CARBON STORAGE OF FINNISH AGRICULTURAL MINERAL SOILS AND ITS LONG-TERM CHANGE

DOCTORAL THESIS

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ACADEMIC DISSERTATION

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CONTRIBUTIONS

I The author participated in the writing and the data analyses of this study

II The author made a significant contribution to the data preparation, interpretation of the data and writing of the manuscript. Statistical analyses were done by a statistician.

III The author made significant contribution to all parts of the study except for the preparation of data for which the author's contribution was minor.

IV The author significantly contributed to the preparation of data, and made a minor contribution to data analyses and writing.

All studies (**I-IV**) are based either on the decades long field experiments/monitoring or modelling, and therefore author did not participate in the planning or sampling of the field experiments.

ABSTRACT

Carbon (C) that is stored in soils is the principal terrestrial C pool. The soil stores twice as much C compared to that which is stored in the atmosphere. Therefore, even a slight change in soil C stock can have a huge effect on atmospheric carbon dioxide concentration and global climate. Agricultural soils play a key role in this system: they cover about 38% of the land area world-wide and are intensively managed. The soil C also greatly contributes to sustainable food production as soil organic matter is to large extent made of C.

The aim of the research carried out for this thesis was to determine the nationwide soil C stock in Finnish arable mineral soils, study historic trends of C stock in soils and examine possible factors that affect those trends. Knowing the past can also give us insights into future trends in soil C and its climatic impact. The data presented in this thesis were obtained from the Finnish national soil monitoring network and long-term field trials in Uppsala Sweden and Pushchino Russia. In addition two process-based soil C models, namely: Yasso07 and RothC, were used and their findings were compared with those of long-term field trials.

Finnish arable lands were found to be rich in soil C. Mineral soils in Finland store between 41 and 67 Mg C ha^{-1} (0-15 cm) depending on the management, soil type and region. Nationwide the C stock in arable topsoil is about 117 Tg and although the deeper soils layers are poorly known the total soil C stock in mineral soils of Finland can be estimated to be about 300 Tg.

The C stock of mineral arable soils has decreased. The decrease was found to be 0.22 Mg C ha^{-1} yr^{-1} (0-15 cm soil layer) according to the national soil inventory network of Finland and 0.29-0.36 Mg C ha^{-1} yr^{-1} (0-100 cm soil layer) according to results obtained by the Yasso07 model. The annual C emissions from agricultural mineral soils are about 0.5 Tg, which represents about 2.5% of the total greenhouse gas emissions in Finland. Process based modelling in which the past land use history was taken into account clearly indicated that the nationwide decrease can be linked to the past change in land use from forest to agricultural land, which has created the ongoing soil C loss.

Finnish arable lands are relatively young and it is likely that they are still losing the soil C that had been accumulated when these lands were part of boreal forest systems. Likewise the thesis indicates that the soil C loss partly results from the intensification in cultivations that took place in the past decades and that cultivation of annual crops has become more common. Cultivation of annual crops increases the soil disturbance due to tillage and decreases the below-ground C influx into the soil. The composition of litter quality was also shown to have considerable effects on the ease of decomposition of organic matter.

Previously, the Yasso07 model had been mainly tested and used in forested soils. Findings in this thesis demonstrate that this model can also

be applied to agricultural mineral soils under boreal conditions. The fact that the model works equally well under various environmental conditions indicates that the accumulation of soil C is largely controlled by litter input, climate and litter chemical quality. The comparison between simulations and experimental data obtained from field trials showed that the results of Yasso07 model are comparable with those of the RothC model, which is currently one of the most widely used soil C models for agricultural applications.

The decreasing trend of soil C stocks found in this thesis has undesirable ramifications for climate, environment and sustainable food production. The changes in topsoil C might reflect the condition of soil C in deeper soil layers. Therefore, it would be crucial to investigate the storage of deep soil C and the long-term effects of agricultural practices on it.

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1 INTRODUCTION

Soil along and especially its organic matter is critical for agriculture and food production as it provides nutrients and water for plants. As a consequence, it is the soils that ultimately provide food and fibers for us to eat, and clothes to wear. As agricultural lands cover as much as 38% of the world's land area (FAOSTAT 2014) and the soils store high amounts of carbon (C), the agricultural soils also have climatic influences. The impact of humans on agricultural soils is high compared humanity's influence on forests and natural grasslands. Therefore, it is most likely that agricultural soils have the greatest potential to capture and store C through improved management and as a consequence play a key role in the battle against ongoing climate change.

According to most soil monitoring studies the C stock of agricultural soils in Europe is decreasing, specifically: England and Wales (Bellamy et al. 2005), Belgium (Sleutel et al. 2006), Finland (Mäkelä-Kurtto & Sippola 2002), Norway (Riley & Bakkegard 2006), Austria (Dersch & Böhm, 1997), France (Saby et al. 2008). Model-based future predictions suggest that arable soils in central-and southern Europe may act as a C sink in the future, whereas C of the northern arable soils continues to decrease (Smith et al. 2005). The decreasing trend in soil C is a concern with respect to the climate and climate change. Decreasing soil C stocks might eventually also lower agricultural yields (Bauer & Black 1994).

Several reasons for the reported soil C loss have been proposed. Soil inventory data in England and Wales showed that soil C is decreasing in all land use types and based on that finding Bellamy et al. (2005) drew the conclusion that C loss is caused by climate warming. Historical changes in land use and management also explain the observed trends in soil C (Van Wesemael et al. 2010). There has been change in Europe towards increasing cultivation of annual crops and intensification of agriculture. For instance in Belgium the soil C loss was linked to reduced manure application and increased erosion (Lettens et al. 2005). Intensification of agriculture might have also involved increased plough depths and compaction of the soil, which is reflected in topsoil C measurements. If soil C stock changes are mainly human induced, then it follows that improved cultivation practices might turn agricultural soils into a net C sink instead.

The Kyoto Protocol sets internationally binding emission reduction targets for member countries of the United Nations Framework of Convention for Climate Change (UNFCCC). Under the Kyoto Protocol, carbon dioxide (CO_2) emissions from soils are among the parameters to be reported. Soil C reporting falls under the category of Land Use, Land-Use Change and Forestry (LULUCF). It is recommended to use direct measurements or modelling (Tier 3) in greenhouse gas reporting, or if this is not available, then country specific emission factors (Tier 2) over the default IPCC emissions factors (Tier 1) (Eggleston 2006). Estimated emissions can vary considerably depending on the method used as was shown by Borgen et al. (2012). Changes in soil C stock are difficult to detect using direct measurements without using prohibitively large sample-sizes (Conen et al. 2003, Smith 2004). Therefore, various models are widely used for reporting annual soil C emissions/removals in UNFCCC greenhouse gas inventories.

1.1 SOILS IN THE GLOBAL C CYCLE

The C pools can be roughly divided into four reservoirs: atmosphere (730 Pg C), ocean (38000 Pg C), land biosphere (2000 Pg C) and geological reservoirs (Solomon et al. 2007). Geological reservoirs, including carbonate rocks and fossil fuel deposits, are by far the largest reservoirs of C (Grace 2004) (Figure 1). The geological reservoirs are relatively stable and make only minor contributions to the global C cycle in the natural-state system (Solomon et al. 2007). The main annual fluxes occur between the atmosphere and biosphere (120 Pg C yr^{-1}) and between the atmosphere and the ocean (90Pg C yr^{-1}) and are approximately in balance each year.

In a natural-state system there can be considerable C fluxes between reservoirs, but the size of the reservoirs remains rather stable. Humans on the other hand have massively altered the global C cycle by releasing C from geological reservoirs through increased use of fossil fuels and cement production (Solomon et al. 2007). Deforestation and land use change from grassland to cropland has also contributed to the change in the global C cycle. Emissions from both fossil fuel burning and deforestation are estimated to be 6.3 and 1.6 Pg yr^{-1}, respectively (Grace 2004) (Figure 1). Emitted C has partly ended up in the atmosphere and as a consequence the CO_2 concentration of the atmosphere has increased from the pre-industrial level of 280 ppm to close to 400 ppm nowadays (NOAA 2014). Emitted C has also been partly taken up by terrestrial and marine ecosystems.

