land cover by the FSO. Figure 28 shows the main categories defined by the FSO areal statistics in settlement areas.

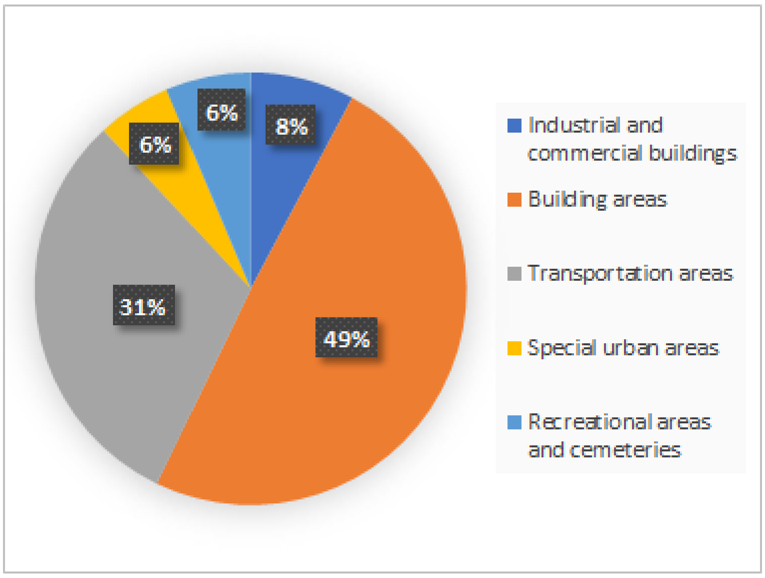

Figure 28: Basic categories of the areal statistics of the Federal Statistical Office in settlements.

About half of them are building areas, mostly used as residential areas but also as public building areas or agricultural building areas. About 8% of the settlement soils are used for industrial and commercial buildings and their surroundings.

Figure 29 shows the subcategories of the building areas and the areas with industrial and commercial buildings with their fractions of consolidated surfaces, buildings and open surfaces. The open surfaces include all unsealed surface types of the land coverage data, such as lawns, shrubs, grass and herb vegetation, etc. Due to their rare occurrence in settlement areas, greenhouses have been neglected in the following analysis. Areas with one- and two-family houses make up the largest part of the building areas, and they consist of the largest fraction of unsealed soil.

Transportation areas make up about a third of the settlement areas in Switzerland (Figure 28). They consist of motorways, roads and paths with their green environments but also of parking areas, sealed railway areas, airports and airfields with their green environments. As shown in Figure 30, roads and paths cover most of the transportation area, and all subcategories consist of a high fraction of sealed surfaces.

The special urban areas cover 6% of the Swiss settlement soils and sum up categories such as supply or waste treatment plants, dumps, quarries & mines but also construction sites or unexploited urban areas (Figure 31). The fraction of consolidated surfaces and buildings is much smaller in the latter four categories than in the different supply and waste treatment plants.

Another 6% of the Swiss settlement soils is used by recreational areas and cemeteries. They sum up categories such as public parks, sports facilities, golf courses, garden allotments and cemeteries. In all these categories, there is a high percentage of open surfaces (Figure 32).

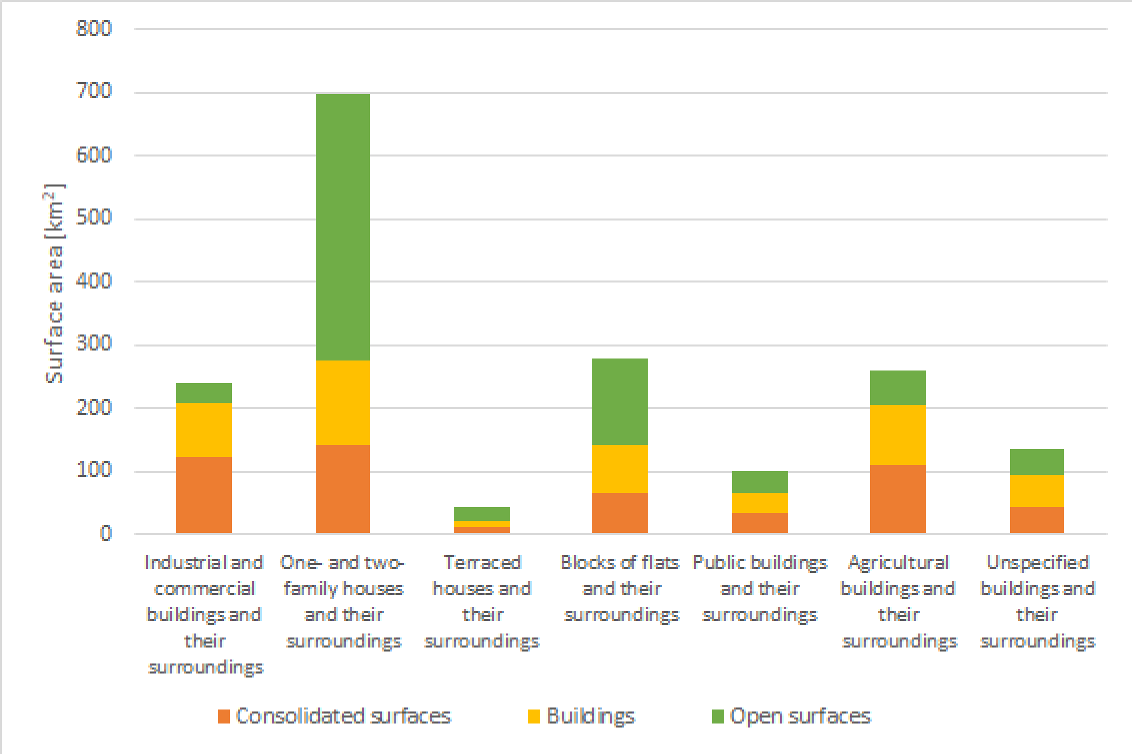

Figure 29: Categories of building areas and their fractions of consolidated surfaces, surfaces covered by buildings and open surfaces.

