UNIVERSITÀ DEGLI STUDI DI SASSARI

A.D. MDLXII

SCUOLA DI DOTTORATO DI RICERCA

Scienze dei Sistemi Agrari e Forestali

e delle Produzioni Alimentari

Indirizzo Agrometeorologia ed Ecofisiologia dei Sistemi Agrari e Forestali

Ciclo XXIII

STUDY OF THE SOIL CARBON DYNAMICS AND REGIONAL ESTIMATES OF CARBON SEQUESTRATION IN SARDINIA SOILS LINKING THE ROTHC MODEL TO GIS DATABASES

Dr.ssa Marina Carta

Direttore della Scuola	Prof. Giuseppe Pulina
Referente di Indirizzo:	Prof.ssa Donatella Spano
Docenti Guida:	Prof.ssa Donatella Spano
	Dr Costantino Sirca

Quadriennio Accademico 2008 - 2011

UNIVERSITÀ DEGLI STUDI DI SASSARI
SCUOLA DI DOTTORATO DI RICERCA

Scienze dei Sistemi Agrari e Forestali
e delle Produzioni Alimentari

Direttore: Prof. Giuseppe Pulina
INDIRIZZO
Agrometeorologia ed ecofisiologia dei Sistemi Agrari e Forestali

Referente: Prof.ssa Donatelle Spano

XXIII Ciclo – Quadriennio Accademico 2008-2011

STUDY OF THE SOIL CARBON DYNAMICS AND REGIONAL ESTIMATES OF CARBON SEQUESTRATION IN SARDINIA SOILS LINKING THE ROTHC MODEL TO GIS DATABASES

Tesi di dottorato:

Dott.ssa Maarina Carta

Direttore della scuola: Prof. Giuseppe Pulina
Docente guida: Prof.ssa Donatella Spano
Docente tutor: Dr. Costantino Sirca

Αδ Αννα

"Soil organic matter is the fuel that runs the soil's engine"(Fischer, 1995)

Abstract

In view of a rapidly changing climate the soils have received an increasing amount of attention over the last two decades. Soil may cause an important feedback to climate change. If C (C) is released from soil to the atmosphere climate change will be exacerbated. On the other hand, if more C is accumulated in soil and the emissions decrease, climate change will be retarded, consequently changes in soil C content can have a large effect on the global C budget. The estimation of soil C content is of pressing concern for soil protection and in mitigation strategies for global warming.

There are still many uncertainties and unanswered questions related to this issue of C accounting and the assessment of soil C amount in Mediterranean regions is even more difficult. There are few studies about C soil content in Sardinia, and consequently in this context drawn on the goal of my thesis.

In this work a modeling approaches based on a Geographic Information Systems (GIS) was used for the estimation soil organic C in Sardinia soil. The overlapping of different layers in a GIS environment has allowed to collect the necessary inputs at Rothamsted C model run, while the validation of this was made possible by a collection of about 160 ground data scattered throughout the island area. By linking GIS, that contain detailed information on soils, land use and climate with dynamic simulation models for the turnover of organic C, it is possible to estimate the impacts of climatic changes on soil C stocks.

Index

INTRODUCTION

The earth's climate is determined by several complex connected physical, chemical and biological processes occurring in the atmosphere, land and ocean.

In view of a rapidly changing climate the soils have received an increasing amount of attention over the last two decades, both from scientists and the public indeed, mitigating the potential large negative impacts of a change in the earth's climate will require strong and definite actions in the different sectors, particularly the soils (link to economic sectors of agriculture and forestry) deserve a close examination due to their large carbon (C) mitigation potential (IPCC, 2007).

Soil is defined as the top layer of the earth's crust through which the lithosphere, hydrosphere, atmosphere and biosphere interact. His training from the parent rock, involves a long series of processes and interactions between various components, living and not interacting with each other creating soil environment (pedosphere) with a depth varying from a few centimeters to several meters.

In the life of our planet the soil plays a key role because it regulates the cycles of water, C, phosphorus and nitrogen, and is a largely non-renewable resource, sensitive to the impacts of climate change and human activity. These resources are listed below:

- Production: almost all vegetation has its roots in the soil and it draws nourishment and water
- Regulatory: soil characteristics depend on the movement of water into it (cycle), the transport of solid particles, the propensity to erosion
- Protection: the soil is a biological filter, capable of retaining pollutants and protecting surface and ground water resources and food chains
- Nature: it represents an important reservoir of biodiversity since is the richest habitat plant and animal organisms
- Climate: a role in the C cycle influencing the Earth's surface energy budget and climate
- Place: home building, road infrastructures and distribution networks and is a source of raw materials such as clay, sand and gravel
- Historical and environmental preserves traces of environmental changes and human history

Climate exerts a significant influence on the soil life generally in equilibrium with climatic conditions that led to its formation. Climate change can alter this equilibrium, exercising a significant influence on soil properties. On the other hand, the soil itself can play a key role in mitigating or even more critical to the increasing trend of atmospheric concentrations of greenhouse gases (carbon dioxide, nitrous oxide and methane) (FAO Conference, 2010). Soil may cause an important feedback to climate change. If C (C) is released from soil to the atmosphere or if methane and nitrous oxide emissions increase, climate change will be exacerbated. On the other hand, if more C is accumulated in soil and the emissions decrease, climate change will be retarded (Rothamsted Research 2005).

If the rate of organic matter accumulation exceeds, the rate of degradation increases the C content in the soil at the expense of C from the atmosphere. If the rate of accumulation is less than the rate of degradation, the C content in soil decreases and the soil acts as an additional source of carbon dioxide (CO_2) to the atmosphere.

This equilibrium very complex is subject of much research throughout the world. Therefore there are many studies of global climate change which took into account the direct effects on the structure and function of soil ecosystems, trying to understand how they may react to climate changes underway (European Commission, 2010).

1. Why is it important to study the soil carbon?

More than twice as much C is held in soils as in vegetation or the atmosphere (Bellamy et al., 2005). Total soil C pools for the entire land area of the world is estimated by Batjes (1996) to be 2157-2293 Pg of C in the upper 100 cm, while soil organic C is estimated to be 684-724 Pg of C in the upper 30 cm, 1462-1548 Pg of C in the upper 100 cm, and 2376-2456 Pg of C in the upper 200 cm.

Changes in soil C content can have a large effect on the global C budget (Bellamy et al., 2005). The estimation of soil C content is of pressing concern for soil protection and in mitigation strategies for global warming (Jones et al., 2005). The greenhouse effect has created great concern and the role of soils in mitigating this effect has been recognized (Bhattacharyya et al., 2007); on the other hand the possibility that climate change is being reinforced by increased C dioxide emissions from soils owing to rising temperature is the subject of a continuing debate (Bellamy et al., 2005).