Estimated quantities of C stocks and fluxes between them, however, greatly vary depending on the study (Watson et al. 2000, Amundson 2001, Grace 2004, Solomon et al. 2007, Beer et al. 2010, Stockmann et al. 2013). Therefore, it is no surprise that estimated soil C stocks have varied from as low as 400 Pg to as high as 9120 Pg in the course of the history (Amundson 2001). PAGE-study (2005) suggests the soil organic C stock in the 1m soil layer to be 1555 Pg. In addition the inorganic soil C pool that consists of carbonate minerals is 695-748 Pg (Batjes 1996). Agricultural soils are a considerable pool of organic C (369 Pg) and account for about 24% of total C stored in soils (PAGE 2005).

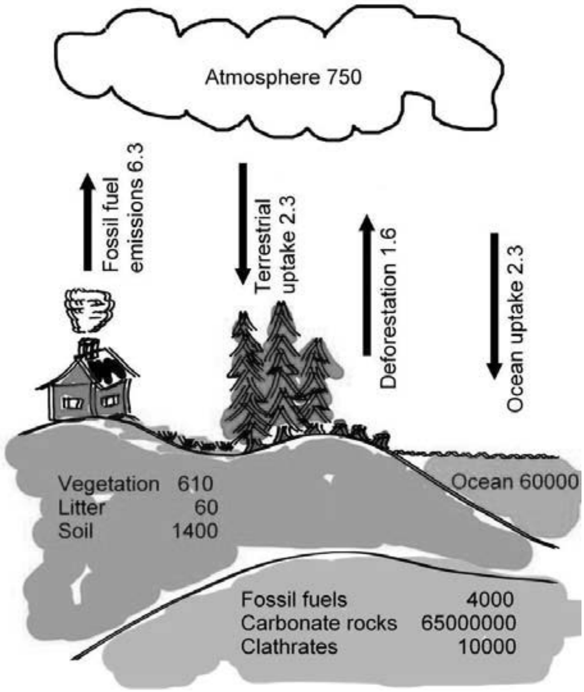

Figure 1. Global C stock (Pg) and net fluxes (arrows) between C pools (Pg yr^{-1}) according to Grace (2004)

1.2 THE C DYNAMICS IN AGRICULTURAL SOILS

The soil C stock in agricultural land reflects the balance between the inputs from plant residues and animal waste and losses due to decomposition and erosion as in any other terrestrial ecosystem (Figure 2). The amount of plant residue derived C that enters the soil depends on the growth of the plants (net primary production) and the portion removed from the field as part of the harvested crop. Soil C return from plants consist of inputs from above-ground and below-ground (Bolinder et al. 2007). The below-ground C input consisting of dead root biomass and root exudates can be considerable especially in grassland ecosystems (Canadell et al. 1996, Kuzyakov and Domanski 2000). Part of the plant biomass is used for feeding livestock and are returned to the arable ecosystem through manure spreading or from the faeces of grazing animals.

Decomposition of soil organic matter (SOM) is driven by microbial communities. The rate of decomposition depends on the chemical quality of organic matter especially its C/N-ratio, environmental conditions such as climate and soil properties, and also on the composition and abundance of microbe communities (Couteaux et al. 1995, Six et al. 2002, Six et al.

2006). It has traditionally been thought that during the decomposition process the organic matter gradually turns into a chemically more recalcitrant form, with the end product in the late stage being extremely resistant to decomposition. The most widely used soil C models are also based on this idea. However, the idea has been recently challenged by Dungait et al. (2012). According to those authors recalcitrance of litter might play only a minor role in soil C accumulation and that the decomposition process is instead governed by the accessibility of SOM for microbes and also the make-up of the populations of the decomposers. Even resistant compounds become decomposable when environmental conditions are suitable and the right decomposers are present.

The idea that biochemical properties of SOM control the decomposition only to some extent marries well with the stabilization mechanism of SOM on soil aggregates and mineral particles. According to the review by Six et al. (2002) SOM binds to mineral soil particles to form either soil aggregates or silt and clay protected organomineral complexes. Both of these processes have been found to reduce microbial activity within the aggregate due to physical protection and therefore to have a positive influence on the accumulation of SOM (Hassink 1997, Paustian et al. 2000). Agricultural cultivation practices on soils such as tillage can break the bindings between SOM and minerals. Recently Segoli et al. (2013) introduced a model that simulates aggregate and aggregate-associated C dynamics, although that model was tested only under simplified laboratory conditions.

Decomposition of SOM releases the C absorbed by plants through photosynthesis back into the atmosphere as CO_2. In waterlogged conditions such as that prevailing in peatland cultivation or ricepaddies the soil C emission can also occur as methane (CH_4) (Le Mer and Roger 2001, Levy et al. 2012). Although it is known that cultivation has an inhibiting impact on CH_4 oxidising bacteria, cultivated boreal mineral soils generally act as a methane sink rather than an emission source (Regina et al. 2007).

Erosion removes the soil C from the agricultural topsoils along with mineral particles and organic matter (Van Oost et al. 2007). Eroded material is thereafter deposited in aquatic or terrestrial ecosystems. Erosion can occur mechanically by wind or water erosion or it can occur chemically. Annual soil C loss due to erosion is typically relatively small according to the study by Van Oost et al. 2007 who reported that global soil C loss varied between 0.3 and 3.2 kg C ha^{-1} yr^{-1}.

Agricultural soils are radically different in respect to the intensity to which they are managed compared to those of natural or near natural state environments (e.g. forests and grasslands). In principle, the aim of the agricultural production system is to remove as much biomass as possible from the field and use it for feeding humans and animals. This possesses persistent challenge whether the system is sustainable in the long run in terms of maintaining SOM. The soil C can be kept in balance by using diverse crop rotation, including green-manure in crop rotation, leaving crop residues in the field and by spreading farmyard manure, which was the case in older times with traditional farming and is the case in contemporary organic farming systems. Mather and Hart (1956) summarized the importance of manure in food production by stating “where there’s muck, there’s luck”. Current agricultural production systems, however, rely to large extent on external inputs such as chemical fertilizers. The use of fertilizers can also have a positive effect on soil C stock in nutrient deficient soils as they boost the growth of agricultural crops and pastures, which increases the C return to the soil.

Agricultural lands in Finland were typically established by clearing the most productive forest types or converting grassland and meadow habitats into arable croplands. Field cultivation is relatively young in Finland: the area of arable land reached its current level as late as the early 20th century (Peltonen 2004). Prior to that time it was common to utilize natural-state grasslands and meadows to feed livestock by fodder and by grazing. Conversion of forest into cropland is reported to result in losses of soil C that amount to about 42% overall (Guo and Gifford 2002). It is commonly thought that soils re-equilibrate with respect of soil C stock within a few decades after conversion (Poeplau et al. 2011, West et al 2004), although it is known that soils also consist of C pools with highly resistant material (Jenkinson and Rayner 1977, Trumbore 1997).

Although the carbon dynamics of organic soils are out of the scope of this thesis, agricultural production in Finland is characterized by commonness of organic soils. Organic agricultural soils are usually established by draining peatlands, in which a thick layer of peat have accumulated under anaerobic conditions over thousands of years. It is estimated that the area under cultivation of organic soils is about 13.6% of the total arable area in Finland (Myllys and Sinkkonen 2004). Cultivated organic soils are more important than their proportional area suggest in respect to their soil CO_2 emissions: most of the agricultural related CO_2 emissions arise from the cultivated organic soils (Statistics Finland 2011).

