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Carbon stocks and carbon sequestration potentials in agricultural soils in Switzerland

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Contents

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Abstract

With the ratification of the Kyoto Protocol, Switzerland has committed itself to reducing its greenhouse gas emissions. In agriculture, mitigation of the greenhouse effect can be achieved by reducing nitrous oxide and methane emissions, and by sequestering carbon in soils. In this report, current carbon stocks of Swiss agricultural soils are estimated, and carbon sequestration potentials are discussed for various activities.

Most of the soil organic carbon in Swiss agriculture is stored under permanent grasslands, which account for more than 70% of the total agricultural area. The amounts of carbon stored in arable soils are distinctly smaller than under permanent grasslands, if site conditions are similar. Although intact and cultivated peatlands account for only a small percentage of the agricultural area, they play a significant role in Swiss carbon stocks due to the large amounts of carbon stored per hectare. In the case of mineral and organic soils, quantification of carbon is subject to major uncertainties regarding area and carbon stocks for the various land-use types. Due to the high share of permanent grassland and leys in arable rotations, sequestration potentials associated with changes in land use are limited for mineral soils. Expanding the area under no-till and converting temporary set-asides to permanent grasslands represent initial measures for enhancing carbon stocks. A larger theoretical potential is given by the conversion of arable land to grassland, but this activity would require significant changes in the agricultural structure. It is not yet possible to quantify the combined effects on soil carbon stocks of tillage, crop rotation, and other cultivation measures as practised in integrated and organic agriculture.

In organic soils, the mitigation potential lies mainly in avoiding CO_2 emissions by reducing oxidative peat losses. Restoration of cultivated peatlands could thus become an option with a high mitigation potential per hectare. As in the case of mineral soils, conflicting aims must be considered when cultivated peatlands are to be restored.

In conclusion, carbon sequestration and carbon savings have the potential to offset significant fractions of the agricultural emissions of nitrous oxide and methane. However, most activities would dramatically alter the agricultural structure and landscape scenery. Compared to studies of other regions (EU, US), the contribution of carbon sequestration to the overall greenhouse gas budget is small. The reasons for these differences are (i) the agricultural structure and natural conditions of Switzerland, which already favour land-use types with higher carbon stocks (grassland) and (ii) historically large carbon losses in other regions, making it possible to restore stocks in carbon-depleted soils by improving management practices.

Zusammenfassung

Mit dem Beitritt zur UN-Klimakonvention und der bevorstehenden Ratifizierung des Kyoto-Protokolls hat sich die Schweiz verpflichtet, die Emission von Treibhausgasen zu senken. In der Landwirtschaft besteht neben der Reduktion der Lachgas- und Methanemissionen die Möglichkeit, durch die Erhöhung der Kohlenstoffgehalte von Böden zusätzliches atmosphärisches CO_2 zu binden und damit dem Treibhauseffekt vorübergehend entgegenzuwirken. Im vorliegenden Bericht werden die Kohlenstoffgehalte der landwirtschaftlichen Böden abgeschätzt und Massnahmen zur Erhöhung der Kohlenstoffgehalte oder zur Verminderung bodenbürtiger CO_2-Emissionen diskutiert.

Entsprechend der Landnutzung in der Schweiz befinden sich die grössten Kohlenstoffvorräte von Böden unter Dauergrünland. Ackerböden enthalten unter ähnlichen Standortbedingungen durchschnittlich weniger Kohlenstoff als die Grünlandböden. Daneben spielen – trotz ihrer vergleichsweise kleinen Fläche – die intakten und kultivierten ehemaligen Moorstandorte eine wichtige Rolle als Kohlenstoffspeicher. Hinsichtlich der Quantifizierung der Kohlenstoffgehalte im Boden bestehen allerdings aufgrund der Datenlage in allen Regionen erhebliche Unsicherheiten.

Aufgrund des grossen Anteils an Dauergrünland und an Kunstwiesen in Ackerrotationen ist das Potenzial für eine Erhöhung der Kohlenstoffgehalte in Mineralböden begrenzt. Die Ausweitung des pfluglosen Anbaus sowie die Umwandlung ökologischer Ausgleichsflächen in Dauerbrachen bieten erste Ansätze zur Erhöhung der Kohlenstoffgehalte. Darüber hinaus stellt die Umwandlung von Acker- zu Dauergrünland ein beträchtliches Potenzial dar, welches allerdings erhebliche strukturelle Folgen für die Landwirtschaft hätte. Die Auswirkung kombinierter Massnahmen, wie sie in den Bewirtschaftungsformen der integrierten und biologischen Landwirtschaft zum Tragen kommen, lassen sich derzeit nicht quantifizieren. Bei organischen Böden steht vor allem die Vermeidung von CO_2-Emissionen durch oxidativen Torfabbau im Vordergrund. Die Renaturierung kultivierter Moore wäre daher eine Massnahme mit hohem Potenzial. Wie bei den mineralischen Böden müssen aber auch hier Zielkonflikte gegeneinander abgewogen werden.

Insgesamt könnten nach derzeitiger Abschätzung grosse Teile der landwirtschaftlichen Lachgas- und Methanemissionen durch Kohlenstoffspeicherung und CO_2-Emissionsvermeidung kompensiert werden, allerdings auf Kosten eines deutlich veränderten Landschaftsbildes. Zudem wären strukturelle Anpassungen (z.B. Moorrenaturierungen) notwendig. Im internationalen Vergleich ist das Verhältnis einer möglichen landwirtschaftlichen Kohlenstoffsenke zur jeweiligen Gesamtemission klein. Dies kann auf die bestehende landwirtschaftliche Struktur, die traditionell und naturräumlich bedingt vergleichsweise kohlenstoffschonend ausgerichtet ist, und auf den übermässigen Abbau an Bodenkohlenstoff in einigen Staaten, welcher zu einem hohen Wiedergewinnungspotenzial führt, zurückgeführt werden.

Résumé

La Suisse s'est engagée a réduire ses émissions des gaz à effet de serre en adhérant à la Convention de l'ONU sur le changement climatique et en ratifiant prochainement le Protocole de Kyoto. Outre la réduction des émissions de protoxyde d'azote et de méthane, l'agriculture a la possibilité de stocker (séquestrer) davantage de CO_2 atmosphérique dans le sol et de contrer ainsi transitoirement l'effet de serre. Le présent rapport donne une estimation de la quantité de carbone présent dans les sols agricoles et discute les mesures permettant d'augmenter ces réserves ou de diminuer les émissions de CO_2 provenant du sol.

Les plus importantes réserves de carbone accumulées dans le sol se trouvent sous les herbages permanents. Dans des conditions locales similaires, les sols affectés aux grandes cultures renferment en moyenne moins de carbone. Par ailleurs, les anciens marais, qu'ils soient encore intacts ou cultivés, sont d'importants réservoirs de carbone en dépit de leur petite surface. Dans toutes les régions, la quantité exacte du carbone présent dans le sol est cependant très incertaine en l'état actuel des connaissances.

Étant donné qu'une grande partie des surfaces herbagères permanentes et des prairies artificielles sont intégrées dans des plans de rotation culturale, le potentiel d'augmentation des teneurs en carbone des sols minéraux est limité. La renonciation de plus en plus courante à la charrue, ainsi que la transformation des surfaces de compensation écologique en jachères permanentes sont en revanche un premier pas vers l'augmentation du réservoir de carbone. La transformation des zones de grandes cultures en herbages permanents est une option qui offre à cet égard un potentiel considérable, mais qui aurait des répercussions sensibles sur les structures de l'agriculture. Il est actuellement impossible de quantifier les effets de mesures combinées au sens du cahier des charges de la production intégrée et de la production biologique. S'agissant des sols organiques, il s'agit avant tout d'éviter les émissions de CO_2 par la dégradation oxydative de la tourbe. La restauration des tourbières cultivées est par conséquent une mesure à fort potentiel. Comme pour les sols minéraux, il faut pondérer les intérêts qui sont en jeu.

On estime qu'à l'échelon global, une grande partie des émissions agricoles de protoxyde d'azote et de méthane pourraient être compensées par le stockage du carbone et la prévention des émissions de CO_2, mais au prix de remaniements sensibles du paysage, auxquels s'ajouteraient des adaptations structurelles (p.ex. restauration des tourbières). En comparaison internationale, le potentiel de stockage du carbone dans l'agriculture par rapport aux émissions globales de carbone est dérisoire en Suisse. Cette différence est d'une part imputable à la structure de notre agriculture, carbo-rétentive par tradition et de par son cadre naturel, et d'autre part, aux pertes excessives en carbone organique dans certains états, lesquelles sont vont de pair avec un potentiel de récupération élevé.

Preface

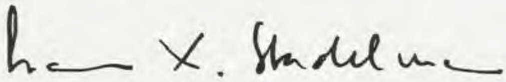
The present report has been prepared against the background of the Climate Convention and the Swiss commitments to report and to reduce anthropogenic emissions of greenhouse gases. This is the last of a series of three reports dealing with the three important agricultural greenhouse gases – methane (CH_4)[1], nitrous oxide (N_2O)[2], and carbon dioxide (CO_2).

Worldwide consumption of fossil fuels, changes in land use over the past 200 years, and food production are driving the increase in atmospheric concentrations of carbon dioxide and other greenhouse gases. This continued increase is enhancing the earth's natural greenhouse effect and leading, as a consequence, to rapid global warming. It has been suggested that by increasing the terrestrial uptake of CO_2 over the next few decades, it would be possible to reclaim some of the 150 or more gigatons of carbon emitted to the atmosphere worldwide as a result of land-use change between 1860 and 1984 alone. Agricultural soils are important reservoirs of terrestrial carbon and may act as carbon sinks through improvements in management or changes in land use. In Switzerland, about 37% of the country's total area is dedicated to agriculture, most of it in the form of permanent grasslands. Thus, by estimating current carbon stocks in these soils and evaluating potential carbon sources and sinks associated with particular agricultural activities, it should be possible to better define the role of Swiss agriculture in mitigating the greenhouse effect.

In view of this, the Swiss Agency for the Environment, Forests, and Landscape (SAEFL) commissioned the Air Pollution/Climate Group at the Swiss Federal Research Station for Agroecology and Agriculture (FAL) to review current knowledge on soil carbon sequestration, to quantify the carbon stocks of agricultural soils in Switzerland, and to evaluate the carbon sequestration potentials of relevant activities under Swiss conditions. With the completion of this report, Switzerland is joining other nations in the endeavour to evaluate the role of agriculture in the mitigation of climate change.

I wish to express my thanks to all those who have played a part in producing this report. This report was mostly funded by the Swiss Agency for the Environment, Forests, and Landscape (SAEFL).

Swiss Federal Research Station for Agroecology and Agriculture
FAL Reckenholz



Franz X. Stadelmann
Department Head Natural Resources / Environmental Protection

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[1] Minzonzio, G., Grub, A. und Fuhrer, J., 1998. Methan-Emissionen der schweizerischen Landwirtschaft. Schriftenreihe Umwelt Nr. 298, BUWAL, Bern.

[2] Schmid, M., Neftel, A. und Fuhrer, J., 2000. Lachgasemissionen aus der Schweizer Landwirtschaft. Schriftenreihe der FAL 33, Zürich-Reckenholz.

Vorwort

Mit dem vorliegenden Bericht erfüllt die Schweiz einen Teil der Vorgaben der Klimakonvention und der eingegangenen Verpflichtungen zur Berichterstattung über die Treibhausgasemissionen und zur Emissionsreduktion. Er ist der dritte Teil einer Serie von Berichten über die landwirtschaftlichen Treibhausgase Methan (CH_4)[3], Lachgas (N_2O)[4] und Kohlendioxid (CO_2).

Der weltweite Verbrauch von fossilen Brennstoffen, die fortschreitende Abholzung und Landnutzungsänderung sowie die Nahrungsmittelproduktion sind die wichtigsten Ursachen für den kontinuierlichen Anstieg der Treibhausgaskonzentrationen in der Atmosphäre. Diese Entwicklung verstärkt den natürlichen Treibhauseffekt und führt zu steigenden Temperaturen. In der Wissenschaft und in der Öffentlichkeit wird darüber diskutiert, ob den steigenden CO_2-Emissionen – weltweit alleine 150 Milliarden Tonnen Kohlenstoff durch Landnutzungsänderung zwischen 1885 und 1984 – unter anderem durch die Nutzung sogenannter Kohlenstoffsenken begegnet werden kann. Landwirtschaftliche Böden sind bedeutende Kohlenstoffspeicher und können über Bewirtschaftungs- oder Landnutzungsänderungen teilweise zusätzlichen Kohlenstoff aufnehmen. In der Schweiz werden etwa 37% der Fläche landwirtschaftlich genutzt. Der grösste Teil der Landwirtschaftsfläche ist Dauergrünland. Die Abschätzung der Kohlenstoffgehalte dieser Böden und ihrer möglichen Quellen- und Senkenfunktion bei unterschiedlichen Bewirtschaftungsoptionen ist daher eine wesentliche Voraussetzung, die Rolle der Landwirtschaft in der Klimadiskussion zu charakterisieren.

Das Bundesamt für Umwelt, Wald und Landschaft (BUWAL) hat daher, aufgrund ihrer langjährigen Fachkompetenz in Fragen landwirtschaftlicher Immissionen und Emissionen, die Arbeitsgruppe Lufthygiene/Klima der Forschungsanstalt für Agrarökologie und Landbau (FAL Reckenholz) gebeten, das gegenwärtige Wissen über Kohlenstoffsenken zusammenzustellen sowie die Kohlenstoffgehalte und möglichen Senkenpotenziale für die Schweiz zu evaluieren. Mit der Fertigstellung dieses Berichts trägt die Schweiz zum weltweiten Bemühen bei, das Potenzial der Landwirtschaft hinsichtlich möglicher Vermeidungsstrategien aufzuzeigen.

Den Autoren und allen, die einen Beitrag zum Gelingen geleistet haben, sowie dem BUWAL für die Mitfinanzierung, danke ich herzlich.

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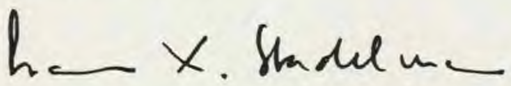
FAL Reckenholz



Franz X. Stadelmann
Leiter Produkt Umweltressourcen / Landwirtschaftlicher Umweltschutz.

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[3] Minzonzio, G., Grub, A. und Fuhrer, J., 1998. Methan-Emissionen der schweizerischen Landwirtschaft. Schriftenreihe Umwelt Nr. 298, BUWAL, Bern.

[4] Schmid, M., Neftel, A. und Fuhrer, J., 2000. Lachgasemissionen aus der Schweizer Landwirtschaft. Schriftenreihe der FAL 33, Zürich-Reckenholz.

Executive summary

Aim and scope

Carbon sequestration in soils is a climate change mitigation strategy based on the assumption that the flux of carbon from the air to the soil can be increased while the release of carbon from the soil back to the atmosphere is decreased. In other words, it is assumed that certain activities can transform soil from a carbon *source* (emitting carbon) into a carbon *sink* (absorbing carbon). This transformation has the potential to reduce atmospheric concentrations of carbon dioxide (CO_2), thereby slowing global warming and mitigating climate change. Under the Kyoto Protocol, it is possible to take carbon sinks into account to a certain extent in calculating national greenhouse gas balances. It is therefore important to have reliable information on current carbon stocks and potential sinks.

Agricultural soils present an important reservoir of organic carbon. In agro-ecosystems, unlike in forest ecosystems, the major carbon pool is located in the soil, not in the biomass. The amount of carbon stored in agricultural soils depends on climatic and site-specific conditions as well as on management decisions. Thus, it is possible to regulate soil carbon stocks by agricultural management within upper and lower limits, which are determined by natural constraints. The aim of this report is to estimate the amount of organic carbon stored in different agricultural soils in Switzerland, and to evaluate the potentials to either enhance carbon stocks by specific agricultural practices, or to reduce current carbon losses which occur mainly due to the cultivation of organic soils.

Agricultural structure

In Switzerland, agricultural soils cover about 1.53 million hectares (ha), or 37% of the country's total area. Of the total agricultural area, about 26% is under arable crop rotation – with leys (i.e. cultivated grassland) accounting for 28% of this area – and 74% is used as permanent grassland; the latter includes 538'000 ha classified as mountain farming (= located in the Swiss Jura mountains and the Alps), some of it lying at altitudes of up to 3000 m above sea level. Intensive agriculture is mainly practised in the Swiss Central Plateau, at altitudes of up to 1000 m. Most of this land (>90%) is subject to the requirements of integrated and organic farming. The proportion of organic soils is estimated at 2.4% of the total agricultural area. About half of the organic soils are intact peatlands, which are managed extensively as traditional litter meadows or pastures, while the remainder have been drained over the past 150 years and are now either under intensive arable cropping and vegetable growing (i.e. cultivated peatlands) or used as grassland.

Soil carbon stocks

By using data from soil surveys and combining these data with geo-referenced information on land use, topography, and soil characteristics, it was possible to calculate soil carbon stocks and assign these values to the various combinations of soil type, land use, and altitude. The analysis shows that soil carbon stocks in Swiss agriculture are primarily a function of i) the prevailing land-use type (e.g. arable land vs permanent grassland), ii) soil properties that limit carbon storage capacity (e.g. soil texture, profile depth), iii) climate, expressed in terms of altitudinal classes, and iv) the extent and duration of peatland cultivation. Figure S1 shows mean carbon stocks for various soils and land-use types.

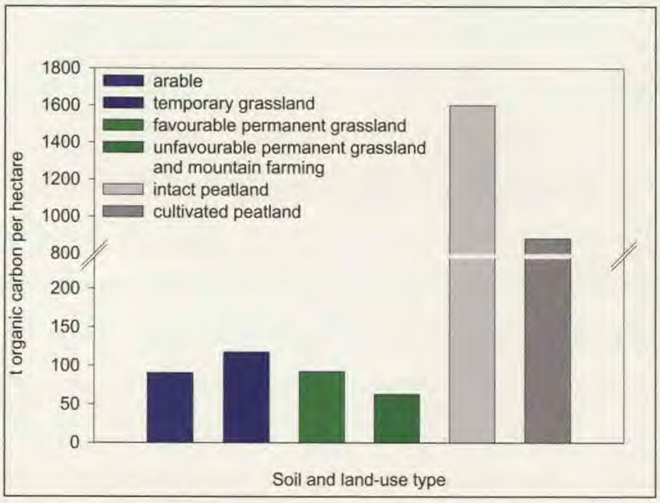

Figure S1. Mean soil carbon stocks per hectare for various land-use types and soils in Switzerland.

For organic soils, carbon stocks per hectare are more than one order of magnitude larger than for mineral soils. The difference between the carbon stocks of intact and cultivated peatlands is attributable to the fraction that was oxidised during prolonged periods of cultivation. Peat oxidation is an ongoing process. Carbon stocks of mineral soils differ across land-use types and are largest for temporary grasslands and smallest for alpine meadows, the latter being characterised by stony and shallow soil profiles, which limit the carbon storage potential of these soils.

Because the area of organic soils (i.e. intact and cultivated peatlands) is small in absolute terms – currently only 37'000 ha of the total agricultural area of 1'525'119 ha – the amount of carbon stored in organic soils is also small, but it still accounts for about 28% of the total carbon in agricultural soils. The main carbon reservoir is to be found in mineral soils covered with permanent grasslands, located on the Central Plateau, followed by carbon stored under alpine meadows (Figure S2). The total carbon stored in Swiss agricultural soils is 170 million tons.

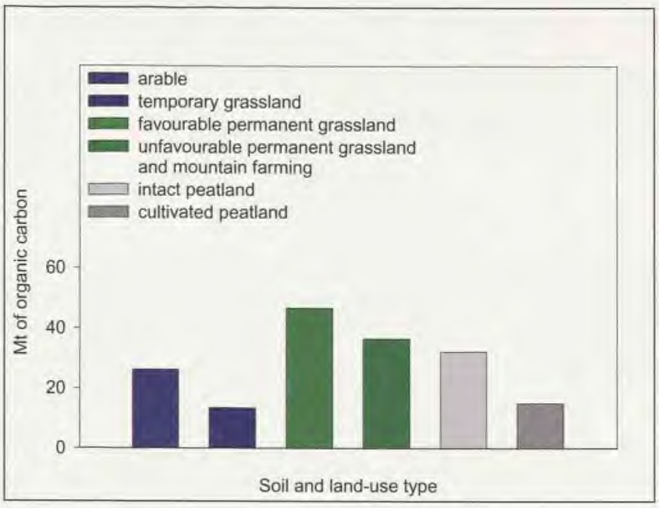

Figure S2. Total soil carbon stocks for various land-use types and soils in Switzerland.

Carbon sequestration potentials

Carbon sequestration is defined here as a site- and management-specific net increase in the amount of soil carbon over the entire course of the management cycle. Consideration of the whole management cycle (e.g. crop rotation) is essential in order to distinguish annual variations in the carbon stock from human-induced long-term accumulation or depletion of soil carbon. Soil carbon storage capacity, i.e. the maximum carbon stock for a particular set of environmental conditions, is attained to different extents by the various management systems. In general, large carbon stocks are promoted by permanent vegetation cover, high productivity, little soil disturbance, and conditions limiting organic matter decay, while arable rotations without leys or soil protection measures and a high tillage intensity lead to highly carbon-depleted soils (Figure S3).

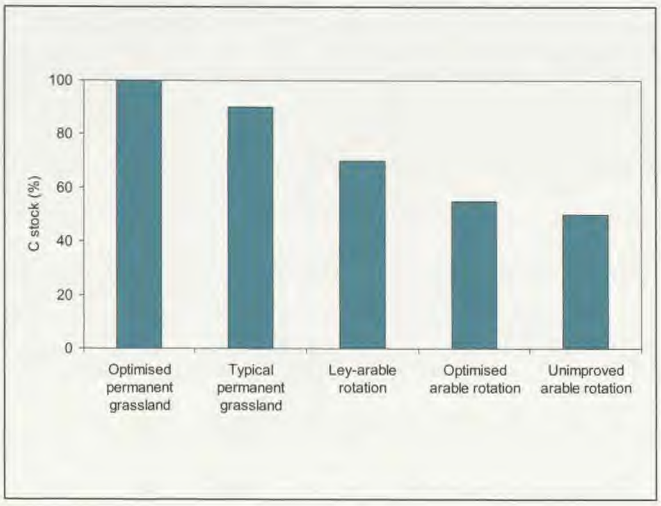

Figure S3. Carbon storage potentials for mineral soils under various management systems.

But it is important to note that the capacity of a mineral soil to store carbon is always finite, and that carbon sequestration is a reversible process. Any change in management that promotes carbon loss will counteract previous sequestration measures. In addition to carbon sequestration, avoidable carbon emissions were calculated in this study, since the avoidance of agricultural emissions also mitigates the increase in atmospheric CO_2.

Several activities with the potential to sequester carbon or to reduce carbon emissions were identified for Swiss agriculture. Currently, the only quantifiable activity is no-till, which is applied to a comparatively small arable area. **Theoretical** potentials were calculated for the following activities:

- conversion of all arable land to no-till
- conversion of all arable land to grassland
- conversion of all cropped peatlands to permanent grassland
- restoration of all cultivated peatlands

The quantification of sequestration rates and avoidable emissions is characterised by a high uncertainty. Calculated theoretical annual rates of carbon sequestration or carbon emissions avoided by the restoration of cultivated peatlands are summarized in Figure S4. They add up to a mean value of around 1.1 million tons of CO_2 per year. The mean theoretical potential corresponds to about 27% of the CO_2 emissions reduction to which Switzerland is committed under the Kyoto Protocol for the period 2008–2012, or 21% of the combined emissions (in CO_2-equivalents) of nitrous oxide (N_2O) and methane (CH_4). Complete realisation of the mean theoretical potential would drastically alter the current agricultural structure in Switzerland since it would require the conversion of all arable land to grassland and the restoration of all cultivated peatlands. A slightly smaller potential could be achieved by converting all cropping land to no-till management and restoring all cultivated peatlands. A more realistic estimate of the potential would lie somewhere between the theoretical maximum and the current sink of around 0.003 million tons of carbon due to no-till. It should be noted that the restoration of cultivated peatlands would result in enhanced CH_4 emissions, which would presumably counterbalance the sequestration effect over a period of decades or centuries.

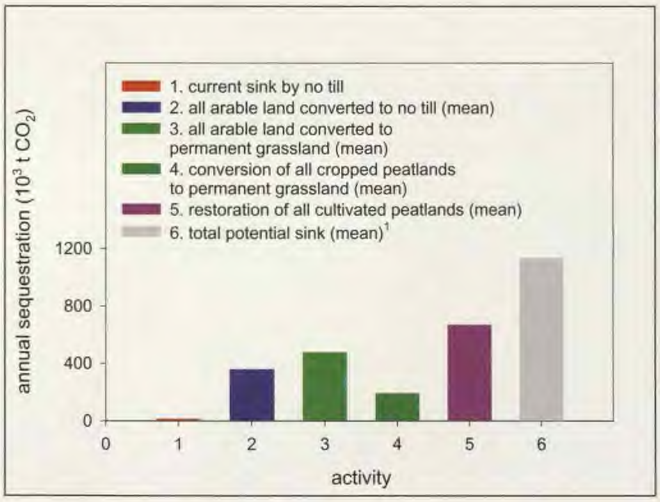

Figure S4. Annual sequestration potentials in Swiss agriculture. (Ranges and uncertainties are described in the report.)

[1] The total potential sink refers to the sum of the means for activities 3 and 5.

National and international comparisons

The mean potential carbon sink which could be achieved by agricultural practices was compared with other fluxes of greenhouse gases in Switzerland (Figure S5). The comparison shows that the potential agricultural sink is smaller than the emissions of CH_4 and N_2O in the agricultural sector (in 2000), and smaller than the estimated forest management sink. However, if agricultural practices designed to sequester carbon and to avoid further emissions from cultivated peatlands were applied to part of the total area, this would have the potential to offset the overall emissions balance of agriculture.

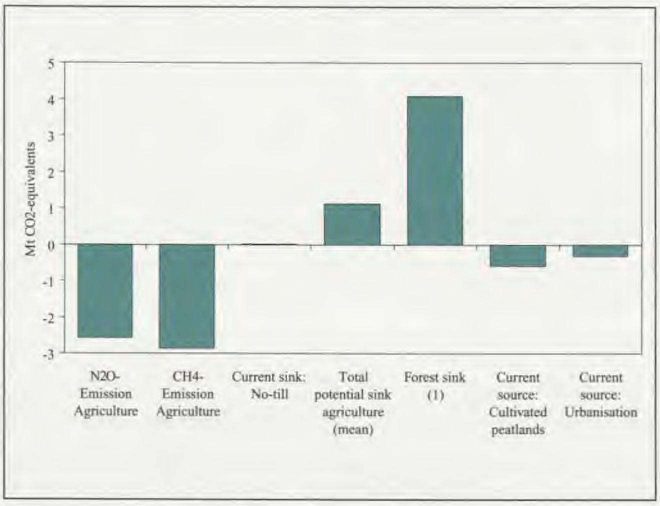

Figure S5. Annual carbon sequestration potentials in agriculture, compared with other sources and sinks of greenhouse gases in Switzerland.

[1] According to the Swiss Greenhouse Gas Inventory, mean value for the period 1990–1999. A maximum of 1.8 million tons is chargeable under articles 3.3 and 3.4 of the Kyoto Protocol.

Compared with data presented in recent studies on carbon sequestration in the US and in the EU, the potential in Switzerland, expressed as a proportion of gross national emissions (in CO_2-equivalents), is small (Figure S6).

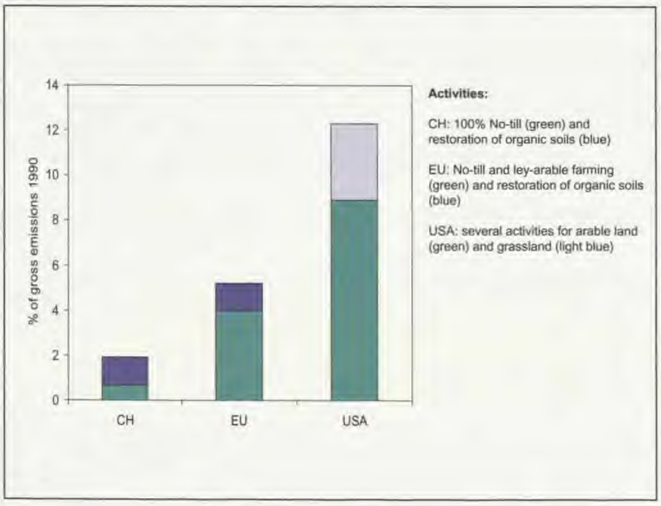

Figure S6. Agricultural sequestration potentials of Switzerland, the EU, and the US relative to gross emissions of greenhouse gases in 1990.

Comparison of carbon sequestration potentials across different countries indicates that the current agricultural structure in Switzerland already conserves agricultural C stocks and avoids excessive C losses in mineral soils. The difference between C sequestration potentials in Switzerland and the US may be attributable to differences in, for example:

- agricultural structure (ley-arable farming, large proportion of arable land),
- land-use histories (e.g. highly carbon-depleted soils in the US), and
- natural conditions (climate, topography, soil types).

Conclusions

The evaluation of soil carbon stocks in Swiss agricultural soils has confirmed the basic mechanisms and drivers for soil carbon stabilisation that have been proposed in the literature and has provided Swiss-specific data on soil carbon stocks and sequestration potentials. It is important to note that these findings rely on a weak data base of soil properties and land-use types, compared with the much larger amounts of information available in other countries.

Due to the country's agricultural structure – with a high proportion of temporary and permanent grassland, and integrated plus organic farming accounting for >90% of arable cropping – carbon sequestration potentials in Switzerland are small when compared with other greenhouse gas fluxes, or with recent estimates for other countries. The restoration of cultivated peatlands shows a considerable potential for avoidance of carbon emissions, which merits further investigation. Achieving the theoretical maximum carbon sequestration potential could compensate for up to 21% of the combined agricultural emissions of N_2O and CH_4. However, the changes that would be required in the agricultural structure are dramatic.

In conclusion, three key points need to be considered for future assessments of carbon sequestration activities as a means of mitigating climate change:

- It is difficult by direct measurement to distinguish between natural changes in soil carbon stocks and those which are directly or indirectly human-induced
- It is necessary to calculate the net greenhouse gas balance for a given activity; i.e. the "agricultural" greenhouse gases N_2O and CH_4 must be included in the budget.
- Such net greenhouse gas fluxes should be included in life-cycle assessments at the farm or regional level in future evaluations of sequestration measures in the agricultural sector.

Kurzfassung

Ziel

Die Speicherung von Kohlenstoff in Böden (Kohlenstoffsequestrierung) ist eine von mehreren Möglichkeiten, um den CO_2-Anstieg in der Atmosphäre zu verlangsamen und damit die globale Klimaveränderung teilweise abzupuffern. Diese Strategie beruht auf der Annahme, dass der Fluss von Kohlenstoff in den Boden erhöht bzw. der Fluss von Kohlenstoff aus dem Boden in die Atmosphäre beispielsweise durch landwirtschaftliche Bewirtschaftungsmassnahmen reduziert werden kann, so dass eine Nettosenke entsteht. Das Kyoto Protokoll ermöglicht grundsätzlich die Anrechnung solcher sogenannten Senken in Land- und Forstwirtschaft, um einem Teil der nationalen Reduktionsverpflichtung nachzukommen. Daher sind solide Informationen über die gegenwärtigen Kohlenstoffvorräte in Böden sowie über mögliche Senken und die entsprechenden Mechanismen, welche im Boden eine Rolle spielen, notwendig.

Böden spielen global betrachtet eine bedeutende Rolle als Speicher von organischem Kohlenstoff. In landwirtschaftlichen Ökosystemen ist – im Gegensatz zu Waldökosystemen – der grösste Anteil an Kohlenstoff in den Böden und nicht in der pflanzlichen Biomasse enthalten. Die Kohlenstoffmenge, die ein Boden enthält, ist abhängig von den Standorteigenschaften und der Bewirtschaftung. Durch Bewirtschaftungsmassnahmen ist es möglich, den Kohlenstoffgehalt des Bodens innerhalb bestimmter Ober- und Untergrenzen zu regulieren. Diese Studie hat zwei Hauptziele: 1. Die Abschätzung der Kohlenstoffmenge in den verschiedenen landwirtschaftlichen Böden der Schweiz; 2. Die Bewertung des Potenzials möglicher Sequestrierungsmassnahmen oder von Massnahmen, die die gegenwärtigen Kohlenstoffverluste aus landwirtschaftlich genutzten organischen Böden verringern.

Die landwirtschaftliche Struktur

Landwirtschaftliche Böden bedecken 1,53 Millionen Hektaren oder 37% der Gesamtfläche der Schweiz. Ungefähr 26% der landwirtschaftlichen Fläche wird ackerbaulich genutzt. Von den Ackerrotationen sind etwa 28% Kunstwiese. Den grössten Anteil (74%) der landwirtschaftlichen Fläche nimmt Dauergrünland ein, wovon ca. 538'000 Hektaren alpwirtschaftliche Nutzflächen sind, die sich im Juragebiet und in den Alpen bis auf 3000 Meter ü. d. M. erstrecken. Intensivlandwirtschaft wird vor allem im Schweizer Mittelland auf Höhen bis ca. 1000 Meter ü. d. M. betrieben. Über 90% dieser Flächen werden nach den Vorgaben der integrierten Produktion und des biologischen Landbaus bewirtschaftet. Der Anteil organischer Böden an der landwirtschaftlichen Fläche ist mit 2,4% relativ gering. Ungefähr die Hälfte dieser Böden wird extensiv bewirtschaftet und weist ein relativ ungestörtes Torfprofil auf. Der andere Teil wurde im Verlauf der letzten 150 Jahre entwässert und kultiviert. Diese Flächen sind heute zum Teil unter intensiver landwirtschaftlicher Nutzung (z.B. Gemüsebau) oder werden als Grünland genutzt.

Kohlenstoffgehalt der Böden

Die Kohlenstoffgehalte der landwirtschaftlichen Böden wurden mit Hilfe von vorhandenen Bodendaten und flächendeckend vorliegenden Informationen zu Landnutzung, Topographie und Bodeneigenschaften geschätzt. Dazu wurden die Kohlenstoffgehalte für alle existierenden Merkmalskombinationen aus Bodeneigenschaften, Landnutzungstypen und Topographie berechnet und mit der jeweiligen Fläche multipliziert. Die Auswertung dieser Informationen zeigt, dass der Kohlenstoffgehalt der Böden im wesentlichen durch 1. den vorherrschenden Landnutzungstyp (z.B. Ackerbau vs. Dauergrünland), 2. die Kohlenstoffspeicherungskapazität bestimmende Bodeneigenschaften (Bodenart, Steingehalt, Gründigkeit), 3. das Klima, abgeleitet aus den topographischen Informationen, und 4. die Dauer und das Ausmass der Kultivierung der Moore bestimmt wird. Mittlere Kohlenstoffgehalte je Hektare sind in Abbildung S1 für verschiedene Landnutzungstypen und Böden dargestellt.

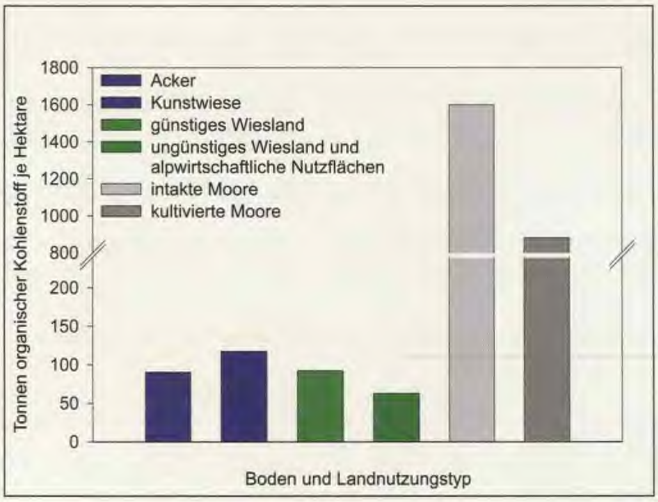

Abb. S1. Mittlere Kohlenstoffgehalte pro Hektare für unterschiedliche Landnutzungstypen und Böden in der Schweiz.

Die Kohlenstoffgehalte organischer Böden liegen um mindestens eine Grössenordnung über denen der Mineralböden. Die Unterschiede zwischen intakten und kultivierten Mooren sind durch den oxidativen Torfabbau bedingt, welcher im Verlauf mehrerer Jahrzehnte zu einem vollständigen Verlust des Torfes führt. Dieser Torfabbau findet auch heute noch statt. Bei den Mineralböden weisen die Kunstwiesen die höchsten und die Alpweiden die geringsten Kohlenstoffgehalte auf. Alpwirtschaftliche Flächen sind häufig durch flache und steinige Böden gekennzeichnet, die die Kohlenstoffspeicherung begrenzen.

