The Carbon Copy of Human Activities

How long-term land use explains spatial variability of soil organic carbon stocks at multiple scales

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at multiple scales

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De twee meest wijdverbreide misvattingen over promoveren zijn “Je bent vier jaar met hetzelfde bezig” en “Je moet het allemaal alleen doen”. Wat betreft de eerste misvatting: Ik heb afgelopen jaren ervaren als een erg afwisselende periode en heb zeker niet het idee gehad dat ik steeds met hetzelfde bezig ben geweest. Ik ben bezig geweest met onderwerpen variërend van Europese politiek tot bosecologie en heb volop gelegenheid gehad voor allerlei extra activiteiten als congressen en workshops bezoeken en veel leuke cursussen volgen, die het nog afwisselender maakten.
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Chapter 1

General introduction

1.1. Introduction

People interact with the environment. They alter the landscape by agricultural or forestry activities, or by removing vegetation for fuel or building. Upon spatial allocation of their activities, people respond to the spatial variation in the landscape. Land use types differ in their requirements for nutrients, water and landscape stability on the one hand and in requirements for accessibility conditions on the other hand, and people tend to allocate land use in the best possible way. The use of natural resources like land, soil and water can change the spatial patterns of the landscape properties and impacts the way the landscape functions. Prolonged agricultural use can for example deplete the soil from nutrients, making it unsuitable for further use. Second, people can induce erosion through tillage (Van Oost et al., 2005) or through land use change (Lesschen et al., 2007), and this influences the topography, both at field scale or at the scale of a complete river basin. Extensive deforestation in Roman times is for example assumed to have caused large-scale changes in erosion and sedimentation and to have influenced the formation of river terraces along the Rhine (Mäckel et al., 2003; Erkens et al., 2009). Humans are long recognized as a soil forming factor as well (Jenny, 1941). Human activities related to soil formation mainly focus on maintaining or enhancing the soil fertility, for example in the plaggen soils in Western Europe (Blume and Leinweber, 2004) or in the Terra Preta in Latin America (Woods et al., 2009). Properties, variability and functioning of the landscape can be seen as a result of a constant interaction between biophysical characteristics of the landscape and the land use. To understand how the current variability of landscape properties like suitability for agricultural use, or the availability of natural resources in the landscape has emerged, and to be able to anticipate how landscapes and landscape functions can change in the future, the interaction between the landscape and the land use has to be understood.

1.2. Soil organic matter and global change

1.2.1. Backgrounds

Soil organic matter (SOM) is an important source of nutrients, improves the water holding capacity, helps maintaining a good soil structure, is a host for the soil biodiversity and is an important reservoir in the global carbon cycle. Loss of SOM increases the erosion and desertification risk, restoring of declined SOM contents is very time-consuming, and SOM losses result in emission of CO_2 to the atmosphere and increased atmospheric CO_2 concentrations (Commission of the European Communities, 2006). Understanding the behaviour of SOM is therefore important for research on climate change and sustainable land use.

Loss of SOM and organic matter (OM) from vegetation through human activities is assumed to have affected the terrestrial carbon stock for millennia. Starting with the spread of agriculture since approximately 8000 BP, deforestation and irrigated rice cultivation have decreased terrestrial carbon stocks and increased atmospheric CO_2 and CH_4 concentrations (Ruddiman, 2003). Human activities are sometimes even seen as the dominant factor shaping the environment (Richter Jr, 2007) and some state that the geological era we live in should therefore be referred to as the Anthropocene (Crutzen and Steffen, 2003; Zalasiewicz et al., 2008). Because of the short human presence on earth compared to geological timescales, the global impact of human activities, especially at the longer term, is however argued to have significantly influenced the environment (Kroonenberg, 2006).

Since 1900, global average temperatures show a pronounced increase compared with 1400-1900, which is strongly correlated with increases of the CO_2 concentration in the atmosphere since the start of the industrial era (Mann et al., 1998). Global emissions of greenhouse gasses including CO_2 due to human activities are rising since pre-industrial times and currently, the atmospheric CO_2 concentration exceeds the natural range over the last 650,000 years. Increased CO_2 emissions are due primarily to fossil fuel use, but land use change provides a small, but significant contribution. Human-induced climate changes are expected to continue in the coming centuries and are expected to strongly impact ecosystems, food production, safety in coastal areas and human health (IPCC, 2007). To understand the impact of these changes, insight in functioning of SOM under continued use is essential.

1.2.2. Carbon dynamics

Worldwide, 1500 billion tonnes of organic carbon is stored in the soil, which is twice the atmospheric carbon pool and three times the vegetation carbon pool (Smith, 2004a). These three pools, together with carbon pools in oceans and in geological reservoirs interact in the global carbon cycle. Dynamics of carbon in soil, vegetation and atmosphere are directly influenced by human activities. Of these three pools, the soil organic carbon (SOC) pool by far has the longest residence time: ages of SOC can range up to millennia (Schlesinger, 1990).

Soil organic carbon makes up on average 58% of SOM (IPCC, 2006). The size of the SOC stock at a certain time and location depends on the balance between input and output of OM (Freibauer et al., 2004). Amount and quality of OM inputs are controlled mainly by vegetation: Forests have a high annual input that is relatively difficult to decompose while OM input from croplands is low and easily decomposable. Both fresh OM and available SOM is partly respired by soil fauna, partly it is turned over to forms of SOM that are resistant to further decomposition (Wild and Russel, 1988). The fraction of fresh OM that is respired depends on the chemical properties of the OM input.

Decomposition of SOM results in CO_2 emission from the soil. The rate and amount of decomposition depend on several factors: When SOM stocks are larger, decomposition speed is higher (Bellamy et al., 2005). Decomposition slows down under lower temperatures or higher moisture contents (Parton et al., 1987; McLauchlan, 2006). Soil texture and structure influence chemical stabilization and physical protection of SOM (Six et al., 2002). Especially the clay type, for chemical stabilization, and the clay content, influencing the aggregate stability, are of importance.

If input of OM exceeds output, the soil sequesters carbon; otherwise the SOC stock will decrease. Under unchanged input and output of OM and unchanged environmental conditions, the SOC stock will eventually move to a dynamic equilibrium where sequestration equals emission (Wild and Russel, 1988). Upon a change of land use or climate, inputs and outputs of OM change and consequently the SOC stock will change to a new dynamic equilibrium (Smith, 2004a).

1.2.3. Human impact on carbon dynamics

People have a clear impact on SOC dynamics through land use and management activities. Land use directly influences OM inputs because land use types differ in the amount and quality of vegetation residues. Land use change alters the input and output of OM and will therefore result in changes of the SOC stock. Management activities like manuring or fertilizing in agricultural land or the choice for a tree species in forests influence the amount and quality of OM input as well. Tillage, cultivation, thinning of forests or harvesting can increase the output of OM by enhanced mineralization due to soil disturbances. Effects of changes in land use and management on SOC stocks are fast in the first years after a change occurs and then quickly level off (Smith, 2004a), but small changes of SOC stocks can be observed up to centuries after changes.
The allocation of land use produces a spatial pattern of inputs and outputs of OM. A close copy of this pattern is printed on the landscape in the shape of spatial variability of SOC stocks: a carbon copy of the human activities. The carbon copy of the land use is very clear if large spatial differences of input or output of OM persist for a prolonged time, while a less pronounced pattern of input and output of OM prints a fainter carbon copy.
When the input or output of OM changes upon a change of land use, the carbon copy of the old land use will get overwritten by the new land use. Because SOC stocks respond slowly on changes in input and output of OM, it can take centuries before the carbon copy of the past land use is overwritten, dependent on how clear and how persistent the carbon copy was, and on how clear and persistent the carbon copy of the new land use is. Several studies show that past land use indeed influences present-day SOC variability. Sonneveld et al. (2002), Pulleman et al. (2000) and Droogers et al. (1997) showed that grasslands with contrasting management over the past decades differ in SOC content. Also historical land use up to centuries ago influences present-day soil conditions (Knohl et al., 2003; Hupy and Schaetzl, 2008; Kätterer et al., 2008; Mueller and Kögel-Knabner, 2009).

1.3. Soil organic carbon stocks

1.3.1. Spatial variability of soil organic carbon stocks at multiple scales

Every plot, landscape, region, country or continent shows spatial variability of the factors that influence the input and output of OM to the soil. At plot scale, small-scale variation in OM inputs, bioturbation, depth of stagnic horizons, clay content or iron oxide contents determine small-scale variations in SOC stocks (Schrumpf et al., 2008). Small-scale variability results from differences in vegetation type and growth conditions, and differences in the species and activity of soil fauna. These are influenced by variation of topography, groundwater dynamics and land use and management at the landscape scale (Pennock and Veldkamp, 2006). Groundwater variations and topography result from geology, climatic conditions and management interventions. Variation in vegetation partly results from small-scale variation in soil and moisture conditions (Schöning et al., 2006; Buis et al., 2009), but patterns are to a large extent controlled by the land use or the management. Biophysical conditions set limits to the suitability for land use at a certain location, both at large scale through climatic patterns, or at small scale, through e.g. soil and drainage conditions, and through that influence the land use pattern (Verburg et al., 2004). At continental scale, patterns of climatic zones influence SOC variability (McLauchlan, 2006). Because of the impact of precipitation and temperature on temporal dynamics of SOC (§1.2.2), the SOC stock increases with increasing precipitation. At a fixed precipitation level, SOC stocks increase with decreasing temperature (McLauchlan, 2006).

1.3.2. Inventories and reporting of SOC stocks

Due to the increased acknowledgement of the importance of the SOC stock for sustainable use of natural resources, several policies require countries to improve the knowledge on national-scale SOC variability. First, the European Soil Strategy identifies seven major threats for the sustainable future use of soil resources, including decline of SOC stocks. Therefore, following the European Soil Strategy, countries need to identify areas where there are risks of declining SOC stocks (Commission of the European Communities, 2006).

Second, the Kyoto protocol (UNFCCC, 1997) aims at limiting the effects of future human-induced climate changes (§1.2.1). This is foreseen both by decreasing greenhouse gas emissions, and by increasing the uptake of greenhouse gases in other reservoirs of the global carbon cycle. The European Union (EU) aims at decreasing greenhouse gas emissions by 8% in 2008-2012 relative to 1990. The Netherlands agreed on a decrease of 6% in 2008-2012 and the Dutch post-Kyoto goal is a decrease of greenhouse gas emissions by 20% in 2020 (Brandes et al., 2007).

The countries with the highest greenhouse gas emissions (the Annex I countries) are following the Kyoto protocol required to submit an annual inventory of their greenhouse gas budget, including comparison with the 1990 situation. The inventory has to include emission and sequestration of all greenhouse gases within the Land use, Land Use Change and Forestry (LULUCF) sector that are a result of human activities. In the LULUCF sector, emission and sequestration have to be reported in six land use types: forest, cropland, grassland, wetlands, settlements and other land (IPCC, 2006). In all land use types, SOC stocks and changes of SOC stocks have to be reported while in forests also the carbon stock in the forest floor has to be reported.

Because of the scale-sensitivity of factors influencing SOC stocks (§1.3.1), upon reporting of SOC stocks the factors describing SOC variability should match the scale of reporting. For reporting SOC stocks at national scale, following the guidelines of the International Panel on Climate Change (IPCC) SOC variability is described using soil classification and land use (Arrouays et al., 2001; Rodriguez-Murillo, 2001; Krogh et al., 2003; Bradley et al., 2005; Lettens et al., 2005a; Tomlinson and Milne, 2006). Forest floor carbon stocks are poorly represented in national inventories. Most countries use a default value or do not report carbon stocks in the forest floor. Uncertainties in inventories of SOC stocks and forest floor carbon stocks are high; the uncertainty of the greenhouse gas budget of the Dutch LULUCF sector for example is around 100%. Uncertainty in SOC and forest floor carbon budgets is an important contribution to this uncertainty (Brandes et al., 2007).

1.4. Contents of the thesis

1.4.1. Objectives

There is a lack of understanding how land use and land use change at the long term influence spatial and temporal variability of SOC stocks. Increased insight in how land use influences carbon dynamics can help explain current and future spatial variability of SOC stocks. Main objective of this thesis is to identify how insight in the spatial variability of SOC stocks can be improved. The objective is subdivided in five research questions:

1. What are the determinants for SOC variability at landscape scale in forests and agricultural land?
2. How are past, present-day and future land use influencing spatial variability and temporal dynamics of SOC?
3. How are biophysical determinants for SOC variability interacting with land use, and what are the consequences of such interactions for explaining SOC variability?
4. Which drivers for SOC variability are dominant at landscape, regional, national and European scale, and what are the consequences of scale sensitivity?
5. Can information on long-term land use be used to improve national-scale inventories of SOC stocks?

The research was done within the framework of the project *“Soil carbon dynamics and variability at the landscape level: its relation to aspects of spatial distribution in national emissions databases”* (ME3) which belongs to the National research programme Climate Changes Spatial Planning (CcSP).

1.4.2. Study area

The research focuses on the Dutch sand area. The sand area covers 43% of the Netherlands and comprises three quarters of the Dutch forests. Therefore, it is an important region for the Dutch terrestrial carbon budget and decreasing uncertainties of the SOC stocks in the sand area is highly relevant.

Landscape formation
The Northwest European sand area stretches from the Netherlands, through northern Germany, Denmark, Poland and the Baltic countries (Fig. 1.1a). The present-day landscape of the area is dominantly formed since the Pleistocene. During the Saalien ice age (300,000-120,000 BP) glaciers transformed mid-Pleistocene (300,000-120,000 BP) river deposits from predecessors of the Meuse, Rhine and Eridanos into ice-pushed ridges at the glacier front. All sediments under the glaciers were crushed, grinded down and compressed to glacial till. Melting of the glaciers by the end of the Saalien resulted in formation of brook valleys and fluvioperiglacial valleys and flats (Fig. 1.1c). During the Weichselien ice age (110,000-10,000 BP) the Netherlands had a cold and dry climate where extensive areas of eolian cover sand flats and ridges formed (Fig. 1.1c).
In the Holocene (since 10,000 BP), the climate became more gentle. Due to sea level rise and vegetation changes, groundwater levels increased. At the high, dry and relatively fertile ice-pushed ridges, open forest vegetation with oak forests became typical and orthic podzols developed. In infertile eolian sands, acid podzolic soils were formed. At the wetter parts of the landscape brook forest with elders was the original vegetation and humic gleysols are typical. Lowest and wettest parts of the landscape were covered with a wet heath and sphagnum vegetation and at many locations there was peat formation (Berendsen, 1997).

Land use history
Land use history differs between the dry (eastern, central and south) and the wet (northern and part of eastern, Fig. 1.1b) sand landscape. In the dry sand landscape, settled agriculture started since the Bronze Age (2000-800 BP) with the Celtic field system. Celtic fields were mainly found at the slopes of ice-pushed ridges, locations with good moisture conditions and relatively fertile sand soils (Kooistra and Maas, 2008). After changing from the Celtic fields agriculture the plaggen agriculture since the early Middle Ages (500-1500 AD), locations of the agricultural lands dominantly remained unchanged and many Celtic fields were buried under the plaggen cover.

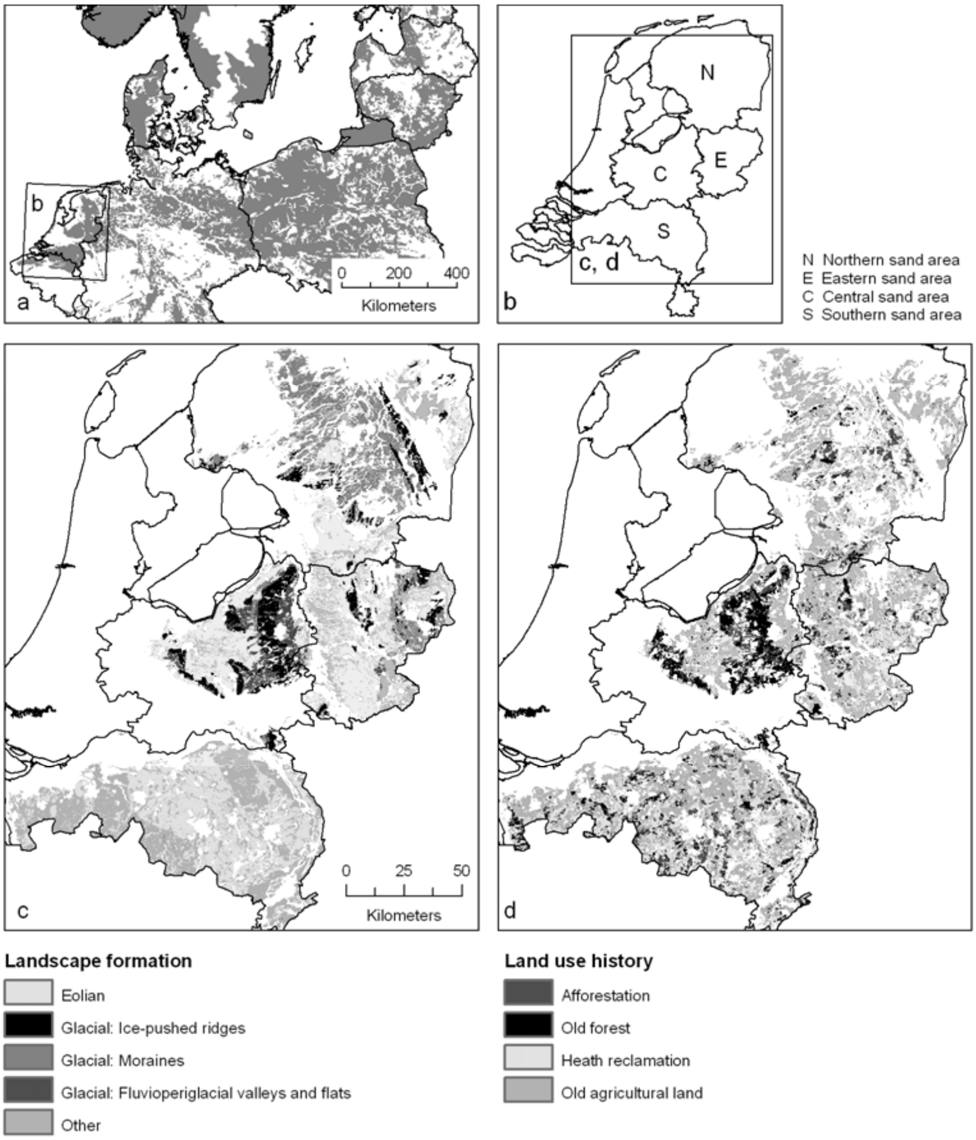

***Fig. 1.1** a. Location of the sand area (grey) in Northwest Europe (European Soil Bureau Network and the European Commission, 2004). b. Regional subdivision of the Dutch sand area. c. Landscape formation of the Dutch sand area, based on (Koomen and Maas, 2004). d. Land use history of the Dutch sand area, based on land use changes between 1900 and 2000 (Knol et al., 2004).*

The plaggen system dominated the sand landscape in large parts of northern Europe for centuries. In this agricultural system, litter or sods were used for animal bedding in stables. The sods together with animal excrements were used as manure (Blume and Leinweber, 2004).
In the earliest phase of the plaggen system only litter was used. Depending on the local vegetation this could include forest, heath or grass litter or straw (Spek, 2004). Litter harvesting and using the forests for sheep grazing resulted in degradation of forests into heathlands. Then, increasing amounts of sand were included in the animal bedding and slowly the litter harvesting transformed into sod cutting. This resulted in further degradation of the heathlands where plaggen were harvested to drift sands.

In the wet sand landscape, settled agriculture started around the Middle Ages with plaggen-based agriculture on relatively dry locations. On areas that were too marginal for the plaggen system, often buckwheat was grown. On soils with a peat cover, the topsoil was burned before sowing of buckwheat (Bieleman, 1992). The peat-buckwheat culture was at its peak of production at around the 1870s and cultivation and burning of peat caused the peat to disappear completely. This was often followed up by incorporating the former peat areas in the plaggen system (Koster and Favier, 2004).

By the end of the 19th century about half of the Dutch sand area was covered with wasteland (Fig. 1.1d), a collective term for heathlands, bogs, dunes and drift sands that were only extensively used (Thissen, 1993). During the 19th century, artificial fertilizers were introduced, enabling the use of marginal lands for agriculture. Together with increasing population, this induced large-scale heath reclamations that continued until the 1960s. The Dutch wasteland area decreased from 3600 km^2 in 1900 to 285 km^2 in 2000 (Knol et al., 2004) (Fig. 1.1d). Heathland areas were mainly converted to agricultural lands while most drift sand areas were afforested (Fig. 1.1d).

The Dutch sand area is divided into four sub-regions differing in parent material, geomorphology, soil formation and land use history (Fig. 1.1b). The southern sand area is a high, dry area, dominated by eolian sand ridges and flats. Large areas of fimic anthrosols and gleyic podzols dominate the landscape. The southern sand area always had a high population density and intensification of land use started around 1350-1450. The central sand area is dominated by the high ice-pushed ridges of the Veluwe. Groundwater is deep and dominant soils are dry podzols and brown forest soils. Intensification of land use started around 1400-1600. The eastern sand area is characterized by low ice-pushed ridges and eolian sand ridges. Gleyic podzols, humic gleysols and fimic anthrosols are frequently found. Intensive agriculture started around 1500-1600. The central and eastern sand areas have an intermediate population density. The northern sand area is a low and flat, originally very wet area with eolian sand ridges and flats, brook valleys and glacial till areas. In the wettest parts of the area peat domes were found which have disappeared since the Middle Ages due to burning, digging off and cultivation. The area is dominated by gleyic podzols and humic gleysols. It is a relatively remote area with low population densities. Intensive agriculture started around 1500-1700 (Spek, 2004).

1.4.3. Outline of this thesis

This thesis comprises seven chapters including the introduction. In Chapter 2-5 determinants and dynamics of SOC variability at multiple scales are discussed (Fig. 1.2). Chapter 2 and 3 are landscape-scale case studies, focusing on the effect of management in forest and land use history in agricultural land on SOC variability. Chapter 3 and 4 discuss the impact of land use history on SOC variability in agricultural land. In Chapter 3 one case study is assessed while in Chapter 4 effects of land use history at different resolution and extent are compared between four case studies across the Dutch sand area. Chapters 4 and 5 are about modelling temporal dynamics of SOC stocks under changing land use. Chapter 4 focuses on the past SOC dynamics to explain current SOC stocks while Chapter 5 presents a projection of future SOC dynamics under several land use change scenarios at European scale. In both Chapter 4 and 5 the modelling of SOC stocks at different spatial and temporal scales and complexities is discussed.

The insights gained in Chapter 2-5 are combined in Chapter 6, to provide a national-scale inventory of SOC stocks. In Chapter 7, results are discussed, research questions (§1.4.1) are answered and a synthesis of the results within the scientific and societal field is provided.

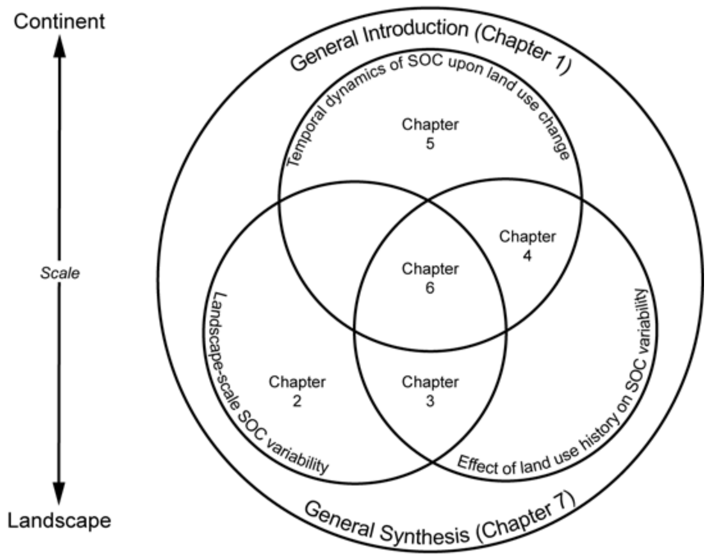

Fig. 1.2. *Schematical outline of this thesis.*

Chapter 2

Effect of tree species and management on carbon stocks in forest floor and mineral soil

Forest soil organic carbon (SOC) and forest floor carbon (FFC) stocks are highly variable. Consequently, the sampling effort required to assess SOC and FFC stocks is large, resulting in limited sampling and poor insight in the size, spatial distribution, and changes in SOC and FFC stocks in many countries. Forest SOC and FFC stocks are influenced by the tree species. Quantification of the effect of tree species on carbon stocks combined with spatial information on tree species distribution could therefore improve insight into the spatial distribution of forest carbon stocks.

This chapter presents a study on the effect of tree species on FFC and SOC stocks for a forest in the Netherlands. FFC and SOC stocks were assessed in stands of beech (Fagus sylvatica), Douglas fir (Pseudotsuga menziesii), Scots pine (Pinus sylvestris), oak (Quercus robur) and larch (Larix kaempferi) at locations with different management intensity.

Under further similar circumstances, FFC and SOC stocks differed between a number of tree species. FFC stocks in the F and H horizon varied between 11.1 (beech) and 29.6 (larch) ton C ha^{-1}. SOC stocks varied between 53.3 (beech) and 97.1 (larch) ton C ha^{-1}. At managed locations, carbon stocks were lower than at unmanaged locations. The Dutch carbon inventory currently overestimates FFC stocks. Differences in carbon stocks between conifer and broadleaf forests were significant enough to consider them relevant for the Dutch system for carbon inventory.

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2.1. Introduction

All signatory countries in the United Nations Framework Convention on Climate Change (UNFCCC) have an obligation to implement a national system for reporting carbon stock changes in the Land Use, Land Use Change and Forestry (LULUCF) sector. As part of such a system, countries must quantify the size, spatial distribution and changes to their soil organic carbon (SOC) stocks. In many countries, information on SOC stocks is rather poor or very scattered (Lindner and Karjalainen, 2007). SOC stock inventories are hampered by large spatial variability and consequently, an enormous sampling effort is needed to reliably assess SOC stocks. Annual changes in SOC stocks are very small relative to total SOC stocks, making monitoring of changes even more difficult (Smit and Heuvelink, 2007).

In the Netherlands a country-wide SOC inventory based on 1:50.000 soil and groundwater maps (Kuikman et al., 2003) represents the current state of knowledge on spatial distribution of SOC. This SOC inventory is used for reporting SOC stocks to the UNFCCC. For a national-scale assessment, the relation between soil, groundwater and SOC stocks can represent SOC stocks correctly. At the landscape scale however, other processes that influence SOC stocks might dominate over the effect of soil and groundwater (Veldkamp et al., 2001). For example, the effects of land use (Lettens et al., 2004; McLauchlan, 2006), land use history (Verheyen et al., 1999; Sonneveld et al., 2002) and management (Dendoncker et al., 2004; Jandl et al., 2007) all influence SOC at finer scales of analysis. At the landscape scale, excluding these diverse landscape effects from SOC stock assessments may lead to erroneous estimates. Including such landscape effects in SOC stock inventories could help to improve national-scale SOC stock estimates.

Forest landscapes comprise 35% of the area in the European Union, Norway and Switzerland, and these areas contain 49% of the SOC stock (Smith et al., 2000; Smith et al., 2005a; Smith et al., 2005c). Thus, forest landscapes comprise an important part of the European SOC stock. In production forests, differences in management and forest age are assumed to influence the rate of soil carbon sequestration and accordingly the SOC stock (Johnson and Curtis, 2001; Jandl et al., 2007). Important in this respect is the choice of tree species (Binkley and Valentine, 1991; Hagen-Thorn et al., 2004; Oostra et al., 2006). Tree species affect SOC stocks by the amount and quality of organic matter (OM) input through litter fall and root activity, and have large effects on SOC stocks relative to other management interventions like harvesting, thinning and fertilizing (Jandl et al., 2007). There is no consensus on the effect of specific tree species on SOC stocks; several studies show different effects (Binkley and Valentine, 1991; Vesterdal and Raulund-Rasmussen, 1998; Jandl et al., 2007).

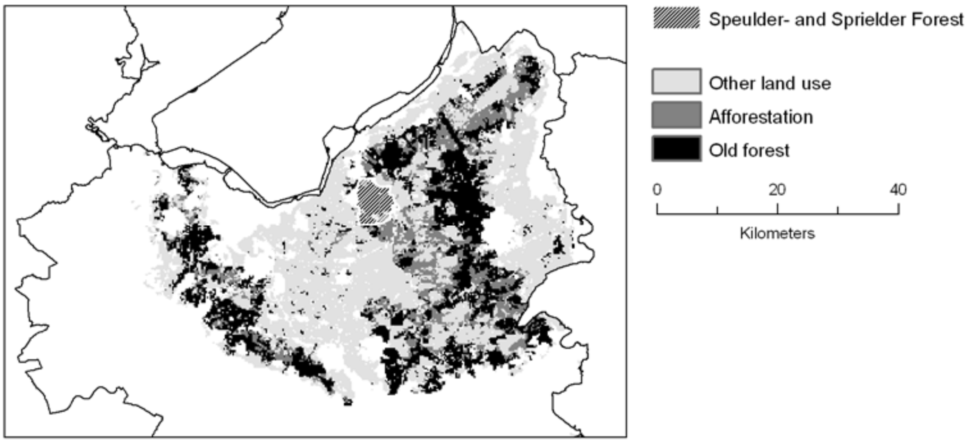

Fig. 2.1. *Location of the Speulder- and Sprielder Forest in the central sand area.*

Spatial data on tree species distribution are often available on regional and national scales. The effect of tree species on SOC stocks combined with information on spatial distribution of tree species can therefore potentially be used to improve insight in spatial distribution of SOC stocks. Differences in SOC stocks between tree species could give an indication of the effects of future management changes, such as increasing areas of broadleaf trees, on carbon sequestration. Additionally, tree species are expected to differ in mitigation potential. This knowledge may help guide tree species choice in case of afforestation with the aim of carbon sequestration.

In forests, large amounts of carbon are stored in organic layers (forest floors). These carbon stocks need to be reported in the UNFCCC national system as well. Inventories of forest floor carbon (FFC) stocks are hampered by large differences in forest floor development between various tree species on the same soils (Hommel and de Waal, 2004; Ladegaard-Pedersen et al., 2005) and by the large variation in depth of forest floors at short distances (Emmer, 1995; Smit, 1999). Forest floor research mainly focuses on semi-quantitative profile descriptions. As a result, many detailed forest floor descriptions are available, but less data exist on carbon content and bulk densities. Consequently, these datasets currently cannot be used for FFC stock calculations. With quantitative data on carbon content and bulk density of different forest floor horizons it is possible to improve insight in differences in FFC stock between different tree species without the need of full cover FFC stock sampling. This can result in a quantitatively more accurate estimation of FFC stocks.

Studies on the effect of tree species on SOC and FFC stocks in Dutch forests, with many introduced tree species, are lacking. This chapter presents a study on the effect of tree species on carbon stock in forest floor and mineral topsoil. The hypothesis is that under relatively homogenous biophysical characteristics, different tree species result in different carbon stocks in the forest floor and mineral soil. Finally, the utility of this information for improving spatial inventories of SOC and FFC stocks is evaluated.

2.2. Materials and methods

2.2.1. Study area and stand description

The study area was located in the Speulder- and Sprielder forest (Veluwe, Netherlands, Fig. 2.1). The forest is approximately 4200 ha and consists of a variation of broadleaf and conifer stands with ages between 20 and 200 years. To focus on the effect of tree species, the following nine factors were controlled for: nitrogen and heavy metal deposition, stand age, soil, groundwater level, other vegetation, stand age, stand history, and site preparation before planting of the trees (Emmer, 1995; Vesterdal and Raulund-Rasmussen, 1998; Jandl et al., 2007). To achieve this, even-aged monospecific stands with specific characteristics were selected: a minimum size of 2.5 ha; an age of approximately 60 years; homogenous soils and a location in the centre of the forest. The tree species investigated were beech (Fagus sylvatica), Douglas fir (Pseudotsuga menziesii), Scots pine (Pinus sylvestris), oak (Quercus robur) and larch (Larix kaempferi) (Table 2.1). These tree species comprise two-thirds of all Dutch forests, making this study representative for forests in the Dutch sand area. Stands that were approximately 60 years of age were chosen because this is the most abundant age class in Dutch forests and is therefore the most relevant class for upscaling. Historically, all studied stands were occupied by forests. Additionally, a beech stand of 170 years was selected. This stand has been relatively undisturbed for the last 50 years and is therefore a reference for SOC stock development.

Parts of all 60-year-old stands have faced some management recently. The Scots pine stand was being invaded by oak, typical of Scots pine forests in the Netherlands (Schelhaas and Nabuurs, 2001). The biophysical characteristics of the stands were very homogenous (Table 2.1), decreasing the possibility that differences between stands were caused by differences in biophysical characteristics. Groundwater was deeper than 1.2 m below the surface in the whole forest and there was no sign of groundwater stagnation in the soil. The slightly loamy ice-pushed ridges on which the forest is situated are also representative of slightly loamy cover sands. Soils were classified as Cambic Podzols (FAO-Unesco, 2003) with a loam content (% particles <50 μm) of 13-18% and are slightly gravelly. Texture, loam content and stone content did not differ between the stands (Table 2.1). The extent of site preparation conducted before tree planting is not known. The mineral topsoil of the larch stand was clearly disturbed, which may indicate that tillage was used as a part of site preparation.

Table 2.1. *Description of the research stands and comparison groups.*

Stand	Tree species	Plant year	Area (ha)	Median sand grain size (µm) min-max	Loam content (% particles <50µm) min-max	Stone content (%) min-max
10D	Beech	1835	6	150-420	10-20	2-8
110C	Beech	1950	2.6	150-600	2-20	1-6
21D	Oak	1948	7	150-600	11-35	1-13
10F	Larch	1945	3.5	150-420	8-20	2-11
111B	Douglas fir	1949	3	150-600	8-20	1-16
111QF	Scots pine	1952	3.5	150-1000	2-25	2-17

Comparison group	Description	n
Broadleaf	Stands 111C and 21D	20
Conifer	Stands 10F, 111B and 111QF	30
Managed	Plots in stands 110C, 21D, 10F, 111B and 111QF.	27
Unmanaged	Plots in stands 110C, 21D, 10F, 111B and 111QF.	22

2.2.2. Sampling and sample treatment

Although long-term accumulation of carbon also takes place in deeper soil layers, focus is on the mineral topsoil because this is the main area of interest in inventories of SOC stocks given Kyoto reporting requirements.

Sample size was estimated based on differences in SOC and FFC content between tree species in other studies (Ladegaard-Pedersen et al., 2005). Sampling was carried out in June - August 2006. In each stand, 10 plots of approximately 100 m^2 were located randomly. In each plot one profile description of the mineral soil up to 1.2 m in depth was made and vegetation was described. Signs of recent management in the plot such as thinning, fresh sawing litter or tree harvesting were registered. Disturbance by wild boars was registered and the disturbed area was measured. One bulk density sample of both the 0-10 cm and 10-20 cm layers of the mineral soil were taken in each plot with a soil core sampler.

In each plot ten points were chosen randomly along a 1×1 m grid. At these points, samples of forest floors and mineral topsoil with a fixed area were taken with a 35 cm^2 monolith profile sampler (Wardenaar, 1987). This sampling device is constructed particularly for forest floor and mineral topsoil description and for sampling with minimal disturbance of the forest floor. Forest floor was described (Table 2.2; Van Delft et al., 2006). Root density was estimated by counting the visible roots and estimating their size. From mineral soil, 0-10 cm and 10-20 cm layers were sampled. From the forest floor, all horizons that could be distinguished separately with sufficient thickness were individually sampled. If horizons were too thin to sample, either the F1 and F2 were combined (abbreviations in Table 2.2) and the Hr and Hh were combined, or the F1 was sampled separately and the F2+H horizons were combined. All horizons together are referred to as the forest floor. In case L material was present on top of the sample, this was removed. Twigs or dead wood in the F or H remained in the sample, the presence of twigs and dead wood was registered.

Table 2.2. *Horizons that are distinguished in forest floor description and sampling (After Van Delft et al. (2006)).*

Horizon code	Description
L	Litter layer. Consists of recently fallen, non-decomposed material with clearly identifiable plant residues. Less than 10% fine organic matter.
F	Fragmented layer. Organic material is partly decomposed. Plant residues are macroscopically recognizable. In this layer, decomposition is most active.
F1	Many plant residues are macroscopically recognizable, 10-30% fine organic matter.
F2	Few recognizable plant residues. 30-70% fine organic matter.
H	Humus layer. Consists of strongly decomposed organic matter, originating from litter fall from decades ago. More than 70% fine organic matter.
Hr	Contains identifiable residues of roots, barks or wood.
Hh	Completely humified.
A/F	Mixture of mineral topsoil and F material (only found at disturbed locations).

Composite samples were made in each plot over all samples and points, per forest floor horizon, or group of horizons, to decrease the effects of short-distance spatial variation. In October 2006, the L horizon was sampled at five locations per stand using a 50×50 cm frame. L thickness was measured at ten random locations per sample.

Samples were dried for 24h at 105°C. The dried forest floor and mineral soil samples were weighed for calculation of bulk densities. Mineral soil samples were sieved with a 2 mm sieve to determine the stone content (Table 2.1). Samples were pre-treated according to NEN5751 (Nederlands Normalisatie Instituut, 1989) and carbon content was determined with a Leco dry combustion element analyzer.

2.2.3. Data analysis

FFC stocks were calculated by multiplying carbon contents with sample mass and dividing this by the area of the sample. SOC stocks were calculated by multiplying carbon content with bulk density and thickness of the mineral soil layer with a correction for stone content.

Differences in carbon contents and bulk densities among forest floor horizons were compared with one-way ANOVA. Forest floor horizon thicknesses, bulk densities of forest floor horizons and FFC and SOC stocks were compared for all stands using a one-way-ANOVA. Additionally, for the 60-year-old stands, differences between tree groups (conifer vs. broadleaf) were tested with t-tests. Parts of the 60-year-old stands had undergone some forest management recently. The effect of whether forests were recently managed or not was tested with t-tests. Interaction between tree group and management was tested with between-subject-ANOVA.

Finally, several easy to measure or easily available variables that can potentially be used to predict FFC and SOC stocks were evaluated. In the Netherlands, these include information on spatial distribution of tree species (Köble and Seufert, 2001; Hazeu, 2005) and management (LNV Directie Kennis, 2007). Many detailed forest floor

descriptions, including thickness of different forest floor horizons, are available (LNV Directie Kennis, 2007). The ratio of H thickness to F thickness gives an indication of turnover of fragmented material to humus and was therefore expected to be a useful index for upscaling and inventory improvement.
First, forest floor characteristics relevant for predicting FFC and SOC stocks were selected with backward conditional linear regression. General linear models (GLM's) were estimated starting with the variable that explains most of the variation in FFC and SOC stocks respectively. For the models with forest floor characteristics as independent variables, all horizon thicknesses were added in a multiple linear regression. For the effect of management and tree species, ANOVA was used. The other independent variables were added one by one in between-subject-ANOVA and mixed models. The significance of the improvements of the GLM's upon adding independent variables was checked with F tests. Coefficients were considered significant at $p<0.05$.

2.3. Results

2.3.1. Forest floor description

The thickness of the forest floors varied between 0-13 cm. Although variability of forest floor horizon thickness was high (CV for L 63%, for F 49%, for H 125%), mean horizon thicknesses were different between most stands (Table 2.3, Fig. 2.2). The forest floor structure differed between stands (Fig. 2.2). The 170-year-old beech stand had the thickest forest floor, mainly because of the Hh horizon which was not present in other stands (Table 2.3). This was not surprising because Hh horizons only develop after 40 to 50 years. F horizon thickness was more variable than H horizon thickness.

Table 2.3. *Mean (Standard Error of the Mean (SEM)) forest floor horizon thicknesses (cm).*

Stand, comparison group	Forest floor horizon				
	L	F1	F2	Hr	Hh
Beech 170	2.64 (0.13)[d]	1.76 (0.14)[ab]	2.79 (0.14)[b]	0.67 (0.09)[a]	2.46 (0.13)[b]
Beech 60	1.38 (0.11)[c]	1.31 (0.09)[a]	1.48 (0.13)[a]	0.36 (0.07)[a]	0.01 (0.01)[a]
Oak 60	2.15 (0.08)[ab]	1.43 (0.13)[a]	1.65 (0.15)[a]	1.26 (0.14)[b]	0.15 (0.04)[a]
Larch 60	1.04 (0.08)[a]	3.29 (0.13)[d]	2.82 (0.12)[b]	0.63 (0.09)[a]	0.12 (0.07)[a]
Douglas fir 60	1.13 (0.09)[a]	2.26 (0.14)[bc]	3.01 (0.17)[b]	0.59 (0.09)[a]	0.00 (0.00)[a]
Scots pine 60	1.67 (0.10)[b]	2.42 (0.16)[c]	1.75 (0.16)[a]	1.27 (0.16)[b]	0.00 (0.00)[a]
Overall mean	1.67 (0.06)	2.08 (0.06)	2.25 (0.06)	0.80 (0.05)	0.46 (0.04)
Broadleaf	1.77 (0.09)*	1.37 (0.08)*	1.57 (0.10)*	0.81 (0.08)	0.08 (0.02)
Conifer	1.28 (0.06)	2.65 (0.09)	2.53 (0.09)	0.83 (0.07)	0.04 (0.02)
Unmanaged	n.d.	2.20 (0.11)	2.16 (0.11)	1.06 (0.09)*	0.12 (0.04)
Managed	n.d.	2.08 (0.09)	2.20 (0.10)	0.58 (0.06)	0.00 (0.00)

[a, b, c, d] Different letters indicate significant differences between tree species ($p<0.05$).
* Significant differences ($p<0.05$) for the tree group and management regime comparisons.
n.d. Not determined.

Table 2.4. p-values of between-subject-ANOVA's on the effect of tree group, management and interactions on forest floor characteristics.

Dependent variables	Independent variables		
	Management	Tree group	Management x tree group
F thickness	0.23	<0.001	0.97
H thickness	0.05	0.01	<0.001
Forest floor thickness	0.02	<0.001	0.05
Forest floor carbon content (%)	<0.001	0.23	0.16
Forest floor carbon stock (ton C ha^{-1})	0.03	0.80	<0.001

Of the 60-year-old stands, conifer stands had thicker F horizons than broadleaf stands. The H horizons did not differ between tree groups. Managed locations had thinner Hr horizons than unmanaged locations and no Hh (Table 2.3). In the F horizons there were no differences between managed and unmanaged locations (Table 2.3). Unmanaged locations showed more variability in H horizon thickness (CV 160%) than managed locations (CV 127%). Variability in F thickness was the same in managed and unmanaged locations (CV 52%).