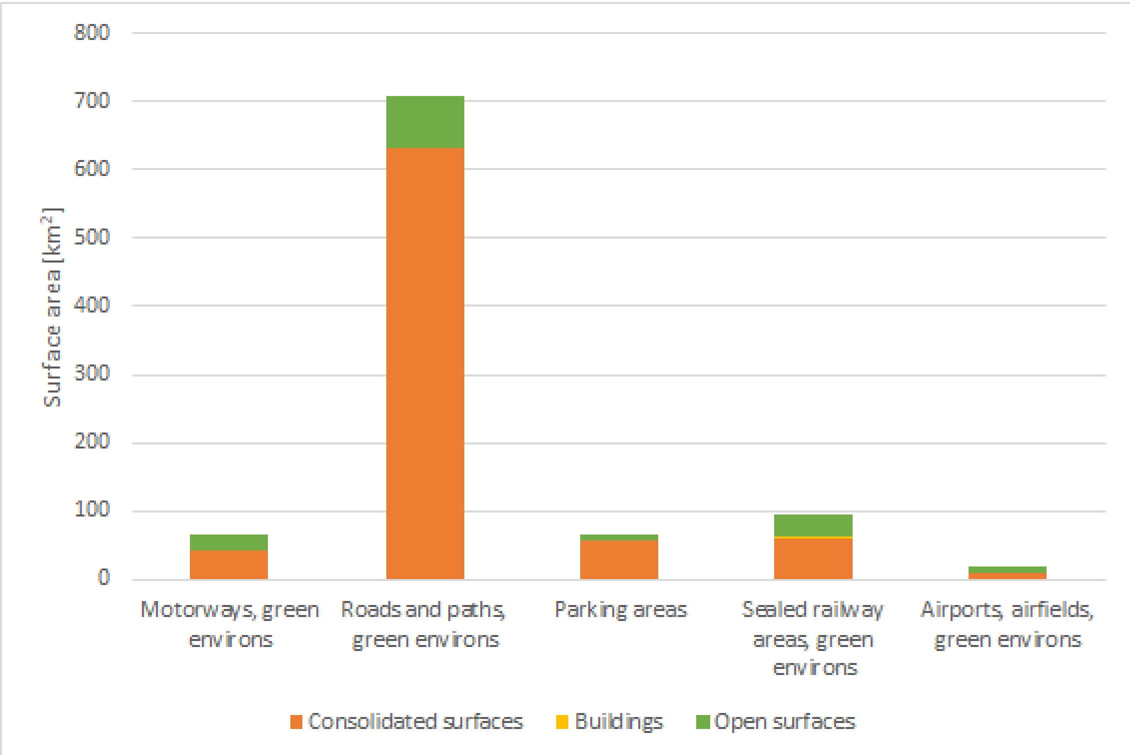

Figure 30: Categories of transportation areas and their fractions of consolidated surfaces, surfaces covered by buildings and open surfaces.

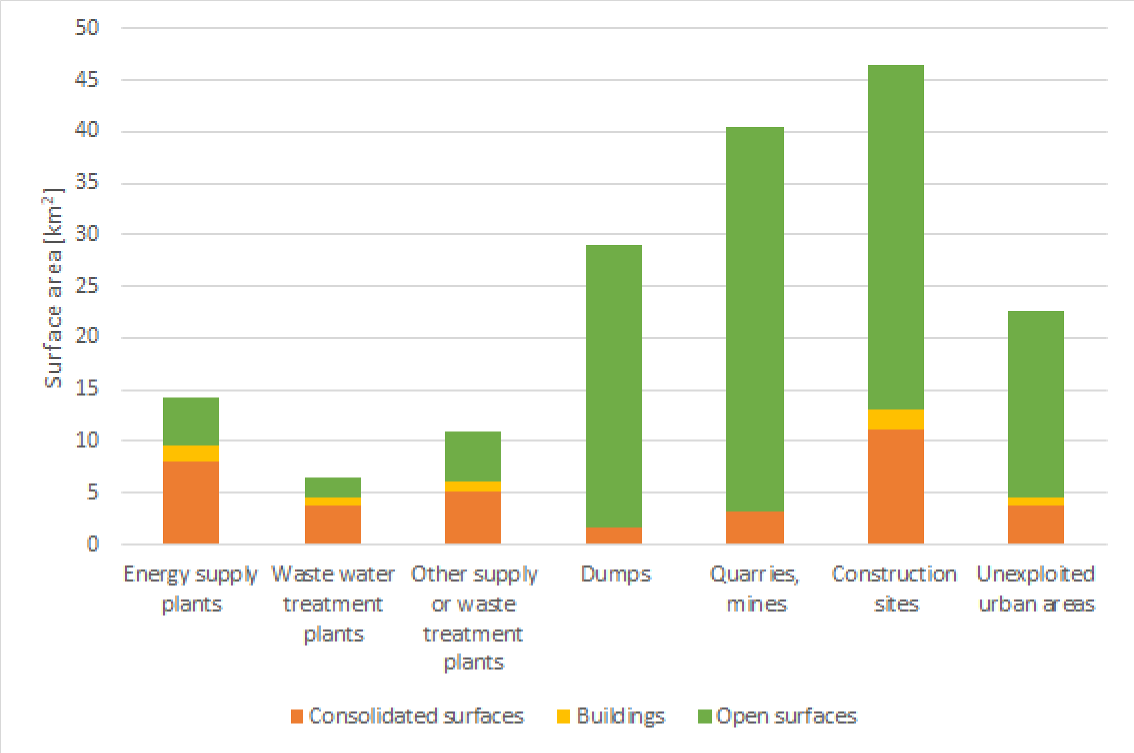

Figure 31: Categories of special urban areas and their fractions of consolidated surfaces, surfaces covered by buildings and open surfaces.