There are still many uncertainties and unanswered questions related to this issue of C accounting. This topic is highly relevant across Europe for the soils more susceptible to desertification (i.e. Spain, Italy) and peat soils with large stocks of organic matter in the north (i.e. Finland, Scotland) or intensively managed soil in major agricultural areas (France, Germany, the Netherlands, the United Kingdom) places where organic matter is expected and anticipated to further decrease. Soil C stocks in Europe are estimated to be average 15.8 Kg C m-2 to a soil depth of 1 m; estimates of mean annual soil C stock changes are almost three orders of magnitude lower, with croplands losing soil C at a rate of about 70 g C m-2 y-1, and forests and grasslands gaining C at a rate between 37-60 g C m-2 y-1 (Janssens et al., 2005).

Over millions of years the CO_2 is removed from the atmosphere through erosion by silicate rocks and through burial in marine sediments of C fixed by marine plants.

Burning fossil fuels returns C captured by plants in Earth's geological history to the atmosphere. Until the 1980s atmospheric CO_2 and oxygen (O_2) data shows that the terrestrial biosphere was largely neutral with respect to net C exchange, but atmosphere became a net C sink in the 1990s. During the last two decades, fossil fuel combustion and cement production emitted 6.4 ± 1.3 Pg C y^{-1} to the atmosphere, while land-use change emitted 1.6 ± 0.8 Pg C y^{-1}. Atmospheric C increased at a rate of 3.2 ± 0.1 Pg C y^{-1}, the oceans absorbed 2.3 ± 0.8 Pg C y^{-1} and there was an estimated terrestrial sink of 2.6 ± 1.3 Pg C y^{-1} (Smith , 2005). The flux of CO_2

between soil and the atmosphere is large and estimated at 10 times the flux of CO_2 from fossil fuels (FAO Conference, 2010).

Human activities contribute to climate change in many direct and indirect ways; burning of fossil fuels and deforestation, but also cement production and other changes in land use and management such as biomass burning, crop production and conversion of grasslands to croplands have determined the increase in atmospheric CO_2 concentration, so CO_2 emissions from human activities are considered the single largest anthropogenic factor contributing to climate change (Denman *et al*., 2007). Through cultivation and disturbance, the man has decrease and still is decreasing the soil C relative to the store typically achieved under native vegetation, totally this human intervention have caused a loss of soil C of between 40 and 90 Pg C globally (Paustian *et al*., 1998; Houghton *et al*., 1999;).

More recently, the mitigation focus has shifted and grown to include the non-CO_2 gases. There is growing recognition that many of the sources of non-CO_2 gases are manageable and that small changes in flux can be important and have long-term impact. A further important issue is that the gases methane (CH_4) and nitrous oxide (NO_2) are also potent greenhouse gases and are emitted from, and absorbed by soils; accordingly soils also have a second role to play, reducing emissions of other trace gases to atmosphere, in combating climate change (*fig. 1, 2*).

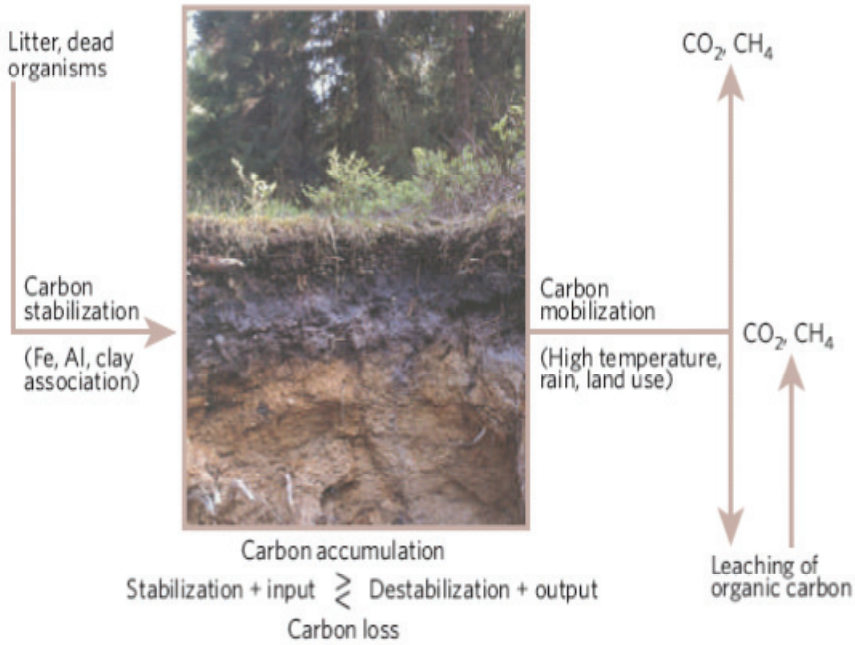

Figure 1. Soil C in context (from Freibauer, 2005)

Nitrous oxide is formed primarily from nitrification and denitrification processes, its fluxes from agriculture soils (0.53 Pg C equivalents y^{-1}) account for more than 50% of the global anthropogenic N_2O flux. The majority (52%) of the methane flux from agriculture arises from enteric fermentation in ruminants, whit biomass burning (19%) and animal waste treatment (8%) accounting for other significant proportions (Robertson *et al*., 2004).

Since C storage and trace gas fluxes from natural ecosystems offer less greenhouse mitigation potential and are harder to manage, is important to deepen the role of agricultural soils. In arable soils harvest reduces C returns to the soil, whereas C losses may be enhanced because of agricultural practices such as tillage. Thus, land conversion from other land uses to cropland is likely to lead to an overall decline in soil C. This soil may have the role of sources and sinks for CO_2 and other greenhouse gases and this depends critically by the soil management. The greenhouse gas mitigation potential for agricultural soils results from reducing emissions or from increasing C inputs.

It is possible to get soil C sequestration by increasing the net flux of C from the atmosphere to terrestrial biosphere by increasing global net primary productivity (NPP) (thus increasing C inputs to the soil), by storing a larger proportion of the C from NPP in the longer term C pools in the soil, by adding additional C containing materials to the soil (such a manure or cereal straw) or by slowing decomposition. Regarding soil C sinks the best solutions are to increase C stocks in soils that have been depleted in C, i.e. agricultural soils and degraded soils, since the capacity for increasing C storage is greatest in these soils (Lal, 2004a).