There was interaction between management and tree group on horizon thickness (Table 2.4). In both conifer stands and broadleaf stands, managed locations had thinner forest floors than unmanaged locations. In broadleaf stands the decrease of forest floor thickness due to management was larger than in conifer stands (1.0 cm vs. 0.3 cm). The same applies to the H horizon. The difference of H horizon thickness between managed and unmanaged locations in broadleaf stands was 1.3 cm while in conifer stands the difference was 0.5 cm. The thickness of the F horizon showed no interaction between management and tree species.

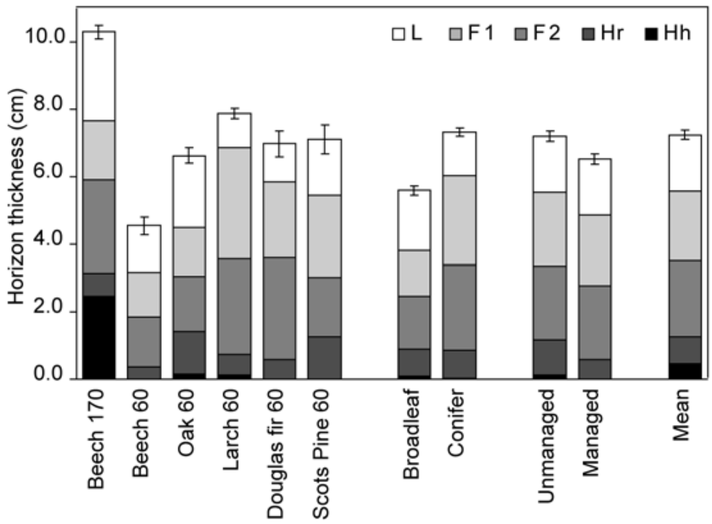

Fig. 2.2. *Mean forest floor thickness (cm) and subdivision in horizons in all stands, including comparison between managed and unmanaged locations and comparison between conifer and broadleaf stands of 60-year-old stands. Error bars indicate SEM of total forest floor thickness.*

2.3.2. Carbon contents

In the forest floor, carbon contents significantly differed between most horizons (p_{ANOVA}<0.05) and decreased from 48% in the L to 12% in the Hh (Table 2.5). Carbon contents in conifer stands were higher in all horizons except A/F than in broadleaf stands ($p_{t\text{-}tests}$ all <0.05) (Table 2.5). For the F and H together, carbon contents were different for some tree species (Table 2.6; p_{ANOVA}<0.001). Conifer forest floors had higher carbon contents than broadleaf forest floors (mean difference 9.8%, $p_{t\text{-}test}$<0.001).

Carbon contents of mineral soil were different for some tree species (Table 2.6; 0-10 cm p_{ANOVA}<0.001; 10-20 cm p_{ANOVA}<0.001). Carbon contents were highest in the larch stand and lowest in the 170-year-old beech stand (0-10 cm) and the 60-year-old beech and Douglas fir stand (10-20 cm). In the 60-year-old stands, unmanaged locations seem to have higher carbon contents than managed locations (0-10 cm: $p_{t\text{-}test}$=0.03; 10-20 cm $p_{t\text{-}test}$=0.06). Between tree groups, carbon contents for the 0-10 cm layer did not differ ($p_{t\text{-}test}$=0.90) while for the 10-20 cm layer carbon content in conifer stands was higher than in broadleaf stands ($p_{t\text{-}test}$=0.04).

2.3.3. Bulk densities

Mean bulk densities differed between forest floor horizons (Table 2.5). Overall, forest floor bulk densities were different for a number of tree species (p_{ANOVA}<0.001). Generally, conifer forest floors had lower bulk densities than broadleaf forest floors, although the difference between coniferous and broadleaved stands was not significant for each horizon. The organic matter content was the main factor explaining forest floor bulk density. The deeper in the forest floor, the stronger the mixing with mineral material in the profile and therefore the higher the bulk density.

Table 2.5. *Mean (SEM) carbon contents (%) and bulk densities (kg m^{-3}) per forest floor horizon for each comparison group.*

Forest floor horizon	Comparison group				
	Overall mean	Broadleaf	Conifer	Unmanaged	Managed
			Carbon contents (%)		
L	48.0	48.2	48.2	n.d.	n.d.
F1	30.1 (1.2)	20.2 (2.1)	32.4 (1.3)	32.4 (2.2)	27.6 (1.9)
F2	29.2 (2.0)	n.d.	26.9 (3.2)	31.0 (6.7)	21.8 (3.1)
F	26.3 (1.9)	21.9 (1.3)	22.8 (2.4)	23.4 (1.4)	19.6 (1.0)
F2+H	23.6 (1.1)	18.3 (1.6)	26.0 (1.0)	24.1 (1.3)	23.2 (1.6)
H	20.3 (1.3)	18.0 (2.2)	25.2 (2.7)	21.4 (2.3)	17.2 (1.7)
A/F	11.8 (1.7)	13.2 (2.2)	8.4 (0.7)	12.5 (1.9)	n.a.
			Bulk densities (kg m^{-3})		
L	17.9	15.8	17.9	n.d.	n.d.
F1	155.2 (15.1)	252.0 (46.9)	118.3 (7.0)	163.4 (31.8)	150.2 (14.8)
F2	292.4 (33.8)	460.5 (46.9)	203.4 (33.4)	290.3 (63.7)	345.3 (49.5)
F	168.7 (6.9)	191.8 (11.0)	154.8 (12.8)	174.3 (12.6)	194.2 (15.6)
F2+H	268.5 (12.9)	323.7 (23.5)	246.0 (13.4)	254.0 (21.6)	279.0 (15.9)
H	379.4 (36.4)	411.4 (60.6)	232.5 (43.2)	337.9 (60.2)	327.0 (61.0)
A/F	473.7 (93.0)	493.5 (111.3)	374.9	473.7 (93.0)	n.a.

n.d. Not determined
n.a. Not applicable

Table 2.6. *Mean (SEM) carbon contents (%) and bulk densities (kg m^{-3}) in forest floor and mineral topsoil for each stand and comparison group.*

Stand, comparison group	Layer			
	L	F+H	0-10 cm	10-20 cm
	Carbon contents (%)			
Beech 170	47.0	24.5 (1.7)	3.4 (0.4)	1.6 (0.1)
Beech 60	46.9	14.9 (2.4)	3.7 (0.3)	1.2 (0.1)
Oak 60	49.5	19.3 (1.0)	4.3 (0.3)	1.5 (0.1)
Larch 60	46.6	32.6 (0.9)	5.2 (0.3)	2.3 (0.1)
Douglas fir 60	50.1	24.0 (1.4)	3.6 (0.3)	1.2 (0.1)
Scots pine 60	47.8	24.0 (1.5)	3.5 (0.2)	1.5 (0.2)
Overall mean	48.0	23.2 (0.9)	3.9 (0.1)	1.5 (0.1)
Broadleaf	48.2	17.1 (1.4)	4.0 (1.1)	1.3 (0.1)
Conifer	48.2	26.9 (1.0)	4.1 (1.1)	1.6 (0.1)
Unmanaged	n.d.	24.4 (1.4)	4.4 (0.2)	1.7 (0.1)
Managed	n.d.	21.7 (1.6)	3.8 (0.2)	1.4 (0.1)
	Bulk densities (kg m^{-3})			
Beech 170	20.8	267 (23.1)	1326.0 (56.1)	1577.5 (45.5)
Beech 60	19.1	236 (17.2)	1105.7 (35.8)	1194.3 (55.1)
Oak 60	13.7	265 (22.1)	1401.4 (55.6)	1588.7 (52.5)
Larch 60	24.9	133 (3.8)	1330.5 (48.2)	1425.6 (51.5)
Douglas fir 60	18.4	201 (22.9)	1365.1 (43.7)	1543.9 (27.3)
Scots pine 60	19.2	222 (8.1)	1448.9 (29.3)	1611.1 (21.4)
Overall mean	19.4	221 (9.1)	1329.6 (57.1)	1490.3 (63.6)
Broadleaf	16.4	251 (14.0)	1253.5 (46.7)	1391.5 (58.3)
Conifer	20.8	186 (10.6)	1381.5 (25.4)	1526.9 (24.9)
Unmanaged	n.d.	204 (15.7)	1391.0 (32.1)	1546.4 (34.0)
Managed	n.d.	218 (11.6)	1278.6 (36.9)	1408.4 (42.7)

n.d. Not determined.

In the mineral soil, dry bulk densities (including stones and organic matter) scarcely differed between stands. The 60-year-old beech stand had a low mean bulk density (Table 2.6). Between tree groups, the bulk density differed: in conifer stands, bulk density was higher than in broadleaf stands (0-10 cm $p_{t\text{-}test}$=0.01, 10-20 cm $p_{t\text{-}test}$=0.05). Managed plots had lower bulk densities than unmanaged plots (0-10 cm $p_{t\text{-}test}$=0.03; 10-20 cm $p_{t\text{-}test}$=0.01).

2.3.4. Carbon stocks

FFC stocks

Mean FFC stocks were different for most stands (p_{ANOVA}<0.001). The 170-year-old beech stand has an FFC stock of 51.1 ton C ha^{-1}. FFC stocks for 60-year-old stands ranged between 12.3 and 30.9 ton C ha^{-1}. Conifer stands had larger FFC stocks than broadleaf stands ($p_{t\text{-}test}$<0.001) and FFC stocks seemed to be larger at unmanaged locations than at managed locations, although not significant ($p_{t\text{-}test}$=0.06) (Fig. 2.3, Table 2.7). There was interaction between management and tree group: in broadleaf stands, FFC stock decreased more due to management than in conifer stands (Table 2.4).

Table 2.7. *Mean (SEM) carbon stocks (ton C ha^{-1}) in forest floor and mineral topsoil in each stand and comparison group.*

Stand, comparison group	Layer				
	L	F+H	0-10 cm	10-20 cm	0-20 cm
Beech 170	2.57	48.50 (3.32)^{c}	43.89 (5.21)^{ab}	24.38 (2.43)^{abc}	68.27 (7.21)^{abc}
Beech 60	1.24	11.06 (1.77)^{a}	39.51 (4.03)^{a}	13.77 (1.40)^{a}	53.28 (5.20)^{a}
Oak 60	1.45	24.62 (1.86)^{b}	59.69 (6.03)^{bc}	21.92 (2.18)^{abc}	81.61 (7.60)^{bc}
Larch 60	1.21	29.64 (1.44)^{b}	66.42 (5.10)^{C}	30.65 (1.73)^{C}	97.08 (6.06)^{C}
Douglas fir 60	1.04	26.33 (1.47)^{b}	45.36 (4.07)^{ab}	17.15 (1.65)^{ab}	62.51 (5.24)^{ab}
Scots pine 60	1.53	26.65 (1.48)^{b}	47.40 (2.08)^{abc}	22.34 (3.84)^{abc}	69.74 (4.83)^{ab}
Overall mean		27.82 (1.66)	50.38 (5.33)	21.70 (2.82)	72.08(7.36)
Broadleaf	1.35	17.84 (1.99)*	49.60 (4.22)	17.85 (1.57)	67.45 (5.54)
Conifer	1.26	27.57 (0.86)	53.06 (2.82)	23.38 (1.78)	76.44 (4.09)
Unmanaged	n.d.	26.24 (0.99)*	58.55 (3.81)*	24.07 (1.52)	82.62 (4.87)*
Managed	n.d.	21.27 (1.94)	45.82 (2.52)	18.70 (1.90)	64.52 (3.97)

a, b, c, d Different letters indicate significant differences between tree species (p<0.05).
* Significant differences (p<0.05) for the tree group and management regime comparisons.
n.d. Not determined.

SOC stocks

Mean SOC stocks in the 0-10 cm layer and the 10-20 cm layer of the mineral soil were different for some stands (p_{ANOVA}<0.001 for both layers separately and together) (Table 2.7). Mean SOC stock in the 0-10 cm layer ranged between 39.5 (60-year-old beech) and 66.4 (larch) ton C ha^{-1}. SOC stocks in the 10-20 cm layer for 60-year-old stands ranged between 13.8 (60-year-old beech) and 30.6 (larch) ton C ha^{-1} and thus were in the same order of magnitude as the FFC stocks. Mineral soil in broadleaf stands contained less carbon than mineral soil in conifer stands but differences were only significant in the

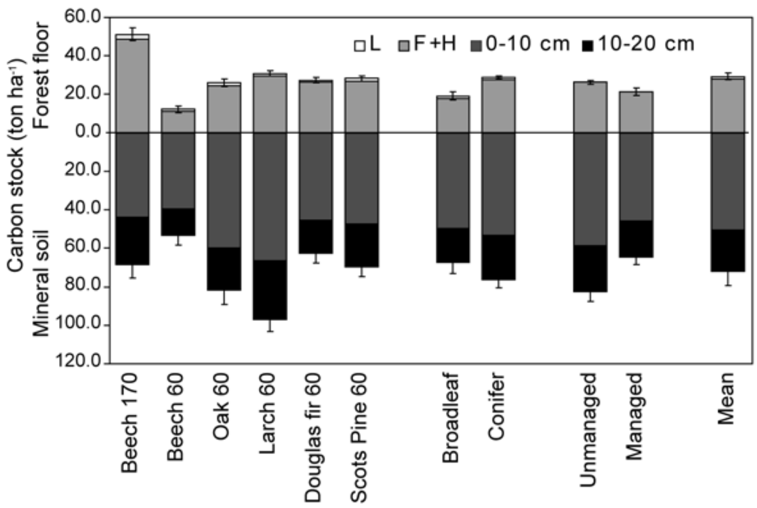

Fig. 2.3. *Carbon stocks (ton C ha^{-1}) in all stands including comparison between managed and unmanaged locations and comparison between conifer and broadleaf stands of 60-year-old stands. Error bars indicate SEM of total forest floor carbon stocks and total mineral SOC stocks.*

10-20 cm layer. The 0-10 cm SOC stock in recently managed plots was 23% lower than in unmanaged plots (Table 2.7, Fig. 2.3). There was interaction between tree group and management: in broadleaf plots, management caused a stronger decrease of SOC stocks than in conifer plots.

The 170-year-old beech stand was expected to have a large SOC stock, because SOC should accumulate in relatively undisturbed forests. The SOC stock in the 170-year-old beech stand was larger than in the 60-year-old beech stand, but compared to the other 60-year-old stands SOC stocks were low, especially in the 0-10 cm layer. The 170-year old beech stand had a different distribution of SOC with depth: the SOC stock in the 10-20 cm layer is 56% of the 0-10 cm SOC stock, while in the 60-year-old stands, SOC stock in the 10-20 cm layer was around 40% of the SOC stock in the 0-10 cm layer (Fig. 2.4).

Overall, carbon stock was highest in the larch stand and lowest in the 60-year-old beech stand (Table 2.7, Fig. 2.3). The distribution of carbon over the layers was comparable in the 60-year-old stands. Broadleaf stands stored a larger percentage of the total carbon stock in 0-10 cm layer and less in the forest floor. In conifer stands, the reverse pattern was found. Management only seemed to influence the amount of carbon in the profile, not the distribution over the layers (Fig. 2.4).

2.3.5. Indices for carbon stocks

Thicknesses of the forest floor horizons and the H:F ratio did not strongly correlate with each other. The FFC stock variability could be explained using thickness of different forest floor horizons and the H:F ratio. Thicker forest floor horizons were related to higher FFC stocks and the H:F ratio showed a negative relation with FFC stocks. Management and tree species explained less of the variability but still had an effect (Table 2.8). When adding management and tree species to the GLM, the GLMs improved significantly. Forest floor characteristics, management and tree species together explained 67% of the variability of FFC stocks (Table 2.8).

Table 2.8. *GLMs to predict FFC stocks and SOC stocks. All models are significant at p<0.05.*

FFC stock		SOC stock 0-20 cm	
Variables	**R^2**	**Variables**	**R^2**
1. Variables independently		*1. Variables independently*	
F1, F2, Hr, Hh, H:F[1] ratio	0.59	Tree species	0.33
Management	0.18	F1, F2, Hr, Hh, H:F ratio	0.12
Tree species	0.04	Management	0.10
2. Combining variables		*2. Combining variables*	
FF characteristics + management	0.61*	Tree species + FF characteristics	0.38*
FF char., management, tree species	0.67*	Tree species, FF char., management	0.39*

[1] F1, F2, etc. are forest floor horizon abbreviations (Table 2.2).
* Significant F changes (p<0.05) upon adding an independent variable.

For predicting SOC stocks, tree species was most important, followed by the forest floor characteristics, the H:F ratio and management (Table 2.8). Forest floor characteristics and management combined with tree species explained 39% of the variability.

2.4. Discussion

2.4.1. Effect of tree species on carbon stocks

Tree species as well as tree groups have effects on FFC and SOC stocks and on the distribution of the total carbon stock over the profile (Fig. 2.4, Table 2.7). Conifer litter contains more components that are difficult to decompose than broadleaf litter, resulting in litter accumulation in the forest floor and formation of acid compounds (Berg, 2000). Therefore, especially the F horizons in conifer stands are thicker than in broadleaf stands. In broadleaf stands there is more biological activity, more fragmentation and humification of F material, resulting in thinner F horizons and thicker H horizons. This is also confirmed by more podzolization features observed in conifer stands than in broadleaf stands. Further, conifers have shallower rooting systems and tend to accumulate more organic carbon in the forest floor (Jandl et al., 2007). Finally, in these acid soils, soil fauna are less active, decreasing the amount of humus mixing through mineral soil (Thuille and Schulze, 2006) and leaving more material in the forest floor. In the mineral soil, the higher amount of podzolization causes carbon contents in the 10-20 cm layer in conifer stands to be relatively higher than in broadleaf stands (Fig. 2.4). These results show that tree group influences the amount of carbon as well as the distribution in the profile.

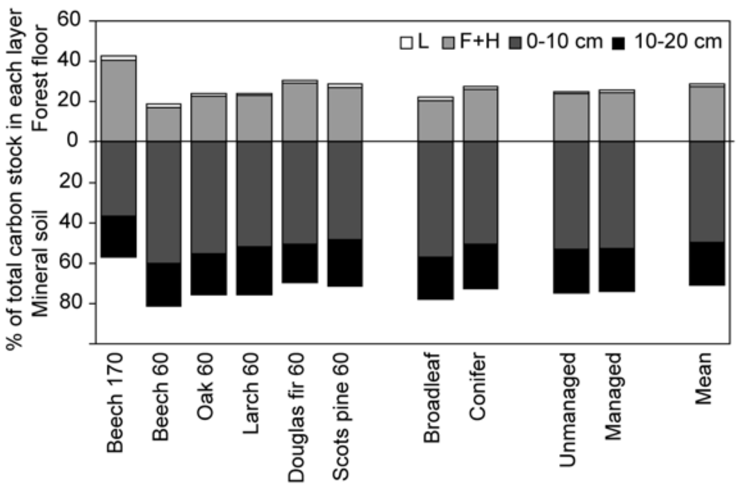

Fig. 2.4. *Distribution of the total carbon stock over forest floor and mineral soil, and over L-F-H and 0-10 cm and 10-20 cm layers.*

The effect of tree species is also demonstrated in the GLMs (§2.3.5) where a higher H:F ratio is related to lower FFC stocks. At the sandy substrate in this study, a high H:F ratio indicates a state where decomposition and turnover is higher than in situations with relatively thick F horizons, resulting in lower FFC stocks. For example, larch litter (with a thick F horizon) is recalcitrant to decomposition on poor soils while oak litter (with a thicker H horizon) decomposes more readily.
Comparison of FFC stocks with other studies is hampered by differences in systems of describing forest floors. Further, formation of FFC stocks and SOC stocks is dependent on stand age (Vesterdal et al., 2002; Thuille and Schulze, 2006) and biophysical characteristics. General trends can, however, be found from comparison with other studies. Vesterdal et al. (1998), Ladegaard-Pedersen et al. (2005), Fischer et al. (2002) and Augusto et al. (2002) found larger FFC stocks and SOC stocks under conifers than under broadleaf trees on the same soil. Further, carbon stocks differed by tree species. Within approximately fifty years, a tree species can substantially change soil chemistry (Binkley and Valentine, 1991). This may explain why Hagen-Thorn et al. (2004) did not find differences in the SOC content of several 40-year-old stands, whereas Oostra et al. (2006) and Fischer et al. (2002) did find differences between tree species in stands of more than sixty years old. Carbon stock differences between tree species are, however, often not consistent along fertility gradients because of interactions between soil characteristics and tree species composition (Vesterdal and Raulund-Rasmussen, 1998).

Compared to the studies mentioned above, FFC stocks and SOC stocks in this study are relatively high. One reason for this result is that older stands are assessed in this study compared to those of other studies (30-40 years vs. 60 years in this study). Second, forest management activities in the Netherlands are less intensive than in many other countries, so in Dutch forests thicker forest floors could accumulate compared to forests in other countries. Finally, differences in carbon stocks between case studies are probably caused by differences in landscape, climate, parent material and other factors like deposition of nitrogen and heavy metals in research stands.

There are two national-scale inventories on forest floor characteristics in Dutch forests. In Leeters and De Vries (2001), forest floor horizon thicknesses and organic matter stocks in 150 Dutch forests are summarized. Trends of differences between tree species are comparable, but Leeters and De Vries (2001) observe FFC stocks 40% larger than the FFC stocks observed in the Speulder and Sprielder forest. The inventory by Leeters and De Vries (2001) includes drift sands with low nutrient content and land dunes with less humification, less mineralization and consequently more accumulation of organic matter in the forest floor than in the more fertile sandy substrate of the Speulder and Sprielder forest. In the Dutch Forest Inventory (LNV Directie Kennis, 2007), at around 1000 locations the thickness of L, F and H horizons was measured. Forest floor thickness for beech, Douglas fir and Scots pine as found in the Speulder- and Sprielder forest were in agreement with the Dutch Forest Inventory data, whereas larch and oak forest floors were clearly thicker than average as measured in the Speulder- and Sprielder forest.

2.4.2. Effect of management on forest floors and carbon stocks

Management activities such as thinning or harvesting can decrease the SOC content: disturbances related to management activities enhance aeration, enhancing mineralization of organic matter. Furthermore, management activities can increase forest floor bulk density by compaction. Mineral soil bulk density can either increase by compaction or decrease through ploughing and stand regeneration (Jandl et al., 2007).
Between managed and unmanaged plots, no difference between F thickness was found, and no interaction between management and tree group (Table 2.3, Table 2.4). This may indicate that between the management interventions, there is enough time for establishment of a new F horizon. Due to enhanced mineralization and removal of material, there is less turnover of fragmented litter to humified OM, causing thinner F and H horizons in managed forests. Further, during management interventions the soil is disturbed, mixing the forest floor into the mineral soil and thereby removing the H (Hedde et al., 2008). Carbon contents in forest floors at managed locations are indeed lower than at unmanaged locations (Table 2.5), although not significant. Altogether, thinner forest floors, lower bulk densities and lower carbon content resulted in lower FFC stocks at managed locations.
Concerning the mineral topsoil, there are contrasting studies on the effect of management. Johnson and Curtis (2001) found no clear effect of management in general. Effects were interrelated with tree species and were highly dependent on the exact type of management. Jandl et al. (2007) found that disturbances consistently enhanced mineralization. This effect might also explain the results of this study. The effect of management is demonstrated in the GLMs (§2.3.5), where management negatively affects the FFC stock.

Parts of the oak, larch and Douglas fir stands have been disturbed by foraging wild boars. As it is observed that F horizons sometimes are completely removed by wild boars, these soil disturbances are expected to enhance mineralization and are expected to influence forest floor structure and FFC stock. F1 and F2 thicknesses decrease by wild boar disturbance (mean difference F1 1.00 cm; $p<0.001$; mean difference F2 1.06 cm; $p<0.001$). In plots that are disturbed, FFC stocks are 14% lower than in undisturbed plots ($p=0.02$). While disturbance by tillage, harvesting or storm damage etc. is frequently mentioned in forest floor inventories, wild boar disturbance is generally not reported (De Vries and Leeters, 2001; Leeters and de Vries, 2001; LNV Directie Kennis, 2007). Since wild boar disturbance is common in northwest European forests, accounting for these effects is important given the lower carbon stocks.

2.4.3. Discussion of methods

The effects of tree species and management on FFC and SOC stocks are based on the measurements of bulk density and carbon contents and the design of the sampling scheme. The choice of methods may affect the results if samples are disturbed or bias is introduced due to the characteristics of the sampling scheme.
Due to difficulties in measuring bulk density in forest soils few data on bulk density are available in literature. In the forest floor, there is the risk of compaction during sampling, causing overestimation of bulk density. In mineral soil, measuring bulk density is mainly hampered in stony areas. With the monolith profile sampler, compaction of F and H horizons during sampling is limited. L material is more strongly compacted during sampling; therefore L horizons were sampled separately as described in §2.2.2.

The sampling method may have caused errors in FFC and SOC stocks because of inaccuracies during sampling or reading off data. Errors are probably larger than when sampling the forest floor as a whole. The sampling of forest floor horizons separately showed, however, that the horizons clearly differ in carbon content and bulk density (Table 2.5). These insights better reflect the effect of factors influencing forest floor characteristics at the landscape scale and therefore provide a better basis for upscaling and refinement of LULUCF carbon stock inventories.

Each of the ten plots within the six stands examined here can be considered pseudoreplicates. This could have led to increased type I error rates in the ANOVAs. Plots are however located as far away from each other as possible, reducing the risk of spatial correlation of forest floor characteristics and carbon contents. Short-distance spatial variation in forest floor characteristics and carbon contents is very high (Emmer, 1995; Smit, 1999), further reducing the risk of spatial correlation. Additionally, site variability of carbon stocks between plots within a stand was not much less than site variability between plots of different tree species (Table 2.7). From this result it is concluded that the ten plots within each stand can be considered as independent replicates.

2.4.4. Effect of future development in forestry on SOC stocks

Management trends in Europe are shifting to more closely mimic natural processes through management. This consists of implementation of single tree selective logging in a continuous cover forestry and a shift towards managing forests with a larger broadleaved species component (Nabuurs et al., 2001). The difference between unmanaged and managed forests in plots in the Speulder- and Sprielder forest does not exactly resemble the traditional management versus close to nature forestry. However, under close to nature forestry, a lower degree of management is often implied. From the unmanaged plots it thus can be hypothesised what a closer to nature forestry in Europe may mean for the FFC stock. There seems to be no saturation yet of carbon accumulation in the forest floor: the 170-year-old beech stand has significantly thicker forest floor horizons and larger FFC and SOC stocks than the 60-year-old beech stand. Further, the difference in C distribution between 170-year-old beech and 60-year-old stands could indicate that the development of a new SOC stock mainly takes place in the upper layer in the first decades after replanting, while, if the stand is undisturbed for longer periods, carbon in the mineral soil is transferred to deeper layers (Jenkinson, 1991; Rumpel et al., 2002).

If the trend in management towards more broadleaf forests continues, then first a reduction in carbon in the forest floors is expected, followed by a slower rate of carbon stock development. Chronosequence studies on conifer to broadleaf transformation (Fischer et al., 2002; Prietzel, 2004; Bens et al., 2006) show that carbon stock changes are found to differ mainly between parent materials; no clear trend within the chronosequence was found in these studies. In this study, differences in SOC stocks between conifers and broadleaf trees were found. As tree species and management regimes were representative for Dutch forests, the same differences may be expected in other forests at sand areas, but results cannot be translated directly to other parent materials.

2.5. Implications for inventories

In the present Dutch National Inventory Report for greenhouse gas emissions (NIR), an FFC stock of 37 ton C ha^{-1} is assumed (Nabuurs et al., 2005) and SOC and FFC stocks are assumed to be stable. To upgrade the NIR from Tier 1 to a higher Tier level, efforts are made to assess changes in FFC stocks using the Yasso model as part of the CO2FIX model (Liski et al., 2005; Van den Wyngaert et al., 2006). For this, FFC stock estimates are needed as input that properly reflect variability resulting from differences in tree species, management regimes, parent material and forest age.
Comparison of several case studies and national inventories showed that national-scale FFC stocks do not reflect local or regional FFC stocks. In this study, mean FFC stocks varied between 12.3 (60-year-old beech) and 51.1 (170-year-old beech) ton C ha^{-1}. For the most abundant age group (40-60 years) using an average FFC stock of 27 ton C ha^{-1} as found in this study would be appropriate. This is confirmed by the fact that forest floor thicknesses, as found in this study, are in agreement with data from the Dutch forest inventory. Using forest floor characteristics from the forest inventory, a more regionalized FFC stock could be calculated. Additionally, in many countries spatial information is available on the extent of conifer and broadleaf forest. As FFC stocks differ between tree groups and tree species, combined with forest floor characteristics and management, such data could be used to improve predictions of FFC stocks (Table 2.8). This distinction within soil-groundwater units could also bring more detail in national scale FFC stocks.

The Netherlands face an average annual afforestation of 3124 ha and deforestation of 2504 ha (Nabuurs et al., 2005). Based on the mean FFC stock as found in this study (27 ton C ha^{-1}), carbon loss from forest floors due to deforestation therefore could be as large as 68000 ton C $year^{-1}$. This is currently not included in the NIR and amply exceeds carbon accumulation in new forests floors.

The forest SOC stock in the Dutch NIR is based on an overlay of the map by Kuikman et al. (2003) and a land use map and is assumed to be 69 ton C ha^{-1} in the upper 30 cm of mineral soil. As well as for FFC stocks, several case studies and national inventories are notoriously different. In this study, mean SOC stocks varied between 53.3 (60-year-old beech) and 97.1 ton C ha^{-1} (larch). Distinguishing tree groups, tree species and management intensities could be used to bring more detail to SOC stock inventories.

The difference in carbon stocks under broadleaf stands compared to conifer stands is expected to be consistent along physiotopes. This difference is often found in other studies and can be explained from soil-vegetation processes. A lower carbon stock at managed locations than at unmanaged locations is also expected to be consistent among physiotopes because the processes causing carbon stock decrease are the same at different physiotopes. Nevertheless, FFC and SOC stocks can be very different on different parent materials and in areas with different nitrogen and heavy metal deposition levels. Research in other physiotopes is therefore still needed to obtain a complete picture of the forest SOC stock.

Chapter 3

Long-term landscape – land use interactions as explaining factor for landscape-scale soil organic matter variability

The present-day land use pattern is often used as a determinant for variability of soil organic matter stocks. Although the effect of historical land use on soil organic matter (SOM) stocks is recognized, this factor is never accounted for in SOM inventories. This chapter assesses if long-term landscape management and the interaction between landscape and land use can improve estimates of landscape-scale SOM variability for a case study in the northern Dutch sand area. Land use history (1780-2000) was reconstructed using topographic maps and land use databases. Potential determinants for SOM contents were analysed. Linear models were used to characterize SOM variability at 50m, 200m and 500m resolutions.

Soil characteristics can partly explain SOM variability. Present-day land use only explains 2% of SOM variability while historical land use patterns explained up to 20% of the SOM variability. At 50m resolution, SOM contents can be best explained with soil and land use history factors. At 200m resolution, soil and groundwater factors yield the best model while at 500m resolution a model including soil, groundwater and historical land use performs best.

Land use history data can significantly improve SOM inventories at multiple scales. As land use history of the Netherlands is relatively well documented, an improvement of national SOM stock estimates can be expected when accounting for the land use history.

Based on: C.J.E. Schulp and A. Veldkamp
Geoderma 146 (2008): 457-465

3.1. Introduction

The global soil organic carbon (SOC) stock is estimated to amount to 1200-1600 billion ton carbon in the upper meter (Batjes, 1999), similar to three times the carbon stock in vegetation or two times the amount of carbon in the atmosphere. Hence, sequestration of carbon in soils is often recognized as an option for mitigation of climate change. To be able to estimate the potential CO_2 uptake of the soil, information on the potential and actual SOC stock is needed.
Spatial distribution of SOC stocks is influenced by several factors. Climate and parent material characteristics define a range of SOC levels for an ecosystem (McLauchlan, 2006). At landscape scale, the actual SOC stock size is to a large extent determined by human impact on the landscape, e.g. land use and management. These determine the input and output of OM to the SOM stock. Therefore, often a direct link between land use and SOM content is assumed (Smith et al., 2000; Jarecki and Lal, 2003; Lettens et al., 2005a). SOM contents in croplands are generally lower than SOM contents in forests and grassland. Consequently, conversion of grassland or forest to cropland is assumed to cause a decrease of SOM content whereas the opposite conversions are assumed to increase SOM contents (Degryze et al., 2004; Lettens et al., 2004; Gerzabek et al., 2005).
At pedological timescales SOM is very dynamic (Richter Jr, 2007): SOM contents can change within decades to centuries. At temporal scales up to a few decades, sometimes clear SOM content changes are observed following land use conversions (Lettens et al., 2004; Falloon et al., 2006). In due time after conversion, the rate of SOM content change decreases until the SOM content reaches a new quasi-equilibrium (Freibauer et al., 2004). These changes can last centuries after land use conversion. Therefore, not only present-day human impact but also land use history up to decades (Pulleman et al., 2000; Sonneveld et al., 2002) to centuries (Springob et al., 2001) or millennia (Verheyen et al., 1999; Kristiansen, 2001) ago can be assumed to still have an effect on present-day SOM variability.

Many countries have only a limited amount of detail in their spatial information on size and quality of SOM stocks (Lindner and Karjalainen, 2007). In the Netherlands a country-wide SOM inventory based on 1:50.000 soil and groundwater maps (Kuikman et al., 2003) represents the current state of knowledge on spatial distribution of SOM. For a national-scale assessment, the relation between soil and groundwater and SOM stocks can represent SOM stocks correctly. At the landscape scale, other factors that influence SOM stocks might dominate over the effect of soil and groundwater (Veldkamp et al., 2001). Spatial variability of SOM stocks at landscape scale might be better represented by patterns of management (Dendoncker et al., 2004; Schulp et al., 2008b), land use or land use history.
Additional to SOM stock size, management, land use and land use history may influence SOM quality as well (Springob and Kirchmann, 2002). SOM stock size and quality together determine the potential amount and rate of carbon sequestration or emission of the landscape. Therefore, insight in variability of SOM quality and quantity is required to be able to quantify the greenhouse gas mitigation potential of the landscape.

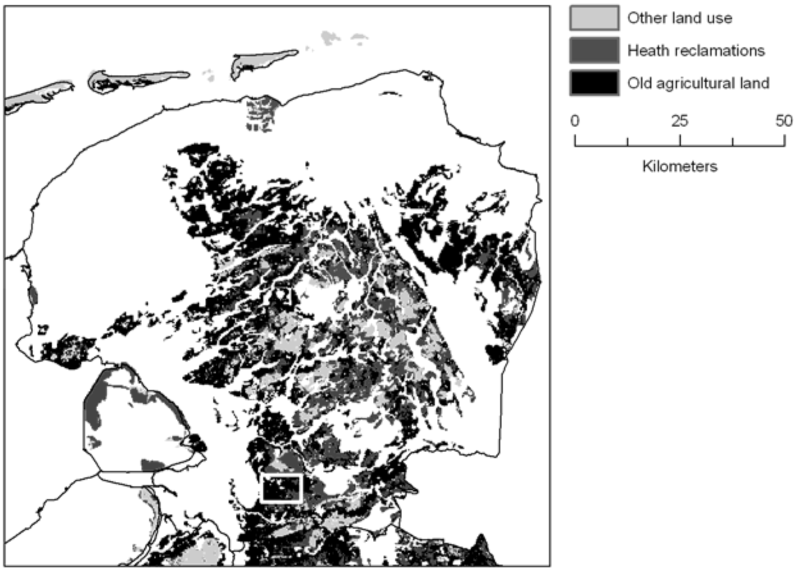

Fig. 3.1. Land use history of the northern Dutch sand area. The white rectangle indicates the case study area.

The effect of present-day land use on SOM stocks is often used for stratification or upscaling in inventories (Arrouays et al., 2001; Rodriguez-Murillo, 2001; Lettens et al., 2005a) or to predict SOC stock changes (Falloon et al., 2006; Schulp et al., 2008a). Although the effect of historical land use on SOM stocks is more and more recognized, this knowledge is as yet never used in SOM inventories.

This chapter assesses if an inventory of long-term landscape management can be used to improve insight in present-day spatial variability of SOM stocks in the Dutch sand area. This area is modified by humans for centuries and is dominated by man-made soils and recent heath reclamations. Soil, present-day and historical land use and their interactions are hypothesized to explain SOM variability at multiple scales.

3.2. Materials and methods

3.2.1. Study area

A case study area was selected that can be held representative for the northern sand area with respect to geology, land use history and soil formation (§1.4.2). The study area is a 60 km^2 area in the surroundings of Nieuwleusen (Fig. 3.1). Mean altitude is around 2 m above sea level. North and south of the area, two rivers drain the area. The study area is an agricultural landscape dominated by permanent grasslands (38% of the area) and rotations between grasslands and arable crops (40%).

Geology, soil formation and groundwater

The surroundings of the study area consist of eolian sand ridges deposited in the late Pleistocene. During the Holocene, lower parts of the area became covered by a thin peat layer with humid heath and sphagnum vegetation (Kuijer and Rosing, 1994). The study area is slightly sloping west and there is an eolian cover sand ridge from north-east to south-west through the area. The cover sand is well-sorted fine sand with loam contents (% particles <50 μm) varying between four and 20%. Soil formation is strongly affected by humans. In the earliest reclamations plaggen soils are found. The lower part of the area is mainly covered by gleyic podzols and humic gleysols (Kuijer and Rosing, 1994). Groundwater levels are homogenous.

Land use history

The area has been permanently populated since the early Middle Ages (500-1000 AD). The Nieuwleusen village was founded in 1635 on the cover sand ridge and settlers started to grow buckwheat in on the surrounding area, causing the thin peat layer to disappear completely. Until 1850, the highest part of the landscape was used for agriculture while the surrounding wastelands were used for grazing sheep and sod cutting. Large-scale reclamation lasted until the second half of the 19th century, as is representative for the Dutch sand area.

3.2.2. Data collection

A GIS database of the study area was constructed containing data on historical and present-day topography, soil and land use (Appendix Chapter 3-4). Actual soil data from a 1:10.000 soil mapping of the area were used (Scholten, 1996). The full dataset contains over 4000 points and has an average spatial resolution of approximately 50m. Mean thickness of the upper soil horizon is 33.4 cm. A soil map of the area was used that also includes groundwater classes.

Land use history was reconstructed using topographic maps from 1780 to 1955 and land use databases from 1960 to 1999. A 5m-resolution DEM of the area was used to quantify the landscape structure.

3.2.3. Land use history reconstruction

Topographic maps since 1780 with scales varying between 1:14.400 and 1:50.000 were used for detailed reconstruction of land use change trajectories (Appendix Chapter 3-4). Dutch historical maps with a scale more precise than 1:50.000 are accurate enough to be used for orientation and reconstructing trajectories of land use change (Koeman, 1963). Maps were scanned and georeferenced to the Rijksdriehoeksstelsel coordinate system. Land use in 1780 and 1850 was traced off from the maps. The 1780 and 1850 maps are known to be very accurate for their time (Donkersloot-de Vrij, 1981). Accuracy is however lower than for modern maps, especially in remote areas. Topographic maps from 1897 and afterwards were classified into seven classes: built-up areas, roads, water, grasslands, arable lands, heathlands and forests, using supervised maximum likelihood classification. Resulting maps had a resolution of approximately 3m. The 1780 map indicates two main types of agriculture: arable lands and buckwheat. The 1850 map sometimes poses difficulties in distinguishing the exact symbology and is therefore classified into mixed land use classes. In the 1930 map, the maximum likelihood classification had difficulties in distinguishing grassland and arable land. In all maps, heathlands and roads were difficult to distinguish. Classification errors were fixed by boundary clean functions and visual checking. Maps were aggregated to 10m resolution using a majority filter. The series of historical land use maps was combined to a reclamation age map.

3.2.4. Data analysis

A 'distance to first order water' map was made based on the 1850 topographic map because here the natural drainage system could be derived. Maps indicating built-up areas in 1780, 1850 and 1999 were made and distance maps to built-up areas were based on Euclidian distance. Using land use datasets from 1986 to 1999, the agricultural area was classified into permanent grasslands and land under rotation with arable crops and grassland.
For assessing determinants for SOM variability at different scales, the 50m base maps of potential determinants and of SOM content were aggregated to 200m and 500m resolution. Map data are extracted to the soil mapping data points. All GIS analyses were done in ArcGIS 9.2.

The structure of the variables was analysed with principle component analysis with varimax rotation. Potential explaining factors for SOM contents were analysed with Pearson's correlation and ANOVAs. When parametric assumptions were violated (tested with Kolmogorov-Smirnov and Levene tests), non-parametric tests were used (Spearman's correlation, Kruskal-Wallace test, Mann-Whitney test, Jonckheere-Terpstra test). To compare magnitudes of observed effects among variables, standardized effect sizes are used. For continuous data, the effect size is the correlation coefficient, for categorical data, the effect size is calculated as the square root of (ANOVA model sum of squares divided by the total sum of squares).

Analysis of determinants for SOM variability was done for 50m, 200m and 500m resolution following the multi-scale methodology as described by Kok and Veldkamp (2001). Spatial autocorrelation in SOM content data was calculated with Moran's I (Legendre and Legendre, 1998). Spatial autocorrelation was high, therefore random samples of the datasets were used: 25% (N=1063) for the 50m dataset; 50% (N=534) for the 200m resolution dataset and 80% (N=146) for the 500m resolution dataset.

Table 3.1. *Dummy variables used in linear models for predicting SOM variability.*

Code	Explanation
Soil classification	
cHn21	Sandy soils with low loam content and a 30-50 cm plaggen cover.
Hn21	Gleyic podzols with low loam content.
Zg23	Sandy soils with medium loam content and a humic topsoil.
Zn21	Sandy soils with low loam content and a humic topsoil.
Groundwater classes	
GWC[1] III	AHG[2] <40 cm below surface, ADG[3] 80-120 cm below surface.
GWC IV	AHG > 40 cm below surface, ADG 80-120 cm below surface.
GWC VI	AHG 40-80 cm below surface, ADG >120 cm below surface.
Historical and present-day land use	
Buckwheat_{1780}	Dummy variable indicating buckwheat areas (1) and other areas (0).
$\text{Arable}_{1780,\ 1850,\ 1900,\ 2000}$	Dummy variable indicating arable land (1) and other areas (0).
$\text{Wasteland}_{1780,\ 1850,\ 1900}$	Dummy variable indicating wasteland (1) and other areas (0).
Grassland_{2000}	Dummy variable indicating grassland (1) and other areas (0).