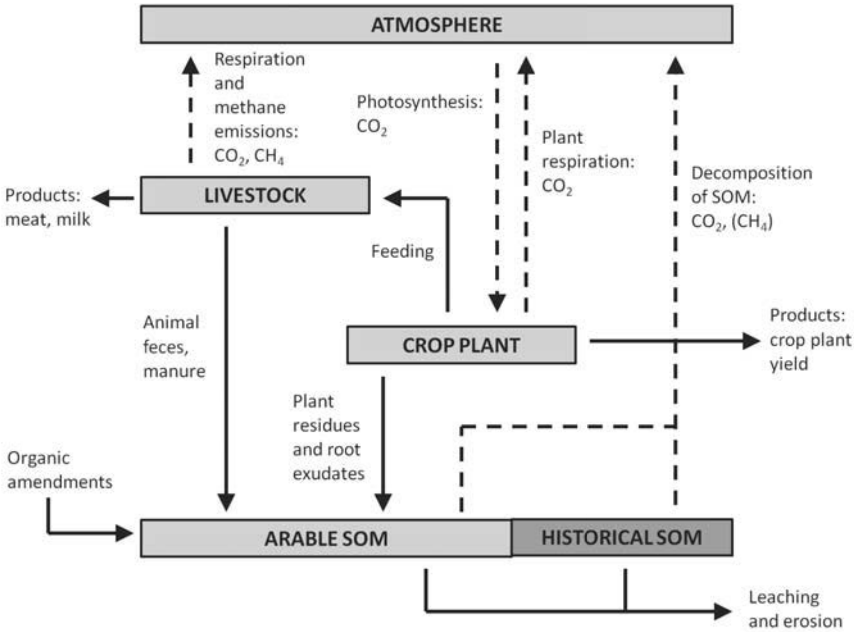

Figure 2. Main C flows between crop plants, livestock, atmosphere and soil organic matter (SOM) in agricultural land. Historical SOM indicates the soil organic matter that had accumulated before land (e.g. forest and grassland) were subsequently converted into agricultural use. Gaseous C flows are shown by broken lines.

1.3 DETECTING THE CHANGES IN SOIL C STOCK

The changes in soil C can be investigated by using five different methodological approaches: a national soil inventory, long-term field trials, soil C modelling, gas measurements and laboratory experiments. This thesis focused on the first three as there are no long-term (up to decades) gas measurements and laboratory experiments available in Finland.

Most European countries have established networks to monitor the changes in soil conditions over time (Morvan et al. 2008, Saby et al. 2008). The geographical coverage of the networks is, however, uneven between countries. Generally the number of sampling plots is higher in northern and central Europe than in the southern Europe. The networks also differ considerably in their sampling protocols, design, location of plots (systematic grids, random irregular networks or locations selected using expert judgment) or the sampling interval. Some countries have separate networks for different land-use types such as forest and arable land whereas others have uniform networks. Recently the Gemas- (Reimann et al. 2014) and LUCAS-projects (Tóth et al. 2013) used a European wide harmonized monitoring network and sampling design for arable soils, which paralleled that for forested soils in the Biosoil-project (Lacarce et al. 2009). Changes in the soil C stocks are small and therefore uncertainties in the measurements due to spatial heterogeneity, soil sampling and analysis are high. Thus, there is an ongoing debate as to whether the accuracy of the monitoring networks is high enough to detect the changes in soil C for greenhouse gas inventory purposes (Saby et al. 2008, Schrumpf et al. 2011).

Long-term experiments are valuable in providing information on the processes that affect soil C and the quantification of the changes in soil C. Experiments have been mainly used to study the effect of various treatments, such as application of organic amendments (Uhlen 1991, Kirchmann et al. 1994), cultivated crop plants (Jenkinson and Rayner 1977, Erviö 1995), land-use and management change (Karhu et al. 2011), on soil C. Long-term field trials also provide an indispensable basis for the calibration and validation of soil C models (Körschens 2006). Laboratory experiments are in principle similar to field trials, but the conditions in a laboratory can be controlled more precisely. Various incubation experiments that are performed in a laboratory are commonplace and such experiments can be used to study the C at the rhizosphere-soil interface (Hütsch et al. 2002) or the response of soil C to elevated temperatures (Holland et al. 2000, Vanhala et al. 2007) or the response to a repeated drying-rewetting cycle (Fierer and Schimel 2002).

The conventional soil sampling followed by the determination of the C content in a laboratory setting, is time-consuming and costly (Mäkipää et al. 2008). The use of spectral analysis for the measurement of soil C has been suggested as an alternative approach to overcome the problem related to the large number of samples needed to cover adequately the spatial variability of soil C (McCarty et al. 2002). Spectroscopy could be used in the laboratory to speed up the analyses of the C content (McCarty et al. 2002, Sørensen and Dalsgaard 2005, Vohland et al. 2011) or extended to the field-scale by the use of portable spectrometers or airborne devices such as probes mounted on aircraft or orbiting satellites (Stevens et al. 2006, Gomez et al. 2008, Stevens et al. 2010). Until now the remote sensing methods have been compared with conventional soil C stock measurement data and in most cases the results were not accurate enough for monitoring purposes.

The C fluxes at the soil-vegetation-atmosphere interfaces can also be quantified by gas measurements. Commonly used methods are chamber- (Regina & Alakukku 2010), flux gradient- (Turcu et al. 2005) and eddy covariance- techniques (Baldocchi 2003). In the chamber method, the surface of the soil is covered by a closed chamber, and the gas flux is derived from the repeated measurements of the gas concentrations inside the chamber. Samples are often taken manually by syringe and analyzed using gas chromatography, but automated sampling systems are also available. In contrast, the gradient method uses probes that measure the soil CO_2 concentration, and these probes are installed at different vertical depths throughout the soil profile. The measured concentrations obtained are thereafter converted into a CO_2 flux by multiplying them by the soil-layer-specific CO_2 diffusion coefficients. This step can, however, be challenging because usually the diffusion coefficients are calculated by taking into account the porosity of soil, the soil water content and the soil texture (Pingintha et al. 2010). It should be noted that both the gradient and the chamber methods give the sum of the CO_2 produced by root respiration and by decomposition of SOM and plant residues. Root respiration alone may account for between 10 to 90% of soil respiration (Hanson et al. 2000). Furthermore, the photosynthesis of the above-ground vegetation contributes to the observed CO_2 flux depending on the sampling design. Therefore, unlike actual soil-sampling, these methods do not directly tell about the changes in soil C stock. A precise quantification of the soil C balance necessitates the use ecosystem level approaches, where all the various C flow routes between the interfaces are taken into account. This could be achieved by e.g. using eddy covariance in combination with the chamber or gradient methods and also with biomass measurements (see e.g. forest ecosystem study by Ilvesniemi et al. 2009). The eddy covariance method is based on micrometeorological theory and it measures the net CO_2 exchange across the canopy-atmosphere interface (Baldocchi et al. 1988).

The modelling approach, when combined with experimental direct measurement data has proven to be a powerful tool for estimating changes in soil C stock. It is cost-effective and models can be applied to wide geographical regions, if the data required by the model are made available. Models differ greatly with respect to their complexity (Smith et al. 1997, Peltoniemi et al. 2007). For example, they can vary from the simple empirical formula (=carbon response functions) that can be used to predict changes in soil C stock as a result of change in management or in land-use, to the complex ecosystem models such as Century (Kelly et al. 1997) and Daisy (Jensen et al. 1997).

2 OBJECTIVES OF THE STUDY

The soil C has only recently become an active topic of scientific interest due to the interaction between soil C and climate. It has not always been the case and e.g. in agricultural land the interest has focused on nutrient status and productivity of the soil. Therefore, there is still knowledge gaps even regarding the fairly basic questions related with soil C in agricultural land.

The purpose of the research for this thesis is to give answers to the following questions:

1. How much soil C is stored in agricultural mineral soils in Finland?
2. Has the soil C stock in agricultural mineral soils in Finland decreased or increased during the last few decades?
3. If there are any changes in soil C stock, what are the possible factors that cause this change?