Der gesamte Kohlenstoffvorrat in Schweizer landwirtschaftlichen Böden beträgt ca. 170 Millionen Tonnen Kohlenstoff. Organische Böden (= intakte und kultivierte Moore) enthalten trotz ihres geringen Flächenanteils

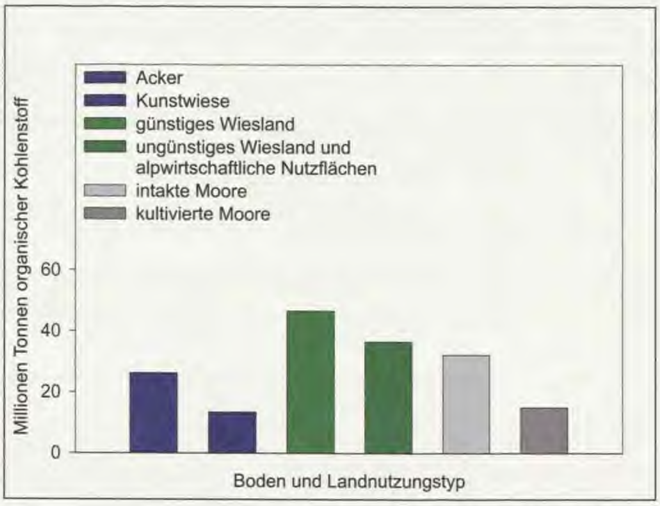

Abb. S2. Gesamtkohlenstoffvorräte unterschiedlicher Landnutzungstypen und Böden in der Schweiz.

von 37'000 Hektaren ca. 28% des gesamten Kohlenstoffs landwirtschaftlicher Böden. Der Hauptanteil des Kohlenstoffs ist in den Mineralböden unter Dauergrünlandnutzung gespeichert, die sich vor allem im Mittelland befinden. Aufgrund der grossen Fläche sind auch die Kohlenstoffvorräte der alpwirtschaftlichen Flächen bedeutend (Abb. S2)

Sequestrierungspotenziale für Kohlenstoff

Kohlenstoffsequestrierung wird hier als standortspezifischer Nettozuwachs an Bodenkohlenstoff über den Gesamtzeitraum einer Fruchtfolge definiert. Schwankungen des Kohlenstoffgehalts im Boden innerhalb der Fruchtfolge, die im Einzeljahr höher sein können als die durchschnittliche jährliche Nettozu- oder Abnahme, werden somit vom gesuchten Nettofluss über den Gesamtbetrachtungszeitraum getrennt. Bewirtschaftungssysteme und Fruchtfolgen nutzen das maximale Bindungspotenzial eines Bodens unterschiedlich stark aus. Das Bindungspotenzial ist hier definiert als der maximale Kohlenstoffgehalt, der unter üblicher landwirtschaftlicher Nutzung in einem Mineralboden erzielt werden kann. Eine dauerhafte Bodenbedeckung (z.B. durch Grünland), hohe Produktivität (diese bewirkt einen hohen Kohlenstoffeintrag in den Boden), minimale Bodenbearbeitung (verringerter oxidativer Kohlenstoffverlust) und limitierende Bedingungen für den Kohlenstoffabbau im Boden unterstützen steigende oder hohe Kohlenstoffgehalte im Boden. Im Gegenzug führt Ackerbau mit intensiver Bodenbearbeitung, kurzen Phasen der Bodenbedeckung und geringem Eintrag von Ernterückständen zu verringerten oder niedrigen Kohlenstoffgehalten. Diese Auswirkung unterschiedlicher Bewirtschaftungstypen auf die Kohlenstoffgehalte des Bodens sind schematisch in Abbildung S3 dargestellt. Die Kohlenstoffsequestrierung in Mineralböden ist stets ein Prozess, der durch das Bindungspotenzial des Bodens limitiert und grundsätzlich reversibel ist. Eine Änderung der Bewirtschaftung bei hohen Kohlenstoffgehalten der Böden kann daher potentiell zu Kohlenstoffverlusten führen.

In dieser Studie werden auch vermeidbare CO_2-Emissionen kultivierter organischer Böden als Sequestrierungspotenziale gerechnet, da solche Aktivitäten zu einem verlangsamten Anstieg des atmosphärischen CO_2-Gehaltes führen.

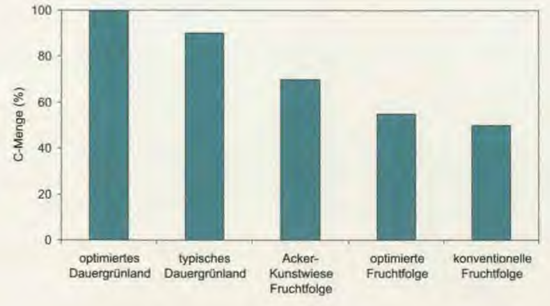

Abb. S3. Schematische Darstellung der Kohlenstoffspeicherung unterschiedlicher Landnutzungstypen und Bewirtschaftungsformen.

Für die Schweizer Landwirtschaft wurden mehrere Aktivitäten identifiziert, die zu einer Kohlenstoffsequestrierung oder einer Verringerung oxidativer Kohlenstoffverluste führen können. Zum gegenwärtigen Zeitpunkt kann nur für die Direktsaat, welche auf einer verhältnismässig kleinen Fläche praktiziert wird, eine Quantifizierung der aktuellen Senke angegeben werden. Theoretische Potenziale wurden für folgende Aktivitäten berechnet:

- Umwandlung der gesamten Ackerfläche in Direktsaatflächen
- Umwandlung der gesamten Ackerfläche in Dauergrünland
- Umwandlung der ackerbaulich genutzten ehemaligen Moore in Dauergrünland
- Renaturierung aller kultivierten Moore

Die kalkulierten **theoretischen** jährlichen Raten der Sequestrierung oder der vermiedenen Emissionen (kultivierte Moore) sind in Abbildung S4 dargestellt. Bei der Quantifizierung bestehen erhebliche Unsicherheiten. Insgesamt könnten im Mittel 1,1 Millionen Tonnen CO_2 pro Jahr mit einer optimalen Massnahmenkombination sequestriert werden. Dies entspricht in etwa 27% der Schweizer Kyoto-Verpflichtung zur CO_2-Reduktion oder 21% der jährlichen Emissionen von Lachgas plus Methan aus der Landwirtschaft. Es ist zu betonen, dass die Ausschöpfung dieses maximalen Potenzials, welches durch eine Umwandlung aller Acker- zu Grünlandflächen und eine Renaturierung aller kultivierten Moore erreicht werden könnte, eine drastische Veränderung der landwirtschaftlichen Struktur in der Schweiz bedingen würde. Ein etwas geringeres Potenzial ergibt sich bei einer Kombination aus der Umwandlung aller Ackerflächen von Pflug zu Direktsaat mit der Renaturierung der kultivierten Moore. Eine Renaturierung der kultivierten Moore würde aller Voraussicht nach zu deutlich erhöhten Methanemissionen führen, die den Senkeneffekt innerhalb einiger Jahrzehnte bis Jahrhunderte kompensieren würden. Ein realistisches Szenario läge irgendwo zwischen dem Maximalpotenzial und der derzeit realisierten Sequestrierung durch Direktsaat von etwa 3000 Tonnen Kohlenstoff pro Jahr.

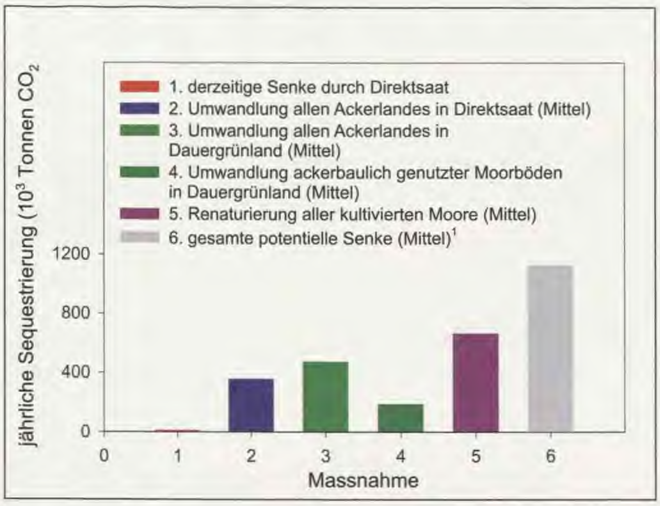

Abb. S4. Jährliche Sequestrierungspotenziale in der Schweizer Landwirtschaft. Spannweiten und Unsicherheiten werden im Bericht beschrieben.

[1] Die gesamte potentielle Senke ergibt sich aus den Massnahmen 3 und 5.

Nationaler und internationaler Vergleich

In Abbildung S5 wird das mittlere erreichbare Senkenpotenzial mit anderen Treibhausgasflüssen in der Schweiz verglichen. Die theoretische Senke in der Landwirtschaft ist deutlich kleiner als die landwirtschaftlichen Emissionen an Lachgas und Methan (Jahr 2000) und ebenfalls deutlich kleiner als die Waldsenke (Mittel 1990 bis 1999). Dennoch könnten die beschriebenen Massnahmen, wie z.B. eine Kombination aus vermehrter Direktsaat und der Renaturierung kultivierter Moore, zu einer verbesserten Gesamtbilanz beitragen.

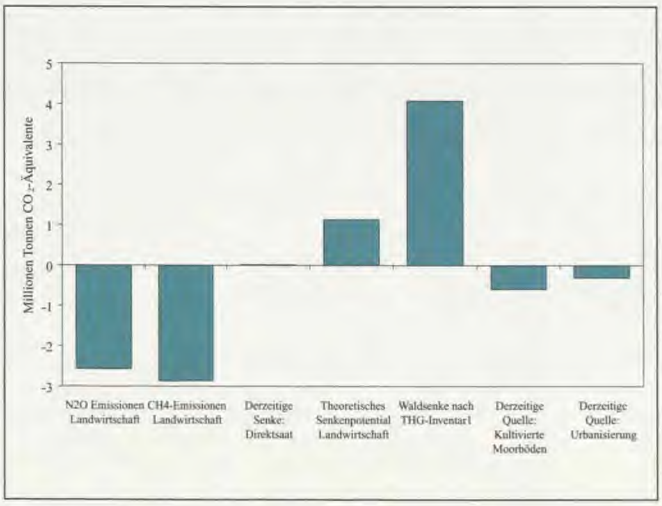

Abb. S5. Jährliches Kohlenstoffsequestrierungspotenzial in der Landwirtschaft im Vergleich zu anderen Quellen und Senken von Treibhausgasen in der Schweiz.

[1] Mittel der Periode 1990 bis 1999 (Schweizerisches Treibhausgasinventar); davon sind maximal 1,8 Mio. Tonnen für Waldbewirtschaftung sowie Aufforstungen für das Kyotoziel anrechenbar.

Wenn Daten anderer Studien aus der EU und den USA herangezogen werden (Abb. 6), ist das Senken- bzw. Vermeidungspotenzial der Schweizer Landwirtschaft in Bezug auf die Bruttoemissionen des Landes vergleichsweise klein. Der in der Studie angestellte Vergleich der Potenziale der unterschiedlichen Länder zeigt deutlich, dass die gegenwärtige landwirtschaftliche Struktur der Schweiz die vorhandenen Kohlenstoffvorräte in Böden weitgehend schützt (mit Ausnahme der kultivierten Moore) und übermässige Kohlenstoffverluste aus Mineralböden vermeidet. Wichtigste Unterschiede hinsichtlich der Senken- bzw. Vermeidungspotenziale zwischen den zitierten Ländern bzw. Ländergruppe sind:

- Die landwirtschaftliche Struktur (Anteil an Kunstwiesen, offenen Ackerflächen Dauergrünland),
- Landnutzungsgeschichte (z.B. stark an Kohlenstoff verarmte Standorte im mittleren Westen der USA),
- Naturräumliche Bedingungen (Klima, Topographie, Bodentypen)

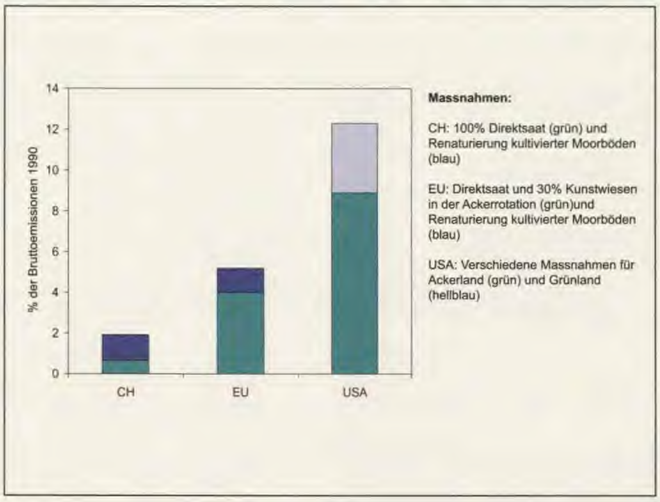

Abb. S6. Landwirtschaftliche Sequestrierungspotenziale in der Schweiz, der EU, und den USA im Verhältnis zu den Bruttoemissionen an Treibhausgasen in den 1990er Jahren.

Schlussfolgerungen

Die Erhebung und Quantifizierung der Kohlenstoffvorräte der landwirtschaftlichen Böden konnte wesentliche Steuergrössen der Kohlenstoffbindung- und speicherung herausarbeiten und zum grossen Teil Angaben aus anderen Regionen bestätigen. Die berechneten Daten bilden eine landesspezifische Grundlage der Kohlenstoffvorräte und sind damit eine wichtige Voraussetzung für die Abschätzung möglicher Senken. Im Vergleich zu anderen Ländern ist allerdings die Datengrundlage für eine Abschätzung bei den Bodendaten schmal, und die flächenhaften Informationen über Bodeneigenschaften und Landnutzungskategorien liegen in nur mässiger Auflösung vor. Das maximale Sequestrierungspotenzial ist aufgrund der landwirtschaftlichen Struktur, die sich durch einen hohen Anteil an Kunstwiesen, Dauergrünland, und einen über 90%-Anteil an integrierter und biologischer Produktion auszeichnet, sowohl in Bezug auf die nationalen landwirtschaftlichen Emissionen an Lachgas und Methan, als auch hinsichtlich der Senkenpotenziale anderer Länder vergleichsweise gering. Ein wesentliches Element einer Emissionsvermeidungsstrategie in der Landwirtschaft, welches noch detaillierter ausgearbeitet werden müsste, wäre die Renaturierung kultivierter Moorstandorte. Abschliessend muss festgehalten werden, dass zur Ausschöpfung des in dieser Studie berechneten maximalen Senken- bzw. Emissionsvermeidungspotenzials, welches sich auf ca. 21% der landwirtschaftlichen Lachgas- und Methanemissionen beläuft, eine wesentliche Veränderung der landwirtschaftlichen Struktur der Schweiz Voraussetzung wäre.

Condense

Objectif

Le stockage de carbone dans les sols (séquestration de carbone) représente une option pour ralentir la progression du CO_2 dans l'atmosphère et pour atténuer ainsi partiellement les changements globaux du climat. Cette stratégie se base sur la supposition qu'il est possible d'augmenter le flux du carbone dans le sol, respectivement de réduire le flux du carbones du sol vers l'atmosphère, par ex. à l'aide de pratiques agricoles pour que, globalement, il y ait une réduction. Le protocole de Kyoto permet de comptabiliser ces piégeurs biologiques en agriculture et en foresterie pour couvrir une partie des engagements de réduction au niveau national. C'est pourquoi, il est nécessaire d'avoir des informations fiables sur les réserves actuelles en CO_2 des sols, sur des piégeurs possibles et les mécanismes y relatif jouant un rôle dans le sol.

Globalement, les sols jouent un rôle important de stockage du carbone organique. Dans les écosystèmes agricoles, contrairement aux écosystèmes forestiers, la plus grande partie du carbone se trouve dans les sols et non dans la biomasse végétale. La quantité de carbone contenue dans un sol dépend des caractéristiques de l'emplacement et de l'exploitation. A l'aide de mesures culturales, il est possible de régulariser la teneur en carbone du sol à l'intérieur de limites supérieures et inférieures précises. Cette étude a deux objectifs principaux: 1) l'estimation des quantités de carbone dans les divers sols agricoles suisses; 2) l'évaluation du potentiel d'éventuelles mesures de séquestration ou de mesures qui réduisent les pertes actuelles de carbone des sols organiques utilisés en agriculture.

La structure agricole

Les sols agricoles couvrent 1.52 mio. ha soit 37% de la surface totale de la Suisse. Environ 26% de la surface agricole sont utilisés pour les grandes cultures, environ 28% de l'assolement des grandes cultures sont des prairies artificielles. La grande partie de la surface agricole (74%) est constituée d'herbages permanents, dont environ 538'000 ha sont des surfaces utiles d'alpage, lesquelles s'étendent dans la région du Jura et dans les Alpes jusqu'à une altitude de 3000 m. L'agriculture intensive est surtout pratiquée sur le plateau suisse jusqu'à une altitude d'environ 1000 m. Plus de 90% de ces surfaces sont cultivées selon le cahier des charges de la production intégrée. La part que prennent les sols organiques dans la surface agricole est relativement basse et atteint 2,4%. Environ la moitié de ces sols est exploitée de manière extensive et présente des profils de tourbe relativement intacts, tandis que l'autre moitié a été drainée et mise en culture au cours des 150 dernières années. Ces surfaces sont aujourd'hui soit exploitées de manière intensive (par ex. culture maraîchère), soit utilisé en tant qu'herbage.

Teneur en carbone des sols

Les teneurs en carbone des sols agricoles ont été estimées en se basant sur des données de sols existantes et sur les informations disponibles concernant l'utilisation du sol, la topographie et les caractéristiques des sols. Les teneurs en carbone ont été calculées pour toutes les combinaisons des caractéristiques existantes : caractéristiques du sol, types d'utilisation du sol et topographie. Elles ont ensuite été multipliées avec la surface correspondante. L'exploitation de ces informations montre que les teneurs en carbone des sols sont déterminées principalement par 1) le type d'utilisation du sol prédominant (par ex. grandes cultures vs herbages permanents), 2) les caractéristiques du sol qui déterminent la capacité de stockage de carbone (types de sol, teneur en pierre, profondeur), 3) le climat, déduit des informations topographiques et 4) la durée et la dimension de l'exploitation des tourbières. Les teneurs moyennes en carbone par hectare sont représentées dans la figure S1 pour divers types d'utilisation du sol et pour différents sols.

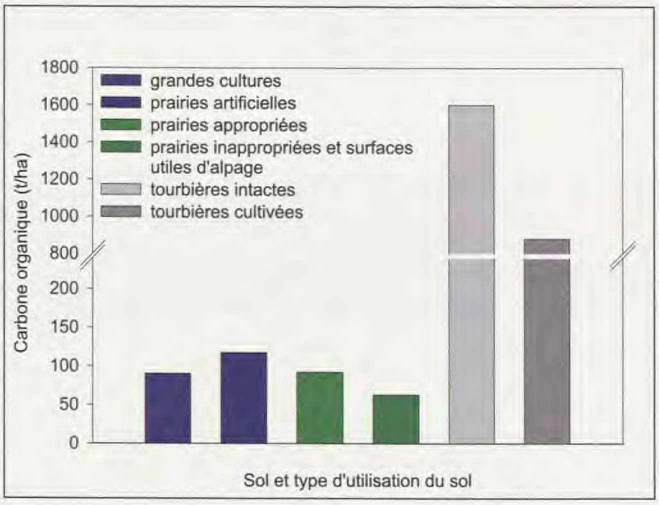

Figure S1. Teneur moyenne en carbone par hectare pour différents types d'utilisation du sol et différents sols en Suisse.

Les teneurs en carbone des sols organiques se situent au minimum un ordre de grandeur au-dessus de ceux des sols minéraux. Les différences entre les tourbières intactes et les cultivés sont dues à la dégradation oxidative de la tourbe, qui mène au cours des décennies à une perte complète de la tourbe. Cette dégradation est actuellement encore en cours. Chez les sols minéraux, ce sont les prairies artificielles qui présentent les teneurs en carbone les plus élevées et les pâturages d'alpage les plus basses. Les terrains d'alpages sont souvent caractérisés par les sols superficiels et pierreux, qui limitent le stockage du carbone.

La réserve totale en carbone s'élève à environ 170 mio. t de carbone. Les sols organiques (tourbières intactes est cultivés) contiennent environ 28% du total de carbone des sols agricoles, bien que leur surface ne représente que 37'000 ha. La plus grande part du carbone est stocké dans les sols minéraux sous des herbages permanents, qui se trouvent surtout sur le plateau. En raison de la grande surface, les réserves en carbone des terrains d'alpages sont également importantes (Figure S2).

Potentiels de séquestration de carbone

La séquestration de carbone est définie ici en tant qu'augmentation nette, spécifique à l'emplacement, du carbone dans le sol durant la période complète d'un assolement. Les variations de la teneur en carbone du sol durant l'assolement, qui peuvent être plus élevées pour une année isolée que l'augmentation ou la baisse nette moyenne annuelle, sont ainsi séparées du flux net pour la période complète d'observation. Les systèmes d'exploitation et les assolements utilisent les potentiels de liaison maximal plus ou moins bien. Le potentiel de liaison est définit ici comme la teneur en carbone maximale pouvant être obtenue dans un sol minéral avec une utilisation agricole habituelle. Une couverture permanente du sol (par ex. avec de l'herbage), une productivité élevée (provoque un flux élevé de carbone dans le sol), un travail du sol minimal (réduit les pertes oxidatives de carbone) et des conditions limitant la dégradation du carbone dans le sol favorisent des teneurs croissantes ou élevées de carbone dans le sol. A l'encontre, les grandes cultures avec un travail du sol intensif, des phases courtes de couverture du sol et peu de résidus de récolte mènent à des teneurs en carbone réduites ou basses. Les effets des différents modes d'exploitation sur les teneurs en carbone des sols sont présentés de manière schématique dans la figure S3. La séquestration de carbone dans les sols minéraux est toujours un processus limité par le potentiel de liaison du sol et qui par principe est réversible. Une modification de l'exploitation de sols ayant des teneurs en carbone élevées peut ainsi potentiellement conduire à des pertes de carbone.

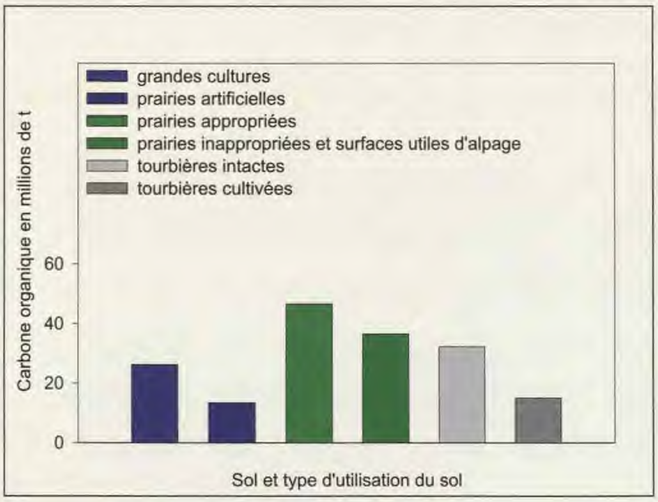

Figure S2. Réserves totales de carbone pour différents types d'utilisation du sol et de différents sols en Suisse.

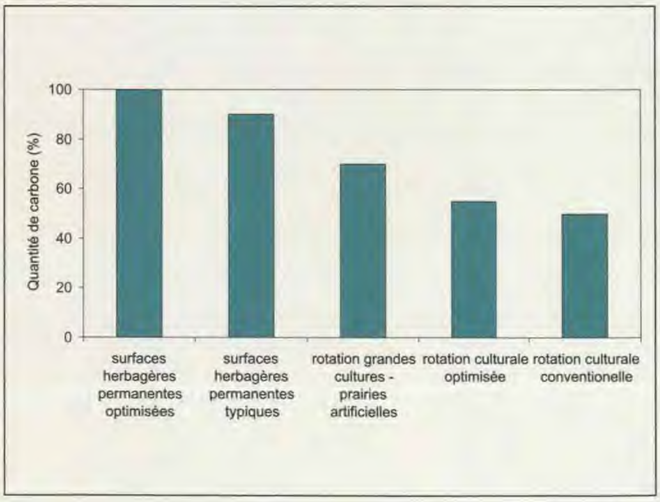

Figure S3. Représentation schématique du stockage du carbone de différents types d'utilisation du sol et de différents modes d'exploitation.

Dans cette étude, les émissions évitables des sols organiques cultivés sont également comptées comme potentiels de séquestration, car de telles activités permettent de ralentir l'augmentation du CO_2 atmosphérique.

Pour l'agriculture suisse, diverses activités ont été identifiées, qui peuvent conduire à une séquestration du carbone ou à une réduction des pertes oxidatives de carbone. A l'heure actuelle, il est uniquement possible de quantifier la baisse actuelle pour le semis direct, qui est pratiqué sur des surfaces relativement petites. Des potentiels théoriques ont été calculés pour les activités suivantes:

- Transformation de la totalité des surfaces de grandes cultures en surfaces de semis directs;
- Transformation de la totalité des surfaces de grandes cultures en herbages permanents;
- Transformation des anciennes tourbières cultivées en herbages permanents;
- Restauration de toutes les tourbières cultivées.

Les taux annuels **théoriques** de séquestration calculés ou ceux des émissions évitées (tourbières cultivées) sont représentés dans la figure S4. La quantification comporte des incertitudes considérables. Il serait possible au total de séquestrer en moyenne 1.1 mio. t de CO_2 par an en combinant de manière optimale les mesures. Cela correspond environ à 27% de l'engagement suisse à Kyoto pour la réduction du CO_2 ou à 21% des émissions annuelles de protoxyde d'azote et de méthane de l'agriculture. Il faut souligner que l'utilisation maximale de ce potentiel, qui pourrait être obtenu en transformant toutes les surfaces de grandes cultures en surfaces d'herbages et en restaurant toutes les tourbières, nécessiteraient une modification drastique des structures agricoles. Un potentiel légèrement réduit pourrait être obtenu en combinant la transformation de toutes les surfaces de grandes cultures labourées en surfaces à semis direct avec la restauration des tourbières cultivées. La restauration des tourbières cultivées provoquerait vraisemblablement des émissions de méthane nettement plus élevées, ce qui compenserait l'effet de baisse pendant quelques décennies ou siècles. Un scénario réaliste devrait se situer quelque part entre le potentiel maximal et la séquestration actuelle par le semis direct d'environ 0.003 Mio. t C par année.

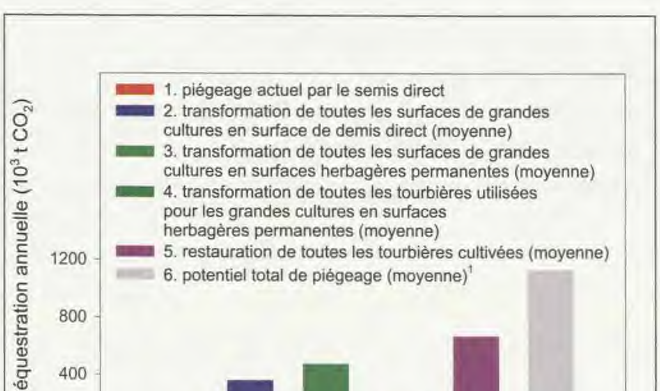

Figure S4. Potentiel de séquestration annuelle du carbone dans l'agriculture suisse. (Les fourchettes et les incertitudes sont décrites dans le rapport).

1. La baisse potentielle totale est constituée des activités 3 et 5.

Comparaison nationale et internationale

La figure S5 compare le potentiel de réduction moyen réalisable avec les flux d'autres gaz à effet de serre en Suisse. La réduction théorique dans l'agriculture est nettement plus petite que les émissions de protoxyde d'azote et de méthane de l'agriculture (année 2000) et également nettement plus petite que le piégeage biologique des forêts (moyenne 1990-1999). Mais les mesures décrites, comme par ex. la combinaison d'une augmentation du semis directe avec la restauration de tourbières cultivées, pourraient tout de même contribuer à améliorer le bilan global.

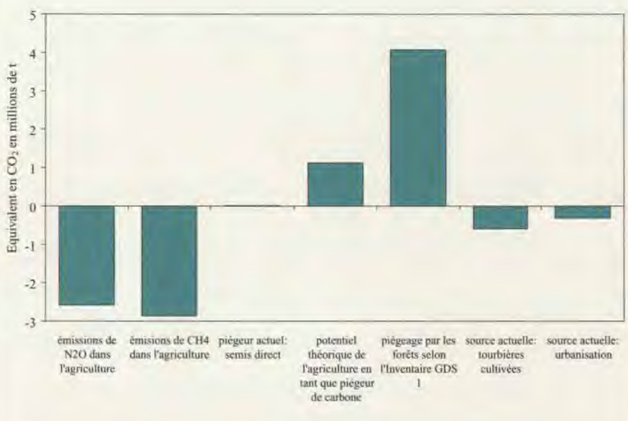

Figure S5. Potentiel annuel de séquestration de carbone dans l'agriculture en comparaison à d'autres sources et piégeurs biologiques de gaz à effet de serre en Suisse.

1) Moyenne de la période 1990 – 1999 (inventaire suisse des gaz à effet de serre), dont 1,8 million de tonnes au maximum ayant trait à l'exploitation forestière et au reboisement imputables dans les engagements du Protocole de Kyoto.

Le potentiel de réduction respectivement d'évitement de l'agriculture suisse est, en ce qui concerne les émissions brut du pays, comparativement petit si on les confronte aux données d'autres études de l'UE et des EUA (Figure S6).

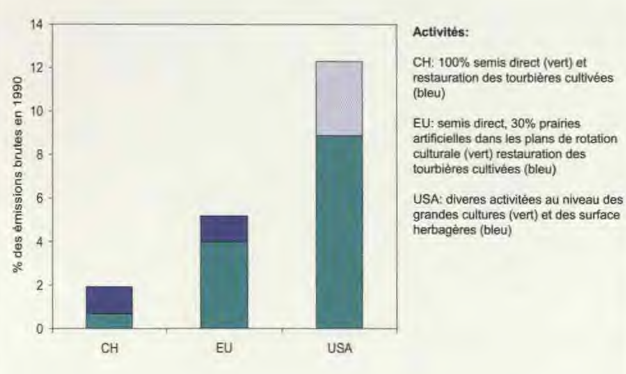

Figure S6. Potentiels de séquestration de l'agriculture en Suisse, dans l'UE et des EUA relativement aux émissions brutes de gaz à effet de serre dans les années 90.

La comparaison des potentiels des différents pays faite dans l'étude démontre clairement que la structure agricole actuelle de la Suisse protège considérablement les réserves en carbone existantes (à l'exception des tourbières cultivées) et qu'elle évite de trop grandes pertes de carbone des sols minéraux. Les principales différences quant aux réductions respectivement évitements entre les pays ou groupe de pays cités sont:

- La structure agricole (taux de prairies artificielles, de terres ouvertes, d'herbages permanents);
- L'histoire de l'utilisation du sol (par ex. emplacements fortement appauvrie en C dans le moyen ouest des EUA);
- Les conditions de l'espace naturel (climat, topographie, types de sol).

Conclusions

Le relevé et la quantification des réserves de carbone dans les sols agricoles a permis de faire ressortir des valeurs régulatrices essentielles de la liaison et du stockage du carbone et de confirmer en grande partie les indications d'autres régions. Les données calculées forment une base spécifique au pays des réserves en carbone et sont ainsi une condition importante pour l'estimation de possibles piégeurs biologiques. En comparaison avec d'autres pays, par contre, la base de données est assez petite pour une estimation et les informations concernant les caractéristiques du sol et les catégories d'utilisation du sol sont assez grossières.

En raison de la structure agricole, caractérisée par un taux élevé en prairies artificielles, en herbages permanents et par un taux de plus de 90% de production intégrée, le potentiel de séquestration maximal est, aussi bien par rapport aux émissions nationales de protoxyde d'azote et de méthane de l'agriculture que vis-à-vis du potentiel de baisse d'autres pays, comparativement petit. Un élément essentiel de la stratégie pour éviter les émissions provenant de l'agriculture serait la restauration des tourbières cultivées. Mais cela doit encore être élaboré plus en détail.

Finalement, il faut retenir que la condition pour utiliser au mieux le potentiel de réduction respectivement d'évitement calculé dans cette étude – s'élevant à environ 21% des émissions de protoxyde d'azote et de méthane de l'agriculture – serait une modification importante des structures agricoles suisses.

1. Introduction

The use of fossil fuels, together with changes in land use over the past 200 years, is driving increases in atmospheric concentrations of carbon dioxide (CO_2) and other greenhouse gases (GHGs), including methane (CH_4) and nitrous oxide (N_2O) (IPCC 1996a). There is a strong consensus among atmospheric scientists that the continued increase in atmospheric concentrations of GHGs will enhance the earth's natural greenhouse effect and thus lead to rapid global warming (IPCC 1996a).

It has been suggested that by enhancing terrestrial uptake of CO_2 over the next 50–100 years the 150 or more gigatons[5] of carbon (C) lost to the atmosphere from vegetation and soils as a consequence of land-use changes could be reclaimed (Batjes 1999; Lal *et al.* 1997a), thus effectively "buying time" for the development and implementation of new, longer-term technological solutions (e.g. the reduction of fossil fuel use through the development of carbon-free fuels). However, given our inadequate understanding of the biogeochemical mechanisms responsible for C fluxes and storage, and of the complex physiological processes controlling key biological and ecological phenomena (Metting *et al.* 2001), the potential for terrestrial carbon sequestration (C sequestration) remains uncertain. Specifically, the structure and dynamics of the below-ground component of terrestrial carbon pools, which accounts for about two-thirds of global terrestrial organic C stocks, is not well understood. But despite these uncertainties, it is agreed that agricultural soils have the potential to sequester additional carbon (IPCC 2000). Because of the political implications of the issue, there is an urgent need to quantify the carbon stocks and the sequestration potentials of various terrestrial ecosystems in relation to their management.

1.1 Greenhouse gas emissions and global climate change

Climate change refers to long-term alterations in temperature, precipitation, wind, and other elements of the Earth's climate system. Natural processes such as variations of the intensity of solar radiation, deviation in the Earth's orbital parameters, or volcanic activity can cause fluctuations in climate (IPCC 1996a). The climate system is also influenced by the concentrations of various atmospheric gases, some of which contribute to the phenomenon of global warming. These gases are called "greenhouse gases" (GHGs). GHGs can absorb long-wave infrared radiation and restrict absorbed heat from reflecting back to outer space. Global warming resulting from the effects of GHGs is important because without it, it is estimated that the Earth's mean temperature would be about 34°C lower (IPCC 1996a). The main naturally occurring GHGs are water vapour, carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). Since about 1750, the concentrations of CO_2, CH_4, and N_2O in the atmosphere have increased by 30%, 145%, and 13%, respectively (IPCC 2001a). This raises scientific and public concern that global temperatures may increase at a more rapid rate than ecosystems can adjust to.

[5] 1 gigaton (Gt) = 10^3 megatons (Mt) = 10^9 tons (t)

To enable direct comparisons between the various GHGs, their radiative properties are normalised to i) their atmospheric lifetime (average amount of time the GHG is present in the atmosphere after release) and ii) the radiative forcing of CO_2. The resulting Global Warming Potentials (GWPs) of 1 kg N_2O and 1 kg CH_4 for a 100-year time horizon are equivalent to 310 and 21 kg CO_2, respectively (IPCC 1996a). With CO_2 as the reference gas, emissions of each GHG are given as CO_2-equivalents.

Long-term measurements of atmospheric CO_2 concentrations show that they have increased primarily during the latter half of the 20th century (Keeling and Whorf 2000). After concentrations had remained near 200–290 ppm for a period of several ten thousand years, this increase coincided with a rapid rise in fossil fuel burning. Records from drilled Antarctic ice cores show that these values are unprecedented in the last 420,000 years (Petit et al. 1999). Although there is no consensus as to how much change will occur in the future, there is general agreement that it is necessary to reduce GHG emissions, or the rate at which they are increasing. From 1850 to 1998, approximately 270 Gt C was emitted into the atmosphere from fossil fuel burning and cement production, and another 136 Gt as a result of land-use changes, predominantly from forest ecosystems (IPCC 2001a). Emissions of CH_4 and N_2O are also influenced by land use, land-use changes, and forestry activities. Examples of relevant activities in the area of land use and land-use change include biomass burning, which currently occurs at high rates in the tropics, agricultural land management, and wetland management. Agricultural activities contributing to CO_2 emissions include fossil fuel combustion, agrochemical production, soil erosion processes and the decomposition of native soil organic matter (SOM).

1.2 Biological carbon sinks

Terrestrial ecosystems, in which carbon is retained in living biomass and soil organic matter (SOM), play an important role in the global carbon cycle. Carbon is exchanged naturally between ecosystems and the atmosphere through photosynthesis, respiration, decomposition, combustion, erosion and leaching. Human activities change carbon stocks (C stocks) in various pools and also affect exchanges between these pools and the atmosphere. Substantial amounts of carbon have been released by forest clearances at high and mid-latitudes in recent centuries, and in the tropics towards the end of the 20th century.

Terrestrial ecosystems exhibit a net carbon uptake into both vegetation and soils. Current C stocks are much larger in soils than in vegetation, particularly in arable and grassland ecosystems at mid- and high latitudes (IPCC 2000). Vegetation biomass can be divided into above-ground and below-ground biomass, and soil C stocks can be classified into organic layers (mainly in forest soils), peat layers (wetland soils), and mineral soils, which are composed of several, in some cases unidentifiable, chemical constituents.

As regards the fate of carbon after its assimilation by photosynthesis, only a very small fraction ends up in the soil over long periods in refractory forms such as charcoal or recalcitrant humus ("net biome productivity", Fig. 1).

So far, only about half of the CO_2 released from fossil fuel combustion can be accounted for as atmospheric CO_2. Global carbon budgets indicate that a significant portion of the additional CO_2 emitted to the atmosphere as a result of human activities must be absorbed by a large terrestrial sink (Schimel 1995) – the so-called missing sink (Table 1).

Fig. 1. Model of the fate of carbon after its assimilation by terrestrial ecosystems, showing the different levels of carbon productivity and loss, carbon flux density, and the time scales of the various productivity levels.