1 Groundwater class.
2 Average highest groundwater level.
3 Average deepest groundwater level.

Based on the determinants identified in the statistical analysis, linear models to explain SOM content variability were constructed for 50m, 200m and 500m resolution. For this, the datasets were split up into a model construction dataset and a validation dataset. Zero-one dummy variables were made for the main soil types, three groundwater classes and historical and present-day land use types (Table 3.1). Independent variables were selected in the 50m resolution model with forward conditional linear regression. Models were calculated using soil, groundwater and historical and present-day land use separately as independent variables, and using combinations of independent variables. In the 200m and 500m resolution models, the independent variables as selected in the 50m resolution model were used. Model performance was checked against the validation datasets by calculating Root Mean Square Errors (RMSE).
Maps of SOM content were made for the 50m, 200m and 500 m resolution dataset with the ArcGIS raster calculator, using the best performing model at different resolutions. Throughout the analyses, $p<0.05$ was used as significance level. All analyses were done with SPSS 15.0.

3.3. Results

3.3.1. Land use history

Area percentages of land use types in the study area between 1780 and 2000 are presented in Fig. 3.2. Most striking land use changes are the disappearance of wastelands until 1950 and the increase of grasslands. The absence of grassland in the 1780 is because grasslands cannot be distinguished on the map. Probably there are areas of grassland included in the arable land classes. Additionally, wastelands were used as common grounds for grazing.

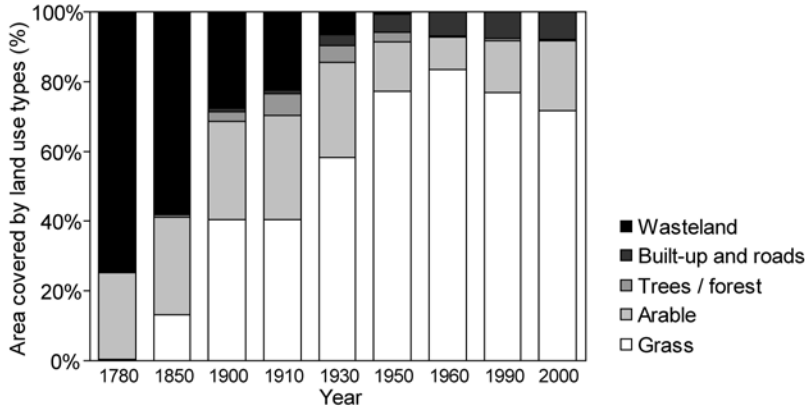

Fig. 3.2. *Land use change in the case study area between 1780 and 2000.*

The arable land area slightly increased until 1910 and decreased afterwards. After 1960, the area of arable land increased again, mainly because of expansion of the maize area after introduction of maize in the seventies.

With the expansion of agricultural area the tree area increased, probably because hedgerows were planted in new agricultural areas. The tree area reduced since 1930. Most likely, hedgerows were removed with land reconsolidations and increased mechanisation.
The reclamation age map (Fig. 3.3) shows that only the central and northern part of the area had been reclaimed before 1850. Since 1850, large scale reclamations were initiated for agricultural expansion, mainly in the south-west of the study area. The south-eastern part of the area remained wasteland up to the 1950s.

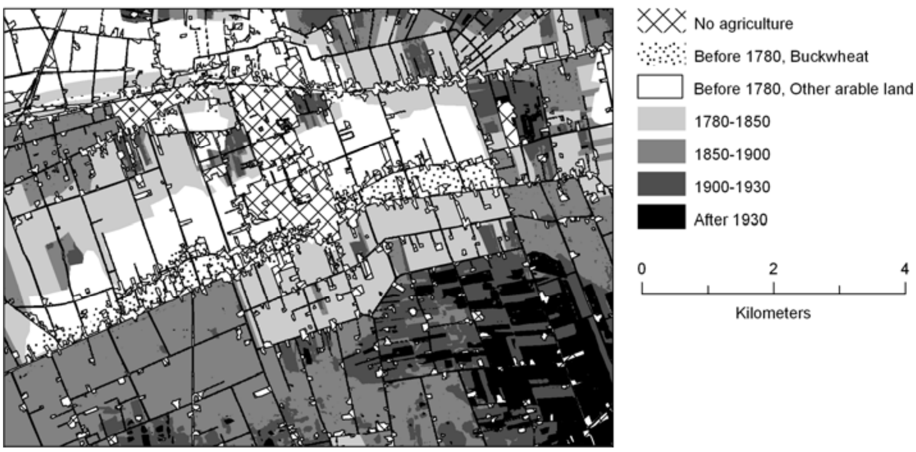

Fig. 3.3. *Reclamation age map of the study area.*

Table 3.2. *Factor loadings and communalities of variables in 50m resolution dataset. Only factor loadings with an absolute value >0.4 are shown.*

Independent variables	Factor			Commu-nalities
	Factor 1	Factor 2	Factor 3	
Median sand grain size (M50)	0.76			0.62
SOM content	0.54			0.40
Loam content	-0.89			0.81
Average highest groundwater level			0.80	0.74
Average deepest groundwater level			0.84	0.78
Reclamation age		0.71		0.65
Distance to built-up area in 1780		0.84		0.71
Distance to built-up area in 1850		0.89		0.80
Distance to built-up area in 1999		0.68		0.61
Distance to first order water	0.72			0.66
Altitude (200 m resolution)	-0.49		0.60	0.62

3.3.2. Principle component analysis – associations with SOM

Principle component analysis showed that many variables in the 50m resolution dataset are highly intercorrelated and can be subdivided in three independent factors (Table 3.2). The factors are not correlated (p=0.000) and together explain 66% of total variance. Factor 1 is interpreted as a soil-landscape factor with high factor loadings for soil variables and landscape structure variables (altitude at 200m resolution and distance to first order water). The landscape consists of two first-order rivers with a cover sand ridge in between. Closer to the rivers finer textured material is found (lower M50 and higher loam content). Table 3.2 suggests that high SOM contents are found at larger distance from first order water. Soil properties including SOM contents thus are strongly intercorrelated with landscape properties. This factor explains 25% of total variance.

The second factor is interpreted as a land use factor. Reclamation age has a positive loading on this factor, as well as the distances to built-up area in 1780, 1850 and 1900. Reclamation age is subdivided into classes ranging from 1 (before 1780) to 6 (after 1950). Young reclamations are found at a larger distance to built-up areas, indicating that areas close by are reclaimed first. Also the location of present-day built-up area is strongly influenced by historical occupation patterns. This factor explains 24% of the total variance.

The third factor is interpreted as a groundwater factor. Altitude has a high positive loading on this factor, indicating that groundwater levels are deeper in higher areas. This factor explains 21% of total variance in the dataset. Other variables as shown in Appendix Chapter 3-4 hardly had common variance.

Table 3.3. *Explained percentage of variance in SOM content and significance level by several variables.*

Independent variables	R^2	Sign.	Independent variables	R^2	Sign.
Loam content	0.25	0.00	Groundwater class	0.10	0.00
M50	0.20	0.00	Land use in 1780	0.14	0.00
Altitude	0.01	0.00	Land use in 1850	0.20	0.00
Slope	0.01	0.00	Land use in 1900	0.04	0.00
Distance to first order water	0.20	0.00	Land use in 2000	0.02	0.02
Distance to built-up area in 1780	0.01	0.00	Reclamation age	0.13	0.00
Distance to built-up area in 1850	0.03	0.00	Permanent grassland	0.00	0.64
Distance to built-up area in 1999	0.06	0.00	Soil type	0.23	0.00
Distance to trenches	0.03	0.00			

3.3.3. Determinants for SOM content

50m resolution

Loam content (R^2=0.25), M50 (R^2=0.20) or soil classification (R^2=0.23) can explain the greatest part of SOM variability among the single determinants (Table 3.3). Present-day land use explains only 2% of SOM variability. Permanent grasslands and land under rotation do not differ in SOM contents. Historical land use patterns have more potential for explaining SOM variability than present-day land use: Land use patterns in 1780 explain 14% and land use patterns in 1850 explain 20% of present-day SOM variability (Table 3.3).

The reclamation age classes (Fig. 3.3) explain 13% of SOM variability (Table 3.3). SOM contents are highest in the oldest reclamations and decrease with decreasing age of reclamation (Fig. 3.4a; J-T statistic$_{std.}$=-8.4; p=0.00). When distinguishing buckwheat areas and other arable land in reclamations before 1780, buckwheat areas have the highest SOM contents (9.6%), followed by other arable lands and areas reclaimed between 1780 and 1850 (6.9% and 7.0% respectively; $p_{Mann\text{-}Whitney}$=0.00). SOM content of

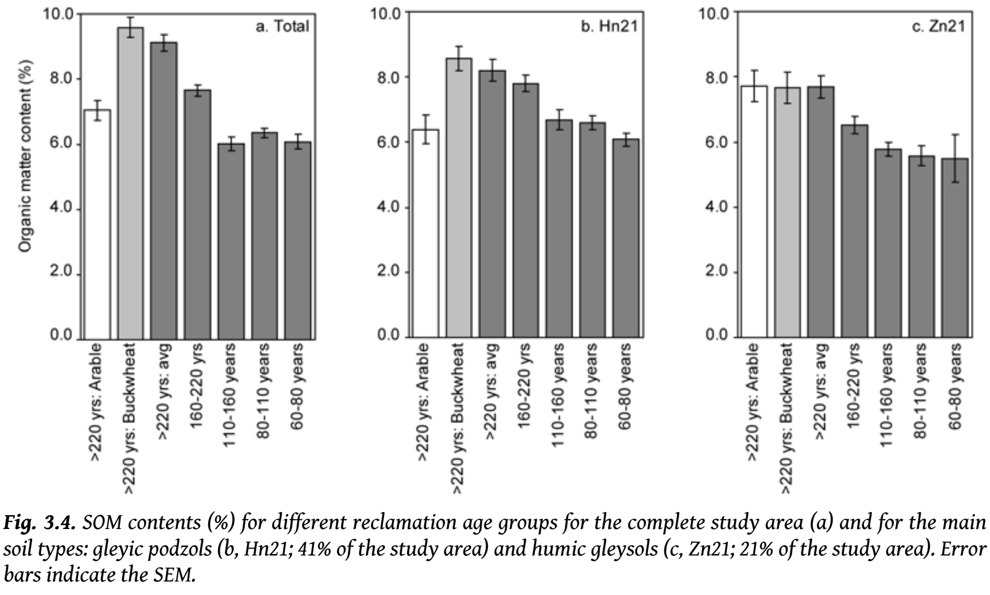

Fig. 3.4. *SOM contents (%) for different reclamation age groups for the complete study area (a) and for the main soil types: gleyic podzols (b, Hn21; 41% of the study area) and humic gleysols (c, Zn21; 21% of the study area). Error bars indicate the SEM.*

other arable land reclaimed before 1780 does not differ from SOM contents of reclamations between 1780 and 1850 ($p_{Mann\text{-}Whitney}$=0.15). Reclamations past 1850 have lower SOM contents than other arable lands reclaimed before 1780 and reclamations between 1780 and 1850 (6.1%; $p_{Mann\text{-}Whitney}$ = 0.00). Also within soil classification units this pattern is seen (Fig. 3.4b and c). Within both dominant soil types in the study area, several reclamation age groups have different mean SOM contents (p_{ANOVA}=0.00; R^2=0.11 for gleyic podzols (Hn21) and p_{ANOVA}=0.00; R^2=0.14 for humic gleysols (Zn21) (Table 3.1)). Apart from soil and historical land use, SOM variability can partly be explained by groundwater class (R^2=0.10) and by distance to 1st order water (Table 3.3). Altitude, slope and distance to present-day and historical occupied areas only explain small amounts of SOM variability. Distance to trenches is correlated weakly but significantly with SOM contents.

Scale resolution effects
At 200m and 500m resolution, several determinants for SOM variability show changing effect sizes (Table 3.5). Spatial autocorrelation increases with decreasing resolution. Moran's I increases from 0.11 in the 50m resolution dataset to 0.16 in the 500m resolution dataset. As a result, effect sizes generally increase with decreasing resolution, a commonly observed effect in spatial multi-scale models (Overmars et al., 2003). Nevertheless, effect sizes for land use patterns in 1780 and 1850 hardly increase and effect size for land use patterns in 1900 decreases. Present-day land use (Land use 2000 and Permanent grassland, Table 3.5) shows an increased effect size that cannot be attributed solely to increasing spatial autocorrelation. Soil classification and groundwater class clearly increase in effect size. Loam content decreases in effect size and the M50 changes sign with decreasing resolution (Table 3.5).

Table 3.5. *Effect sizes of determinants for SOM content at different resolutions.*

Determinants	Resolution		
	50m	200m	500m
Loam content	-0.50	-0.44	-0.29
M50	-0.45	0.29	0.26
Altitude	-0.10	0.05 ns	0.20
Slope	0.10	0.27	0.41
Distance to trench	-0.17	-0.06 ns	ns
Distance to first order water	0.45	0.60	0.71
Distance to built-up area in 1780	-0.09	-0.18	-0.28
Distance to built-up area in 1850	-0.19	-0.29	-0.10
Distance to built-up area in 1999	-0.24	0.13	0.07 ns
Groundwater class	0.31	0.46	0.55
Reclamation age	0.37	0.47	0.56
Land use in 1780	0.37	0.44	0.38
Land use in 1850	0.45	0.49	0.53
Land use in 1900	0.20	0.15	0.16 ns
Land use in 2000	0.15	0.50	0.75
Permanent grassland	0.04 ns	0.19	0.25
Soil type	0.48	0.65	0.80

ns Effect is not significant at p<0.05.

Altitude has a weak negative effect on SOM content at 50m resolution, changing to a medium positive effect at 500m resolution. At 50m resolution, this is because in trenches and former ditches SOM contents are higher, as shown by the negative correlation with distance to trenches (Table 3.5). At coarser resolution, altitude reflects the landscape structure. SOM contents are high on the cover sand ridge and low in the lower areas due to the reclamation history, resulting in positive correlation at coarser resolutions.

3.3.4. Improving SOM inventories using land use history information

Spatial variability of SOM contents was assessed at 50m, 200m and 500m resolution using soil, groundwater and present-day land use, because these factors are commonly used in inventories. Additionally, the suitability of historical land use patterns for predicting spatial variability of SOM contents was assessed.
At 50m resolution, SOM content can be predicted with an RMSE of 2.32 (R^2=0.23) using soil types:

$$SOM\% = 7.22 + (cHn21*3.57) + (pZg23*-1.59) + (pZn21*-0.73) \qquad (eq.\ 3.1)$$

In which cHn21, pZg23 and pZn21 are zero / one variables indicating if at a certain location a certain soil type (Table 3.1) is present (1) or not (0). Including groundwater classes did not improve the model (Table 3.6).

Using historical land use, the best model (RMSE=2.28; R^2=0.21) to predict SOM content is:

$$SOM\% = 6.84 + (Arable_{1850} * 2.26) + (Buckwheat_{1780} * 1.31) + (Wasteland_{1850} * -0.70) \qquad (eq.\ 3.2)$$

A linear model using present-day land use as independent variables resulted in an R^2 of 0.000.
Adding land use history variables (Table 3.1) to the soil model (eq. 3.1) significantly improved the model (RMSE=2.19; R^2=0.28) and resulted in:

$$SOM\% = 7.32 + (2.47*cHn21) + (-0.76*pZn21) + (-1.13*pZg23) + (1.17*Buckwheat_{1780}) + (0.54*Arable_{1850}) + (-0.76*Wasteland_{1850}). \qquad (eq.\ 3.3)$$

The predicted spatial distribution of SOM content is shown in Fig. 3.5a.

Upon decreasing resolution, the R^2 for the soil model, groundwater model and the present-day land use model increase while the R^2 for the historical land use model decreases (Table 3.6). The R^2 increases are partly an overrating due to increasing spatial autocorrelation and averaging out variability at coarser resolution. The R^2 increase for groundwater and present-day land use is however too large to be solely attributed to these effects.

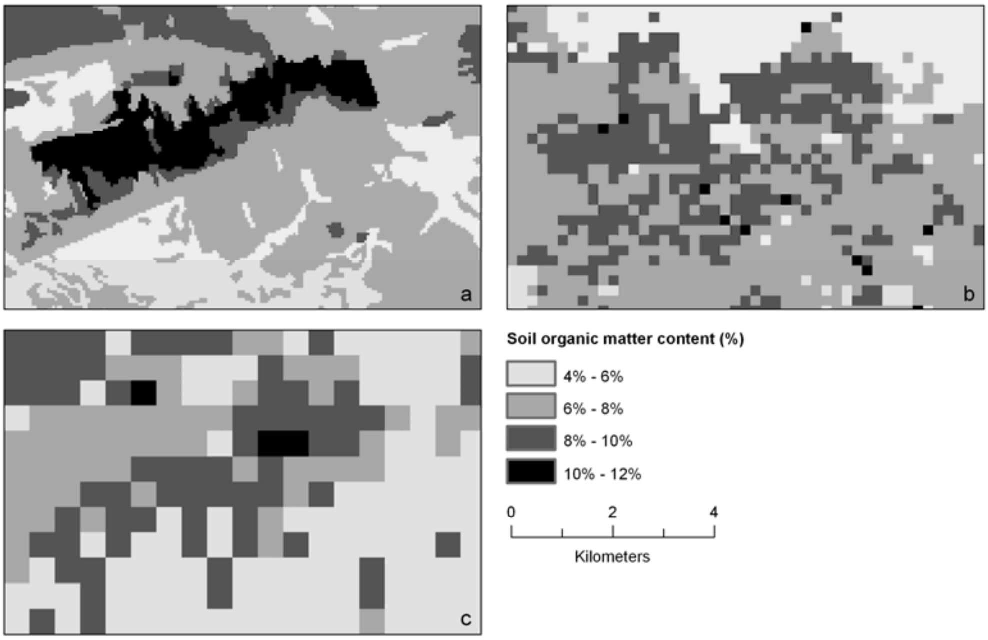

Fig. 3.5. *Maps of the spatial distribution of SOM content in the study area at (a) 50m resolution, (b) 200m resolution and (c) 500m resolution.*

At 200m resolution, SOM contents can be explained best with soil and groundwater (R^2=0.42, Table 3.6). Historical land use (R^2=0.32) is more important for explaining SOM variability than present-day land use (R^2=0.23). The predicted spatial distribution of SOM contents using the soil-groundwater model is shown in Fig. 3.5b. At 500m resolution, present-day land use can better explain SOM variability than historical land use. A model using soil, groundwater and historical land use yields the highest R^2 (0.75). The predicted spatial distribution of SOM contents using the soil-groundwater-land use history model is shown in Fig. 3.5c.

Higher SOM contents on the cover sand ridge, related to old reclamations, can be observed at all resolutions. In the 200m resolution map, predicted SOM contents are low in the north of the study area whereas in the 50m and 500m resolution maps the south has lowest SOM contents. Tested against the 50m resolution validation dataset, the 50m modelled map has a RMSE of 2.7. RMSE of the best-performing 200m resolution model is 3.5 and RMSE of the best-performing 500m model is 2.8.

Table 3.6. *R^2 and RMSE for predicting SOM contents at different resolutions.*

Independent variables	R^2			RMSE		
	Resolution			*Resolution*		
	50m	200m	500m	50m	200m	500m
Soil types	0.23	0.25	0.32	2.32	2.45	2.18
Groundwater classes	0.05	0.35	0.62	n.d.	2.56	2.29
Historical land use	0.21	0.32	0.14	2.28	2.46	1.92
Present-day land use	0.00	0.23	0.57	n.d.	2.68	2.09
Soil-Groundwater	n.d.	0.42	0.70	n.d.	2.43	1.80
Soil- Historical land use	0.28	0.39	n.d.	2.19	2.24	n.d.
Soil- Present-day land use	n.d.	0.38	0.70	n.d.	2.49	1.89
Soil-Groundwater-Historical land use	n.d.	0.39	0.75	n.d.	2.27	1.46
Soil-Groundwater-Present-day land use	n.d.	n.d.	0.73	n.d.	n.d.	1.85

n.d Not determined because regression model was not significant or did not significantly improve relative to a simpler model.

3.4. Discussion

3.4.1. Effects of landscape-land use interactions on SOM variability

Landscape characteristics are important for explaining SOM contents: SOM contents generally increase with an increasing percentage of fine particles (McLauchlan, 2006) and wet conditions have a restraining effect on SOM mineralization rates. Therefore, wet areas generally have higher SOM contents than dry areas. In this study opposite relationships were found: loam content is negatively correlated with SOM content and areas with deeper groundwater levels (higher class number) have higher SOM contents (Table 3.5). M50 is negatively correlated with SOM contents at 50m resolution, indicating higher SOM contents with smaller grain size, as expected, but at 200m and 500m resolutions smaller grain sizes are related to lower SOM contents (Table 3.5).

These contradictory relations between SOM contents and explaining factors probably are caused by interactions between landscape and land use. Humans are often recognized to be an important factor in formation of landscape and soils (soilscapes) and are sometimes assumed to be the most important landscape and soil engineer (Pennock and Veldkamp, 2006; Richter Jr, 2007; Zalasiewicz et al., 2008). Also in the Dutch sand area there is a clear link between landscape and long-term human influence. The earliest patterns of occupation and agriculture reflect the landscape suitability. Originally, the landscape in the study area was wet and only the driest parts were suitable for occupation and agriculture (central ridge in Fig. 3.3). The traditional heathland farming in the study area resulted in accumulation of SOM on the cover sand ridge by manuring, but caused depletion of nutrients in heathlands north and south in the study area by sod cutting (Webb, 1998). Additionally, the plaggen manure is assumed to contain relatively inert SOM (Springob et al., 2001; Springob and Kirchmann, 2002; Heumann et al., 2003; Dercon et al., 2005). SOM in ancient arable lands therefore decomposes slower than would be expected in arable land. This

difference in SOM content between old arable lands and heathlands persisted after reclamation and explains the present-day difference.
The parts of the landscape originally suitable for agriculture within the topographic and hydrological constraints are the areas with coarser sand, lower loam content and deeper groundwater. In this way, the land use history overrules soil factors as determinant for SOM contents.

Historical land use is especially suitable for explaining SOM contents at fine resolution while the relevance of present-day land use increases with decreasing resolution (Table 3.5, Table 3.6). Probably this is because historical land use reflects biophysical characteristics of the landscape taking into account local variation in suitability. At coarser resolutions, the impact of biophysical variation of the landscape on historical land use patterns is averaged out, decreasing the explaining value of historical land use. Present-day agriculture has a larger scale than historical agriculture and is less dependent on soil and landscape characteristics due to improved drainage, fertilization and mechanization. Therefore, present-day land use reflects broad, large-scale differences in biophysical characteristics. For inventories at national scale with a coarse spatial resolution, soil, groundwater and present-day land use thus might be able to explain SOM variability. At smaller scales, taking into account effects of historical land use on SOM variability is more relevant.

3.4.2. Applications

Improving SOM inventories
Information on land use history can improve models for predicting SOM contents and could therefore be relevant for improving insight in spatial variability of SOM relative to the current Dutch inventory.
Land use history is partly accounted for in the Dutch soil classification system by including several types of plaggen soils. As the soil map serves as a basis for stratification in the current Dutch SOC inventory, land use history is partly accounted for. However, soil classification leaves a large amount of SOM variability unexplained and land use history can be used to explain part of this variability (Fig. 3.4, §3.3.4). As an example, soils with a plaggen cover thinner than 30 cm are not recognized as such on soil maps, but differ in SOM content and composition from soils with the same classification but without plaggen cover. Using land use history in SOM inventories can help to include such variability.

Land use history has the highest explaining value for SOM variability at 50m resolution, decreasing at 200m and 500m resolution. Present-day land use shows an opposite trend. The National Reporting System for the United Framework Convention on Climate Change (UNFCCC) requires insight in SOC stocks at a 100m resolution (Nabuurs et al., 2004). At such a high resolution, historical land use proved to be better for improving insight in spatial variability of SOC contents than present-day land use.

Land use and management influence both amount and turnover rates of OM input. Therefore, historical land use patterns are expected to influence present-day chemical composition of the SOM stock. The plaggen system caused development of inert SOM with high C:N ratios (Springob and Kirchmann, 2002), while present-day management with high fertilizer input results in easily decomposable SOM with low C:N ratios. Therefore, no standard C content of SOM can be applied when calculating SOC stocks from SOM stocks in agricultural landscapes with a varied land use history. SOM quality controls decomposition rates of SOM upon land use conversion or due to heavy soil tillage. Therefore, the historical land use pattern is assumed to cause spatial variation in the mitigation potential of the landscape.

Upscaling
The land use history in the case study area resulting in development of plaggen soils with subsequent reclamation of heathlands is common in north-west Europe and is described among others in Denmark, Germany, Scotland and Belgium (Blume and Leinweber, 2004). General land use history trends and resulting SOM content trends, like high SOM contents in old arable soils that faced plaggen management can be assumed applicable in large parts of these areas. High SOM contents in anthropogenic soils are also found in among others Amazonia (Lima et al., 2002), China (Lindert et al., 1996) and Russia (Blume and Leinweber, 2004). A trend of SOM accumulation in intensively long-term used soils thus may be generally applicable.

The northern Dutch sand area is a relatively flat landscape with infertile eolian cover sand ridges and originally wet conditions. The geomorphology of the landscape results in a specific historical spatial arrangement of occupation and agriculture (§3.4.1). Although general trends of land use history and resulting trends in SOM contents are assumed to be applicable all over this area, interactions between landscape and land use result in specific spatial and temporal differences in SOM contents that cannot be applied one-to-one in other sand landscapes. Although land use history over the past 200 years all over the Dutch sand area is dominated by heath reclamations, different regions differ in geomorphology and parent material, resulting in spatial and temporal differences in reclamation history and SOM variability. Additionally, demographic developments over the past 200 years clearly differ between regions (Statistics Netherlands, 2007). SOM content differences within the landscape between areas with different land use histories as found in this case study are therefore only representative for the northern Dutch cover sand area. For a complete picture of the effect of land use history on SOM contents in the Dutch or north-west European sand area more research in different regions would be required.

In north-west Europe, detailed topographical mappings including mapping of land use have been done since centuries and information on historical land use is often available through cadastral information (Verheyen et al., 1999; Caspersen and Fritzboger, 2002; Knol et al., 2004; Bender et al., 2005; Hamre et al., 2007). As land use history, especially at smaller scales, is found to be important for explaining variability in SOM contents, also in other countries with a long history of detailed mapping data on land use history might be interesting for improvement of SOC inventories.

3.5. Conclusions

This chapter assessed how long-term landscape-land use interaction can be used to explain SOM variability in a case study area that can be held representative for the northern Dutch sand area with respect to geology, soil and land use history. Historical land use, up to 230 years ago, influences present-day SOM content variability. Historical land use can explain up to 20% of SOM variability and a reconstruction of reclamation history can explain 13%. At landscape scale, historical land use is more important for explaining SOM variability than present-day land use which can explain only 2% of SOM variability in the case study. Partly this is a result of clear interaction between the biophysical landscape and the historical land use pattern; partly historical land use is independent of the biophysical characteristics and therefore can be used to refine spatial information on SOM variability.

At 50m, 200m and 500m resolution, different determinants are important for explaining SOM variability. At 50m resolution, soil and land use history together are best for explaining SOM variability. At 200m resolution a model using soil and groundwater yields the best result while at 500m resolution, a model using soil, groundwater and historical land use has the highest fit.
In the case study area, topography and hydrology explain the historical land use pattern. The land use pattern in turn influences SOM variability. As a result, the landscape-land use interaction can overrule the effect of biophysical characteristics on SOM stocks as they are generally assumed and applied in predicting SOM variability.
Differences in explaining factors for SOM variability at different resolutions should be considered in SOM inventories. Currently, SOM inventories only consider soil, land use and groundwater for stratification or prediction. For low-resolution national-scale inventories this might represent SOM stocks correctly. At smaller scales, land use history is proven to be important for explaining SOM variability.

In large parts of north-west Europe, land use history is documented very well since 1850 and in the Netherlands, data on land use history since 1900 are available in GIS. Therefore, land use history data are expected to be an interesting factor for improving insight in spatial variability of SOM stocks in large parts of north-west Europe. For that, more specific insight is needed in how landscape and human influence interact in other regions of north-west Europe.

Chapter 4

Effect of land use history and site factors on spatial variation of soil organic carbon across a physiographic region

Chapter 3 showed that the land use history was important for explaining soil organic matter variability in a case study area. Small case studies however are often not representative for a complete region, and consequently, case study results have to be treated with great care when upscaling them to regional scale. To provide a better basis for upscaling, the representativeness should be increased and insight in the processes underlying the importance of land use history should be strengthened.

This chapter assesses how land use history and several site factors are associated with SOC variability in four sites across the Dutch sand area. Determinants for SOC variability were identified with empirical analysis. Causality of the associations was tested with a process model.

Empirical analysis showed that historical land use has a stronger association with SOC variability than present-day land use, because long-term agricultural areas have higher SOC contents than recent reclamations. The results of the process modelling indicate that land use systems with high carbon input accumulate a 1.5 times larger SOC stock than low-input land use systems after 200 years. Significant differences between high-input and low-input systems only emerge after several decades and are preserved for over 100 years after conversion. Due to the long-lasting imprint of historical land use systems combined with the high spatial variability of historical land use, current variability of SOC stocks is related to the land use history.

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4.1. Introduction

Recently, several studies have assessed determinants for soil organic carbon (SOC) variability at the global scale (Batjes, 1996), continental scale (Jones et al., 2005) or national scale (Arrouays et al., 2001; Rodriguez-Murillo, 2001; Lettens et al., 2004; Bradley et al., 2005). Reason for the growing interest in SOC stock inventories is the notion that the SOC stock is an important pool in the global carbon cycle. Land management is among the important determinants for SOC stocks, because land management influences the input and output of OM to the soil (Guo and Gifford, 2002). Sequestration of carbon in soils through land management is therefore often seen as a method for climate change mitigation (Lal, 2003).

To evaluate the potential sequestration of carbon in the soil, insight in the size of the SOC stock and its determinants is needed. It is common to base a SOC inventory on a soil map or survey combined with a land use map. For each combination of soil type and land use type, a mean SOC stock is calculated. This value is assumed to be representative for all mapping units with the same soil-land use combination (e.g. Lettens et al., (2004)). This approach allows identifying large-scale patterns in SOC stock variability related to soil characteristics and land use. Owing to the attribution of all the spatial variation in SOC stocks to soil and land use alone, uncertainty is often large with coefficients of variation between 32% and 165% (Arrouays et al., 2001; Lettens et al., 2004).
Soil and land use are however not the only determinants for SOC stock variability. A second approach for SOC stock inventories is the use of a wider range of determinants. Key determinants for SOC variability have been determined through empirical analysis of observed data in several studies at different scales (Mueller and Pierce, 2003; Tan et al., 2004). Generally, parent material and climate are assumed to define large-scale patterns of SOC stocks (McLauchlan, 2006), while other determinants, like topography, land use and management, define the SOC stock variation at smaller scales (Dendoncker et al., 2004; Sleutel et al., 2007; Schulp et al., 2008b). Because of the slow turnover of SOC, the effect of land use change on SOC stocks is slow. Therefore, also past land use, up to decennia ago, can influence SOC stocks (Verheyen et al., 1999; Sonneveld et al., 2002; Hupy and Schaetzl, 2008).

For a case study in the north of the Netherlands, land use history was found to explain soil organic matter (SOM) variability (Schulp and Veldkamp, 2008). Areas that have been used for agriculture since the 18th century have higher SOM contents than young heath reclamations. However, due to regional differences in land use history one case study cannot represent the complete country and therefore upscaling to a larger area is not possible. The same applies to most studies that use local datasets to quantify the determinants for SOC variability.

This chapter quantifies the effect of land use history and site factors on SOC variability in four sites representing variation in geological, historical and management conditions in the Dutch sand area. Based on detailed soil surveys of the sites, empirical relations between determinants and SOC variability were quantified. Quantification of determinants was tested with a process-based model to assess the causality of the empirical results. Finally, the implications of the findings for upscaling SOC contents to a complete physiographic region are discussed.

4.2. Methods: Data collection and editing

To assess determinants for SOC variability across a wide range of variation of site factors and land use histories, in each sub-region of the Dutch sand area (Fig. 1.1b) one site was selected where a detailed soil mapping with approximately one georeferenced augering per hectare was available (Fig. 4.1, Table 4.1). During the soil mappings, OM content, clay content and loam content of each horizon were estimated at each location. For part of the locations, samples were taken and analysed. Estimates and analysis results were strongly correlated (De Groot et al., 2005). 0-30 cm SOC stocks were calculated using a carbon content in OM of 58% (Kuikman et al., 2003) and a bulk density of 1300 kg m^{-3} (Finke et al., 2002). Each site is considered representative for one sub-region of the Dutch sand area (§1.4.2).

A GIS database for the entire physiographic region was constructed with data on biophysical characteristics, historical and present-day land use and agricultural management (Appendix Chapter 3-4). A reconstruction of land use in each site in 1850 was based on the topographical map from 1850, distinguishing meadows, arable land, forest and heathlands. Based on the 1850 land use reconstruction and historical land use datasets (Appendix Chapter 3-4), a reclamation age map was constructed for each site. A map indicating permanent grasslands and land under rotation was made based on recent land use datasets (Appendix Chapter 3-4). To assess the impact of manure and crop inputs on SOC contents, effective organic carbon (OC_{eff}) inputs (Sleutel et al., 2007) were calculated at zip code scale using farm-scale livestock numbers (Naeff, 2006) and crop areas from the 1999 land use map.

Table 4.1. *Population densities of sub-regions of the Dutch sand areas and characteristics of representative sites.*

Population density (persons km^{-2}); *year, source*	**Region**			
	North	**East**	**Central**	**South**
1500 (Spek, 2004)	8	12	17	27
1800 (Statistics Netherlands, 2007)	33	41	43	44
2000 (Statistics Netherlands, 2007)	195	320	342	418
Site characteristics	**Site name**			
	Nieuwleusen	**Achterhoek**	**Veluwe**	**Den Bosch**
Area (km^2)	60	150	200	245
Altitude range (m above sea level)	0-9	14-36	2-85	2-28
Number of samples	1061	1896	3002	1159
Dominant soil type (FAO)	Gleyic Podzol	Gleyic Podzol	Humic Gleysol	Fimic Anthrosol
Dominant groundwater regime	Intermediate	Shallow	Very deep	Deep
Dominant reclamation age group	<1850	1900-1938	<1850	<1850
Average lutum content (%)	2.6	10.8	12.3	5.2

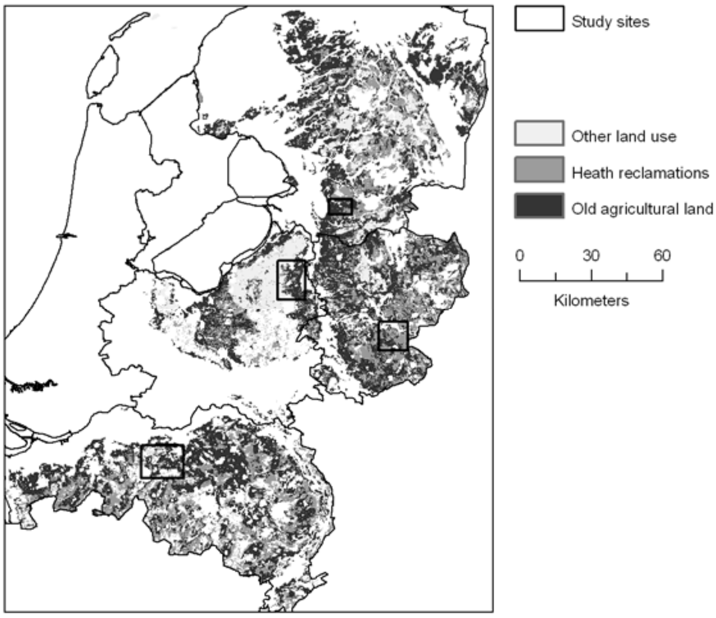

Fig. 4.1. *Locations of the study sites within the agricultural sand area.*

4.3. Methods: Analysis of determinants for SOC variability

Determinants for SOC variability were first identified by empirical analysis of observed data (§4.3.1). Second, the causality of the identified associations was tested with a process-based model (§4.3.2). The results from both methods were compared and the robustness of the results was checked with a sensitivity analysis (§4.3.3).

4.3.1. Empirical modelling

For each augering location all site conditions were extracted from the GIS database. Random samples (25%) from these datasets were used for the empirical analyses to decrease the potential influence of spatial autocorrelation. The effects of site factors, historical and present-day land use and management on SOC variability were analyzed with ANOVA and Pearson's correlations. When parametric assumptions were violated, non-parametric tests were used.

Multivariate models explaining SOC variability using site factors, land use and land use history were estimated. Main soil types, groundwater classes, geomorphology classes, reclamation types and historical and present-day land use types were treated as categorical variables (Table 4.2). First, independent variables were selected per group of variables with forward conditional linear regression. Then, models were constructed

Table 4.2. *Variables for multivariate modelling.*

Code	Explanation
Soil type	
cHn21	Fine sand gleyic podzols with low loam content and a 30-50 cm anthropogenic topsoil.
zEZ23	Loamy fine sand fimic anthrosols with a black anthropogenic topsoil thicker than 50 cm.
pZg23	Loamy fine sand humic gleysol with thin humous topsoil.
Hn21	Fine sand gleyic podzol with low loam content.
pZn23	Loamy fine sand humic gleysol with thick humous topsoil.
Groundwater class (GWC)	
II	AHG* <40 cm below surface, ADG* 40-80 cm below surface.
III	AHG <40 cm below surface, ADG 80-120 cm below surface.
V	AHG <40 cm below surface, ADG >120 cm below surface.
VI	AHG 40-80 cm below surface, ADG >120 cm below surface.
VII	AHG >80 cm below surface, ADG >160 cm below surface.
Geomorphology	
Plateaux, plateau-like forms, rolling areas, low hills, ridges, rolling areas and flats in between, flats, shallow valleys, medium valleys.	
Other site factors	
Loam content (%), elevation, median sand grain size.	
Reclamation types	
Small-scale and large-scale old agricultural reclamations, afforestations on heath, former peat reclamations, old forests.	
Land use 1900	
Pasture, arable land, heathland, deciduous forest, coniferous forest, low meadows.	
Land use and management 2000	
Grassland, maize, potatoes, cereals, other agriculture (From LGN4 map).	
OC_{eff} crops, OC_{eff} livestock: Effective organic carbon input (ton) (Sleutel et al., 2007) per zip code region.	

* AHG: Average highest groundwater level.
ADG: Average deepest groundwater level.

using soil and groundwater factors and using all site factors, respectively. Finally, models were constructed that combined site factors and historical land use, and site factors and present-day land use and management. The variance inflation factor (VIF) was checked in all models to detect multicollinearity. Variables causing multicollinearity were excluded from the models. All empirical analyses were done both site-specific and over all sites. Throughout the analyses, $p<0.05$ was used as significance level.

4.3.2. Process modelling

Several process-based models are widely used for simulating changes in SOC stocks for a wide range of environments and land use conditions, in particular RothC (Coleman et al., 1997) and Century (Parton et al., 1988). Both models have different input requirements in view of differences in underlying model assumptions, often resulting in different projections for similar scenarios (Smith et al., 1997; Falloon and Smith, 2002). RothC was selected here because it is most compatible with the available data. In this section, parameterization of the RothC model is described.

Table 4.3. Ranges of mean monthly temperature (°C) and precipitation sum (mm) in each site, and the weather stations used to compile the data.

Site	Time period					Weather station
	before 1875	1876-1930	1931-1950	1951-1975	1976-2005	
Range of mean monthly temperature (°C)						
Nieuwleusen	0.8-16.8	1.7-17.4	1.0-16.9	1.3-16.0	2.1-16.8	De Bilt, Groningen
Achterhoek	0.8-16.8	1.8-16.7	1.5-17.2	2.1-16.5	2.8-17.7	De Bilt
Veluwe	0.8-16.8	1.8-16.7	1.5-17.2	2.1-16.5	2.8-17.7	De Bilt
Den Bosch	0.8-16.8	1.8-16.7	1.5-17.2	2.1-16.9	2.8-17.9	De Bilt, Eindhoven
Range of monthly precipitation sums (mm)						
Nieuwleusen	34-83	39-88	38-86	45-93	46-86	Groningen
Achterhoek	42-82	41-87	46-81	50-89	46-80	Utrecht, Winterswijk
Veluwe	42-82	44-87	42-91	49-86	44-81	Utrecht, Heerde
Den Bosch	36-65	40-83	44-74	48-94	45-86	Maastricht, Oudenbosch

Source (Royal Netherlands Meteorological Institute, 2008)

Soil and weather data

Topsoils were slightly thicker than 30 cm in all sites. A topsoil thickness of 30 cm was used in all sites for consistency. Mean lutum content was calculated for each site from the soil data (Appendix Chapter 3-4). Mean monthly temperature (°C) and precipitation sums (mm) were calculated from national weather station data (Royal Netherlands Meteorological Institute, 2008) (Table 4.3). Monthly open pan evaporation was calculated from data by Muller (1982).

Land use trajectories

In each site, SOC dynamics were simulated for three land use trajectories that reflect historical developments in the Dutch sand area:

- Long-term agriculture: areas that are used for agriculture since the first half of the 19th century or longer;
- Old reclamations: areas reclaimed for agriculture around 1875;
- Young reclamations: areas reclaimed for agriculture around 1930.

In the Achterhoek site, no old reclamations were found. Two types of young reclamations (on dry heath and on wet heath) were simulated in this site. The age of the first reclamations differed between sites (Fig. 4.2), consistent with differences in the development of land use intensity across the Dutch sand area. Land use history was based on narratives on historical geography for the Netherlands (Spek, 2004), completed with site-specific information by STIBOKA (1976), Teunissen van Manen (1985) and Leenders (1992) for Den Bosch, information by Harbers and Rosing (1983) and Dekkers (1997) for the Achterhoek, information by Eilander et al. (1982) and Van der Werff (1999) for the Veluwe and information by Scholten (1996) for Nieuwleusen.