From CO_2 this entails reducing the CO_2 efflux from the soil or sequestering C in the soil. For N_2O, this entails reducing N_2O emissions. For CH_4 this entails reducing CH_4 emission from soils emitting CH_4 (e.g. rice paddy soils) and maximizing the methane oxidation potential of other soils (Smith *et al*., 2001). The agricultural soils can be a large source of CO_2 (Janssen *et al*., 2003). There is significant potential to reduce the efflux of C from agricultural soils and to sequester C in them.

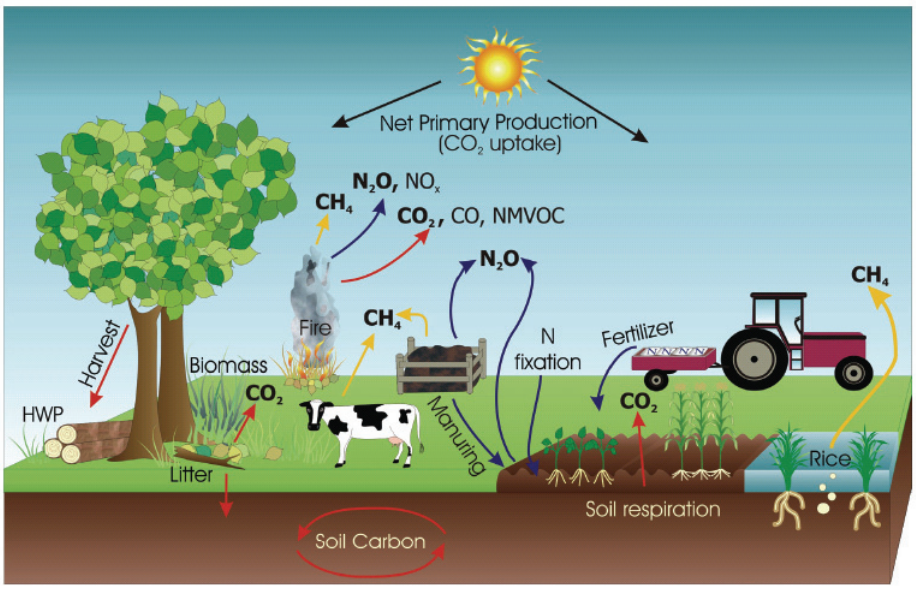

Figure. 2 The main greenhouse gas emission sources/removals and processes in managed ecosystems. (from IPCC AFOLU, 2006-)

Estimates of the potential for additional soil C sequestration vary widely. The most recent global estimate shows 0.9 ± 0.3 Pg C y^{-1}; over 50 years, this level of C sequestration would potentially restore a large part of C lost from soils historically (Lal, 2004a). However, soil C sequestration rates have a limited duration and cannot be maintained indefinitely.

Robertson et al. (2004) insisted on need for a systems approach for assessing greenhouse gas mitigation potential in agriculture. Increasing soil C stocks in the soil through reduced tillage can lead indeed to anaerobic zones in some soils and thereby increase N_2O emissions (Smith *et al.,* 2001; Six *et al.,* 2004). Also, management to reduce CH_4 emissions in rice paddy fields might increase N_2O emissions. Trade-offs between the greenhouse gases are complex, but should always be considered (Robertson *et al*., 2004).

Smith (2004b) recently calculated how future C emissions and CO_2 stabilization targets might influence the relevance of soil C sequestration as a greenhouse gas mitigation measure. The future trajectory of C emissions over the next century depends upon many factors.

The IPCC developed a range of standard reference emission scenarios (SRES) to provides estimates of possible emissions under a range of different possible futures. These possible future depend upon the degree to which greenhouse gas mitigation policies become global and upon whether environmental or economic concerns take precedence over the next century (IPCC, 2000).

C emissions gaps by 2100 could be as high as 25 Pg C y^{-1} meaning that the C emission problem could be up to four times greater than at present. The maximum annual global C sequestration potential is 0.9 ± 0.3 Pg C y^{-1} meaning that even if these rates could be maintained until 2100, soil C sequestration would contribute only a maximum of 2 to 5 % towards reducing the C emissions gap under the highest emissions scenarios (Lal, 2004a).

Considering the limited duration of C sequestration options in removing C from the atmosphere, we see that soil C sequestration could play only a minor role in closing the emissions gap by 2100. It is clear that if we wish to stabilize atmospheric CO_2 concentrations by 2100, the increased energy demand can only be supported if there is large-scale switch to non-C emitting technologies for producing energy. Given that soil C sequestration can play only a little role in closing the C emissions gap by 2100 still retains an important utility for C sequestration in future climate mitigation: if atmospheric CO_2 levels are to be stabilized at reasonable concentrations by 2100 (e.g. 450-650 ppm), drastic reductions in emissions there will be over the next 20 to 30 years.

During this critical period, the sum of all measures to reduce net C emissions to the atmosphere will play an important role, there will likely be no single solution (IPCC, 2000a).

Given that soil C sequestration is probably to be most effective in its first 20 years of implementation, it should form a central role in any portfolio of measures to reduce atmospheric CO_2 concentration over the next 20 to 30 years while new energy technologies are developed and implemented (Smith, 2004b).

The potential of soil organic C (SOC) sequestration is finite in magnitude and duration. It is only a short-term strategy to mitigating anthropogenic enrichment of atmospheric CO_2. The atmospheric concentration of CO_2 at the observed rate of 1990 (3.2 Pg C y^{-1}) will continue to increase at the rate of 2.0–2.6 Pg C y^{-1} even with soil C sequestration. Thus, a long-term solution lies in developing alternatives to fossil fuel. Yet, SOC sequestration buys us time during which alternatives to fossil fuel are developed and implemented. It is a bridge to the future. Soil C sequestration is something that we cannot afford to ignore (Lal 2004a).

1.1. Policy needs: the key role of soils in the Kyoto Protocol

Climate change is widely recognized as one of the most critical challenges the world has ever faced. For several years now the European Union has been committed to tackling climate change both internally and internationally and has placed it high on the EU agenda, as reflected in European climate change policy. In the political arena, soil C sequestration is perhaps more hotly debated than in the purely scientific arena. Terrestrial sinks have received close political scrutiny since their inclusion in the Kyoto Protocol at the Fourth Conference of Parties (COP4) to the United Nations Framework Convention on Climate Change (UNFCCC) (Smith 2004b).

Under the Kyoto Protocol international treaty, developed countries must take domestic action to reduce greenhouse gas emissions; the Annex B lists the Quantified Emission Limitations or Reducing Commitments for the parties that ratified the UNFCCC. The European Union is committed to an 8% reduction in CO_2 emissions compared to baseline (1990) levels during the first commitment period (2008-12). Soils can be used as biospheric C sinks under the Kyoto Protocol and soil organic C is potentially an important global C sink, if the sink can be maintained.