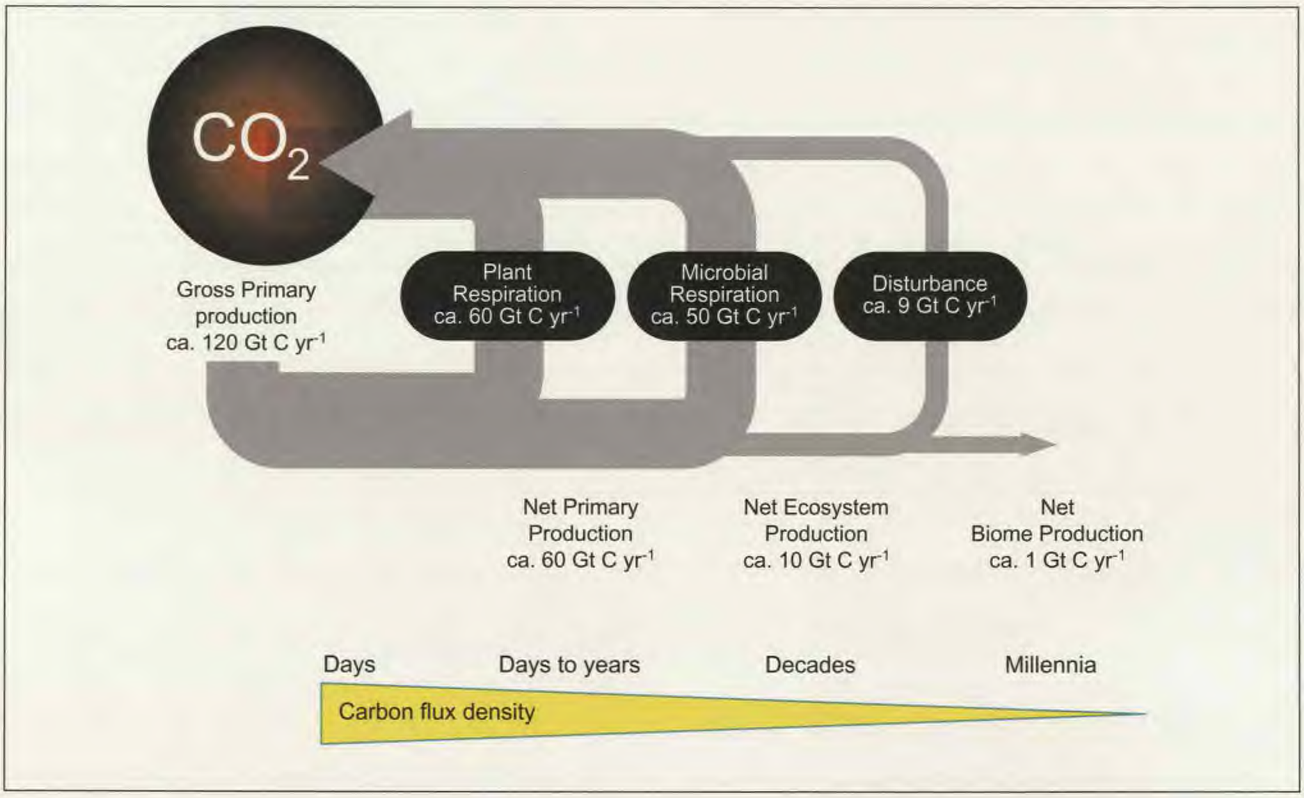

Table 1. Average annual global CO_2 budget for 1980–1989 and for 1989–1998 (in Gt C a^{-1}) (± 90% confidence interval) Source: IPCC (2000).

		1980 to 1989	1989 to 1998
(1)	Fossil fuel combustion and cement production	5.5 ± 0.5	6.3 ± 0.6
(2)	Storage in the atmosphere	3.3 ± 0.2	3.3 ± 0.2
(3)	Ocean uptake	2.0 ± 0.8	2.3 ± 0.8
(4)	Net terrestrial uptake = (1)-[(2)+(3)]	0.2 ± 1.0	0.7 ± 1.0
(5)	Emissions from land-use change	1.7 ± 0.8	1.6 ± 0.8
(6)	"Missing sink": Residual terrestrial uptake = (4)+(5)	1.9 ± 1.3	2.3 ± 1.3

The residual terrestrial uptake (6) compensates for the difference between Σ (1) + (5) and Σ (2) + (3). Thus, the missing sink is a numerical difference between individual values in Table 1. Which C pools may contribute to this missing sink? There is evidence that the activity and the length of the growing season of terrestrial vegetation respond to shifts in temperature (Braswell *et al.* 1997) and to increased atmospheric CO_2 concentrations (Schimel 1995). Increasing plant productivity might result in higher biomass C stocks and higher inputs of plant residues into soils, with a subsequent accumulation of SOM. The suggestion is that the C sink is the result of a disequilibrium between increasing primary production and the respiration of litter and soil carbon pools, which lags behind by some years to decades; the net balance remains positive as long as atmospheric CO_2 continues to grow. Due to the high protective capacity of soils, either mineral or organic, which serves to reduce carbon turnover rates, a long-term sink of additionally sequestered carbon can be expected (e.g. Buyanovsky and Wagner 1998; Lal *et al.* 1997a). However, it is expected that rising global temperatures will counterbalance the higher residue C inputs by increasing turnover rates (e.g. Rounsevell *et al.* 1999; Kirschbaum 1995), although results concerning the response of SOM turnover to temperature remain equivocal (Liski *et al.* 1999).

1.3 Kyoto Protocol and Swiss commitments

In 1992, following the recognition of global warming during the 1980s, the United Nations adopted a Framework Convention on Climate Change (UNFCCC) in Rio de Janeiro (United Nations 1992). The aim of the Convention is the 'stabilisation of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system'. In December 1997, the Parties to the Convention met in Kyoto, Japan, and drafted a Protocol which imposes binding limits on emissions, in order to start the process leading to the stabilisation of the atmospheric concentrations of GHGs (Kyoto Protocol, United Nations 1997). It was recognised that the objectives of the Kyoto Protocol can be met either by decreasing the rate at which GHGs are emitted to the atmosphere or by increasing the rate at which they are removed from it. The Kyoto Protocol established the concept of accounting for carbon sinks and defined a limited number of accepted 'human-induced activities'. This list was expanded and the details were specified during subsequent negotiations of the Conference of the Parties (COP).

For the first commitment period of the Kyoto Protocol, between 2008 and 2012, Switzerland agreed to reduce its mean annual GHG emissions by 8% compared with the base year of 1990. Total emissions (CO_2-£equivalents) in 1990 were 53'232 Gg CO_2, i.e. 14'518 Gt CO_2-C or 14.5 Mt C (BUWAL 2002)[6], [7]. The reduction target is therefore set at 0.08 x 53.232 Gg CO_2 = 4.259 Gg CO_2 a^{-1} in the commitment period, or 1.16 Mt C. To what extent C sequestration by terrestrial sinks will be taken into account in efforts to meet the target remains an open question. However, any decision must be based on the best available knowledge concerning current C stocks and the potential of terrestrial ecosystems to sequester additional carbon under a range of realistic management scenarios.

[6] 1 t C * 3.67 = t CO_2

[7] BUWAL: Swiss Agency for the Environment, Forests and Landscape

1.4 Scope and objectives of the study

The aim of the present study was to quantify soil carbon stocks and to assess the most relevant land-use types and cultivation practices for carbon sequestration in Switzerland. This report summarises the results in the following main areas:

- Quantification of C stocks in Swiss agricultural soils
- Determination of the main factors influencing C stocks in Swiss agricultural soils
- Definition of the most relevant potential sequestration activities for agricultural soils in Switzerland
- Assessment of agricultural C sequestration potentials including trade-offs, considering Swiss-specific data and land-use scenarios
- Evaluation of uncertainties

The study focused on organic carbon only, since maximum accumulation rates for pedogenic inorganic carbon produced by biogenic precipitation, mainly as $CaCO_3$, are at least two orders of magnitude below organic C sequestration rates (Monger and Martinez-Rios 2000), and the formation of pedogenic carbonate is restricted to areas with an annual rainfall of less than 500–700 mm, accompanied by periodic droughts enabling carbonate precipitation.

The Kyoto Protocol calls for the determination of soil C stocks in 1990 and for estimates of changes in C stocks in subsequent years (Article 3.4). A reliable quantification of C stocks provides the basis for estimates of the sequestration potential and, conversely, of the vulnerability of the system in terms of potential carbon losses. Soil C pools and rates of change can be determined by extrapolation from relationships developed on the basis of data from plot and field measurements, or by means of geostatistical techniques. In this study, correlations between measured C stocks and abiotic and biotic site factors – including past and current land use, soil properties, climate, and altitude – were used as the main accounting method.

Carbon sequestration is the net long-term increase in biomass and soil carbon. Thus, measuring C stock changes over time or carbon flux balances yields the sequestration rate of a particular system. Estimates for C sequestration rates for agricultural practices in Switzerland were derived from literature data and from calculations involving Swiss soil and land-use data.

For a comprehensive evaluation, it must be considered that concurrent fluxes of N_2O and CH_4 are also strongly affected by agricultural practices. This may lead to a net GHG balance which differs in magnitude or sign from the net C flux. Consequently, the net GHG balance needs to be assessed for any management adopted or changed. Additional fluxes that play a role in the assessment of agricultural practices are energy costs deriving from fossil fuel use. The energy costs may offset any economic benefit of the improved net carbon balance.

A complete GHG budget for agriculture as a whole would thus include the following elements:

- Emission and sequestration of soil carbon
- Emission and consumption of N_2O and CH_4

- CO_2-equivalents produced by consumption of fossil fuels for land management, production and transport of fertilisers, lime, etc.
- Integrated farm-level GHG budgets could be elaborated by means of life-cycle assessments.

The scope of the present study is more narrowly defined. It includes only a quantitative estimate of net carbon fluxes (i.e. sequestration or emissions) for a set of human-induced activities as proposed by IPCC (2000). The activities considered are selected according to Swiss agricultural conditions. The study also includes a qualitative assessment of possible trade-offs caused by consumption and/or emission of N_2O and CH_4 in cases where data were available. Overall, the scope of the study includes the biotic and abiotic aspects of C sequestration, whereas accounting for CO_2-equivalents consumed by the use of fossil fuels for land management, fertilisers, etc. is outside its scope.

The quantification of C stocks and C sequestration potentials as developed in Chapters 4 and 5 is discussed against the background of i) indirect human-induced and natural effects, e.g. CO_2 fertilisation, and ii) the Swiss national greenhouse gas budget. These topics are addressed in Chapters 6 and 8, respectively. Based on the assessment of sequestration potentials, agricultural and political recommendations are formulated and summarised in Chapter 9.

2. Conceptual framework of the Kyoto Protocol and the IPCC approach

2.1 Human-induced activities according to Article 3.4, and their carbon sequestration potential

The Kyoto Protocol allows for "additional human-induced activities related to changes in greenhouse gas emissions by sources and removals by sinks in the agricultural soils and the land-use change and forestry categories [to be] added to, or subtracted from, the assigned amounts [of reduction commitments]" (Article 3.4). The agricultural activities outlined in this study refer to this article of the Protocol. The Kyoto Protocol specifies that carbon accounting is to consider only the land area subject to "direct human-induced" (Article 3.3) or "human-induced" (Article 3.4) activities, but these terms are not clearly distinguished. Article 3.3 refers to "afforestation, reforestation, and deforestation" and is thus outside the scope of this study. In this Chapter, possible definitions of human-induced activities are discussed in accordance with the proposals of the IPCC (2000), together with the associated C sequestration potentials.

Human-induced activities can be defined in a **broad** (e.g. management of all cropland) or **narrow** sense (e.g. tillage practices on a particular piece of land). For both definitions, either **land-based** or **activity-based** accounting systems are possible (see 2.2). The difficulty with the term "human-induced" is the distinction between anthropogenic and non-anthropogenic sources and sinks. C stocks change over temporal and spatial scales for different reasons. Even if C stock changes could be measured with sufficient accuracy, attributing a given change to a particular cause can be highly challenging. Yet identifying the cause of an observed change in C stocks may be crucial to deciding whether or not it should be included in the accounting system.

At the global scale, C stocks maintain a natural balance between uptake and release due to the annual cycle of plant growth and residue decay. Natural climate variations such as the El Niño Southern Oscillation have resulted in lower-than-normal uptake of atmospheric CO_2 by the terrestrial biosphere and the oceans during 3 El Niño events since 1980 (Keeling and Whorf 1999). At the regional scale, other natural factors could affect C stock changes over a 5-year commitment period. These include natural cycles of disturbance and recovery, and occasional disasters like flooding or drought. Some of the C stock will be destroyed by such factors and subsequently be recreated. Besides natural variations, soil C stocks are also significantly affected by indirect human-induced effects (see Chapter 6). Hence, it is difficult, if not impossible, to distinguish by direct measurement between natural changes in C stocks and those which are directly or indirectly human-induced.

To overcome these limitations, the distinction between direct and indirect effects can be made by comparing experiments in which different activities were applied under the same environmental conditions (baseline versus activity). This may involve, for instance, long-term agricultural experiments with a single- or multi-factorial design, providing key information on driving factors for soil C stocks (Paul *et al.* 1997). Among the human-induced activities proposed by IPCC (IPCC 2000), four categories are presumably of relevance for Swiss agriculture (Table 2). Most agricultural activities have an estimated annual C sequestration potential of between 0.1 and 1 t C ha^{-1}. On the basis of global data, rates for some activities (e.g. species introduction to marginal grasslands) are higher, but the activities are of minor importance for Switzerland. Importantly, emissions occurring from drained organic soils are about one order of magnitude larger than potential sinks in mineral soils.

Table 2. Potential rates of carbon gains and losses (-) (t C ha^{-1} a^{-1}) for human-induced activities that are potentially relevant for Swiss agriculture (IPCC 2000).

Activity	Key practice	Rate	Time interval[2] (years)	Confidence[3]
Cropland management	Conservation tillage	0.34	50–100	H
	Fertilisation, crop rotation, organic amendments	0.1 to 0.3		H
	Incorporation of biosolids, manure, straw, etc.	0.2 to 1.0	50–100	M
	Ley-arable farming	0.54	100	M
	Forages in rotation	0.3	37	M
Grazing land management	Improved management[1]	0.22	40	M
	Increased productivity[1]	0.51		M
	Species introduction[1]:			
	Legumes	1.09		
	Grasses	3.34		
Conversion of agricultural land	Conversion of arable to permanent grassland	0.5 to 1.0	50	M
	Set-asides	0.52	50	M
Wetland management	Conversion to agriculture	-1 to –19	>100	M
	Wetland restoration	0.1 to 1.0	>100	M

[1] Global estimates, otherwise data for temperate regions

[2] Time interval to which estimated rate applies

[3] Relative confidence: H = high, M = medium, L = low

2.2 Mechanisms of carbon accounting proposed by IPCC

A carbon accounting system records, summarises, and reports the quantity of carbon emissions by sources and removals by sinks due to given land use, land-use change, and forestry activities for a specific time period. It is thus different from full carbon accounting, which would involve accounting for changes in C stocks across all carbon pools in a given time period, irrespective of the cause of the observed changes (other than direct human-induced or human-induced activities). "The mere presence of C stocks is excluded from accounting"[8]. In a first approximation, carbon accounting over an inventory period of *n* years would include net changes in soil C stocks based on soil types and land-use systems (IPCC 1996b). Carbon stocks are calculated for any combination of soil type and land-use category at time *t* (using default values or country-specific data) and are multiplied by the area concerned. The resulting stock is then subtracted from the stock after $(t+n)$ years, to obtain the net CO_2-C flux. Separate calculations are performed for organic soils, using default values for CO_2 emissions.

A carbon accounting system should provide "transparent, consistent, comparable, complete, accurate, verifiable, and efficient recording and reporting of changes in C stocks and/or changes in greenhouse gas emissions by sources and removals by sinks from applicable land use, land-use change, and forestry activities" (IPCC 2000). Two modalities for carbon accounting are possible:

- **Land-based accounting**
- **Activity-based accounting**

Both methods rely on activities under Articles 3.3 and 3.4 of the Kyoto Protocol which are implemented on land units. Land-based accounting first involves the definition of applicable activities and the identification of land units on which these activities occur. Next, the total change in C stocks on these land units during the commitment period is determined. The resulting change is the sum over land units during the commitment period. With activity-based accounting, the impact of applicable activities on C stocks is determined per unit of time and area. The starting point is the change in C stocks attributable to the designated activities. This impact is then multiplied by the area on which the activity occurs, and by the number of years of application, or the number of years of the commitment period. The resulting change is the sum over activities during the commitment period. Thus, with activity-based accounting, a given area of land could potentially be counted more than once if it is subject to multiple activities. Double-counting could result in inaccurate accounting if the effects of activities are not additive. Stock changes prior to the start of the activity would not be accounted for in activity-based accounting, even if they occurred during a commitment period.

Additional aspects of carbon accounting systems to be considered concern modifications regarding:

- **Baselines** (a reference to changes in stocks and/or emissions not induced by the activity itself),
- **Leakage** (the indirect impact that an activity at a certain place and time has on carbon storage at another place or time),

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[8] UNFCCC FCCC/CP/2001/13/Add.1; 21 January 2002

- **Permanence** (the longevity and stability of C stocks for a given environmental and management set-up), and
- **Uncertainties** (measurement uncertainty, uncertainty in defining baselines, definition of key terms).

COP7 requested IPCC to develop definitions for some of these key terms, and also to specify the details of the accounting systems. While some of these definitions are still outstanding, other open questions have been resolved:

- The emissions by sources and removals by sinks resulting from Article 3.4 activities in the base year need to be accounted for in the first commitment period: "For the first commitment period, accountable anthropogenic greenhouse gas emissions by sources and removals by sinks resulting from [Article 3.4 activities] shall be equal to anthropogenic greenhouse gas emissions by sources and removals by sinks in the commitment period, less five times the emissions by sources and removals by sinks resulting from these eligible activities in the base year" (UNFCCC Marrakesh Accords[9]).
- Questions concerning good practice guidance (quality control and quality assurance), quantification of un certainties, and verification issues for national GHG inventory systems are treated by IPCC (2001b). However, the topic of verification of agricultural CO_2 sinks and sources is not addressed by IPCC (2001b).

2.3 Monitoring and verification of changes in soil organic carbon stocks

The measurement of management impacts on the accumulation and loss rates of SOC presents many challenges. In designing methods to monitor and project changes in soil C stocks, it is essential to consider the temporal and spatial heterogeneity of soil properties, environmental conditions, and management history. As pointed out by Post *et al.* (2001), there is an "urgent need to develop robust, science-based, flexible, and practical tools for monitoring and verifying temporal changes in soil C". Several methods exist to quantify C stocks and carbon fluxes:

Direct methods include the measurement of SOC concentration and soil bulk density to quantify C stocks per unit area. For this method, the content of stones or coarse fragments >2 mm, horizon thickness, and the equivalent soil mass rather than soil volume must be taken into account. The heterogeneity of SOC and its dynamic nature usually prevent direct detection of changes on annual or finer time scales. In spite of the large annual fluxes associated with primary production and respiration processes, the net SOC changes are small relative to the large C stock present in soil. If, for example, an annual sequestration of 1% is to be detected over a 5-year period, and a coefficient of variation of 10% is further assumed for the organic C content due to spatial heterogeneity, a total of 62 samples (31 for each sampling date) will be necessary to detect a significant increase with a 90% confidence interval (one-tailed t - test). If this number of samples is multiplied by the num-

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[9] UNFCCC FCCC/CP/2001/13/Add.1; 21 January 2002

ber of sites and selected activities, sampling and analytical costs for verification purposes may exceed any economic value of C sequestration by a large margin. However, after a sufficient length of time (5–10 years), sta-tistically significant differences in soil organic C have been observed in experiments (Paul *et al.* 1997). To reduce the uncertainty of projections of future changes in soil C stocks due to human-induced activities, analysis of SOM fractions with higher sensitivities to management may be a better option (see 3.3).

Instead of measuring C stocks, fluxes of CO_2 between the soil and the atmosphere can be determined. The vertical component of turbulent air movement over a vegetated surface (eddies) can be isolated and quantitatively measured, as can CO_2 concentrations associated with each eddy. By correlating eddy size and CO_2 concentration for each upward- and downward-moving eddy (eddy covariance), it is possible to calculate the net uptake or release of C from soil and vegetation – the net ecosystem carbon exchange. The integration of fluxes over longer periods of time yields an estimate of net changes in ecosystem C. The net ecosystem exchange can be considered to have two components: changes in soil C stocks and changes in vegetation C. Changes in vegetation C stocks are generally easier to measure directly, and therefore changes in soil organic C are usually calculated as the difference between net ecosystem exchange and change in vegetation C. Eddy covariance flux measurements may improve our understanding of changes in C fluxes in time scales of less than one year and will therefore be useful in conjunction with the direct sampling and measurement of SOC.

Indirect methods include the determination of soil C pools and rates of C change from extrapolation of relationships between C flux and other biotic or abiotic variables developed at the plot and field scale, or the use of process models. The accounting methods include correlations between C stocks and abiotic and biotic factors, such as land use, soil properties, climate, and altitude. Estimates for each patch with similar management and ecological conditions are combined to yield a regional estimate. This approach was used here to determine soil C stocks. Availability of complete and detailed spatial and geographical information increases the accuracy of this approach. Furthermore, analysis of the sensitivity of C measurements to various environmental factors should help to decide whether the accuracy of spatial data layers is adequate or needs improvement. Remote sensing can be used in particular for regions where detailed geographical information is lacking.

Process models can be used to describe SOM dynamics and ecosystem processes. Such models have been used to project changes in SOC through time (e.g. Parton *et al.* 1995). Since the amount and the rates of SOC changes depend on previous management, the site history must be known for realistic estimates. Heterogeneity attributable to variations in initial conditions must also be considered. If information on management and climatic history is incomplete, the definition of adequate initial conditions for the model is difficult, and trajectories of C stocks may be miscalculated.

Verification of SOC changes must include estimates based on incomplete knowledge of some fluxes and/or some temporal interpolation from long-term trends derived from experiments which may not directly apply to the specific situation. Thus, monitoring and verification of changes in C stocks will have to rely to a certain extent on information which cannot readily be obtained through observations.

3. Carbon storage and carbon sequestration in soils: Mechanisms and controls

The genesis of SOC is a major process of soil formation. The amount of carbon stored under an area of land to a defined depth is referred to as the C stock. This stock is subject to several processes and may be at equilibrium (input = output), or in a transitional state. Carbon sequestration is a term commonly used to describe the process of net organic carbon (OC) accumulation in the terrestrial biomass and in soils. Because of the transient nature and the relatively small amount of OC in plant biomass (see 3.1), attention is focused on SOC as the most important aspect of C sequestration in agricultural ecosystems. However, in the case of energy crops to substitute for fossil fuels, plant biomass plays a significant role in the overall agricultural C budget.

The term **C sequestration** in soil and climate change research implies an enrichment of SOC, which is intended to be persistent. Hereafter, C sequestration is defined as the site- and management-specific net increase in the amount of SOC, *dSOC/dt*. Since C stocks fluctuate during the year and during a management cycle, C sequestration is "any increase in mean C stocks over the entire course of the management cycle" (Paustian *et al.* 1997).

The C stock (C) under steady-state conditions is defined as

$$C = OC_i(t) / r_t \qquad [\mathrm{t\ ha^{-1}}] \qquad [1]$$

where OC_i is the organic carbon input [t ha^{-1} a^{-1}] into the soil compartment at any given time (t), and r_t is the turnover rate [a^{-1}], and for steady-state or equilibrium conditions,

$$OC_i = OC_o \qquad [\mathrm{t\ ha^{-1}\ a^{-1}}] \qquad [2]$$

where OC_o = organic carbon output from the soil compartment.

It follows that changes in the C stock (either an increase, i.e. sequestration, or a depletion) always occur under non-equilibrium conditions. Such conditions can be achieved either by changing OC_i / OC_o, or r_t. If $OC_o > OC_i$, the C stock will decrease. If OC_i / OC_o, or r_t are altered for an extended period of time, then the C stock will reach a new equilibrium after years, decades or in some cases only after centuries. In practice, management will most often alter the input, e.g. by modified crop rotations, or the turnover rate, e.g. by changing the quality (decomposability) of the residue input. Following this outline, it is important to recognise that C sequestration is not an infinite process, but is limited by site-specific environmental and anthropogenic conditions.

The relationship between input, output, and turnover rate is the basis for any C sequestration (or depletion) activity in agriculture.

In fact, real equilibria of soil C stocks are never reached, as soil formation is a continuously ongoing process. However, in the context of short- to medium-term C sequestration, such changes in soil conditions can be neglected. Apart from the accumulation or depletion of SOC, C stocks are effectively influenced by soil transport processes such as **erosion** and **leaching,** which both contribute to a higher OC_o. Erosion is often counterbalanced by the deposition of the eroded material where slopes level off. This accumulation of the eroded material may even lead to a net accumulation, if the conditions for carbon turnover are less favourable at the deposition site. In each case, the borders of the system must be defined to characterise the net flux of the system.

3.1 Carbon stock and carbon turnover in agroecosystems

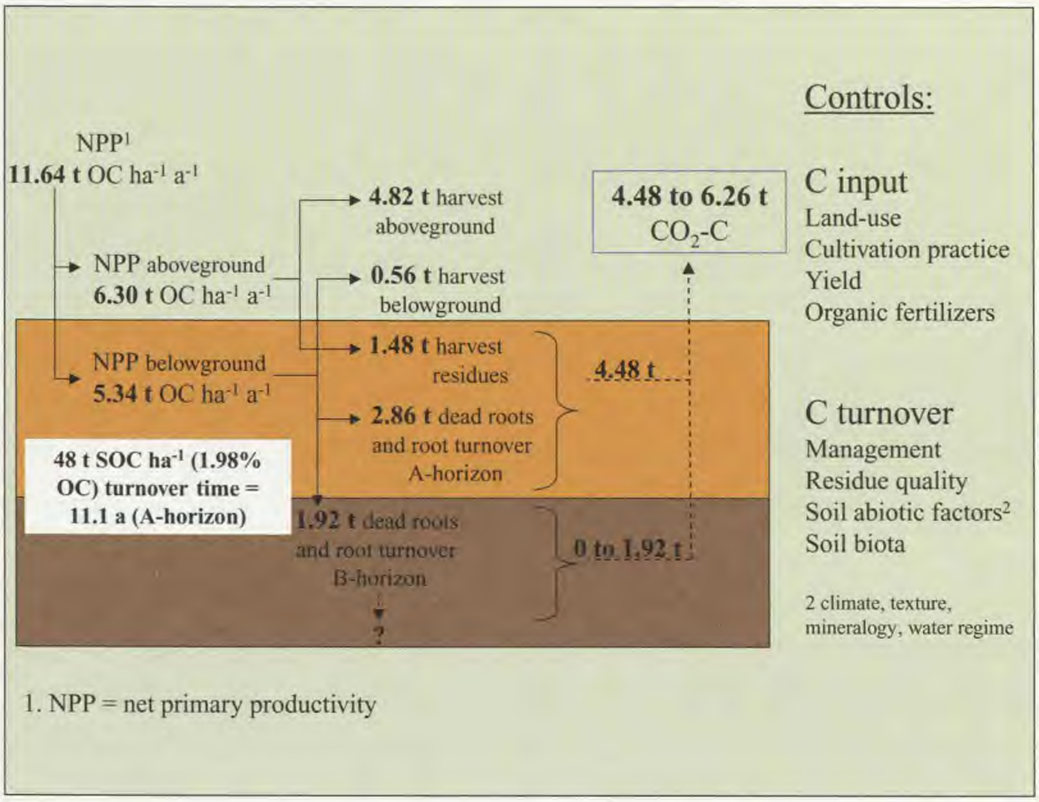
Management practices such as crop rotation, soil tillage, and fertilisation affect the carbon input and turnover in arable cropping systems. Similarly, the soil C stock of temporary or permanent grasslands is governed by grazing and mowing, or by any combination of the two practices.

The main fraction of the C stock is contained in dead SOM (93 to 94%) and soil micro-organisms (1 to 2%), with plant biomass contributing less than 5%. Carbon from root litter and root exudates represents a major soil input in arable cropping systems. In addition, a significant portion of the above-ground biomass is transferred to the soil as non-harvested plant residues, and as organic fertilisers excreted by the livestock. The ratio of roots to shoots and of harvested to non-harvested plant materials varies between crop types. The amount of plant material remaining on the field also depends on the farmers' management decision. For example, the cropping of corn for silage will leave distinctly smaller amounts of residues on the field as compared with exclusive harvesting of the cobs.

Depending on the type of crop, type of fertiliser (organic or mineral), fertiliser application technique, and tillage method, the input material is distributed between the soil surface and several soil horizons. Generally, carbon inputs to the soil decrease exponentially with depth and are in line with above-ground litter input and rooting density. If the carbon input and the soil C stock are known, turnover times for soil organic carbon can be calculated (Fig. 2). Although the values in Fig. 2 are based on yield measurements only, while below-ground inputs are calculated using SOM modelling and literature data, they give a first approximation of fluxes and stocks for a typical Swiss arable rotation.

The data in Fig. 2 illustrate how management decisions affect soil C sequestration by altering the amount and localisation of residue inputs. But management can also affect conditions for SOM turnover. Again, this concerns crop rotation and residue quality, but also soil tillage and fertilisation. Each management option for C sequestration is constrained by additional factors, mainly abiotic conditions for plant growth and SOM turnover.

Fig. 2. Annual carbon fluxes for a 7-field crop rotation with mean yields on the Swiss Central Plateau (winter wheat, corn, potato, winter wheat, corn, summer barley, 2 x temporary grassland). Soil input and vertical distribution was estimated from model runs with CENTURY (Parton *et al.* 1994) and literature data, and from root:shoot ratios and root profiles of the corresponding crops.



3.2 Soil organic matter (SOM) formation and stabilisation

The amount and composition of SOM in the soil profile is determined by the interaction of several abiotic, biotic and anthropogenic factors. Since the processes of formation and stabilisation influence carbon turnover in mineral and organic soils in different ways, these two soil types are discussed separately.

Mineral soils are defined by SOM contents not exceeding 30% (18% OC by weight) in any of the upper soil horizons (FAO 1998). The five major drivers of SOM **formation** are (in descending order of importance) (Baldock and Nelson 2000):

Land use > climate > soil texture and mineralogy > topography > time

Land use affects the amount, composition and localisation of the organic input, i.e. plant residues, root exudates and organic fertilizers. Climate provides the basis for physical constraints on biological activity, i.e. temperature and water supply. Soil texture affects soil structure and provides the habitat for soil organisms. Soil mineralogy is of fundamental importance for the nutrient status of a soil. Together with soil texture, it provides the reactive surface of the soil matrix, which plays a major role in SOM stabilisation processes. Soil mineralogy and climate also have a major effect on weathering and soil chemical properties, which in turn affect soil biological activity. Topography includes micro-meteorological factors, as well as hydrological and textural gradients. Time is a major driving force since SOM accumulation and depletion often require time scales of years, decades or even centuries.

Plant residues entering the soil are subject to several processes of degradation and turnover, resulting in the formation of SOM. The rate constants of these processes depend on site conditions. Two major processes can be distinguished – mineralisation and humification of OM. Mineralisation involves the depolymerisation of complex organic compounds into simple inorganic compounds, such as CO_2, H_2O, or NH_3. Most of this transformation is effected by microorganisms. Humification is the transformation of plant or microbial biomass into humic substances, which are relatively resistant to microbial decomposition. Significant amounts of SOM can be classified neither as plant/microbial tissue nor as humic substances, but are in a transitional stage.

Stabilisation of OM is an important prerequisite for keeping C stocks at a desired level. The relative recalcitrance[10] of humic substances contributes to this effect. Improved understanding of the mechanisms leading to the formation of humic substances would help to improve the design of C sequestration activities. Despite the long tradition of research on humic substances, soil science is still a long way from attaining this goal, since even the definition of the structure of humic substances is still under discussion (Hayes and Clapp 2001). Besides humic substances, refractory macromolecules of an aliphatic nature (cutans and suberans), polyphenols and proteins such as glomalin also contribute to soil C sequestration (Derenne and Largeau 2001). Another important source of recalcitrant SOM is black carbon deriving from incomplete combustion during fires and from anthropogenic deposition (Swift 2001). Although forest and other fires result in the injection of CO_2 into the atmosphere, the simultaneous production of charcoal is a sequestration process, resulting in the retention of substantial amounts of carbon in the soil for long periods. The amount retained can be manipulated by human activities.

Another major mechanism of SOM stabilisation is physical protection, which occurs at the micro-level, e.g. through adsorption of SOM on mineral surfaces. At higher levels, clustering of mineral and organic compounds with biota, known as aggregation, protects SOM against degradation by reducing accessibility for microorganisms and enzymes. Aggregation can be managed by considering site-specific conditions such as soil texture. Many human-induced activities resulting in agricultural C sequestration promote soil aggregation by reducing soil disturbance and by extending the periods during which the soil is covered with plants or layers of mulch.

The turnover time of the various fractions of SOM – including degraded plant residues, microorganisms and their by-products, humic substances, and associations of various organic molecules with the mineral matrix – ranges from hours to millennia. Thus, based on carbon and nutrient turnover rates, SOM is often classified into virtual (physical and chemical) fractions or (mathematical) pools distinguished by turnover times. In general, fast-cycling pools react immediately to management changes, whereas the measurable response of intermediate and stabilized pools has a time scale of years to decades. A long-term strategy to increase SOM content should target the intermediate and stable OM fractions, which make up the major part of SOM. The determination of fractions from the active and intermediate pools may thus reveal changes in soil C stocks before any

[10] resistance to microbial decomposition

significant changes in bulk SOC have occurred. Analysis of various physical fractions revealed a wide range of mean residence times, the highest for non-occluded plant fragments in the sand fraction and the lowest for OM in microaggregates or for OM associated with finer particles of higher specific density (Table 3).

Table 3. Comparison of mean residence times (MRT; in years) of C in various virtual SOM pools and in soil physical fractions (adapted from Collins *et al.* 1997).

Pool	Definition I[1]	MRT	Definition II[2]	MRT	Physical fractions[3]	MRT
I	Decomposable plant material	0.24	Metabolic plant residues	0.5	Plant fragments >0.2 mm	0.5–1
II	Resistant plant material	3.33	Structural plant residues	3.0	Plant fragments >0.053 mm	1–2
					Plant fragments <0.053 mm	2–3
					Macroaggregates 2–1 mm	1–4
III	Soil biomass	2.44	Active soil C	1.5–10	Aggregates 1–0.5 mm	2–10
					Aggregates 0.5–0.1mm	3–10
					Non-aggregated soil	7
IV	Physically stabilized	72	Slow soil C	25–50	Fine silt	400
V	Chemically stabilized	2857	Passive soil C	1000–1500	Fine clay	1000

[1] Jenkinson and Rayner (1977)

[2] Parton *et al.* (1988)

[3] Buyanovsky *et al.* (1994)

Functional and structural attributes of SOM that have also been used to indicate the beginning of shifts in soil C stocks include mineralisable C, extractable C, carbohydrates and soil enzymes (Gregorich *et al.* 1994). But it should be noted that while the determination of such parameters may be useful in identifying changes in C stocks, it cannot in the current state of knowledge replace measurements designed to quantify carbon for accounting purposes. Although some fractions reveal higher sensitivity to management than others, the accumulation of C does not account for the total change in SOC since even small variations in the stable fraction may be quantitatively more important, given the size of this fraction.

Organic soils are characterised either by the presence of an organic horizon (≥ 18% OC) at least 10 cm thick extending from the surface to a lithic or paralithic contact, or an organic horizon at least 40 cm thick starting within 30 cm of the soil surface (FAO 1998). Plant roots are adapted to the anoxic conditions, and plant productivity can be higher than in intensively used mineral soils (Succow 1999). The C stock of organic soils may exceed 2000 t ha^{-1} (Bergkamp and Orlando 1999). Oxygen is the limiting factor for residue decomposition. Oxygen limitation results from a high water table (reducing the oxygen supply because of the distinctly lower diffusivity of O_2 in water) and oxygen depletion due to residue decomposition. The imbalance between plant residue carbon input and carbon output by peat mineralisation results in the accumulation of organic carbon; in some regions, this occurred throughout the Holocene. Thus, it is noteworthy that in contrast to mineral soils, the carbon storage capacity of peatlands might be inexhaustible, at least over historical time scales.

In contrast to mineral soils, OM of intact[11] organic soils (peatlands) often exhibits a low degree of decomposition, as quantified by the fraction of identifiable plant residues. This material provides a good carbon source for microbial decomposition when oxygen limitation is reversed by lowering the groundwater table during cultivation. In contrast to fens[12], nutrient deficiencies in ombrotrophic raised bogs[13] may reduce peat decomposition rates under cultivation, but the peat body remains exposed to enhanced decomposition.

3.3 Influence of human-induced activities on carbon sequestration

3.3.1 Mineral soils

From the list of factors determining SOM formation, it is apparent that only land use and cultivation practices are under direct human influence. Other factors are either independent of human activities (soil texture, mineralogy, topography and time) or subject to indirect human influence (climate). Land use influences SOM formation through the amount and characteristics of the input material, and by affecting the conditions for OM turnover. Changes in cultivation practices induce a shift in soil C stock, until a new equilibrium is reached. Active and intermediate SOM pools respond more quickly to management and contribute preferentially to changes in C stocks (Fig. 3). The final stock again depends on site-specific conditions.

Two major agricultural land-use systems can be distinguished:

- **Arable cropping,** with various crop rotations and tillage intensities (including no-till)
- **Permanent grasslands** with various levels of management intensity, i.e. amount and type of fertiliser application, cutting frequency, and stocking density of pastures.