Land use systems

Thirteen land use systems were parameterized (Fig. 4.2; Table 4.4). Forest vegetation inputs were calibrated by running the RothC model until an equilibrium SOC stock of 107 ton ha^{-1} was reached, corresponding with the 0-30 cm SOC stocks in ancient Dutch broadleaf forests. Locations of ancient forests were identified using the historical map material (Appendix Chapter 3-4). Mean SOC stock was calculated using the source data

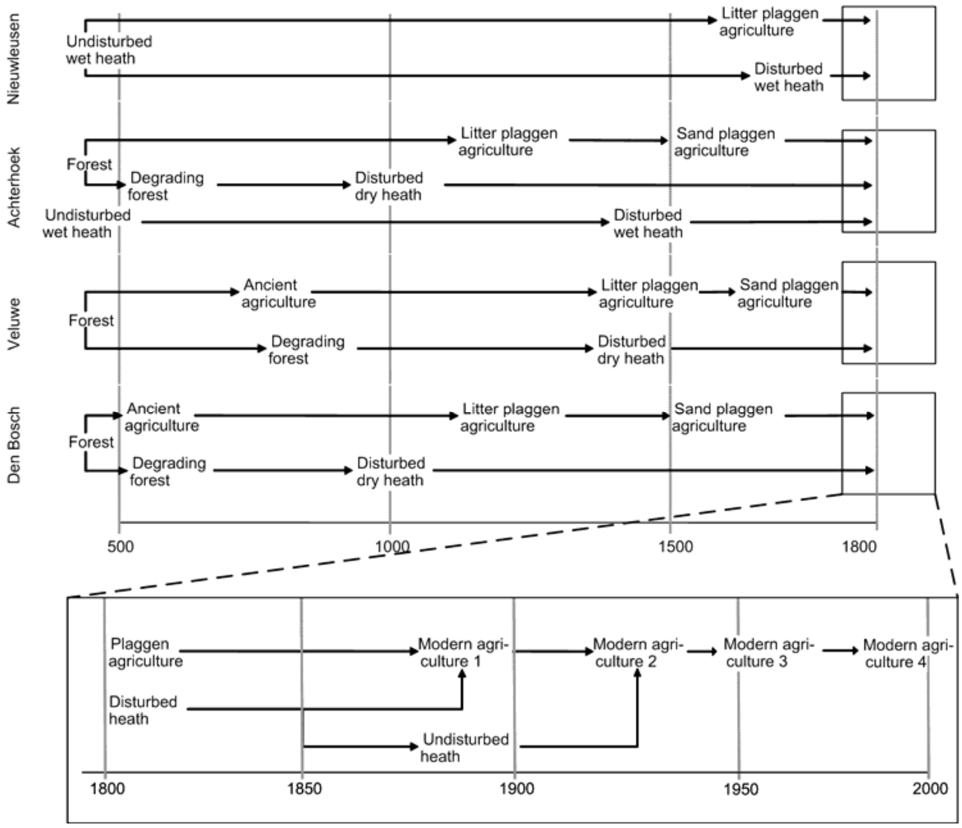

Fig. 4.2. *Land use trajectories from 500 to 2000 that are used in the process modelling.*

from Finke et al. (2002) at ancient broadleaf forest locations. A ratio of decomposable plant material to resistant plant material (DPM/RPM ratio) of 0.25 (Coleman and Jenkinson, 1999) was used.

Dry and wet heathlands were distinguished (Fig. 4.2). For both dry and wet heath, a DPM/RPM ratio of 0.04 was used based on the N to lignin content (Aerts, 1989; Anderson and Hetherington, 1999; Coleman and Jenkinson, 1999; Van Meeteren et al., 2007). Vegetation inputs were calibrated by running the RothC model to an equilibrium SOC stock of 97 ton ha^{-1} (wet heathlands) or 80 ton ha^{-1} (dry heathlands) (Finke et al., 2002). The calibrated vegetation inputs (Table 4.4) agreed with literature values (e.g. Aerts (1989)). Vegetation input of heathland disturbed by plaggen harvesting was set at 80% to 90% of vegetation input by undisturbed heath, based on differences in land use intensities (Spek, 2004).
The land use system “degrading forest” (Fig. 4.2) simulates the transition of forest to heathlands. Vegetation residue input was assumed to be the average of forest and heath (Table 4.4) and a DPM/RPM ratio of 0.17 was used.

Ancient agriculture includes rotations of arable crops and grass with regular fallow periods. Vegetation input was set at 70% of plaggen agriculture and farmyard manure (FYM) inputs at one-third of plaggen agriculture (Table 4.4).

Plaggen agriculture was subdivided into two phases (Fig. 4.2). In the first phase, litter from natural vegetation or straw was used as animal bedding. Litter harvesting resulted in land degradation and therefore in the second phase, smaller amounts of litter supplemented with sand were used as animal bedding (Spek, 2004). Different types of litter were used in each site. In Den Bosch straw and forest litter was used, in the Veluwe straw and litter from forest and heath, in the Achterhoek grass, heath and forest litter and in Nieuwleusen only heath litter. To estimate FYM inputs, decomposition of regionally different amounts of manure application (Spek, 2004) combined with litter was simulated with RothC. In the litter plaggen system, the FYM amount was assumed to be one-third of literature values that mainly describe the 19th-century situation (Van Zanden, 1985; Spek, 2004).

Modern agriculture was subdivided into four periods (Table 4.4). The subdivision is based on temporal changes in fertilizer and manure input and changes in dominant crop types (LEI and Statistics Netherlands, 2008; Statistics Netherlands, 2008) (Fig. 4.2). FYM input for 1976-2005 was derived from data on manure application and on land use data per administrative region. For the other periods, FYM input was scaled to the 1976-2005 input based on livestock numbers in the Netherlands from 1900 to 2005 (Statistics Netherlands, 2008). Vegetation inputs in modern agriculture were calculated site-specific for 1876-1930 and 1976-2005. Area percentages per crop and crop yields at province scale in 1875 (Van Zanden, 1985) were combined with harvest indices (Sinclair, 1998; Kuikman et al., 2003) to calculate vegetation inputs for 1876-1930 (Table 4.4). For 1976-2005, crop areas from recent land use databases (Appendix Chapter 3-4),

Table 4.4. *Farmyard manure input and vegetation input (ton C ha^{-1} year^{-1}) in each site in each land use system.*

Land use system	Site				Month of input
	Nieuwleusen	Achterhoek	Veluwe	Den Bosch	
	Farmyard manure input (ton C ha^{-1} year^{-1})				
Ancient agriculture	n.a.	n.a.	0.1	0.1	November
Litter plaggen	22.9	2.2	3.6	3.3	November
Sand plaggen	n.a.	4.3	5.2	5.2	November
Modern agriculture 1876-1930	0.2	0.2	0.3	0.3	November
Modern agriculture 1931-1950	0.3	0.4	0.5	0.4	November
Modern agriculture 1951-1975	0.8	0.8	1.1	0.9	November
Modern agriculture 1976-2005	1.6	1.8	2.4	2.0	November
	Vegetation input (ton C ha^{-1} year^{-1})				
Forest	n.a.	3.24	3.24	3.24	Jan-Dec
Forest degrading to heath	n.a.	2.65	2.65	2.65	Jan-Dec
Undisturbed dry heathland	n.a.	2.07	2.07	2.07	Jan-Dec
Undisturbed wet heathland	2.79	2.79	n.a.	n.a.	Jan-Dec
Disturbed dry heathland	n.a.	1.76	1.76	1.66	Feb-Dec
Disturbed wet heathland	2.51	2.37	n.a.	n.a.	Feb-Dec
Ancient agriculture	n.a.	n.a.	0.67	0.89	Apr-Jul
Litter plaggen	1.01	0.67	0.67	0.97	Apr-Jul
Sand plaggen	n.a.	1.44	1.44	1.54	Apr-Jul
Modern agriculture 1876-1930	2.68	2.60	2.65	2.87	Apr-Jul
Modern agriculture 1931-1950	3.10	2.92	2.67	3.23	Apr-Jul
Modern agriculture 1951-1975	3.08	3.24	2.64	5.12	Apr-Jul
Modern agriculture 1976-2005	3.35	2.72	3.27	3.14	Apr-Jul

n.a. Not applicable because land use system is not present at the site.

yields per agricultural region (LEI and Statistics Netherlands, 2008) and harvest indices were used to calculate vegetation inputs. For 1931-1950 and 1951-1975, yields per crop were linearly interpolated between the values for 1876-1930 and for 1976-2005. From the interpolated yields the vegetation input was calculated using the area percentage per crop across the Netherlands (Statistics Netherlands, 2008) and harvest indices.

4.3.3. Sensitivity analysis

To assess the causal relations between site conditions and land use history on one hand and SOC dynamics on the other hand, the sensitivity of SOC dynamics was analyzed with the RothC model. Similar effects of these factors were expected in each site and therefore sensitivity analysis was done only for the Veluwe site. Sensitivity to clay content, precipitation and temperature was tested by simulating SOC dynamics for the land use trajectories as found in the Veluwe site (Fig. 4.2), varying these factors between the range of one standard deviation. The effect of land use history was tested by running RothC starting with an identical SOC stock of 107 ton ha^{-1} and four contrasting sets of model input settings: low-input and high-input historical and present-day land use, based on the land use systems disturbed heath, plaggen agriculture, Veluwe (1976-2005) and Achterhoek (1876-1930) (Table 4.4) for 200 years followed by 200 years of high-input present-day agriculture.

4.4. Results and interpretation

4.4.1. Empirical modelling

The analysis of SOC contents across the region indicates that the sites significantly differ in mean SOC content. Mean (± standard error) SOC contents are highest in the Nieuwleusen site (4.1 ± 0.05%); followed by the Achterhoek site (2.7 ± 0.02%) and the Veluwe site (2.6 ± 0.02%). The Den Bosch site has the lowest SOC contents (2.1 ± 0.02%).
The ANOVA and correlation analysis indicate that site factors separately can only explain up to 10% of the SOC variability over all sites and up to 23% within each site (Table 4.5). All site factors combined can explain between 11% (Veluwe) and 36% (Nieuwleusen) of SOC variability within the sites and 10% of SOC variability over all sites.
Elevation is found to better explain SOC variability within the sites than over all sites. Elevation is identified before as a determinant for SOC variability at small scale (Mueller and Pierce, 2003). The lower R^2 for elevation over all sites indicates a scale dependency; the relative position in the landscape is a more important determinant for SOC variability than the absolute elevation. This matches results by Tan et al., (2004). Also for texture, weak associations with SOC variability at large spatial extent are identified before. R^2 values below 5% are common (Springob et al., 2001; Tan et al., 2004; Meersmans et al., 2008).

Table 4.5. *Associations with SOC content (R^2) per site and for the total dataset.*

Independent variables	Site				
	Nieuwleusen	Achterhoek	Veluwe	Den Bosch	All sites
	Associations with SOC content – Determinants separately				
Site factors					
Loam content	23%	11%	6%	5%	0%
Median sand grain size	6%	1%	4%	6%	2%
Elevation	2%	8%	8%	4%	0%
Groundwater class	5%	3%	2%	3%	2%
Soil type	21%	6%	2%	12%	10%
Geomorphology	0%*	6%	3%	8%	4%
Land use history					
Reclamation type	14%	1%	2%	3%	17%
Land use 1900	15%	4%	3%	8%	2%
Reclamation age	12%	1%	1%	1%	1%
Present-day land use and management					
Land use 1999	0% *	1%	1%	3%	1%
Permanent grassland	0% *	0% *	1%	1% *	0%
OC_{eff} input by crops per zip code region	19%	4%	0%*	1%	2%
OC_{eff} input by livestock per zip code region	16%	0%*	1%	3%	2%
OC_{eff} input by crops per municipality					6%
OC_{eff} input by livestock per municipality					9%
	Associations with SOC content – Multivariate regressions				
Soil type – Groundwater class	16%	7%	2%	10%	6%
Site factors	36%	19%	11%	15%	10%
Site factors-Reclamation type	40%	19%	14%	16%	20%
Site factors-LU 1900	39%	20%	11%	18%	12%
Site factors-LU 1900 – Reclamation type	41%	21%	14%	-	21%
Site factors-LU2000	-	19%	-	-	10%
Site factors-LU2000-OC_{eff}	41%	20%	-	18%	11%
Site factors-LU1900-Recl. type-LU2000	-	-	-	-	21%
Site factors-LU1900-Recl. type-LU2000-OC_{eff}	42%	21%	-	21%	21%

* Not significant at $p<0.05$.
– No significant R^2 increase upon adding a variable.

Soil type has the strongest association with SOC variability among the site factors tested. SOC variability between soil types is often found (Tan et al., 2004) and used for stratification for national-scale SOC inventories, e.g. Arrouays et al. (2001). Geomorphology explains SOC variability in a few sites. In other studies, associations between geomorphology and SOC variability are observed as well (Bedard-Haughn et al., 2006; Follain et al., 2007). For both soil type and geomorphology, the R^2 over all sites is in the same range as within each site.

Both reclamation type and land use in 1900 are significantly associated to SOC variability within the sites (Table 4.5). Over all sites, reclamation type explains 17% of SOC variability. This reflects differences in dominant reclamation type: In the Nieuwleusen site former wet heathlands are common while the other sites are dominated by reclamations of dry heath and old agricultural reclamations. Within the sites, the land use map of 1900 has a stronger association with SOC than the reclamation type map.

Table 4.6. *Results of Mann-Whitney tests (effect size) on the impact of reclamation age before or after 1850 on SOC content at the dominant groundwater class-texture combinations[1].*

Texture	Site	Groundwater class[2]	
		III	VI
Fine sand with low loam content	Nieuwleusen	-0.30	-0.47
	Achterhoek	-0.06*	-0.37
	Veluwe	0.04*	-0.16
	Den Bosch	n.p.	-0.07*
	All sites	-0.31	-0.35
Fine sand with high loam content	Nieuwleusen	-0.18	-0.15*
	Achterhoek	-0.04*	-0.33
	Veluwe	-0.15	-0.03*
	Den Bosch	-0.25	-0.16
	All sites	-0.08	-0.03*

1 Negative effects indicate that old reclamations have higher SOC content than young reclamations.
2 Groundwater classes are explained in Table 4.2.
* Not significant at p<0.05.
n.p. Not present at site.

In Table 4.6, the effect of reclamation age within texture-groundwater combinations is shown. In most texture-groundwater class combinations, areas reclaimed before 1850 have higher SOC contents than areas reclaimed after 1850 (shown by the negative effect size). Differences are largest in the Nieuwleusen site with mean SOC contents of 4.0% in areas reclaimed before 1850 and mean SOC contents of 3.5% in areas reclaimed after 1850. In the Den Bosch site, differences are smallest (2.2% vs. 2.1%).
Present-day land use hardly explains SOC variability (Table 4.5). This applies for both the land use map indicating arable crops, and the map distinguishing permanent and temporary grasslands. Crop types probably cannot explain SOC variability because large parts of the sites are under crop rotations. Total effective carbon (OC_{eff}) production by livestock per municipality explains 9% of SOC variability, OC_{eff} input by crops explains 6%. Historical land use explains more SOC variability than present-day land use, as well within the sites as over all sites (Table 4.5).

When combining site factors and historical land use in a multivariate regression, between 14% and 41% of SOC variability within the sites can be explained. Multivariate regressions using historical land use yield higher R^2's than models using present-day land use. Present-day land use only improves a multivariate regression model for the Achterhoek site and a model for the total dataset. OC_{eff} input can increase the R^2 in each site but the Veluwe.

By combining site factors, historical and present-day land use, up to 42% of SOC variability within the sites can be explained (Table 4.5). This is lower than found by Sonneveld et al. (2002). This difference can be explained by the smaller extent of that study.

4.4.2. Process-based modelling: Effect of land use trajectory on SOC dynamics

With RothC, SOC dynamics were modelled for three land use change trajectories. In all sites, modelled SOC stocks in 2005 are larger in the long-term agriculture than in the old and young reclamations (Table 4.7). Long-term agriculture has higher percentages SOC in humified OM than young and old reclamations. Young reclamations and old reclamations do not differ in SOC stocks and in percentage SOC in humified OM. In most sites and land use trajectories, modelled SOC stocks are within the range of one standard deviation of the empirical data. In all sites, the simulated difference between long-term agriculture and the young and old reclamations is larger than found in the empirical data.

Temporal change of SOC stocks for the long-term agriculture and the young reclamations between 1850 and 2000 is shown in Fig. 4.3. Generally, SOC stocks in long-term agriculture start to decrease upon conversion of plaggen agriculture to modern agriculture. This is due to the lower and less resistant inputs in modern agriculture compared to plaggen agriculture. In the Veluwe site, SOC stocks are increasing since 1976 because of increased FYM input. In the Den Bosch site, vegetation inputs are high in 1950-1975 because of a high area percentage of potatoes within the land use system. This results in an increase of SOC stocks.
Areas that are reclaimed in the late 19th century have been depleted from nutrients and carbon during the plaggen harvesting period. Upon reclamation, inputs increased, followed by an increase of the SOC stock (Fig. 4.3). No increase occurs in the Nieuwleusen site because of the shorter history of depletion and the higher SOC stock in the wet heathland at the start of the simulation. Between 1930 and 1950, SOC stock decreases in all sites. The main reasons are innovations in crop production that increase the DPM/RPM ratio.

Since 1950 FYM inputs increase (Table 4.4), resulting in increased SOC stocks. The decrease in SOC stock in the Den Bosch site is due to the introduction of large-scale silage maize production. Vegetation inputs in maize production seem to be insufficient to maintain the SOC stock.

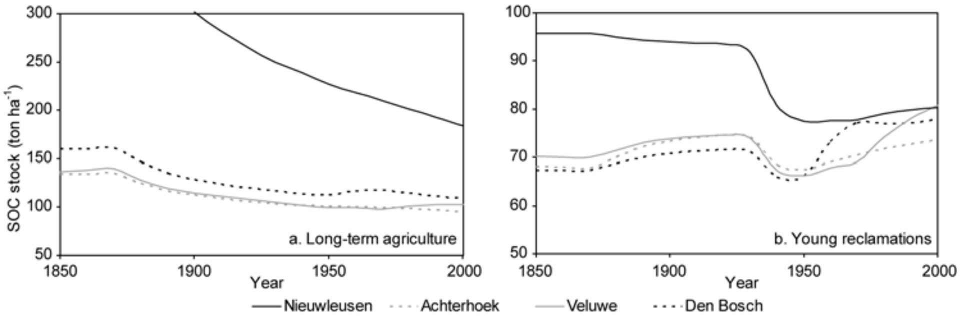

***Fig. 4.3.** Temporal changes of SOC stocks between 1850 and 2000 in all sites in long-term agriculture (a) and young reclamations (b).*

Table 4.7. *Comparison of SOC stocks in 2005 based on process modelling and SOC stocks derived from empirical analyses.*

Land use trajectory, pool	Site			
	Nieuwleusen	**Achterhoek**	**Veluwe**	**Den Bosch**
Long-term agriculture				
SOC stock (ton ha^{-1}):				
2005 process model	179.6	95.3*	103.1	108.5**
Empirical model mean (SD)	188.9 (107.2)	108.3 (38.9)	101.0 (35.4)	84.5 (19.3)
Process model % SOC in humified OM	85%	74%	72%	74%
Old reclamations – dry / wet heath				
SOC stock (ton ha^{-1}):				
2005 process model	77.4*		82.2*	79.0
Empirical model mean (SD)	131.1 (56.0)		105.2 (100.1)	85.7 (97.8)
Process model % SOC in humified OM	66%		64%	64%
Young reclamations - dry heath				
SOC stock (ton ha^{-1}):				
2005 process model		74.0*	81.9*	78.5
Empirical model mean (SD)		99.3 (50.5)	94.9 (52.7)	96.1 (126.8)
Process model % SOC in humified OM		66%	64%	64%
Young reclamations - wet heath				
SOC stock (ton ha^{-1}):				
2005 process model	80.5**	84.6*		
Empirical model mean (SD)	137.6 (43.6)	99.6 (36.7)		
Process model % SOC in humified OM	67%	72%		

* Modelled values are not within 95% confidence interval of mean

** Modelled values are not within mean ± one standard deviation

4.4.3. Sensitivity analysis

Modelling of SOC stocks under different temperature regimes shows that SOC stocks increase when temperature decreases (Fig. 4.4). SOC stocks decrease on increasing precipitation and increase upon increasing clay content. Variation of climate and clay content results in larger differences in SOC stocks in the long-term agriculture than in the old and young reclamations. Variation in clay content has a less strong effect than variation in climatic conditions (Fig. 4.4).

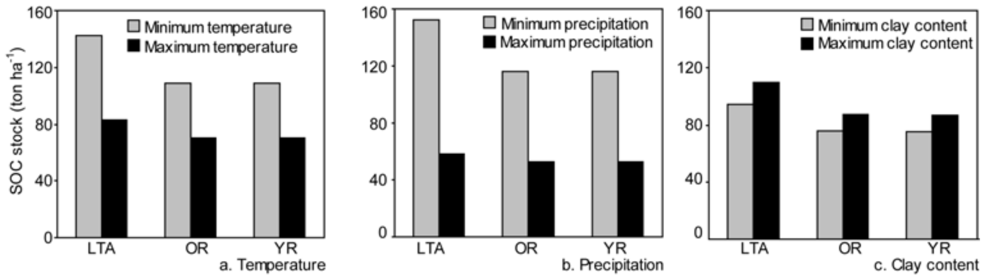

Fig. 4.4. *SOC stocks modelled with RothC with variation of temperature (a), precipitation (b) and clay content (c) in the Veluwe site under different land use trajectories. LTA = long-term agriculture, OR = old reclamations, YR = young reclamations.*

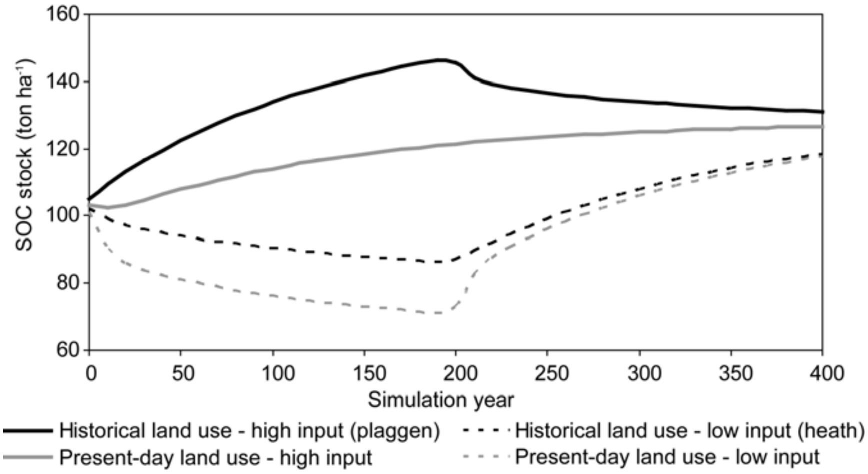

Fig. 4.5. *Temporal changes in SOC stocks under contrasting land use systems in the Veluwe site. From year 200 onwards, conversion of all four land use systems into present-day land use with high inputs is simulated.*

Modelling changes of SOC stocks under two contrasting historical land use systems results in SOC stock differences of 67% after 200 years (Fig. 4.5). From year 200 onwards, conversion of both land use systems to high-input modern agriculture is simulated. SOC stocks then start to converge, resulting in a SOC stock difference of 24% in year 300. Contrasting present-day land use systems have a similar effect on SOC stocks; the difference between high-input and low-input systems is 66% after 200 years and a difference of 20% lasts in year 300.

Both disturbed heath and low-input agriculture result in a SOC stock decrease. The difference between the low-input systems is maximally 19% and quickly decreases after conversion to high-input agriculture. The results suggest that although land use has strong impact on SOC stocks, significant differences only emerge after several decades of unchanged land use and differences between high-input and low-input systems will last decades after conversion to identical land use.

4.5. Discussion

4.5.1. Determinants of SOC content

Associations between site factors and SOC contents found in the empirical analysis are site-specific and not consistent across the studied physiographic region. Elevation, original drainage and texture determined the patterns of historical land use. Effects of groundwater and texture on SOC contents can be overruled by the effect of land use history on SOC contents at landscape scale. Additionally, SOC contents could be influenced by climatic differences across the Netherlands. SOC contents are highest in the site with highest annual rainfall (Nieuwleusen) and lowest in the site with the lowest annual rainfall (Den Bosch). This is consistent with studies by McLauchlan (2006) and Olsson et al. (2006) but contrasting with the process modelling results and with Parton et al. (1987) and Bellamy et al. (2005) who suggest that in freely drained soils, drier conditions result in slower decomposition if lack of moisture limits soil microbes. Moisture availability in RothC is controlled by precipitation and evaporation. At many locations in the Dutch sand area, the soil water regime is principally controlled by groundwater dynamics. In such soils, conditions for decomposition are expected to improve under drier climatic conditions and no decrease of decomposition is expected in groundwater-controlled situations. RothC thus might overestimate SOC stocks in such situations.

Historical land use explains SOC variability better than present-day land use (Table 4.5). Several explanations can be provided for this result. First, in the traditional heathland farming plaggen manure was a source for large amounts of resistant OM (Springob and Kirchmann, 2002; Heumann et al., 2003). This is consistent with the process modelling results, where long-term agriculture has higher percentages SOC in humified OM than young and old reclamations. At the same time, plaggen harvesting resulted in depletion of nutrients and SOC in heathlands (Webb, 1998). Historical land use thus results in a highly variable carbon copy on the landscape with large differences in nutrient balances at small distances. Second, plaggen agriculture has lasted several centuries while present-day management with high manure inputs is only present since a few decades. The sensitivity analysis indicated that contrasting present-day land use systems result in similar SOC stock differences as contrasting historical land use systems, but clear differences only emerge after a few decades (Fig. 4.5). Because of the long-lasting enrichment and depletion, and the slow response of SOC stocks to land use changes, still an effect of historical land use on SOC stocks can be seen. This could also explain the lack of difference in SOC stocks between old and young heath reclamations, as they only have 30 years of different land use history (Fig. 4.2).
A final reason explaining the effect of historical land use on SOC stocks is the dependence of historical land use systems on resource availability, especially good drainage conditions. The small scale of agriculture (Hamre et al., 2007) provided the possibility to follow patterns of landscape suitability in land use allocation. Combined with the carbon copy of the traditional heathland farming, this results in a relatively high percentage explained SOC variability (Table 4.5). Present-day land use in contrast has a larger scale and is less constrained by resource availability. Nutrient inputs are uniform over a large extent and are mostly controlled by governmental regulations. Additionally, large parts of the sites are under a rotation with grass and arable crops. Therefore, present-day land use is not expected to result in clear differences in input and output of OM and in strong variability of SOC at the landscape scale. Over all sites, regional differences in rotations result in regional differences in input balance. Therefore OC_{eff} inputs explain variability of SOC over all sites.

4.5.2. Uncertainties

Within the sites, a large proportion of SOC variability cannot be explained by the factors considered in this study. Possible reasons are local variability within fields due to small local disturbances by management or redistribution of manure within fields through water flow (Sonneveld et al., 2006). At the field scale, SOC variability is high and characterized by a lot of noise and outliers. Low levels of explanation are therefore expected even for models that include all possible relevant factors explaining SOC variability (Meersmans et al., 2008). Additionally, the use of datasets developed at different scales will have caused inaccuracies in the analysis due to mismatch of mapping units.
In the process-based modelling results SOC stocks of long-term agriculture are overestimated relative to the empirical data. Due to sand addition in plaggen agriculture, SOC stocks could have been diluted. A similar effect was observed in Flanders by Van Wesemael et al. (2005) due to increased plough depth. However, the sand addition could also have caused an extra addition of inert OM (Springob et al., 2001; Blume and Leinweber, 2004).
In the Nieuwleusen site, process-based modelling underestimates SOC stocks in young and old reclamations compared to the empirical data. Heath areas under wet conditions such as found in this site normally have thick litter layers. Upon reclamation, litter layers are sometimes mixed with the mineral topsoil. Calibrating RothC to the observed SOC stocks with a one-year fallow in the reclamation year with only FYM input to simulate this, required an addition of 400 ton C ha^{-1} of FYM. This contains 8 ton C ha^{-1} in the form of humified OM. As litter layers in heathlands can contain up to 55 ton C ha^{-1} (Finke et al., 2002) this seems a plausible explanation for the observed difference.

To calculate carbon stocks from the OM contents that are observed in the soil surveys, a standard bulk density of 1300 kg m^{-3} was used, based on (Finke et al., 2002). Bulk density is however influenced by agricultural management practices. Present-day management using more heavy machinery than ancient agriculture probably has increased bulk density by compaction (Tomlinson and Milne, 2006). Conversion of native vegetation to cropland is generally found to increase bulk densities as well (Bauer and Black, 1981; Conant et al., 2001). The use of a standard bulk density thus will have caused inaccuracies but differentiation was not possible as observations on bulk density in the different agricultural systems assessed are not available.

4.5.3. Complementarity of empirical and process modelling

Empirical models are often used to describe SOC variability on scales varying from field (Mueller and Pierce, 2003) and landscape (Bedard-Haughn et al., 2006) via region (Meersmans et al., 2008) and country (Tan et al., 2004) to continents (Jones et al., 2005). The advantage of an empirical model is that it is a straightforward and reproducible method to quantify SOC variability. Disadvantage is uncertainty about the causality behind the associations. Processes are not explicitly quantified and, consequently, an empirical model is only applicable to the system on which it is estimated. This is confirmed in this chapter, where sites show different statistical associations between SOC variability and determinants (Table 4.5). Relations are also scale-dependent (Table 4.5) and can change upon changing the spatial scale of analysis (Nol et al., 2008; Schulp and Veldkamp, 2008). Therefore, the empirical results of individual sites cannot easily be used in upscaling.

In process-based models, SOC stocks are described with equations that represent known processes. There are several models for SOC dynamics, with parameter and structure complexity ranging from relatively simple in the RothC model (Jenkinson et al., 1990)) to the highly complex DNDC model (Li et al., 1992). With a well-established process-based model for SOC stock changes, causalities between determinants and SOC stock changes can be assessed. Advantage is that if a cause-effect relation between a determinant and SOC stock change is correctly represented, this provides a universal basis for upscaling. Second, with a process model SOC stock changes can be tested under circumstances where empirical analysis is not possible (e.g. future climate scenarios). A disadvantage is the need to calibrate the process model specifically for each site. If a variable can have contrasting effects on SOC stocks, this might have different effects under different circumstances. Secondly, process models for SOC dynamics are mostly developed at plot scale. At other scales, a plot scale model might not work correctly (Dendoncker et al., 2008; Schulp et al., 2008a). Finally, the data requirements of complex process models are often high and cannot be fulfilled at sufficient detail in regional assessments (Verburg et al., 2006b).

4.6. Conclusions

This study explained between 14 and 41% of SOC variability for four sites across the Dutch sand area individually and 21% of SOC variability over all sites (Table 4.5). The different associations between SOC and its determinants between the sites reflect spatial differences in processes governing carbon dynamics. Such regional differences in SOC behaviour should be considered in upscaling. Differences in associations with SOC within the sites when compared with effects over all sites stresses the scale dependency of determinants for SOC variability. The empirically determined associations are confirmed by simulations with a process-based model. The consistent results give confidence that the empirical modelling explains causal relations between SOC variability and its determinants. The spatial variation of land use and management history, both within the sites and across the physiographic region is an important determinant for SOC variability that should be considered when upscaling SOC contents. Inclusion of land use history as a determinant will lead to improved regional estimates of SOC.

Appendix Chapter 3-4

Table A3-4. *Overview of data used in Chapter 3 and 4.*

Name	Scale	Description	Source
		Soil survey data	
SOC content M50 Loam content Lutum content	1:10.000 1:10.000 1:10.000 1:10.000	Detailed soil mapping of each site that is assessed in Chapter 3 and 4. Includes loam content (% particles <50μm), lutum content (% particles < 2 μm), median of sand size (M50) and SOM content (%). SOC content is SOM content * 0.58. All data are observed for each soil horizon separately.	(Leenders, 1992; Scholten, 1996; Dekkers, 1997; Van der Werff, 1999)
		Biophysical spatial datasets	
Soil map	1:50.000	Spatial distribution of soil types and groundwater classes in the Netherlands.	(STIBOKA, 1964-1987; Kuijer and Rosing, 1994)
Geomorphology	1:50.000	Spatial distribution of geomorphological units, simplified to main land form classes.	(Koomen and Maas, 2004)
AHN	5m resolution 100m resolution	Elevation map (cm +asl) of the Netherlands.	
Slope	5m resolution	Derived from AHN.	This study
		Topographical data	
Topography 1780	1:14.400	Map of the northern part of the Netherlands.	(Versfelt, 2003)
Topography 1850	1:50.000	Military topographical map, based on surveys from the 1830s, partly updated up to 1850.	(Kuiper and Kersbergen, 1864 / 2008)
Topography 1897	1:50.000	Topographical map of the Nieuwleusen site, surveyed in 1883-1897.	
Topography 1908	1:25.000	Topographical map of the Nieuwleusen site, surveyed in 1884, updated until 1908.	
Topography 1932	1:25.000	Topographical map of the Nieuwleusen site, surveyed in 1932.	
Topography 1952	1:50.000	Topographical map of the Nieuwleusen site, surveyed in 1952.	
Distance to water	10m resolution	First order streams in the Nieuwleusen site, derived from 1850 topographic map.	This study
Distance to built-up (DBU) 1780	10m resolution	Euclidean distance (km) to houses in the Nieuwleusen site at the 1780 map.	This study
DBU 1850	10m resolution	Euclidean distance (km) to houses in the Nieuwleusen site at the 1850 map.	This study
DBU 1999	25m resolution	Euclidean distance (km) to built-up area in the Nieuwleusen site in the LGN4 dataset.	This study

Table A3-4. *(Continued).*

Name	Scale	Description	Source
		Spatial land use databases	
LGN1	25m resolution	Land use in 1986. Agricultural land use is subdivided into grassland, maize, sugar beets, cereals and potatoes.	(Thunnissen et al., 1992)
LGN2	25m resolution	Land use in 1992. Classification of agricultural land similar to LGN1.	(Noordman et al., 1997)
LGN3	25m resolution	Land use in 1996. Classification of agricultural land similar to LGN1.	(De Wit et al., 1999)
LGN4	25m resolution	Land use in 1999. Classification of agricultural land similar to LGN1.	(Hazeu, 2005)
HISTLAND	1:50.000	Historical-geographical GIS, indicating 54 reclamation types based on land cover before reclamation and age and characteristics of reclamation patterns.	(De Bont, 2004)
HGN1900	50m resolution	Land use in 1900, based on topographical maps, distinguishing in 11 land use types.	(Knol et al., 2004)
Forest survey 1938	25m resolution	Forest and nature in 1938, distinguishing tree species, age class and classification of other nature.	(Clement and Kooistra, 2003)
HGN1960	25m resolution	Land use in 1960. Classification matching HGN1900.	(Knol et al., 2004)
HGN2000	50m resolution	Land use in 2000. Classification matching HGN1900 and HGN1960.	(Knol et al., 2004)
		Management data	
Permanent grassland	25m-resolution	Derived from LGN1,2,3,4. Indicates permanent grassland, permanent cropland and land under rotation.	This study
Reclamation age	25m-resolution	Classes of reclamation age. Based on maps LU1850, HGN1900, HGN1960 and Forest survey 1938.	This study
GIAB 2005		Annual farmer survey, including crop areas, livestock numbers and manure production per farm.	(Naeff, 2006)

Chapter 5

Future carbon sequestration in Europe - Effects of land use change

Important land use changes are expected in the European Union (EU) the coming decades, having effects on carbon stocks in soil and vegetation. In this chapter, the impact of future land use change (LUC) on future carbon stock change in soil and vegetation in the EU is assessed.
Because LUC is the most dynamic driver of terrestrial carbon stock change, it is important to account for the dynamics of LUC in carbon stock change modelling. LUC was modelled with a high-resolution LUC model while for assessing carbon stock changes a bookkeeping approach was used that takes into account effects of soil and forest age on carbon stock changes. Four scenarios that cover a range of possible future developments were evaluated.
If land use remains unchanged, carbon sequestration rates are expected to decrease by 4% in 2030 relative to 2000. LUC causes an additional sequestration rate decrease of 2% in 2030 in one of the scenarios. In the other scenarios, sequestration rate increases by 9-16% in 2030 relative to 2000. In 2030, the terrestrial biosphere in the EU is expected to sequester between 90 and 111 Mton C yr^{-1}. This is 6.5% – 8% of the projected anthropogenic emissions. Clear differences are found in the spatial distribution of sinks and sources between the scenarios, illustrating that land use is an important factor in future carbon sequestration changes that cannot be ignored.

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5.1. Introduction

Important land use changes are expected in Europe in the coming decades (WRR, 1992; Rounsevell et al., 2005), driven by, among others, changes in agricultural policies, demography and globalization of trade in agricultural commodities. These processes influence the amount of land required for agricultural, forestry, residential, industrial and recreational purposes and influence the diversity of the landscape.
Land use change (LUC) has effects on greenhouse gas emissions and carbon stocks in soil and vegetation (Feddema et al., 2005). Land use types differ in the amount of carbon stored in soil and vegetation (Arrouays et al., 2001; Rodriguez-Murillo, 2001; Lettens et al., 2004; Bellamy et al., 2005) and differ in the potential rate of carbon stock change. Generally, soil organic carbon (SOC) stocks under cropland are lower than SOC stocks under pasture or forest. Forest SOC stocks tend to be higher than pasture SOC stocks. Conversion of forest to pasture or cropland or conversion of pasture to cropland is found to decrease SOC stocks, the opposite conversions usually lead to increased SOC stocks (Guo and Gifford, 2002; Lettens et al., 2005b; Falloon et al., 2006). Besides storage of carbon in the soil, forests store large amounts of carbon in biomass (Freibauer et al., 2004). Future LUC is assumed to have the largest impact among factors that will influence future SOC stock changes (Smith et al., 2005c).

Following the Kyoto protocol, countries have to report SOC stock changes due to LUC. Countries also have the possibility to charge carbon uptake to the Land Use, Land Use Change and Forestry (LULUCF) sector as carbon credits for achieving their greenhouse gas emissions reductions (UNFCCC, 1997). However, large uncertainties in estimates of carbon storage in the LULUCF sector prevail (Post and Kwon, 2000; Ogle et al., 2003). These uncertainties can be attributed to gaps in our understanding of both future LUC and quantification of the response of carbon sequestration to LUC. Therefore, the role of future LUC in carbon sequestration is uncertain.

The aim of this chapter is to assess how LUC between 2000 and 2030 can influence carbon sequestration in soil and vegetation in the European Union (EU). The emphasis is on the role of LUC in the overall carbon balance of the European biosphere. Using the available knowledge of the carbon balance of each land use type, the relative importance of LUC is assessed. Specific attention is given to the role of different LUC trajectories and the spatial distribution of LUC. To accomplish this objective, a detailed LUC assessment at a one-km^2 resolution was used, and a carbon budgeting approach extended with information on soils and forest age that best fits the chosen scale and available data. To account for uncertainties in future LUC, four scenarios that cover a range of societal and political developments are evaluated.

5.2. Methodology

Changes in terrestrial carbon sequestration in the EU between 2000 and 2030 resulting from LUC are assessed for four scenarios. Scenarios as developed in the EURURALIS 2.0 project (Wageningen UR and Netherlands Environmental Assessment Agency, 2007) are used. An overview of the methodology is shown in Fig. 5.1 and the components are discussed in more detail in §5.2.1-§5.2.4. Scenario storylines (Fig. 5.1) are elaborated by specifying parameter values for a model of the global economy and an integrated assessment model that together determine changes in the required agricultural areas (cropland and pasture) at the national scale (Van Meijl et al., 2006; Eickhout et al., 2007). Spatial patterns of LUC are simulated with a LUC allocation model (Verburg et al., 2006a) (Fig. 5.1). Size and spatial distribution of potential carbon sinks and sources are derived with a carbon budget accounting approach using the LUC allocation maps (Fig. 5.1).

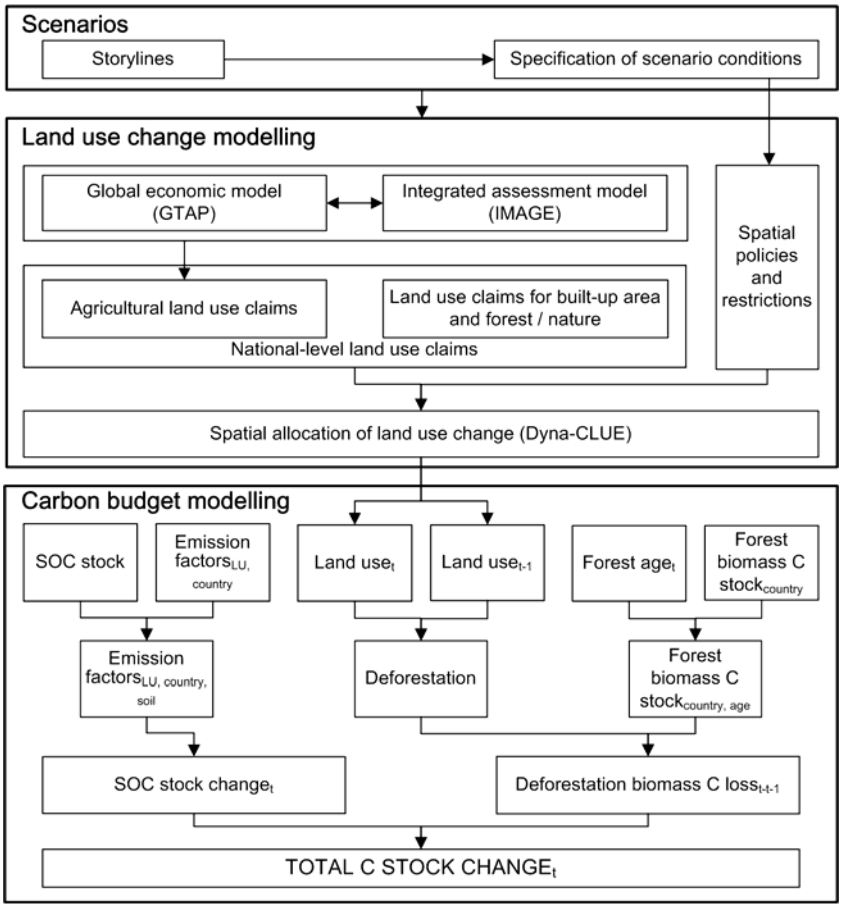


Fig. 5.1. *Flowchart of the carbon budget calculation approach.*

5.2.1. Scenarios

The LUC scenarios used in this study are based on the four IPCC SRES scenarios (IPCC, 2000). The scenario set is structured along two axes, one ranging from increasing globalization (indicated with a 1) to a world with regional economic and cultural focused blocks (2). The second axis ranges from a lean government (A) to a world with a high level of governmental intervention (B). The resulting four scenarios are Global Economy (A1): lean government, strong globalization; Continental Markets (A2): lean government, regional cultural and economic development; Global Co-operation (B1): much governmental intervention, strong globalization; Regional communities (B2), much governmental intervention, regional cultural and economic development. Narratives are made based on these scenarios elaborating the European context including anticipated changes in gross domestic product (GDP), demography and policies such as the EU's Common Agricultural Policy (CAP), Bird and Habitat Directives and Water Framework Directive (Westhoek et al., 2006). Model parameters for macro-economic development and policy settings are quantified based on literature, policy documents and expert judgement.