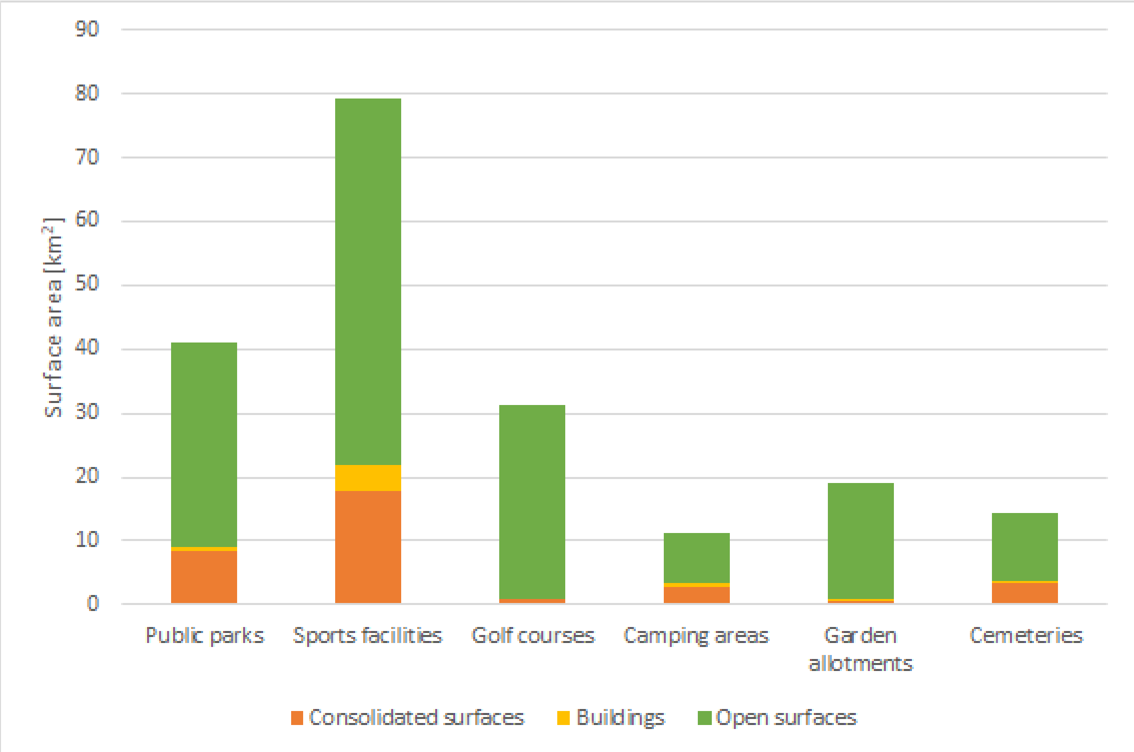

Figure 32: Categories of recreational areas and cemeteries and their fractions of consolidated surfaces, surfaces covered by buildings and open surfaces.

2.4.1.3. Categories of open surfaces

The more natural the soil in settlement areas is, the better the soil properties and its carbon storage potential can be assessed. Soils can be subdivided into classes based on their composition of mineral components or organic matter and on the use of a soil. Soils which have undergone strong anthropogenic change are of limited transferability. Such soils would require a more detailed soil analysis (*Arbeitskreis Stadtböden der deutschen Bodenkundlichen Gesellschaft 1996*). However, settlement soils can also be classified into different urban structure types according to their use (*Sauerwein 2006*), such as residential areas, industrial and commercial areas, recreational areas like parks, etc.

Using the data provided on land cover by the FSO areal statistics, we have in addition to the information if an open soil is located along a road or within a building area also the information if it is for instance covered by lawn or trees. This provides more information for the assessment of the open surfaces.

Based on the categories of land cover given by the FSO areal statistics, after some aggregation, we defined the following categories of land cover on open surfaces, for which the C concentration probably is in a similar range. Figure 33 shows the fractions of these categories with respect to the total amount of open surfaces.

- Lawns on artificial areas
- Trees on artificial areas
- Mix of small structures on artificial areas
- Gardens with border and patch structures on artificial areas
- Grass and herb vegetation
- Tree and shrub vegetation: aggregation of all types of tree and brush vegetation on not artificial area such as shrubs, brush meadows, short-stem fruit trees, vines, permanent garden plants and brush crops, closed forest, forest edges, forest strips, open forest, brush forest, linear woods, and clusters of trees
- Bare land: aggregation of the categories solid rock, granular soil, and rocky areas
- Watery areas: aggregation of the categories water, glacier/perpetual snow, wetlands, and reedy marshes
- Sealed soils may be Technosols or Anthrosols in general. They cover a very large surface area (e.g. 27% of the Geneva Canton surface area; *Viganò et al. n.d.*) and can potentially be opened to infiltration-depuration of surface waters (see section II.4.2.2 and *Embrén 2016*).