With respect to C sequestration, one of the most crucial aspects of **arable cropping** on mineral soils is **tillage.** Most studies show that tilled soils have lower C stocks than non-tilled soils (see 5.3.1), even after correcting for changes in bulk density by making the calculation for equivalent soil depths (Ellert and Bettany 1995). A number of studies of C sequestration under no-till management showed that biological aspects of aggregate formation, which are closely related to the intermediate SOM pool (Fig. 3), are of key importance. The physical occlusion or "protection" of OM, leading to restricted accessibility for microorganisms and enzymes, is an important mechanism controlling SOM turnover. On the basis of a conceptual model proposed by Six *et al.* (1999,

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[11] peatlands without or with little disturbance by peat harvesting and/or drainage

[12] peatlands which intersect the groundwater table. Fens receive water that passed through mineral soils, often nutrient-rich

[13] peatlands supplied by precipitation only (ombrotrophic), often nutrient-poor

2000), major mechanisms of carbon accumulation under no-till are outlined in Fig. 4. These equally apply to grasslands.

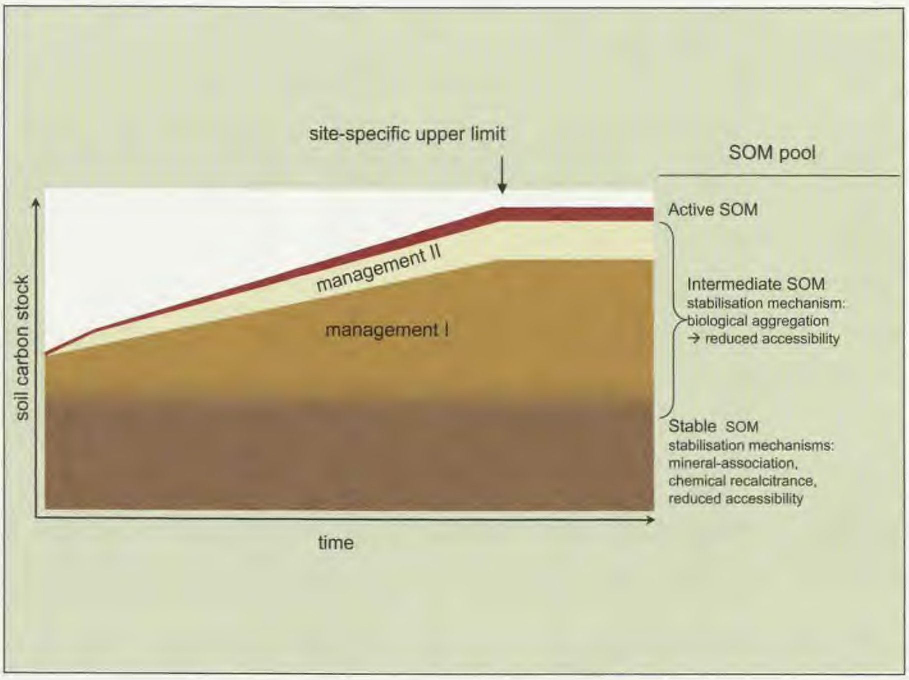


Fig. 3. Management-induced changes in SOM pools. Pools are characterised by their turnover rates and stabilisation mechanisms. The soil carbon stock shifts to a new equilibrium between input and output as a result of changes in input and/or turnover rates. Major changes are expected for the intermediate carbon pool. Changes in stable SOM are disregarded for the time horizon of human-induced activities (several decades).

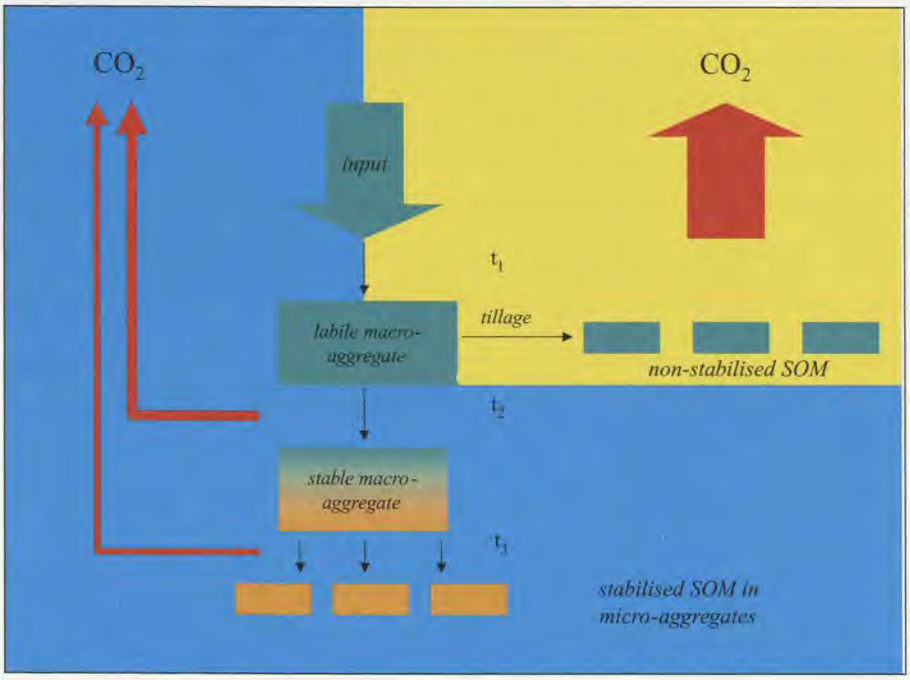


Fig. 4. Simplified life cycle of soil aggregates for no-till and grassland soils (*blue background*), and for tilled soils (*yellow background*). No-till and grassland soils show three time steps (t_1 - t_3) of aggregate formation, resulting in stabilised SOM in micro-aggregates. The life cycle is cut short for tilled soils, where early macro-aggregate disruption causes enhanced decomposition and less formation of micro-aggregates and stabilised SOM.

In this model, plant residue inputs act as a nucleus for the formation of macroaggregates through enhanced biological activity in the immediate proximity of the residue particles. During subsequent degradation of these particulate residues (particulate organic matter, POM), microaggregates containing smaller units of POM are formed. Under no-till, the relative proportion of protective microaggregates, as well as the amount of SOM stored within macro- and microaggregates, is about twice as high as in tilled soils. Tillage disrupts young macroaggregates, prevents the formation of new microaggregates, leads to depletion of microaggregates and enhances aggregate turnover. Thus, the protective capacity of aggregates – a major driving force in physical stabilisation – is significantly reduced. OM is then exposed in well-aerated sites within the soil matrix. Conversely, reduced soil disturbance leads to shifts in the microbial community. Guggenberger *et al.* (1999) detected significant differences in the ratio of fungi-derived to bacterial-derived organic compounds for till vs no-till soils. The authors suggested that less soil disturbance promotes the accrual of hyphal cell walls, which contribute to the formation of SOM. Also, no-till soils have lower temperatures due to insulation by the mulching layer covering the surface (Doran *et al.* 1998). The mulching layer itself decomposes more slowly due to drier conditions and reduced contact with soil microorganisms, which again slows the turnover of SOM. However, less soil disturbance and stronger aggregation under mulch may be accompanied by higher soil moisture, lower spring temperatures and the potential for higher denitrification when climatic conditions are cool-humid. Such differences in soil physical properties may result in lower yields for sites with high precipitation in some Swiss regions (e.g. for corn; Anken *et al.* 2003).

The concept of C sequestration being the result of better protection of soil aggregates can be extended to **permanent grasslands**. In general, permanent grasslands show higher C stocks than croplands. Again, the determinants of the C stock are residue input and conditions for turnover. In addition to the high protective capacity of the undisturbed soil, higher annual residue inputs from roots, together with a different residue quality, lead to the higher soil C stocks found in the soil of permanent grasslands. Mechanically dispersible clay was found to be indicative for the carbon-dependent structural stability at the micro scale for arable-grassland sequences in South Germany (Leifeld and Kögel-Knabner, 2003).

Effects of arable cropping and grassland management on C sequestration are mainly attributed to residue input and carbon turnover in the topsoil, but modification of the carbon storage in deeper soil horizons may also be possible. The long turnover time is related not only to physical protection or chemically more resistant fractions, but also to substrate placement in the deeper soil, which is often characterised by limiting conditions (aeration, nutrients) for substrate turnover and by a higher protective capacity. Thus, the cultivation of deep-rooting annual or perennial species can significantly enhance carbon storage in subsoils, as documented for South American savannahs (Fisher *et al.* 1994).

3.3.2 Organic soils

The five factors involved in SOM formation are the same for organic soils, with carbon storage being additionally influenced by hydrology. Since oxygen limitation is the major prerequisite for peat accumulation, manipulations of the water table alter the conditions for SOM turnover in the peat body. Often, the water table reaches the surface, but a lowering is necessary for intensive agricultural use. As a consequence, peatland drainage, together with other cultivation measures, leads to excessive oxidation of peat. Peat decomposition follows the stepwise decline of the water table until the SOM content reaches the level typically found in mineral soils.

Maintaining a high water table under agricultural management, e.g. in extensive grassland use, is one measure that can be taken to reduce peat oxidation.

Natural peatlands are among the most important soil carbon reservoirs and active carbon sinks worldwide (Bergkamp and Orlando 1999). Cultivation of peatlands not only stops the carbon accumulation of the intact ecosystem, but also produces a flush of CO_2 within relatively short periods. Therefore, peatlands are key ecosystems in the discussion of C sequestration.

4. Soil organic carbon stocks in agricultural soils

4.1 Aim of the estimation of carbon stocks

Knowledge of current carbon stocks in various types of agricultural soil is a key element of any carbon accounting system, and for the evaluation of carbon sequestration potentials. The present estimate is based on statistical data for land use, soil type, and topography, together with soil data taken from several soil surveys. This is an extension of the earlier evaluation of OC in Swiss forests and agriculture undertaken by Paulsen (1995). The approach adopted involves elements proposed by IPCC (1996b), such as (i) to include land use and soil types to quantify C fluxes in the agricultural sector, and (ii) to treat mineral and organic soils separately. This approach is extended to include climate as a third factor affecting soil C stocks. Using these refined and Swiss-specific data reduces the uncertainty for estimates of sequestration potentials.

By combining information from different sources, it was possible to consider quantitatively three main drivers of SOM formation:

- Land use
- Soil type
- Climate

4.2 Methodological approach

4.2.1 Area and land-use data

The boundary of the system is given by the agricultural land area. Soil profile depth is considered to **1 m** for mineral and **2 m** for organic soils. This differentiation takes into account the fact that the main C stock in mineral soils is located in the uppermost decimetres, whereas the C stock of organic soils is limited by the total soil depth, which is typically 2 m or less.

Area-based information was available for land use, for soil classes (without information on SOC), for topography, and for the area and spatial distribution of peatland. The total land area under agriculture and its distribution among different land-use types is calculated from statistical data in Table 4.

Organic soils include intact and cultivated peatlands[14], both belonging to the same pedological unit (histosols). Intact peatlands are further divided into (raised) bogs, which are not in contact with the groundwater table, and fens, which exhibit groundwater contact or even partial water logging. The total area of intact peatlands is given by the three peatland inventories[15], which together cover 25'745 ha for fens and another 1'490 ha for bogs (data provided by BUWAL in 2001). Fens <1 ha are not included in the inventories. The actual fen

[14] "intact" designates peatlands without major disturbance of the peat body by peat harvesting and/or drainage and natural or semi-natural vegetation; "cultivated" designates peatlands with visible peat degradation induced by drainage, peat harvesting, and agriculture.

[15] National and Regional Inventory of Fens, National Inventory of Raised Bogs (Nationales und regionales Flachmoorinventar, Nationales Hochmoorinventar)

area might be about 40% higher if the smallest fens were taken into account (Dalang and Fischbacher 1992). About 19›400 ha of fens and 700 ha of bogs are used agriculturally. Fens are mainly used as traditional litter meadows, with one cut in autumn and/or some grazing (Grünig 1994). About one-third of the raised and transition bogs are undisturbed, and the rest are partially degraded by human activities (Grünig *et al.* 1986). For the calculation of C stocks of intact peatland under agriculture, the area of intact peatland was subtracted from the total agricultural area, taking into account their distribution by altitude (Fig. 5).

Table 4. Area (ha) and distribution of agricultural land among different land-use types

Land use[1]	Area	Source of data	Comment
Open arable land	293’947	Swiss Federal Statistical Office	Area in 1999
Temporary grassland	115’933	Swiss Federal Statistical Office	Area in 1999
Favourable permanent grassland	518’908	Swiss Land-Use Statistics 1992/97	Area in 1992/97
Unfavourable permanent grassland	58’530	Swiss Land-Use Statistics 1992/97	Area in 1992/97
Mountain farming	537’801	Swiss Land-Use Statistics 1992/97	Area in 1992/97
Total	1‘525’119		

[1] Fruit trees and vineyards are classified as permanent grassland; favourable grassland = area classes 81, 82 minus arable and temporary grassland; unfavourable grassland = area classes 83, 84; mountain farming = area classes 85–89 according to Swiss land-use statistics.

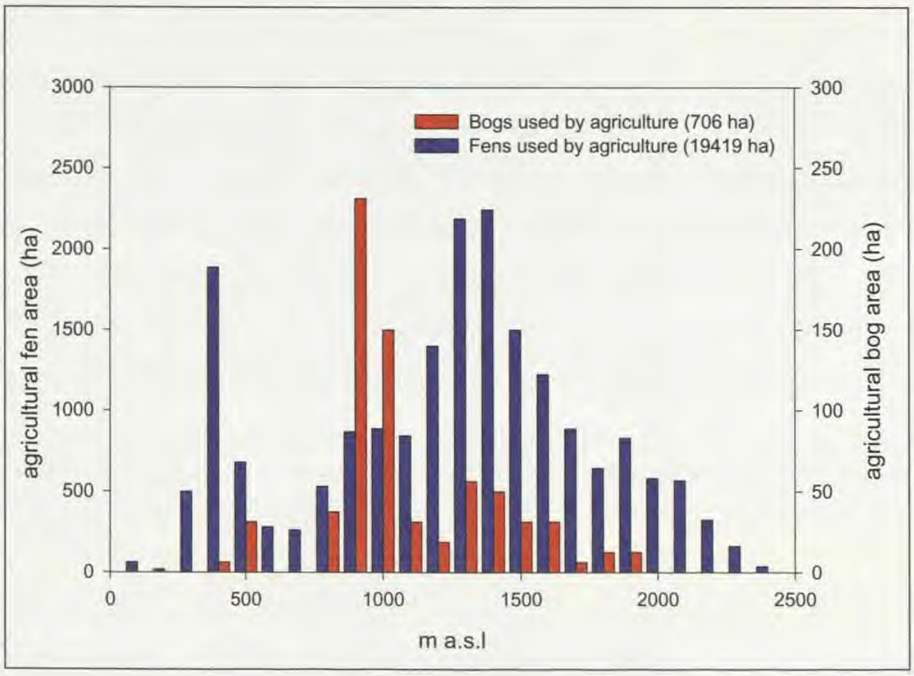


Fig. 5. Distribution of agriculturally used fens and bogs by altitude.

The area of cultivated peatlands which to some extent is utilized intensively by agriculture was estimated from historical and recent surveys. This estimate is more uncertain than the calculation of the area of intact peatlands on the basis of the inventories. The most reliable estimate was obtained by combining information from different sources after checking for consistency (Table 5).

Table 5. Overview of methods used to estimate the current area (ha) of cultivated peatlands. The mean area calculated is 17'000 ha, and the lower and upper calculated areas are 12'000 and 22'000 ha, respectively.

Data source	Approach	Area (ha)	Comment	Reference
Peatland inventories (fens), inventory of cultivated organic soils	Extrapolation of size distribution of peatland sites to cultivated organic soils	18'000	Considers organic soils <50 ha excluded from the inventory of organic soils	BUWAL; Presler and Gysi 1989
Inventory of cultivated organic soils, historical peatland survey	Extrapolation of distribution of organic soils to main natural regions with previous peatland distribution	15'000	Includes regions (Jura Mountains, Pre-Alps, Alps) other than Swiss Central Plateau	Presler and Gysi 1989; Früh and Schröter 1904
Historical peatland survey	Extrapolation of surveyed peatland objects (mean weighted area) to all peatland sites described	22'000	High uncertainty due to unknown ratio of peatland to total peatland area	Früh and Schröter 1904
Soil map (Canton of Zürich), digital soil map	Extrapolation of the proportion of organic soils in the detailed map of histosols to the digital soil map for the whole of Switzerland	12'000	Analysis shows low suitability of digital soil map for estimation of organic soils	Digital soil map; detailed soil map Canton of Zürich
Inventory of cultivated organic soils, estimate of C stock, digital soil map	Extrapolation of two crucial mapping units to the whole of Switzerland	19'000	High uncertainty due to dependence on digital soil map	Paulsen 1995; Presler und Gysi 1989; Digital soil map

Estimates of the area of cultivated organic soils range from 12'000 to 22'000 ha, with a mean of 17'000 ha. For the calculation of C stocks and carbon fluxes, mean, lower and upper values of the range were used. Previous figures for the area of cultivated peatlands ranged from 6'400 ha (Presler and Gysi 1989) to as much as 180'000 ha (Grünig 1994; based on documented melioration activities in Switzerland since 1885, as given by the Eidgenössisches Meliorationsamt Bern, 1954). The figure of 6'400 ha is considered an underestimate, since it includes only sites located on the Central Plateau with a minimum area of 50 ha, and which are used for intensive agriculture. On the other hand,180'000 ha is probably an overestimate of the actual area of cultivated peatlands, since the melioration activities on which it is based included drainage of non-organic soils, e.g. gleysols, or soils with only a shallow organic horizon. The maximum area of agricultural organic soils given by the digital soil map is 127'000 ha which is also regarded as an overestimate because it includes non-organic soils in the same soil classes.

Estimates of the spatial allocation of cultivated peatlands were carried out by comparing the altitudinal distribution of peatlands in the historical survey by Früh and Schröter with the corresponding distribution of histosol soil classes in the digital soil map (Fig. 6). The altitudinal distribution of former intact peatlands, which are considered to represent a significant fraction of current cultivated peatlands, is similar to that of the histosol soil classes. For the calculation of OC in cultivated peatlands, the area of cultivated peatlands was subtracted from the total agricultural area, taking into account the altitudinal distribution of histosols.

The resulting total area (in hectares) of mineral and organic soils was then calculated as follows:

$$ha_{tot} = ha_{mineral} + (ha_{peat} + ha_{org}) \qquad [3]$$

where ha_{tot} = total agricultural area; $ha_{mineral}$ = area of mineral soils under agriculture; ha_{peat} = area of intact peatlands under agriculture and ha_{org} = area of cultivated peatlands under agriculture.

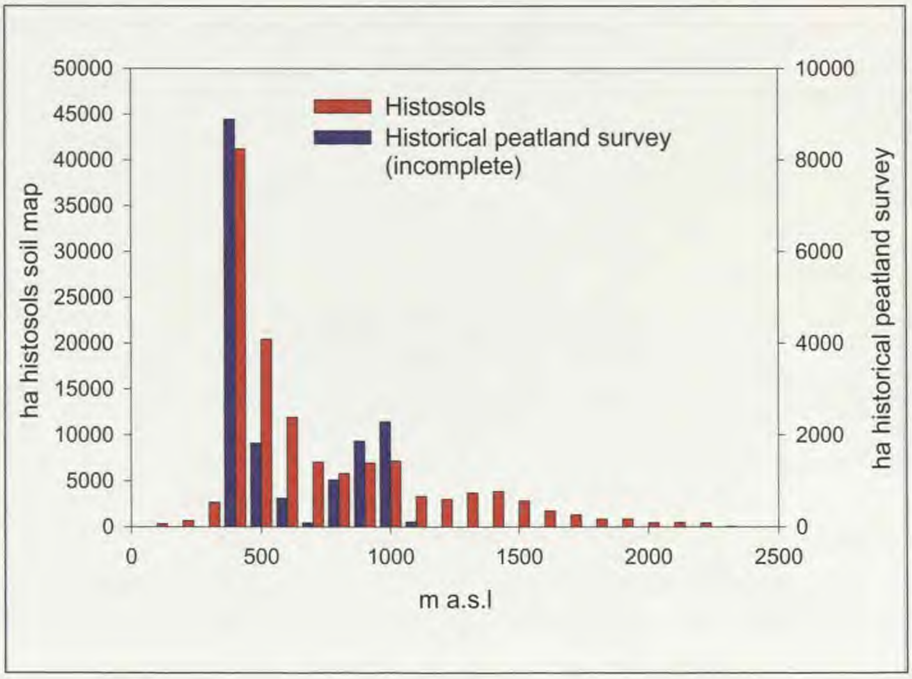


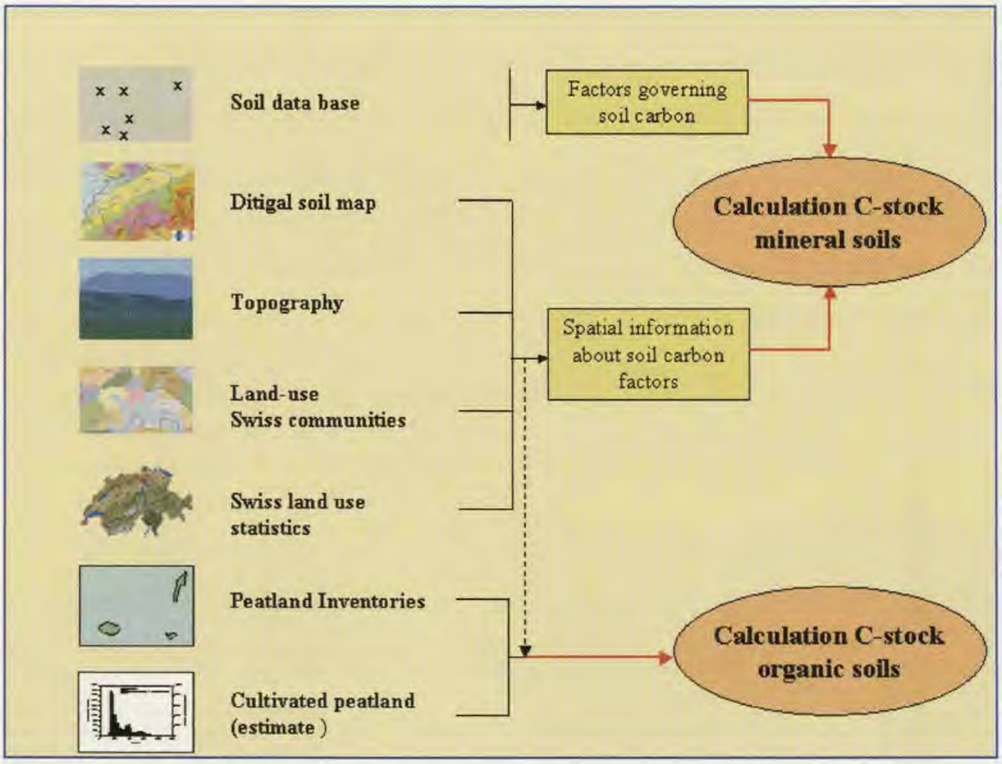
Fig. 6. Altitudinal distribution of current histosols according to the digital soil map and of peatlands according to the historical survey.

The land-use classification schemes employed in the Swiss statistical data bases were modified according to the needs of this study and to the available soil carbon data. Different grassland types, as given by the land-use statistics, could not be considered for the calculation of C stocks because the existing soil data made no distinction between them. Instead, permanent grasslands were distinguished according to soil and altitudinal classes (see below). This classification scheme also provides qualitative information on differences in C stocks between the various land-use types.

4.2.2 Soil carbon stocks

To calculate SOC stocks, data from profiles in mineral soils were combined with the existing spatial information (Fig. 7). Variables in the soil data base with a statistically significant relationship to SOC were identified and used as predictors. They were classified for the area-based information in order to obtain the spatial distribution of C stocks in agricultural soils. Amounts of SOC (0–20 cm) were calculated for 660 possible factorial combinations ("patches") of land use x soil characteristics x altitude for mineral soils, and multiplied by the frequency of the corresponding patches. Carbon stocks were calculated separately for peatlands.

Fig. 7. Overview of data sources and procedures used to estimate soil carbon stocks in mineral and organic soils.



The data sets which were combined for the estimate of SOC in **mineral soils** included:

- A non-georeferenced data base of 544 soil profiles with information on land use, site altitude, soil type, soil texture, OC, bulk density and, in some cases, additional physical and chemical soil properties. Organic carbon or humus contents[16] are given for each profile for 0–20 cm, while C contents for subsoil horizons and values of other soil variables are not complete for the overall data set. Data were taken from federal (BUWAL 1993, n = 73; NFP[17] 22, n = 94) and regional soil surveys (KABO[18] Bern, n = 72; KABO Aargau, n =

[16] humus content = organic carbon x 1.72

[17] Nationales Forschungsprogramm 22 NFP22, 1988. Mehrfachnutzung des Bodens in Übergangsbereichen.

[18] Kantonale Bodenbeobachtung (Cantonal Soil Monitoring)

24; KABO Zug, n = 113; Basel-Land[19], n = 84); from Swiss soil maps (Grindelwald and Davos, FAL[20], n = 73,), and from the literature (Zeller *et al.* 2001, n = 6; Jacot *et al.* 2000, n = 5).

- Georeferenced data for topography (DHM25, Federal Office of Topography 1994), land-use statistics[21] (Swiss Federal Statistical Office 2000) with three major land-use classes (without differentiation between arable and grassland), and 140 major soil classes from the Swiss digital soil map BEK[22] (Federal Office for Spatial Development 1980). Georeferenced data for BEK, peatland surveys, and land-use statistics were all converted from polygon data to a 250 x 250 m grid size. In general, data consistency between polygon and grid data was excellent (e.g. <2% deviation for the peatland surveys). The BEK soil classes, which subsume different soil types[23], are based on landscape units, including data on soil physical parameters, profile depths, and stone content for each soil class. Means for each class were used for the calculation of C stocks. Information on soil clay content was obtained by non-linear regression between 6 classes of field capacity (FC; m^3/m^3) in the BEK and corresponding clay contents (AG Boden, 1994) using the following relation ship:

$$Clay = 0.0093 \times \% FC^{2.19} \quad [\%]; \qquad R^2 = 0.99 \qquad [4]$$

- Non-georeferenced land-use data for all Swiss communities[24] providing detailed information for several types and intensities of arable and grassland use. The spatial resolution of the Swiss community data is determined by the boundary of each community. The land-use patches, as described above, were separated into 100-m increments for altitude. For each community, the relative proportion of each type of patch was then calculated to obtain the C stock. The resulting patches indicate the most likely distribution of arable, temporary grassland, and favourable permanent grassland among soil and altitude classes, but a spatial allocation with a resolution higher than that of the community boundaries cannot be attained.

  SOC contents in the soil data set revealed significant differences across land use, three classes of clay content and four classes of site altitude (Table 6).

[19] by Landwirtschaftliches Zentrum Ebenrain, unpublished data

[20] Swiss Federal Research Station for Agroecology and Agriculture (FAL), Zürich, unpublished data

[21] Arealstatistik 1992/97 (land-use statistics)

[22] Digitale Bodeneignungskarte (digital soil map)

[23] Each soil class comprises several soil types. Any particular soil type may be a subset of more than one soil class.

[24] Gemeindeerhebung (community statistics)

Table 6. SOC content (%) in the 0–20 cm layer for various classes of land use, soil texture, and altitude[1]

	Land use				% Clay		
	Arable	Temporary Grassland	Permanent Grassland		<20	20 ≤ 40	>40
			<1000 m a.s.l.	>1000 m a.s.l.			
Median	1.80a	2.33b	2.91c	5.35d	2.09a	2.73b	3.20c
Mean	1.98	2.48	3.09	6.10	3.10	3.27	5.43
S.E.	0.09	0.10	0.08	0.28	0.60	0.50	0.87
n	157	96	189	102	186	269	89

[1] Letters following values indicate statistically significant differences (P = 0.05) (Kruskal-Wallis Z-test).

To calculate SOC contents (0–20 cm) as a function of the main determinants, the following linear relationship was used for arable land and temporary grassland:

$$SOC = \%\ clay \times f + c \qquad [\%] \qquad [5]$$

Where $f = 4.60 \times 10^{-2}$ ($\pm 4.8 \times 10^{-3}$) and 4.90×10^{-2} ($\pm 8.1 \times 10^{-2}$), and $c = 0.75$ (± 0.13) and 0.92 (± 0.25), for arable and temporary grassland, respectively (mean values ± 95% C.I.[25]). Variability in clay content accounted for 36% (arable) and 27% (temporary grassland) of the variation in SOC.

For permanent grassland, a multiple linear regression describing SOC as a function of altitude and clay content accounted for 44% of the variation in SOC concentrations (0–20 cm):

$$SOC = altitude\ [\mathrm{m}] \times f_1 + \%\ clay \times f_2 + c \qquad [\%] \qquad [6]$$

where $f_1 = 2.38 \times 10^{-3}$ ($\pm 2.16 \times 10^{-4}$), $f_2 = 3.92 \times 10^{-2}$ ($\pm 0.98 \times 10^{-2}$) and $c = 0.38$ (± 0.39).

For SOC (0–5 cm), 53% of the variation was attributable to altitude alone. For Equations 5 and 6, the range of values calculated was not allowed to lie outside the range of SOC concentrations measured.

SOC concentrations derived by regression analysis were assigned to each patch. Corresponding soil bulk densities ρ_d for each class were calculated from the 544 soil profiles as:

$$\rho_d = 1.49 \times \%\ SOC^{-0.29} \qquad [\mathrm{g\ cm^{-3}}]; \qquad R^2 = 0.69 \qquad [7]$$

This relationship was used to calculate the C stock (C) in the 0–20 cm layer:

$$C = \%\ SOC \times 20 \times \rho_d \qquad [\mathrm{t\ ha^{-1}}] \qquad [8]$$

[25] Confidence interval

Subsoil layers (20–40, 40–60, 60–80 and 80–100 cm) also differed in SOC concentrations between land-use types. Permanent grasslands showed consistently higher SOC concentrations down to 1 m profile depth, compared with arable soils (Fig. 8). Median SOC contents and bulk densities of subsoil layers were used to calculate C stocks for each soil class in each of the three land-use systems. This simplified approach was used because

- the correlation between clay content or clay x altitude and SOC content was not significant for most of the combinations,
- the data base for subsoil layers was small for some of the land use x soil depths combinations, and
- the largest changes in SOC concentrations due to human-induced activities were expected to occur in the upper soil horizon.

Since for subsoil, it was necessary to take profile depths into account, a total of 90 SOC classes for possible land use x soil depth x stone content combinations were calculated. Carbon stocks (t ha^{-1}) for the subsoil layers (20-cm increments) were calculated as above and added to the stock of the topsoil.

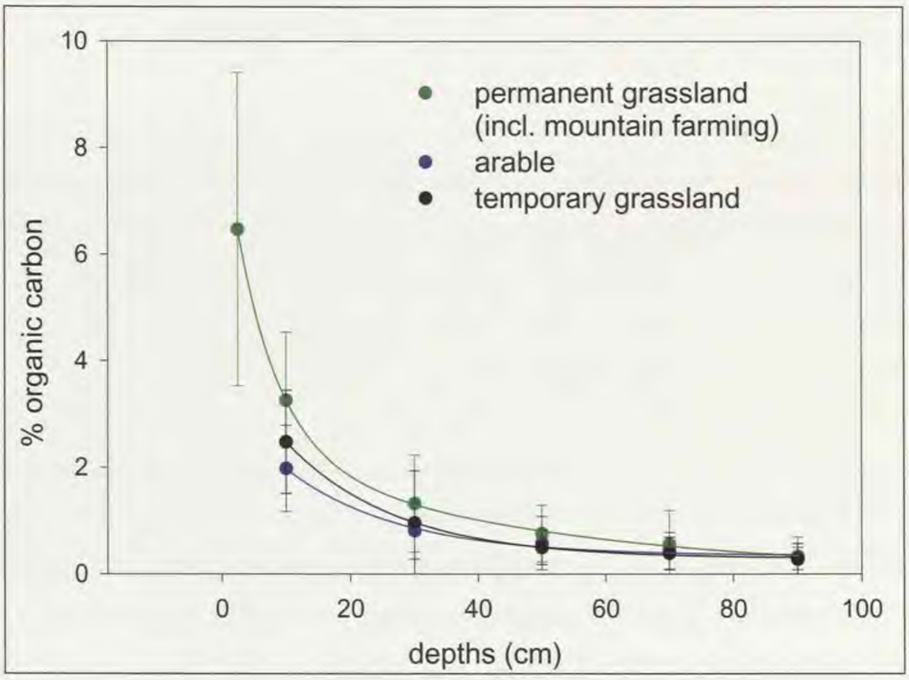


Fig. 8. Soil organic carbon (SOC) concentrations in fine earth for different land-use types (mean ± SD).

For comparison, C stocks were estimated on the basis of equivalent soil depths (Ellert and Bettany 1995), corresponding to the mean C stock stored in the upper 20 cm of arable land with a bulk density of 1.30 g cm^{-3}:

$$Dt = 0.2 \times (1.30 / \rho_{deq}) \quad [m] \quad [9]$$

with Dt = equivalent soil depth, and ρ_{deq} being the soil bulk density for each SOC concentration, as calculated in Equation 7. The comparison of SOC calculated for equivalent soil masses and for equal soil volumes (Equation 8) permitted an estimation of the uncertainty of the volumetric approach.

Estimating the amount of C stored in **peatlands** was difficult because of the often unknown thickness and bulk density of peat layers. A reasonable guess of the C stock of intact peatlands for the upper 2 m was based on Presler and Gysi (1989), who found bulk densities of about 0.2 g cm^{-3} in intact peat layers. Assuming a carbon content of 40% (Zeitz 1997) and a peat depth of 2 m, the C stock was estimated at 1600 t ha^{-1}. The assumed peat depth is relatively low, since weighted mean peat depth calculated from historical surveys (Früh and Schröter 1904) is 2.9 m. On the other hand, the peatland inventories mainly involve classification based on vegetation surveys and do not necessarily represent a soil classification for peatlands (minimum peat layer in the topsoil: 0.4 m). The overall error resulting from incorrect estimates of peat layer depth cannot be quantified.

A suitable methodology for cultivated peatlands should also provide information on net changes in SOC resulting from progressive melioration activities. For this estimate, the mean, upper, and lower area estimates in Table 5 were used as the initial area meliorated since 1885. The main melioration activities were carried out in two phases from 1885 to 1949 (Eidgenössisches Meliorationsamt Bern 1954), and continued less intensively after the Second World War (Grünig 1994). For the calculation of carbon remaining in cultivated peatlands, a simplified chronology of melioration activities was assumed, with a steady increase in area between 1885 and 1970. Expected annual losses of organic carbon due to peat oxidation were taken from literature data. References were only considered if they involved studies carried out under climatic conditions similar to those in the Central Plateau, the Jura Mountains and the Pre-Alps (Table 7).

Table 7. Oxidative peat losses (t OC ha^{-1} a^{-1}) estimated from literature data[1]

Mean peat loss	Upper 95% C.I.	Lower 95% C.I.
9.52	11.68	7.34

[1] Presler und Gysi, 1989; Kasimir-Klemedtsson *et al.,* 1997; Zeitz 1997.

Some of the literature data are based on subsidence rates rather than on oxidative losses. Subsidence and shrinkage occur at early stages after drainage and lead to an overestimation of oxidation rates if all of the subsidence is attributed to oxidation. Kasimir-Klemedtsson *et al.* (1997) considered the contribution of subsidence by taking a conversion factor of 0.7, as proposed by Eggelsmann (1976). The rates of peat loss calculated were similar to the values reported by Freibauer and Kaltschmitt (2001), i.e. 10 ± 5 t C ha^{-1} a^{-1} for grassland and 15 ± 5 t C ha^{-1} a^{-1} for arable cropping systems. Local observations in Witzwil (Switzerland) revealed an annual decrease in the organic layer of about 1.5 cm over the past three decades (P. Trachsel, personal communication), which is comparable to the data shown earlier. Further discrimination between arable and grassland use is not feasible, since the relative proportions of each type of land use for cultivated peatlands are based only on statistical probabilities.

4.3.2 Consistency of combined spatial data

The combination of several different spatial data sets could lead to inconsistencies, as different data sources and methods of evaluation are used. For mineral soils, the reliability of the combined data sets was examined for the combinations i) Swiss land-use statistics and digital soil map and ii) community land-use statistics and digital soil map (Fig. 9).