Several topics arise in the protocol that are relevant to the soil sequestration strategy. Some of these topics are the following: eligibility of sinks, treatment of soils, international trading, verification, leakage, and baselines.

- Eligibility of Sinks

Although the prospect of implementing the Kyoto Protocol raises many complex issues, the Protocol does establish international concurrence on some basic ideas and principles. For example, the Protocol recognizes that net emissions may be reduced either by decreasing the rate at which greenhouse gases are emitted to the atmosphere or by increasing the rate at which greenhouse gases are removed from the atmosphere using sinks. The focus is on agricultural soils as a sink. Even without ratification the Protocol helps to define this setting. The Kyoto Protocol clearly establishes credits for C sinks in Article 3.3. Three principles appear to underlie the sink provisions in the protocol and supporting documents.

First, credits are to be limited to activities undertaken purposefully. Second, credits are to be granted only for those items that could be accurately and reliably measured. Third, countries are not allowed to meet their full commitment through sink enhancement, emission reductions must also be pursued.

- Treatment of Soils

In terms of sinks, the Protocol permits credits for a limited list of activities (Article 3.3) while making it possible for additional activities to be added to this list later (Article 3.4). The Article 3.3 sinks involve afforestation and reforestation since 1990, and prescribe debits for deforestation. However, even there the role of forest soils is somewhat ambiguous. The Protocol does not clearly indicate that soils are part of the forest. In fact, the Protocol fails to define 'forest', 'afforestation', and 'reforestation'. The Intergovernmental Panel on Climate Change has prepared a Special Report on Land Use, Land Use Change, and Forestry (IPPC, 2000) that explores these and other matters. That report generally does assume that forest soils should be considered as part of the forest. Agricultural soils are treated in Article 3.4 but only as a possible item for future inclusion. Specifically, the Protocol states that the Conference of the Parties will 'decide upon modalities, rules and guidelines as to how, and which, additional human-induced activities related to changes in greenhouse gas emissions by sources and removals by sinks in the agricultural soils and the land-use change and forestry categories shall be added to, or subtracted from, the assigned amounts.

- International Trading

Sinks are not only relevant within a country, to offset its own emissions. The Protocol establishes the principle of international emission permit trading, which at least partially extends to sinks. Article 6 discusses Joint Implementation, whereby one Annex I country1 can pursue projects in another Annex I country and use the C benefits toward its own emissions commitments. Article 12 discusses the Clean Development Mechanism, under which projects can be sponsored by an Annex I country in a non-Annex I host. Article 17 discusses trading of emissions credits among Annex I countries. Article 6 specifically mentions 'removals by sinks' but this language is missing from Article 12 and coverage needs to be established. Transfer of emissions credits under either Article 6 or Article 17 would not create additional global emissions permits. What appears in the accounts of one party must be subtracted from the other. Transfer of emissions credits under Article 12 would create additional global emissions permits since a country with emissions restrictions would obtain additional emissions permits from a country without such. An issue open to discussion with respect to international trading is the allocation of responsibility for failure to sequester C. Does responsibility fall to the buyer or the seller? Sequestration in a non-Annex I country under Article 12 raises additional questions. Presumably this sequestration would be undertaken under a defined contract and the sponsoring country would receive emissions permits as C was sequestered in the host country. After

expiration of the contract, however, the host country would have no emissions limits under the Kyoto Protocol and the question is whether there is any liability if the sequestered C is subsequently released. Strategies for renting emissions permits are poorly developed but are beginning to be discussed. There are a number of related concerns about international leakage under a system in which some, but not all, countries participate.

- Monitoring and Verification

A recurring theme in the Kyoto Protocol is monitoring and verification of C emissions and sinks. Potential sinks must somehow be internationally certified with changes in C stocks monitored. What is of particular importance here is that the Kyoto Protocol includes a variety of mechanisms that permit, and would surely stimulate, trading in C emissions permits. C sequestered by one Party could be used to offset emissions in another sector of the national economy (and thus help to meet national commitments under the Kyoto Protocol) or it could be traded or sold to a Party in another country to use in fulfilling its national commitments. In order for a viable market in C credits to develop there needs to be a commodity that can be clearly identified and reliably and consistently measured in a country wide setting, not just on a project by project basis. We also note the possibility that the quantity of C credits generated by an endeavour might depend on the uncertainty of sequestration achievement. For example, Canada (1998) has outlined a proposal in which the amount of C sequestered by a mitigation measure would be reported along with the uncertainty in this measurement. Credits could be claimed only to the extent that there was 95% certainty in the amount of C sequestered. Under this procedure, the greater the uncertainty, the lower the mitigation credit that could be claimed in order to be 95% confident that the achievement was indeed at or above the credited amount.

- Leakage

An important concept in terms of C credits for sinks is that of leakage. Much of the implementation of C sinks will occur through specific C enhancement projects. When a project is implemented it may stimulate C gains through its own activities, but there is the possibility of offsetting C losses through changes in activity outside of the project focus area. For example, C sequestration end eavors involving transfer of land from agriculture to forestry might stimulate substantial countervailing land transfers out of forestry, thus offsetting the C gains. Such losses are called leakage. C credit projects will need to account for leakage due to altered economic activity in other parts of the economy.

Countrywide accounting may sometimes be required. Furthermore, in international C trading, the accounting system will need to consider leakage in the source and host countries.

However, the leakage issue is further complicated by the fact that some changes in C stocks are reportable under the Kyoto Protocol while others are not.

- Baselines

Finally, the Protocol raises the issue of baselines. Most notably, under Article 12 emissions credits are required to arise from 'reductions in emissions that are additional to any that would occur in the absence of certified project activity'. This would require that countries not only monitor and verify the C that has been sequestered, but that they measure a baseline of C that would have been sequestered without the project. Measurement of C stock changes along the path-not-traveled might rely on modeling or control plots. In either case, countries would have to distinguish between what did happen and what would likely have happened. Baselines are likely to be an issue under Article 3.4 of the Protocol as well (Marland *et al*., 2001).

As exposed before, land-use/ land-management change and forestry activities, that are shown to reduce atmospheric CO_2 levels, can be included in the Kyoto emission reduction targets. Soil C changes occurring during afforestation, reforestation or deforestation are included under Article 3.3 of the Kyoto Protocol, while soil C sequestration in croplands, grazing lands, in managed forests and in land subject to revegetation is included under Kyoto Article 3.4. The additional C emissions caused by land-use change (e.g. due to deforestation) have to be accounted. These C emissions as well as the offset by C sequestration have to be reported as part of national greenhouse gas inventories. The respective guidelines (IPCC, 2006) define precise rules for preparing annual greenhouse gas inventories in the Agriculture, Forestry and Other Land Use (AFOLU) sector.