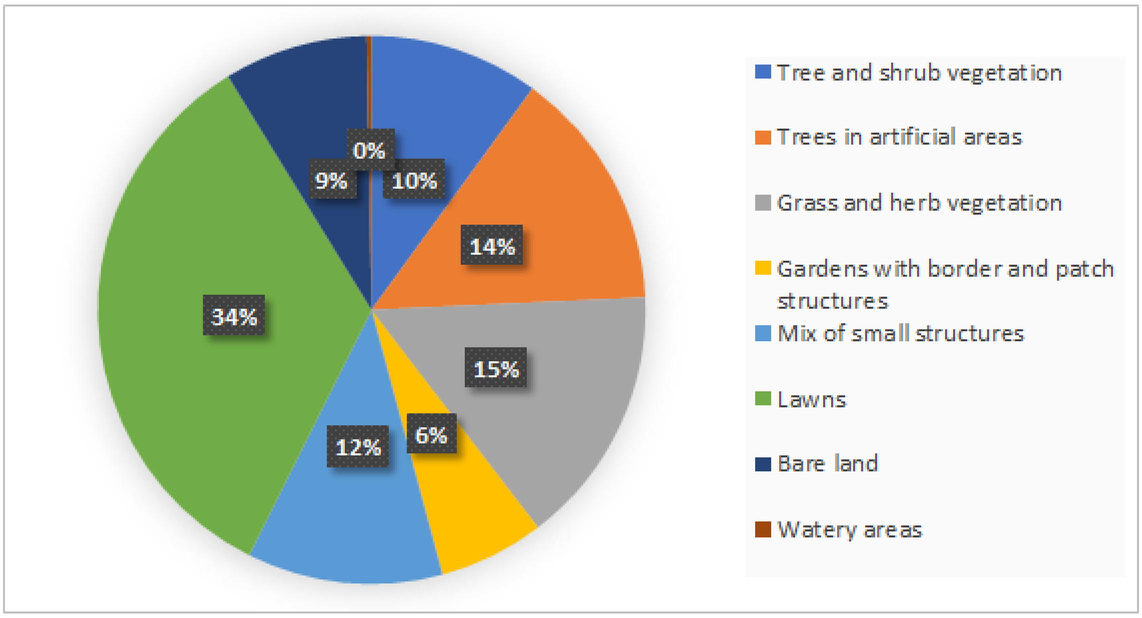

Figure 33: Fractions of land cover on open surfaces in settlements.

As Figure 33 shows, about a third of the area of open surfaces is covered by lawn, whereas watery areas cover less than 0.5% of the open surfaces on settlement soils. Not only the fraction of the land cover by the different categories differs but also the environment on which these land cover types are found (Figure 34).

Lawns make up 34% of open settlement areas. 56% of them are located in the surroundings of one- and two-family houses, 26% in building areas.

Trees in artificial areas make up 14% of open settlement areas. They are mainly found in the building area. 63% of the trees on artificially created areas are found in the surroundings of one- or two-family houses, 19% in the surroundings of blocks of flats.

Areas with mixed small structures make up 12% of the open settlement areas. 69% of these are in the surroundings of one- and two-family houses, 14% in the vicinity of blocks of flats.

Gardens with border and patch structures make up 6% of the open settlement surfaces and are mainly found in the building areas. There, a good half of the gardens with border and patch structures can be found in the surroundings of one- and two-family houses and more than a quarter in the vicinity of agricultural buildings. The area around blocks of flats accounts for 11% of the gardens with border and patch structures.

Grass and herb vegetation covers about 15% of the open settlement surfaces. More than half of it is located in transportation areas. About 20% can be found in building areas, the largest part of which is located in the vicinity of agricultural buildings, followed by the surroundings of one- and two-family houses.

Tree and shrub vegetation makes up about 10% of the open settlement surfaces. This type of land cover is mainly found in building and transportation areas. Although none of the subcategories aggregated in this category were assigned to artificially created areas, it is possible that soils along roads and railroad lines have been disturbed.

Bare land accounts for 9% of open settlement areas. It occurs mainly on special urban areas, such as landfills, mining areas, construction sites and energy supply facilities. Water and wet areas cover less than 0.5% of open surfaces on settlement soils. Therefore, they are not included in the further evaluation.

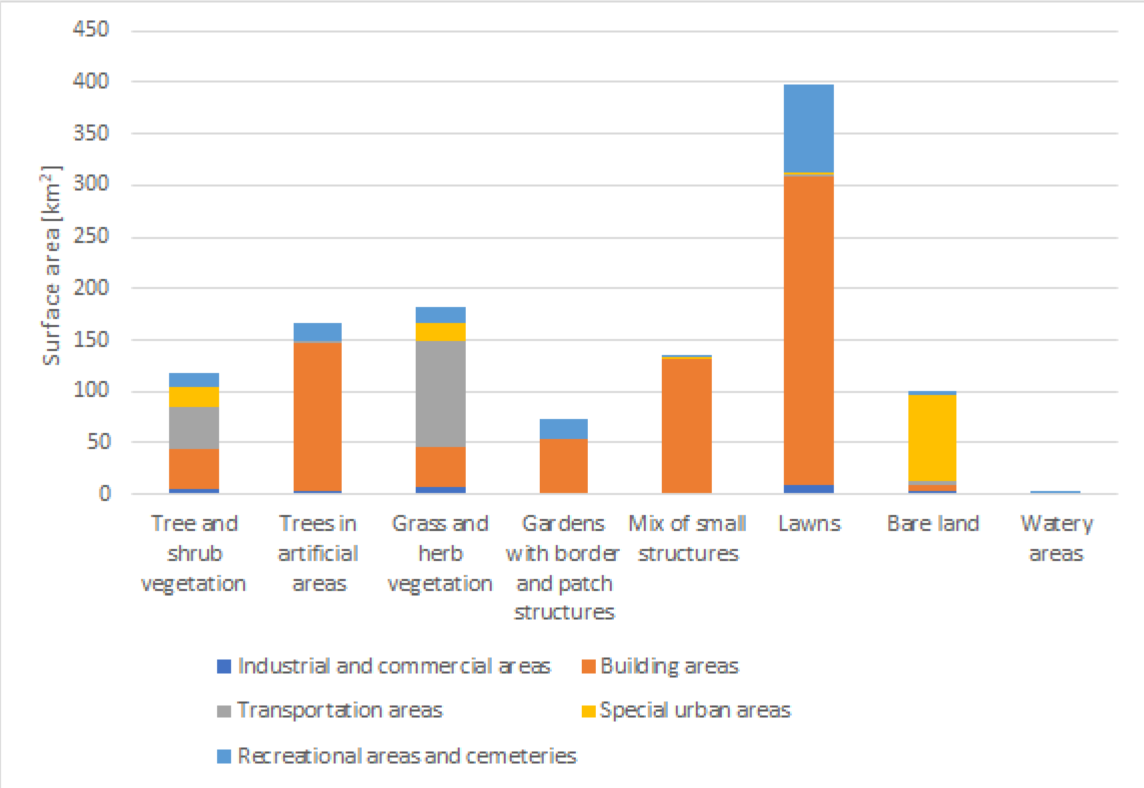

Figure 34: Occurrence of the land cover categories in the main categories of the areal statistics for open settlement soils.