In the A1 scenario, increased productivity and increased production is expected. Set-aside policies are abolished (Van Meijl et al., 2006; Verburg et al., 2006a; Eickhout et al., 2007). In combination with increased global trade of agricultural commodoties a decrease in agricultural area may be expected. In the A2 scenario claims on land for croplands and urban areas result in a high pressure on land resources. Although productivity increases in A2, the demand for agricultural land is high due to a high economic growth, unchanged CAP and a high level of market protection (Eickhout et al., 2007). In the B1 scenario, agricultural subsidies are abolished after 2010. Up to 2010, the 12 countries that entered the European Union in 2004 and 2007 (the EU12) profit from agricultural subsidies. Set-aside policy is abandoned from 2020 onwards.
In the B2 scenario, agricultural subsidies remain largely unchanged. Population numbers decrease and crop productivity increases (Eickhout et al., 2007). In the profit-driven scenarios, nature development and protection have no priority. In the regulation-driven scenarios nature protection is important and development of new nature is stimulated.

5.2.2. LUC modelling

Requirements for agricultural land are simulated per world region using a combination of a model of the global economy (GTAP) and a land use and environmental development model (IMAGE) (Van Meijl et al., 2006) (Fig. 5.1). Requirements for urban areas (comprising residential, industrial, recreational and infrastructural areas) are a function of demographic changes and changes in area use per person (including prosperity and spatial planning considerations) (Verburg et al., 2006a). The remaining area is covered by different types of forest and (semi-)natural vegetation. If agricultural land is abandoned, it is assumed that re-growth of natural vegetation can occur on the

abandoned area. The speed of re-growth is determined by soil and climate conditions, and the proximity of forests related to seed dispersal. Grazing or population pressure near cities can retard natural re-growth while active management (e.g. afforestation) can provide an advantage as compared to natural succession.
The resulting changes in national land use claims are spatially allocated using the Dyna-CLUE LUC model (Verburg et al., 2002). Dyna-CLUE is a spatially explicit high resolution model in which national scale land use requirements are downscaled to a one-km^2 resolution while changes between natural vegetation types are simulated based on local growth conditions. The assumed spatial policies are incorporated in the land use allocation, e.g. nature protection in designated areas and policies concerning spatial spread of urbanization. Advantage of using a high resolution spatially explicit approach is the possibility to account for specific characteristics of each location, like land use history and the LUC pathway. The LUC allocation uses a one-year time step. Sixteen land use types are distinguished. The methodology of LUC allocation is described in more detail in Verburg et al. (2006a) and Verburg et al. (2008)

5.2.3. Carbon budget calculation

A common approach to model the effect of LUC on carbon stocks is the use of process models describing the dynamics of carbon sequestration based on spatially heterogeneous input parameters (Smith et al., 2005a; Smith et al., 2005c; Falloon et al., 2006). For large scale studies it is often difficult to find sufficiently detailed input information to match the scale at which the processes are described and avoid aggregation bias (Verburg et al., 2006b). An alternative approach that is used in large scale studies are bookkeeping models as used by e.g. Gitz and Ciais (2004), Houghton et al. (1999) and Nabuurs et al. (2003). In a bookkeeping model for carbon sequestration, stock changes are calculated with discrete time steps using empirical data on stock changes. For Europe, large empirical datasets on carbon sequestration of forests are available and empirical data at national scale on processes influencing carbon sequestration or emission. Given the large spatial scale of analysis and the limited detail on land management practices provided by the land use data and scenarios a bookkeeping approach was chosen to make most consistent use of the available information.

Carbon budgets are calculated for each gridcell following:

$$soilstockchange_{ktl} = EF_{kl}$$
$$biomassstockchange_{ktl} = (deforestation\ area_{tl} * forest\ biomass\ carbon\ content_{tl} * removal\ factor)$$
$$stockchange_{ktl} = soilstockchange_{ktl} - biomassstockchange_{ktl}$$

Where $stockchange_{ktl}$ for both soil and biomass (ton C yr^{-1}) is the sequestration or emission of land use type k at time t at grid cell l; EF_{kl} (ton C yr^{-1}) is the emission factor, the emission or sequestration per unit area for land use type k at grid cell l. $Deforestation\ area_{tl}$ (km^2 yr^{-1}) is the area where forest is removed between time t-1 and time *t* at grid cell *l*; *forestbiomasscarboncontent*$_{tl}$ (ton C km^{-2}) is the amount of carbon in biomass in forest at time *t* at grid cell *l* and the *removal factor* is the fraction of biomass that is removed upon deforestation. To calculate stock changes per region or for Europe as a whole, stock changes for all gridcells in the region considered are added up.

Country-specific EF's are available for cropland, pasture, peatland (Janssens et al., 2005) and forest (Karjalainen et al., 2003) (Table 5.1). For other land use types, EF's are derived from these EF's (Table 5.1) following the decision rules given in Table 5.2. For all land use types but forest and forest under succession, only SOC stock changes are considered, because carbon stocks in biomass are negligible compared with SOC stocks (Janssens et al., 2005).
Carbon emissions from croplands depend on the SOC content (Sleutel et al., 2003; Bellamy et al., 2005). Therefore, cropland EF's are modified based on a SOC map (Jones et al., 2005) by multiplying the default EF with a SOC content dependent fraction (Fig. 5.2a). The modification factors range from zero for soils with a SOC content of zero percent to 3.5 for soils with high SOC contents and are derived from Bellamy et al. (2005) and Sleutel et al. (2003). For pastures on mineral soils the pasture EF is used, on peat (European Soil Bureau Network and the European Commission, 2004) the peatland EF is used. This corresponds better with the definitions of pasture and peatland by Janssens et al. (2005). All SOC stock changes consider the upper 30 cm of the soil.

Table 5.1. *EF's (ton C km^{-2} yr^{-1}) and forest biomass carbon stocks (ton C km^{-2}). Negative EF's denote emission, positive EF's denote sequestration.*

Country	Land use type				Forest biomass C stock[2] (ton C km^{-2})
	Pasture[1]	Cropland[1]	Wetlands[1]	Forest / nature[2]	
	Emission factors (ton C km^{-2} yr^{-1})				
Austria	25.5	-16.2	0.1	127	8210
Belgium + Luxembourg	15.8	-9.1	-9.1	127	6977
Bulgaria	6.8	-19.8	-0.3	54	3630
Cyprus	2.8	-10.1	-0.5	42	3818
Czech Republic	6.6	-35.8	-0.7	23	6830
Denmark	2.6	-39.9	-6.0	119	4110
Estonia	2.2	-39.7	-26.2	87	3500
Finland	5.6	-5.5	-12.8	43	2970
France	12.0	-19.1	-0.7	43	5520
Germany	13.6	-28.3	-6.4	134	7190
Greece	2.8	-10.1	-0.5	42	3818
Hungary	6.3	-44.8	-6.4	111	6180
Irish Republic	21.2	-12.3	-52.7	192	2910
Italy	12.7	-19.5	-2.8	67	5390
Latvia	2.9	-44.1	-7.9	87	3500
Lithuania	3.2	-60.8	-2.4	87	3500
Netherlands	18.4	-25.4	-47.1	111	4950
Poland	8.5	-36.6	-26.2	87	5410
Portugal	-4.5	-28.1	-2.0	92	2080
Romania	11.1	-30.7	-0.2	166	6640
Slovakia	12.2	-24.7	-0.7	91	6460
Slovenia	3.7	-8.2	0.5	65	7330
Spain	20.7	-4.7	-0.4	33	1330
Sweden	1.2	-6.5	0.4	68	4070
United Kingdom	24.2	-13.7	-27.5	165	3990

[1] Janssens et al. (2005); [2] Karjalainen et al. (2003)

Table 5.2. *Emission / sequestration behaviour including data source of land use types.*

Land use type	Emission / sequestration behaviour
Built-up areas; Glaciers and snow; Sparsely vegetated areas; Beaches, dunes and coastal flats; Salines	No emission or sequestration
Cropland (Non-irrigated and irrigated)	Cropland EF[1]; depends on initial SOC content[2]
Pasture	Pasture EF[1] on mineral soils, peat EF[1] on peat soils
Inland wetlands	Peatland EF[1]
Heath and moors	Pasture EF[1]
Permanent crops	0.2 * Forest EF[3]
Recently abandoned agricultural land; Forest; nature	Forest EF[3]; age dependent[4]

1 Janssens et al. (2005)
2 Sleutel et al. (2003), Janssens et al. (2005), Bellamy et al. (2005)
3 Karjalainen et al. (2003)
4 Nabuurs (2001), Pussinen et al. (2001)

For forest and forest under succession, carbon sequestration in biomass is taken into account because of the large carbon stock in biomass and the significant changes in biomass in the timeframe considered compared to changes in SOC stocks (Schlesinger and Andrews, 2000). Consequently, carbon losses occur upon deforestation. Estimates of the biomass fraction remaining in the forest after deforestation vary between 15% (Dirkse et al., 2003) and 30% (Gitz and Ciais, 2004). It is assumed here that 20% of the biomass remains in the forest as dead wood and 80% is removed. Therefore the removal factor is set at 0.8. LUC other than deforestation are assumed not to cause sudden releases of carbon from biomass, but only result in change in EF. Carbon sequestration by forests and forest biomass carbon content vary with forest age (Nabuurs, 2001; Pussinen et al., 2001). Based on empirical relations between forest age and Net Biome Productivity (NBP) and biomass carbon stock (Nabuurs, 2001; Pussinen et al., 2001), the default EF and biomass carbon content are modified by age by multiplying the default EF with an age dependent modification factor (Fig. 5.2b and 5.2c). For the forest age in 2000, data from EFISCEN (Nabuurs et al., 2006) are used that report average forest ages at federal state to national scale. The average age of an existing forest is assumed to increase by two years per ten year and the age of a new forest by one year per two years. Such slow age increases are based on average age increases of forest populations in a larger area due to rejuvenation or harvest with subsequent replanting (Nabuurs et al., 2006). Climate changes are expected to be relatively small over the timeframe considered (Eickhout et al., 2007) and are therefore not assessed.

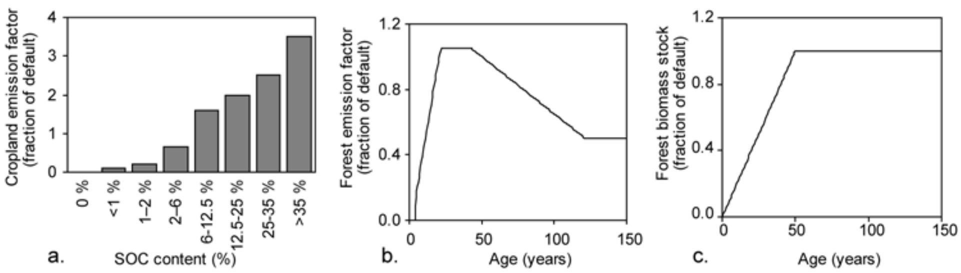


Fig. 5.2. *a. Differentiation of cropland emission factor as a function of initial SOC content. b. Change of forest carbon sequestration with age. c. Change of forest biomass with age.*

Additional to the LUC scenarios, carbon sequestration was calculated for a "No LUC" scenario. This provides the possibility to identify the effect of LUC only. To estimate the sensitivity of the approach to the uncertainty in EF's, carbon budget changes were calculated using the lower and upper confidence limit of all EF's.

5.3. Results

5.3.1. LUC results

Aggregated change in area as simulated for the EU27 for the major land use types that emit or sequester carbon are presented in Fig. 5.3. Most striking is the decrease in cropland area in all scenarios except A2. This decrease can be attributed to decreases in agricultural subsidies, globalization of markets, increasing productivity and increasing land prices due to demands for urban development (Van Meijl et al., 2006; Eickhout et al., 2008; Verburg et al., 2008). In the A2 scenario the decrease in cropland area is limited due to the high population growth, the high level of market protection and relatively low agricultural efficiency. In B1 and B2, large areas of abandoned cropland are converted to forest through afforestation or natural re-growth, while in A1 large areas are converted to urban land (Table 5.3).

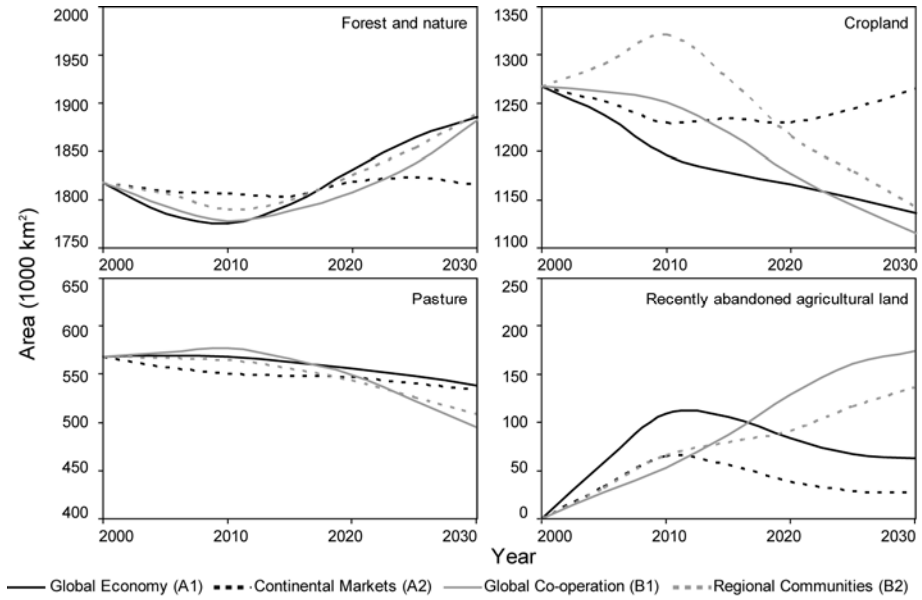
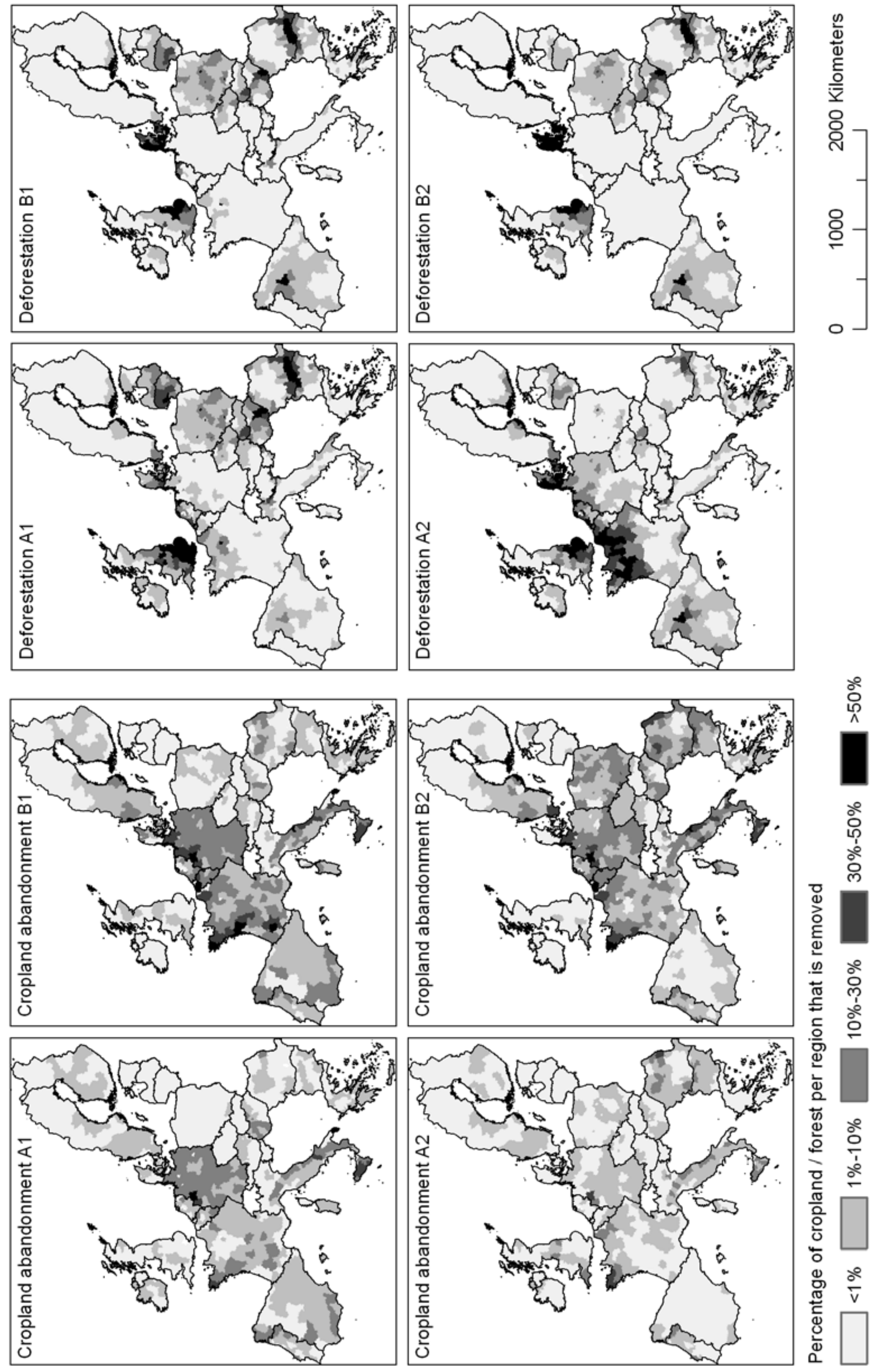


Fig. 5.3. *Aggregated LUC 2000-2030 (*1000 km^2) under the four scenarios.*

Page 89:
Fig. 5.4a. *Percentage of cropland within each HARM2 region that is abandoned or converted to nature between 2000 and 2030 in each scenario (left).*
Fig. 5.4b. *Percentage of forest within each HARM2 region that is deforested between 2000 and 2030 in each scenario (right).*



Cropland abandonment A1
Cropland abandonment B1
Deforestation A1
Deforestation B1
Cropland abandonment A2
Cropland abandonment B2
Deforestation A2
Deforestation B2
Percentage of cropland / forest per region that is removed
<1%
1%-10%
10%-30%
30%-50%
>50%
0
1000
2000 Kilometers

Table 5.3. *Distribution of cropland area and deforested area (km^2) converted to other land use types between 2000 and 2030 (%). Percentages do not sum up to 100% due to conversion to land use types not included in this table.*

Scenario	Area of converted land (km^2)	Land use type after conversion			
		Pasture	Urban	Forest	Abandoned
		Cropland conversion			
Global Economy (A1)	150956	24%	14%	41%	19%
Continental Markets (A2)	58888	15%	5%	58%	17%
Global Co-operation (B1)	182474	15%	3%	36%	45%
Regional Communities (B2)	166482	12%	1%	40%	44%
		Pasture	**Urban**	**Cropland**	
		Deforestation			
Global Economy (A1)	50044	54%	13%	32%	
Continental Markets (A2)	65726	14%	9%	72%	
Global Co-operation (B1)	42011	33%	1%	66%	
Regional Communities (B2)	34311	11%	0%	85%	

Succession of abandoned farmland into forest is especially foreseen in regions that have marginal conditions for agriculture and face little pressure on land resources, such as mountain regions. The transition from agricultural land to forest is consistent with the ongoing changes in these regions over the past decades (Schneeberger et al., 2007; Rutherford et al., 2008).

Additional to differences in amount of LUC, the scenarios differ in spatial distribution of LUC. Most important in the context of this study is the location of cropland abandonment, because this determines the possibilities for development of forest and other natural vegetation types. The location of abandonment determines the regrowth rate, e.g. abandonment of pastures high up in the mountains leads to a slower regrowth as compared to abandonment on fertile lowland locations. The location of deforestation is of importance because of the large immediate carbon loss from biomass upon deforestation.

Agricultural abandonment in A1 and B1 is found mainly in the old EU member states (the EU15), while in B2 abandonment occurs all over Europe (Fig. 5.4a). Differences in spatial pattern result from, among others, differences in policies and market conditions (Verburg et al., 2006a). In the A1 scenario, agriculture tends to concentrate in highly productive areas, leading to abandonment in less productive areas. In B2, regional production and protection of cultural-historic landscapes is favoured. Therefore, several policies are maintained or introduced that compensate farmers in less favourable areas. Additionally, CAP supports are maintained in B2. Due to its late introduction in the EU12 there is a strong effect on land allocation of these policies in the EU12, leading to a different pattern of land allocation across the EU. Deforestation shows a reverse spatial pattern (Fig. 5.4b) since it mainly occurs in regions of agricultural expansion or strong urbanisation. In all scenarios, deforestation is found in Denmark, parts of the United Kingdom and in the border region of Romania and Bulgaria.

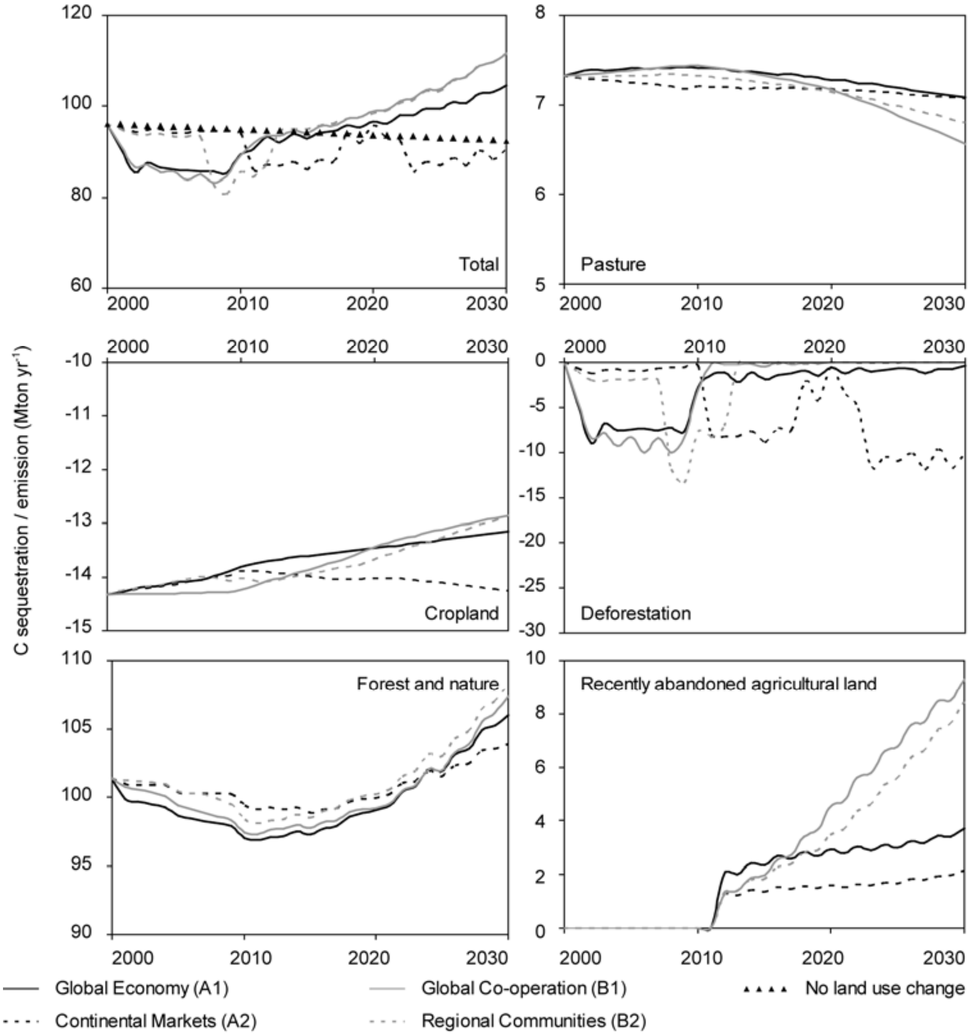


Fig. 5.5. *Temporal changes of carbon sequestration / emission (Mton C yr^{-1}) 2000-2030 for the EU as a whole in each scenario. Total net changes are given as well as changes for the main land use types considered and for deforestation separately.*

5.3.2. Temporal dynamics of EU27 carbon sequestration

Carbon sequestration in 2000 is estimated to amount 96 Mton C yr^{-1} based on the EF's. If land use remains unchanged, carbon sequestration rates are expected to decrease by 4% in 2030 relative to 2000 (Table 5.4, Fig. 5.5) because of ageing of forests. As forest carbon sequestration rates are age dependent (Fig. 5.2b) and European forests are currently in a stage of fast carbon sequestration (Nabuurs et al., 2003), carbon

Table 5.4. *Carbon sequestration in 2030 (Mton C yr^{-1}) in the four scenarios.*

Scenario	Region		
	EU25	EU15	EU12
Global Economy (A1)	104.6	79.1	25.5
Continental Markets (A2)	90.4	61.7	28.8
Global Co-operation (B1)	111.5	83.9	27.6
Regional Communities (B2)	111.6	80.3	31.3
No LUC	92.4	67.4	24.9

sequestration in existing forests will decrease the coming decades.

LUC causes an additional sequestration decrease of 2% in A2 in 2030. In the other scenarios, sequestration increases by 9-16% in 2030 relative to 2000. Total carbon sequestration is mainly determined by high sequestration in forests, emissions from cropland and deforestation, and small amounts of sequestration in pastures and abandoned land (Fig. 5.5). Emission or sequestration rates in soils of cropland, pasture, forest and abandoned land change gradually. For pasture, sequestration changes do not completely follow area changes (Fig. 5.3, Fig. 5.5). In B1, the pasture sequestration continuously decreases, while the area increases until 2010 because pasture is dominantly abandoned at locations where sequestration is high and new pasture is found in areas with lower sequestration. The contrary is seen in B2. Here, pasture sequestration decreases by 7% while the pasture area decreases by 11%. For croplands, especially in B2 there are differences between area changes and emission changes. As cropland emission depends on SOC content, the location of cropland abandonment is of importance for cropland emission. In forests, sequestration changes due to area changes are strengthened by the effect of forest age on sequestration. Forest sequestration will decrease due to ageing and young forests sequester only small amounts of carbon. Abandoned farmlands are assumed to start to sequester carbon only several years after abandonment. Therefore, when sequestration in abandoned farmland starts, it continuously increases in the profit-driven scenarios, even while the area decreases (Fig. 5.3, Fig. 5.5). In A2, the increasing sequestration by recently abandoned agricultural land from 2010 onwards dampens the forest sequestration decrease.

Deforestation has a two-fold effect on carbon sequestration. First, forest carbon sequestration decreases due to area decrease. Second, biomass loss causes a carbon emission that can equal up to 15% of the annual terrestrial carbon sequestration. Because of the large and immediate impact of deforestation on terrestrial carbon sequestration (Fig. 5.5), gross deforestation and timing are important factors in the annual carbon balance. Especially in the profit-driven scenarios, deforestation at a number of locations temporally causes carbon emission although there is net afforestation over Europe as a whole (Fig. 5.5; Fig. 5.6). Finally, the impact of deforestation depends on the land use type to which forest is converted. In the regionalization scenarios (A2 and B2), deforestation is mainly done for cropland expansion, while in the globalization scenarios (A1 and B1) considerable areas of forest are converted to pasture (Table 5.3). Therefore, in the regionalization scenarios

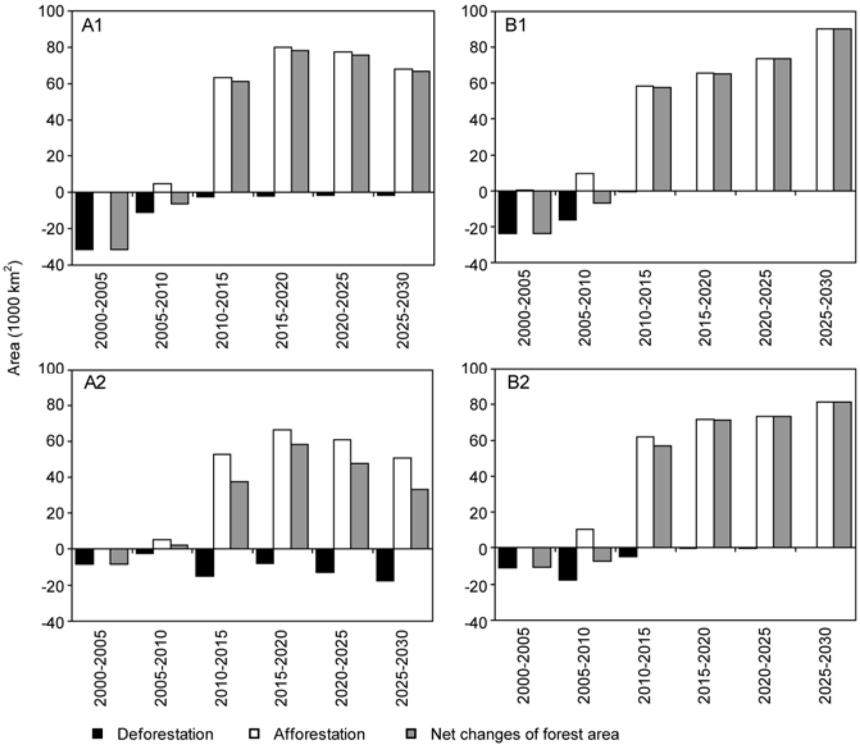


Fig. 5.6. *Deforestation, afforestation and net change (*1000 km²) 2000-2030 in each scenario.*

deforestation has more impact on carbon sequestration because converting forest to cropland causes a change of the soil from a sink into a source.

Altogether, temporal changes of carbon sequestration show irregular patterns because of the interaction between different LUC's and the location of LUC.

In A1, total carbon sequestration decreases up to 2008, mainly because of deforestation, and after that increases to 104.6 Mton C yr^{-1} in 2030 (+9%; Fig. 5.5). In A2, there is little deforestation until 2010. Deforestation after 2010 is variable in time (Fig. 5.6) and causes large temporal variation in net carbon sequestration (Fig. 5.5). The sequestration decrease is 6% in 2030 relative to 2000 in the EU27 (Fig. 5.5). The B1 scenario shows an increase in sequestration of 16%, although net sequestration decreases up to 2005 (Fig. 5.5). In the B2 scenario, annual sequestration increases by 16% in 2030.

The Kyoto protocol requires the EU27 to reduce greenhouse gas emissions by 7.7% relative to the 1990 emission level during 2008-2012 (UNFCCC, 1997). Carbon sequestration in 2000 is estimated to amount to 96 Mton C yr^{-1} based on the EF's (Karjalainen et al., 2003; Janssens et al., 2005), corresponding to 8.5% of the EU27 anthropogenic CO_2 emissions. Compared with baseline emission projections for 2010

(EEA, 2005), the terrestrial ecosystem is expected to sequester around 7% of CO_2 emissions in the EU15 and 7% (B2) to 10% (A2) in the EU12.

5.3.3. Spatial distribution of carbon sinks and sources

Colour plate 5a shows the size and spatial distribution of carbon sources and sinks between 2000 and 2030. Results are aggregated to HARM2 regions. Carbon stock changes between 2000 and 2030 are expected to range between a carbon loss of 1100 ton C km^{-2} and a carbon gain of 3200 ton C km^{-2} within the HARM2 regions. Patterns of cropland abandonment and deforestation from Fig. 5.4 show comparable patterns as the maps of total sequestration or emission in Colour plate 5a because of the direct link between agricultural land abandonment, deforestation and carbon sequestration. The strongest carbon sink is found in the forested areas of Sweden, Central Europe and Romania. Few regions are expected to show net carbon emission in all scenarios: Denmark, East UK, South Hungary, the Czech Republic, the border region of Rumania and Bulgaria, and Lithuania. These areas are expected to be continuously dominated by cropland in all scenarios. The major part of Europe is expected to sequester carbon between 2000 and 2030. In Sweden, Finland and Romania, total carbon sequestration is hardly different between scenarios. Especially in France, Denmark and the remainder of the EU12, there are large differences between the scenarios (Fig. 5.7).

Policy measures assumed in the scenarios like changes in agricultural subsidies work out differently for the EU12 as compared to the EU15. In B1 and B2 the EU12 will profit from agricultural subsidies up to 2010, leading to agricultural expansion. This does not happen in the EU15. Therefore deforestation in these scenarios is concentrated in the EU12, causing emission of carbon between 2000 and 2030 at several locations in the EU12 while the EU15 is dominated by sequestration.

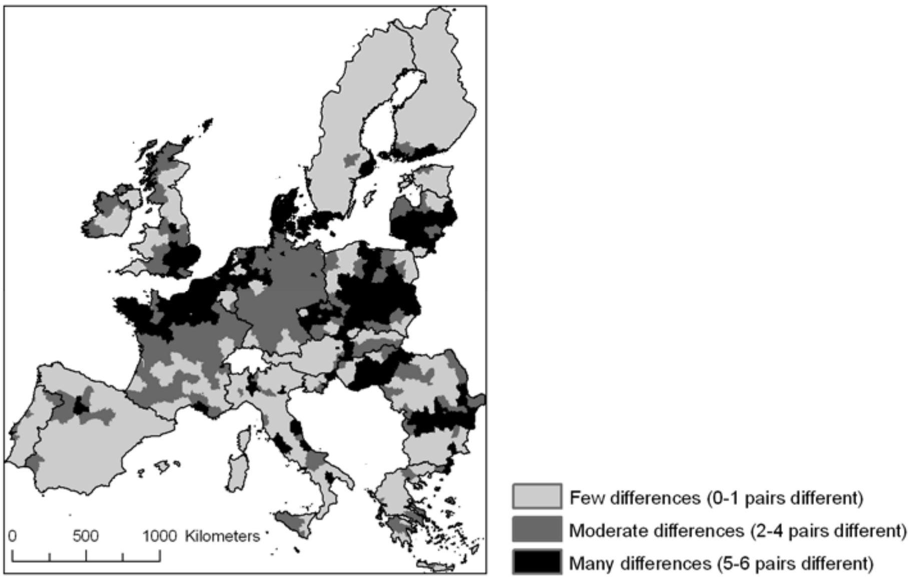


Fig. 5.7. *Similarity of carbon sequestration and emission based on pairwise comparison of the scenarios (pairs are considered different when difference exceeds 10%).*

In the A2 scenario, deforestation is concentrated in the EU15 because the demand for agricultural land is high (Fig. 5.4), resulting in emission in large areas of the EU15. In the EU12 only the major cropland areas emit carbon in A2. For B1 and B2, amounts of emission and sequestration are comparable. In B1, there are more sources than in B2 because in B1 deforestation is more equally distributed over the EU12 than in B2 (Fig. 5.4b). The emission per km^2 in the B1 scenario is however somewhat lower than in B2 (Fig. 5.7), mainly because in B2 more forest is converted to cropland (Table 5.3).
In 2000, sequestration per km^2 in the EU12 is about 80% of the EU15 sequestration because of the larger fraction of land used for cropland in EU12 compared with the EU15. In Romania and Bulgaria separately, sequestration increases in all scenarios because of the relatively low amount of cropland abandonment and the limited deforestation in most of both countries (Fig. 5.4).
The different LUC trends in the regions are amplified by differences in EF's between regions: the pasture EF's in the EU15 are higher than in the 10 countries that entered the European Union in 2004 (EU10), while the EU10 countries have higher cropland EF's and higher forest EF's (Table 5.1) (Janssens et al., 2005).

5.3.4. High-resolution changes in carbon sequestration

To provide insight in how carbon sequestration can differ due to specific LUC trajectories, results for two contrasting scenarios (A2 and B1) for the Netherlands are shown, representing a multitude of processes influencing carbon stocks (Colour plate 5b). Comparison of the land use change maps with the carbon stock change maps indicates high emission levels on locations where cropland is found continuously during the time frame assessed. Differences in carbon emission of cropland are observed because of differences in initial SOC content. The location of cropland and cropland abandonment thus influences carbon stock changes due to differences in soil conditions. Cropland emission has a large impact on the total carbon budget because of the strong emission per unit area, especially in areas with high initial SOC contents. Conversion of cropland to other land use types can cause net sequestration between 2000 and 2030, depending on when the land is abandoned or converted.
High sequestration rates are found in continuous forests. As carbon stocks of forest and nature under succession are age dependent, the location of deforestation is important. Removal of old forests will lead to larger carbon losses than removal of young forests. In the Netherlands, deforestation is however limited in both scenarios.

Differences in spatial patterns of sources and sinks between the scenarios are found at several locations. Along the eastern border (box 1 in Colour plate 5b), removal of natural vegetation for urban expansion in the A2 scenario causes large biomass losses, resulting in a net carbon source. In the B1 scenario, urbanization is less than in A2. Pasture is maintained and natural vegetation expands, resulting in a net sink. In the centre of the Netherlands (box 2 in Colour plate 5b), in the A2 scenario the cropland persists, causing carbon emission, while in B1 extensive land abandonment results in carbon sequestration. In the most south-eastern part of the Netherlands (box 3 in

Colour plate 5b), in A2 the landscape is dominated by cropland while in the B1 scenario a lot of abandonment and urban expansion on former croplands is expected. As a result, in A2 carbon emission is expected while in B1 sequestration in the abandoned farmland is expected. Urban areas do not emit or sequester carbon, resulting in a net zero carbon balance. Differences between box 2 and box 3 in the initial SOC content and in the timing of cropland abandonment result in larger emissions in box 2 in the A2 scenario and differences in the net C balance in the B1 scenario.

In the total carbon budget the location of LUC thus is of importance because of differences in EF related to soil characteristics at one-km^2 resolution. Because of LUC during the period assessed, spatial differences in forest age at one-km^2 resolution emerge. The high spatial resolution of the LUC simulations provides the possibility to identify the most likely locations of forest conversion and to assess their effect on carbon sequestration.

5.3.5. Uncertainty assessment

The EF's used in this study have a high uncertainty level with coefficients of variation around 90% for pasture, 75% for cropland and 20% for wetlands for individual locations (Janssens et al., 2005). For the forest EF's, uncertainties are not quantified (Karjalainen et al., 2003). Comparable forest EF's by Janssens et al. (2005) have a coefficient of variation of approximately 40%. Forest biomass carbon inventories are estimated to have an uncertainty of around 5% for Germany and Norway (Kaipainen et al., unpublished); further data on uncertainties are not available. Carbon budgets were recalculated using the lower and upper confidence limit of the reported mean EF's (Fig. 5.8). The confidence interval of sequestration within one scenario in 2030 is around 65 Mton C yr^{-1}. Difference in sequestration between the minimum and maximum scenario in 2030 when using the mean EF's is 21 Mton C yr^{-1}.
Notwithstanding the large uncertainties in EF's, temporal trends and differences between the scenarios are consistent: sequestration increases in all scenarios but A2, sequestration decreases up to 2006 in the globalization scenarios and sequestration decreases up to 2010 in B1.