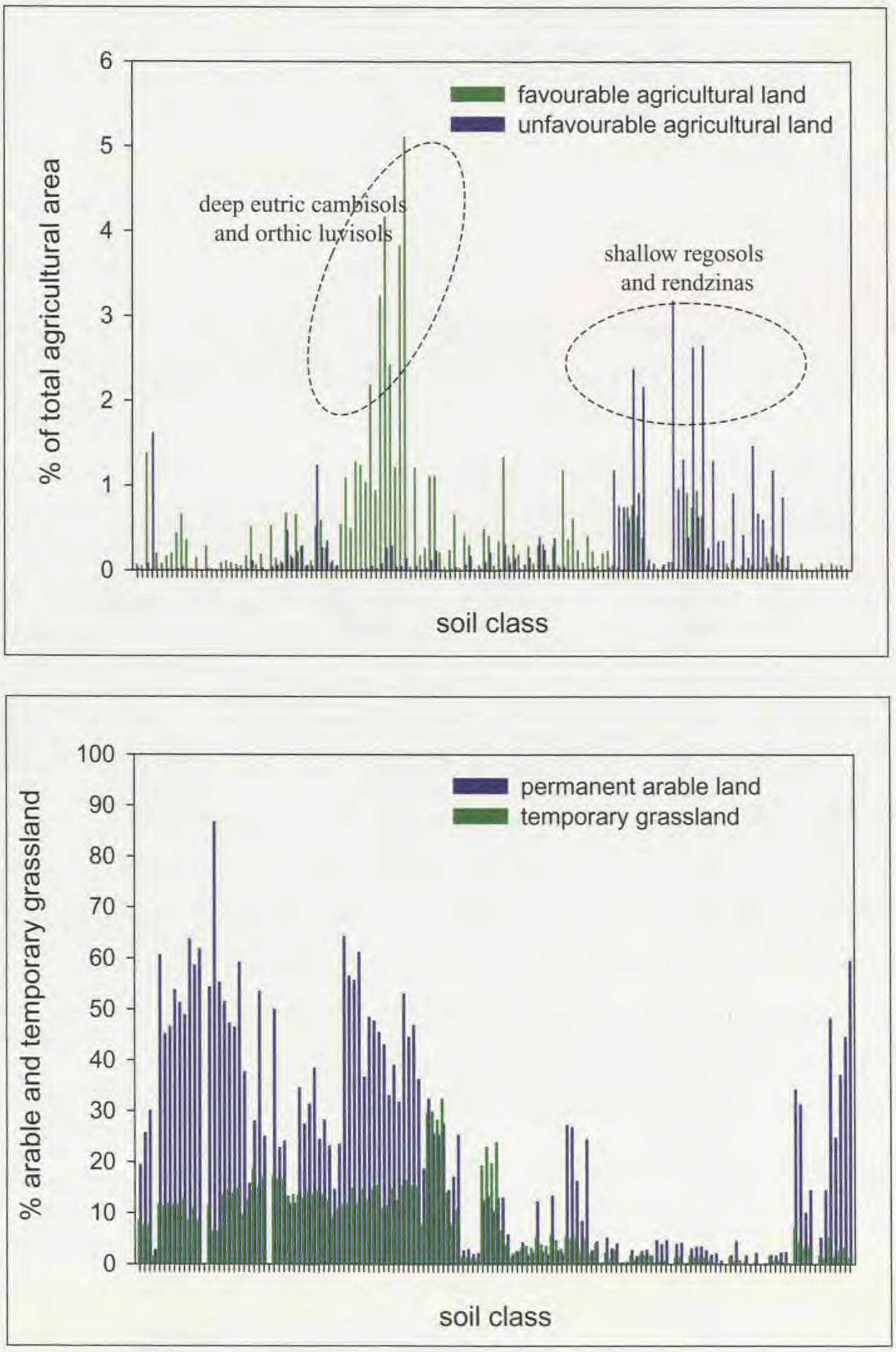


Fig. 9. Distribution of land-use types among digital soil map units. *Top*: favourable and unfavourable agricultural land. *Bottom*: permanent arable and temporary grassland. Soil classes are given according to the Swiss digital soil map.

The combined land-use and soil map (Fig. 9, upper panel) shows a consistent picture regarding soil classes. Favourable agricultural land (which includes all arable land) is associated mainly with deep and fertile cambisols and luvisols, whereas shallow soils with a high stone content are predominantly occupied by unfavourable agricultural land (e.g. alpine meadows). The combination of the soil map and community land-use statistics (Fig. 9, lower panel), which provides higher resolution of land-use types at the expense of spatial precision, agrees with the finding that the most unfavourable soils have a very low share of intensive agriculture and are used almost exclusively for extensive grassland.

The allocation of peatlands in the inventories were combined with the spatial digital soil map data and the data for land-use statistics to obtain the total area of intact peatlands (Table 8).

Table 8. Total area (ha) of intact peatlands, derived from peatland inventories, the area of intact peatlands used for agriculture, and the area of intact peatlands covered by corresponding soil classes in the digital soil map.

	Total area	Peatland area used for agriculture	Total area of histosols (digital soil map)
Fens	25'745	19'419	7'194
Bogs	1'490	706	944

Comparison of the peatland area and the digital soil map reveals a discrepancy between the soil classes that should include all peatlands (i.e. histosols) and the estimated proportion of peatlands covered by these soil classes. Only 63% and 28% of the bogs and fens, respectively, are found in the histosol soil classes. This result is striking since histosol soil classes in the digital soil map represent all potential organic soils. It thus appears to be inappropriate to use the digital soil map for estimating the area or allocation of cultivated organic soils.

4.2.4 Measurement of soil carbon

Analytical methods for SOC differed between data sources, but most analyses were carried out according to FAC (1989). Soil bulk density was measured i) as the weight per volume using soil cores, corrected for volumetric stone content, or ii) as the weight per volume of fine earth. SOC was measured either by complete dry combustion in an O_2 stream, with correction for carbonates, or by wet oxidation in sulphuric acid with hot potassium dichromate, according to the modified Mebius procedure cited in Nelson and Sommers (1982). This procedure avoids the low and variable recovery of organic C observed with the Walkley-Black method by heating the sample after addition of the reagents. A comparison of the two methods showed a slightly lower OC yield of 92–95% with dichromate oxidation (H-J. Bachmann, personal communication). This is in agreement with Arrouays *et al.* (2001). An examination of two samples from different surveys – one using the O_2 method, and the other $K_2Cr_2O_7$ – revealed no statistically significant difference for the mean OC content. Therefore, analytical differences for the determination of SOC contents were disregarded.

4.2.5 Statistics

Relationships were tested using Spearman's rank correlation coefficient for three levels of significance ($P = 0.05$, $P = 0.01$, and $P = 0.001$). Linear regression was calculated with robust regression, using weighted least squares to minimize the effect of outliers. Significant variation ($P = 0.05$) was tested by one-way analysis of variance for the error sources land use, soil clay content, and altitude. Significant differences for medians are given by the Kruskal-Wallis test for the same levels of significance as above. The error Δx_{rel} for the estimate of C stocks (expressed as 95% C.I.) is given by the errors of the regressions for soil carbon concentrations (equations 5, 6) and the estimate for the soil bulk density (equation 7) as:

$$\Delta x_{rel} = \sqrt{\Delta u_{rel}^{2} + \Delta w_{rel}^{2}} \qquad [10]$$

where Δu_{rel} and Δw_{rel} are the relative errors of soil carbon concentration and bulk densitiy, respectively. Relative errors for C stocks are ±22%, ±24%, and ±25% for arable land, permanent grassland, and temporary grassland, respectively.

4.3 Results

4.3.1 Carbon stocks in mineral soils

The calculation of C stocks in mineral soils revealed a total stock of around 123 Mt OC (0–100 cm) for a total area of 1'488'000 ha (Table 9).

Table 9. SOC stocks in mineral agricultural soils (Mt C) for various land-use types (± 95% C.I)

Land-use type	Area (ha)[1]	Soil organic carbon stock		
		0-20 cm	0-100 cm	0-20 cm (equivalent soil depth)
Arable	289'339	11.7 (±2.6)	26.2 (±5.8)	12.1 (±2.7)
Temporary grassland	114'147	4.9 (±1.2)	13.4 (±3.4)	5.1 (±1.3)
Permanent grass-land (favourable)[2]	504'467	25.6 (±6.4)	46.6 (±11.2)	27.4 (±6.6)
Permanent grass-land (unfavour-able)[3]	580'047	27.6 (±6.6)	36.5 (±8.8)	29.5 (±7.1)
Total	1'488'000	69.8 (±9.6)	122.6 (±15.8)	74.1 (±10.2)

[1] Area of organic soils subtracted

[2] Classes 81 and 82 of land-use statistics

[3] Classes 83–89 of land-use statistics, including mountain farming

On average, more than 50% of SOC is stored in the upper 20 cm of the soil. The fraction in the topsoil is higher in permanent grasslands (55–76%) than in arable and temporary grasslands (37–45%) due to higher SOC concentrations in the topsoil, as well as more shallow soil profiles under permanent grassland. The difference between volumetric C stocks and stocks calculated for equivalent soil depths (Table 9, last column) is 5.6% of the SOC stock (0–20 cm). The difference is most pronounced in permanent grasslands, as their bulk densities are significantly lower than those of arable topsoils (100%). Mean SOC stocks per hectare for various land-use types are given in Table 10.

Table 10. Mean SOC stocks (t OC ha^{-1}) for various land-use types

	Arable	Temporary grassland	Permanent grassland	
			Favourable	Unfavourable
0–20 cm	40.60	43.43	50.71	47.65
0–100 cm	90.38	117.39	92.28	62.88

The data in Table 10 show that topsoil C stocks are highest under permanent grassland, but not for deeper soil horizons. As regards the total C stocks, differences in soil depth and stone content offset the higher concentrations of SOC found in the fine earth of permanent grasslands.

4.3.2 Carbon stocks in organic soils

Using various assumptions for the total area of cultivated peatlands and peat decay rates, the time-course of changes in C stocks was determined, beginning with the first melioration programme in 1885 and projecting future trends (Fig. 10).

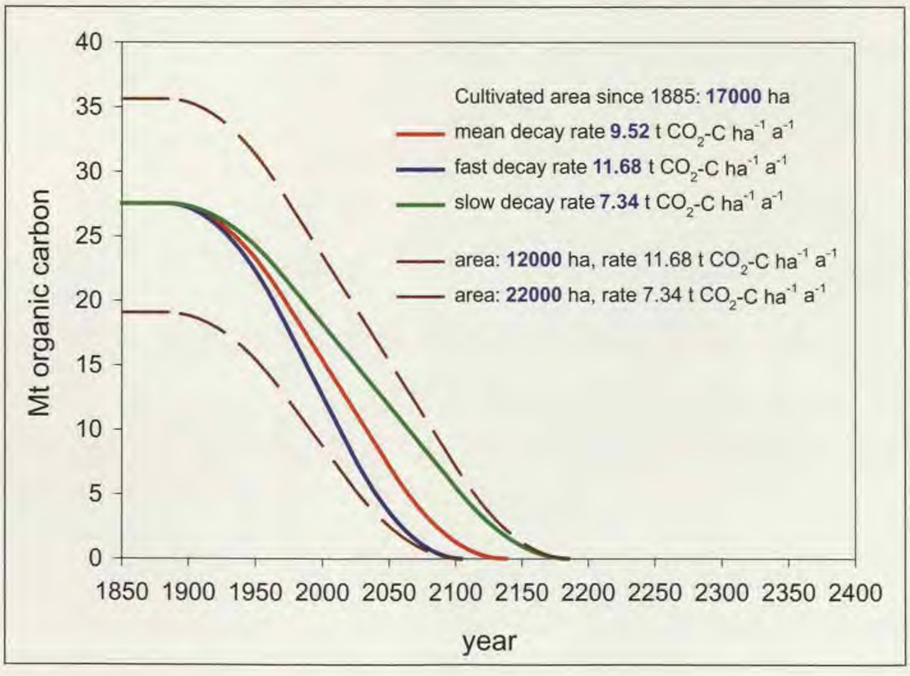


Fig. 10. Estimated time-course of changes in carbon stocks in cultivated organic soils since 1885. *Solid lines* indicate variability of decay rates only, *dashed lines* variability of decay rates and area estimates.

Today, the total C stock in cultivated peatlands has decreased to about half the level that existed before the beginning of the melioration activities in 1885. The decline in C stocks will continue until at least 2100, when the peat layer in the 12'000 ha scenario with the highest decay rate is fully oxidised. If peatland areas were higher in 1885, and the lowest decay rate is assumed, the decay of organic soils will continue until about 2150. Consequently, annual carbon losses from cultivated peatlands range from 88 x 10^3 to 257 x 10^3 t CO_2-C. Calculated C stocks of cultivated peatlands in 2000 are in the range of 710 to 1050 t OC ha^{-1}. The weighted mean of C stocks in cultivated peatlands of the Central Plateau, covering around 7000 ha, as estimated by Presler and Gysi (1989), is 980 t OC ha^{-1}. This corresponds to a mean carbon concentration of 31%, a mean bulk density of 0.27 g cm^{-3}, and a weighted mean peat thickness of 1.16 m. Thus, the assumed values for the former peat thickness and carbon concentrations in order to estimate C stocks in intact and cultivated peatlands would appear to be reasonable, since these estimates were calculated independently of the data of Presler and Gysi (1989).

Together with the amount of carbon stored in intact peatlands (about 32.2 Mt OC, corresponding to 1600 t OC ha^{-1}), the total stock of carbon in organic soils is about 47 Mt OC (Table 11). Intact peatlands account for 68% of C stocks in organic soils, and cultivated peatlands about 32%.

Table 11. Area (ha) and carbon stocks (Mt OC) of intact and cultivated peatlands

	Area	Carbon stock
Intact peatlands	20'125	32.2 (± 6.4)[1]
Cultivated peatlands	17'000 (12'000–22'000)	15.0 (8.5–23.2)
Total (mean)	37'125	47.2

1 Estimated error for C concentrations and bulk densities in intact peatlands.

The sum of organic C stocks of mineral and organic soils gives the total C stock of Swiss agricultural soils, i.e. 170 Mt OC, with 72% in mineral soils and 28% in organic soils.

Carbon stocks in agricultural soils thus exceed C stocks in mineral forest soils; the latter were estimated recently at 110 Mt OC (Perruchoud et al. 2000). Together, Swiss forest and agricultural soils contain 280 Mt SOC.

4.4 Main factors governing soil carbon stocks

The available data indicate that the main factors governing soil carbon stocks include land use, climate, and soil characteristics such as profile depth and stone and clay content. Additional factors may play an important role but have yet to be identified, due to a lack of data. Carbon stocks of organic soils comprise intact and cultivated peatlands. Arable land and temporary grasslands are mainly found at altitudes up to 1000 m above sea level, whereas the relevant areas of permanent grasslands extend up to 2500 m above sea level (Fig. 11). Organic soils used agriculturally account for only 2.3% of the total agricultural area.

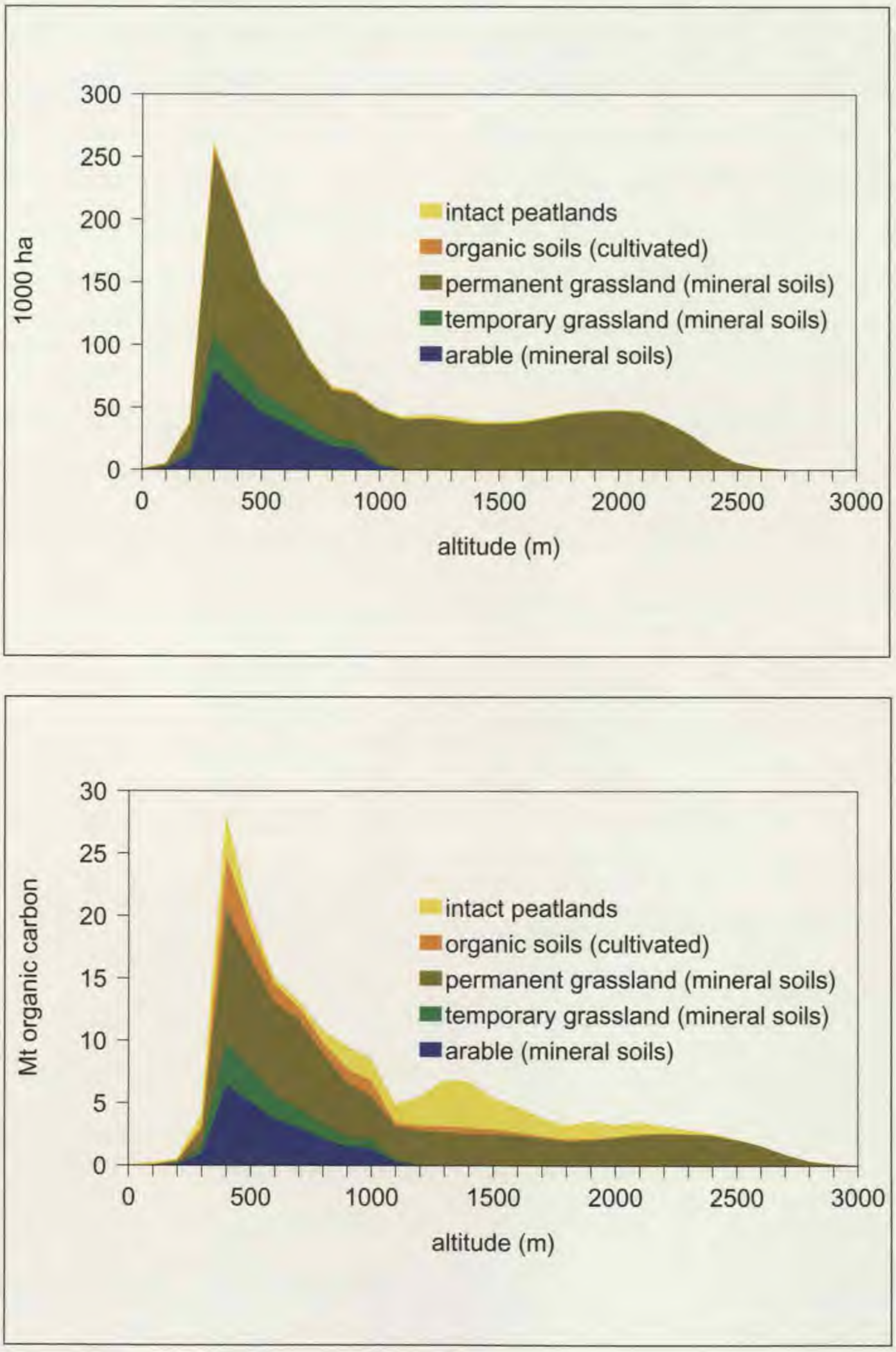


Fig. 11. Distribution of land-use types (*top*) and soil C stocks (*bottom*; 0-100 cm for mineral, and 0-200 cm for organic soils) along an altitudinal gradient.

The largest fraction of the total of 170 Mt OC is concentrated at altitudes <1000 m, mainly in arable land and intensively managed grassland on the Central Plateau. Here, cultivated peatlands contribute significantly to the total C stock. Intact peatlands constitute the main carbon reserve between 1300 and 1500 m and still play a relevant role up to 2200 m.

The change in the SOC content of mineral soils with altitude should reflect increasing SOC concentrations, as calculated by Equation 6. However, comparison of SOC concentrations in fine earth with the stocks calculated for favourable and unfavourable permanent grasslands (0–20 cm) reveals only a minor influence of altitude (Fig. 12). The apparent discrepancy between C concentrations and C stocks can be attributed to (i) increasing rock fraction, (ii) decreasing profile depths and (iii) decreasing bulk density with altitude (see above). Thus, the effect of higher SOC concentrations on soil C stocks is largely offset by other soil properties. It is interesting to note that unfavourable permanent grasslands contain smaller stocks per hectare than favourable ones. Again, this reflects the different soil qualities of intensively and extensively managed grasslands, if the latter are located mainly on unfavourable sites. The increasing SOC concentrations of fine earth contrast with decreasing yields of permanent grasslands with altitude (Fig. 13).

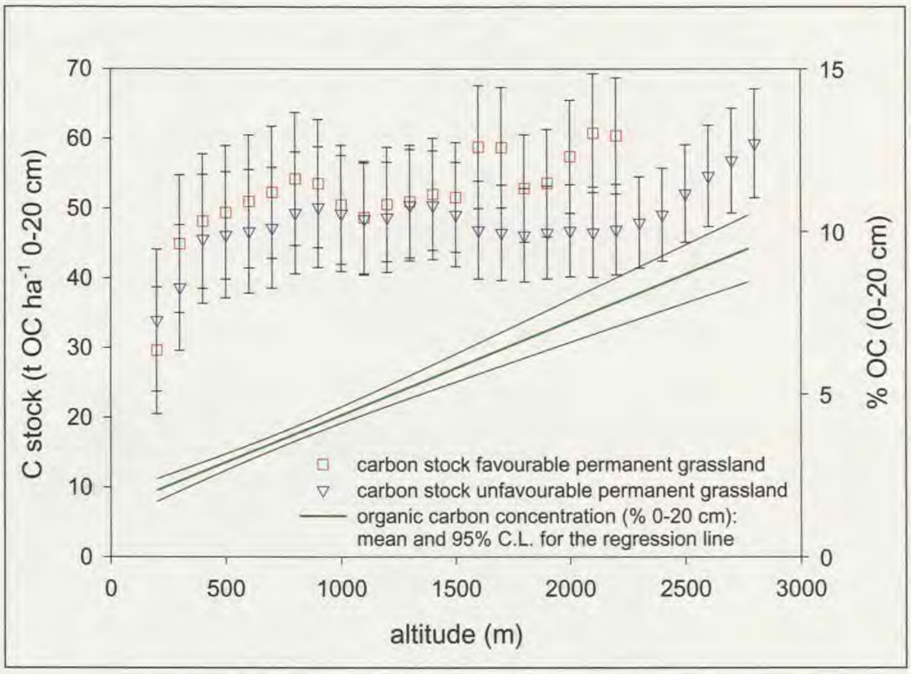


Fig. 12. Soil carbon stocks ($t\ ha^{-1}$) and SOC concentrations (%) in permanent grasslands.

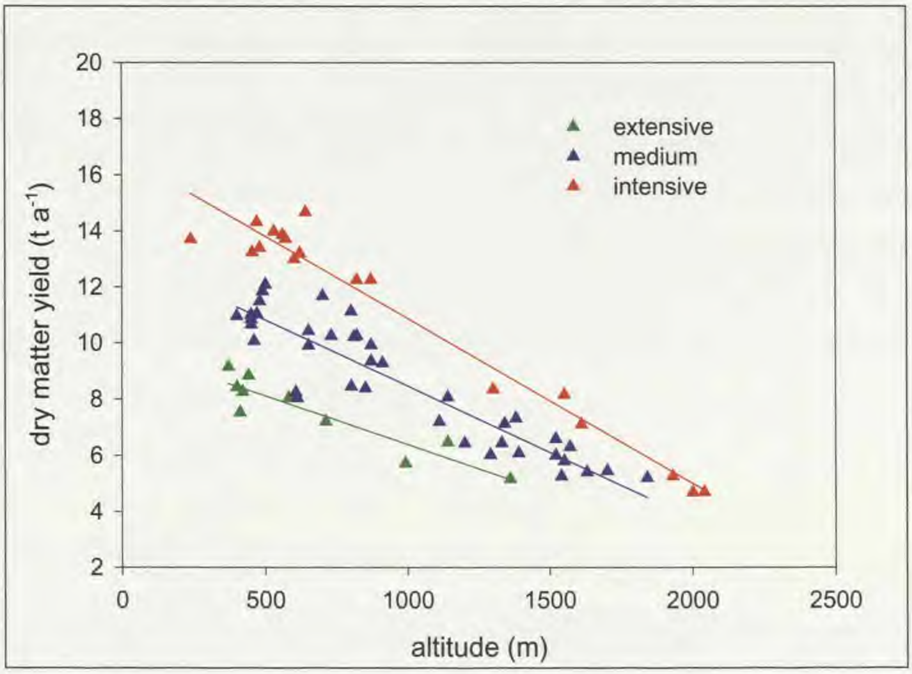


Fig. 13. Altitude dependence of yield of permanent grasslands for different management intensities. Extensive: 1 cut per year; medium: 3 cuts per year; intensive: 5 cuts per year (for yields at 700 m a.s.l).

The apparent disagreement between higher SOC concentrations and concomitant lower yields raises questions about the mechanisms accounting for increasing SOC concentrations with altitude. C concentrations would be expected to increase with altitude if (i) the residue input from above- and below-ground litter is supposed not to be proportional to the dry matter yield and (ii) the quality and quantity of the input materials are affected in a different way by altitude than SOC turnover.

Regarding residue inputs, Hitz *et al.* (2001) determined that the above-ground yield of alpine grassland decreases along a climatic gradient from 1665 to 2525 m above sea level in the Canton of Graubünden. Although the ratio of root:shoot biomass production increased with altitude from 1 to 5, total root mass was similar at all sites, suggesting that root turnover decreases with altitude. Also, root C/N ratios (calculation based on C and N inputs) were almost identical among sites, indicating similar residue quality. Although residue inputs from above- and below-ground litter were not proportional to dry matter yield, below-ground inputs did not increase with altitude. From this study, the hypothesis that SOC concentrations increase due to residue C input can be excluded. Thus, it seems more reasonable to assume that SOC turnover is more limited by altitude than by net primary productivity. Among the potential variables accounting for differences in SOC turnover and NPP, the limiting effect of temperature on SOC turnover seems to be crucial. NPP depends more on radiation flux (see 5.3.2). The findings of increasing SOC concentrations with altitude associated with decreasing temperature agrees well with studies by Trumbore *et al.* (1996) and Townsend *et al.* (1995) along elevation and temperature gradients in the Californian Sierra Nevada, and on Hawaii. These studies also showed decreasing SOC turnover times with decreasing temperature.

The proportion of SOC that can be explained by soil clay content decreases as SOC content increases (Fig. 14). For permanent grasslands at altitudes greater than 1000 m above sea level, the relationship to clay content is not significant. This relationship, which is based on the 544 soil profiles, solidly supports the mechanism of SOM stabilisation discussed in Section 3.2.

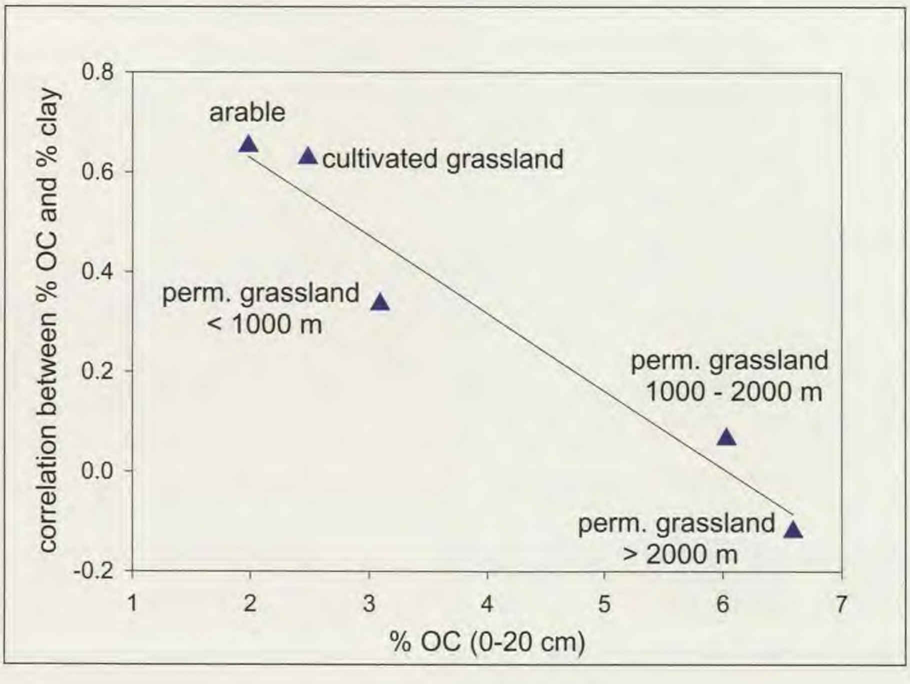


Fig. 14. Relationship between % SOC and % clay for agricultural soils at different altitudes.

Thus, clay content is only one factor in SOM stabilisation, and its effect is negligible at higher altitudes under permanent grassland, where SOM turnover is affected by other physical/chemical constraints, such as temperature.

One major assumption for the calculation of carbon sequestration potentials is that of a site-specific carbon stock, which is given by site conditions as discussed above, land-use, and management. Carbon sequestration potentials for an individual land-use change can then be derived from the difference in carbon stock under a given land-use relative to the carbon stock of the land-use type which will follow. Data in Fig. 9 showed that some soil classes are preferentially used as arable land or as grassland. In particular, soil classes with only a small share of arable land are characterised by small SOC stocks. Since the C stock of an arable field will in most cases not equal the C stock which would be attained under permanent grassland with identical site conditions, the potential for carbon storage is reduced by farmers' preferences for using favourable sites for arable cropping instead of grassland. Thus, current C stocks are not only the result of the conditions for productivity and C turnover under different land-use types, but are also determined by farmers' decisions to use a site in a specific way due to the demands of a crop or the suitability of a site, e.g. regarding machine use.

Quantification of this selectivity of land use for particular soil classes would facilitate assessment of the sequestration potential achievable by land-use changes. The selectivity is obtained by comparing the current C stocks per hectare (Table 10) with the stocks that would result from a proportional distribution of each land-use type across the 140 soil classes (i.e. hypothetical stocks) (Table 12).

Table 12. Carbon stocks (t OC ha^{-1}, 0–100 cm) for the current distribution of land-use types among soil classes, and for a distribution of each land-use type proportional to all soil classes (hypothetical stock)

Land use	Arable land[1]	Temporary grassland[1]	Permanent grassland[2]
Current stock	90.4	117.4	80.7
Hypothetical stock	78.1	113.3	89.3
Factor	1.16	1.04	0.91

[1] Hypothetical stock calculated for agricultural area < 1000 m above sea level.

[2] Stocks are weighted means of favourable and unfavourable grasslands.

Current stocks per hectare are 1.16- and 1.04-fold larger for arable cropping and temporary grassland, respectively, than hypothetical stocks, and 0.91-fold lower for permanent grassland. The hypothetical stock for permanent grassland is lower than for arable land and temporary grassland because it includes shallow alpine soils, whereas arable and temporary grassland soils are restricted to the Central Plateau, with deeper soil profiles. The factors calculated indicate that arable rotations are located on soils with a higher C storage capacity, whereas soils with smaller C storage capacity are mainly used as permanent grasslands. The difference between typical arable soils and the mean grassland soil can be explained by deeper soil profiles, smaller stone contents and a slightly higher clay fraction, even when only soils in the Central Plateau are considered. The selection of soils with larger C storage capacity (and thus higher fertility) for intensive agriculture is logical from a farmer's point of view. The extensive management of permanent grasslands (mainly alpine meadows and sites with steep slopes) largely accounts for the smaller current C stock. If permanent grassland were to be implemented exclusively on all current arable and temporary grassland soils, hypothetical stocks for this combination of soil classes and land use would amount to 113.29 t OC ha^{-1};[26], due to the higher SOC concentrations in the fine earth of grassland soils (Table 6). The difference between actual and hypothetical stocks has some consequences for the calculation of carbon sequestration potentials, which will be addressed in Chapter 5.

4.5 Historical changes in soil carbon stocks of agricultural soils

Historical changes in the total soil C stock can be attributed either to shifts in agricultural land use or to changes in the total agricultural area. Shifts in agricultural land use, and in total agricultural area, are well documented for recent decades (Fig. 15). The total agricultural area has decreased fairly steadily over the past 150 years, largely as a result of changes in permanent grassland and mountain farming. Changes in the relative proportion of each land-use type cannot be correlated with changes in soil C stocks, since it is not known which areas were converted to a different land-use type and at what time. However, it is possible to estimate the changes in C stocks resulting from decreases in the peatland area, and from the decline in the total agricultural area.

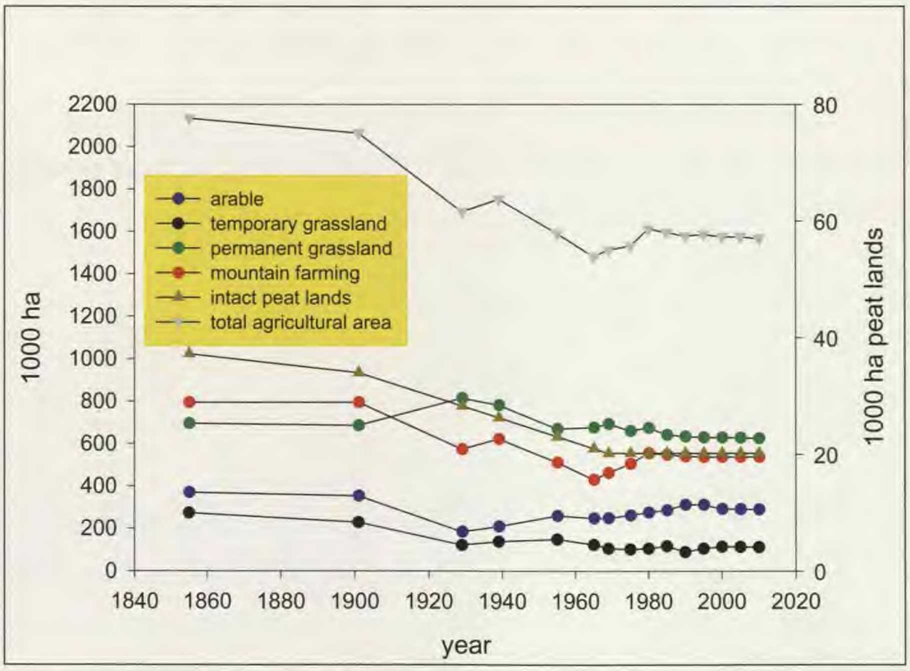


Fig. 15. Changes in agricultural land use in Switzerland over the past 150 years.

For cultivated peatlands, annual emission rates between 1885 and 2002 ranged from 58 to 165 x 10^3 t C, corresponding to a total loss of 6.8–19.4 Mt C since the beginning of peatland cultivation. Annual emission rates between 1885 and 1910 are lower than reported for current peat oxidation, since the area of cultivated peatland increased between 1885 and 1970.

[26] The closeness of this value is coincidentally, but not causative close to that for the hypothetical stock of temporary grassland, in Table 12

Changes in the total agricultural area lead to shifts in total C stocks if the soil C storage per unit land differs between land-use types. It is thus necessary to know to what type of land use agricultural land is converted. A good estimate can be given for changes in land use between the two surveys conducted for the periods 1979/1985 and 1992/1997. These statistics provide an accurate allocation of the converted land. According to the Swiss Federal Statistical Office (2001), 48'200 ha of agricultural land was lost over the 12 years between the two surveys, or 4'017 ha annually. About two thirds of this was used for infrastructure (urban areas) on the Central Plateau. The remaining area, mainly located in alpine regions, developed into set-aside land[27] and forests. Of the urbanised area, 50% is assumed to be covered by gardens and other open spaces. For these areas, no change in soil C stocks is assumed. The remaining 16'200 ha of new urban areas (equivalent to an annual loss of 1'350 ha) developed on former agricultural land was used for buildings. Can all of the SOC stored under this area be accounted for as oxidative carbon loss? A reasonable estimate should take into account the proportion of the stabilised SOC fraction which is not likely to be oxidised in the medium term. This proportion amounts to around 30 % of the total SOC (calculated from Table 2 in Falloon *et al.* 1998), as determined by means of various chemical or physical fractionation methods. Long-term agricultural experiments reveal that SOC becomes depleted by around 50% during decades of bare fallow and/or no fertilisation (Kiem *et al.* 2000). For the calculation of C remaining in the soil material after excavation, it is assumed that the 30 % value represents the lower limit of SOC surviving construction activities.

Taking the mean C stock of 90 t ha^{-1} for arable soils, the total loss due to conversion of arable land to urban areas during the12-year period under investigation was 16'200 ha x 90 t C ha^{-1} x 0.7 = 1020 x 10^3 t OC, or 85.1 x 10^3 t OC a^{-1}. This same annual carbon loss has also to be expected in the future, if the rate of urbanisation remains at the current level. The total urban area in Switzerland is 279'000 ha (Swiss Federal Statistical Office 2001). If the same conditions are applied to the whole urban area, 279'000 ha x 90 t C ha^{-1} x 0.7 = 17.6 Mt SOC has been lost historically due to building activities. It is interesting to note that total SOC losses, as well as annual oxidation rates, are in the same order of magnitude for both peatland cultivation and construction.

Overall, 24–37 Mt SOC has been lost historically as a result of peatland cultivation and construction, corresponding to 14–22% of the current agricultural soil C stock.

Besides historical changes in the C stocks of agricultural soils, the total change induced by conversion from natural woodland to agricultural land is calculated in order to obtain the cumulative anthropogenic effect on C stocks since the beginning of clearances. For this calculation, it is assumed that today's C stocks per hectare of mineral soil are equal to the prehistoric C stock. Only topsoil changes were considered. According to Perruchoud *et al.* (2000), the current C stock is estimated at 62 t OC ha^{-1} (0–20 cm). Taking mean stocks and the total area of the various types of agricultural land use, and subtracting areas above the timberline (2300 m above sea level), a total of 22.6 Mt C has been lost to date due to deforestation and conversion to agricultural

[27] former agricultural land that is abandoned and on which natural vegetation establishes

land. Thus, historically "young" SOC losses due to peatland cultivation and urbanisation exceed the decrease in soil C stocks that occurred as a result of deforestation from the beginning of settlement and agricultural activities in the Neolithic period.

5. The potential of agriculture to sequester carbon in the soil

5.1 Aim of the assessment of carbon sequestration potentials

Carbon sequestration in terrestrial ecosystems can be defined as the net removal of CO_2 from the atmosphere into organic carbon pools. This process is believed to be a measure that could help to mitigate the greenhouse effect, but in effect it is merely a way of "buying time" until improved technologies are available to reduce anthropogenic GHG emissions. Nevertheless, it is necessary to assess a nation's C sequestration potential in relation to overall GHG emissions before the relative importance of this measure can be evaluated. For a general (global) outline of possible ecosystem management options for reducing CO_2 concentrations, the reader is referred to a recent report by Batjes (1999). Here, the most relevant carbon sequestration activities for Swiss agriculture are described and evaluated. Local structures, e.g. the high proportion of permanent grassland, and integrated farming[28] practices for arable cropping are taken into account. Measures that could be taken to avoid further oxidative losses of SOC and possible trade-offs with other GHG emissions are discussed.