The first objective can be achieved in a fairly straightforward manner by using data that were obtained from the national soil inventory network. However, there is no single correct way or dataset that would achieve the second and third objectives. While experimental data can give some insights into the factors that affect soil C nationwide, the modelling approach turned out to be the most appropriate and effective way to study it. In this thesis the Yasso07 model for soil C was used. Yasso07 has already been widely used in forest applications and for example it is used to estimate soil C stock changes in forested land by the Finnish national greenhouse gas inventory. However, there is little experience on the performance and applicability of the model for soils in agricultural land. Therefore, this thesis also aims to meet the following objectives:

4. test the validity of the Yasso07 model for evaluating soils of agricultural land and compare the results with those of other models
5. compare the limitations and advantages of different methods in monitoring changes in soil C stock

Soil C has an essential role in sustainable food production and its loss has several adverse environmental implications. Hopefully, this thesis will increase the awareness of the current state of our environment and encourage farmers to use environmentally friendly cultivation methods. The findings of the thesis might also lead to the improvement of methods in soil C monitoring, which is applicable e.g. in national greenhouse gas inventory.

3 MATERIALS AND METHODS

3.1 NATIONAL SOIL INVENTORY

Finland has a relatively dense soil monitoring network for its agricultural soils (Figure 3). The original aim of the network was to observe the state and temporal trends in soil quality; especially those of nutrient status, heavy metal contents and SOM (Sippola & Tares 1978). The network was established in 1974 on fields that were growing timothy grass and therefore the sampling design was not completely random (see **II**-study for discussion on its effect on results). Sites were also sampled in

1987, 1998 and 2009 (Erviö et al. 1990, Mäkelä-Kurtto & Sippola 2002). The number of sampling plots decreased from 2042 in 1974 to 611 in 2009. A subset of sites were randomly selected in later inventories, but these selections still ensured the regional representative of the data. The number of sampling plots for the mineral soils was 515 in 2009. The network covers the whole country except for the northernmost part, where the total agricultural land area is small and fragmented (Figure 3). Soil samples were collected from 10m*10m sampling plots from a ploughed layer. Soil C was determined by using the dry combustion method (Matejovic 1993) except in 1974 when wet oxidation was used (Graham 1948). The national inventory data were used to determine the overall soil C stock in agricultural land and the temporal trend in soil C stock (**II**). Data were classified by the region, soil type and cultivation history. Data were analysed using generalized linear mixed effect models. Inventory data were also compared to Yasso07 modelling results (**IV**).

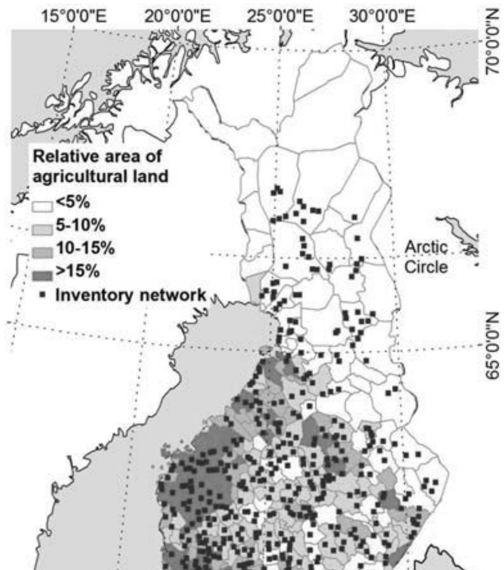

Figure 3. Relative area of agricultural land and national soil inventory network in Finland in 2009

3.2 LONG-TERM EXPERIMENTS

3.2.1 Ultuna

Ultuna long term experiment (59°85' N, 17°63' E) (Figure 4) commenced in 1956, and is one of the longest and most extensively studied field trials in Europe. The main aim of the experiment is to examine the effects of the application of various organic amendments on soil properties (Kirchmann et al. 1994). The experiment consists of 15 treatments each of which had four replicates. Eight of these treatments were selected for this present study: bare fallow, straw, straw+N-fertilization, crop only, crop+N fertilization, green manure, farmyard manure and peat. Organic amendments were added in 1956, 1960, 1963 and thereafter every second year. Each amendment was added at the rate of 8000 kg ash-free dry matter ha^{-1}. The size of plots were 2*2 m. Soil samples were taken from the topmost 20 cm in 1956, 1967, 1975 and thereafter every second year just before the application of the organic amendments. Samples were dried, passed through a 2 mm sieve and the C content was determined. Apart from the fallow plots, all treatments had the same crop rotation, which included cereals, oilseeds and root crops. All above-ground biomass was removed before adding organic amendments.

Data obtained from Ultuna were utilized in this thesis to examine the effect of quality and quantity of C input on soil C balance (**I**). The site was also used to validate the Yasso07-soil C model for agricultural land (**I**). The site has been widely studied and several empirical studies (Witter et al. 1993, Gerzabek et al. 1997, Otabbong et al. 1997, Kirchmann and Gerzabek 1999) and modelling studies (Falloon and Smith 2002a, Nilsson et al. 2005, Juston et al. 2010) have been published.

Figure 4. Location of Ultuna and Pushchino long-term experiments.

3.2.2 Pushchino

Puschino is located in Russia in the Moscow region (54°50' N, 37°35' E) (Figure 4). The land-use experiment was established to study the effect of cropland conversion to grassland on various properties of soil and vegetation (Ermolaev and Shirshova 2000, Larionova et al. 2003, Larionova et al. 2009). The site was withdrawn from arable use in 1980. The experiment was divided into four treatments according to the fertilizing and mowing regimes. Fertilized plots were treated annually with mineral NPK fertilizer at the rate of 60 kg N ha^{-1} yr^{-1}. Soil sampling was carried out in 1980, 1999 and 2004 by collecting samples from the soil layers of three depths: 0-20, 20-40 and 40-60 cm. The bulk densities of the soil layers were also determined. The C content was estimated by the dichromatic oxidation procedure (Orlov and Grishina 1981). Weights of the above- and below-ground biomasses were measured for several years over the time period of 1980-2007 (Ermolaev and Shirshova 2000, Kurganova et al. 2007, Kurganova et al. 2008, Kurganova et al. 2010). Data from Puschino was used to validate Yasso07 and RothC models (**III**).

3.3 SOIL C MODELS

3.3.1 Dynamic soil C models

Yasso07 (Tuomi et al. 2011) and RothC (Coleman and Jenkinson 1999) are dynamic soil C models. They can be used to estimate the changes in soil C stocks in mineral soils. Yasso07 has mainly been used for forest soils (Repo et al. 2011, Ortiz et al. 2013), but has also been tested for other applications such as modelling C stock changes due to land-use change (Karhu et al. 2011). RothC is probably the most widely used soil C model for agricultural land (Falloon and Smith 2002b, Smith et al. 2005, Romanovskaya 2006, Van Wesemael et al. 2010). Both models can be run using a monthly time step, although an annual time step is more commonly used with Yasso07. RothC predicts the C stocks in the topmost 20 cm soil layer whereas Yasso07 predicts the soil C stock of a soil layer at 1m of depth.

The RothC model is structurally similar to that of Yasso07. The soil C in both models is divided in five compartments according to the chemical properties (Figure 5, Table 1). The chemical quality of the compartments varies from easily decomposable material to highly resistant material with slow decomposition rates. One of the compartments in the RothC model is assumed to be inert, i.e. it does not decompose at all. Environmental conditions also modify the decomposition rates in both models. The decomposition rates in the Yasso07 model depend on the temperature and precipitation, whereas the decomposition rates in the RothC model are affected by temperature, soil moisture content and soil cover. Figure 6 shows the temperature dependencies of decomposition of both the Yasso07 and the RothC models.

Table 1: Parameter values of the Yasso07 model. Decomposition depends on the temperature and precipitation. Table 1 shows the decomposition rates when temperature is 0°C degrees and soil-water is sufficient (≈precipitation is high). The decomposition in Yasso07 results in mass flows between all A, W, E and N compartment combinations, but table 1 shows only the most important mass flows as shown in Figure 5. Other mass flows are less than 0.05.