2.4.1.4. Trends in land use and land cover on settlements soils

Settlement areas are expanding. One of the main results of the FSO areal statistics after the first two surveys (1979/85 and 1992/97) was a growth rate of 0.86 $m^2\ s^{-1}$ of urban areas at the expense of agricultural land. Historically, settlements were often built near especially fertile soils. Therefore, the expanding settlements are likely to replace crop rotation areas. The trend seemed to be less pronounced after the third survey (2004/09); between the second and third surveys, the growth in settlement areas reached only 0.69 $m^2\ s^{-1}$. The data of the latest survey (2013/18) has not been analysed for all of Switzerland; the eastern part of the country is still missing. However, based on the areas already evaluated, it can be concluded that settlement areas are still growing, but that this growth is continuing to slow down.

As the total coverage of settlement areas increases between surveys, most categories of land use and land cover show an increase in their surface, although not to the same extent in all categories. To allow considering the differences in the development of different types of surfaces independently of the growth of the total settlement areas, the percentage of a given category in the total settlement area is evaluated here instead of the area covered. The resulting graphs show an increase for categories that grow overproportionally strongly and a decrease for categories with a growth rate lower than the growth rate of the total settlement areas. Because data from the cantons of Grisons and St. Gallen are not yet available from the latest survey, these two cantons are excluded for the estimation of trends for all surveys to ensure comparability.

Figure 35 shows temporal changes in the main land use categories on settlement soils. Obviously, the building areas are the only ones that are growing at a larger rate than the settlement areas themselves. Therefore, we focus on the building areas for our further observations.

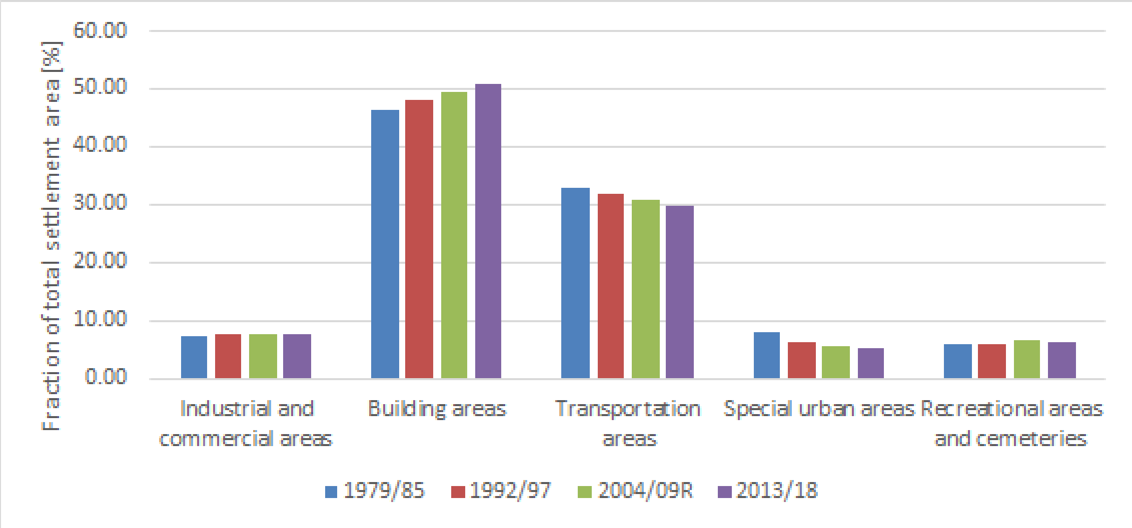

Figure 35: Changes in the fractions of the main categories of land use in settlement areas. Increase occurs for categories that grow overproportionally strongly compared with the overall growth of the Swiss settlement areas.

Within the building areas, the residential buildings and their surroundings show increasing fractions of the settlement areas over the years (Figure 36). The areas covered by public, agricultural and unspecified buildings and their surroundings show a similar or smaller growth rate compared with the total expansion of settlement areas. One- and two-family houses and their surroundings seem to be expanding at a comparable speed to blocks of flats and their surroundings.

With regard to the aim of spatial planning to increase the inward density of settlement areas, one could have expected that areas with blocks of flats would grow faster than other residential areas. This trend cannot be shown from the data. However, because there is no information on the building heights in these areas, the data cannot be used to make any statement about inward densification. Only in the areas with one- and two-family houses, it appears that surrounding areas grew less than the area of the buildings themselves between the last two surveys. This could be a sign that a slight inward densification has taken place in these areas.

The surroundings of one- and two-family houses are much larger in relation to building areas themselves than in areas with blocks of flats. Therefore, it is likely that future inward densification, by transforming quarters with one- and two-family houses into areas with blocks of flats, may lead to an increase in sealed areas at the expense of open surfaces in the residential areas.