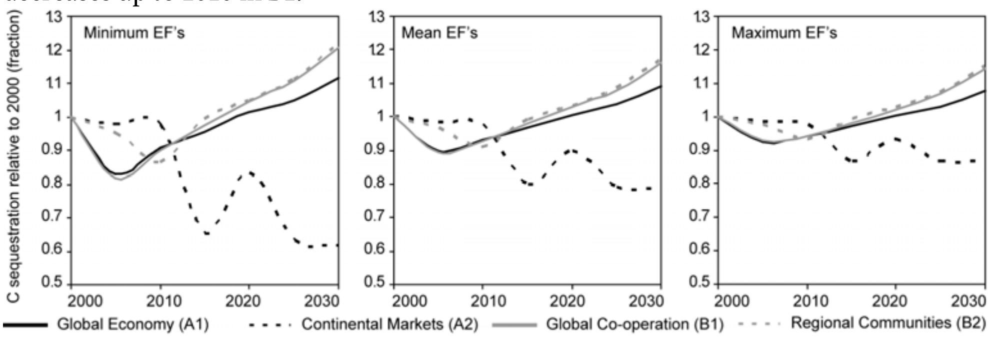


Fig. 5.8. *Trend of minimum, mean and maximum carbon sequestration (fraction of sequestration in 2000) between 2000 and 2030 as calculated from uncertainty assessment.*

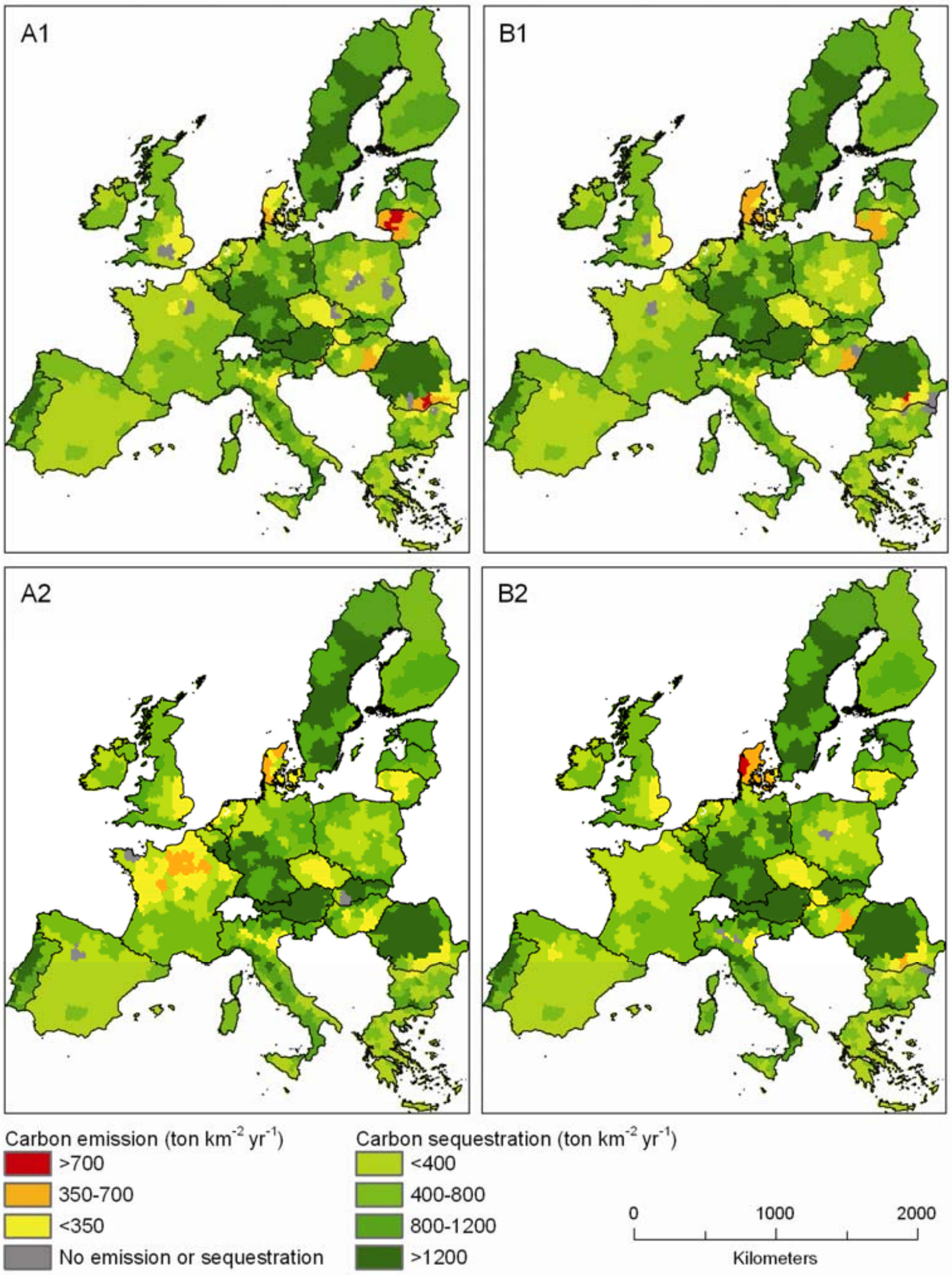
5.4. Discussion

5.4.1. Evaluation of methods

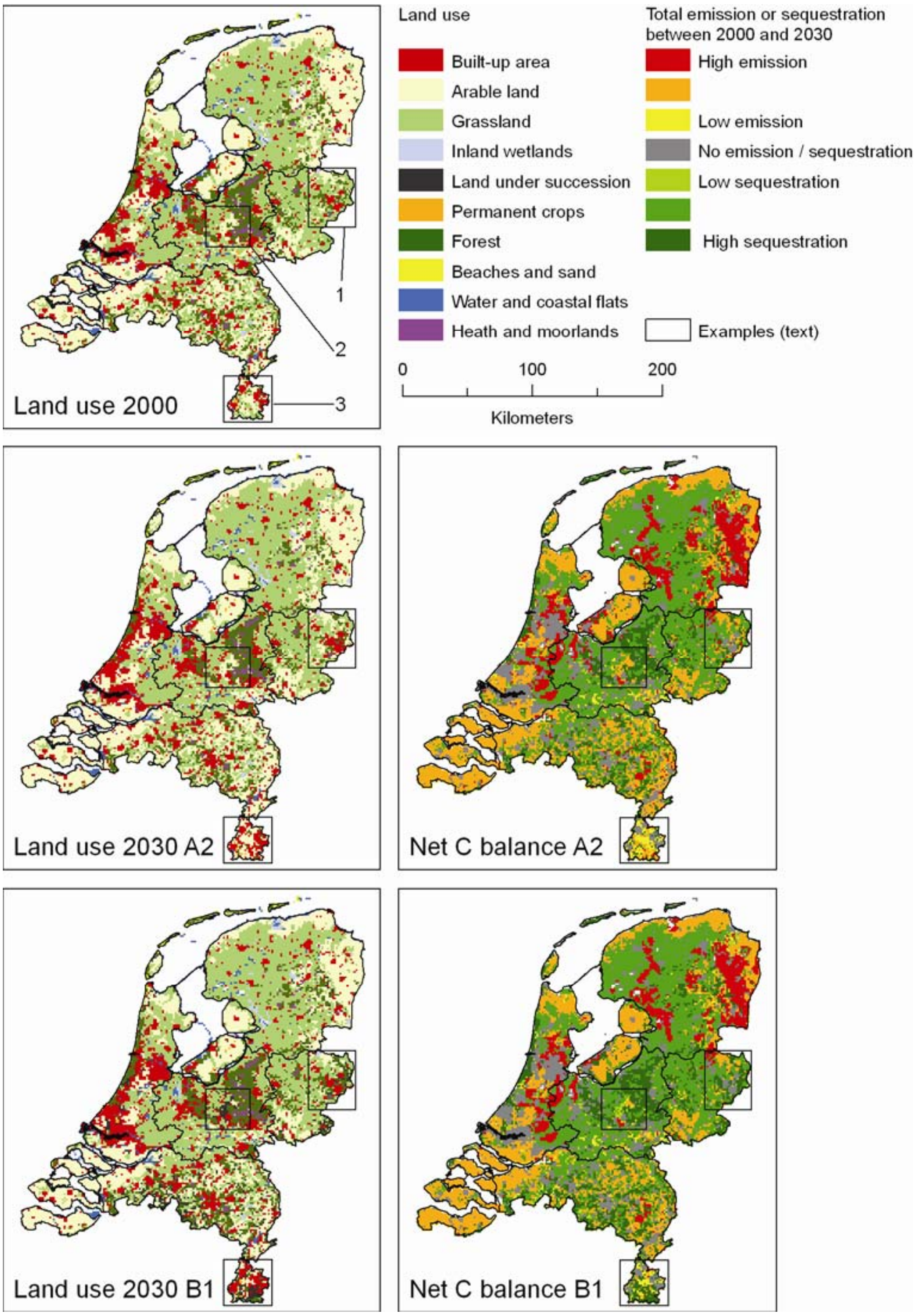
To model the effect of future LUC on carbon stocks, a LUC model and a carbon model were coupled. Such coupling is difficult due to different spatial and temporal resolutions of available LUC and carbon models. The most detailed LUC models for the European extent currently available work on spatial resolutions varying between one km^2 and 10 minutes (approx. 200 km^2) (Rounsevell et al., 2005; Verburg et al., 2006a), temporal resolution varies between one and 30 years. Carbon models are typically process-based point or stand models which are parameterized and calibrated on a site-specific basis and work with monthly or weekly time steps (Parton et al., 1987; Coleman and Jenkinson, 1999). Such process-based models have high data requirements to parameterize all processes involved. At plot scale, carbon sequestration depends on, among others, soil conditions, moisture and the decomposition rate of organic matter input. These parameters typically have a high spatial variation and non-linear responses potentially leading to scaling errors if average values for larger regions are used (Dendoncker et al., 2008).

National or continental scale carbon assessments simulate carbon stock changes with process models for polygons or grids that are assumed to be homogenous with respect to the parameters of the carbon model. This is, e.g., done in the GEFSOC system (Milne et al., 2007) where the RothC and Century model are used for national and sub-national assessment of soil carbon stocks, and in several other studies (Vleeshouwers and Verhagen, 2002; Smith et al., 2005a; Smith et al., 2005c; Falloon et al., 2006). If, however, these polygons or grids have a large spatial extent, the assumption of homogeneity does no longer apply. This causes inaccuracies in the aggregated estimates, especially if non-linearity's in the model response are involved (Post et al., 2001; Falloon et al., 2006; Dendoncker et al., 2008). At European scale, LUC is the carbon sequestration influencing factor with the quickest changes and the strongest effect (Smith et al., 2005a). At this scale it is therefore important to be able to account for differences in LUC trajectories.

Most assessments that address the effect of LUC on carbon sequestration strongly simplify the dynamics of LUC (Smith et al., 2005a; Smith et al., 2005c; Falloon et al., 2006; Al-Adamat et al., 2007) and put most emphasis on understanding the carbon sequestration behaviour of one land use type separately under changing management or changing climate. The attention for the dynamics of LUC trajectories is limited. In this study an approach was chosen that links LUC and carbon models at approximately the same level of complexity and makes best use of the available data. Disaggregation of the country-specific EF's using forest age and SOC content upgrades the EF's, reflecting other soil properties like texture and soil depth.



Colour plate 5a. *Total sequestered (green) and emitted (yellow-red) carbon (ton C km^{-2}) between 2000 and 2030 in each scenario.*



Colour plate 5b. *LUC between 2000 and 2030 for the A2 and B1 scenarios in the Netherlands and net carbon balance between 2000 and 2030.*

The combined LUC and carbon modelling avoids some of the problems with process-model based approaches as described above. However, this does not mean that the chosen methods are without problems. First, there is large variation in estimates of present-day carbon sequestration. Present-day carbon sequestration is estimated at 96 Mton C yr^{-1} by applying the EF's from Janssens et al. (2005) to the 2000 land use map for the EU27. Zaehle et al. (2007) report a carbon uptake by land of 29 Mton C yr^{-1} for the EU15 plus Norway and Switzerland. Also for land use types separately estimates of sequestration / emission rate are very different. Vleeshouwers and Verhagen (2002) estimate pastures in Europe to sequester 52 ton C km^{-2} yr^{-1} and croplands to emit 84 ton C km^{-2} yr^{-1} while Smith et al. (2005a) estimate pastures to sequester 11 ton C km^{-2} yr^{-1} and croplands to emit 11 ton C km^{-2} yr^{-1}. Due to large uncertainty in the EF's (§5.3.5), uncertainty in the results is high, especially when compared to the differences in sequestration between the scenarios. This stresses the importance of reducing uncertainties of the EF's.

Differences in LUC trajectories and timing cause differences in carbon sequestration (§5.3.4). Therefore it is important to be able to account for these differences. The approach presented in this chapter can account for the spatial and temporal dynamics of LUC at a detailed scale. This provides the possibility to account for effects of different land use transitions on carbon stocks.
Future LUC is however uncertain. The scenarios represent relatively large differences and it is likely that LUC is developing within the range of options indicated. Still the LUC model itself is a source of uncertainty. Validation of a high-resolution LUC model for the EU27 extent is difficult due to lack of consistent high resolution historical land use data for the entire EU27. Further uncertainties related to LUC comprise the utilization of recently abandoned agricultural land and the behaviour land use types for which no EF was available. As the extent of these land use types is limited, this is not expected to have large effects on carbon sequestration.

A third group of uncertainties results from model assumptions. EF's are assumed to be constant and upon LUC, the EF's are assumed to change immediately. In reality, SOC stocks start to change to a new equilibrium after LUC. Immediately after conversion this is a fast process which levels off when the SOC stocks approach equilibrium (Freibauer et al., 2004). Also the possibility of sink saturation is not taken into account in this study. Saturation is assumed to be possible from approximately 20 years after conversion onwards (Freibauer et al., 2004). As the EF's are based on large sets of data and model outputs representing the present-day situation, saturated conditions are partly accounted for in the EF's. Further, forests in Europe are relatively young and in the phase of fast biomass enhancement (Janssens et al., 2005). Sink saturation in forests is therefore not expected during the timeframe considered.

Improvement of the approach could be achieved with the use of spatial variation in factors determining carbon emissions. This requires a further refinement of the EF's or the use of process-based models. EF's that are not only country-specific and land use type specific, but also depend on biophysical characteristics like soil texture, drainage and soil depth, could potentially be combined with the detailed assessment of LUC dynamics. Information on these factors at a high level of spatial detail is however lacking.

5.4.2. Comparison with other carbon sequestration assessments

Methodology comparison
Several studies have made an assessment of future carbon sequestration in Europe. Smith et al. (2005a) and Smith et al. (2005c) studied future SOC stock changes of pasture and cropland, and forest respectively. For agricultural land, effects of climate change and technology change are assessed while for forest the effect of climate change and change in litter input are assessed. In both studies the effect of LUC is assessed: SOC stocks per land use type resulting from climate change and technology change and litter input respectively per time step are multiplied by areas per time step. LUC scenarios from Rounsevell et al. (2005) are used. These LUC scenarios are developed for the EU15 plus Norway and Switzerland and are extrapolated to the EU25 plus Norway and Switzerland. Only effects on SOC stocks are calculated.
Zaehle et al. (2007) modelled effects of future changes in climate, management and LUC on the terrestrial carbon flux between 2000 and 2030 in the EU15 plus Norway and Switzerland using LUC scenarios by Rounsevell et al. (2005). Per time step, the land-atmosphere flux is calculated using the LPJ-DGVM model. In this study the effect of LUC is integrated in the terrestrial carbon balance model.

Results of different studies on future carbon sequestration and land use change are hard to compare for several reasons. Studies differ in the initial terrestrial carbon flux in 2000 (§5.4.1) and in temporal extent and resolution. Other difficulties arise from the scenarios used: The scenario elaboration differs and the studies named above use scenarios that are developed for the EU15 plus Norway and Switzerland but Smith et al. (2005c) and Smith et al. (2005a) apply them on the EU25. As shown in §5.3.1, the EU12 is expected to react differently on driving factors of LUC and this may cause differences in future LUC trends. Further, studies differ in the carbon stocks they assess. Smith et al. (2005c) report SOC stock changes only for forests while this chapter reports NBP changes. Finally, studies differ in the driving factors they assess. This chapter only assesses effects of LUC while Smith et al. (2005c), Smith et al. (2005a) and Zaehle et al. (2007) also assess the effects of climate change, NPP change, technology change and management change.

Results comparison
Carbon stock changes from (Smith et al., 2005a) are recalculated to total emission or sequestration between 2000 and 2030 resulting from LUC only (Table 5.5). Spatial patterns of sinks and sources cannot be compared because Smith et al. (2005a) only report spatial patterns of sequestration / emission between 1990 and 2080.

Table 5.5. *Total emission (negative) or sequestration (positive) (Mton) between 2000 and 2030 by pasture and cropland in this study and compared with Smith et al. (2005a).*

Scenario	Study		Study	
	Pasture sequestration (Mton)		*Cropland emission (Mton)*	
	This study	**Smith et al.**	**This study**	**Smith et al.**
A1	183	199	-424	-313
A2	211	198	-436	-293
B1	189	244	-426	-335
B2	220	203	-427	-351

When incorporating only effects of LUC, for pastures, sequestration of carbon between 2000 and 2030 in this study is in the same order of magnitude as found by Smith et al. (2005a). Highest sequestration is expected in B2 while in Smith et al. the B1 scenario has the highest carbon sequestration. This is because of differences in scenario elaboration and extrapolation of EU15 trends to Europe as a whole in Smith et al. (2005a). Further deviations are because sequestration of pastures per km^2 is assumed to be slightly higher according to Janssens et al. (2005).
For croplands, 20-50% higher emissions are expected compared to Smith et al. (2005a). Main reason is that Smith et al. (2005a) expect cropland area to decrease by -24 to -41% while in this study decreases up to -12% are expected. Differences in cropland area decrease are partly because of extrapolation of EU15 trends to Europe as a whole by Smith et al. (2005a) while an increase of cropland areas is expected in this study in the EU12, with a smaller net decrease as a result. Further deviations in both pasture and cropland carbon stock changes can be caused by differences in area, because Smith et al. (2005a) do not take into account organic soils. Finally, as is demonstrated by Dendoncker et al. (2008), spatial scale can strongly influence SOC stock change modelling results.

Zaehle et al. (2007) model a net land-atmosphere flux of -29 Mton C yr^{-1} (uptake) for the 1990's, consisting of losses from urbanization (3.3 Mton C yr^{-1}), agriculture (19.3 Mton C yr^{-1}) and pastures (14.5 Mton C yr^{-1}) and carbon uptake by forests and wood products of 59.1 ± 31.4 Mton C yr^{-1} for the EU15 plus Norway and Switzerland, based on an analysis with a 10*10 minutes spatial resolution.
Forest carbon sequestration in 2000 in the EU15 in this study is in the same order of magnitude as reported by Zaehle et al. (2007) (69 Mton C yr^{-1}) and differences can be explained by differences in area. Main difference between the initial (2000) carbon budgets is that agricultural land emission is smaller in this study (6.4 Mton C yr^{-1}) and that pastures are expected to sequester instead of emit carbon. Therefore, net carbon sequestration is higher in this study (70 Mton C yr^{-1} uptake).

Temporal trends of carbon sequestration for the B2 scenario are comparable with this study, but for the other scenarios the elaboration of the LUC scenarios is considerably different, resulting in differences in land-atmosphere flux changes. Difference in land-atmosphere flux in 2030 between the most extreme scenarios in Zaehle et al. (2007) is a difference in carbon uptake of around 35 Mton C yr^{-1}, considerably larger than in this chapter (22 Mton C yr^{-1} for the EU15). Also with this study spatial trends are difficult to compare because of differences in time span of the study.

Altogether, studies on the effect of future LUC on carbon sequestration are different in approach, time span, resolution, estimates of terrestrial carbon balance in 2000 and scenario elaboration. Therefore, comparison can only be done in general terms. Cropland emission and pasture sequestration between 2000 and 2030 is in the same order of magnitude as found by Smith et al. (2005a). Compared with Zaehle et al. (2007) results are considerably different, mainly because of basic assumptions in the studies. For an in-depth comparison of results, a study using the same land use dataset and scenarios in different carbon models should be performed.

5.5. Conclusions

Europe's terrestrial biosphere is expected to sequester between 90 and 111 Mton C yr^{-1} in 2030. In three of the four scenarios evaluated net sequestration increases, mainly due to a decrease in cropland area. The highest sequestration enhancement is expected to be 15 Mton C yr^{-1} in the B2 scenario. Clear differences between the scenarios in spatial distribution of sinks and sources, and differences in the change of the sink size illustrate that LUC is an important factor in future carbon sequestration changes.

Uncertainty assessment and comparison with other studies revealed that the estimates have a high level of uncertainty. Further refinements in data and emission factors, and validation and comparison studies are needed to increase the certainty of the estimates. However, the study has indicated the importance of LUC as a main driving factor of carbon sequestration and the need to account for the effects of LUC as a potential source or sink of carbon. The high carbon losses upon deforestation of old forest and potential for carbon sequestration upon cropland abandonment stress the importance of carefully assessing ongoing and future land use changes.

This is one of the first Europe-wide assessments of carbon stock changes fully accounting for the dynamics of LUC. The method makes most consistent use of available data at the scale of analysis and is able to take into account the role of soils and forest age in carbon sequestration and can make a contribution to continental scale assessments and inventories of future carbon sequestration potentials under a wide range of LUC conditions.

Chapter 6

Upscaling landscape-scale knowledge to improve national-scale carbon stock inventories

Long-term land use is, in local studies, often identified as a determinant for spatial variability of soil organic carbon (SOC) stocks and forest floor carbon (FFC) stocks. Quantification of the effects of long-term land use is, however, never used to explain variability of carbon stocks in national-scale inventories. This chapter assesses if knowledge on the impact of long-term land use on SOC and FFC stocks can improve national-scale inventories of SOC and FFC in the Dutch sand area. Determinants for SOC and FFC stocks derived from landscape-scale studies were used to map national-scale spatial variability and to calculate national totals. Calculations were validated, and the resulting national-scale spatial distribution was compared with the SOC stock map from the Dutch greenhouse gas inventory.

Using long-term land use to explain SOC stocks decreased the error of the SOC stock estimate in 60% of the area as compared to the current Dutch greenhouse gas inventory. The error in FFC stocks decreased in half the forest area. Increasing the sample density in agricultural land is only expected to decrease errors of SOC stocks when the sampling strategy accounts for the effects of long-term land use.

Considering the increasing availability of data on long-term land use, the approach presented in this chapter is applicable to improve the Dutch SOC and FFC inventory as well as inventories in other countries.

Based on: Schulp, C.J.E. P.H. Verburg, P. Kuikman, G.J. Nabuurs,
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Global Change Biology, submitted

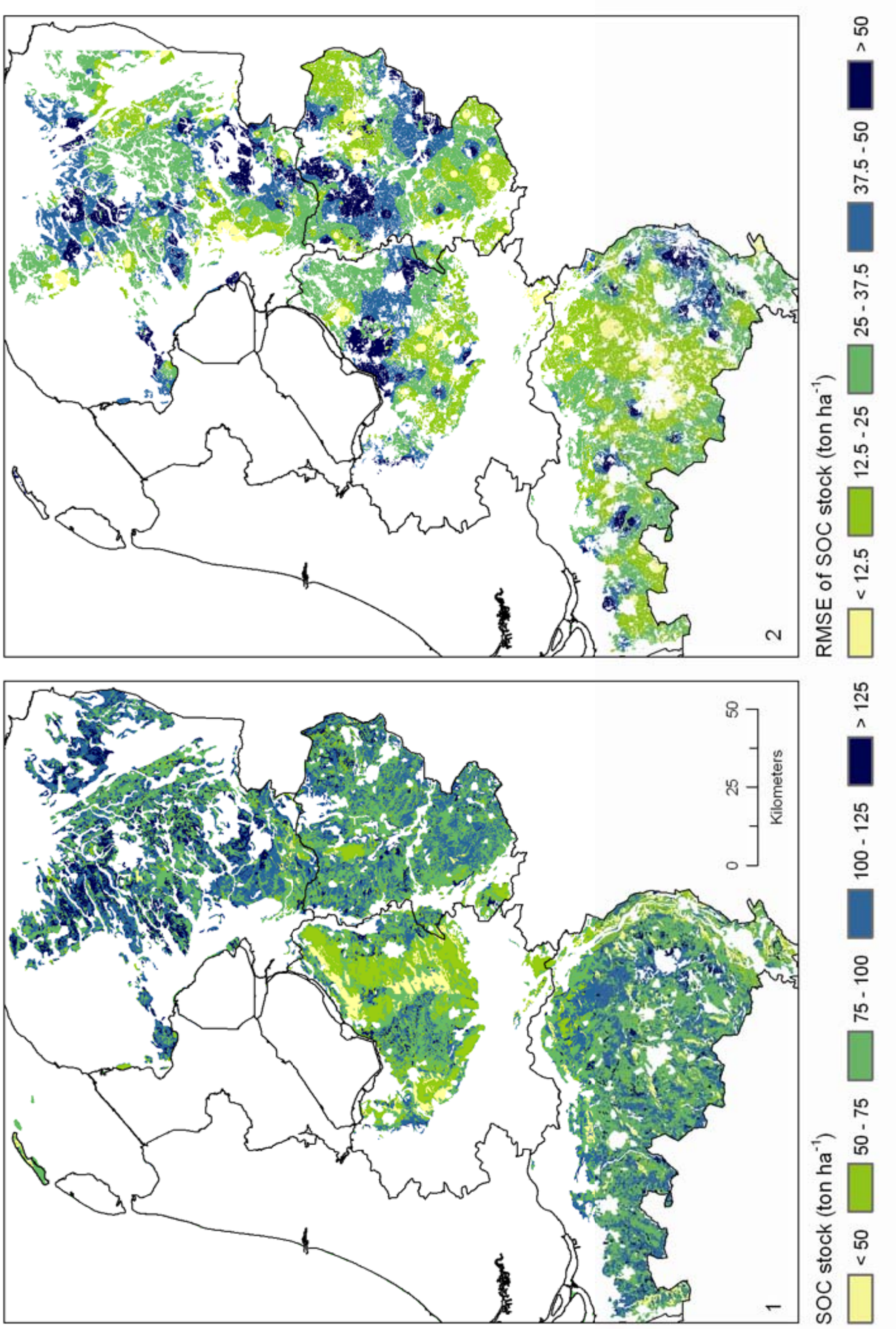
6.1. Introduction

Worldwide, there is an increasing interest in functioning and spatial variability of soil organic carbon (SOC) and forest floor carbon (FFC) stocks. The impact of several factors influencing SOC and FFC dynamics has been assessed, mainly at the scale of small sites. One of the important determinants indicated is land management, both in agriculture and in forests (Dendoncker et al., 2004; Vesterdal et al., 2008; Hanegraaf et al., 2009; Johnson et al., 2009). Also the history of land use, up to centuries ago, is an important determinant for SOC patterns in local studies (Verheyen et al., 1999; Sonneveld et al., 2002; Hupy and Schaetzl, 2008; Mueller and Kögel-Knabner, 2009). Dynamics and stocks of FFC at the local scale are influenced by site conditions, management, tree species and forest age (Thuille and Schulze, 2006; Jandl et al., 2007; Vesterdal et al., 2008).

Despite the increasing insight in the determinants of SOC and FFC variability, uncertainties in national-scale inventories are high. In the European Union (EU) for example, uncertainty in the greenhouse gas budget of the Land use, Land Use Change and Forestry (LULUCF) sector is estimated at 41%. Uncertainty in SOC and FFC dynamics is an important contributor to this uncertainty (EEA, 2009).

Not all information on carbon stocks and dynamics obtained in local studies is currently used in national-scale inventories of carbon stocks. A possible reason for this is the upscaling required for national-scale inventories, which makes it difficult to include all factors identified in local studies. In many countries the SOC inventory is based on a national-scale sampling, stratified by soil classification and land use type (Arrouays et al., 2001; Rodriguez-Murillo, 2001; Lettens et al., 2004; Bradley et al., 2005). Soil and land use classification in these inventories is often distinguishing the main FAO soil classes and the land use types used in Kyoto reporting (forest, cropland, grassland, wetland, settlements and other land, (IPCC, 2006)). This upscaling approach results in a correct representation of SOC variability if the strata are homogenous at all scales. In soil (King et al., 1994) and land use data (Nol et al., 2008) at larger spatial scale strata are, however, never homogenous, resulting in uncertainties (Verburg et al., 2006b).

An other upscaling approach describes SOC variability by a regression model parameterized at small scale, and applies this regression model at larger scale. This is e.g. done by Meersmans et al. (2008) for Flanders (Belgium). This upscaling approach also has several drawbacks. First, at different scales, different processes are determining SOC variability. Generally, parent material and climate are assumed to define large-scale patterns of SOC stocks, while other determinants, like topography, groundwater dynamics, land use and management, define the SOC variability at smaller scales (McLauchlan, 2006). When estimating a regression model, only the dominant processes resulting in SOC variability are considered. Processes dominant at another scale can be excluded from the model because they are not relevant at the scale of analysis. Second, local studies are never representative for the national scale because they do not cover the variability in determinants for SOC exhibited at national scale.



Colour plate 6a. *Spatial distribution of SOC stocks (1) and related RMSE (2) (ton ha^{-1}) in the Dutch sand area as used in the National Inventory Report for greenhouse gas emissions (based on (Kuikman et al., 2003; De Groot et al., 2005)).*

Third, input data availability is generally smaller at larger scales. Therefore there is often the need to simplify models when applying them at larger scale. This can result in aggregation errors (Heuvelink, 1998; Verburg et al., 2006b). Finally, relations between determinants and SOC stocks are often non-linear. Linearization of relations in aggregated data can lead to errors as well (Easterling, 1997).

To summarize, upscaling of data from local studies to national scale has many limitations. Knowledge on functioning of SOC and FFC stocks provided through local studies is nonetheless expected to be applicable at others scales and is expected to improve national-scale inventories. The aim of this chapter is therefore to assess how knowledge of determinants for carbon stocks obtained at local scale can improve national-scale carbon inventories.

6.2. Methods

6.2.1. Background

The Netherlands is the only EU country where the LULUCF sector is a net source of CO_2 emissions (EEA, 2009). The net source is 2.3 Mton CO_2 and contributes 1.1% to the total Dutch greenhouse gas budget. Forests sequester 2.5 Mton CO_2 annually while grasslands emit 4.2 Mton CO_2 (Brandes et al., 2007). Several categories of the LULUCF sector are identified as a key source because of their high uncertainty. The Dutch National Inventory Report of greenhouse gas emissions (NIR) estimates the uncertainty of SOC stock changes in mineral soils at 38% (Brandes et al., 2007).

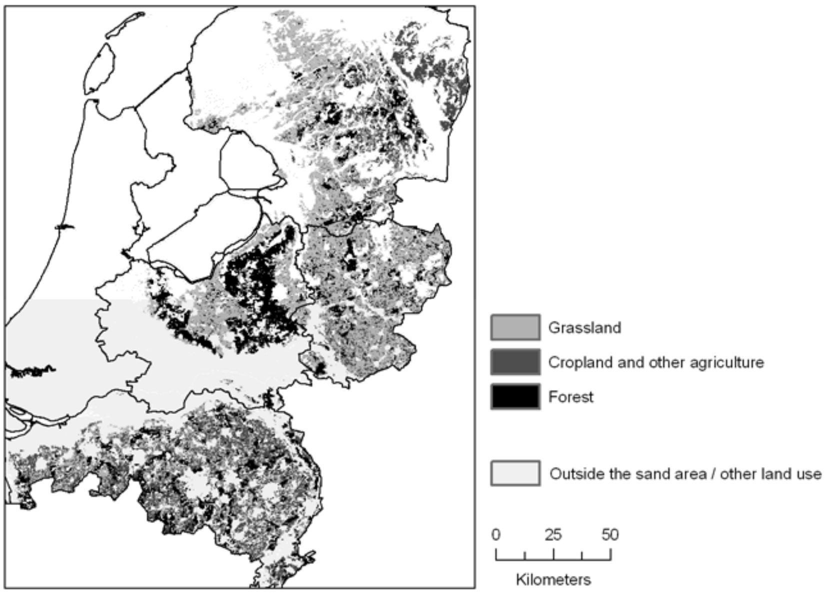


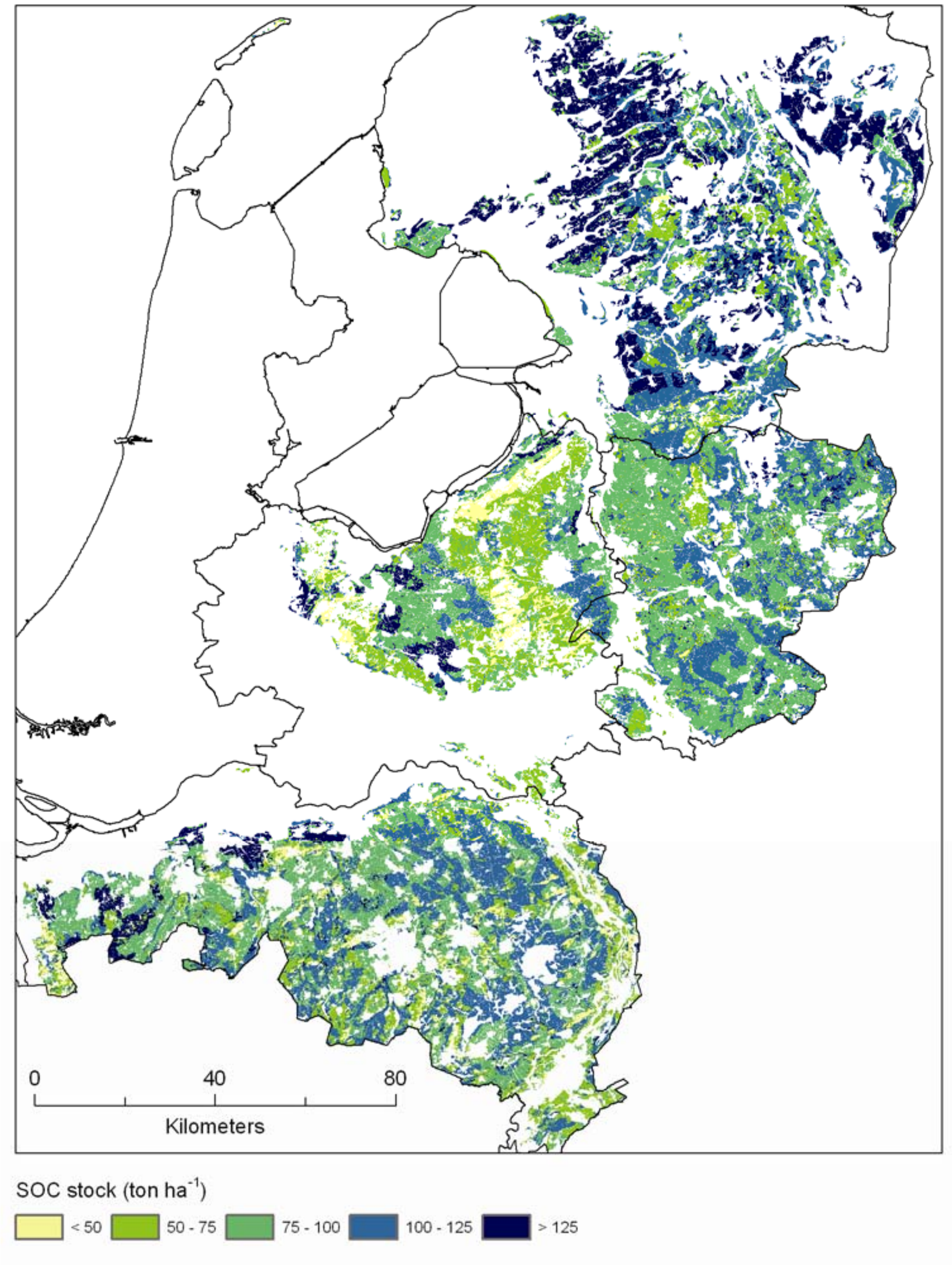
Fig. 6.1. *Simplified land use map of the Dutch sand area.*

Spatial variability of 0-30 cm SOC stocks and FFC stocks was assessed in the Dutch sand area. Sandy soils in the Netherlands cover 14536 km^2 (De Vries et al., 2003), 43% of the land area. Of the Dutch forests, 73% (2613 km^2) is in the sand area (Fig. 6.1) and therefore the sand area is an important sink for CO_2. Forests are relatively young with a dominant age of 40-60 years (LNV Directie Kennis, 2007) and are mainly afforestations of heathlands and drift sand areas. About half of the forests are coniferous. Agriculture occupies 8767 km^2 of the sand area and is dominated by intensive livestock keeping with strongly manured grasslands and silage maize cultivation (Fig. 6.1). Large areas (40% of the agricultural sand area) have been used for agriculture since the early 19th century or longer, 41% consists of late-19th century large-scale heath reclamations.

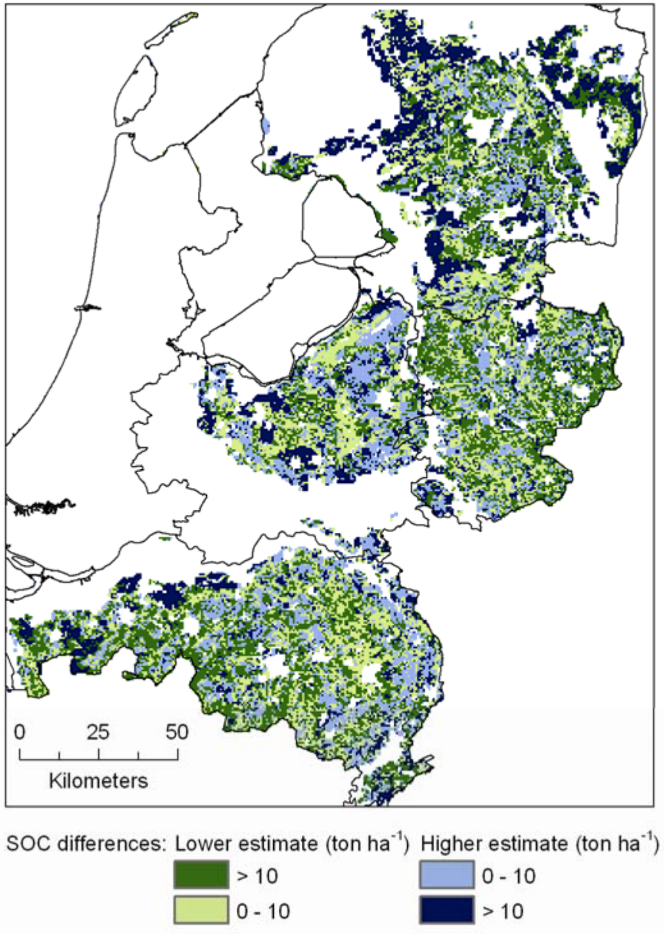
As a reference, the SOC inventory from the Dutch NIR (2005) was used. This SOC inventory is based on the National Soil Sampling Programme (NSSP) where 1392 points were sampled in 1994-2001 (Finke et al., 2002; Visschers et al., 2007). Data were collected following a stratified sampling. Stratification was first done by groundwater class. Some groundwater classes were stratified by soil type, in the other groundwater classes a subdivision between nature and agriculture was made. No further stratification was applied in the nature class while the agricultural land was then stratified by soil type. At all sample points, SOC stocks were determined. The mean SOC stock over all points in each groundwater class-soil type combination was calculated and was assumed to be representative for all mapping units with the same groundwater class-soil type combination (Kuikman et al., 2003; De Groot et al., 2005) (Colour plate 6a.1). Spatial distribution of FFC stocks across the Netherlands is unknown. The NIR uses a mean of 37 ton ha^{-1} (Nabuurs et al., 2005), based on measurements described by De Vries and Leeters (2001). Because the spatial distribution of FFC stocks is not known, the impact of future land use changes on FFC stocks cannot be properly included in the LULUCF greenhouse gas budget. This limitation is criticized in the international review of the Dutch NIR.

6.2.2. Landscape-scale determinants for SOC and FFC stocks

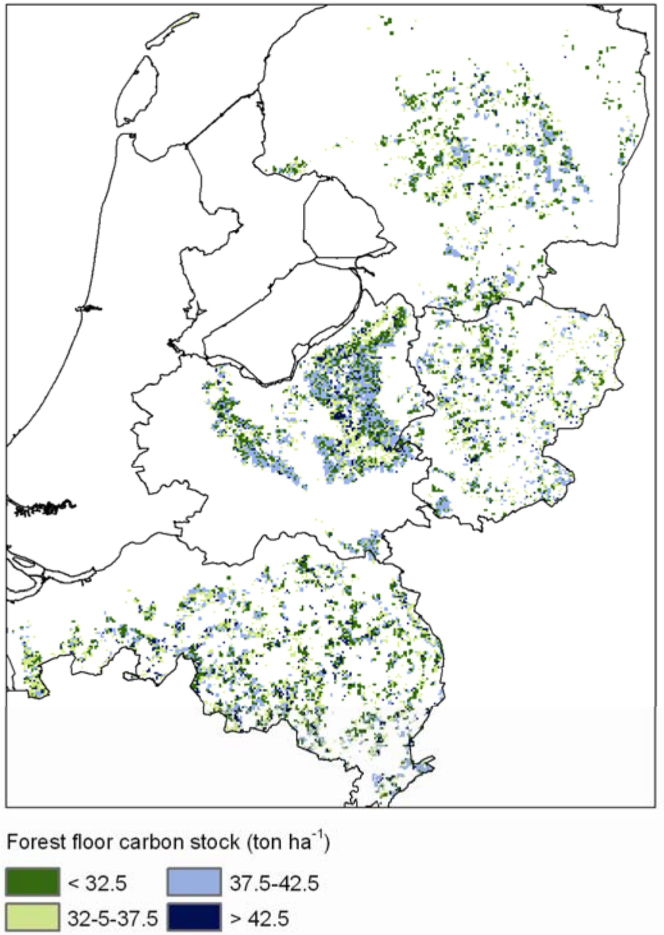
In landscape-scale case studies, determinants for SOC and FFC variability in the Netherlands were identified (Fig. 6.2). The effect of tree species and management intensity was assessed in a study with six even-aged monospecific forest stands on fertile sandy soils in the centre of the Netherlands (Schulp et al., 2008b). SOC stocks significantly differed between stands with different tree species, and broadleaf stands had lower SOC stocks than conifer stands. SOC stocks were lower at unmanaged locations than at managed locations. Tree species, tree group (conifer vs. broadleaf) and management intensity explained part of the FFC variability (Schulp et al., 2008b). A similar case study with identical sampling strategy was done in forest stands on infertile sandy soils in the centre of the Netherlands. Comparison of these two case studies showed the importance of soil fertility and the interaction between soil fertility and tree species for explaining SOC and FFC variability (Van den Wyngaert, 2009).



Colour plate 6b. *SOC stocks (ton ha^{-1}) in the Dutch sand area using a soil-aggregated + reclamation type stratification for agriculture and a NIR-aggregated + tree group stratification for forest.*



Colour plate 6c. *Difference between the NIR SOC map (Colour plate 6a) and the SOC map presented in this study (Colour plate 6b).*



Colour plate 6d. *FFC stocks (ton ha^{-1}) in the Dutch sand area, using a tree species + age group + soil fertility stratification.*

For agricultural land, land use history was identified as an important determinant for SOC variability. In one case study, areas that have been used for agriculture since the early 19th century or longer had significantly higher SOC contents than late-19th-century and 20th-century heath reclamations (Schulp and Veldkamp, 2008). Similar effects were found in three other case studies across the Dutch sand area. The causality of these empirical results was tested in a sensitivity analysis (Fig. 4.5). The association of several other determinants with SOC and the scale sensitivity of the determinants was assessed, indicating that at landscape scale the effect of present-day land use on SOC variability is limited. Soil and geomorphology were important for explaining SOC variability both within case studies and across the physiographic region (Table 4.5). At a large extent or a coarser resolution, present-day land use and management did explain part of the SOC variability (Schulp and Veldkamp, 2008; Schulp and Verburg, 2009).

6.2.3. National-scale SOC and FFC stock variability

The National Soil Sampling Programme (NSSP) dataset (Finke et al., 2002; Visschers et al., 2007) was used to assess national-scale SOC variability. The NSSP dataset was subdivided into sample datasets for agriculture (N=404) and forest (N=78) based on the 2005 land use map (Table 6.1). National-scale SOC variability was assessed for both forest and agriculture separately in six steps (Fig. 6.2):

1. In case studies, determinants for SOC variability at landscape scale were identified. Additionally, the causality behind the associations between the determinants and the SOC variability was assessed and scale sensitivity of the determinants was explored (§6.2.2). Relevance of these potential determinants at national scale was evaluated (Fig. 6.2). This evaluation was made by an ANOVA analysis, using the sample dataset and national-scale data on the spatial distribution of potential determinants (Table 6.1).
2. The determinants that significantly explained part of the SOC variability in the sample dataset were used as possible stratification options of the sample dataset (Fig. 6.2). A map of the spatial distribution of the strata of each determinant or combination of determinants was coupled to the sample dataset to stratify the sample points.
3. For each stratification option, the mean SOC stock of each stratum was calculated (§6.2.4). This mean value per stratum was assumed to be a representative SOC stock for all mapping units in the stratum.
4. Spatial distribution of SOC stocks was mapped for each stratification option by coupling the stratum mean values calculated in step 3 to the mapping units of the maps of the spatial distribution of the strata. This procedure was repeated for each determinant or combination of determinants.
5. For each stratification option, root mean square errors (RMSE) for each sample point were calculated using the observed SOC stock and the stratum mean calculated in step 3. The procedure for calculating RMSE's is described in §6.2.4.