5.2 Methodological approach

Some potential C sequestration activities can be evaluated by using the data for soil C stocks given in Chapter 4 – in particular, the conversion of arable land to grassland, and wetland management. All other activities (soil tillage, low-input farming, grassland management, set-aside) were selected according to IPCC (2000). Only activities with an obvious potential under the circumstances prevailing in Switzerland were considered. Where no data were available or data were of low quality, only qualitative estimates of C sequestration potentials could be made. The activities considered to have a significant C sequestration potential in Switzerland are shown in Fig. 16 in relation to the activities listed by IPCC (2000).

Other measures suggested by Smith *et al.* (1997) – including amendment of animal manure and sewage sludge, or straw incorporation – are not treated here, as these activities are either already being applied (manure amendment, straw incorporation) or will remain inappropriate in the near future (application of sewage sludge).

The activities included under the heading of **cropland management** are tillage, ley-arable farming[29], and farming intensity. Most of the agricultural land in Switzerland (86%, excluding mountain farming, in 2000) is subject to the requirements of integrated farming, which is the "required standard of ecological performance"[30] (REP). Integrated farming can be regarded as a management system with lower inputs than in conventional farming. Some of the measures included are relevant for C sequestration and are discussed under Section 5.3.1.

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[28] Key elements of integrated farming are a balanced whole-farm nutrient budget, diversified crop rotation, an appropriate share (7%) of ecological compensation areas, as much ground cover as possible in winter and specific rules for the use of pesticides.

[29] Crop rotations with annual or perennial temporary grassland (German: *Acker-Kunstwiese Rotation)*

[30] German term: *ökologischer Leistungsnachweis*

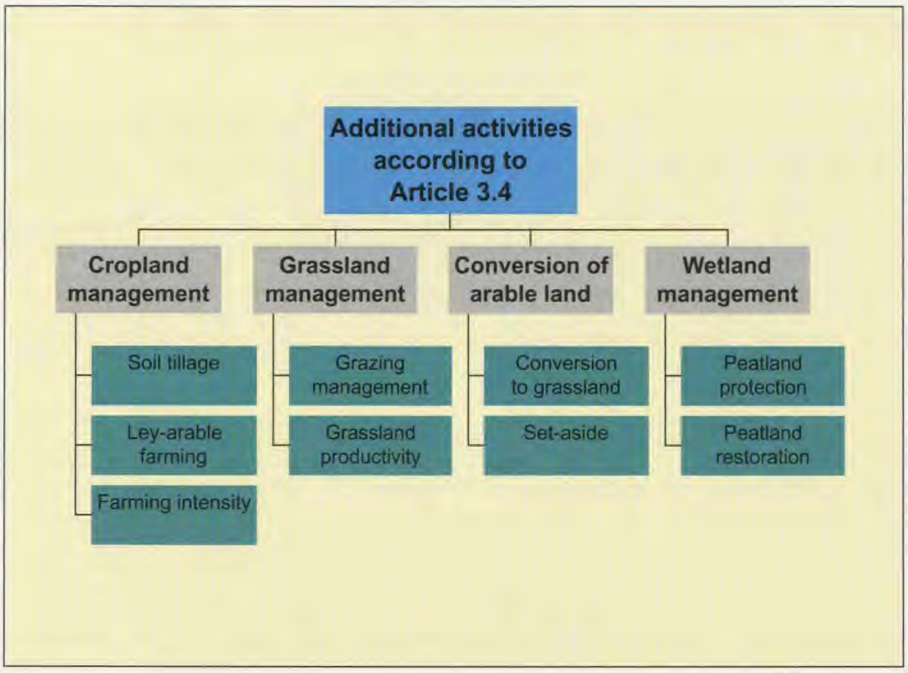


Fig. 16. Additional human-induced activities (according to Article 3.4 of the Kyoto Protocol) with C sequestration potential for Switzerland. Terms in *grey boxes* refer to IPCC (2000).

The agricultural practice of REP is also compared with other low-input agricultural practices, such as organic farming (see Section 5.3.1).

Studies comparing the effects of different **tillage systems** on SOC mostly distinguish between two levels of tillage intensity – conventional tillage and conservation tillage. Conventional tillage includes the use of mouldboard ploughing, followed by any type of seedbed preparation. Conservation tillage is defined as any tillage and planting system in which 30% or more of the crop residue remains on the soil surface after planting (IPCC 2000). Conservation tillage can include specific tillage types, such as no-till, ridge-till, or mulch-till, which meet the residue requirements. In most studies, the two extremes, i.e. conventional till and no-till, are compared. To assess the effect of no-till systems on SOC storage, a total of 34 studies were evaluated. The data given in Section 5.3.1 (source: studies from regions with climatic and pedological conditions similar to the Swiss Central Plateau or from Switzerland) show the potential for no-till without any cultivation other than direct drilling, as this can be considered the most effective tillage system for carbon sequestration. Only for the Swiss site of Changins were experiments comparing mouldboard ploughing with minimum tillage (rotary harrow) included, as this is one of the few experimental sites in Switzerland where SOC amounts have been measured systematically for different tillage systems.

Ley-arable farming as an SOC enhancement measure can be assessed by comparing published effects on SOC stocks with current agricultural practice in Switzerland. Unfortunately, it is difficult to make a quantitative estimate of the C sequestration potential since the data base for this activity is small, and the data are often from studies performed under different climatic and pedological conditions. The high proportion of ley (temporary grassland) already existing in crop rotations makes an evaluation even more challenging, since pronounced effects are only expected for crop rotations in which ley is newly implemented. However, available data for ley-arable rotation and temporary set-asides[31] are discussed in order to evaluate their relative C sequestration potentials.

For other cropland activities, such as soil mulching and under-sowings, only a qualitative assessment can be undertaken. These activities are treated under **production methods and farming intensity.** In 2000, more than 94% of Switzerland's intensively used agricultural area (excluding mountain farming) was subject to the requirements of integrated or organic farming (86% IP, 8% organic farming). Only limited data were available to assess the impact of integrated farming on SOC stocks compared with so-called conventional farming; therefore, integrated farming is evaluated only qualitatively. The data base is slightly better for the comparison of conventional or integrated vs low-input farming[32] in terms of their effects on SOC. Many of the characteristics of low-input farming are related to different types of crop rotation, and different treatments with organic fertilisers, making a qualitative assessment of any specific effect more difficult. Two field experiments where identical crop rotations and rates of organic amendments were compared for conventional and low-input agriculture are discussed in Section 5.3.1.

Grassland management includes **grazing land management** and **grassland productivity.** Limited data are available describing the effects of stocking densities, rotational frequencies, and grazing vs mowing on soil C stocks for Western and Central Europe and the alpine region. Most studies are from the US, with different climatic and pedological conditions. The regional data base for grassland productivity is little better, and estimates of the direction of changes in soil C are highly uncertain. The most relevant results with respect to grassland management reported in the literature are discussed in Section 5.3.2.

Conversion of arable land includes conversion to grassland and to set-aside land. For both measures, quantitative estimates can be made either on the basis of the data for soil C stocks, or from literature data. Set-aside leads to gradual regrowth of forests on former grassland. This management-induced change can be observed in some regions of the Swiss Alps where economically unviable extensive management of alpine meadows has been abandoned by farmers (Swiss Federal Statistical Office 2001). The consequences of set-aside for C sequestration are compared with the conversion of arable to grassland.

Wetland management[33]

Consequences for C fluxes due to changes in the area of peatlands are calculated with the data used for the estimation of Swiss soil C stocks. Since intact peatlands are active carbon sinks, the current C sink of these areas is estimated from literature data. Net GHG effects[34] further include changes in fluxes of N_2O and CH_4, which respond highly sensitively to altered peatland management. Flux data for the three GHGs are not available for Switzerland; therefore, literature data from several European studies were used to assess the potential

[31] Used here as a synonym for ecological compensation area *(German: ökologische Ausgleichsfläche)*

[32] Used here as a synonym for organic farming

[33] Wetlands comprise marshes, peatlands, floodplains and other shallow flooded areas that undergo periodic or continuous flooding. The term is used here for consistency with the terminology for potential Article 3.4 activities as proposed by IPCC (2000).

[34] The cumulative flux of the three agricultural GHGs (CO_2, CH_4, N_2O), expressed as CO_2 equivalents

of wetland management for C sequestration. In addition to C sequestration in intact peatlands, the avoidance of emissions through restoration of cultivated peatlands is discussed as a GHG mitigation measure.

How long will the sink persist?

The effect of management practices on C sequestration is of limited duration. The time required by SOM to achieve equilibrium after a change in management depends largely on the chemical, mineralogical and physical properties of the soil, as well as on climatic conditions. For newly adopted activities, the duration required for the system to approach a new SOM equilibrium is in the order of some decades to >100 years (IPCC 2000). Thus, for the first commitment period (2008–2012), the duration is not critical, but estimates will become more uncertain as time proceeds. For the activities considered for mineral soils, the corresponding sink was assumed to be inexhaustible until 2020; i.e. the annual C sequestration rate depends only on the area. Carbon losses from organic soils will remain constant as long as carbon is available to decompose. After 2100, the total area of cultivated organic soil will begin to decrease as a consequence of its transformation to mineral soils.

5.3 Carbon sequestration activities and their net greenhouse gas effect

5.3.1 Cropland management

Cropland management includes crop rotation, soil tillage, fertilisation and other measures taken for the purpose of soil protection. These include under-sowings and mulching. Crop rotation affects the amount of residue input and SOM turnover. Several individual activities relating to crop rotation are expected to have positive effects on SOC stocks (IPCC 2000). Among them, ley-arable cropping, cultivation of perennial forage crops, use of cover crops and residue retention are relevant for Swiss agriculture. At the systemic level, the most common practices include integrated and low-input farming. Conservation tillage can also be applied in conventional farming. Dick *et al.* (1998) studied the impacts of agricultural management practices on C sequestration in forest-derived soils of the US Eastern Corn Belt. Annual C inputs and tillage intensity were the most important factors affecting C sequestration. The impact of crop rotation on C sequestration was primarily related to its effect on total C inputs. The removal of above-ground plant biomass and the use of cover crops were less important. Measures taken to prevent soil erosion, e.g. conservation tillage, are often accompanied by reduced SOC loss or by SOC accumulation (see below).

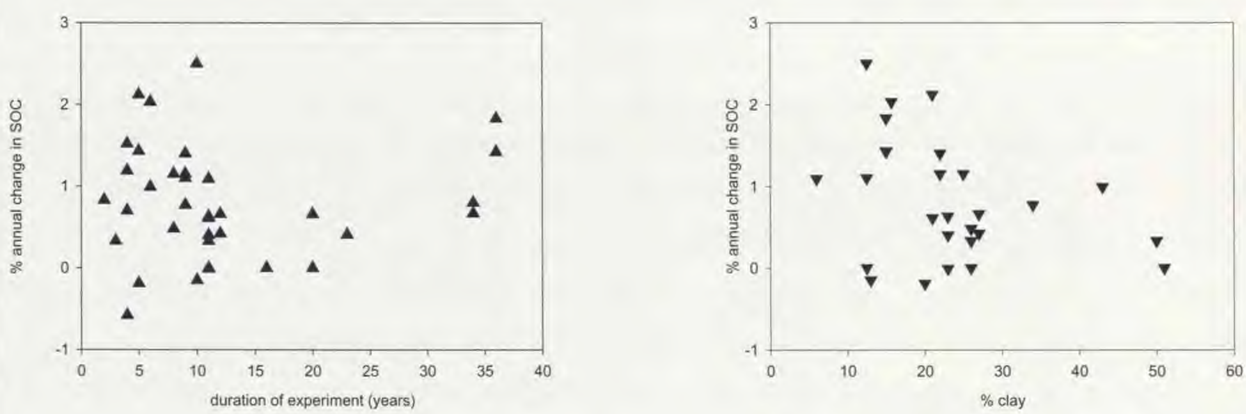
No-till

No-till is considered an effective method for carbon sequestration. Smith *et al.* (1998) estimated the mean annual mitigation potential from no-till in Europe at 0.34 t OC ha^{-1}. The same rate was proposed by IPCC (2000) for Europe, including 0.02 t C ha^{-1} a^{-1} in fossil fuel savings. In the present study, the data base with results from 14 experimental studies cited in Smith *et al.* (1998) was combined with data from an additional

20 studies, mainly from Central and Western Europe[35], including Swiss experiments at Changins, in the Canton of Vaud (Vullioud 2000), and at Zollikofen, in the Canton of Berne (Zihlmann and Weisskopf 2001). A third Swiss site located at Tänikon was excluded because it had been used as a 10-year permanent grassland before the implementation of the tillage system, with concomitant carbon losses for all types of tillage (Anken *et al.* 2002).

With the available data, it was evaluated whether the annual change in C stock relative to the control was related to the duration of the experiment or to the soil clay content (Fig. 17).

Fig. 17. Effects of duration of experiment (*left*) and clay content (*right*) on annual changes in SOC under no-till systems. Data are from 34 experiments (see text).



The plots in Figure 17 reveal no significant relationship between C stocks and the duration of the experiments conducted under various environmental conditions. Similarly, changes in C stocks were not related to soil clay content. The response of C stocks to tillage intensity varied strongly between experiments, and was even negative in some cases. Similar results were reported in a review by Paustian *et al.* (1997), based mainly on North American sites. However, in many studies, the pre-experimental cultivation practice is not described and it thus remains uncertain whether the SOC has been in equilibrium or not. This analysis underlines the need for additional on-field measurements for any no-till activity that is to be accounted for in a national carbon accounting system. Mean annual changes calculated from the combined data set are 0.81 % ± 0.24% (95% C.I)

[35] Data cited in Angers *et al.* (1995); Bajracharya *et al.* (1997); Dick and Durkalski (1997); Stockfisch *et al.* (1999); Tebrügge and Düring (1999); Balesdent *et al.* (2000); Vullioud (2000); Zihlmann und Weisskopf (2001)

SOC. Thus, taking a mean soil C stock (0–20 cm) of 40.6 t ha^{-1} for Swiss arable soils, annual accumulation rates of 0.33 ± 0.10 t C ha^{-1} can be derived. This rate agrees almost exactly with that cited by IPCC (2000). Table 13 summarises the C sequestration potential under a no-till system for the current area of no-till, and for two scenarios with either a 50% or 100% share of no-till in arable cropping.

Table 13. Swiss potential for soil carbon sequestration with no-till (1000 t OC a^{-1}).

	No-till in 2001	50% of arable land under no-till	100% of arable land under no-till
Area (ha)	8230	144'670	289'339
Annual sequestration	2.7 ± 0.8	47.7 ± 14.5	95.5 ± 28.9

Potential carbon gains for other conservation tillage systems (e.g. chisel-till), as reported in some studies, are lower than for no-till or have no effect on SOC. The effectiveness of no-till depends on its continuous application since even a single ploughing event may lead to the loss of all additionally sequestered carbon (Stockfisch *et al.* 1999). Therefore, crop rotations where occasional ploughing is necessary (e.g. including potatoes) have no or only a small sequestration potential (see also Buhtz *et al.* 1970).

Apart from the possibility of stored carbon being lost due to occasional tillage, no-till may also result in higher N_2O emissions than conventional tillage. Possible mechanisms for the observed differences are

- a larger pool of substrate N in the mulching layer under no-till and
- a reduction in soil disturbance and aeration, promoting N_2O production (Smith *et al.* 2000a).

Based on data from MacKenzie *et al.* (1998), Smith *et al.* (2000a) estimated that in the worst case 32% of the carbon gained by no-till may be offset by additional N_2O emissions. However, no-till does not lead in all cases to elevated N_2O emissions relative to conventional tillage, as shown by Robertson *et al.* (2000).

Ley-arable farming

A significant contribution to carbon storage can be achieved through ley-arable farming. In particular, in agroecosystems where this type of rotation has not been practised for decades (e.g. in the Mid-Western US), the integration of leys has a significant C sequestration potential (Lal *et al.* 1998). Increases in SOC under ley-arable cropping are due to the higher input of plant residues (mainly by roots) during the ley phase, accompanied by less soil disturbance. Although some of the additional SOC will be decomposed during the subsequent arable crop phase, the mean level of SOC for ley-arable cropping tends to be higher than under exclusive arable rotation systems. The higher residue input is an obvious difference from no-till, where higher amounts of SOC are mainly achieved by reductions in soil disturbance, making the surplus SOM stored under no-till more vulnerable to oxidative losses caused by a change in management.

Use of ley in arable rotations is widespread in Switzerland, with a proportion of 22–39% during the last century. During the first four decades of the 20th century, the share remained relatively constant at around 39%. A minimum of 22% was recorded in 1990, and the figure has increased steadily since then. Future projections assume that the proportion of ley will remain constant at the current level of around 28%. With respect to C sequestration, temporary set-asides in arable rotations can be treated as leys. These account for approximately 10% of the leys in arable rotations (10'700 ha of temporary set-asides are projected for 2020). To assess the likely direction of changes in C stocks due to ley-arable farming by 2020, three scenarios regarding shares of ley/ temporary set-asides were used (Fig. 18):

- "Business as usual": the historical (until 2000) and projected share of ley
- Temporary set-asides included: the projected share of ley + temporary set-asides
- Temporary set-asides + 34% ley: the proportion of ley in 2020 equals that in 1960 (= 100-year mean = 34%) + the proportion of temporary set-asides from scenario II.

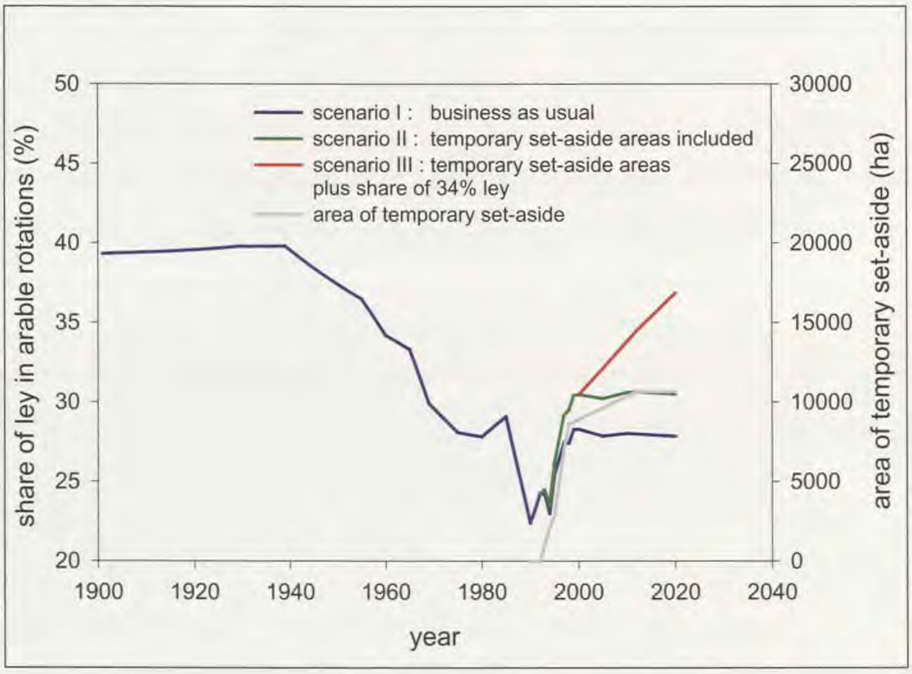


Fig. 18. Proportions of ley and ley + temporary set-aside in three different land-use scenarios, and projected area of temporary set-asides. For details, see text.

The response of C stocks is expected to be proportional to the share of ley if the distribution of each land-use type (arable and ley) is proportional to the distribution of soil types. The time required to reach a new SOC equilibrium is considered to be 100 years (IPCC 2000). During the first half of the last century, an equilibrium of C stocks was achieved in ley-arable rotations, which was subsequently disturbed. Carbon stocks reflected the changes in ley area with a lag of some decades. Carbon sinks due to the increasing share of ley from 1990 onwards did not fully compensate for the previous loss.

To quantify C fluxes resulting from changes in the share of ley, long-term field experiments can be used. Ley-arable rotations were shown to increase C-stocks in several field experiments, mainly conducted in the US (Paustian *et al.* 1997). For Europe, Smith *et al.* (1997) estimated a relative annual increase in SOC of 1.02% for a scenario in which 30% of the area is in a ley phase at any given time, based on data from long-term field experiments (UK, Sweden, Netherlands). This corresponds to a mean annual sequestration rate of 0.54 t C ha^{-1} (IPCC 2000). As a first approximation, a linear relationship can be assumed to exist between the annual sequestration rate of 1.02% for 30% ley, and higher, or lower proportions of ley (with 1% change in ley = 1.02/30

= 0.034% change in C stock). The pattern of the resulting fluxes is plausible and follows the trend shown in Fig. 18, but absolute fluxes become unrealistically high. For example, the introduction of temporary set-asides over the past 10 years would cause larger fluxes than the sequestration rates expected from the conversion of temporary set-asides to permanent grassland. Realistically, higher sequestration rates should result for the conversion to permanent grassland than for the introduction of leys. Therefore, it is suggested that the high proportion of leys in arable rotations contributed to the recent high level of soil C stocks, which is confirmed by the fact that C stocks are significantly higher under temporary grassland than under arable land (see Section 4.3). However, calculation of sequestration potentials using the IPCC data considerably overestimates the real potential for Switzerland of <0.034% for each 1% increase in ley.

Ley-arable farming and temporary set-aside is promoted by the commitments imposed by the REP system. The current conditions under which farmers obtain subsidies by including set-aside land could lead to improvements. Instead of taking these areas out of rotation for only a few years, permanent conversion or set-aside land would be more beneficial for C sequestration, not only due to higher C sequestration rates but also due to lower expected N_2O emissions. Conversely, C sequestration attained by implementing leys may be partially offset by N_2O emissions. Comparing three different arable and intensively used (permanent or temporary) grassland sites, Goossens *et al.* (2001) reported generally higher N_2O emissions from the grasslands. Since Swiss temporary grasslands are used intensively, with fertiliser applications of 130–180 kg N ha^{-1} a^{-1} on the Central Plateau (i.e. 5–6 cuts per year; FAL 2001), N_2O emissions partially offsetting the gains produced by ley-arable farming will play a significant role in the net GHG balance for this practice, excluding CO_2 emissions from the use of machinery.

Production methods and farming intensity

Other measures under the REP system which are relevant for C sequestration are included under the heading of soil protection activities, i.e. ley-arable farming and set-aside areas, plus no-till, cropping (of corn) with under-sowings, cover-cropping and mulching. All these measures are expected to lead to additional C sequestration through enhanced inputs of plant residues, less soil disturbance and longer periods of soil coverage. Enhancing residue returns to the soil is generally believed to be an important way of increasing soil C stocks (Paustian *et al.* 1997). Thus, under REP the use of under-sowings and winter cover crops is likely to increase soil C. The data compiled by Paustian *et al.* (1997) also provided evidence for a positive effect of corn in crop rotations on soil C stocks, if the corn is harvested as grain and not used for silage production. For mulching, Duiker and Lal (2002) reported efficiencies (percentage of C applied in mulch converted to SOC per year) of 10% for no-till, 8% for plough-till, and 0% for ridge-till. Effects of mulching on SOC were significant for the upper 10 cm of the A-horizon. The study site had climatic conditions (mean annual temperature 11°C; mean annual precipitation 932 mm) similar to those of the Swiss Central Plateau. Therefore, mulching may also be expected to have positive effects on SOC stocks under Swiss conditions. However, as for other activities, the exact sequestration rates must be verified under the specific conditions. Additionally, the starting point (SOC in equilibrium or not) must be known. It is not possible to make separate quantitative assessments of mulching, under-sowings and cover-cropping for Switzerland on the basis of the available data.

Alternatively, a whole-farm approach may give additional information about the sequestration potential of different farming intensities. In a German study involving nine different sites, each comparing **integrated vs conventional farming** over a 10-year period, different practices were assessed with respect to their effect on a range of soil parameters (Emmerling *et al.* 2001). This approach is not sufficiently sensitive to detect specific effects of individual measures, but it can be used to assess the response of the whole farming system to extensification[36]. The study included different crop rotations, tillage, fertilisation and soil protection activities. SOC content and microbial parameters showed no significant response to extensification. SOC levels were largely determined by temporal and spatial variability and soil texture. These levels decreased slightly during the first years after implementation of integrated farming and were somewhat higher during the last years of the various studies. Thus, integrated farming cannot be shown to have had a beneficial effect on soil C stocks, but the duration of the investigations might have been too short to detect significant changes.

About 8% of the agricultural area in Switzerland is managed using **organic farming** practices. Here, the practice is referred to as "low-input" farming, with the main characteristics relevant to soil C being restricted input of mineral and organic fertilisers and pesticides. The total area of low-input farming increased steadily from 19'000 ha in 1993 to about 78'000 in 2000. The structure of low-input farming differs from the average values for Swiss agriculture. In low-input farming, arable land accounts for 15% of the total agricultural area, compared with a national average of 38%[37], and the share of ley in arable rotations (47%) is larger than the national mean (28%). Since higher proportions of permanent grassland and ley in arable rotations promote C sequestration (see above), low-input farming can be expected to be favourable for soil C stocks. As for integrated farming, a whole-farm approach provides an initial insight into the potential of low-input farming in terms of C sequestration. Von Lützow *et al.* (2002) compared SOM qualities for integrated vs low-input farming over a 7-year period at a research farm in Southern Germany (Scheyern). Low-input farming was characterised by a 30% share of ley, under-sowing and ploughing, while integrated farming had a four-field crop rotation without ley, but with minimum tillage and winter cover crops. During the observation period, the two farming systems showed similar values for SOC, microbial parameters and decomposable OM. Soil C stocks depended mainly on soil texture, but not on management. This study gives no evidence of any extra C sequestration potential for low-input compared with integrated management. In contrast to the studies by Emmerling *et al.* (2001) and Von Lützow *et al.* (2002), Robertson *et al.* (2000) reported C sequestration for integrated and low-input farming relative to conventional farming at experimental sites in the Midwestern US. They attributed C gains to the integration of legumes as cover crops following maize and wheat.

[36] Extensification measures in the study by Emmerling *et al.* (2001) comprise: intermediate crops, green manure, winter cover crops, conservation tillage, restricted use of N fertilisers depending on spring soil mineral N analysis, adoption of integrated pest management, less intense soil tillage at several locations as compared with the conventional system. The main crops in the conventional system were similar to the IP in this study, while differences were mainly due to (a) higher mineral N fertilisation and (b) non-restricted use of pesticides.

[37] The mountain farming area is converted to an area of agricultural land with equivalent productivity.

To distinguish the effects of low-input farming on SOC from those of crop rotation and tillage, experiments involving low-input plots with identical crop rotations and tillage are necessary. Another important characteristic of low-input farming is the exclusive use of organic fertilisers, which requires a mixed system of cash crops, forage crops and livestock. With respect to organic fertilisers, similar carbon loads and fertiliser types are necessary in order to distinguish the effects of low-input farming on C stocks from effects due to C inputs from organic amendments[38].

These conditions were met by two long-term field trials comparing low-input vs conventional or integrated farming, cited by Raupp (2000) and Fliessbach *et al.* (2000). At these experimental sites (Darmstadt, Germany, and the DOC trial in Therwil, Switzerland), biodynamic farming was compared with conventional (Darmstadt) or with integrated (called "conventional" in the experimental design) and low-input farming (Therwil). Both experiments confirmed the dependence of SOC on manure inputs, showing higher C stocks for the low-input systems with increasing manure application. The experiments also revealed a tendency towards higher SOC contents in biodynamic farming. These differences were statistically significant for Darmstadt, but not for Therwil. In Therwil, differences in C stocks at the beginning of the experiment and significant decreases during three crop rotations made it more difficult to evaluate the effects of low-input farming (Fig. 19).

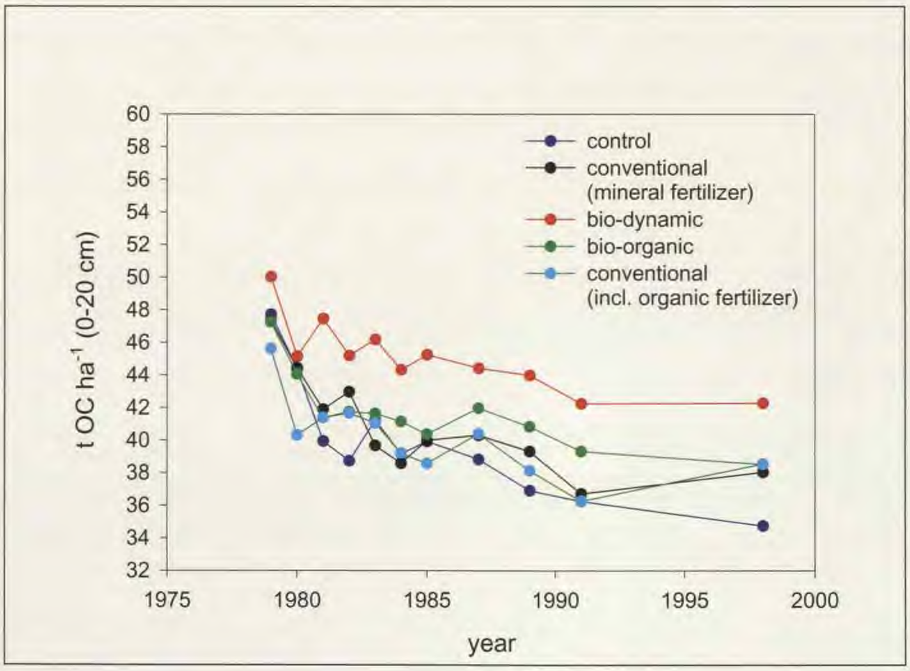


Fig. 19. Amounts of SOC with different farming systems in the DOC trial over a period of 19 years. Rates of application of organic manure are similar for the biodynamic, the bio-organic and the conventional system.

Mean annual SOC loss is highest in the unfertilised control (0.65 t a^{-1}) and in plots treated with mineral fertilisers (0.47 t a^{-1}), but it is in the same order of magnitude for the three organic fertiliser treatments, i.e. conventional, bio-organic and biodynamic (0.36, 0.44 and 0.39 t a^{-1}, respectively). On the other hand, considering the results reported by Raupp (2000) and literature cited therein, and in view of the significantly higher micro-

[38] Discussions of the effects of organic fertilisers on soil C stocks are sometimes misleading, since different qualities of organic residues are compared. For example, farmyard manure has been shown to be effective in enhancing SOC compared with wheat straw application (Paustian *et al.* 1992), but an assessment of such effects should also consider C losses during animal digestion and the storage and composting of farmyard manure. This issue falls within the scope of C balances at the whole-farm level.

bial activities and species diversity at the Therwil site (Fliessbach and Mäder 2000; Fliessbach *et al.* 2000; Mäder *et al.* 2002), long-term effects on SOC stocks may be expected, particularly with biodynamic agriculture, although they remain uncertain. For example, higher microbial activities in Therwil may be due to differences in pH that developed over time. It must be noted that in low-input farming residue returns are smaller due to lower crop yields.

This brief evaluation shows that:

- carbon sink benefits from low-input farming may be due to effects of crop rotations rather than being a result of the system per se and
- the amount of organic fertilisers applied is one of the main factors governing C stocks in both conventional and low-input farming systems. At a higher spatial scale, differences due to organic fertiliser application level off, since each unit of manure is processed at the cost of reduced residue input on the field.

Calculating the net GWP[39] over a 9-year period for arable rotations under conventional, conventional no-till, integrated and low-input farming, Robertson *et al.* (2000) showed that all farming systems were net GHG emitters, but that net GWP was highest for conventional farming and lowest for conventional no-till farming. Emissions from integrated and low-input farming amounted, respectively, to 36% and 55% of those from conventional farming. Differences among farming systems were attributed to fossil fuel inputs and C sequestration in integrated and low-input farming, while N_2O emissions were similar in all farming systems. In addition, emissions of N_2O were in the same order of magnitude for alfalfa as a perennial crop as for the crop rotations. The data reported by Robertson *et al.* (2000) indicated that resource-use efficiency[40] was higher for the integrated and low-input farming systems. Higher resource-use efficiency was also postulated for low-input farming at the Therwil site (Mäder *et al.* 2002).

5.3.2 Grassland management

Grazing

Grazing influences SOC stocks via changes in plant productivity, root allocation of plant C, changes in species composition and changes in physical soil properties. In most studies, grazing has been reported to produce positive effects on grassland soil C. Increases in SOC levels under grazing, as compared with mowing, can be attributed to

- a higher return of ingested nutrients to the pasture as excreta (60–95%, Schnabel *et al.* 2001),
- enhanced physical breakdown and soil incorporation of plant materials, which accumulate in ungrazed exclosures as standing dead and surface litter (Manley *et al.* 1995) and
- a more rapid annual shoot turnover (Reeder and Schuman 2002).

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[39] Global Warming Potential. The cumulative radiative forcing of GHG over a specific time horizon, usually expressed as CO_2 equivalents.

[40] Yield per unit energy input.

Heavier foot traffic results in enhanced breakdown of above-ground litter and its incorporation into the soil (Schuman *et al.* 2001). Upon incorporation into the soil, C turnover rates are slower due to physical protection (see Section 3.2). However, overgrazing will result in damage to the vegetation cover, leading to C losses from soil erosion. Although grazing tends to increase SOC stocks compared with types of management that do not involve livestock, the results are far from being consistent, and several studies indicate carbon losses due to grazing (Potter *et al.* 2001; literature cited in Schuman *et al.* 2001). To evaluate the effect of grazing management on C stocks in Swiss grassland soils, studies considering country-specific conditions would be necessary, including stocking densities, soil and climate. In particular, lower OM return under mowing, as proposed by Schnabel *et al.* (2001), may not apply since the manure is used as organic fertiliser and C is returned to the field.

Grassland productivity

Since the soil C stock is determined by C input and turnover time, improved grassland productivity is expected to increase soil C stocks because of higher inputs. Lal *et al.* (1997b) reported a higher NPP for fertilised than for unfertilised grassland, leading to a higher residue input. This effect was more pronounced in the presence of grasses than with legumes. Nyborg *et al.* (1997) reported increases in SOC due to a higher NPP with increasing N and S fertilisation. They also showed that the light organic fraction ($\rho \le 1.7$ g cm^{-3}) was much more responsive to different mineral fertiliser application rates than total SOC. In a review of 115 studies, Conant *et al.* (2001) showed that conversion of native to cultivated grassland led to a mean relative annual increase in SOC. Using ^{13}C pulse-labelling, Crawford *et al.* (1997) found that grass cutting resulted in lower above-ground NPP compared with the uncut control, but in a higher allocation of assimilated C to roots. This may lead to higher soil C stocks in the long term.

In a few studies, soil C stocks have been investigated for alpine grasslands differing in productivity. Gisi and Oertli (1981) observed higher C stocks at sites in the Swiss Jura Mountains and the Alps under a more extensive grassland use, but the observed differences between intensively and extensively managed grasslands were small, and the data were not analysed for statistical significance. Zeller *et al.* (1997) reported soil C stocks for four Austrian sites with different types of management (1 to 3 cuts per year) at altitudes of 1500 m a.s.l. They found no consistent trend in SOC stocks for the upper 10 cm. Further evaluation of their data is complicated by the fact that two of the sites were characterised by soils from colluvial deposition of different origin, which may confound the *in situ* dynamics. Likewise, Bitterlich *et al.* (1999) found no uniform pattern among three alpine sites under intensively vs lightly managed hay meadows. SOM concentrations tended to increase with decreasing intensity, but these shifts were accompanied by changes in horizon depth, and C stocks were not calculated in this study. Schinner (1978) observed higher decay rates in litter bags buried in an extensive alpine grassland (occasional grazing and mowing) compared with bags placed in an adjacent summer pasture. However, higher decay rates and decomposition could not compensate for the larger amount of litter supply, resulting in residue accumulation and formation of litter horizons.

The studies from alpine regions do not support the hypothesis that extensification leads to a lower NPP and to decreasing SOC stocks. This apparent inconsistency may derive from a difference in the response of NPP and of residue decomposition to temperature. As Burke *et al.* (1997) pointed out for the North American central grassland region, the decay of plant residues is more constrained by temperature, while NPP is more limited by precipitation. If such findings apply to the alpine region, possible management effects on soil C may be masked by limited decomposition. In a study of root production and turnover under two UK grasslands, Fitter *et al.* (1998) observed that root growth and the resulting input of C to the soil was controlled by radiation flux rather than temperature. Therefore, any evaluation of the effects of extensification on SOC needs to take into account multifactorial interactions, as well as different residue qualities associated with changes in grassland species composition. Caution should be exercised when extrapolating findings about C dynamics in grassland systems from one region to another.

Grassland management also affects N_2O emissions. Lower rates of fertiliser application and reduced cutting frequency both tend to decrease N_2O emissions in managed grasslands (Velthof *et al.* 1996; Kammann *et al.* 1998). Area-related GWPs were progressively lower for intensive, extensified and organic grassland at dairy farms in southern Germany (Haas *et al.* 2001). Differences in GWP were attributed to lower N_2O emissions in the field, and to lower CO_2 emissions for extensified and organic grasslands as a result of fossil fuel savings. Consequently, the net effect of extensification on the GHG balance is likely to be negative in these cases (i.e. net sink).