To prepare inventories for the AFOLU sector, emissions and removals of CO_2 and non-CO_2 greenhouse gases are estimated separately for six land-use categories: Forest Land (FS), Cropland (CL), Grassland (GL), Wetlands (WL), Settlements (SL) and Other Land (OL). Each land-use category is further subdivided into land remaining in that category (e.g., Forest Land Remaining Forest Land) and land converted from one category to another (e.g., Forest Land converted to Cropland). Countries may choose to further stratify land in each category by climatic or other ecological regions, depending on the choice of the method and its requirements.

Within each land-use category, C stock changes and emission/removal estimations can involve the five C pools that are defined in Table 1. The emissions and removals of CO_2 for the AFOLU sector, based on changes in ecosystem C stocks, are estimated for each land-use category and C stocks changes within a stratum are estimated by considering C cycle processes between the C pools (*Fig 3, 4*):

$$\Delta C_{LUi} = \Delta C_{AB} + \Delta C_{BB} + \Delta C_{DW} + \Delta C_{LI} + \Delta C_{SO}$$

Where ΔC_{LUi}: C stocks changes for a stratum of a land-use category. Subscripts denote the C pools described in *table 1*.

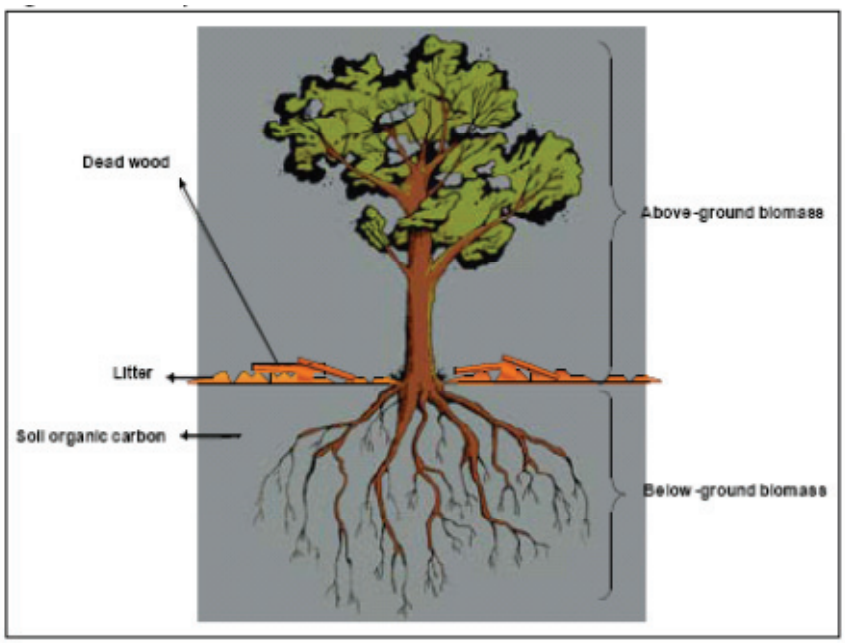

Figure 3: The five C pools indicates by the AFOLU guidelines.

According to relevance, data and model availability, as well as resource and capacity to collect and analyze additional information, the reporting country has to choose between different tiers in estimating changes in C pools and fluxes. In general, moving to higher tiers improves the accuracy of the inventory and reduces uncertainty, but the complexity and resources required for conducting inventories also increases for higher tiers, and if needed, a combination of tiers can be used.

Table. 1 Definition for C pools used in AFOLU for each land use category

POOL		DESCRIPTION
Biomass	Above ground biomass (**AB**)	All biomass of living vegetation, both woody and herbaceous, above the soil including stems, stumps, branches, bark, seeds and foliage
	Below ground biomass (**BB**)	All biomass of live roots. Fine roots of less than suggested 2 mm diameter are often excluded because these often cannot be distinguished empirically from soil organic matter in litter
Dead organic matter	Dead Wood (**DW**)	Includes all non-living woody biomass not contained in litter, either standing, lying on the ground, or in the soil. Dead wood includes, wood lying on the surface, dead roots, and stumps, larger than or equal to 10 cm in diameter
	Litter (**LI**)	Includes all non living biomass with a size graer than he limit for soil organic matter (suggested 2 mm) and less than the minimum diameter chosen for dead wood (e.g. 10 cm), lying dead, in various states of decomposition above or within the mineral or organic soil. This includes the litter layer as usually defined in soil typologies.
Soils	Soil Organic Matter (**SO**)	Includes organic C in mineral soils to a specified depth chosen by the country and applied consistently through the time series. Live and dead fine roots and DOM within the soil, that are less than the minimum diameter limit (suggested 2 mm) for roots and DOM, are included with soil organic matter where they cannot be distinguished from it empirically

The guidelines distinguish between lands that remain in the same category but are affected by management changes and lands which are converted to another category. Changes in managements or land-use category are called *activities* (reported in area) and have to be multiplied with an emissions factor that describes the source/sink strength of each kind of activity (reported in t CO_2 per area and year) (IPCC, 2007).

Seen the important role for C sequestration in climate mitigation in the next 20 to 30 years, it will be necessary to accurately monitor the amounts of C sequestered in order to estimate its role in closing the C emission gap (Aalde *et al*., 2006).

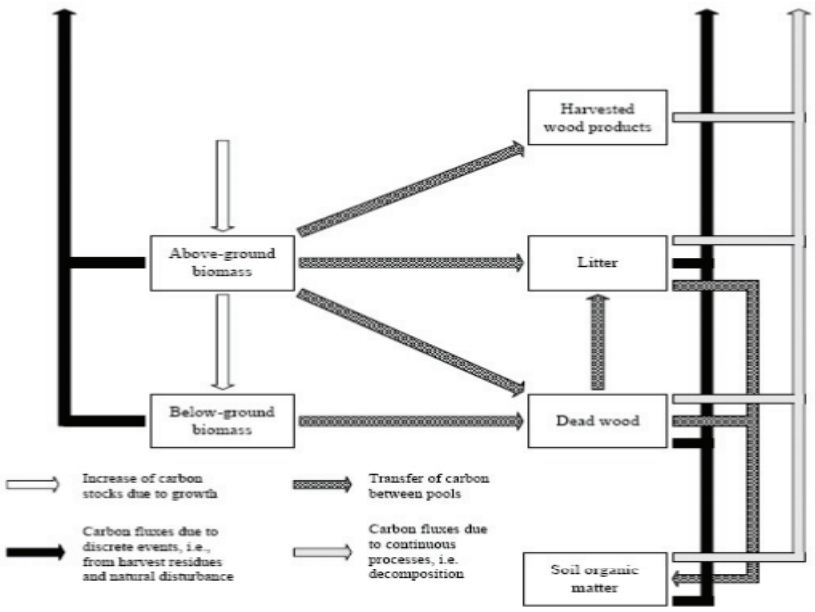

Fig. 4: Generalized C cycle of terrestrial AFOLU ecosystems showing the flows of C into and out of the system as well as between the five C pools within the system. (from Aalde, 2006)

The soil C sequestration strategy is the common link between three UN Frameworks Conventions: Climate Change, Biodiversity and Desertification Control. Soil C sequestration can reduce the rate of enrichment of atmospheric CO_2 by 0.6 to 1.2 Pg C y^{-1} while enhancing biodiversity and controlling desertification. Restoration of degraded/ desertification soils and ecosystems is an important ancillary benefit of soil C sequestration.