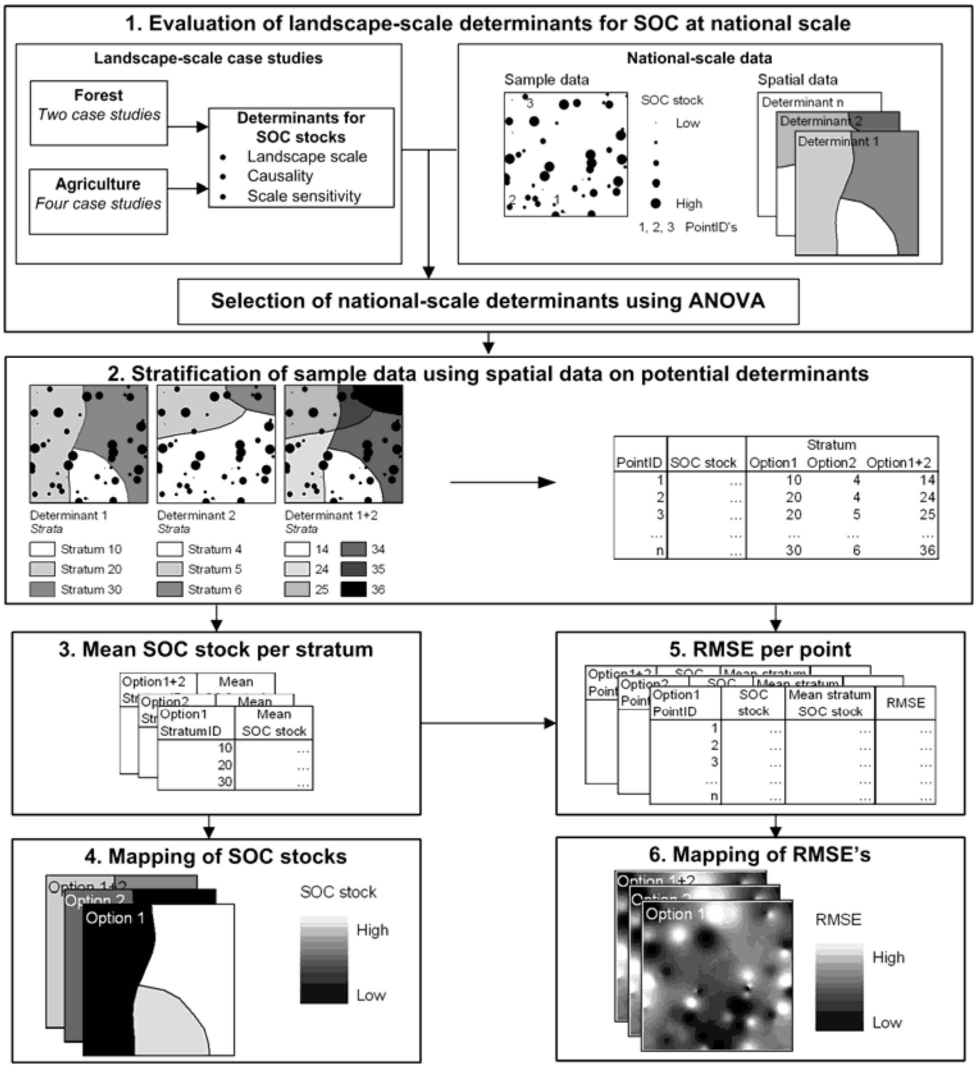


Fig. 6.2. Overview of the methodology for assessing national-scale SOC variability.

6. Spatial distribution of the RMSE's was mapped for each stratification option by interpolating the RMSE's per point using inverse distance weighting.

The potential determinants assessed include:

- The groundwater class-soil type stratification currently used in the Dutch NIR;
- Potential determinants identified in the case studies, including relevant combinations of determinants (Fig. 6.2);
- An aggregated version of the NIR stratification and soil stratification in which strata with similar SOC stocks were aggregated (§6.2.5).

Table 6.1. *Overview of the sample data and the spatial data that are used as stratification options.*

Abbreviation	Explanation	Source
	Sample data: SOC and FFC stocks	
NSSP	National Soil Sampling Programme. Approx. 500 sampling points across the sand area where 0-30 cm SOC stocks are determined.	(Finke et al., 2002; Visschers et al., 2007)
FFC stock data	140 data points where FFC stocks in the sand area are determined under several tree species and on different soil types.	(De Vries and Leeters, 2001)
	Spatial data: Potential determinants for SOC and FFC variability	
NIR	Stratification by groundwater class and soil type.	(De Vries et al., 2003), (Finke et al., 2002), (Kuikman et al., 2003)
Soil type	Spatial distribution of soil types across the Netherlands, clustered into 21 groups of pedologically similar mapping units	(De Vries et al., 2003; Visschers et al., 2007)
Soil fertility	Soil type map reclassified to distinguish fertile and infertile sand soils only.	(De Vries et al., 2003)
Groundwater class	Classification of groundwater dynamics into eight classes based on average highest groundwater level and average deepest groundwater level (Table 4.2).	(Finke et al., 2002)
Geomorphology	Classification of geomorphological genesis (eolian, fluvial, glacial, anthropogenic). The glacial class is split-up further in ice-pushed ridges, fluvioperiglacial areas, glacial till.	(Koomen and Maas, 2004)
Reclamation type	Map indicating reclamation types, based on land cover before reclamation and age and characteristics of reclamation patterns.	(De Bont, 2004)
Land use	Land use in 2005. Agricultural land use is subdivided into grassland, maize, sugar beets, cereals and potatoes.	(Hazeu, 2005)
Tree species Age class	Map from the Fourth Dutch forest statistics. Contains forest stand delineations, and tree species and regeneration year at stand level.	(Clement, 2001)
Tree group	Tree species map reclassified to conifers and broadleafs.	(Clement, 2001)
	Spatial data: Evaluation of the NIR stratification	
NIR-aggregated	Stratification by groundwater class and soil type, aggregated by joining strata with similar SOC stocks.	This study
Soil-aggregated	Stratification by soil type, aggregated by joining strata with similar SOC stocks.	This study

National level total SOC stocks and associated errors were calculated using the mean SOC stocks of each stratum, the RMSE's (§6.2.4) and area covered by each stratum.
To assess national-scale variability of FFC stocks, the same approach was followed using sample data with FFC stocks across the Dutch sand area (N=140) from a study by De Vries and Leeters (2001) and national-scale spatial data on potential determinants (Table 6.1).

6.2.4. Validation

SOC and FFC maps resulting from each stratification option were validated (Fig. 6.2). Because of the relatively small sample sizes, it was not feasible to exclude points from the analysis to create independent validation datasets. Therefore, validation was done based on a jackknife method (Efron and Gong, 1983). In this method, a sample of N-1 points was taken. With this sample, stratum means were calculated (Fig. 6.2). If a stratum only contained one sample point, the stratum mean was replaced by the

overall mean. The stratum means were validated by calculating an RMSE for the excluded point. The procedure was repeated with replacement until each point was used for validation once. Finally, a mean RMSE over all sample points was calculated for each stratification option.
RMSE maps for each stratification option (Fig. 6.2) were compared with the RMSE map of the NIR SOC map (Colour plate 6a.2) or an RMSE map of the FFC stock from the national reporting system for forest (Nabuurs et al., 2005).

6.2.5. Evaluation of the NIR carbon maps

The aim of the NSSP was to quantify information regarding a wide range of soil quality parameters for mapping units for the 1:50,000 soil map of the Netherlands (De Vries et al., 2003). Because the SOC stock was not the prime variable of interest, the grouping of soil types used for stratification in the NSSP might not be optimal for explaining national-scale SOC variability. To evaluate this, strata with significantly different SOC stocks were identified in the ANOVA analysis (§6.2.3). Then, groups of strata that did not have significantly different SOC stocks were united. The aggregation was evaluated by testing if coefficients of determination showed a significant decrease upon aggregation, using an F test. A significance level of $p<0.05$ was used throughout.

The adequacy of sample size of the agriculture SOC sample dataset and the FFC stock sample dataset was evaluated by assessing the changes of the RMSE when samples were excluded from the analysis. Random sub-samples of 10%-20%-...90% of the total datasets were taken. For each sample percentage, ten different random samples were taken, resulting in 90 datasets for the agricultural land SOC stock and 90 datasets for the FFC stock. For each of the 180 datasets the total SOC stock was calculated and validated. This was done for the NIR stratification and one other stratification option, using the methods described in §6.2.3 and §6.2.4. For each sample percentage, the mean RMSE's and standard deviations over all ten datasets were calculated.

6.3. Results

6.3.1. National-scale determinants for SOC and FFC

Evaluation of landscape-scale determinants of SOC variability in the national-scale sample dataset indicated that for agriculture, the combination of soil type and reclamation type explained a larger part of the SOC variability (R^2 = 0.38) than the stratification currently used in the NIR (R^2 = 0.35) (Table 6.2). Soil type or a combination of geomorphology and reclamation type explained around 30% of SOC variability. Present-day land use (the agricultural crop type) (R^2 = 0.04) and groundwater class (R^2 = 0.05) hardly explained SOC variability.

Table 6.2. *Coefficients of determination of SOC and FFC stocks for different stratification options.*

Stratification	R^2 Mineral topsoil Agriculture	R^2 Mineral topsoil Forest	R^2 Forest floor Forest
NIR	35%	39%	
Soil type	26%	28%	1%
Soil fertility			0%
Groundwater class	5%	25%	11%
Geomorphology	21%		
Reclamation type	16%		
Land use	4%		
Tree species		29%	17%
Tree group		13%	0%
Age class			26%
Soil + Reclamation type	38%		
Geomorphology + Reclamation type	30%		
NIR + Tree group		38%	
Soil + Tree group		27%	
Tree species + age class			45%
Tree species + soil fertility			21%
Tree species + soil fertility + age class			59%
NIR-aggregated	21%	40%	
Soil-aggregated	23%		
Soil-aggregated + reclamation type	31%		
NIR-aggregated + reclamation type	30%		
NIR-aggregated + tree group		43%	

For forests, none of the landscape-scale determinants explained more SOC variability in the national-scale sample dataset than the NIR stratification (R^2 = 0.39). Stratifying by groundwater class and soil type (NIR) combined with tree group was second best (R^2 = 0.38) and the other stratification options explained around 30% of SOC variability (Table 6.2).

Tree species (R^2 = 0.17) and age class (R^2 = 0.26) explained part of the FFC variability. Soil type and soil fertility only explained the FFC variability when combined with tree species and age class.

In the NIR stratification and the soil stratification many strata did not have significantly different SOC stocks in both agriculture and forest. This may be caused by either the similarity of the strata or by lack of sufficient sample data. Therefore, distinguishing these different strata is not useful. Aggregating NIR strata with similar SOC stocks for forest significantly increased the R^2 to 0.40. A combination of the aggregated NIR stratification with tree group explained 43% of SOC variability. For agriculture, aggregating strata significantly decreased the R^2 to 0.21 (NIR) and 0.23 (Soil type) (Table 6.2).

Table 6.3. *Total mean (error) SOC and FFC stocks (Mton) in the Dutch sand area under different stratification options. Errors are one standard deviation.*

Stratification	Total mean (error) SOC stock (Mton)		Total mean (error) FFC stock (Mton)
	Agriculture	*Forest*	*Forest*
NIR	92.2 (27.6)	20.3 (6.8)	11.7 (4.6)
Soil type	91.9 (27.1)	19.7 (7.7)	9.7 (3.9)
Soil fertility			9.4 (3.6)
Geomorphology	91.4 (28.7)	22.1 (9.9)	
Reclamation type	90.7 (28.5)		
Tree species		18.8 (7.4)	13.0 (5.1)
Tree group		18.2 (8.0)	11.7 (5.1)
Age class			14.1 (5.4)
Soil + Reclamation type	90.3 (26.4)		
NIR + Tree group		21.1 (7.5)	
Soil + Tree group		20.5 (8.1)	
Tree species + age class			9.2 (3.4)
Tree species + soil fertility			8.8 (3.1)
Tree species + soil fertility + age group			8.9 (3.2)
NIR-aggregated		20.4 (7.6)	
Soil-aggregated + Reclamation type	89.7 (26.3)		
NIR-aggregated + Reclamation type	89.4 (26.6)		
NIR-aggregated + Tree group		16.1 (6.2)	

6.3.2. National totals of SOC and FFC stocks

Carbon stocks in the mineral topsoil

The total SOC stock of the Dutch sand area was estimated at 92.2 (±27.6) Mton in the agricultural area and 20.3 (±6.8) Mton in forests based on the NIR (Table 6.3).

For agriculture, the largest decrease in the error of the total SOC stock was achieved using the aggregated NIR combined with reclamation type for stratification. Using soil type slightly decreased the error while stratifying by reclamation type only or by geomorphology slightly increased the error. The stratification options resulted in different estimates of the total mean SOC stock; estimates vary between 89.4 Mton and 92.2 Mton.

In forests, the error of the total SOC stock decreased by 10% when using the aggregated NIR combined with tree group for stratification. This stratification option estimated the forest SOC stock to be 1.3 Mton lower than currently indicated in the NIR. All other stratification options had a larger error than the NIR estimate.

Forest floor carbon stocks

Total FFC stock in the Dutch sand was estimated at 11.7 (± 4.6) Mton in the NIR (Table 6.3). Stratifying by soil type decreased the error by 15-20% and stratifying by tree species combined with soil fertility decreased the error by 32%. Stratifying by tree species or age group increased the error of the total FFC stock. The options that decreased the error for the total FFC stock also estimated a smaller FFC stock. The tree species-soil fertility stratification had a 25% smaller FFC stock than the NIR.

Table 6.4. *RMSE of spatial variability of SOC and FFC stocks (ton ha^{-1}) under different stratification options and the percentage of the Dutch sand area with lower RMSE's compared to the NIR.*

Stratification	Mineral topsoil				Forest floor	
	Agriculture		*Forest*		*Forest*	
	RMSE (ton ha^{-1})	**Area improved** (%)	**RMSE** (ton ha^{-1})	**Area improved** (%)	**RMSE** (ton ha^{-1})	**Area improved** (%)
NIR	33.3		33.3		11.5	
Soil type	32.8	60%	33.8	22%	12.0	18%
Soil fertility					11.5	49%
Geomorphology	33.0	58%	40.2	15%		
Reclamation type	32.7	59%	n.d.			
Tree species			36.1	50%	11.3	60%
Tree group			36.4	42%	11.6	1%
Age class					10.8	65%
Soil + Reclamation type	31.8	56%				
NIR + Tree group			35.0	20%		
Soil + Tree group			36.1	22%		
Tree species + age class					10.7	56%
Tree species + soil fertility					11.2	52%
Tree species + soil fertility + age class					10.3	45%
NIR-aggregated	32.1	66%	30.7	39%		
Soil-aggregated			35.8	47%		
Soil-aggregated + Reclamation type	30.5	66%				
NIR-aggregated + Reclamation type	30.7	65%				
NIR-aggregated + Tree group			31.4	45%		

6.3.3. National-scale spatial variability of SOC and FFC stocks

Carbon stocks in the mineral topsoil

The NIR SOC map had a mean RMSE of 33.3 ton C ha^{-1} (Table 6.4), ranging up to 186 ton C ha^{-1} in agricultural land and up to 146 ton C ha^{-1} in forests (Colour plate 6a.2). In agricultural land the RMSE decreased slightly when stratifying by soil type, reclamation type or geomorphology (Table 6.4). Combining the soil-aggregated classification with reclamation type resulted in the lowest RMSE (30.5 ton ha^{-1}).

In forests, the best stratification options were the aggregated NIR stratification whether or not combined with tree group (Table 6.5). These stratification options decreased the RMSE by 6%-8%. All other stratification options increased the RMSE relative to the NIR SOC map.

All stratification options decreased the RMSE in parts of the Dutch sand area (Table 6.4). In the agricultural area, the RMSE decreased in two-third of the sand area using the soil-aggregated + reclamation type stratification. This option decreased the RMSE in the improved area by 23%. Several other stratification options decreased the RMSE in half to two-third of the agricultural sand area (Table 6.4). In 7% of the agricultural sand area, no decrease of the RMSE could be achieved.

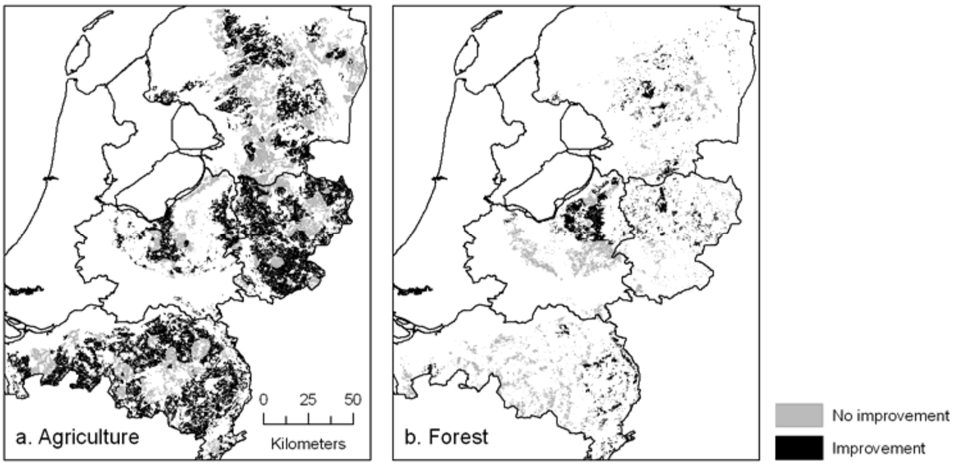


Fig. 6.3. *Areas where the RMSE can be decreased relative to the NIR SOC map using a soil-aggregated + reclamation type stratification (a, agriculture, left) or a NIR-aggregated + tree group stratification (b, forest, right).*

The areas where decrease of the RMSE in agricultural land was achieved (Fig. 6.3a) are dominated by a landscape characterized by a variation of old arable land and wasteland reclamations. The areas where no improvements were possible are dominated by old peat reclamations with less variation in land reclamation types.

In forests, the stratification options decreased the RMSE in 15%-50% of the area. The best stratification option was the NIR-aggregated + tree group stratification (Table 6.4). This decreased the RMSE in the improved area by 19%. In 17% of the forests in the sand area, no decrease of the RMSE relative to the NIR was possible. Decrease of the RMSE is concentrated in the north of the central sand area and the northern sand area (Fig. 6.3b), which are areas with more broadleaf trees and more variation of tree species. As Scots pine dominates Dutch forests, distinguishing tree groups is especially beneficial for estimating SOC stocks in broadleaf forests.

The map resulting from the stratification options that gave the largest RMSE decrease relative to the NIR is shown in Colour plate 6b for both agriculture and forest. Compared to the NIR SOC map (Colour plate 6a.1), including reclamation type or tree group in stratification resulted in higher SOC stocks in areas in the northern and central sand area (Colour plate 6c) with increases up to 20 ton ha^{-1}, also in areas where the RMSE decreases. A higher SOC stock as compared to the NIR is mainly found in areas where a thin peat cover was present before reclamation. Upon reclamation, the peat disappeared due to cultivation and burning (Koster and Favier, 2004). These areas are therefore not indicated as peat on the soil map. The reclamation patterns are typical for peat reclamations and consequently it is possible to distinguish them using a reclamation type map. The historical management, including the burning of peaty topsoils, could have caused accumulation of relatively large amounts of resistant organic matter in the soils. In the central and southern sand area there are areas where a higher SOC stock is expected when including the tree group in stratification. These are areas dominated by old forests with Scots pines. In half of the sand area, the SOC stock is estimated to be higher than in the NIR.

In the agricultural area, lower SOC stocks relative to the NIR were found in old agricultural areas. In forests lower SOC stocks were dominantly found in the afforestations of former heathland.

Forest floor carbon stocks

The RMSE of the FFC stock in the NIR is 11.5 ton ha^{-1} (Table 6.4). Using tree species or age class for stratification slightly decreased the RMSE while using soil type or soil fertility for stratification increased the RMSE. Combining soil fertility, tree species and age class decreased the RMSE by 10%. Low FFC stocks were found in the northern sand area while in the central and southern sand area FFC stocks around 60 ton ha^{-1} were expected (Colour plate 6d). In the northern sand area relatively more oak forests are found, where FFC stocks are lower (Schulp et al., 2008b). Scots pine, dominant in central and southern sand area, has higher FFC stocks. Additionally, in the central and southern sand area forests are older than in the northern sand area. As FFC stocks increase with age, this can explain the higher FFC stocks in the central and southern sand area.

6.3.4. Evaluation of the sample size

The sample size of the SOC stock sample dataset for agricultural land used in the analyses was evaluated for the NIR stratification and the soil-aggregated + reclamation type stratification. The sample size of the FFC stock sample dataset was evaluated for the determination of the overall mean value currently used in the NIR and the tree species + age group + soil fertility stratification (§6.2.5). Fig. 6.4 indicates that upon increasing sampling size the RMSE decreased and the standard deviation between the ten datasets for each sample size decreased. The decrease in standard deviation is due to an increasing overlap between the datasets. The soil-aggregated + reclamation type stratification had a lower RMSE than the NIR stratification, with differences being significant at sample sizes larger than 150. For both stratifications, there was hardly a decrease in RMSE upon sample sizes exceeding 325. It is, therefore, doubtful if RMSE's will further decrease upon increasing sample size.

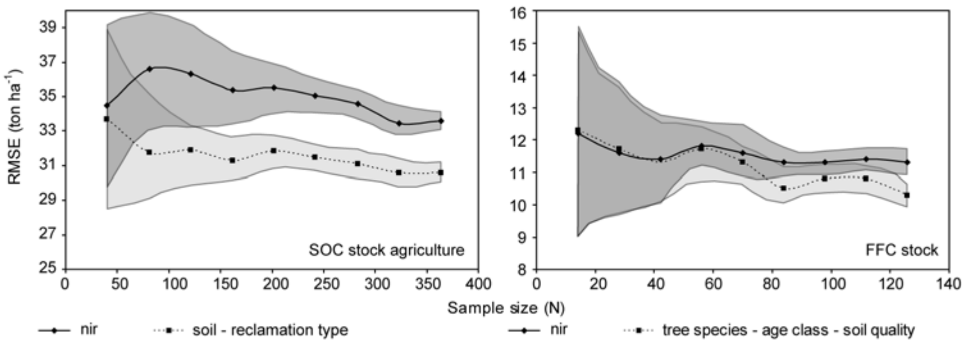


Fig. 6.4. *Effect of sample size on RMSE (ton ha^{-1}) of SOC stocks in agricultural land (left) and FFC stocks (right) for the NIR maps and the best stratification option. The grey areas indicate the standard deviation of the RMSE.*

For the FFC stock dataset, RMSE's hardly decreased at sample sizes larger than 80 without stratifying the data (NIR). When stratifying by tree species, age class and soil fertility, RMSE's decreased upon increasing sample size (Fig. 6.4). Further increase of sample size thus might further improve the estimate of the FFC stocks, especially when the sampling scheme accounts for the factors explaining FCC variability and focuses specifically on sampling of currently underrepresented strata.

6.4. Discussion

6.4.1. Determinants for spatial variability of carbon stocks at multiple scales

For agriculture, using reclamation type as a determinant for SOC variability improved the estimate of SOC variability both at the landscape scale and at the national scale. Getting an imprint of the land use on the SOC stock takes decades to centuries (Schulp and Verburg, 2009). As a result, the long-term land use (represented by the reclamation type) is better for explaining SOC stocks than present-day land use. Additionally, the reclamation type is a proxy for past soil conditions that are not indicated on the soil map but still explain SOC variability. In forests, spatial information on present-day management can decrease the error of the SOC stock. The forest management assessed (the choice for a certain tree species) is almost by definition long-term management and therefore influenced SOC and FFC stocks. Additionally, forest age is identified as determinant for FFC stocks (Thuille and Schulze, 2006) in case studies and was found to be relevant at national scale as well.

Groundwater class is hardly relevant as determinant for national-scale SOC variability in the agricultural sand area. Groundwater levels in the Dutch sand area have been frequently altered in the past, mainly to drain agricultural areas. Because SOC reacts slowly to changes in management conditions, SOC stocks are probably not in equilibrium with current groundwater levels. In forests in the sand area the effect of drainage is less strong and as a result, groundwater class can still explain SOC variability. The interaction between soil type and groundwater class results in forest in a higher percentage explained variance when soil type and groundwater class are combined (Table 6.2). In wet parts of the Dutch sand area further identical soil types can have different SOC dynamics than in dry parts of the sand area.

Current SOC variability is a result of SOC dynamics over the past centuries and past land use is an important driver of these temporal dynamics. The history of interactions between the biophysical landscape and land use results in a spatial SOC imprint of the temporal dynamics. Thus, to improve insight in spatial patterns of SOC and FFC stocks, information on both biophysical characteristics and land use is needed to narrow down the errors in SOC and FFC inventories both at the landscape scale and at national scale.

6.4.2. Evaluation of methods

The method for upscaling SOC stocks used in this study has several inaccuracies. First, the NSSP dataset was treated as a random dataset. During construction of the NSSP, the sampling was stratified by groundwater class and soil type (Visschers et al., 2007). As a result of the sampling design, in some areas sampling density is low and at some locations there is clustering of samples. Second, because the NSSP is stratified by groundwater class and soil type, samples are quite well distributed over groundwater and soil strata. In other stratification options, samples are sometimes divided unequally over the strata. Combined with the relative small sample size, this caused a lack of degrees of freedom in several stratification options. However, only increasing the sample size is not expected to further decrease the error for SOC stocks (Fig. 6.4). Improving the distribution of samples over the soil-reclamation type strata by sampling underrepresented strata might further improve the accuracy.
Finally, for estimating FFC stocks, a dataset from a sampling campaign in 1990 was used (De Vries and Leeters, 2001) while sampling for the NSSP was done in 1994-2001. For SOC stocks, it can be expected that the SOC stock did not change significantly since the sampling campaign. FFC stocks can have changed considerably in the past 19 years (Vesterdal et al., 2002; Thuille and Schulze, 2006). Intensifying the sampling might further improve the estimate of the FFC stock, especially when combined with using a stratification that enables identifying national-scale spatial variability (Fig. 6.4).

6.4.3. Comparison with other studies

The best SOC stock estimate was achieved using a soil-aggregated + reclamation type stratification for agricultural land and a NIR-aggregated + tree group stratification for forest (Colour plate 6b). The total SOC stock in the Dutch sand area using these stratification options is estimated at 105.8 Mton, which is 6.6 Mton lower than the SOC stock currently used in the NIR (Table 6.5). SOC stocks in agricultural land are slightly lower and forest SOC stocks are 10% lower. Including long-term land use to explain national-scale SOC variability results in a change of spatial pattern of SOC at national scale with a more pronounced north-south gradient. A change of the spatial pattern of SOC and FFC stocks will alter the inventory of greenhouse gas budgets in the LULUCF sector and has consequences for studies that use SOC stocks in modelling of future carbon sequestration (Schulp et al., 2008a).

Compared to other countries, SOC densities estimated in the Dutch sand area, both in the NIR and in this chapter (Table 6.5), are high in agricultural land and low in forests. Lettens et al. (2005a) estimate 50 ton SOC ha^{-1} in arable lands, 78 ton SOC ha^{-1} in pasture and 85-95 ton SOC ha^{-1} in forests up to 30 cm depth in Belgium, while Krogh et al. (2003) estimate 80 ton SOC ha^{-1} in agricultural and 116 ton SOC ha^{-1} in forests in Denmark up to 28 cm. Estimates for the UK (Bradley et al., 2005) are in the same order of magnitude as in Denmark and Belgium, while estimates for France (Arrouays et al., 2001) and Spain (Rodriguez-Murillo, 2001) are even lower. Although comparison is hampered because results often are only grouped by land use type and not by soil, there are several explanations for the higher SOC stocks in Dutch agricultural land. Krogh et al. (2003) indicate that in northwest Europe SOC stocks are high due to relative cold and wet conditions, and also indicate that heavy management with large inputs of farmyard manure and slurry can result in large SOC stocks. This is supported by Springob et al. (2001) who found SOC stocks up to 230 ton ha^{-1} in the topsoils of grasslands across northern Germany. Also under cropland Springob et al. (2001) observed SOC stocks in the same range as estimated in this study. They attribute the high SOC stocks to the historical land use, mainly plaggen manuring.

There are two possible reasons for the low SOC stocks in Dutch forests. First, forests in the Netherlands are young compared to other areas in Europe (Pussinen et al., 2001). Second, Dutch forests are often afforestations of infertile drift sands with low initial SOC stocks and poor growing conditions, which slow down the accumulation of SOC.

Table 6.5. *Ton ha^{-1} SOC stock and national total SOC stock (Mton) per land use type as found in the NIR and in this study.*

Land use type	Mean SOC stock (ton ha^{-1})		Total SOC stock (Mton)	
	NIR	*This study*	*NIR*	*This study*
Cropland	105.5	103.3	53.7	52.5
Grassland	106.6	103.2	30.4	29.4
Other agriculture	96.2	94.0	8.1	7.9
Forest	77.4	69.7	20.3	16.1
Total			112.4	105.8

6.4.4. Applications

Insight in the spatial variability of SOC and FFC stocks is of vital importance for insight in the greenhouse gas budget of the LULUCF sector. Without proper insight in SOC and FFC stocks, an accurate estimate of SOC and FFC stock changes is not possible. The decreased errors in the SOC and FFC stock estimates resulting from this study (Table 6.3 and 6.4; Colour plate 6b; Colour plate 6d; Fig. 6.3) might contribute to a better estimate of the LULUCF greenhouse gas budget for the Dutch sand area. For that purpose, the SOC and FFC maps (Colour plate 6b; Colour plate 6d) have to be evaluated based on the Quality Assurance / Quality Control (QA/QC) criteria for national inventories of greenhouse gas inventories (IPCC, 2006):

- *Transparency* (assumptions and methodologies should be clearly explained and should be reproducible);
- *Consistency* (similar data should be used for all sectors and all reported years);
- *Comparability* (approach and results should be comparable with other countries);
- *Completeness* (all source and sink categories should be covered);
- *Accuracy* (errors should be as low as possible).

An important focus of QA/QC issues specific for the LULUCF sector is the consistency of land use areas, correctness of describing management within the country and international comparability of the management correction factors relative to the defaults.

Basic difference between the SOC inventory presented in Colour plate 6b and the NIR SOC map (Colour plate 6a.1) is the explicit inclusion of long-term land use as determinant for SOC variability. This increases the accuracy (Table 6.4) while the transparency, consistency and completeness of the SOC inventory are not affected. In the Dutch NIR SOC map, the use of soil and groundwater as determinants for SOC variability limits the comparability as it is international good practice to use soil and land use (IPCC, 2006). A stronger focus on land use therefore enhances the comparability.

The accuracy of FFC stocks can be improved as well (Table 6.3, Table 6.4). The method used for inventory of the spatial variability of the FFC stocks is transparent and consistent. The default FFC stock that is currently applied in the NIR limits assessing the carbon loss from FFC stocks upon deforestation. With improved insight on the spatial distribution of FFC stocks, assessment of FFC loss upon deforestation will be possible. As a result, more sectors can be covered in reporting, increasing the completeness.

Worldwide, information on long-term land use is increasingly available from local-scale case studies (Van Eetvelde and Antrop, 2005; Kasel and Bennett, 2007; Hupy and Schaetzl, 2008; Kätterer et al., 2008; Stevens and van Wesemael, 2008). For forests, spatial data on tree species (Köble and Seufert, 2001; Skjøth et al., 2008) and forest age (Nabuurs et al., 2006; Schelhaas et al., 2007) are available covering the complete EU and internationally from the ICP project data on forest management and FFC stocks is available (Institute for World Forestry, 2008). Increased accuracy of SOC and FFC stock estimates as achieved in this study is therefore also expected in other areas with comparable land use history and high data availability. Global-scale information on land use history is also increasingly available (Ramankutty and Foley, 1999; Klein Goldewijk, 2001; Klein Goldewijk and Ramankutty, 2004; Hurtt et al., 2006) and might be interesting for explaining SOC variability. However, these datasets are often coarse-scale data based on model-based reconstructions with limited value for explaining landscape-level variation.

6.5. Conclusions

The accuracy of the national-scale inventory of SOC and FFC stocks in the Dutch sand area can be improved by explicitly accounting for the impact of long-term land use on SOC and FFC variability. The errors of the SOC maps can be decreased by 19-23%, in about 60% of the Dutch sand area. Errors of the FFC map were decreased in half the sand area. The improved insight in the spatial variability of FFC stocks provides the possibility to quantify carbon losses from the forest floor upon deforestation, providing the possibility to increase the completeness of the Dutch NIR. Increasing the sample intensity is expected to further improve the insight in national-scale SOC variability only when the sampling design accounts for the long-term land use in stratification and focuses on sampling on currently underrepresented strata.

The internationally increasing availability of spatial data on long-term land use will further enhance possibilities for accounting for the long-term land use in SOC and FFC inventories.

Chapter 7

Synthesis

7.1. The carbon copy of human activities

7.1.1. Determinants for SOC variability at landscape scale

Spatial variability of soil organic carbon (SOC) and forest floor carbon (FFC) stocks at the landscape scale is determined by the interaction between biophysical characteristics of the landscape and long-term land use. Biophysical characteristics, like the loam content or the size of the sand fraction of the parent material, the topography and combinations of these determinants explained a significant part of the variability of SOM contents (Chapter 3). The same biophysical characteristics completed with information about spatial distribution of soil types and geomorphological units explained SOC variability within four case study sites (Chapter 4). Chapter 6 showed the importance of soil fertility for SOC and FFC stocks in forests.

Land use history in agricultural land and management in forests influence SOC and FFC dynamics, because the land use and the management control the quantity and quality of organic input to the soil. Chapter 2 showed that different tree species significantly altered the SOC and FFC stocks within 60 years compared to a uniform starting condition. Chapter 4 showed that in agriculture with low organic matter (OM) input SOC stocks could not be maintained, while land use with high OM input accumulates SOC in due time (Fig. 4.5).
Land use history in agricultural land and forest management also influence the output of SOC. In forests, locations that faced management interventions or were recently disturbed by wild boars had lower SOC and FFC stocks (Figure 2.3, §2.4.2) than undisturbed plots. Removing vegetation upon plaggen harvesting will have decreased SOC stocks in heathlands in the Dutch sand area for centuries (Fig. 4.5).

7.1.2. Long-term landscape-land use interactions

Several studies indicate the slow and prolonged response of the SOC stock to changes in land use (Freibauer et al., 2004; Leifeld and Kogel-Knabner, 2005). Significant changes of SOC stocks upon land use change are only found after longer times, both in forests (Chapter 2) and in agricultural land (Chapter 3 and 4). FFC stocks on the other hand can change relatively quick due to land use change (Thuille and Schulze, 2006) or disturbances like forest fires (De Groot et al., 2009). Land use interventions that result in a clear and long-lasting change in inputs and outputs of OM will print a clear and persisting carbon copy on the landscape, while small or short-lasting changes result in a faint carbon copy that can be easily overwritten. In short-term studies, effects of land use indeed only are observed in specific labile fractions of OM (Schjonning et al., 2007), or otherwise no changes are observed (Geissen et al., 2009).

Consequently, the present-day agricultural land use pattern at landscape scale is only weakly correlated with the current SOC variability because the present-day agricultural land use is only a snapshot of the cumulative OM input that created the current SOC variability. Forest management interventions like choosing a tree species or a rotation length on the contrary have almost by definition a long-term effect and therefore do influence present-day SOC variability in forests. Second consequence of the slow and prolonged response of SOC stocks to land use change is that SOC dynamics and the resulting SOC stocks always lag behind the dynamics of the land use. To understand present-day spatial patterns of SOC stocks, the past temporal dynamics of SOC and the past temporal dynamics of land use have to be understood. Similarly, although the effect of present-day land use on SOC stocks cannot be observed right now, the current and future land use also print a carbon copy on the landscape that will become gradually visible in the future.

When deciding about the allocation of land use and management, humans always adapt the land use to the landscape properties. The land use in turn constantly influences the biophysical characteristics of the landscape. To understand the carbon copy of past, present and future human activities and the resulting SOC variability, the long-term interaction between the landscape and the land use thus has to be understood. Land use change in the Dutch sand area is a clear example of this. The initial settled agriculture in the Dutch sand area (Celtic Fields) is found at relatively fertile locations with high moisture retention (Kooistra and Maas, 2008). The location of Celtic fields influenced the allocation of agricultural fields in the plaggen management. The already less fertile parts of the landscape were used for plaggen harvesting and as a result further impoverished, resulting in formation of drift sands. Simultaneously, the more fertile locations were enriched with nutrients and OM that is assumed to be turned over into relative resistant forms of SOC (Springob and Kirchmann, 2002). In the northern sand area burning of peat for growing buckwheat created inert OM that can probably be compared with biochar (Lenting, 1853; Bieleman, 1992; Woods et al., 2009). The carbon copy of the human activities thus amplified the natural variability of SOC stocks, both at landscape scale and at national scale.

Upon large-scale reclamations in the 19th century, the biophysical landscape, combined with socio-economic conditions like tenureship and accessibility (Thissen, 1994) determined whether wastelands were reclaimed for agricultural use or afforested. Fertile but wet locations were drained and reclaimed; drift sands were afforested. The scale of the land use increased both in agricultural land and forests. The carbon copy of the pre-19th-century land use was therefore overwritten by uniform inputs of OM over larger areas, either by the larger-scale agricultural management or by uniformity of tree species over larger areas. Additionally, newly reclaimed 19th-century agricultural lands and forests can have clearly different initial SOC stocks because of the different past land use.

Agriculture since the 1950s in the Netherlands and in Belgium had a well-defined carbon copy due to heavy application of pig slurry. This has probably increased SOC stocks in 1990 relative to the 1950s in agricultural land (Van Meirvenne et al., 1996; Sleutel et al., 2007), and even further amplified the difference between SOC stocks in forests and agriculture, because no manure is applied in forests. OM in slurry is relatively easy to decompose and therefore, the increased slurry application probably resulted in the build-up of large pools of labile SOC, while accumulation of more stable forms of SOC probably was limited. Negative side effect of slurry application was a large increase of nitrate leaching to the groundwater. To cope with this, policies have been implemented to curtail the environmental pollution due to leaching of N (Sonneveld and Bouma, 2003; Commission of the European Communities, 2007). These policies were an incentive for changed management practices: slurry application has decreased in the past decades. The carbon copy of this management change is that the labile SOC originating from the slurry application is lost again, decreasing the SOC stocks (Goidts et al., 2009).

In the coming decades, extensive abandonment and afforestation of agricultural land is expected in Europe (Table 5.3). The location and extent of abandonment is influenced by a combination of political, demographical, accessibility related and biophysical conditions. Areas that are unfavourable for agriculture are most likely to be abandoned, and at the abandoned locations biophysical conditions impact the speed of regrowth of natural vegetation: Abandonment of agricultural land high up in the mountains will result in slower regrowth than abandonment at fertile lowland locations. Management options on abandoned agricultural land can range between extensive use for e.g. horse keeping and regrowth of natural vegetation or active afforestation. The use of the abandoned land influences the amount of carbon sequestration or emission. Under extensive agricultural use, small amounts of carbon sequestration could be expected. Natural vegetation on the other hand is expected to sequester large amounts of carbon (Fig. 5.5), first in shrubland or forest vegetation, but this is probably soon followed by development of forest floors and sequestration of carbon in more stable forms of SOC. The present-day land use patterns and biophysical conditions combined with decisions on land use change thus will have clear impact on future SOC dynamics: considerable differences in potential sequestration up to 2030 can emerge under different land use trajectories (Colour plate 5b).

The constant interaction between land use change and the biophysical landscape can result in different patterns of SOC variability than would be expected based on assessing biophysical effects or land use effects separately. This is e.g. shown in Chapter 3, where SOM contents are negatively correlated with loam content. These interactions are location-specific. In each landscape the biophysical conditions induce an other spatial arrangement of land use, resulting in a specific carbon copy pattern, depending on the local conditions. Accounting for this interaction helps explaining SOC variability at landscape scale (Chapter 3 and 4) and results in different future SOC dynamics (Fig. 5.5, Colour plate 5b). The slow response of SOC stocks to changes in land use thereby has as a consequence that land use probably always changes before the theoretically expected dynamic equilibrium SOC stock (§1.2.2) is reached. This probably results in SOC stocks to be much more dynamic than expected when assuming equilibrium conditions.

7.1.3. Scale sensitivity of carbon stocks and determinants

The level of spatial variation of environmental variables differs across scales. When environmental processes are analyzed at a large spatial scale, data on variables are often aggregated, which decreases the level of variation. As a consequence, aggregated data cannot be used to explain relations between environmental variables at smaller scale (ecological fallacy; (Steel and Holt, 1996)). Scale sensitivity has been widely discussed in environmental sciences (Lambin et al., 2001; Rindfuss et al., 2004; Wimberly and Ohmann, 2004) and also drivers of land use are scale-sensitive. Dominant drivers of land use at farm scale are social and accessibility related drivers. At landscape scale, topography and agro-climatic potential drive decisions on land use change while at regional to national scale, climatic conditions, macro-economic factors and demographic factors often are dominant (Veldkamp et al., 2001).

Consequently, different determinants are relevant for explaining SOC variability at different resolution (Chapter 3) and extent (Chapter 4, Chapter 6) and the importance of land use for explaining SOC variability is not consistent across scales. Also interactions between long-term land use and the landscape play out differently at different scales. At different resolutions, a digital elevation model describes different landscape features, resulting in different associations with SOC (Chapter 3). At different extents, historical land use patterns either reflect local variability or national-scale variability in processes governing SOC dynamics (Chapter 4). E.g. at local scale, land use adapts to the local circumstances with respect to drainage and soil fertility (Chapter 3), while at national scale land use history in the Netherlands can be seen as a proxy for large-scale differences in soil formation and a climate gradient across the country. At the landscape scale, historical land use interacting with the biophysical landscape is important for explaining SOC variability and the present-day land use pattern is of less importance. At coarser resolution or larger extent, present-day land use is a better determinant for SOC variability because then the scale of analysis better matches the spatial variation of the land use (Chapter 4). However, both at landscape scale and at national scale the long-term land use has stronger associations with SOC stocks than present-day land use. At European scale, location specific land use trajectories are important to explain future SOC dynamics (Chapter 5). The extent and impact on carbon sequestration of future land use changes is expected to be large and long-lasting. At the long term, these large-scale landscape-land use interactions will print a clear carbon copy throughout Europe.

Scale sensitivity of processes causing SOC variability has to be accounted for in sampling, monitoring and inventories of SOC. Currently, in national-scale inventories of SOC stocks it is common practice to use soil classification and present-day land use as a determinant for SOC variability (IPCC, 2006). Chapter 6 showed that in the Dutch sand area explaining SOC variability using soil classification and reclamation type decreased the error of the SOC maps relative to the National Inventory Report for greenhouse gas emissions. Analysis of determinants for national-scale SOC variability and using this knowledge to set up inventories is expected to further improve SOC inventories (§7.3.2).