Decomposition rates (a^{-1})	Relative mass flows	
5.8	A→W	0.99
0.73	W→A	0.48
0.29	N→A	0.83
0.031	E→N	0.92
0.0017	A,W,E,N→H	0.0045

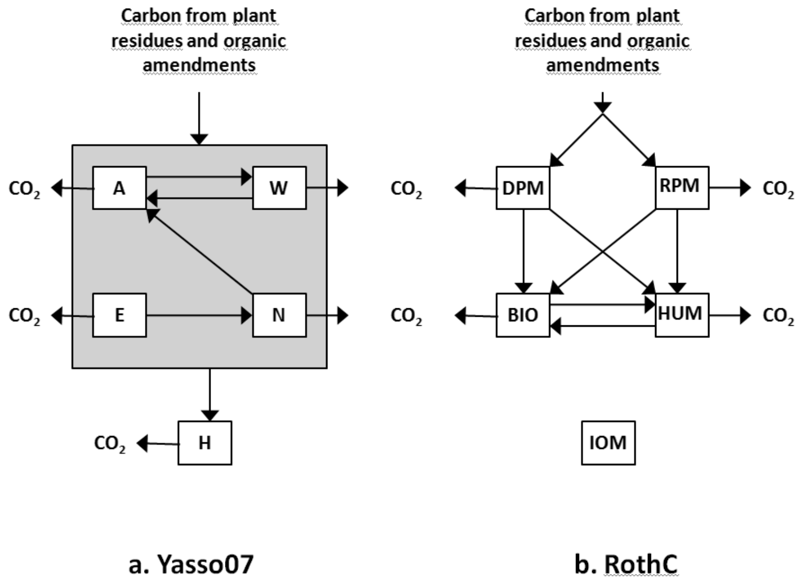

Figure 5: Flow chart of Yasso07- (a) and RothC soil C models (b). The grey shaded box in Yasso07- model indicates that the incoming C is divided into acid-(A), water-(W), ethanol- (E) and non-solubles (N) compartments and the decomposition of each of those compartments leads to the formation of humus (H). Only major flows of the Yasso07- model are shown. The soil C in the RothC model is divided in decomposable plant material (DPM) and resistant plant material (RPM), humified (HUM)- and inert organic matter (IOM) and microbial biomass (BIO).

The Yasso07 and RothC models only cover processes related to the decomposition of SOM, unlike ecosystem models such as Century (Kelly et al. 1997). Therefore, the amounts of organic matter entering the soil have to be estimated and provided for the models separately. Soil C input can be determined either by direct measurements or indirectly by deriving it from crop yield statistics using allometric functions. The latter method is commonly used in regional studies (**IV**), but can also be used to estimate below-ground C inputs, which are difficult to quantify by measurements alone. Therefore, in many cases allometric functions are used in combination with direct measurements as was done in studies **I, III** and **IV**. Carbon from animal waste at the regional level was estimated by obtaining the number of livestock and multiplying that by the annual faeces extraction per livestock (**IV**).

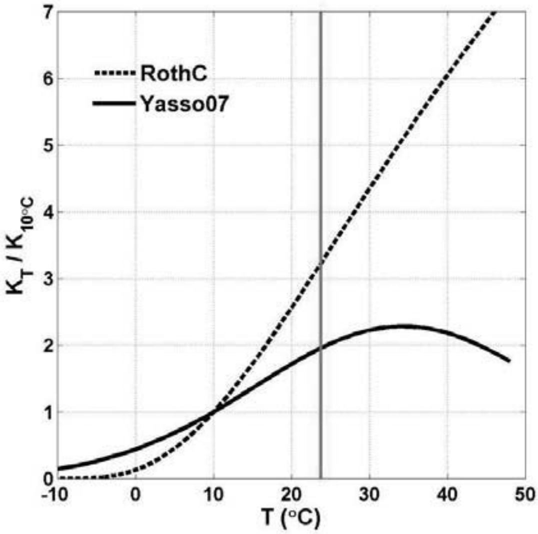

Figure 6: Temperature (T) dependence of decomposition rate (K). $K_{10°C}$ denotes the decomposition rate at 10 °C. The red vertical line (23°C) refers to observed maximum monthly mean temperature in Finland.

The model initialization refers to the assumptions that are made at the beginning of the modelling process. The model initialization stage is thus critical for accurate modelling results. Usually the soil C compartments of the models are not measurable (**III**). Therefore, model initialization can be done either by assuming that agricultural soil C stocks are in a steady state with the current agricultural soil C inputs (**I** and **III**) or by taking account of the historical land-use (**IV**). Taking account the historical land-use is challenging as the contemporaneous soil C measurements do not exist. In this thesis, the initial soil C stock that would have been in the soils of the forest that preceded the present day arable land were estimated on the basis of tree growth tables from the early 20th century, which were converted to soil C input using allometric biomass equations and turnover rates (**IV**). In addition, the modelled soil C stocks were compared to the current C stock for the most productive forest types to make sure that modelled soil C stocks were at a realistic level.

In the present study, the models were validated by using national soil monitoring datasets (**IV**) and data that were obtained from the long-term experiments of Pushchino (**III**) and Ultuna field-studies (**I**). The model performances were evaluated by using the regression coefficient (R^2), modelling efficiency (EF), root mean square error (RMSE) and normalised average error (NAE) (Loague and Green 1991, Janssen and Heuberger 1995).

Yasso07 was also used to study the factors that might have caused the nationwide decrease in C observed in agricultural soils (**IV**).

3.3.2 Carbon response function

Carbon response functions (CRFs) are simple equations that describe the mean annual rates of soil C change as a result of management or land-use change (West et al. 2004). The CRFs are based on the quantitative analyses of large numbers of field measurements. Various CRFs have been developed for different geographical regions and purposes (West et al. 2004, Poeplau et al. 2011). The CRF developed to predict the change in soil C after land use-change from arable land to grassland in the temperate zone (Poeplau et al. 2011) was used in this study (Figure 7). Results were compared to dynamic soil C models and field measurements (**III**).

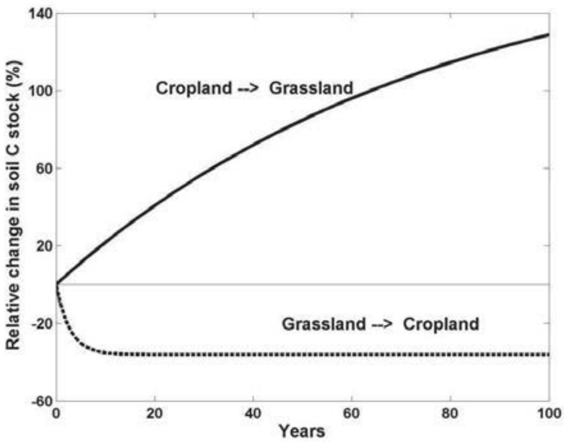

Figure 7. The change in soil C stock after converting cropland to grassland according to the carbon response function presented by Poeplau et al. (2011). The change in soil C stock after converting grassland to cropland (dotted lines) is shown as a reference.

4 RESULTS AND DISCUSSION

4.1 SOIL C STOCK OF FINNISH MINERAL ARABLE SOILS

Finnish mineral arable soils stored from 41 to 67 Mg C per hectare in the topmost 15 cm layer in 2009 (**II**) according to the national soil inventory network. The highest regional soil C stocks were in the west and the lowest stocks in the in the east. The soil C stock tended to be slightly higher in fields under perennial croplands and crop rotation than in fields that grew annual crops. Soil C stock of the topsoil did not clearly depend on the soil type. The geographical distribution of soil C is presented in figure 8 and shows that the soil C stock of the topsoil mineral soil is highest in the coastal region. Nationwide soil C stock was estimated to be about 117 Tg (0-15 cm) in the inventory data study (**II**). For comparison, modelling study suggested that soil C stock in the topmost 1m layer was between 92 and 124 Mg C ha^{-1} (**IV**).

The studies in the soil sciences about arable lands have traditionally focused on the ploughed layer due to its importance in plant production. Therefore, extensive soil profile studies do not exist and C stock in deeper soil layers have remained poorly known. According to soil profile study by Yli-Halla (2000), it can be estimated that total C stock in arable land is about 2-3 times the C stock in topsoil 15 cm. The total C stock in arable land can, therefore, be estimated to be about 300 Tg based on the value of 117 Tg found in the uppermost 15 cm (**II**).