Of all the ancillary benefits, the impact of soil C sequestration in advancing global food security can neither be ignored nor overemphasized. Severe soil degradation is a common problem among all regions threatened by food insecurity including sub Saharan Africa, Central and South Asia, the Andean Regions etc.

There is a important link between soil C sequestration and global food security. Soil C sequestration indeed is an important ancillary benefit of the inevitable necessity of enhancing soil quality for feeding the global population of 6 billion in 2000, which is expected to reach 7.5 billion by 2020 and 9.4 billion by 2050 (Kutsch *et al*., 2009).

2 The soil carbon dynamics

Soil science studies the properties physical, chemical, biological and soil fertility, and these properties in relation to the use and exploitation of terrestrial soil more recently, soil science focused also in the role soil in a context of increasing atmospheric CO_2 concentration and the resulting greenhouse effect.

The soil science is important for diagnostics and maintenance of soil health (or fertility): indeed the soil health is defined as the continued capacity of soil to function as a vital living system, by recognizing that it contains biological elements that are key to ecosystem function within land-use boundaries These functions are able to sustain biological productivity of soil, maintain the quality of surrounding air and water environments, as well as promote plant, animal, and human health (Kutsch *et al*., 2009).

Knowledge of the dynamics of soil C is essential for a better understanding of the terrestrial C balance. The pools of actively cycling C are atmospheric CO_2, biota (mostly vegetation), soil organic matter (including detritus), and the ocean. Of these, the oceans contain the largest reserves of C (about 39,000 Pg C) though most of this (all but about 1000 Pg C) is in deep ocean layers and not in active circulation, at least in times measured in human generations (Janzen, 2004). World soils represent a major component of the global C cycle, because the soils are the second largest terrestrial C reservoir, yet because the soil–plant system and the pedosphere-atmosphere interface are sites of intense C exchange with 10% of atmospheric C passing through soils annually, and finally for the capacity of soils for long-term storage of photosynthetically fixed C (Jacinthe *et al*., 2002).

All of these C pools (the atmosphere, vegetation, soil, and ocean) are connected. Atmospheric CO_2 enters terrestrial biomass via photosynthesis, at a rate of about 120 PgC per year (gross primary productivity). But about half of that is soon released as CO_2 by plant respiration, so that NPP is about 60 Pg C per year. This amount is stored at least temporarily in vegetative tissue, but most eventually enters soil upon senescence. At the same time, heterotrophic respiration (largely by soil microorganisms) and fire return an amount roughly equivalent to NPP (60 Pg C per year) back to atmospheric CO_2, closing the loop. Averaged over the total area of continents, these C inputs and losses amount to about 4Mg C ha^{-1} per year (*Fig. 5*). Exchange of CO_2 between atmosphere and ocean are even larger than those between air and land ecosystems (about 90 Pg C per year, both ways).

But a surprisingly large exchange also occurs via biological processes; though ocean biomass is a mere 3 Pg C, its NPP almost equals that of all land plants, with a mass of about 600 Pg C. The C flows among the various pools, and their feedbacks, have kept the global C cycle quite stable for millennia at least, until recent decades when human activity began exerting increasing stresses on the C cycle (Janzen, 2004).

Soil organic C is the largest C reservoir in many terrestrial ecosystems including grasslands, savannas, boreal forests, tundra, some temperate forests, and cultivated systems, comprising as much as 98% of ecosystem C stocks in some systems. The global soil C pool of 2500 gigatons (Gt) includes about 1550 Gt of soil organic C and 950 Gt of soil inorganic C (SIC) (Conant and Paustian, 2002).

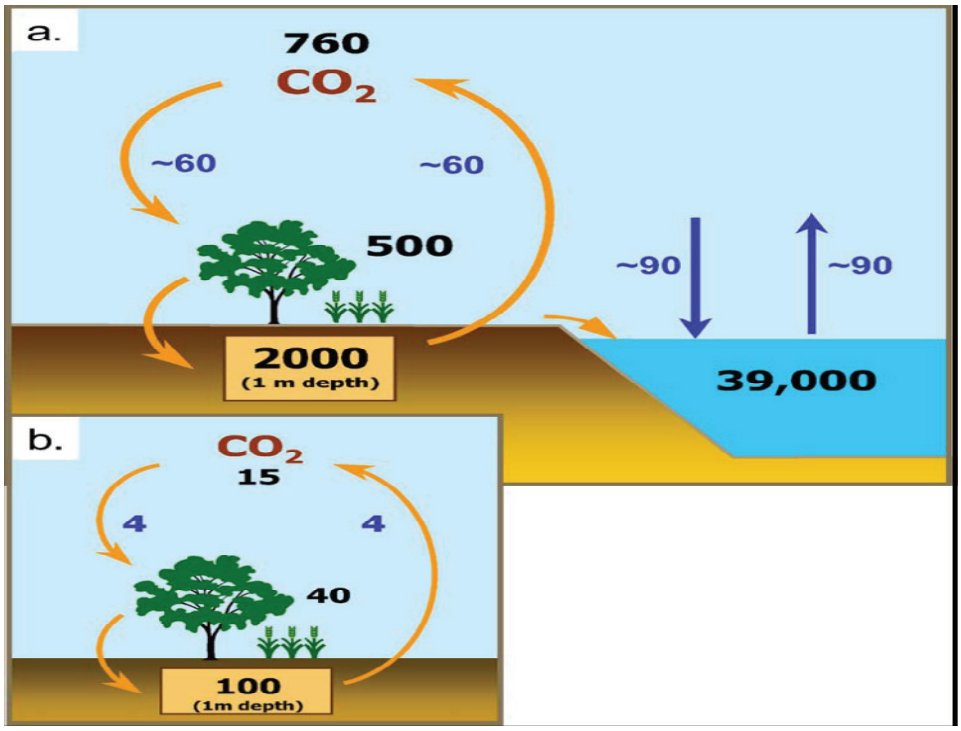

Figure 5 An overview of the global C cycle, as it was in the 1990s 2001 a) All C stocks are in units of Pg C, and flows are in units of Pg C per year. (b) Global C stocks and flows, expressed as averages per ha of continental area. Values are expressed in units of $MgCha^{-1}$ or $MgCha^{-1}$ per year.