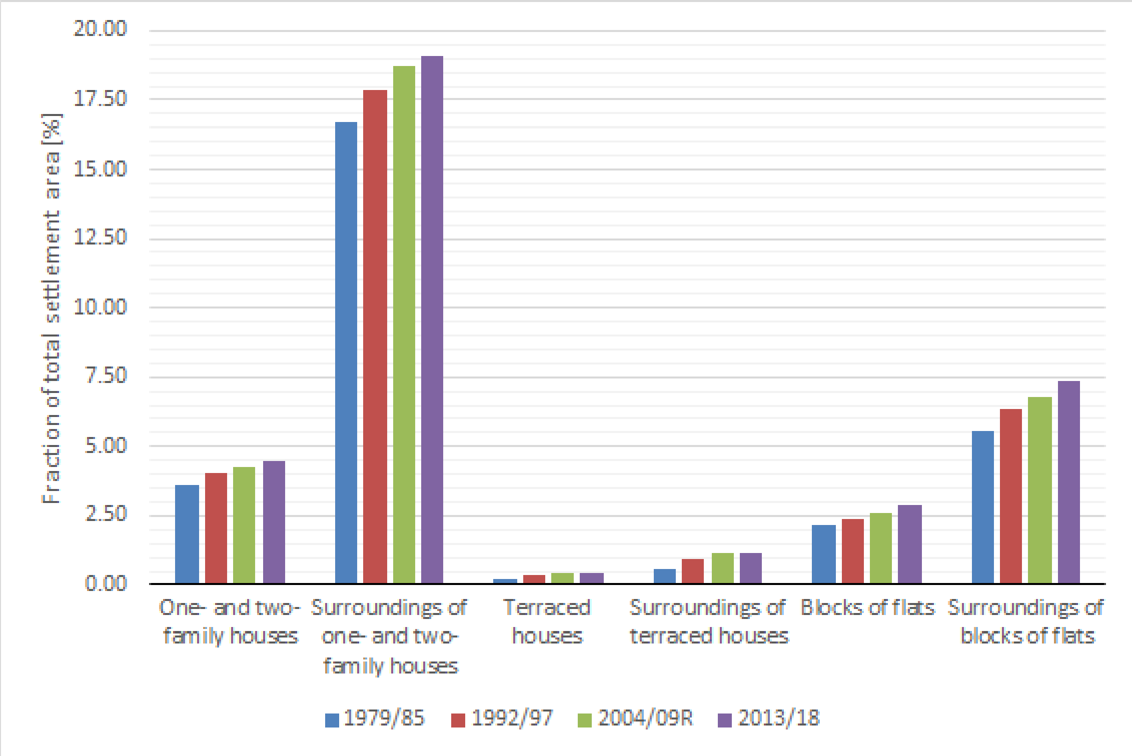

Figure 36: Changes in the fractions of buildings in settlement areas. Increase occurs for categories that grow overproportionally strongly compared with the overall growth of the Swiss settlement areas.

It can also be expected that the percentage of lawn will increase and the proportion of other categories of open settlement areas will decrease in case of a shift from one- and two-family houses to blocks with flats. Whereas 41% of the surroundings of one- and two-family houses are covered with lawn, the lawn cover for areas around blocks with flats is almost 60% for the survey 2004/09 of the FSO areal statistics.

To evaluate changes in land cover on the open settlement soils, the four categories related to artificial areas (see section II.4.1.3) from the land cover data of the FSO areal statistics are here analysed for all four surveys carried out by the FSO since the 1980s. The open surfaces belonging to the categories tree and shrub vegetation, grass and herb vegetation, bare soils, and watery areas are left out because they mainly occur on agricultural land. Without a detailed analysis of the intersection of the categories for land use and land cover of the FSO areal statistics, which has only been conducted for the survey 2004/09, it is not possible to make a statement about their change in the settlement area. As before, the cantons of Grisons and St. Gallen are excluded from the analysis of all the surveys to allow for the comparison of the data.

The fraction of open surfaces in artificial areas in relation to the total artificial areas is slightly decreasing with time. As shown in Figure 37, especially the gardens with border and patch structures are decreasing while the areas covered with lawns are increasing. Trees on artificial areas and mix of small structures do not show a clear trend.

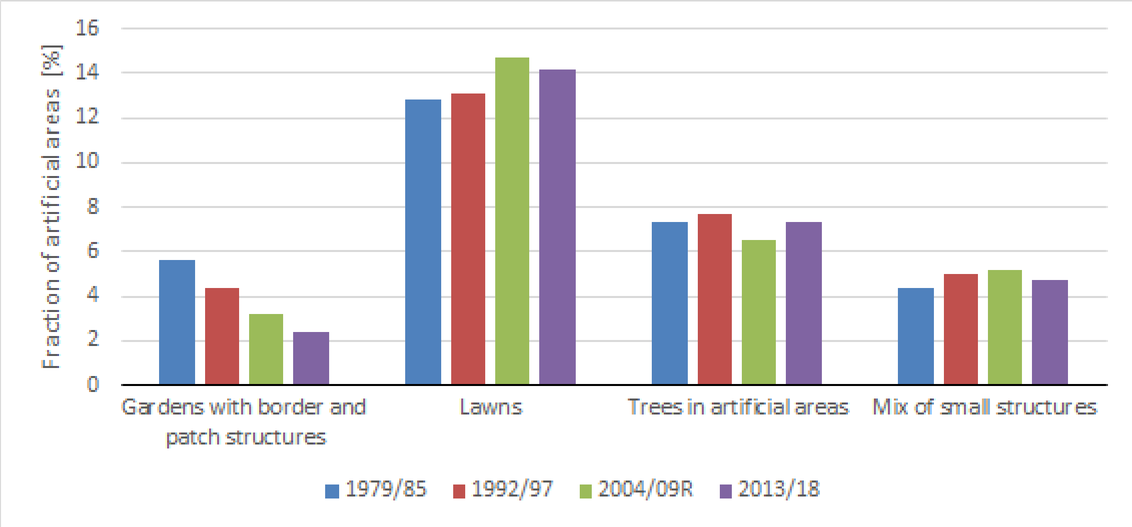

Figure 37: Changes in the fractions of land cover on open surfaces in artificial areas. Increase occurs for categories that grow overproportionally strongly compared with the overall growth of the Swiss settlement areas.

2.4.2. Measures to increase carbon storage in settlement soils

In settlement areas, two categories of measures can be distinguished. The first category can be described as constructional measures (such as green roofs, trees on artificial areas [tree lawns], biochar in tree substrates, unsealing). The second category includes adapted management and maintenance of current areas.

2.4.2.1. Green roofs – constructional measure

The creation of new areas in urban space can be achieved through structural measures within the framework of modern urban planning and building services engineering.

The total area of roofs in Switzerland amounts to about 50,529.7 ha. According to *Wüest und Partner (2017)*, 70% of roof areas in Switzerland are built as flat roofs. The potential for flat roof greening is thus available in an area of approximately 35,370 ha. An estimated 10% of this area is already green roof area (*written communication with S. Brenneisen 2020*).