5.3.3 Conversion of arable land

Conversion to permanent grassland

Estimates of the C sequestration potential associated with the conversion of arable land to permanent grassland are based on data presented in Chapter 4. Only sites below 1000 m above sea level are considered here, since arable land use is restricted mainly to the milder climate of the Central Plateau. Changes in C stocks are expected to occur mainly in the upper 20 cm. A first approximation was based on the difference in mean C stocks for this upper horizon between arable land (40.6 t OC ha^{-1}) and permanent grassland (49.9 t OC ha^{-1}; weighted mean for sites <1000 m above sea level). For an estimated time to reach new equilibrium of 50 years (IPCC 2000), annual C sequestration rates would be around 0.19 t OC ha^{-1} a^{-1}. This sequestration rate is distinctly lower than the estimate of 0.5–1.0 t OC ha^{-1} a^{-1} proposed by IPCC (2000), and even below the rate of 0.54 t OC ha^{-1} a^{-1} estimated for a 30% share of leys. However, it must be considered that permanent grasslands are currently located on soils with a carbon storage capacity lower than that of arable soils. Comparison of current and hypothetical C stocks (see Section 4.4) indicates that the sequestration potential in Swiss agriculture is limited as long as soils with high C sequestration potential are used for arable cropping instead of grassland. Permanent grasslands installed at sites currently used as arable land are expected to store 62.3 t OC ha^{-1} (0–20 cm; 113.3 t OC ha^{-1} 0–100 cm), or 21.7 t OC ha^{-1} (0–20 cm) more than their arable counterparts. For a 50-year time horizon, C sequestration rates with the second, more realistic, assumption amount to 0.43 t OC ha^{-1} a^{-1}. This rate is closer to the IPCC value, but still at the lower end. Current C stocks in arable soils are relatively high due to the large proportion of leys, as well as the high rate of residue and manure return. Hence, the present situation already promotes soil C storage and thus limits the future C sequestration potential of conversion to permanent grassland.

In order to identify the most suitable sites for conversion of arable land to permanent grassland with a view to C sequestration, it is necessary to consider soil texture, which has a strong influence on the SOC content of both arable and permanent grassland. Figure 20 shows how the C sequestration potential for the conversion of arable land to grassland decreases as soil clay content increases. The sequestration potential for different clay contents as derived from the soil data base follows the exponential function:

$$C \text{ sequestration} = 20.19 + 3.94 \times \exp(-0.042 \times \% \text{ clay}) \quad [\text{t OC ha}^{-1}] \quad [11]$$

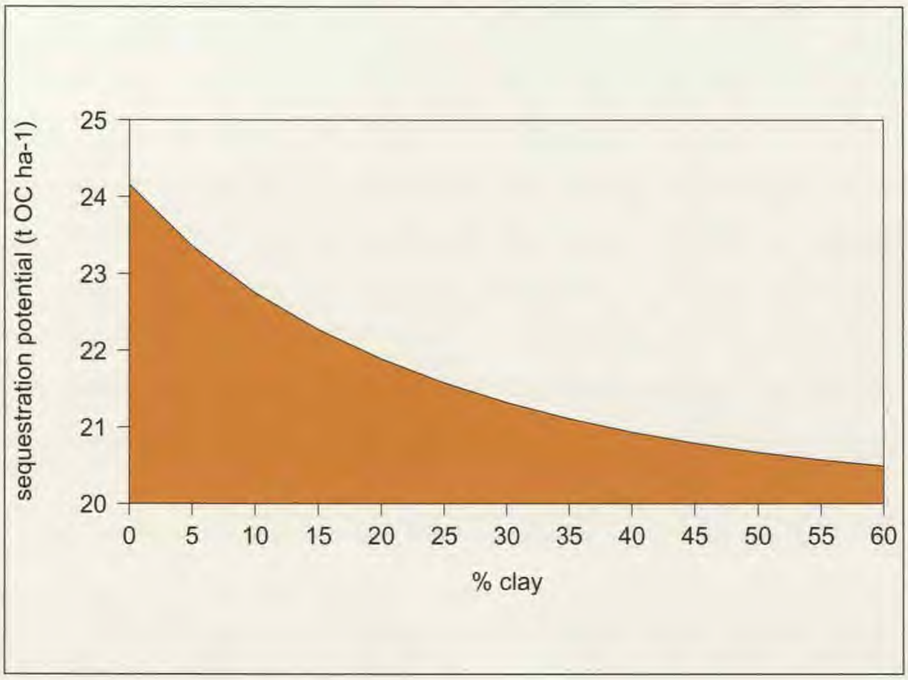

Fig. 20. Estimated sequestration potential for the conversion of arable land to permanent grassland in relation to soil clay content.

Carbon sequestration potentials for the conversion of arable land to permanent grassland are calculated for different soil textures using the data for estimated C stocks given in Section 4.3 (Table 14). The area estimates for this calculation correspond to the total area of temporary set-asides, on the assumption that they would be managed as permanent grassland. The analysis shows a significant C sink from the year 2000 on, when the area of set-asides stabilises at a high level.

Table 14. Annual sequestration rates (t OC ha^{-1} a^{-1}) and C sequestration potentials (1000 t OC) for different time periods of the conversion of temporary set-asides in agricultural rotations to permanent grassland, in relation to soil clay content (calculated for a 50-year time horizon).

Soil texture	Annual sequestration	1990-2000	1990-2020	2000-2020	2008-2012
10% clay	0.46	20.2	112.9	96.7	23.6
25% clay	0.43	19.7	107.1	91.7	22.4
40% clay	0.42	18.6	103.9	89.0	21.7

Set-aside land

The term *set-aside* is defined as the exclusion of agricultural land from any further cultivation. In most regions of Switzerland, long-term set-aside will result in the regrowth of dwarf shrubs or forest within a few decades. Whether this is classified under the heading of agricultural (conversion) or forest (reforestation) activities is a matter of definition. It has been estimated that over the past 20 years 18'000 ha of mountain farming has been abandoned, with subsequent natural regrowth of dwarf shrubs and forest (Swiss Federal Statistical Office 2001).

The relevant areas for this activity are located in subalpine and alpine meadows and pastures, and forest re-growth – also referred to as human-induced promotion of natural seed sources *(Vergandung)* – is part of forestry. However, the initial period of set-aside will involve management-induced changes in vegetation and productivity, particularly at sites where fertilisers have been applied before the activity started. In general, set-aside will alter the pedological conditions, with higher C:N ratios of the residues, decreasing soil reaction, and lower shoot:root ratios. Decreasing soil reaction will result in enhanced podzolisation on silicate rock and in leaching of dissolved OC down the profile. This process is most evident at higher altitudes with correspondingly high precipitation. Along the succession of intensively and extensively managed grassland, adjacent abandoned areas and forests in the Swiss Jura mountains and the Alps, Gisi and Oertli (1981) found an oscillation of several soil parameters (OC, N, pH). Soil C stocks increased with extensification, but decreased again from extensive grassland to set-aside land characterised by *Vaccinietum* vegetation. Succession to forest would further decrease SOC stocks, but increase the C stocks of litter horizons. A similar result was found in the region of the Vals valley (Bouma *et al.* 1969). These authors reported slightly lower SOC concentrations for the upper 20 cm of a *Vaccinium-Picea* set-aside area, compared with a nearby pasture. Bitterlich *et al.* (1999) observed higher SOC concentrations for forest sites, compared with adjacent set-aside patches and meadows. However, as already discussed above, the results reported by Bitterlich *et al* (1999) correspond to the A-horizon with variable depth, and the influence of changing horizon depth during succession was not accounted for.

Robertson *et al.* (2000) compared the contribution of individual GHGs to the GWP in different agricultural systems of the Midwestern US. Of all the ecosystems investigated – including integrated farming, low-input farming, perennial alfalfa, and early, mid, and old successional forests – the system reported to have the most negative GWP (i.e. net sink) was an early successional management (2–10 years after abandonment of arable cropping). The net sink was attributed to a soil C sequestration rate of around 0.6 t C ha^{-1} a^{-1}, and to lower N_2O emissions. These results indicate that set-aside programmes yield benefits for both C sequestration and net GWP.

The effects of set-aside on C sequestration are still far from being fully understood. Apart from the uncertainty regarding SOC stocks, the accumulation of organic horizons (e.g. how are these to be treated in C stock accounting ?) and intensified podzolisation make it difficult to assess the C sequestration potentials.

5.3.4 Wetland management

Peatland protection

Intact peatlands are important sinks for carbon and nutrients, often accompanied by high NPP. For example, the dry matter yield of reeds of extensively used peatlands in temperate zones is as high as 6–12 t ha^{-1} (Schäfer and Degenhardt 1999), which is comparable to the yield of intensively managed grasslands on mineral soils. The NPP of ombrotrophic raised peatlands is distinctly lower, and C accumulation in this system is due not only to lack of oxygen but also to the low nutrient status.

In Switzerland, about 20'000 ha of intact peatlands are currently under moderate or extensive grassland cultivation (see Section 4.2). They began to be utilised as extensively grazed (wooded) meadows several centuries ago. Later, between 1860 and 1914, they were used as traditional litter meadows, mainly in the Central Plateau. Since 1885, large areas of former peatlands have been destroyed by drainage, cultivation and peat harvesting.

Peat accumulation rates depend on the peatland type, climatic conditions and hydrological status. Mean accumulation rates (± 95% C.I.) calculated from 23 data sets from studies[41] in temperate and cold climates are 0.45 (± 0.2) t OC ha^{-1} a^{-1}, but they can be as low as 0.16 t OC ha^{-1} a^{-1} (Kasimir-Klemedtsson *et al.* 1997). Based on literature data, the total current rate of C sequestration for intact peatlands in Switzerland is estimated at 9 (± 4) x 10^3 t OC a^{-1}.

Intensive peatland management leads to peat degradation and, consequently, to CO_2 emissions. A degree of protection can be achieved by maintaining a higher water table, e.g. in semi-natural grasslands and pastures. Organic soils used as drained grasslands emit about half the amount of CO_2 emitted by organic soils used for arable cropping (Freibauer and Kaltschmitt 2000). Since intact peatlands are emitters of methane (CH_4), and may become an important source of N_2O during cultivation, a total GHG balance is needed in order to fully assess the contribution of peatlands to GHG mitigation (see below).

Peatland restoration

Restoration of cultivated organic soils to intact peatlands provides an opportunity to sequester additional carbon due to growth of the peat body combined with the reduction of C losses from peat oxidation (see Section 4.2). Conditions for the restoration of intensively managed organic soils include:

- a minimum peat thickness of 1 m and
- the option (hydrological, ecological and social) of raising the water table up to the surface (Pfadenhauer 1999).

A field study from Finland (Tuittila *et al.* 1999) confirms the opportunity to restore cut-away peatlands, even with highly decomposed *Sphagnum-Eriophorum* peat, and to turn a former CO_2 source into a CO_2 sink within a few years. Volk (1972) showed that the relative rate of peat decomposition was up to 5 times higher for

[41] Armentano and Menges 1986; Augustin *et al.* 1996; Bergkamp and Orlando 1999; Gorham 1991; Kasimir-Klemedtsson *et al.* 1997; data cited in Schäfer and Degenhardt 1999; data cited in Schopp-Guth 1999; Tolonen and Torunen 1996; Tuitilla *et. al.* 1999.

a water table depth of 25 cm, compared with 5 cm. The relationship between decomposition and water table depth also holds for deeper groundwater tables, as demonstrated by Mundel (1976) for cultivated German fens (Fig. 21).

This study also provided evidence of a coupling of peat thickness and decomposition rates, with deep groundwater tables being accompanied by reduced CO_2 emissions, probably due to desiccation. Under higher water table conditions, decomposition still proceeded, but at a lower rate. Since utilisation as pasture is less restricted by a high water table than arable use, conversion of organic soils currently used as arable land to pastures, with a concomitant rise in the water table, would probably decrease the rate of peat decomposition.

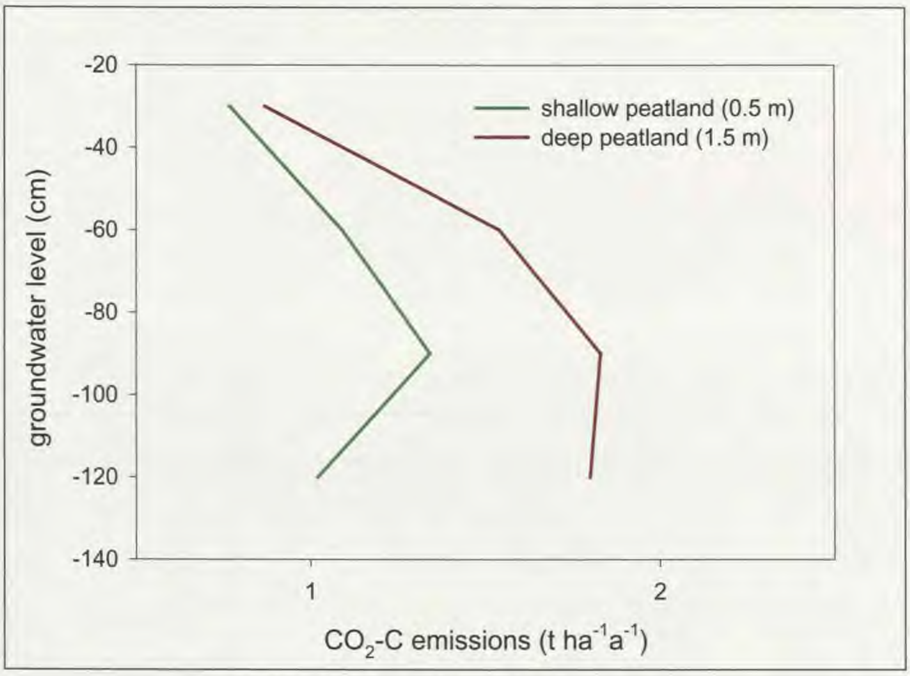

Fig. 21. Annual oxidative carbon losses for two fens with different water table depths. Data adapted from Mundel (1976).

Actual C losses, avoidable C emissions and potential C sequestration in Swiss peatlands can be calculated for different management options (Table 15). One underlying assumption is the technical feasibility of peatland restoration, which still has to be investigated.

Five scenarios have been evaluated:

- business as usual (no change in the area of cultivated peatlands)
- restoration of 5% of cultivated peatlands,
- restoration of 20% of cultivated peatlands,
- complete restoration of cultivated peatlands (100%) and
- conversion of all cultivated peatlands into permanent grassland.

For scenarios I-IV, flux data for CO_2-C were taken from Section 4.2. Mean fluxes were calculated from data obtained for agricultural practices ranging from intensive agriculture to extensive grassland management. For scenario V (conversion to permanent grassland), literature data were used for oxidative losses from grazed organic soils.

A change in the management of organic soils has a significant potential for the avoidance of C losses and for additional C sequestration, depending on the restoration scenario. Restoring 5% of the area of currently managed organic soils to intact peatlands would reduce C fluxes by 5–12 x 10^3 t OC a^{-1}, and restoring 20% would lead to a reduction of 18–51 x 10^3 t OC a^{-1}. Extensification (conversion to permanent grassland) would result in even greater emission savings of 52–191 x 10^3 t OC a^{-1}. The restoration of all organic soils to intact peatlands would result in C accumulation rates of 3–14 x 10^3 t OC a^{-1}, plus another 88–257 x 10^3 t a^{-1} from CO_2-C savings. These estimates involve major uncertainties for both fluxes and areas. But even the most moderate scenario of 5% restoration would have an effect equal to the current rate of C sequestration in intact peatlands.

Table 15. Annual OC loss (-) and gain (+) (1000 t OC a^{-1}) for cultivated peatlands under various restoration and conversion scenarios[1].

Scenario	I	II	III	IV	V
	Business-as-usual	5%	20%	100%	Converting all managed organic soils to grassland (extensification)
Oxidative loss	-88 to -257	-83 to -245	-70 to -206	±0	-36 to –66
Sequestration	±0	0.2 to 0.7	0.6 to 2.9	3 to 14.3	±0
Net C flux	-88 to -257	-83 to -244	-69 to -203	3 to 14.3	-36 to -66

[1] The area of managed organic soils is taken to be 17'000 (12'000 to 22'000) ha (see Table 5), rates for oxidative peat losses are 9.5 ± 2.2 t OC ha^{-1} (± 95% C.I.; see Section 4.2), and the sequestration rate for restored organic soils is 0.45 (± 0.2) t OC ha^{-1}. A mean oxidative loss of 3 t OC ha^{-1} a^{-1} was assumed for drained grassland (Kasimir-Klemdtsson *et al.* 1997). Other GHGs are not considered in this scenario. Fluxes are given as upper and lower limits. For references see Sections 4.2 and 5.3.5.

Intact peatlands are important sources of CH_4, and cultivated peatlands may become a significant N_2O source (Hensen and Vermeulen 1999). Since fluxes of all three GHGs are generally higher for organic soils than for mineral soils, calculation of the net GHG effect is essential in relation to any peatland activity. Ranges for annual fluxes of the three GHGs for intact peatlands, and for different management options, are given in Table 16, based on reviews by Kasimir-Klemedtsson *et al.* (1997) and Augustin *et al.* (1996).

Table 16. GHG fluxes from peatlands expressed as CO_2 equivalents. (t ha^{-1} a^{-1})[1]. (Negative numbers denote net release from the soil.)

	CO_2		N_2O		CH_4	
	a[2]	b[2]	a	b	a	b
Undrained fen	0.6	0.5 to 8.2	±0	-0.06 to -0.4	-4.9	-0.3 to -26.6
Undrained bog	0.9	0.7 to 2.6	±0	-0.03	-2.5	-0.3 to -6.3
Drained fen		-10.6 to -24.5		-0.8 to -3.2		±0 to -0.01
Drained bog		-4.2 to -12.9		-0.03		-0.01 to -0.3
Drained grassland	-11(±4)		-2.9		-0.002	
Drained cereals	-20		-4.8		-0.03	
Drained row crop	-70		n.d.		n.d.	

[1] Calculated using GWPs according to IPCC (1996a).

[2] Data source: a = Kasimir-Klemedtsson et al., 1997; b = Augustin et al., 1996. n.d. not determined.

The flux data in Table 16 are characterised by wide ranges, making a precise quantitative assessment of peatland activities difficult. Nevertheless, the data show that intact peatlands are always C sinks and CH_4 sources. The net effect in CO_2 equivalents is negative or positive; i.e. C sequestration is in some cases counterbalanced by CH_4 emissions. Methane emissions are higher for fens than for bogs. Conversion of land to agriculture by drainage results in a shift to CO_2 and higher N_2O emissions, while CH_4 emissions decline. The net emission is negative for all types of land use. Taking the highest measured CH_4 emission from the review by Augustin *et al.* (1996), the net warming potential of intact fens would exceed that of drained fens. However, based on measurements by Bridgeham *et al.* (1995), this is an extreme value, and average CH_4 fluxes are at least one order of magnitude smaller. Thus, it is suggested that in most cases the net flux of cultivated peatlands exceeds that of intact peatlands, and that peatland restoration provides a way of mitigating the net greenhouse gas effect, as long as oxidative peat loss occurs due to cultivation of organic soils.

In the medium term (100–200 years), the CO_2 and N_2O source from cultivated peatlands will be exhausted, and emissions will reach rates typical of mineral soils, while recultivated or intact peatlands will remain a net CH_4 source as long as the hydrological and ecological conditions favour peat accumulation (long term, several hundred years). However, the time required by cumulative CH_4 emissions of the intact or recultivated peatland to reach the level of CO_2 emitted from the cultivated peatland may exceed the time horizon of the commitment period of the Kyoto Protocol.

5.3.5 Summary of relevant activities under Swiss conditions

Today, only the area under no-till land represents a significant C sink (<3 x 10^3 t OC $a^{-1)}$. Intact peatlands sequester another 9 x 10^3 t OC a^{-1}, but this is not regarded as a human-induced activity. A second realistic option would be to include the conversion of temporary set-asides to permanent grassland, with a sink potential of 4.5–4.9 x 10^3 t OC a^{-1}.

Much larger potentials could be achieved by expanding the area under these activities, and from the restoration or extensification of cultivated organic soils. Theoretically, all arable land could be converted to no-till or, alternatively, to permanent grassland, and all organic soils could be either restored or used as extensive grassland. This hypothetical maximum increase in the area under selected activities would generate the following C sink potentials:

- **No-till: 96 x 10^3 t OC a^{-1}**
- **Conversion of all arable land to permanent grassland: 122−133 x 10^3 t OC a^{-1}**
- **Conversion of all cultivated peatlands to grassland with raised groundwater table:**
- **36−66 x 10^3 t OC a^{-1} or, alternatively,**
- **Restoration of cultivated peatlands with high groundwater tables of natural peatlands:**
- **91−271 x 10^3 t OC a^{-1}**

The hypothetical options represent significant potentials when compared with the total CO_2 emissions in the Swiss Greenhouse Gas Inventory (see Section 8.1). The agricultural activities discussed in this Chapter and the corresponding C sink potentials are summarised in Table 17.

Table 17. Summary of agricultural sequestration potentials. Units: Area (ha), sequestration rate (t C ha^{-1} a^{-1}), sink (1000 t C a^{-1}). For organic soils, emission savings from wetland restoration or extensification are presented as sinks.

Activity	Area	Sequestration rate	Actual sink	Maximum sink	Other GHGs[1]	Comments
Cropland management						
No-till[2]	8'230 289'339	0.33 ± 0.10	2.72 ± 0.82	 95.5 ± 28.9	+ N_2O	a
Ley-arable farming	n.n.[3]	n.n.	n.n.		+ N_2O	b
Farming intensity	1'072'492[4]	n.n.	n.n.			
Grassland management						
Grazing management	n.n.	n.n.	n.n.			c
Grassland productivity	n.n.	n.n.	n.n.		- N_2O[5]	c, d
Conversion of arable land						
Conversion arable → permanent grassland	10'700[6] 289'339[7]	0.42 – 0.46	n.n.	4.5 – 4.9 122 – 133	± N_2O	e
Set-aside	14'499[8]	n.n.	n.n.		± N_2O	f
Wetland management						
Restoration of organic soils	n.n. 17'000[9] ± 5'000	0.45 ± 0.2 plus 9.5 ± 2.2	n.n.	 91 – 271	- N_2O + CH_4	g
Conversion organic soils → permanent grassland[3,10]	n.n. 17'000 ± 5'000	6.5	n.n.	 36 – 66	n.n.	h
Total annual sink			**2.72 ± 0.82**	**309** **(213 – 404)**		

[1] Negative sign: sink/avoided emission, positive sign: source; [2] Area in 1999 or total arable area (mineral soils); [3] n.n. = sink potential not known or activity currently not implemented; [4] Area under REP in 2000 (includes organic farming); [5] Negative N_2O effect for extensification; [6] Projected area of temporary set-asides in 2020; [7] Arable area (mineral soils) in 1999; [8] Set-aside of mountain farming areas with forest regrowth between 1979/84 and 1993/97;[9] Area of cultivated organic soils; [10] data calculated for mean emissions of different agricultural land-use types.

Comments: **a** Maximum area refers to 100% no-till; N_2O effect is likely; **b** N_2O effect is possible; **c** Sequestration is uncertain; **d** N_2O effect is likely; **e** Potential, if temporary set-asides were to become permanent! N_2O effect: positive under intensive grassland use, negative for long-term set-asides; **f** N_2O effect negative, if grassland were used intensively; **g** Sequestration = sum of sequestration and emissions avoided; net effect (CO_2, + N_2O + CH_4) is estimated to be negative for at least 200 years, maximum area for this activity: 17'000 ha (12'000 to 22'000); **h** 6.5 t OC ha^{-1} a^{-1} emission reduction for conversion → permanent grassland; assumption: current use of organic soils as arable and grassland

Hereafter, 309 x 10^3 t OC a^{-1} is designated as the mean theoretical sequestration potential, and 404 x 10^3 t OC a^{-1} as the maximum theoretical sequestration potential (bottom row).

5.4 Additional aspects of carbon sequestration

Enhancing soil C stocks is not only a way of mitigating the greenhouse effect. SOM is an important factor in soil quality by virtue of its effects on physical (change in soil structure, aggregation, porosity, water retention), chemical (nutrient storage and availability) and biological properties (population and species diversity, microbial biomass, mineralisation of nutrients, degradation and detoxification of organic pollutants). The protection of SOM levels thus prevents erosion and degradation, and promotes high yields in agricultural soils. In extensive grasslands and in forest ecosystems, accumulation of organic litter horizons may have conflicting effects, by promoting acidification and leaching of dissolved carbon, causing reduced microbial activities. Litter accumulation and concomitant podzolisation have been observed in extensively managed alpine permanent grasslands (see Section 5.3.2).

Depending on the type of activity, removal of atmospheric CO_2 by sequestration may be offset by enhanced emissions of N_2O or CH_4. Human-induced activities aiming to sequester C by increasing crop yields and/or residue returns are expected to promote denitrification, since the high N turnover under intensive agriculture is accompanied by a sufficient supply of substrate C. This applies to conservation tillage and to the situation with a higher proportion of intensively used leys in arable rotations. In these cases, the net GHG effect is reduced or even completely offset by N_2O emissions. Given the increasing recognition of environmental problems associated with the loss of excessive quantities of N to the environment (Jenkinson 2001), it would seem difficult to argue for a greater use of inorganic N fertilisers as a means of increasing the C sink in soils through stimulated NPP. Therefore, grassland extensification may be a promising option for reducing N_2O emissions. Grassland extensification may also have effects on soil C stocks, but such effects still need to be assessed under Swiss conditions. Increasing CH_4 emissions are to be expected following the restoration of cultivated peatlands to intact peatlands with high groundwater tables. Though the net effect of this activity will be positive (i.e. net sink) for as long as the decay of the organic matter would otherwise last, CH_4 emissions in the long-term (hundreds of years) will offset the emissions avoided. The situation is less clear in the case of an extensification activity, e.g. conversion of cropped organic soils to grazed or mown organic soils.

A full GHG budget for agriculture at the farm or regional level should also include CO_2 equivalents released by the consumption of fossil fuels for land management, production and transport of fertilisers, lime, CO_2 and CH_4 production by livestock, etc. For such a life-cycle assessment, the contribution of any activity to C sequestration must be discounted by CO_2 emitted during the production process. These so-called "carbon costs" (Schlesinger 1999) can shift the C balance of any activity significantly in the opposite direction, as demonstrated for nitrogen fertilisation under different crop rotations, and for irrigation in semi-arid areas. In addition, the application of manure is not likely to result in a net sink for C in soils, since greater SOC levels in manured fields can be expected to be associated with lower inputs of plant residues on a proportionally larger area of off-site lands used for fodder production (Schlesinger 2000). Conversely, the reduction in fossil fuel consumption under conservation tillage shifts the CO_2 balance of this activity towards carbon benefits. System-level GHG assessments of this kind are beyond the scope of this report, but they clearly need to be considered in future evaluations of measures in the agricultural sector.

Some of the activities discussed in Section 5.3 affect biodiversity. Since the implementation of land-use and land-use change activities should contribute to the conservation of biodiversity according to the COP accords of Marrakesh[42], C sequestration activities are discussed briefly from the viewpoint of biodiversity. Positive effects, leading to a win-win situation, would be desired. Integrated management practices imply an extensification of agriculture. There is a general agreement that cropland extensification increases floristic as well as faunistic biodiversity (Mclaughlin and Mineau 1995; Hansen *et al.* 2001). However, benefits are not explicitly correlated with C sequestration activities such a no till, but rather with additional environment protecting measures in the context of integrated production, like reduced pesticide input and temporary set-aside programs (Hyvonen and Salonen 2002). Grassland extensification may lead to improvements in biodiversity (Haas *et al.* 2001). Plant species richness can be increased under certain conditions by complete conversion to a low-input/low-output regime, which includes no fertilisation during decades, as well as reducing cutting frequencies to one or two cuts a^{-1} (Baumberger 1996; Nösberger and Kessler 1997). Such extensification measures are successful on dry and calcareous soils with a large seed bank typical for the Jura mountains. By contrast, the soils of the Central Plateau have been intensively managed over decades which makes a satisfactory removal of accumulated nutrients practically impossible. Under such conditions grass-cut reduction and low-input management have often only small effects on plant diversity, and the floristic composition is dominated by a few species which prefer sites with high nutrient availability (Fisher and Rahmann 1997). Plant species number can therefore only be increased by over-sowing with seed mixtures or hay flowers (Koch 1996). Similar problems may occur by transforming arable land to extensively managed grassland. Grazing generally has a positive impact on plant diversity due to micro-patterns of grazing, trampling and dung deposition. A comparison of grazed and mown calcareous grassland plots showed a slightly higher floristic diversity in cattle-grazed plots (Schläpfer *et al.* 1998). With high stocking rates, the benefits of grazing will turn into disadvantages due to over-grazing and erosion, which results in a rapid decrease in biodiversity (Fisher and Rahmann 1997). One important question is whether grassland plant diversity directly influences soil C stocks. Although several large investigations on ecosystem functioning of biodiversity have been carried out (Hector *et al.* 1999; Tilman *et al.* 2001), there is a lack of information on relationships between soil C storage in grasslands differing in plant diversity (Schmid *et al.*, in press).

Peatland conservation and restoration to enhance C sequestration would have positive incidental benefits for biodiversity, as the extinction of peatland habitats means the extinction of the whole species community. Peatland species are dependent on the specific conditions of their habitat. Their survival can only be secured by determined efforts to protect the remaining wetland areas (Grünig 1994). Successful restoration mainly depends on the condition of the residual peatland – eutrophication level, water regime, seed bank and connectivity with adjacent intact peatlands. But the biodiversity restoration process may take decades, and the original state of biodiversity will never be attained (Schopp-Guth 1999).

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[42] UNFCCC/FCCC/CP/2001/13.Add.1

6. Indirect human-induced and natural effects on soil carbon

Soil carbon is not only influenced directly by human interventions, e.g. in agriculture, but also indirectly via effects on the global atmospheric system, with consequences for ecosystem processes such as NPP or residue decay. This indirect route involves the increase in atmospheric CO_2 concentrations, global warming resulting from increasing GHG emissions, or increasing emissions of reactive nitrogen, which is deposited to N-limited ecosystems. According to the IPCC (2000), intent and foreseeability may be of relevance in distinguishing between direct and indirect human-induced activities. Indirect human-induced effects cannot be taken into account in national accounting systems, according to the Marrakesh Accords of the UNFCCC[43].

Natural effects include annual variation in the cycling of terrestrial carbon[44], inter-annual variability associated with natural climate variations, and variations at the regional scale induced by disturbance and recovery of ecosystems (fire, flooding) (IPCC 2000). These natural effects cannot be considered here in detail, as they do not contribute significantly to C stock changes in the short to medium term (e.g. annual growing season of agricultural crops), or are often limited to small temporal and spatial scales (floods, storms). Natural effects need to be taken into account when GHG sources and sinks are to be attributed to human-induced activities for a specific area under observation.

6.1 Climate change

Climate change will impact on SOM quality and quantity directly, through temperature- and precipitation-mediated effects, and indirectly, by inducing changes in land-use patterns. Such changes can have far-reaching implications for agriculture. Because soils are major sinks/sources in the global terrestrial carbon cycle, considerable scientific attention has been paid to the impact of climate change on SOM turnover (for relevant findings see Rounsevell *et al.* 1999, and references cited therein):

- Changes in SOM depend on the balance between carbon inputs and carbon losses through decomposition. In the absence of other climate change factors, increasing atmospheric CO_2 alone will increase SOM (see also Section 6.2).
- SOM increases with water availability and decreases with temperature.
- Increasing temperature will strongly stimulate OM decomposition, and carbon is likely to be lost from soils in the cooler regions where decomposition rates are currently slow.
- In general, decomposition is more likely to outstrip NPP with elevated temperatures, leading to reduced soil carbon contents, except where limited soil water supply affects decomposition rates.

The catalogue of possible effects of climate change on soil carbon storage reveals some important implications for Swiss agriculture. In view of the results discussed in Section 4.5, the current distribution of soil carbon along the altitudinal gradient in the Swiss Jura Mountains and the Alps strongly supports the hypothetical re-

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UNFCCC FCCC/CP/2001/13/Add.1, 21 January 2002 sponse of soil C stocks to climate change. If only the projected change in temperature is considered, the current distribution of C stocks with altitude, and thus with temperature, may serve as an empirical tool for the estimation of changes in C stocks due to climate change. A more sophisticated approach would involve the use of process-based ecosystem models (e.g. Riedo *et al.* 2001).

The following empirical estimate of shifts in soil C stocks for permanent grasslands is based on the soil data presented earlier (Section 4) and on recent downscaling of expected changes in regional temperatures. The calculated shifts in soil C stocks refer to the current distribution of permanent grassland across soil classes and altitudinal gradients. For the adaptation of the C stocks to elevated temperatures, it was assumed that a new equilibrium would be achieved, together with the temperature increase, by the end of the 21st century. The possibilities of a delayed response of SOC, non-equilibrium conditions and changes in precipitation patterns are not considered. The altitudinal gradient of SOC mainly occurs at more than 1000 m above sea level, and soil C stocks under elevated temperatures were only calculated for these higher altitudes. The response of lowland soil C stocks to climate change cannot be determined with the available soil data.

6.1.1 Soil carbon concentrations and mean annual temperature

Equation 6 was modified to a single regression between OC concentration and altitude:

$OC = 1.56 + 0.0028 \times altitude$ (m a.s.l.) [%]; $R^2 = 0.41$ [12]

The mean annual temperature was then expressed as a function of altitude, using data published by Z'graggen (2001):

$MAT = 11.36 + (-0.0049 \times altitude)$ [°C]; $R^2 = 0.89$ [13]

Solving [12] for the altitude and inserting in [11] yields:

$OC = 8.05 - 0.57 \times MAT$ [%] [14]

Thus, OC can be obtained from temperature. Equation 13 shows that OC decreases by about 0.57% for each 1°C increase in MAT, corresponding to a temperature gradient of $0.49°C \times 100\ m^{-1}$.

6.1.2 Climate change projections for Switzerland

Projections for the mean annual temperature during the 21st century were obtained from the 18 regional climate scenarios provided by D. Gyalistras (personal communication, 2002). The regional climate scenarios are given on a 5 x 5 km grid covering the whole of Switzerland. They were derived using a statistical downscaling technique (Gyalistras *et al.* 1998; Gyalistras and Fischlin 1999) from 18 global scenarios published by IPCC (2001a).

[44] Atmospheric CO_2 concentrations decrease during the Northern Hemisphere spring and summer, and increase during autumn and winter.

The global scenarios represent 6 general circulation models (GCM) and 4 emissions scenarios denoted by GHG (greenhouse gas forcing corresponding to a 1% increase in CO_2 up to 2100), GHG-A (same as GHG, but including the effects of aerosols), A2 and B2 (two of the scenarios included in the IPCC Special Report on Emissions Scenarios/ SRES; see IPCC 2001c).

The four SRES B2 scenarios project the lowest increase, ranging from about 1°C (0–500 m above sea level) to about 1.5°C (2500–3000 m above sea level). The SRES A2 scenarios (greenhouse gas forcing, greenhouse gas forcing + sulphate aerosols) show a larger increase and a more substantial spread, with a range by 2099 from 1.5°C to 2.5°C (0–500 m above sea level), and from 1.5°C to 3.5°C (2500–3000 m above sea level). For altitudes >1000 m above sea level, the temperature increase by 2100 ranges from about 1–1.5°C for the four B2 scenarios, to about 1–3°C for the GHG and GHG-A scenarios. Therefore, the minimum and maximum temperature increase by 2100 can be estimated at 1°C and 3°C. These two values are used to assess changes in carbon stocks.

6.1.3 Projected decline in soil carbon stocks

Projected changes in carbon stocks by 2100 are presented in Table 18. Temperature increases of 1°C and 3°C were considered, to cover the range of projected temperature changes.

Soil C stocks are estimated to decline by 0.45–1.25 Mt OC for favourable, and by 2.36–6.50 Mt OC for unfavourable grasslands. This corresponds to a decrease of 9.0–11.4% for a projected increase in MAT of 1°C, and to a decrease of 24.7–31.3% for a projected increase of 3°C. Soil C stocks are more constrained by fine earth C concentrations in favourable grasslands, while C stocks are limited by stone contents and – to a certain extent – profile depths in unfavourable grasslands.

Table 18. Current and projected soil carbon stocks (0–20 cm) of favourable (F) and unfavourable (UF) permanent grasslands in relation to increases in mean annual temperatures by 2099. Data refer to altitudes ≥ 1000 m above sea level.

	Current C stock		+1°C		+3°C	
Land use	F	UF[2]	F	UF	F	UF
Mt C	4.0	26.3	3.6	24.0	2.8	19.8
t C ha^{-1}	51.9	47.7	46.0	43.4	35.6	35.9
Relative change (%)			-11.4	-9.0	-31.3	-24.7
Mean annual decrease (1000 t OC)[1]			4.5	23.6	12.5	65.0

[1] Calculated for a 100-year time horizon. It is assumed that C stocks will achieve equilibrium with the new temperature conditions by the end of the 21st century.

[2] Including mountain farming.

The expected decline in soil C stocks of permanent grasslands at more than 1000 m above sea level corresponds to annual CO_2 emissions ranging from 28.1 x 10^3 t OC (+1°C) to 77.5 x 10^3 t OC (+3°C). Thus, climate change may reduce the mean theoretical sequestration potential estimated for Swiss agriculture (309 x 10^3 t OC a^{-1}) by 9–25%.

It is possible that increasing MAT will also affect C stocks of permanent grasslands below 1000 m above sea level, but only a weak statistical relationship exists between SOC and temperature for lower altitudes. Soil respiration as a measurement of decomposition activity is related to temperature not linearly, but exponentially. From this relationship, the relative temperature sensitivity of soil respiration was calculated for the temperature range of -10°C to 30°C by Lloyd and Taylor (1994) (Fig. 22).