7.2. Scientific relevance

7.2.1. Research framework

The effect of present-day management on SOC variability at landscape, regional and national scale is overrated in current SOC inventories. Causality of the effect of present-day land use on SOC variability is lacking because of the slow dynamics of SOC relative to the dynamics of land use, and often no effects of present-day land use on SOC variability are observed at all (Breuer et al., 2006; Geissen et al., 2009; Jelinski and Kucharik, 2009).
Effects of long-term land use are more relevant for explaining current SOC variability. This is often overlooked in research on SOC variability and in SOC inventories. This thesis demonstrated the importance of the long-term interaction between the biophysical landscape and the land use by quantification of explaining factors for SOC variability at landscape scale. This interaction can explain spatial variability of SOC stocks in both agricultural lands and forests and although quantification is scale-specific, it can be used to explain SOC variability at multiple scales.

The results of this study indicate that quantification of long-term landscape-land use interactions can help to improve insight in spatial variability of SOC stocks in other case studies as well. Furthermore, quantification of long-term landscape-land use interactions probably can help to explain spatial variability of SOM quality parameters, including the resistance against decomposition. The resistance against decomposition influences both the risk for decline of SOM stocks and the possibilities for future sequestration of carbon in the soil. Insight in spatial variability of SOM quality thus is important both for the European Soil Strategy and for greenhouse gas reporting under the Kyoto protocol.

A research approach focusing on SOC dynamics at the landscape scale, that integrates quantification of land use history and SOC dynamics (Follain et al., 2009) could further improve insight in spatial variability of SOC quality. The landscape scale is an important scale for human-environment interactions (Veldkamp et al., 2001) which are important for explaining SOC stocks at multiple scales (§7.1, Chapter 2, 3, 4, 6). Thereby, the landscape scale is generally overlooked in the few existing multi-scale assessments of SOC variability (e.g. (Schrumpf et al., 2008; Ogle et al., 2009)). For such an approach, data on variability and dynamics of SOC and data on land use dynamics should be combined. Methods for this are evaluated in §7.2.2 - 7.2.4.

7.2.2. Methods for SOC inventories

SOC inventories comprise sampling and laboratory analysis of SOC stocks, followed by upscaling to the area of interest. For sampling of SOC stocks, recently, the use of visible and near-infrared (VNIR) spectroscopy is becoming established (Mutuo et al., 2004; Brown et al., 2006; Gomez et al., 2008). With VNIR spectroscopy, reflectance signals in the wavelength range from 400 to 2500 nm are used to estimate soil properties (Gomez et al., 2008). Spectroscopy has been used to determine SOC contents both on-site with portable devices, and airborne (Stevens et al., 2008). On-site spectroscopy can achieve almost the same accuracy as traditional laboratory methods and is therefore an interesting option for monitoring SOC changes. Airborne spectroscopy has no acceptable accuracy yet, but because many spectra can be recorded easily, this can outweigh the lower accuracy, providing a potential for SOC inventories as well (Stevens et al., 2008).

Main drawback of VNIR spectroscopy is that only bare soil can be analyzed and that only the SOC content at the surface is determined. Only if the topsoil is homogenous, like in croplands that are frequently ploughed, this number is representative for the complete topsoil. When soil moisture contents are high or when a vegetation cover is present the signal is too much disturbed to be applicable for determining SOC contents. Impact of land use change on SOM quality has been demonstrated at plot scale (Paul et al., 2008; Mueller and Kögel-Knabner, 2009). Relatively simple methods for analyzing SOM quality are available. The C:N ratio for example gives a reasonable indication for how easy material can be decomposed. More complex methods for analyzing SOM quality e.g. subdivide OM in different density fractions with different decomposition speed (Basile-Doelsch et al., 2009) or characterize the chemical composition of the SOM in such a way that the source of the OM can be identified (Kögel-Knabner, 2002).

For upscaling SOC stocks or soil quality indicators from sampling points to a landscape, several geostatistical or digital soil mapping techniques are used. Soil properties at a certain location depend on the soil properties at neighbouring locations (McBratney et al., 2003). Kriging (using spatial dependency alone) and co-kriging (using ancillary data that are correlated with the variable of interest) can be used to interpolate soil properties at unsampled locations. This is commonly done in small-scale studies on SOC variability with high sampling densities, e.g. (Mueller and Pierce, 2003; Don et al., 2007). Digital soil mapping relates soil observations to readily available ancillary data. These relationships are extended across the survey area to predict soil properties at unsampled locations (Kempen et al., 2009). Digital soil mapping includes several methods to predict soil properties at unsampled locations, including generalized linear models, classification and regression trees, and geostatistics (McBratney et al., 2003).

Kalman filtering is a technique that merges empirical and process knowledge on the behaviour of a system to estimate the state of the system through time (Heuvelink et al., 2006). This gives the possibility to assess the temporal changes of dynamic systems. The methods for assessing SOC quality indicators described above are often expensive and time-consuming. Sampling intensity in studies that use such methods is therefore often low. Geostatistical upscaling methods on the other hand require large amounts of data: to assess spatial dependency, a high sample density is needed and Kalman filtering is even more data-intensive than common geostatistical techniques. Although techniques for inventories, upscaling and data assimilation are important for providing information on SOC variability, they lack applicability on larger spatial scales and empirical analysis alone is insufficient for increasing process knowledge on SOC dynamics at larger scales.

7.2.3. Methods for long-term land use inventories

Next to data on SOC quantity and quality, spatial data on long-term land use is required for quantification of the impact of long-term land use on SOC variability and dynamics. For reconstructing past land use changes, historical data sources can be used. Large parts of northwest Europe have been mapped regularly at a detailed scale since the 18th century, mainly for military and cadastral purposes. Military maps indicate terrain properties and cadastral maps sometimes indicate suitability of the land for agriculture (Knol and Noordman, 2002). Detailed topographical maps since the 18th century are accurate enough to use them for reconstructing land use history (Koeman, 1963). Regular topographical mappings have been done during the 20th century. Data on recent land use with high spatial and temporal resolution are increasingly available through remote sensing (Appendix Chapter 3-4). In many studies, small-scale historical land use maps have been digitized (Caspersen and Fritzboger, 2002; Petit and Lambin, 2002; Bender et al., 2005; Van Eetvelde and Antrop, 2005; Hupy and Schaetzl, 2008; Stevens and van Wesemael, 2008). Projecting future land use changes is commonly done using a land use change model under contrasting future scenarios (Busch, 2006).
Quantification of inputs by ancient land use systems is challenging but a lot of data is available that gives an indication, e.g. (Van Zanden, 1985; Bieleman, 1992; Spek, 2004). Data from regular farm surveys (Statistics Netherlands, 2008) provides a great potential for quantifying current OM inputs in agricultural land. National-scale forest statistics (LNV Directie Kennis, 2007) or European-scale models (Schelhaas et al., 2007) can provide data on OM inputs in forests. Such data can also be used for estimating future developments of OM inputs when they are combined with spatial land use data.

7.2.4. Methods for linking long-term land use and SOC dynamics

To quantify landscape-scale relations between SOC variability and underlying processes, either empirical data on this relation is needed or the relations can be explored using a model for SOC dynamics. Long-term field trials that establish

empirical relations between SOC variability and underlying processes are scarce, especially when considering SOC dynamics over a century or longer (Smith et al., 1997). Further disadvantage is that long-term field trials are always plot-scale studies where SOC dynamics are controlled by other processes than at larger scales (§7.1.3).
An integrated model for SOC dynamics and land use change could provide a tool to evaluate the role of long-term landscape-land use interactions in explaining variability of present-day SOC quantity and quality, and could be used to evaluate the future impact of changes of land use on SOC quantity and quality. There are studies where both land use changes and SOC dynamics are modelled (Falloon et al., 2006; Schaldach and Alcamo, 2006). As discussed in Chapter 5, in such model exercises the scale difference between SOC models and land use change models often hampers dynamically including the effect of location-specific land use trajectories on SOC dynamics. The spatial distribution and temporal dynamics of input and output of OM through the land use is of prime importance for explaining or predicting the carbon copy of the land use and therefore a high flexibility in modelling land use trajectories is important for modelling SOC dynamics. These land use trajectories and their interaction with the landscape should either be known based on historical sources (§7.2.3) or modelled with a LUC model (Chapter 5; (Claessens et al., 2009)).

Second, the complexity of a model has to be suitable for the scale of interest, the processes included and the data availability. Increasing the complexity of a model by including more parameters and algorithms increases the risk for error propagation (Heuvelink, 1999) and as a result, a less complex model can result in a more certain output (Passioura, 1996). Third, models for carbon dynamics are often developed to describe SOC dynamics at plot scale based on data from long-term experiments. Applying them at large scale can result in upscaling errors, either because processes relevant at the scale of interest are not included in the model algorithms or because of lack of data. Complex models need more data for parameterization which is often not available at larger scales (Verburg et al., 2006b).

Last considerations upon long-term modelling of SOC quality and quantity variation at landscape scale are the pools included in the modelling and the quantification of the start SOC stock. SOC models often distinguish several pools of SOC with different decomposition rates, that mimic the range of chemical composition of SOM (Zimmermann et al., 2007). SOC turns over through these pools one by one. Inclusion of multiple pools in a SOC model at larger scales can be questioned, because the calibration of different pools is impossible due to limited sampling at larger scale (§7.2.2) and because of model complexity issues described above. However, the amount of stabilization of OM into slow pools with residence times exceeding a century is of importance in evaluating effect op past land use and management on SOC dynamics and resulting variability of SOC quality and quantity.
The start SOC stock in modelling exercises is generally quantified by calibrating the model to an equilibrium SOC stock based on data that are assumed to reflect natural circumstances in the study area as closely as possible. Reaching the equilibrium mostly takes 7000-10000 years under which all conditions including climate, vegetation and

management are assumed to remain the same. Although there are hardly alternative approaches for this, this is an unrealistic assumption. As a consequence, soils will never have a SOC stock that is in a dynamic equilibrium, also at the start of a simulation. Not assuming equilibrium conditions but using alternative calibration methods can have strong impact on SOC stocks resulting from such models (Wutzler and Reichstein, 2007).

Integrated modelling of SOC and land use dynamics thus also has several drawbacks and requires data for calibration and validation. However, an integrated model is more flexible for assessing SOC dynamics resulting from land use changes than empirical analysis and is easier to use at scales larger than a single plot. Such model exercises provide the opportunity to simulate future dynamics of SOC quantity and quality. Then, the carbon copy of future policies affecting future land use or management can be evaluated. Such evaluations can be used to guide policy decisions.

7.3. Societal relevance

7.3.1. Applications of methodology and results

The importance of SOM for sustainable use of natural resources has been stressed in Chapter 1. The Commission of the European Communities (2006) estimates the annual costs of the impact of declining SOM contents in the European Union at 3.4-5.6 billion euro, due to greenhouse gas emissions, erosion and loss of soil productivity. EU member states are therefore obliged to identify areas where there is a risk of SOM contents declining to less than 3.4% (Commission of the European Communities, 2006). The improved insight in national-scale spatial variability of SOC stocks in the Dutch sand area (Chapter 6) can contribute to identification of these risk areas.

In identification of risk areas for SOM losses in the northwest European sand area the amount of inert SOM in old arable lands should be accounted for, because loss of SOM from inert pools is unlikely. Old arable lands therefore probably have lower risks for SOM losses. Further quantification of the effects of long-term land use on stabilization on OM would be needed for this. Then, spatial information on historical land use can be a valuable proxy for spatial variability of SOM quality.

Second, the improved accuracy of spatial variability of SOC and FFC stocks as provided in this thesis can increase the accuracy and completeness of the Dutch National Inventory Report for greenhouse gas emission (Table 6.3 and 6.5). A rough indication of the potential annual FFC losses due to deforestation was given in Chapter 2. The impact of gross deforestation on carbon stock changes (Fig. 5.6) stresses the importance of deforestation on SOC stock changes. The national-scale spatial variability of the FFC stock (Colour plate 6d) can be used to provide a better estimate of FFC losses and gives the possibility to include FFC loss upon deforestation in the National Inventory Report for greenhouse gas emissions (Chapter 6). Finally, quantification of the effects of long-term land use on SOC stocks could be used to adapt international IPCC default SOC stocks for intensively managed agricultural land.

Focus of this thesis is on the Dutch sand area. For the remaining, non-sandy, part of the Netherlands, the method used in this thesis to explain national-scale spatial variability of SOC stocks can be easily applied as well, because all necessary legacy data and a national-scale SOC dataset are available. Specific quantification of landscape-land use interactions for the other main landscapes of the Netherlands would be needed. The carbon copy of the land use in other landscapes of the Netherlands will be different from the sand area and might be less clear. In the Dutch peat district anyhow a clear carbon copy of past human activities on present-day soil variability is expected (Van Wallenburg and Markus, 1971; De Bont, 2008). The method used in this thesis could be applied to better explain SOC and FFC stock variability in the complete northwest European sand area (Fig. 1.1a), but then spatial data on the long-term land use is needed.

For land use types other than agriculture and forest, national-scale quantification of SOC stocks remains poor and identification of risk areas for SOM loss is challenging. Urban areas could potentially store considerable amounts of SOC (Mestdagh et al., 2005; Pouyat et al., 2006; Rawlins et al., 2008) but a quantification of the carbon copy of urban land use in the Netherlands is lacking. Because the expected large extent of future urbanization in some scenarios in the EU (Table 5.3) and in the Netherlands (Colour plate 5b), urbanization will have a huge impact on future carbon stock changes resulting from land use and land use change. Types of nature other than forests are poorly sampled and poorly quantified (Visschers et al., 2007). These areas can be strongly impacted by future policies like the allocation of Natura2000 areas and changes in management schemes (Chapter 2). Management interventions resulting from future policies and changes in management schemes can be expected to impact SOC stocks and related emission or sequestration, but quantification of the expected changes is highly uncertain.

In future national-scale sampling of SOC stocks, e.g. to quantify the carbon copy of urban land use or other nature, long-term land use should be accounted for in the sampling design. Upon increased sampling density, a further decrease of the uncertainty of SOC stocks is only expected when the sampling scheme is adapted based on knowledge about the processes that caused the spatial variability of SOC (Fig. 6.4). With the stratification currently used in the Dutch National Inventory Report for greenhouse gas emissions, important information describing SOC variability is excluded from the analysis and as a result no improvements in describing SOC variability can be expected. Data availability on long-term land use is increasingly available worldwide from several case studies, increasing the possibilities for this. Clever use of available ancillary data combined with readily available SOC data can improve national-scale SOC estimates more than setting up new extensive sampling campaigns.

7.3.2. Sustainable future management of SOC stocks

Land use and management activities can result in considerable SOC stock changes at the long term. The importance of land use history for explaining SOC variability relative to the current management indicates that present-day SOC stocks are for a large part determined by factors that cannot be changed right away. Because a small SOC stock change in percentage terms results in a considerable emission or sequestration of CO_2, management strategies aiming at avoiding SOC losses or increasing SOC stocks are however often seen as important strategies to mitigate increases of atmospheric CO_2 concentrations (Dendoncker et al., 2004; Smith, 2004b; Lindner and Karjalainen, 2007), although the effects of management on the SOC stock in Kyoto timeframes probably are limited (Smith et al., 2005b).

One management strategy that is assumed to contribute to climate change mitigation is increasing the application of artificial fertilizer, manure (Lu et al., 2009) or slurry (Van Meirvenne et al., 1996). Next to increasing SOC stocks, applying slurry or fertilizing can however result in increased emissions of N_2O which can easily set aside the potential mitigation (Schlesinger, 2009). Emission of greenhouse gases during production and transport of fertilizers further limits the net mitigation effects.
A second category of potential management strategies for mitigation of climate change focuses on changing tillage practices: Reducing tillage can decrease mineralization of SOC. Also for these strategies for limiting SOC losses, a complete impact assessment is necessary. The influence of tillage on N_2O emissions from mineral soils for example is not clear, small-scale studies suggest that tillage can increase emission of N_2O (Bertora et al., 2007).
In forests, changing the rotation length or the thinning regime, or applying fertilizers, are seen as measures to sequester carbon in the soil (Jandl et al., 2007). The effect of tree species however dominates over the impact of other management strategies.

Avoiding certain land use changes, like deforestation (Chapter 5) or conversion of grassland to cropland can help to avoid future SOC losses (Freibauer et al., 2004) while stimulating woodland regeneration at abandoned agricultural land can increase carbon sequestration. At European scale abandonment of agricultural land and subsequent woodland regeneration is expected to sequester large amounts of carbon (Fig. 5.5) because of the extent of the abandonment and the clear carbon copy of the land use. Increasing the use of biofuels can help decrease the use of fossil fuels and the related emission, but when native vegetation or grassland is converted to produce biofuels, the land use change needed for biofuel production can significantly increase greenhouse gas emissions (Searchinger et al., 2008).

The efficiency of management strategies for mitigation of climate change, both in forests and in agricultural land, are small relative to effects of land use changes such as woodland regeneration (Dendoncker et al., 2004). The potential impact of management on SOC stocks is thereby limited because many land use changes are more irreversible than management practices, and because several practical considerations hamper

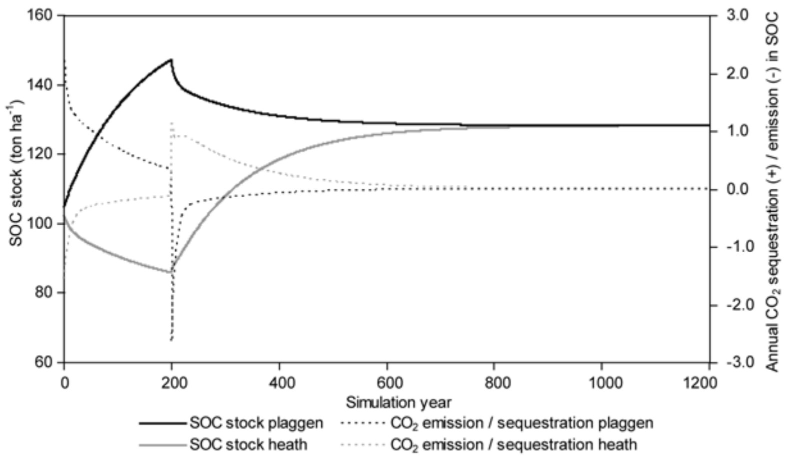


Fig. 7.1. *Temporal dynamics of SOC stocks and annual CO_2 emission / sequestration in the SOC stock under plaggen agriculture and disturbed heathland up to year 200, followed by 1000 years of high-input modern agriculture.*

adaptation of alternative management strategies. The practically achievable carbon sequestration by improved cropland management in Europe is estimated at less than 20% of the biological potential (Smith, 2004b). An additional effect of changing management is however that intensification of agriculture or changing the tillage practices can change the land area that is required for agricultural production. Such land requirement changes strongly influence the extent of future abandonment of agricultural land. Future land use change thus is expected to dominate future carbon balances due to the larger extent and the more pronounced carbon copy.

How the carbon copy of present-day and future land use will overwrite the carbon copy of past land use is illustrated in Fig. 7.1. Temporal dynamics of SOC under two contrasting land use histories (plaggen agriculture and heathland disturbed by plaggen, compare with Fig. 4.5) were simulated over 200 years with RothC. After that, the land use is converted to high-input modern agriculture and this land use is kept for 1000 years. Fig. 7.1 suggests that high-input present-day agricultural land use can need centuries to overwrite the carbon copy of the 19th-century land use.

Currently, OM inputs in agricultural land are to a large extent controlled by governmental regulations. As a consequence, OM inputs will generally be at the maximum allowed amount and have limited spatial variability at national scale. In forests, comparable trends are assumed due to uniformity of tree species over large areas. Dutch forests are dominated by afforestations of heathlands and drift sands, mainly with Scots pines and many ancient forests have been replaced by Scots pines as well. These spatially uniform inputs in theory will finally average out SOC stock variability over larger areas in the future. At the long term, the present-day land use and the future land use will constantly keep on printing carbon copies. The present-day land use is eventually expected to overwrite the carbon copy of past land use, but this can take centuries. As SOC stock size influences CO_2 emission from the soil (Chapter 5), SOC stock differences resulting from past land use result in differences in CO_2 balance during 450 years (Fig. 7.1).

7.4. Conclusions

Spatial variability of present-day SOC stocks is a carbon copy of past human activities interacting with biophysical characteristics of the landscape. Human activities are much more dynamic in time than the biophysical characteristics and consequently have a large impact on dynamics of the SOC stock as well. Although SOC is dynamic at pedological timescales, SOC stocks change slowly when compared to the dynamics of land use. Therefore, the carbon copy of present-day human activities will influence future SOC dynamics. Just like the past land use shaped the present-day SOC variability, the present-day and future land use prints a carbon copy on the landscape that will emerge in spatial patterns of SOC variability in the future. Human activities with a clear carbon copy and a large spatial extent, like land use changes that have pronounced spatial pattern of input and output of OM and that are not easily reversed again, will strongly influence the future spatial variability of SOC quantity and quality.

Management strategies interact with land use dynamics, and can impact the extent and trajectory of land use changes. Through this, future management impacts the future SOC dynamics. Management changes within forest or agriculture have a fainter carbon copy than land use changes and are more easily reversible. The carbon copy of present-day and future land use is therefore of prime importance for explaining variability and dynamics of SOC stocks in the future.

At the short term, no strong impact of human activities on SOC stocks and dynamics is expected, but the long-term effect of present-day land use on SOC stocks and dynamics is of prime importance for the future use of soil resources. The effects of long-term land use and management on SOC dynamics therefore have to be considered upon future decisions on land use change.

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Summary

Soil organic matter (SOM) is a key element for the structure, fertility and water holding capacity of the soil. Moreover, SOM is an important pool in the global carbon cycle and can store large amounts of soil organic carbon (SOC) in a sustainable way. Human activities have a profound impact on the balance between uptake and loss of SOC: land use is an important determinant for the amount of carbon that a soil can store, and influences the potential rate of loss or uptake. Spatial patterns of human activities that control SOC dynamics persist in spatial patterns of SOC variability over a long time. Spatial variability of SOC thus can be seen as a close copy, a carbon copy, of human activities.

Because SOC is essential for sustainable future use of natural resources, several international policies including the European Soil Strategy and the Kyoto Protocol oblige countries to provide insight in national-scale spatial variability of SOC stocks. In many countries including the Netherlands, information on national-scale spatial variability of SOC stocks is rather poor. Aim of this thesis is to assess the functioning of the carbon copy of human activities and to use this to improve insight in spatial variability and dynamics of SOC stocks for the Dutch sand area. The Dutch sand area comprises 43% of the Netherlands and three-quarters of the Dutch forests, making it an important area for sequestration of carbon. The sand area has a varied land use history, both in agricultural lands and in forests. Some agricultural land is in use since the iron age. Large heath or drift sand areas on the other hand have been only recently reclaimed or afforested.

After the general introduction (Chapter 1), Chapter 2 identifies and quantifies factors determining spatial variability of SOC and forests floor carbon (FFC) stocks at the landscape scale. This is done in a forest case study in the centre of the Netherlands. Tree species differ in the amount and quality of the litter they produce. As a result, different tree species under further similar circumstances have significantly different carbon stocks in both the forest floor and the mineral topsoil. In a larch stand the largest carbon stocks were observed: 29.6 ton per ha in the forest floor, 97.1 ton per ha in the top 20 cm of the mineral soil. The beech research stand had the lowest carbon stocks: 11.1 and 53.3 ton per ha in the forest floor and the mineral topsoil respectively. Carbon stocks were significantly altered by the management intensity. Plots with signs of recent management activities like thinning or harvesting had lower carbon stocks in forest floor and mineral topsoil than unmanaged plots.

The SOC stock responds very slowly to changes in factors that control the input and output of organic matter to the soil. Consequently, present-day variability in SOC stocks could be a carbon copy of the land use in the far past. In Chapter 3 this is proved for a case study in the northern part of the Dutch sand area. Old agricultural lands had higher SOM contents than recent heath reclamations and patterns of past land use had a stronger association with SOM variability than patterns of present-day land use. At a coarser resolution, the importance of present-day land use for explaining SOC stocks increases because then the scale of analysis more closely matches the scale of the land use.

Chapter 4 focuses on the land use history of agricultural lands as well. The impact of land use history on variability of SOC stocks is shown to apply in other case studies across the Dutch sand area as well. A sensitivity analysis of the effect of land use history on temporal dynamics of SOC stocks reveals why historical land use is more strongly associated with SOC variability than present-day land use: the SOC stock at a certain time and location is a result of SOC dynamics over the past centuries. Historical land use types like plaggen agriculture and the associated degradation of heathlands lasted for a long time and were highly variable at a small scale. High-input present-day agriculture has had too short time to overwrite this carbon copy. Thereby, present-day land use patterns vary at a much larger scale than historical land use patterns, resulting in less importance for explaining SOC variability at small scale.

Policy, demography and globalization are expected to cause large changes in EU land use in coming decades. Chapter 5 assesses how these land use changes between 2000 and 2030 would influence sequestration of carbon in soil and vegetation across the complete European Union. Future land use changes and carbon sequestration were modelled under four scenarios that cover a range of globalization options and a range of governmental intervention levels. In one of the four scenarios a decrease of annual carbon sequestration was expected. In this scenario there was no decrease of cropland area. The other three scenarios demonstrated an increase in sequestration rates by 9-16% in 2030 relative to 2000. The highest sequestration rate is expected in the scenario where agricultural subsidies remain largely unchanged and development occurs regionally. Total carbon sequestration is mainly determined by high sequestration in forests, emissions from cropland and deforestation. As cropland emissions depend on SOC content and forest sequestration depends on age, the location of land use change is important for the net sequestration of the European land use. For example, removing old forests leads to larger carbon losses than removing young forests (EC DG ENV News Alert Service, 2009).

In Chapter 6, insight is provided in national-scale variability of SOC and FFC stocks using the knowledge on SOC variability gained through Chapter 2-5. Because Chapter 3 and 4 showed that determinants for SOC variability are different at different scale of analysis, first the relevance of determinants identified in case studies for national-scale SOC variability was checked. Using the relevant determinants, national-scale spatial variability and the associated uncertainty of SOC and FFC stocks was mapped. For forests, errors in the national-scale spatial variability of SOC stocks can be decreased in half of the sand area by distinguishing between conifer and broadleaf forests. For agricultural lands, including the impact of historical land use decreased errors in national-scale spatial variability of SOC stocks in two-third of the sand area. Total SOC stocks in the Dutch sand area did hardly change compared with the current state-of-the art, while the spatial distribution of SOC stocks across the sand area changed to a stronger north-south gradient with lower SOC stocks towards the south.

Chapter 7 is a synthesis of the results and conclusions from Chapter 2-6. Long-term landscape-land use interactions are important determinants for SOC variability at multiple scales, and can be highly relevant internationally for explaining spatial variability of SOC quantity. Thereby, an approach integrating biophysical factors and historical land use could shed more light on the spatial patterns of quality of organic matter, which is highly relevant for climate change research and identification of areas with a risk of declining SOM contents. The importance of past land use for explaining present-day SOC variability indicates that the present-day land use will leave a carbon copy on the future landscape. As land use on the long term is of prime importance for SOC dynamics, the longevity of the response of SOC stocks to changes in land use has to be considered in decisions upon land use change to secure the sustainable future use of soils.

Samenvatting

Organische stof is essentieel voor de bodemvruchtbaarheid en de bodemstructuur en verbetert het vermogen van de bodem om vocht vast te houden. Bovendien kan de bodem grote hoeveelheden CO_2 uit de atmosfeer opnemen en duurzaam opslaan in de vorm van organische koolstof. Mensen hebben veel invloed op de opname en uitstoot van koolstof in de bodem: vegetatie, en dus landgebruik, is doorslaggevend voor de hoeveelheid koolstof die de bodem op kan slaan of uit kan stoten, en hoe snel dat kan gaan. Ruimtelijke patronen van landgebruik en beheer drukken een stempel op het landschap in de vorm van ruimtelijke variatie van koolstofvoorraden, die lange tijd zichtbaar blijft. Ruimtelijke variatie van de hoeveelheid koolstof in de bodem kan dus gezien worden als een afdruk, oftewel een “carbon copy”, van menselijk handelen.

Diverse internationale richtlijnen, waaronder de Europese Bodemrichtlijn en het Kyotoprotocol, schrijven voor dat landen inzicht moeten hebben in de omvang van de hoeveelheid koolstof in de bodem en de onzekerheidsmarges hiervan. In veel landen, waaronder Nederland, ontbreekt dit inzicht grotendeels. Doel van dit proefschrift is om na te gaan hoe landgebruik een stempel drukt op de variatie van de hoeveelheid koolstof in de bodem, en om met deze informatie het inzicht in de variatie van koolstofvoorraden in de bodem van de Nederlandse zandgebieden te verbeteren. Zulke informatie is essentieel voor inzicht in duurzaam gebruik van natuurlijke hulpbronnen. Het Nederlandse zandgebied beslaat 43% van Nederland en driekwart van de Nederlandse bossen bevindt zich in het zandgebied. Daarom is het een zeer belangrijk gebied voor opname van CO_2 uit de atmosfeer. Het landgebruik in het Nederlandse zandgebied heeft een gevarieerde historie; sommige gebieden zijn in gebruik voor landbouw sinds de ijzertijd. Daarentegen zijn grote voormalige heide- en stuifzandgebieden pas sinds de 19^{e} eeuw ontgonnen of bebost.

Dit proefschrift begint met een algemene inleiding (Hoofdstuk 1). Daarna beschrijft en kwantificeert Hoofdstuk 2 factoren die variatie van de hoeveelheid koolstof in de bodem van een bos beïnvloeden op de schaal van een landschap. Het effect van de boomsoort en de intensiteit van het bosbeheer op koolstofvoorraden in de bodem en de strooisellaag is onderzocht in een bos op de Veluwe. Bomen verschillen in strooiselproductie en strooiselkwaliteit, en daardoor hebben bosopstanden met verschillende bomen onder verder identieke omstandigheden, significant verschillende koolstofvoorraden. Larixen bleken de grootste hoeveelheden koolstof op te slaan: 29.6 ton per hectare in de strooisellaag, 97.1 ton per hectare in de minerale bovengrond. Het onderzochte beukenbos had de kleinste koolstofvoorraden: 11.1 ton per hectare in de strooisellaag en 53.3 ton per hectare in de minerale bovengrond. Bovendien bevatten de bodem en de strooisellaag minder koolstof op plekken waar recent beheersingrepen hadden plaatsgevonden, zoals oogsten of uitdunnen.

Koolstofvoorraden reageren zeer langzaam op veranderingen in aanvoer en afvoer van organische stof. Daarom valt het te verwachten dat de huidige koolstofvoorraad in de bodem beïnvloed is door historische veranderingen van het landgebruik. Hoofdstuk 3 laat zien dat dit in een proefgebied in het noorden van Nederland inderdaad het geval is. Oude landbouwgronden hadden hogere organische-stofgehaltes dan 19^{e}- en 20^{e}-eeuwse heide-ontginningen, en bij een analyse op kleine schaal waren historische patronen van landgebruik sterker gecorreleerd met patronen van organische-stofgehaltes dan hedendaags landgebruik. Als de analyse werd herhaald op een grotere schaal werd de correlatie met hedendaags landgebruik sterker. Dit kan worden toegeschreven aan de grotere schaal van hedendaags landgebruik.

Hoofdstuk 4 laat zien dat historisch landgebruik ook in drie andere proefgebieden in andere delen van het Nederlandse zandgebied van belang is voor de ruimtelijke variatie van de hoeveelheden koolstof. Een analyse van de gevoeligheid van koolstofvoorraden toont aan waardoor historisch landgebruik meer invloed heeft op de variatie van koolstofvoorraden dan huidig landgebruik: De koolstofvoorraad op een bepaalde plaats en tijd is het resultaat van aanvoer en afvoer van organische stof in de afgelopen eeuwen. Eeuwenlang is in het Nederlandse zandgebied de heide verarmd door onttrekking van organisch materiaal in de vorm van plaggen. Met dit materiaal werden de akkers verrijkt. Deze vorm van landbouw wordt aangeduid als plaggenlandbouw. Deze heeft een zeer sterke koolstofafdruk achtergelaten met een grote mate van ruimtelijke variatie, dit in tegenstelling tot de moderne landbouw met hoge mestgiften die nog maar relatief kort bestaat en nog geen koolstofafdruk heeft achtergelaten. Tevens is hedendaags landgebruik is veel grootschaliger en heeft dus op kleine schaal minder invloed op variatie van de koolstofvoorraad, terwijl op grote schaal de invloed van hedendaagse patronen van landgebruik juist belangrijker is.

Net als Hoofdstuk 4 gaat ook Hoofdstuk 5 over veranderingen van de koolstofvoorraad in de tijd. Politieke keuzes, demografische ontwikkelingen en de mate van globalisering hebben waarschijnlijk de komende decennia veel invloed op het landgebruik in Europa. In Hoofdstuk 5 wordt gesimuleerd hoe veranderingen van het landgebruik tussen 2000 en 2030 de vastlegging van koolstof in bodem en vegetatie in de hele Europese Unie kunnen beïnvloeden. Hiervoor zijn een model voor landgebruikveranderingen en een model voor koolstofvastlegging gebruikt. De simulatie is uitgevoerd voor vier verschillende toekomstscenario's. De scenario's verschillen in de mate van globalisering en in de mate van vrije-marktwerking. In een van de vier scenario's wordt verwacht dat de jaarlijkse koolstofvastlegging afneemt. In dit scenario neemt het akkerlandareaal niet af. In de andere drie scenario's laten de simulaties zien dat de jaarlijkse koolstofvastlegging toeneemt met 9-16%. De sterkste toename wordt verwacht in het scenario waarin landbouwsubsidies ongewijzigd blijven en er een sterke mate van regionale ontwikkeling is. De netto vastlegging van koolstof in Europa wordt gedomineerd door vastlegging in bossen en uitstoot door akkerlanden en ontbossing. Hoe sterk de uitstoot van koolstof uit akkerlanden is, hangt af van de hoeveelheid koolstof die in de bodem zit. De jaarlijkse vastlegging van koolstof in bossen hangt sterk af van de leeftijd van het bos. Daarom is het voor de netto vastlegging van koolstof van belang waar in Europa welke veranderingen in landgebruik zullen plaatsvinden (EC DG ENV News Alert Service, 2009).

De kennis uit Hoofdstuk 2 t/m 5 wordt in Hoofdstuk 6 gebruikt om op nationale schaal meer inzicht te geven in patronen van de koolstofvoorraad in de bodem van het Nederlandse zandgebied. Omdat Hoofdstuk 3 en 4 aantoonden dat op verschillende ruimtelijke schalen verschillende factoren van belang zijn om de variatie van hoeveelheden koolstof te verklaren, is eerst nagegaan of de relevante factoren uit kleinschalige proefgebieden ook relevant zijn op nationale schaal. Vervolgens is met behulp van de relevante factoren een nieuwe kaart van de ruimtelijke variatie van de koolstofvoorraad gemaakt voor het gehele Nederlandse zandgebied. Ook zijn de onzekerheidsmarges van de koolstofvoorraad in kaart gebracht. De koolstofkaart en de kaart van de onzekerheidsmarges zijn vergeleken met de actuele koolstofkaart en de actuele kaart van de onzekerheidsmarges. In de helft van de bossen kon de nauwkeurigheid van de koolstofkaart verbeterd worden, door onderscheid te maken tussen loofbossen en naaldbossen. In het landbouwgebied verbeterde de koolstofkaart in twee-derde van het areaal. Hier zijn gebieden met verschillende ontginningstypes onderscheiden bij het maken van de koolstofkaart. De totale koolstofvoorraad in het Nederlandse zandgebied is nauwelijks veranderd, maar de nieuwe koolstofkaart vertoont een sterkere noord-zuidgradient, waarbij de koolstofvoorraden in zuidelijke richting afnemen.

Hoofdstuk 7 is een algemene synthese van de resultaten en conclusies. Alle hoofdstukken hebben gemeen dat ze laten zien hoe belangrijk de langdurige interactie van het landgebruik met het landschap is voor het verklaren van variatie van koolstofvoorraden. Deze interactie is van belang in kleine proefgebieden, op regionale schaal en voor patronen op nationale schaal en zou ook gebruikt kunnen worden om meer duidelijkheid te krijgen over ruimtelijke patronen van de chemische kwaliteit van organische stof. De chemische kwaliteit van organische stof bepaalt in sterke mate het risico van verliezen van organische stof als er veranderingen in het landgebruik optreden. Dit is zeer relevant voor toekomstig onderzoek naar de invloed van klimaatveranderingen en voor het aanwijzen van gebieden waar organische stof-verlies een risico is voor toekomstig gebruik van de bodem. Het doorslaggevende belang van historisch landgebruik voor het verklaren van de huidige ruimtelijke variatie van de koolstofvoorraad geeft aan dat het huidige landgebruik tot in de verre toekomst invloed zal hebben op de ruimtelijke variatie van de hoeveelheden koolstof in de bodem, en op de uitstoot en opname van CO_2 die in de toekomst te verwachten is.

About the author

Curriculum vitae

Catharina Johanna Elizabeth (Nynke) Schulp was born on April 21st, 1979 in Sneek. After attending the Bogerman secondary school in Sneek, she studied Soil, Water and Atmosphere at Wageningen University, starting in 1997. Her major MSc thesis at the Soil Inventory and Land Evaluation group of Wageningen University lead to the thesis "Pliocene marine-fluvial interactions in the lower Guadalhorce Basin, Southern Spain". A second thesis research at Alterra and the Historical Geography chair group of Wageningen University was on the influence of the biophysical landscape on past and present land use patterns. After that, Nynke did an internship at DHV Consultancy and Engineering where she was involved in soil policy consultancy.
After graduating in 2003, Nynke worked as junior researcher at the Soil Inventory and Land Evaluation group in 2004. She contributed to modelling future land use changes in the European Union in the EURURALIS project. In May 2005, she started her PhD (part time) at the Land Dynamics group.
Next to doing science, Nynke is active as saxophonist in several orchestras and ensembles ranging from classical music to jazz, and is an enthusiast photographer.

List of publications

Schulp, C.J.E. P.H. Verburg, P. Kuikman, G.J. Nabuurs, J.G.J. Olivier, W. de Vries, A. Veldkamp, 2009. Upscaling landscape-scale knowledge to improve national-scale carbon stock inventories. Global Change Biology, submitted.

Schulp, C.J.E. and P.H. Verburg, 2009. Effect of land use history and site factors on spatial variation of soil organic carbon across a physiographic region. Agriculture, Ecosystems and Environment 133: 86-97.

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Verburg, P.H., C.J.E. Schulp, N. Witte, A. Veldkamp, 2006. Downscaling of land use change scenarios to assess the dynamics of European landscapes. Agriculture, Ecosystems and Environment 114: 39-56.

PE&RC PhD Education Certificate

With the educational activities listed below the PhD candidate has complied with the educational requirements set by the C.T. de Wit Graduate School for Production Ecology and Resource Conservation (PE&RC) which comprises of a minimum total of 32 ECTS (= 22 weeks of activities)

Review of Literature (5 ECTS)
- Dynamics and spatial variability of soil organic carbon and its determinants at European, national and landscape scale.
  Presentation European scale in Spatial Methods discussion group (2007);
  Presentation on SOC variability in forest soils in Climate change & Soil-water-atmosphere Interactions "CSI Wageningen" discussion group (2007)

Writing of Project Proposal (6 ECTS)
- The role of land use history and management in SOC pool behaviour at the landscape level in the Netherlands (2005)

Post-Graduate Courses (7.6 ECTS)
- Land Science: concepts, tools and uncertainties in land use studies and landscape dynamics; DOW-SIL and PE&RC (2005)
- Advanced statistics; WGS (2006)
- Summer school Experimental Assessment of Soil Organic Carbon in Mountain Forests; Institute of Forest Ecology, University of Natural Resources and Applied Life Sciences-BOKU (2006)
- Land Science: Bringing concepts and theory into practice; DOW-SIL and PE&RC (2007)
- Statistical Methods for Spatial Data Analysis and Modelling; Climate changes Spatial Planning research programme; PE&RC (2007)

Deficiency, Refresh, Brush-up Courses (2.5 ECTS)
- Basic statistics; WGS (2005)
- Techniques for writing and presenting a scientific paper; WGS (2006)

Competence Strengthening / Skills Courses (2.4 ECTS)
- PhD competence assessment; WGS (2005)
- Professional communication strategies for PhD students; WGS (2005)
- Workshop scientific publishing; WGS (2005)
- Organising and supervising MSc thesis; WGS/OWU (2008)
- NWO Talent Day; NWO (2008)
- How to write an FP7 proposal; Wageningen International Helpdesk (2008)

Discussion Groups / Local Seminars and Other Scientific Meetings (5.7 ECTS)
- PE&RC Discussion group Spatial Methods (2005-2008)
- BSIK Climate changes Spatial Planning-Mitigation conference Groningen; oral presentation (2006)
- Synergy in CcSP Research meeting; oral presentation (2007)
- Climate changes Spatial Planning-Mitigation meeting; oral presentation (2007)
- PE&RC discussion group Climate change & Soil-water-atmosphere Interactions- "CSI Wageningen " (2007-2009)
- Climate changes Spatial Planning-Mitigation meeting; oral presentation (2009)

PE&RC Annual Meetings, Seminars and the PE&RC Weekend (1.8 ECTS)
- PE&RC Annual meeting "The truth of science" (2005)
- PE&RC 10th Anniversary (2005)
- PE&RC Introduction weekend (2006)
- Annual meeting "Collapse" (2007)

International Symposia, Workshops and Conferences (13.2 ECTS)
- Soil and Water symposium; Stichting Kennisontwikkeling Bodem, Driebergen, the Netherlands; oral presentation (2006)
- Climate changes Spatial Planning mid-term review conference; Den Haag, the Netherlands; oral presentation (2007)
- COST 639 WG IV meeting; Copenhagen, Denmark; oral presentation (2008)
- EUROSOILS; Vienna, Austria; oral presentation (including chairing of a discussion session at COST 639 workshop) (2008)
- Technical workshop on projections of GHG emissions and removals in the LULUCF sector; Ispra, Italy; invited oral presentation (2009)
- EGU; Vienna, Austria; oral presentation (2009)
- 8th International Carbon Dioxide Conference; Jena, Germany; poster presentation (2009)

Courses in Which the PhD Candidate Has Worked as a Teacher
- International course "Ecoregional tools"; ILRI, Kenya (2005)
- Soil and water I; DOW-LAD (2006-2009)
- International course Regional and National Biodiversity modelling and analysis; ITC, Enschede and MNP, Bilthoven (2007)
- Field practical Geology, soils and landscape of the Netherlands; DOW-LAD (2008, 2009)

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