According to *Getter et al. (2009)*, extensive green roof systems can store between 150 and 375 g carbon per m^2 (or 1.5–3.75 t C ha^{-1}) depending on the plant species composition. These values refer to the above- and belowground biomass. The sequestration rate for initially low-carbon or carbon-free substrates is around 50 g m^{-2} (0.5 t C ha^{-1}), due to C inputs to soil from leaf and root loss of plants during the year, depending on the maintenance management. If soil with a higher initial SOC concentration is used this rate will be lower. For 35,000 ha of roof area, this results in a soil carbon sequestration potential of 17.7 kt yr^{-1} for about 20 years (until a new steady state is reached). This estimate applies for the case that the potential areas are completely extensively greened. The effect could be further increased by intensively greened roofs.

2.4.2.2. Trees on artificial areas (tree lawns), biochar in tree substrates – constructional measure

Tree substrates in inner-city areas consist of mineral and organic components. These are usually gravel, sands, crushed stone, chippings and clays, as well as other organic components such as compost or humus (*FLL 2010*). To ensure the structural stability of the substrates and to minimize the problem of anaerobic processes (due to sealed surfaces or compaction), the proportion of organic matter must be kept to a minimum. For substrates that can be compacted and used as a roadbed, this is a maximum of 1–2 w/w% organic material (*FLL 2010*). The canton of Geneva has defined a standard for the tree plantation soils (*État de Genève 2013*) which recommends a minimum organic matter content of 1.5% and a minimum clay content of 10%.

With regard to the substrates which are structurally stable ('Pflanzgrubenbauweise 2') according to 'Forschungsgesellschaft für Landschaftsentwicklung und Landschaftsbau' (FLL) and 'Zusätzliche Technische Vertragsvorschriften München' (ZTV-Vegtra-Mü) (*Schönfeld 2017*), it is noticeable that these substrates are recommended for footpaths, bicycle paths and parking spaces but not for areas and roads with heavy traffic. These areas with substrates according to current standards remain unrootable or only slightly rootable for trees and contain very little carbon.

With a new approach and an adapted tree substrate with biochar as organic substance, a substrate can be produced that meets the physical requirements of civil engineering as well as the basic physiological conditions of the roadside greenery. Research results from the 'Zürcher Hochschule für Angewandte Wissenschaften' show that it is possible to work with about 10 vol% biochar (*Saluz 2017*). With 1 m^3 substrate, this corresponds to a mass of about 60 kg (a standardized weight [earth-moist] of the biochar of about 300 kg m^{-3} was assumed). Depending on the method of production, the biochar contains between 30% and 80% carbon, which would correspond to a C concentration of about 9–24 kg m^{-3} of substrate. This amount could be used in substrates under road surfaces both as a sink and as a plant-available substrate.

Full biochar tree plantation substrates have been used for more than 10 years in Stockholm (*Embrén 2016*), and experimental sites with Technosols made of 80% biochar are currently monitored in the cantons of Geneva and Vaud (*Plante&Cité Suisse 2020*). This experimental application is part of a currently developed multifunctional Technosol strategy, which aims at i) infiltrating urban stormwater (mitigation of urban flood hazard), ii) improving tree plantation condition and iii) promoting soil organic carbon (SOC) sequestration in urban settlements. The C concentration of these Technosols is about 70% of their dry mass, namely 560 kg m^{-3}. Combined with the finding that 27% of the surface area of the canton of Geneva are sealed soils that could be transformed into infiltrating Technosols, there is a huge perspective in terms of C sequestration. However, this strategy must respect some safety guidelines such as biochar being produced with urban wastes (circular economy and recycling).

The tree pits according to FLL provide a standardized size of 12 m^3 at a depth of >1.5 m (*FLL 2010*). The adjacent substrate layers should be rootable. In practice, the root zone of urban trees usually is smaller. In Bern, 6 m^3 is the minimum, and in other Swiss cities, the substrate volume is usually less than 12 m^3 due to lack of space. With the mentioned substrate property, the volume of these tree pits can be increased. According to *ZTV-Vegtra-Mü (2019)*, a volume of 36 m^3 should be aimed for. Based on the areas from the analysis of the FSO areal statistics, the following carbon sequestration potential can be assumed:

Trees on artificial areas occupy an area of about 16,738.1 ha. The average volume is assumed to be 12 m^3 at 1.5 m depth per tree. The associated total volume thus amounts to 251,071,368 m^3. The potential for an effective C storage in tree substrates is therefore about 4,519,284 t carbon (at 18 kg m^{-3}) for the existing areas. However, this calculation is for already existing areas. It is not possible to sequester 18 kg C in these substrates at once. This calculation can be used to estimate the potential of newly created areas within these given parameters. Primarily relevant is a sequestration potential of 18–48 kg C per m^3 substrate. This value can be assumed for new areas and areas to be developed.

2.4.2.3. Biochar under streets (in road beds) – constructional measure

Substrates including biochar could also be used in road beds under smaller streets. Between 18 and 48 kg C m^{-3} of substrate can be installed without static restrictions. For the existing areas, this would result in a potential of 17.8 Mt biochar-C (at 18 kg m^{-3}). However, adding biochar under existing streets would be very costly. According to the FSO areal statistics, roads (excluding highways) have grown by about 13% between 1985 and 2009. This growth rate corresponds to an area of about 8,223 ha. Assuming a similar increase between 2009 and 2043, this growth rate corresponds to a potential C sink under newly built road surfaces of about 2.22 Mt biochar-C until 2043.

2.4.2.4. Unsealing – constructional measure

Unsealing and re-cultivation can be used to restore soils on previously sealed surfaces. However, existing soil material, which is excavated at some other location, is generally being used for this purpose. This redistribution of soil does not result in any change with regard to SOC storage. An exception are unsealed areas that are not covered with redistributed existing soil but with substrates. There, additional C could be stored that is present in the organic fractions of the substrates. This is a measure that can most likely only be realized on small areas and, depending on the case, should be treated in a similar way as green roofs or substrates beneath single trees along sealed roads or squares.