The estimated nationwide agricultural C stock of 300 Tg is comparable for example to the C stock in forested soils, which has been reported by various studies to range from 921 to 1315 Tg (Kauppi et al. 1997, Liski and Westman 1997, Ilvesniemi et al. 2002). Liski and Westman (1995) estimated the mean forested mineral soil C stock per hectare to vary from 40 Mg C ha^{-1} to 119 Mg C ha^{-1} (1m soil layer) depending on the productivity of the forest type. The high soil C stock in the arable land can be explained by the fact that cultivated lands had been established by converting most fertile forest types with high soil C stock into agricultural land. The C stock in the pre-cropland forest soils was estimated to be 159-191 Mg C ha^{-1} (1m soil layer) based on the modelling study (**IV**). This range of values is comparable to the soil C stock of 92.1-165 Mg ha^{-1} (0-40cm) currently found in most productive forest types (Karhu et al. 2011).

The soil C stocks of Finnish cultivated soils are comparable to the 94 Mg C ha^{-1} in the topmost 25 cm soil layer of soils in Sweden (Andrén et al. 2008), but are higher than those reported for Central European countries. Lettens et al. (2005) estimated that cropland soils in Belgium store from 47 to 56 Mg C ha^{-1} (20 cm) depending on the region. Soil C stock in Swiss arable lands has been estimated to be 41 Mg C ha^{-1} on average in 20 cm soil layer (Leifeld et al. 2005).

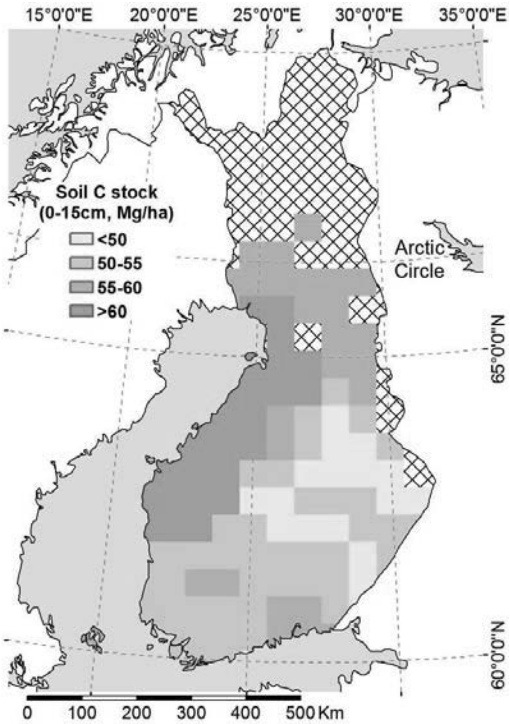

Figure 8. Geographical distribution of soil C stocks (Mg ha^{-1}) in Finnish mineral agricultural land (0-15 cm). Soil C stock is shown in a 50km*50km grid, each square grid represents the mean soil C stock within that grid. Soil C stocks are shown only for those square grids that have more than 1000 ha agricultural land.

4.2 ARABLE SOILS IN FINLAND ARE LOSING C

The arable mineral soils in Finland annually lose 0.22 Mg C ha^{-1} nationwide based on the inventory data (**II**). The change in soil C stock between 1998 and 2009 suggest that the rate of decrease was greater for annual croplands than for perennial croplands or in fields under crop rotation. In addition, the rate of change in soil C stock seems to depend on the soil type especially in annual cropland: the coarser the soil type the greater the decrease in C tends to be (**II**). The modelling study indicates the decreasing rate that range from 0.29 to 0.36 Mg C ha^{-1} depending on the region (**IV**). The modelling study also suggests that the rate of decrease of C is highest in the west and lowest in the north. Both the modelling and experimental results are of similar magnitude as one must take into account that modelling results represent the 1m soil layer and inventory data represent the topmost 15 cm layer.

An annual decrease in soil C of 0.22 Mg C ha^{-1} estimated and reported in this thesis corresponds to the nationwide C loss of 0.5 Tg (**II**), which amounts to 2.5% of the total greenhouse gas emissions of Finland. Soil C emissions from cultivated mineral soils were found to be higher than previously thought. For instance, the greenhouse gas inventory report suggested that mineral soils in Finland were acting as a C sink using the IPCC default methods (Statistics Finland 2011).

Monitoring and modelling studies elsewhere in Europe have shown contrasting trends in soil C. Soil C in Belgium croplands were found to decrease at the rate of 0.19 Mg C ha^{-1} (Sleutel et al. 2006), which is comparable to the result obtained in this thesis. Soils of all types of land use in England and Wales overall are losing C at the rate 0.31 Mg C ha^{-1}, although in arable soils the decrease was found to be considerably less (Bellamy et al. 2005). Soil C was also reported to have decreased in Norway (Riley & Bakkegard 2006), Austria (Dersch & Böhm 1997) and France (Saby et al. 2008), but a direct comparison with those results is difficult as those studies expressed the changes in C concentration. The relative change in C is, however, at similar magnitude to one observed in Finnish arable lands. Other studies in Europe show either no particular trend as in Denmark (Heidmann et al. 2002) and Sweden (Andrén et al. 2008) or a slight increase as found in the Netherlands (Reijneveld et al. 2009).

It is likely that the modelling study (**IV**) overestimated the changes in soil C to some extent. The study was based on the assumption that the present day Finnish agricultural lands were cleared of forest about 110 years ago. A recent case study on historical changes in agriculture in Nummenpää-village in southern Finland (Schulz 2014), however, showed that the area of agricultural land reached its present level as early as 1870 at that particular site. In the late 19th century food production relied to a large extent on meadow-based agriculture and field cultivation had replaced the meadows by the end of 1920. Typical agricultural land is therefore likely to be older than assumed in study **IV.** It is also the case that some of the agricultural fields might have been established on natural grasslands rather than on cleared forest sites. The overestimation of the soil C decline in study **IV** is, however, relatively small because the long-term changes in soil C in Yasso07 results mainly in the changes in the humus (H) and the non-solubles (N) pools (figure 4 and study **III**), and the decomposition rates in both these pools are slow (Table 1). Nonetheless, this phenomenon emphasizes the need for gaining more knowledge about agricultural history at the regional level.

For comparison, the emissions per hectare from Finnish mineral soil (0.22 Mg C ha^{-1}) are an order of magnitude smaller than the emissions from cultivated organic soils. Drained peatlands are the hotspots of soil CO_2 emissions in Finnish food production chain: emissions are estimated to be about 3.2-5.9 Mg C ha^{-1} yr^{-1} (Maljanen et al. 2007).

4.3 REASONS FOR THE DECREASING TREND IN C IN FINNISH ARABLE SOILS

Inventory study **II** showed that soil C tends to decrease more in annual croplands than in perennial croplands or from fields under a crop rotation, which indicates that the shift towards the increasing cultivation of annual crop plants has contributed to the noted nationwide soil C loss. Soils of coarse texture were also found to be more sensitive to management practices than clay soils (**II**). The long-term field trial at Ultuna also showed that the soil C sequestration rates are clearly affected by the quantity and quality of soil C input (**I**), which further supports the idea of management induced changes in soil C. The application of decomposed material such as farmyard manure and peat in Ultuna increased the soil C stock more than the application of similar quantities of fresh plant residues. The fallow treatment which had no soil C input resulted in the highest soil C losses (**I**).

However, the fact that soil C decreases at similar magnitudes in the different regions of Finland (**II,IV**), despite having varying percentages of the land under annual cropping, suggests that the deforestation and the implementation of cultivation is likely to be the major driving forces for the observed C losses from Finnish arable mineral soils. Soil C accumulated during the forested or natural grassland stage is still decomposing and the agriculturally related C inputs (crop residues, manure…) are not enough to counterbalance the soil C losses (see Figure 2). This explanation is further supported by the findings of two long-term field experiments at Ultuna (**I**) and Pushchino (**III**), which show that despite the decadal long field trials, the soil C still do not show clear signs of being in a steady state. Possible climatic warming contributes only slightly to the nationwide decrease in soil C (**IV**).