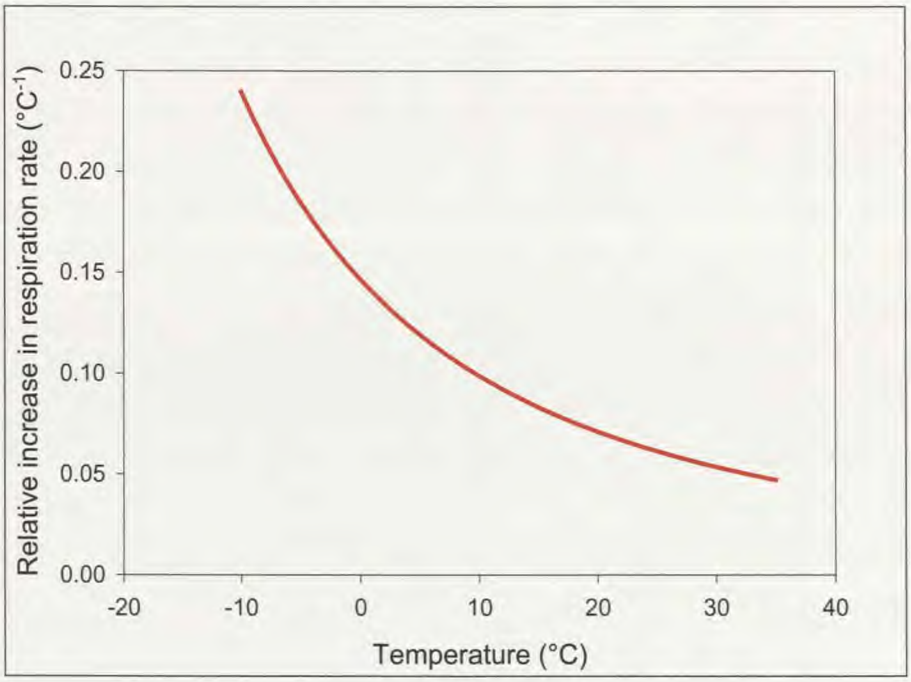

Fig. 22. Relative increase in soil respiration rate with temperature (based on Equation 12 from Lloyd and Taylor, 1994).

The calculation reveals that at 0°C a 1°C increase in temperature would cause a 15% increase in respiration rate, whereas at 25°C, the increase would only be about 5%. Thus, in the absence of moisture limitation, ecosystems associated with low soil temperature, such as alpine grassland ecosystems, have the highest relative sensitivity of soil respiration rates to changes in soil temperature.

The latest coupled climate-carbon cycle models, which consider feedback effects of elevated atmospheric CO_2 concentrations, reveal a significant positive feedback of climate change on terrestrial soil C stocks (Cox *et al.* 2002). Two coupled GCM experiments showed accelerated climate change as a result of suppression of the land carbon sink, resulting in a predicted rate of climate warming for the 21st century which is much more rapid than projected with earlier models. The positive feedback was associated with the conversion of the global net land carbon sink to a source by the middle of the 21st century. The results of these modelling experiments support the hypothesis that soil C stocks of permanent grasslands at higher altitudes are likely to decrease with climate warming.

6.2 Carbon dioxide fertilisation

The increase in atmospheric CO_2 concentrations affects terrestrial ecosystems indirectly, by the forced change in global climate, and directly, by stimulation of plant growth. The observed stimulation of photosynthesis led to speculations about the effect of increased plant carbon assimilation on SOC pools, and the possible feedback effect on atmospheric CO_2 concentrations (Schimel 1995). Some of the additional carbon accumulated in phytomass is likely to be transferred to the SOC pools. If this input is not offset by increased C export, SOC should accumulate and soils will contribute to global C sequestration (Amthor 1995).

Studies of ecosystem gas-exchange in CO_2 enrichment experiments found large net C gains unaccounted for by biomass, implying an undetected C sink in the soil (e.g. Drake *et al.* 1989). Recent evidence from temperate grasslands in the vicinity of natural CO_2 springs suggested soil C accumulation (Ross *et al.* 2000). However, on an intensively managed temperate *Lolium/Trifolium* pasture subjected to four years of CO_2 enrichment, van Kessel *et al.* (2000) found no evidence of a CO_2 fertilisation effect on soil C inputs. In extensively managed grassland under elevated CO_2, potential extra soil C was considered to occur only in fractions with rapid turnover (Niklaus *et al.* 2001; Volk and Niklaus 2002). Increased soil C sequestration due to reduced water stress was found after eight years of CO_2 enrichment in a Kansas tallgrass prairie (Williams *et al.* 2000). But due to the enormous amount of background soil C, it is difficult to detect significant changes in SOC content even in multi-year experiments (Rice *et al.* 1994; Canadell *et al.* 1996; Hungate *et al.* 1996).

The inconsistency of soil C responses to increasing CO_2 leaves the following questions open: (1) is extra C sequestered under elevated CO_2 conditions and (2) if so, where in the ecosystem is it stored? To date, no successful attempt to identify the nature of a putative large below-ground C sink has been reported from any long-term grassland CO_2 enrichment experiment.

6.3 Nitrogen deposition

Today, humans fix 98 Mt nitrogen (N) a^{-1} by the Haber-Bosch process, mainly to produce fertilisers, an additional 40 Mt N a^{-1} by cultivation of nitrogen-fixing legumes, and another 22 Mt N a^{-1} by combustion processes, mainly to generate energy (Jenkinson 2001). In pre-industrial times, biological processes accounted for 90–130 Mt N a^{-1} over the land surface of the earth, with an extra global contribution of <10 Mt N a^{-1} from lightning. The human-induced doubling of global N fluxes has resulted in large N losses (NO_3, NH_3, NO_x, N_2O), resulting in growing atmospheric N inputs to terrestrial ecosystems (Jenkinson 2001). This input is expected to increase by another 25% in developed countries over the next 25 years (Galloway *et al.* 1994).

In N-limited ecosystems, additional N supply leads to extra C uptake and storage because of the tight link between the N and C cycles (Vitousek and Horwarth 1991). Where N is limiting, its efficient retention within the system results in extremely low N losses to both aquatic and atmospheric environments. The "decoupling of terrestrial carbon and nitrogen cycles" (Asner *et al.* 1997), i.e. the fertilisation of N-limited ecosystems by anthropogenically enhanced levels of atmospheric N deposition, may stimulate terrestrial carbon uptake by vegetation and soils, and thus partly account for the "missing sink" (see Section 1.2). Deposition of atmospheric N might also influence the effectiveness of CO_2 fertilisation and thus the complexity of coupled CO_2 fertilisation/N deposition. Few long-term empirical studies are available concerning the combined effects of N deposition and CO_2 fertilisation on C sequestration. An analysis by Jenkinson *et al.* (1994) for fertilised and unfertilised grasslands revealed no measurable change in hay yield over the past 100 years for the respective treatments, despite substantial increases in N inputs via precipitation and a 21% increase in atmospheric CO_2 concentrations during the study period. Schimel (1995) suggested that future increases in N deposition will not result in proportional increases in C sequestration. In contrast to the findings of Jenkinson *et al.* (1994), Asner *et al.* (1997) propose that where N supply is limited, "increases in nitrogen should be matched by significant increases in carbon, but as nitrogen inputs continue to rise, the response of the carbon cycle must eventually approach zero and may even become negative in some regions". The combined effects of CO_2 and N fertilisation were studied by Loiseau and Soussana (1999) for a temperate grassland ecosystem. They found that higher rates of N fertilisation stimulated soil C accumulation at ambient and elevated CO_2 concentrations. Under elevated temperature conditions, not only C but also soil N mineralisation rates will increase. A combined climate and N mineralisation effect is expected to cause increases in production which, reduce or offset enhanced soil C respiration (Schimel 1995).

The effect of anthropogenic N deposition via the atmosphere, if any, will be more pronounced in agroecosystems that currently receive only low N fertilisation and are thus more N-limited in their productivity and in SOM turnover. These conditions are met by some of the extensively managed permanent grasslands in Switzerland, mainly located in the sub-alpine and alpine regions, but also by extensively used wetlands in the lowlands. For these systems, enhanced N deposition together with increasing temperatures may result in enhanced NPP, but higher residue inputs into the soil may largely be offset by temperature-induced increases in SOM turnover rates.

7. Uncertainties

The estimation of C stocks and the assessment of the C sequestration potential of different activities is based on soil surveys, experiments and land-use data, which are subject to statistical errors. Literature data used for some calculations were derived from studies conducted at various locations under different climatic conditions. Projections of future land use and activities are to some extent speculative. The major uncertainties and shortcomings regarding the data base, quantification of activities and climate change projections are discussed below.

7.1 Data bases for land use, soil types and carbon stocks

Land use

- The distribution of arable land and temporary and permanent grasslands among soil classes was calculated by combining different georeferenced data sets. This approach revealed a plausible distribution of land-use types among soil classes. However, the share of each land-use type for each soil class is based on proportionality only, assuming that its fraction within a community is proportional to the number of soil classes. **Georeferenced data with a high spatial resolution, such as the Swiss land-use statistics, which distinguish between arable and grassland use – the main factor governing soil C stocks – would significantly enhance the reliability of estimates of C stocks.**
- Historical land-use changes are based on data from the Swiss Federal Statistical Office (BFS), from the Swiss Farmers' Association (SBV) and from the Swiss land-use statistics, some of them dating back more than 100 years. While high data quality and data consistency can be expected from statistical offices, the accuracy of historical data cannot be assessed. In particular, large mountain areas cannot be directly assigned to land-use types. For mountain farming, it is even difficult to identify the precise management from aerial photographs.

Soil type

- The area of cultivated peatlands and their spatial distribution are among the most uncertain data used in this study. The range of uncertainty may even be higher if the pre-industrial peatland area exceeds the assumptions made in Section 4. The calculated represents the best guess, but a higher degree of certainty could only be achieved by identification of organic soils in the field, including also organic soils under permanent grasslands, which are not covered by the inventory of Presler and Gysi (1989).
- The spatial distribution of cultivated peatlands could only be calibrated sufficiently against an altitudinal gradient derived from historical surveys. Therefore, cultivated peatlands were assigned to a soil class according to altitude. The lack of correspondence between intact peatlands and histosols in the digital soil

map also indicates a gap in our knowledge of cultivated peatlands. However, the error caused by misassignment to soil classes is rather small compared with uncertainties regarding the total area.

- Carbon stocks in mineral soils were calculated according to the digital soil map (BEK) – the only georeferenced source of soil characteristics. The two main reservations are that (i) BEK classes are derived from an assignment of soil classes to landscape units (i.e. topography and parent materials) extrapolated to the whole area and (ii) each BEK class covers two to six soil types, with different physical and chemical properties. Thus, the information obtained from the BEK is much less sound than it would be for a soil map based on a soil survey.
- For mineral soils, the calculation of C stocks required data on soil clay and stone contents, and on soil profile depths. Clay contents were derived from relationships between physical soil attributes (field capacity), which themselves rely on extrapolations from limited data. Profile depths are given in the BEK for each soil class. In a recent study (Vökt and Pazeller 2002), the authors point out that the physiological depth may be overestimated for most of the soil profiles. **A marked improvement in the quality of the soil data base could be achieved by georeferencing the soil profiles used for BEK, which would permit geostatistical approaches.** A georeferenced approach has recently been adopted for the estimation of C stocks in French soils (Arrouays *et al.* 2001).

Carbon stocks

- The quantification of C in organic soils is subject to major uncertainties due to the unknown area of cultivated peatlands (see above). Additional problems arise from the estimation of the former peat layer thickness, the carbon content of the (degraded) peat and the decay kinetics. The mean thickness of the peat layer before cultivation (2 m) is supported by only a few measurements. The thickness of peat horizons ranges from a few decimetres up to several metres. The most promising source even today is the historical peatland survey by Früh and Schröter (1904). Compared with the peat layer thickness, estimates concerning the bulk density of the undisturbed peat and the OC content of peat are more certain, being supported by data from Switzerland and other European regions. The use of decay constants derived from a wide range of literature values gave reasonable estimates. Current C stocks measured in cultivated organic soils (Presler and Gysi 1989) agreed well with those calculated as described above, which confirms the relatively high certainty of C stocks per unit land.
- The variables in the regression analysis used to scale SOC concentrations from single observations to C stocks for whole areas explain only 27–44% of the variability for a particular land-use type. The confidence limits of the regressions were used to quantify the corresponding errors for C stocks in mineral soils, ranging from 22 to 25%. The degree of uncertainty could be reduced if more variables determining SOC concentrations were identified. Promising candidates include more detailed information on land-use types (tillage or intensity of grassland use), local climate, soil mineralogy and soil hydrology.
- Although the carbon stocks of subsoils did not differ significantly between textural or altitudinal classes, they did among land-use types, which indicates that land-use and/or selection pressure by farmers for fa-

vourable soils is also a major determinant of subsoil C stocks. Coefficients of variation for SOC concentrations within each land-use type were always higher for subsoils than for the corresponding topsoils. It will not be possible to identify the factors regulating C stocks in subsoils until georeferenced soil data are available to provide information on the major factors determining SOM formation.

7.2 Quantification of activities

In the absence of international guidelines on good practice in carbon accounting, it is difficult to assess current uncertainties in the quantification of activities. If it is decided to use mean C flux rates for a particular activity, perhaps modified by climatic or management constraints, only a few further assessments of uncertainties will be needed in this regard. If the mechanisms of carbon accounting stress verifiability at the farm or site level, uncertainties will need to be reduced by generating regional or farm-specific rates for a set of activities.

The quantification of C sequestration potentials for activities relating to Article 3.4 of the Kyoto Protocol is limited, since the origin of the relevant studies is highly variable. Clearly, a reasonable assessment of a given activity cannot be achieved simply by taking mean rates from the literature, as exemplified by the variable response of C stocks to no-till or to grassland management. Verification of literature data on sequestration potentials for ley-arable rotations demonstrated a significant overestimation for Swiss conditions, even when the currently high share of ley was considered. It is most probable that C stocks measured in Swiss arable soils already reflect the ley-arable rotation practised for more than a century, and that literature data derived from experiments where long-term arable rotations were modified by introducing leys show a systematically different response. The country-specific data for the conversion of arable land to permanent grassland shows the potential to reduce the uncertainty in calculations of sequestration rates and underlines the need for country-specific data also for other activities. Net C fluxes for organic soils are urgently needed in order to confirm the assessment of sink potentials under different management options. Listing the activities in the order in which they were discussed in Section 5.3, the main uncertainties, together with some research needs, can be summarised as follows:

- **No-till:** A number of studies from temperate regions are available, providing a range of sequestration potentials. Site-specific estimates at the farm level are needed in order to assess management effects and to address accounting requirements.
- **Ley-arable farming:** IPCC default values overestimate the Swiss potential due to the high share of leys already existing. A country-specific estimate requires field studies.
- **Farming intensity:** Consideration of individual aspects of integrated farming suggests that this type of management should have a sequestration potential, but assessments at the farm level failed to confirm this expectation. Low-input farming shows some signs of higher substrate-use efficiency, thereby promoting C sequestration, but hard evidence is still lacking. The magnitude of the effect of farming intensity also depends on actual differences between farming practices, which vary across countries, regions and individual

farms. If integrated and low-input farming have any effect on C stocks, then this is due to a higher share of leys and permanent grasslands, organic fertiliser inputs[45], higher residue returns and better soil protection.

- **Grassland management:** Major uncertainties are due to the limited data base concerning effects of management intensity on C stocks. Most studies were performed under different agricultural and environmental settings and cannot be extrapolated to Swiss conditions. Systematic research on this issue is urgently needed if e.g. grassland extensification is to be included in carbon accounting.
- **Conversion of arable land to grassland and set-aside land:** With the soil data base compiled in this study, a reliable estimate of the sequestration potential seems feasible. Annual sequestration rates are more uncertain, since the time required to reach a new equilibrium is provisional.
- **Peatland protection and restoration:** The sign of possible fluxes of the three main GHGs is based on a well-established data base, but data exhibit a wide range. Since absolute fluxes are one to two orders of magnitude higher than for mineral soils, and fluxes of CO_2 and CH_4 are in the opposite direction, site-specific measurements under representative conditions are necessary to improve the quantification of wetland activities.

7.3 Uncertainties in climate change projections

Uncertainties in the regional climate scenarios arise from various sources: uncertainties in the emissions scenarios, differences among the general circulation models used to infer the global climate scenarios, and uncertainties introduced by the downscaling procedure. It is not possible to discuss these aspects in detail, but comparison of the output of different models gives a measure of the uncertainty involved.

[45] See footnote in Section 5.3.1 "Production methods and farming intensity".

8. Agricultural carbon sequestration potential in a national and international context

8.1 Carbon sequestration and the Swiss greenhouse gas budget

The total value given for 1990 CO_2 emissions in the Swiss Greenhouse Gas Inventory was 14.52 Mt C, and the 8% reduction target corresponds to 1.16 Mt C a^{-1}. Today, no-till contributes 0.3% to the total reduction target (CO_2 equivalents for the base year 1990). Combining this with the conversion of temporary set-asides to permanent grassland would increase the proportion to 0.7% of the reduction target.

Substantial changes in agricultural structure would provide additional sink potentials. The restoration of all agricultural organic soils would contribute 8–23% of the reduction target, and the conversion of all arable land to no-till another 6–11%. Alternatively, converting all arable land into permanent grassland would contribute 11–12% of the reduction commitment, and CO_2 savings from the conversion of all organic soils to permanent grassland would amount to another 3–6% of the reduction target.

Altogether, the mean theoretical contribution of carbon sinks and carbon savings in agriculture amounts to 27% (18–35%) of the Swiss Kyoto commitment, corresponding to 2.1% (1.5–2.8%) of the total national CO_2 equivalents emitted in 1990.

This statement assumes a balanced CO_2 budget for all agricultural land with activities not considered under Article 3.4, or for which sequestration rates cannot be given (e.g. integrated production).

The current carbon sequestration by no-till of 0.01 Mt CO_2 is much less than the agricultural emissions of N_2O and CH_4 in 2000, i.e. 2.6 and 2.9 Mt CO_2 equivalents, respectively (BUWAL 2002). These agricultural GHG emissions account for 10.3% of the national total. The mean theoretical agricultural sequestration potential resulting from combining peatland restoration and conversion of all arable land to permanent grassland would amount to 1.13 Mt CO_2, corresponding to 21% of current N_2O and CH_4 emissions from agriculture. **Therefore, agriculture would remain a net GHG source even if all of the theoretical C sequestration potential were to be realised.**

The current agricultural GHG source is counterbalanced to a large extent by increases in forest and other woody biomass stocks of 4.1 Mt CO_2 (mean for 1990–1999), or 75% of the agricultural N_2O + CH_4 emissions in 2000. Altogether, C sequestration activities in the forestry and agricultural sectors covered under Articles 3.3 and 3.4 of the Kyoto Protocol could offset about 95% of the agricultural N_2O and CH_4 emissions.

The mean theoretical agricultural sequestration potential can be compared with other fluxes. The current CO_2 source resulting from conversion of agricultural land to urban areas, i.e. 0.31 Mt CO_2, is equivalent to 28% of the mean agricultural C sequestration potential of 1.13 Mt CO_2, but 25 times more than the current sequestration by no-till. Projected losses of SOC due to the expected rise in temperature by 2099 amount to 0.10–0.29 Mt CO_2, or 9–26% of the mean theoretical sequestration potential. Agricultural sources and sinks of GHGs in

2000, together with sources arising from current peatland utilisation and land consumption for urbanisation, are shown in Fig. 23.

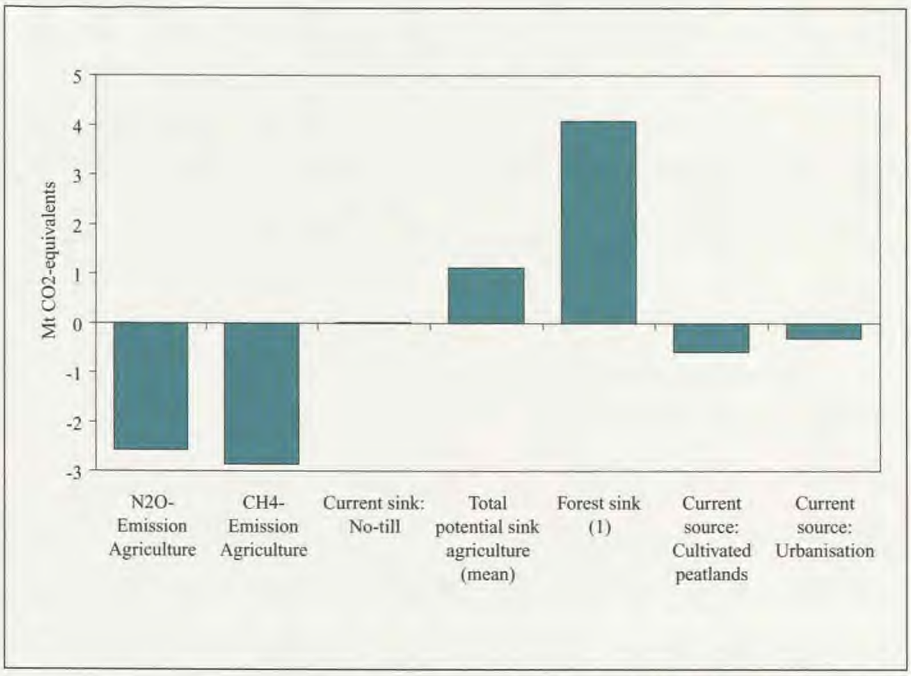

Figure 23. Agricultural GHG sources (-) and sinks (in Mt CO_2-equivalents), forest sink, and sources arising from peatland cultivation and urbanisation (consumption of agricultural land).

(1. Mean of the period 1990–1999; Swiss Greenhouse Gas Inventory[46]). Recent estimates by Fischlin *et al.* (2002) indicate a significantly smaller forest sink.

8.2 Carbon sequestration in an international context

In recent years, several efforts have been made at the national and international level to identify the mitigation potential of C sequestration. Although the Swiss contribution to overall global emission reductions is small, relative potentials can be placed in an international context.

EU15[47] and US sequestration potentials

In a study by Smith *et al.* (2000b), several agricultural scenarios regarding C sequestration were discussed for EU15, some of which are outside the scope of the present study. Scenarios similar to those discussed here concern the maximum enlargement of the no-till area and the introduction of leys, each contributing about 50% of the calculated total effect. Combining no-till and arable extensification (i.e. 30% ley in arable rotations) would yield an annual carbon mitigation potential of about 4% of the CO_2 equivalents emitted in 1990. Emissions from organic soils were estimated by Freibauer and Kaltschmitt (2001) for EU15. Total CO_2 equivalents emitted by organic soils were 73 ± 47 Mt a^{-1} or 52 ± 33 Mt a^{-1}, depending on the reference used. If all of the organic soil emissions, as in this study, were included as savings, the emission reduction achievable would be

[46] www. climatereporting.ch

[47] The 15 members of the European Union

1.6 ± 1.0% or 1.2 ± 0.7% of the total 1990 emissions by EU15. Including no-till and the introduction of arable-ley rotations, maximum sequestration potentials amount to around 5–6% of annual emissions.

For the US, Lal *et al.* (1998) calculated the C sequestration potential of cropland. Depending on the assumptions for sequestration rates under different scenarios, 4.7–13.0% of the annually emitted CO_2 equivalents could be stored in soils. Activities included mainly conservation tillage and residue management, improved cropping systems, irrigation, land restoration and land-use change. In a study of US grasslands, Follett *et al.* (2001) estimated the C sequestration potential to account for 1.1–5.7% of overall annual emissions. Main activities under the heading of grassland management were soil erosion control, land conservation and restoration, improved and intensified grazing land management and improved management of rangelands. Together, cropland and grassland activities could sequester 5.8–18.7% of gross annual emissions in the US.

Comparison of EU15 and US scenarios with Switzerland

The sequestration potentials presented by the various authors for EU15 and for the US exceed by a wide margin the mean theoretical sequestration potential calculated for Switzerland. By analysing differences in agricultural structures and land-use histories, some conclusions can be drawn regarding the discrepancies in sequestration potentials:

- The EU15 potential derives mainly from ley-arable rotations, and from no-till. The effects of ley-arable rotations are nearly exhausted in Switzerland, as the share of ley is close to the level set as the target in the EU15 scenario. Moreover, calculating Swiss potentials for larger ley areas using the data provided by Smith *et al.* (1997) resulted in highly misleading projections. The C sequestration rate per hectare for no-till is similar to that given in Smith *et al.* (1997). However, arable land accounts for 51% of the total agricultural area in EU15, compared with 26% in Switzerland (including temporary grasslands). Thus, arable activities have a larger impact on the total C budget at the European level. Carbon savings due to peatland restoration are in a similar range for EU15 (1.2–1.5% of gross emissions) and Switzerland (0.6–1.9 of gross emissions).
- Sequestration potentials for the US are distinctly higher than for the EU. As a consequence, agricultural options for mitigation of the greenhouse effect have a larger impact on the political and technological discussions in the US. However, the US potential needs to be discussed against the background of the country's specific agricultural conditions. Amounts of SOC in North American agricultural soils have been depleted as a result of intensified agriculture and inappropriate management in recent decades. For example, soil C depletion in the Corn Belt and parts of the Great Plains was about 47% compared with the SOC under native vegetation (Lal *et al.* 1998). This depletion is much larger than the calculated decrease due to forest clearance and the spread of agriculture in Switzerland, which have resulted in a decrease in C stocks (0–20 cm) of no more than about 24%. Significant contributions to the US potential derive from conservation tillage, residue management and land conservation, involving the conversion of highly erodible land from active crop to permanent grassland over a 10-year period. The adoption of sustainable management systems, with modified crop rotations and conservation tillage, as well as the set-aside of marginal land,

will lead to the recovery of pre-agricultural SOM levels in the long term. For example, the current share of leys in arable rotations amounts to only 14% in the US, compared with 28% in Switzerland. Another means of enhancing productivity in the US is to expand irrigation areas. Soil C stocks for this activity would even exceed SOC levels under native vegetation in the semi-arid regions. For grasslands, erosion reduction and increased use of nitrogen fertilisers to improve soil fertility are the main strategies for soil C recovery. Other relevant aspects of the US sequestration capacity are climatic and pedological conditions. Large areas of soils in the Mid-Western US are classified as Mollisols. These soils are characterised by high SOM contents, which developed under prolonged prairie vegetation. Prehistoric C stocks in these soils were around 70 t OC ha^{-1} (0–20 cm), compared with current levels of 40 OC ha^{-1} (Lal *et al.* 1998). If the SOC stock under natural vegetation is supposed to be the maximum level achievable, the resulting theoretical sequestration potential is 30 t OC ha^{-1} for the topsoil only. This figure exceeds the Swiss potential arising from the conversion from arable land to permanent grassland (22 t OC ha^{-1}) by 8 t OC ha^{-1}.

Concluding remarks

The mean theoretical C sequestration potential of Switzerland is below the potentials estimated for EU15 or the US, relative to gross GHG emissions. The difference in C sequestration potentials is due to differences in the agricultural structure (ley-arable farming, fraction of arable land), land-use histories (highly C-depleted soils in the US), and natural conditions (soil types).

Potential C savings due to the restoration of organic soils are in the same order of magnitude for EU15 and Switzerland, indicating a similar fraction of cultivated peatlands. From the comparison of C sequestration potentials across different countries, it can be concluded that the current agricultural structure in Switzerland makes it feasible to maintain agricultural C stocks and to avoid excessive C losses from mineral soils. In addition, during the last century, agriculture promoted high SOC stocks in mineral soils by means of site-adapted cultivation and land use. In an international context, agricultural structure and natural conditions should be considered when agricultural sources and sinks are included in carbon accounting.

9. Conclusions

Quantification of carbon stocks

The quantification of agricultural soil C stocks takes into account several factors leading to human-induced and natural variability in C stocks, i.e. land-use types, altitudinal gradients and soil characteristics. The data base of 544 soil profiles makes it possible to estimate agricultural soil C stocks, but it still needs to be expanded if it is to serve as a tool for a national carbon monitoring system. The following points, in particular, need to be addressed: soil data for organic soils, an improved Swiss soil map, more detailed differentiation among land-use types (permitting international comparability), and a consistent area specification by different statistical sources.

Historical, current and future soil C losses

Historically, Swiss C stocks in agricultural soils have undergone significant changes, mainly promoted by land use and land-use changes, and they are expected to vary in the future due to climate change. On the basis of contemporary figures for forest SOC stocks and calculations of overall C losses, the prehistoric SOC stock has been estimated at 330–343 Mt OC. Today's forest and agricultural stocks add up to 280 Mt SOC, which yields a loss by deforestation of 15.2–18.4% of the former stock. Historically "young" C losses from peatland cultivation and urbanisation over the past 120 years account for 7.4–10.8% of the prehistoric or 13.9–21.4% of today's agricultural soil C stock. It can be concluded that the total soil C stock will further decrease if agricultural use of organic soils continues and urbanisation proceeds at the current rate. Projected regional warming is expected to decrease C stocks of permanent grasslands at more than 1000 m above sea level by another 2.8–7.8 Mt OC, corresponding to 0.9–2.3% of prehistoric and 1.6–4.5% of today's agricultural soil C stocks.

Carbon sequestration potentials

Carbon sequestration potentials resulting from human-induced activities have to be discussed against the background of the large historical C loss caused by human activities and in the light of natural conditions which constrain the potential of each specific management scenario. Swiss agriculture has a considerable potential for mitigation of the greenhouse effect. Although today's sink accounts for less then 0.3% of the Swiss Kyoto commitment, a maximum theoretical sink potential of up to about 35% can be estimated. Within this range, several combinations of activities are conceivable, depending on social and political decisions. One example of a readily achievable, but small, carbon sink is the conversion of temporary set-asides to permanent grasslands. What proportion of GHG emissions from direct activities and from indirect effects can be compensated for by C sequestration? To take an example, a scenario with no-till applied to 88% of the arable land would allow annual C losses from urbanisation to be counterbalanced. The mean theoretical sequestration potential would approximately compensate for the sum of CO_2 from urbanisation, peat oxidation and regional warming.

With respect to the gross agricultural GHG balance, the mean theoretical C sequestration potential could compensate for up to 21% of N_2O and CH_4 emissions combined. Swiss agriculture would remain a net source of GHG even if the maximum theoretical sequestration potential were to be tapped. Implementation of only some elements of all the possible activities, which is much more realistic, may counteract the expected SOC depletion due to global warming. Therefore, with respect to historical C losses, most sequestration scenarios will reclaim only a fraction of the C losses induced by land-use changes. However, restoration of cultivated peatlands and reduction of the loss of agricultural land due to urbanisation would favourably affect the Swiss GHG balance. Among the activities discussed in the report, peatland restoration and protection would be the most potent measure for mitigation of the greenhouse effect, but its effects on the net GHG balance, which also includes fluxes of CH_4 and N_2O, still need to be evaluated under Swiss conditions. Another current gap in our knowledge concerns the preconditions for and practicability of peatland restoration.

As a proportion of total GHG emissions, the C sequestration potential of Swiss agricultural soils falls below the potential for EU15 and the US. The differences are attributable to characteristics of the agricultural structure (higher share of permanent grasslands and of leys in Switzerland than in EU15 and the US), land-use history (historically high SOC depletion in the US vs moderate losses in Switzerland) and natural conditions (high C sequestration potential in US Mollisols). The current structure of Swiss agriculture, as determined by natural conditions as well as policy decisions, already promotes C sequestration and prevents additional C emissions, except for the intensive use of organic soils. However, with the limited information currently available, it is not possible to quantify any carbon flux into the soil that may arise e.g. from integrated farming.

Full carbon balances of agroecosystems, including carbon costs and offsets by other GHGs, were outside the scope of this study. However, it is important to recognise that some potential additional mitigation options exist in terms of the full carbon balance. The studies cited for sequestration potentials in the EU15 and the US both stress the potential of the cultivation of biofuels to offset fossil fuel use. Unlike other options with finite potentials, the potential of biofuel use is infinite. In the study by Smith *et al.* (2000b), bioenergy crops show by far the greatest potential for carbon mitigation in all scenarios (offset amounting to 6% of 1990 gross emissions). Thus, the role and the potential of biofuels for the Swiss greenhouse gas budget should also be evaluated from the viewpoint of Switzerland's Kyoto commitments.

Consequences of a mitigation strategy involving carbon sequestration

The implementation of carbon sequestration activities in Swiss agriculture as discussed in this report would have severe consequences for the agricultural structure and the characteristic appearance of the landscape. Adoption of mitigation strategies through conversion of arable land to grassland and restoration of cultivated peatlands would dramatically affect the agricultural production mix, with enhanced production of meat and dairy products and a reduction in the overall production of annual crops. While the application of no-till to a larger area would have almost no effect on the characteristic scenery of the agricultural landscape, the conversion to permanent grassland would clearly alter the agricultural scenery, particularly in the Central Plateau. Changes in the agricultural structure and the scenery are two examples of conflicting aims that would arise if sequestration activities were to be implemented on a large scale. On the other hand, restoration of cultivated peatlands would be beneficial in terms of enhanced biodiversity. However, in many cases, a well-balanced combination of sequestration activities is also valuable from the agricultural viewpoint when implemented together with other measures to maintain or enhance soil fertility. A precise evaluation of management practices that also promote C stocks needs to be carried out at the regional or farm level, where site- and management-specific constraints can be considered in accordance with the aim of sustaining soil fertility and productivity.

10. Outlook and research needs

Carbon stocks and sequestration potentials in agricultural soils in Switzerland have been calculated as a best estimate with the data and projections available, despite pronounced uncertainties. Since the accuracy, and in some cases even the trajectory, of such estimates largely depend on data quality, the question of research needs is addressed in the following concluding remarks.

- The quantification of current stocks and sequestration potentials **of organic soils** shows pronounced uncertainties in both area and flux estimates. There is a strong need to enhance the quality of the area data by soil mappings covering the whole area of Switzerland. Replacing the current digital soil map with an area-based soil map would significantly improve the quality of C stock estimates for mineral soils. Measurements of CO_2, N_2O and CH_4 fluxes of organic soils under Swiss conditions at different stages of degradation are needed to reduce uncertainties for net GHG fluxes.
- Different intensities of **grassland** use are one potential activity discussed in this study. However, C stocks and sequestration potentials could not be evaluated as there are almost no data available concerning the response of C stocks to changes in grassland management. There is a need for process-oriented studies clarifying the influence of management practices on GHG fluxes, or considering climatic gradients. This applies not only to carbon, but in particular also to nitrogen dynamics. Some activities are in progress under the Fifth Framework Programme of the European Union.
- The response of grassland systems to climate change has been addressed using a straightforward empirical regression approach. More sophisticated **modelling** approaches, which again rely on our experimental knowledge of feedback between climatic, biological, and pedological processes, will help to reduce uncertainties and to improve our understanding of GHG fluxes and their controls. Ecosystem models have already been used, and the results are encouraging (Riedo *et al.* 2001). However, further improvement and testing of models is necessary.

Several studies and research papers on the topic of C sequestration have come to the conclusion that a real potential exists, varying among agricultural, social, and climate regions. The potential role of C sequestration in mitigating the greenhouse effect has been studied using the integrated assessment model MiniCAM 98.3, compiled by Rosenberg and Izaurralde (2001). With this model, the anticipated increase in C emissions to the atmosphere from 2000 to the end of the 21st century was calculated for a "business-as-usual" scenario and compared with a stabilisation trajectory, in which C emissions were allowed to increase to a maximum level by 2035, followed by a reduction by the end of the century. The results of this and similar studies clearly show that if emissions are to be lowered to the desired level, major changes in the current energy system will be required – increased efficiency in the use of fossil fuels, development of non-carbon-emitting fuels, improvements in power generation, a greater role for renewable energy and other technological advances. Soil C sequestration can play a strategic role and help to "buy time", but it cannot in itself solve the problem of climate change. Terrestrial C sinks are finite, particularly in a country such as Switzerland where the C sequestration potential is comparatively low, relative to gross GHG emissions.

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Glossary of terms and abbreviations

a	Year
a.s.l.	Above sea level
BEK	Digital soil map *(Bodeneignungskarte)*
Clay	Used here for the particle size fraction < 2µm., which consists of several soil minerals (e.g. clay minerals, other silicates, Fe and Al (hydr)oxides)
C sequestration	Site- and management-specific net increase in the SOC stock over a management period
CH_4	Methane
CO_2	Carbon dioxide
C stock	Soil organic carbon stock
COP	Conference of the Parties
Extensification	Agricultural management practice with low inputs. For grasslands: low stocking densities or cutting frequencies.
GHG	Greenhouse gas
GWP	Global warming potential. The GWPs used in this report are 1 for CO_2, 21 for CH_4, and 320 for N_2O for a 100-year time horizon.
ha	Hectare
Integrated farming	Farming system according to REP
IPCC	Intergovernmental Panel on Climate Change
Ley-arable farming	Arable rotation with annual or perennial leys (grassland)
MAT	Mean annual temperature
Mineral soil	Soil with SOC content not exceeding 18% in any of the upper soil horizons
NPP	Net primary productivity. Carbon assimilated by plants through photosynthesis minus carbon consumed by autotrophic respiration (per year)
N_2O	Nitrous oxide
OC	Organic carbon
OM	Organic matter
Organic farming	Low-input farming system, avoiding the use of mineral fertilisers and chemical herbicides and pesticides
Organic soil	Soil having either an organic horizon (≥ 18% OC) with a thickness of 10 cm or more from the soil surface to a lithic or paralithic contact, or an organic horizon with a thickness of 40 cm or more starting within 30 cm of the soil surface

REP	Required standard of ecological performance. Farming system that includes standards for crop rotations, soil protection, plant protection measures and ecological compensation areas
Set-aside	Former agricultural land that is abandoned and on which natural vegetation becomes established
SOC	Soil organic carbon (the carbon fraction of SOM)
SOM	Soil organic matter. Chemical composite of several elements, mainly C, O, H and N
Temporary set-aside	Used as synonym for ecological compensation area *(ökologische Ausgleichsfläche)*
UNFCCC	United Nations Framework Convention on Climate Change

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