2.4.2.5. Adapted management and maintenance of lawns and of grass and herb vegetation

A long-term study by ‘Universität für Bodenkultur Wien’ (BOKU) (*Schönthaler et al. 2007*) concluded that mulch mowing contributes positively to both soil activity and soil fertilization. Due to the increased oxygen content of the soil, an improved nutrient availability could be determined. The relevant main nutrients (NPK) were proven to be available to the plants when the cuttings were consistently returned as follows: 20–23 g nitrogen, 4–5 g phosphorus (P_2O_5) and 12 g potassium (K_2O). These nutrients were directly available to the lawn, and no fertilization was necessary. According to *Chen et al. (2014)*, the C/N ratio of lawn cuttings is 10:1. It can therefore be assumed that soil C storage in the lawn will increase if no cuttings are removed. At this C/N ratio, approximately 200–230 g organic C m^{-2} (or 2–2.3 t C ha^{-1}) and year can be returned to the soil. With a total area of 39,700 ha, this results in a potential of at least 79,460 t C yr^{-1} to be returned to Swiss lawns. How much of this additional C input would remain in the soil needs to be assessed.

2.4.2.6. Adapted management and maintenance of trees on artificial areas

Today's anthropogenically modified settlement soils offer little basis for urban green spaces and functioning soil processes with C storage. The substrates to be incorporated and future green space management must therefore provide the basis for sustainable soil development. This soil development must be achieved by means of adapted and dynamic planning and strategic management.

The potential of biochar in urban substrates has already been discussed in section 2.4.2.2. In addition, biochar could be applied as a fertilizer. According to *Beuttler et al. (2019)*, between 2.5 t and 6.7 t biochar-C (depending on which biochar is used) could be applied each year on existing areas of the category ‘trees on artificial areas’.

The adapted maintenance of green spaces for a sustainable C storage in the soil includes the return of the leaves to the substrate and the establishment of underplanting. Considering the huge N loads in the atmosphere and the low N levels in urban soil or substrate, leguminous plants, which can fix atmospheric N, could be included in the underplanting. Thus, additional N could be added to the substrate. The underplanting with partly deep rooting systems is an optimal instrument for active aeration of the soil. The resulting biomass and N input favours an optimization of the C/N ratio. With this method, fertilization for urban trees could become superfluous. In addition to the incorporation of plant C as described in section 2.4.2.5, levels of C storage as shown in Table 11 could be expected.

For the calculation, the mean value of C sequestration of the total litter layer of trees is used, which was calculated by *Liu et al. (2018)* from 27 forests. Root growth, as well as deadwood, is neglected in the following estimation. This C can be retained on the area if the foliage produced is consistently recycled. *Liu et al. (2018)* assumed 4.0 (±0.2) t C ha^{-1} yr^{-1} in the total litter layer. The calculations are also used for the areas ‘mix of small structures on artificial areas’, ‘gardens with border and patch structures on artificial areas’ and ‘tree and shrub vegetation’ (Table 11).

Table 11: Estimations of minimum and maximum soil organic carbon (SOC) storage with an adapted maintenance of trees per total area of the different area types.

Area type	**Min. SOC [t C yr^{-1}]**	**Max. SOC [t C yr^{-1}]**
Tree and shrub vegetation	44,745	49,455
Trees on artificial areas	63,604	70,299
Gardens with border and patch structures on artificial areas	27,849	30,781
Mix of small structures on artificial areas	51,568	56,997

2.4.3. Summary of settlement soils

Open surfaces make up around 40% of the settlement soils. They can mainly be found in the vicinity of one- or two-family houses and in recreation areas. In areas with blocks of flats, the fraction of open surfaces is smaller. Open surfaces are often covered with lawn, followed by grass and herb vegetation and trees on artificial areas. The analysis of the trends over the last 30 years shows that surfaces covered by buildings and sealed surfaces have been increasing strongly, while the fraction of open surfaces of the artificial surfaces have slightly been decreasing. Within the open surfaces, a change of land cover from gardens with border and patch structures towards lawns can be observed. Because inward densification with respect to living space within settlement areas will probably lead to less one- and two-family homes and more blocks with flats, it is likely that the fraction of open surfaces of the total artificial areas will decrease further.

The data basis for settlement soils is insufficient to allow reliable statements about the potential of additional carbon storage. The potential to create new areas, e.g. on roofs, and/or to achieve carbon accumulation and storage through structural measures is reasonabe. For green roofs the C sequestration potential is in the order of 18 kt C yr^{-1}. In addition, inducing a kind of succession on artificially constructed sites with less-developed soils could lead to carbon sequestration in these areas. A distinction must be made between natural conversion processes and enrichment and artificial sinks.

Lawns have a certain potential for C sequestration with some adjustments of their maintenance. For example, leaving the cuttings after mowing would result not only in an increase in SOC stocks, but also in a return of plant nutrients. This would reduce fertilization needs or even make fertilization unnecessary. However, the specified measures only make sense for green spaces where there are no extensive ecological compensation areas.

With the addition of biochar underneath newly built roads, theoretically about 2.2 Mt C could be sequestered in the next 20 years. This would increase the estimated C concentration in the soils of Switzerland's settlements by 20%. However, artificial sinks with biochar under sealed road surfaces are currently still very expensive and therefore not necessarily economical. Furthermore, biochar has to be produced from biomass grown within settlement areas to count as a true sequestration measure.

Most of the measures for C sequestration in settlement soils and surfaces are associated with a certain effort or cost. Nevertheless, these measures often lead to further benefits beyond C sequestration. Green roofs have a positive effect on the urban climate by counteracting the formation of heat islands and are good for local biodiversity. Using tree substrates with high fractions of stable C, such as biochar, has positive effects on water uptake and retention and, in the case of newly planted trees, does not cause much higher costs or work than common methods. Adapted management and maintenance of lawns and of grass and herb vegetation may lead to a certain restructuring or adjustment of current practices and might cause acceptance issues with residents, especially in parks. However, some larger cities such as Zurich are already investing in near-natural maintenance of open spaces and are seeing benefits of such a practice.

2.4.4. References

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