The soil C pool is 3.3 times the size of the atmospheric pool (760 Gt) and 4.5 times the size of the biotic pool (560 Gt,). The SOC pool to m^{-1} depth ranges from 30 tons/ha in arid climates to 800 tons/ha in organic soils in cold regions, and a predominant range of 50 to 150 tons/ha. The SOC pool represents a dynamic equilibrium of gains and losses (*Fig. 6*) (Lal *et al.*, 2004).

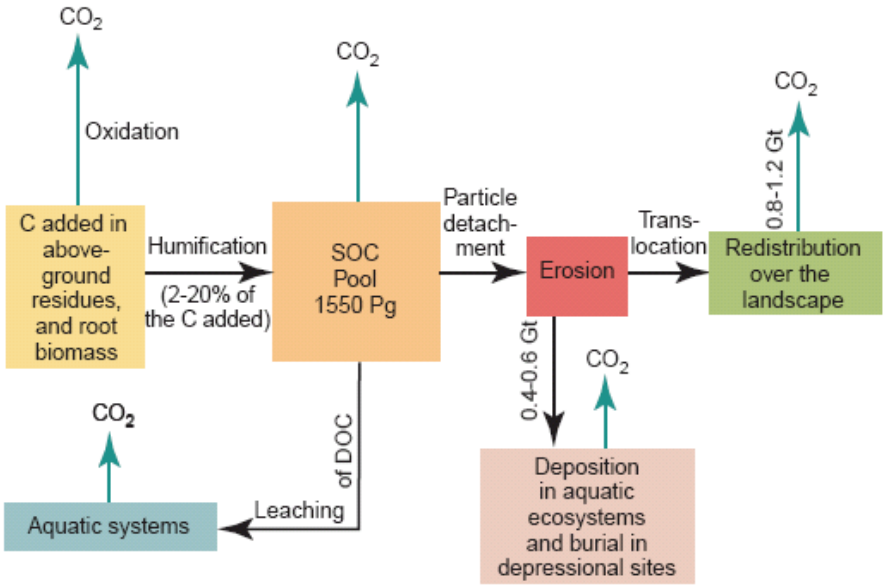

Figure 6. Processes affecting SOC dynamics. There are indicated emissions of CO_2 into the atmosphere.

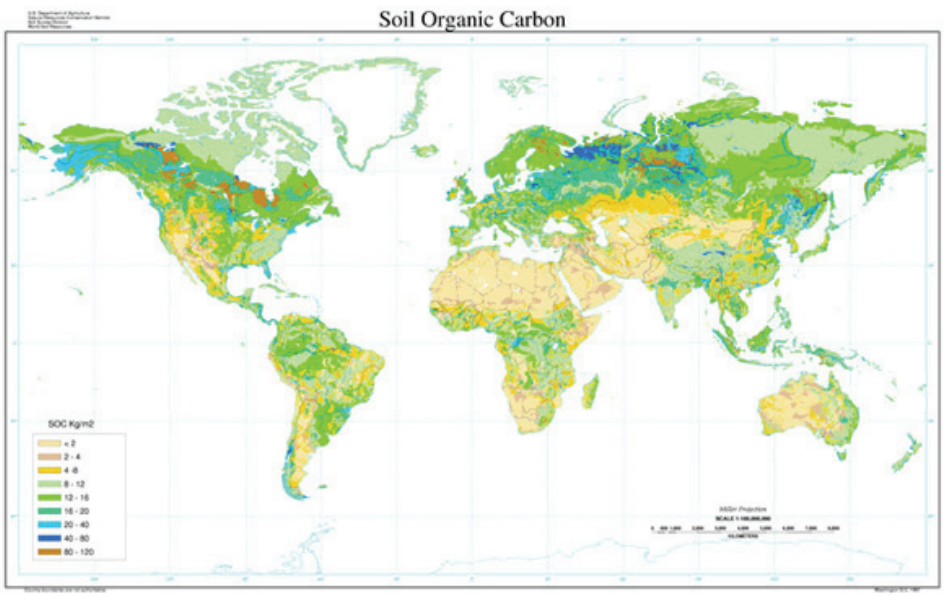

Figure 7 The global soil organic C map derived from International Satellite Land Surface Climatology Project (ISLSCP) dataset

About one third of organic soil C occurs in forests and another third in grassland and savannas, the rest in wetlands, croplands and other biomes. The global soil organic C map (*fig 7*) shows the areas of high soil organic C predominantly in cold boreal (e.g. Northern Canada) and warm and humid tropical regions (e.g. South East Asia), reflecting areas of deep organic soils (i.e. peat lands). However even temperate zones, for example the United Kingdom, can contain considerable amounts of soil organic C in wet and cold upland regions. Most of the soil organic is not inert, but in a continuous dynamic state of accumulation and decomposition (Janzen, 2004), the schematic soil C cycle in *fig 8* indicates this continuous exchange of C between the soil and the atmosphere, mostly as C dioxide (CO_2) and methane (CH_4). Consequently, any net C loss from soils will increase the CO_2 concentration in the atmosphere and in water bodies, whereas net accumulation in soil C (or sedimentation in rivers or lakes etc.) can contribute to the reduction of the atmospheric C pool (Ellert *et al*., 2001; Lal, 2004). This cycling of C is increasingly influenced by human activities (IPPC, 2007). On an annual basis, global soil respiration estimates amount to about 80 Pg C (Schlesinger and Andrews, 2000; Raich *et al* ., 2002), roughly ten times the annual flux from fossil fuel combustion (7.2 Pg C; IPCC, 2007). Crucially, past and current cultivation of soils led to significant soil C losses of 50 Pg or more; conversely land-use or management change can offer an opportunity for sequestering atmospheric C in soils (Janzen 2006). Importantly, in the long term, these soil C changes can be greater than any above-ground C gains.

As even small changes in soil organic C pools, due to climate changes or to human activities, might have large impacts on the global C cycle, it becomes vital to assess soil organic matter pools and their changes accurately.