Previous studies also indicate that cultivation of annual crops, tillage and bare fallow decrease the C in the soil (Paustian et al. 1997, West and Post 2002). Regular tillage and over-wintered fallow land are usual management practices especially in annual crop cultivation, and therefore it is not surprising that perennial crops are known to be beneficial in respect of soil C (Guo and Gifford 2002, Post and Kwon 2008). Tillage breaks up soil structure and releases the aggregate associated C (Beare et al. 1994, Six et al. 1999, Mikha and Rice 2004). The effect of tillage on aggregate associated C in the boreal region, however, is not likely to be that obvious (Sheehy et al. 2012, Singh et al. 2015).

The area of annually cropped arable lands in Finland has increased by about 50% between 1910 and 2010 (Luke 2015). At the same time with the shift towards increasing cultivation of annual crops, there have been other major changes in the management of agricultural soils. Machinery has become heavier, the use of pesticide and inorganic fertilizers have increased, liming, open drainage ditches have been replaced by sub-surface drainage systems and the use of slurry has increased whereas the use of farmyard manure has decreased. Although nowadays there are signs that more conservative cultivation practices are becoming more common, this demonstrate that on the course of agricultural history there have been dramatic changes in management practices. These changes can have an effect on soil structure and soil C balance either directly or through their impact on soil flora and fauna (Stoate et al. 2001). All these changes in management practices, such as drainage, the use of fertilizers and liming, do not necessarily accelerate the loss of soil C. They can even be associated with increasing the soil C balance, as they can improve the growing condition of crop plants and therefore result in higher-levels of C inputs into the soil. The beneficial effects on soil C conservation of these practices must, however, be further assessed on the regional scale to provide more evidence.

Apart from the management practices of arable land, this thesis point to land-use change as being one possible explanation for the observed C loss from arable land. Lower soil C return and more readily decomposing litter in agricultural ecosystems compared to forest ecosystems led to a net loss of soil C (**IV**). It is generally thought that soil C reaches a new steady state a few decades after land-use change (Poeplau et al. 2011). Data in this thesis, however, indicate that the changes might take considerably longer than previously thought. This has got less attention than it probably should. The soil C pool consists of carbon compounds of varying levels of resistance to decomposition. Chemically resistant C pools might have turnover rates that range from decades to even millennia (Jenkinson and Rayner 1977, Trumbore 1997). Therefore, arable lands will continue to act as a C source for a long time after the conversion from forest to agriculture. As a consequence it is likely that the decreasing trend in soil C will also continue in the future.

Retaining the soil organic C would be beneficial for the climate and also for the environment and sustainable food production. The decrease in soil C could be reduced through improved management practices. Possible measures to reduce soil C loss and increase the retention of soil C include the following: keeping the ground covered all year round, decreasing the tillage intensity, leaving the greater proportion of plant residues in the field and using farmyard manure as a source of nutrients in plant production (Maa- ja metsätalousministeriö 2014, Paustian et al. 1997, Smith et al. 2008, West and Post 2002). Likewise, increasing rotation complexity and including legumes can have a beneficial impact on soil C stocks (West et al. 2004). However, as the bulk of the soil C losses result from the long-term effect of forest clearance, the opportunities to increase soil C stock by management practices are limited. Furthermore, many of the management practices listed above, which are considered to be sustainable such as no-tillage farming and leaving residues on the field are already common practices in Finland.

Because significant net loss of C takes place after the deforestation of farmland, in respect of food production related soil C emissions, it is critically important to avoid yield gaps, obtain as high yields as sustainably possible per unit of arable land, and use as little land area for sustainable diet as possible (Foley et al. 2011). In cases where there is an ongoing net loss of C from the arable land, these needs are even more important. However, it should be noted that soils and their CO_2 emissions are just one factor in the food production chain. The other factors such as transportation, manufacturing, production of fertilizers, waste management etc. should all be assessed to evaluate the overall environmental impacts of food production and consumption systems (Risku-Norja 2011).

4.4 APPLICABILITY OF SOIL C MODEL YASSO07 FOR AGRICULTURAL LAND

The long-term experiments of the Ultuna field-study (**I**) and the land use study of Pushchino (**III**) both suggested that the Yasso07 model simulates soil C stocks and changes in C stock relatively well for agricultural land. The modeling efficiency of Yasso07 for the Pushchio data was 0.60 and the results were consistent with the RothC model. Compared to simple carbon response function (CRF) the use of the dynamic models increased the precision of the estimated soil C stock change. In the Ultuna Yasso07 model accurately predicted the changes in soil C stock that arose from application of various organic amendments for all other treatments. The modelling results generally deviated from the observed soil C stock by less than 10% except for the application of peat for which the chemical quality was uncertain. Performance of the Yasso07 model was similar to other soil C models that used data from the Ultuna field trial, namely: ICBM (Andrén and Kätterer 1997, Andrén et al. 2004, Juston et al. 2010), RothC (Falloon and Smith 2002a), Century (Paustian et al. 1992, Falloon and Smith 2002a), Q-model (Hyvönen et al. 1996, Nilsson et al. 2005) and CN-SIM (Petersen et al. 2005). In addition, the predicted nationwide soil C stock and soil C stock changes were comparable to data from national soil monitoring network (**IV**). Therefore, this present study indicate that the most important factors that affect the decomposition of organic matter are well-represented in the Yasso07 model and the changes in soil C stock in boreal agricultural land can be largely modelled using climatic data and quantity and quality of litter only.

The Yasso07 model predicts the changes in total soil C stock (≈1m). Although this simplifies the use of the model in the ecosystems with deep rooting vegetation such as forest and grassland, it makes the comparison between modelling results and measurements difficult due to a lack of data of soil C in deeper soil layers. The Ultuna (**I**) and Puschino (**III**) experiments provided soil C measurement data for 20 cm and 60 cm soil layer, which were compared to results of Yasso07 model. Therefore, the comparison was based on the assumption that soil C stock below that layer (20 cm or 60 cm) is small compared to topsoil C stock or alternatively that there is no changes happening in the underlying soil C stock. In the comparison of the national soil monitoring data with Yasso07 results (**IV**), the soil C stock of deeper soil layers were estimated based on the literature.

Although Yasso07 and RothC are structurally fairly similar (see Materials and methods), they also have certain differences. In the RothC the decomposition rate is affected by temperature and soil moisture content, whereas the Yasso07 model uses temperature and precipitation as a proxy for decomposition activity in the soil. Therefore, it can be argued that RothC reflects the decomposition in the soil more directly. The use of soil moisture content, on the other hand, requires data on evapotranspiration, which can be estimated relatively simply for arable land (Allen et al. 1998), but not so easily for forested land. This limitation restricts the use of the RothC model to mainly agricultural soils. Contrary to RothC model the effect of soil texture is not taken into account in the Yasso07 model either. The soil inventory study data (**II**) showed, however, that soil C stocks are fairly similar regardless of the soil-type, and also relatively good performance of the Yasso07 model in studies **I** and **IV** indicate that the soil texture might play only a minor role under boreal conditions. Boreal agricultural soils are rich in organic matter. Therefore it is possible that boreal soils are saturated with C and the soil C stock changes are to a large extent controlled by the dynamics of the non-protected fractions of soil C (see Figure 1 in Six et al. (2002)) that are not affected by soil texture or aggregate formation.

It should be noted that both Yasso07 and RothC models only simulate the decomposition of soil organic matter and therefore the amount of C entering the soil has to be estimated separately. The modelling results therefore also depend on how accurately soil C input can be estimated. As noted by Kuzyakov and Domanski (2000), Kuzyakov and Schneckenberger (2004), Bolinder et al. (2007) and confirmed in present thesis (**III**) especially the below-ground soil C inputs are considerable but highly variable. Furthermore, little knowledge of the ages of the fields exists, and such knowledge is important for soil C dynamics of agricultural land (**IV**).

The findings of this thesis can be used to improve the reporting the soil C emissions in UNFCCC greenhouse gas inventories (UNFCCC). The Yasso07 soil C model is used to some extent in Finland (Statistics Finland 2011), Austria (Radunsky 2013), Norway (Rosland 2013) and Switzerland (Heldstab 2013) to report soil C emissions. Currently the model is used in Finland to estimate the emissions from forested soils or from land whose use has changed. This thesis indicates that application of Yasso07 also on agricultural lands could improve the accuracy of the estimated soil CO_2 emissions compared to currently used more basic Tier2 method, which suggests that agricultural soils in Finland act as a C sink instead of source of C emissions.

4.5 TOWARDS MORE ACCURATE MEASUREMENTS AND MODELLING

The changes in soil- C stock are negligible (nationwide annual change being order of 1%) compared to soil C stock. Furthermore, the soil C stock can vary considerably within a vertical soil profile but also from one place to another even within a short distance (Röver and Kaiser 1999, Zhang et al. 2011). Sampling/analysis related errors can also be large (Goidts et al. 2009). Therefore it is no wonder, that it was the interest in soil C research due to climate change that highlighted the need for more accurate soil sampling methods. Uncertainties in soil sampling and analysis combined with the inadequacy of monitoring networks have raised the issue of whether nationwide changes soil C stock can be reliably detected by solely using inventory based methods (Saby et al. 2008, Schrumpf et al. 2011). The need for developing carbon monitoring, including not only the use of equivalent soil mass (see next paragraph), but also the use of deep enough sampling, dense enough sampling networks and accurate marking of the sampling points is widely recognized (Goidts et al. (2009) and study **II**). Generally the same requirements also apply to any other soil C studies such as long-term field trials or chronosequence studies.

The fixed depth method is a commonly used method for studying the properties of soils in the environmental sciences (Figure 9). The method relays on the measuring off the soil layers at certain fixed depths of a chosen thickness in centimetres, which are then used to study temporal changes or treatment effects. This is timely issue also in respect of this thesis as all experimental data used in this study were obtained by using the fixed depth method (**I, II, III**). The fixed depth method rests on the assumption that soil is vertically homogenous i.e. there is no vertical gradient of characteristics in the soil profile. This assumption, however, is very rarely the case and as a consequence the use of fixed depth method can lead to an erroneous interpretation of data. Ellert and Bettany (1995) and Wendt and Hauser (2013) proposed the use of equivalent soil mass method, whereby depth of soil is measured as a mass in kilograms rather than by centimeters of depth. The equivalent soil mass method is the method of choice in recent soil C studies (Mishra et al. 2010, Poeplau et al. 2011, Mcleod et al. 2013) and it is likely that equivalent soil mass method is also gradually replacing the fixed depth method also in other environmental applications such as soil nutrient-, biological and physical property- studies.

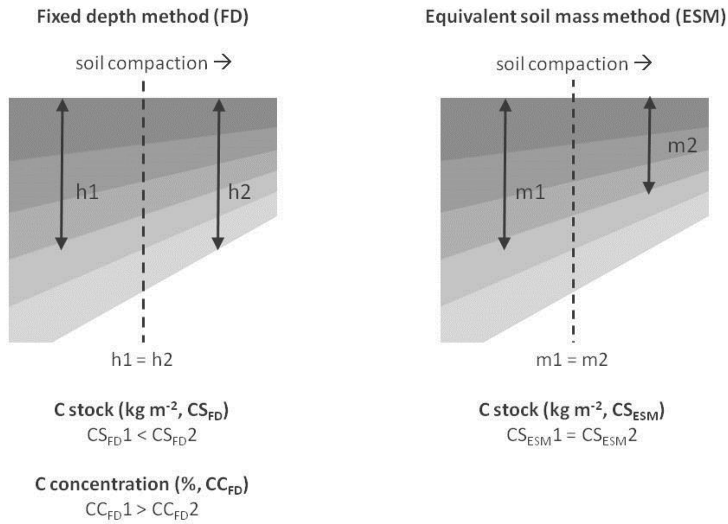

Figure 9. Fixed-depth method versus equivalent soil mass method. Soil layers in a profile are defined according to their depth in centimetres in the fixed depth method (h), whereas the equivalent soil mass method is based on the soil mass (m) in kilograms. Soil compaction is used as an example to demostrate how the two methods can produce different results. The varying grey shades of the profile refer to the C concentration (darker the colour, the higher the C concentration).

Dynamic soil C models provide a cost-efficient way to explore the changes in soil C stock. Soils consist of several C pools for which the decomposition rates range by several orders of magnitude and C continuously enters and leaves the soil, thus soil C models can also be used as efficient tools to study the factors that affect the soil C balance (**IV**). In other applications, such as future scenarios including soil C response to a warming climate or to changing management, soil C models can be the only feasible options. The use of models can also be proposed as a way to improve the efficiency of sampling; that is to reduce the number of sample plots needed without reducing the precision of the determinations (Peltoniemi et al. 2007).

The results of soil C modelling, whether accurate or inaccurate, do not depend only on model performance. In many cases the input data required by the model are imprecise (i.e. random errors) or possible even inaccurate (i.e. systematic errors). This makes the validation of a model difficult to achieve. Modelling results are only as reliable as their input data. While climatic records and soil properties are usually well represented, in many cases soil C measurements are not repeated often enough for good model comparisons to be made. This research also highlighted the need to improve the knowledge on the below-ground C inputs, the age of the fields and the past land use of arable land (**IV**). Information required for modelling depends on the complexity of the model (Peltoniemi et al. 2007). A valid comparison between modelled results and actual measurements is subject to difficulties, because the soil C compartments in most models (including Yasso07 and RothC used in this thesis) are conceptual (Smith et al. 2002). This is understandable as the highly diverse C compounds occurring in soils are described in only a few C compartments in the models.

5 CONCLUSIONS

Over last decades interest in soils has increased due to climate change and elevated CO_2 concentration in the atmosphere. This study showed that cultivated mineral soils in the boreal zone are considerable pools for C despite of their relatively small land area. Furthermore, the present study indicated that this pool is decreasing, which is unfavourable in terms of the climate, water systems and sustainable food production. Although the decrease in soil C can partly result from climatic change such as rising temperatures, study data indicate that the major part of this decrease is anthropogenic. Land-use change from natural-state forests to cultivated agricultural land triggers the long term decrease in soil C due to the lower levels of annual C inputs into farmland than is deposited in forest soils and due to more easily decomposable litter. Cultivation practices have also changed during the course of agricultural history. Two examples of these practices are the change towards increasing cultivation of annual crops and the simplification of cropping system (monoculture), which have obviously contributed to the decrease in soil C stock.

Soil C can be maintained through improved management practices. However, as the causes of soil C losses can mainly be traced back to deforestation the optimizing the efficiency of food production per cultivated land area is an equally important. Maintaining or improving soil fertility and using good cultivation practices will enable the use of agricultural land for the production of high yields and thus minimize the climatic impacts.

Detecting the changes in soil C stock by measurement based methods is not straightforward and this study highlighted the clear need for developing monitoring networks and sampling techniques. Extensive networks with adequate numbers of sampling points are costly but necessary to detect the small changes in soil C stock compared to overall soil C stock. Therefore, it is no surprise that in many applications, such as in greenhouse gas inventories it is tempting to use various process-based models rather than direct measurements. Modelling can be the only alternative for other applications such as predicting the outcomes of future scenarios. The research in this thesis indicated that Yasso07- soil C model, which is extensively used in forested land, can also be applied in boreal agricultural soils.

This research mainly focused on the topsoil C stock. Considerably larger C pools, however, lie underneath the topsoil in deeper soil layers. Whether the results of this thesis reflect the changes of C pool of these deeper soil layers is not known. Therefore, it would be crucial to investigate the storage of deep-soil C and the effect of long-term agricultural practices on it in future studies. Due to high costs, it is hardly ever achieved using large scale soil sampling. Rather, by establishing a few intensively studied ecosystem level research stations on boreal agricultural land, similar to the SMEAR stations (Station for Measuring Ecosystem-Atmosphere Relations) on forested sites, could give us information about C fluxes in the land ecosystem-atmosphere continuum and possibly could lead us to a better understanding of the fate of the deep soil C.

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