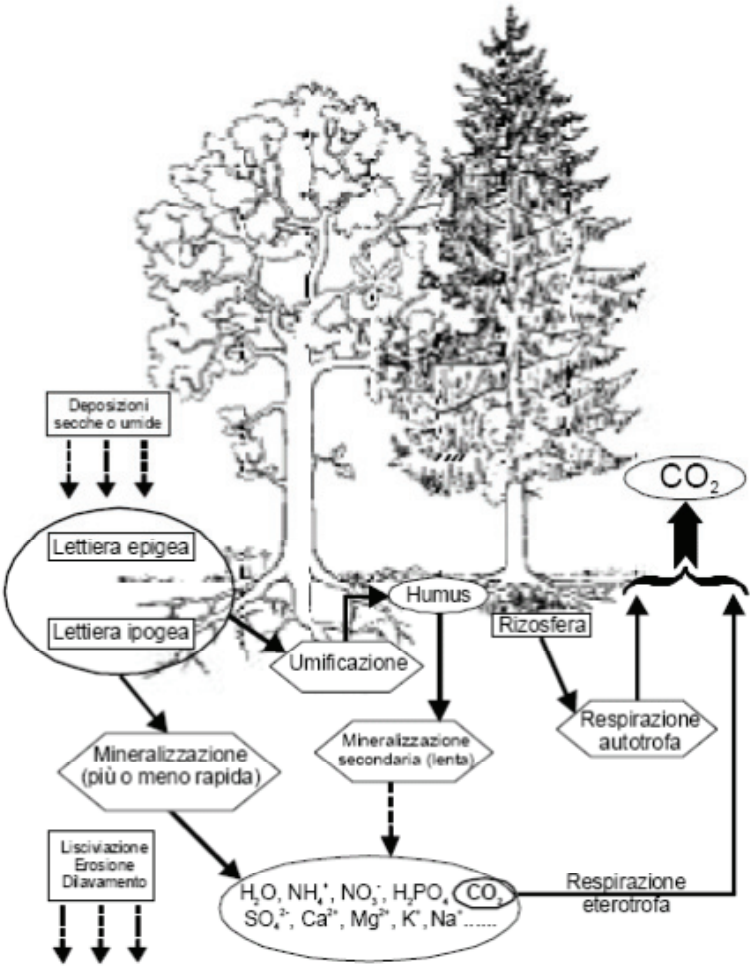

Figure 8 Schematic soil C cycle (from Rodeghiero 2006)

SOC is the C content of SOM, which is mostly around 50 to 60%. Most soils contain large amounts of SOC in the litter and organic layers. Commonly soils are characterized by a SOM rich topsoil with decreasing soil organic C content down the soil profile, its thickness determined by soil mixing. In Histosols organic matter layers can extend to many metres depth. The simplest way to report SOC content is a concentration (i.e. mass of C per unit mass of soil; g kg^{-1}). More frequently SOC is expressed on an area (Kg m^{-2}) or volume (Kg m^{-3}) basis, although for inventory purposes SOC concentration, unless specified as per total soil volume, should be defined as C density (e.g. kg C m^{-2} per unit of soil depth), which requires data on soil bulk density, stone content, depth of sampling and root content.

The total amount of organic C stored in a soil is the resulting net balance of all C fluxes entering and leaving the soil over time (*fig 8*). Most organic C matter entering the soil originates

from plants. Above ground litter, mainly derives from plants. Above-ground litter mainly derives from woody tissue (e.g. twigs, bark) leaves, flowers, fruits, mosses, lichens and fungi; below-grounds inputs include dead roots and their associated mycorrhizal hiphae, root exudates (mainly sugars and organic acids) and sloughing of root surface tissues. However, any above-ground litter will eventually become below-ground input, for example due to bioturbation and cultivation. Although the living fine root biomass constitutes only a small fraction of the total stand biomass, the contribution of fine roots to total stand biomass, the contribution of fine roots to total soil C inputs can be quite substantial. For example, fine root production in forest ecosystems can be assumed to be around 50 to 75% of total net primary productivity (Majdi, 2001; Nadelhoffer and Raich, 1992).

In forest ecosystems, this is due to the high biomass of woody tissue relative to production above ground, coupled with fast rates of growth and turnover relative to below-ground biomass. Some recent studies indicate that roots and mycorrhiza are even more important for the accumulation of SOM than above-ground leaf litter. Other considerable source of SOM can be animal faeces, dead animals and wet or dry organic matter soil surface deposition. At agricultural sites, applications of fertilizer including organic matter such as manure or compost can significantly increase the SOM pool. Furthermore, there is evident that a small amount of CO_2 can be directly fixed by microbes in the soil, contributing to SOM

In natural ecosystems, losses of C from the soil derive mainly from decomposition and mineralization processes of SOM, which lead to the release of CO_2 and some other trace gases (CH_4 and CO). The production of methane is of importance in particular in wetland and peatlands, varying between less than 1 to more than 40 g C m^{-2} y^{-1} (Miltner *et al*., 2005).

Leaching can also cause C losses from any soil via dissolved organic C (DOC) and dissolved inorganic C (DIC). These losses, although high during peak rainfall events (Freeman *et al*., 2004), are generally small and account for up to 20 g C m^{-2} y^{-1} for DOC and DIC in Europe

The largest annual flux components is generally heterotrophic soil respiration (calculated as the difference between net primary productivity and net ecosystem productivity), estimated to range between 280-970 g C m^{-2} in temperate forest ecosystems (Pregitzer and Euskirchen, 2004). Still, hydrological fluxes of C can be significant, especially in relation to a sometimes quite small net ecosystem exchange (Fahey *et al*., 2005). In arable or other highly disturbed systems, a depletion of soil C also occurs due to lower plant litter input through harvesting, faster SOM breakdown (as it is more accessible in soil aggregates disrupted by

ploughing) and increased displacement of C-rich surface soil through wind and water erosion. In fact erosion can lead to considerable C losses at the plot scale, but net effect are harder to predict when up-scaling since it depends on the fate of the C in the deposition area. It is estimated that about 1.14 Pg C may be annually emitted into the atmosphere through erosion-induced processes (Janzen, 2006).

All the ecosystems in the world will be likely to experience fire activity. Mediterranean ecosystems, for example, are fire dominated and have been subjected to fire for centuries (Duguy *et al*., 2007). The fires in tropical savannas (grassland, woodland and forest) are principal sources of CO_2 emission to atmosphere; in these ecosystems, in fact, fire is commonly used to convert forest to agriculture purposes. In forest ecosystems prescribed fires of low intensity are sometimes used to consume the understorey and part of the forest floor layers to prevent fuel build-up and resulting large-scale forest damage (Ferran *et al*., 2005). Fire events mostly consume all litter as well as part of the organic matter; depending on the intensity, this "burning" can reach quite low into the mineral layers (Duguy *et al*., 2007) or also peat (Page *et al*., 2002). Moreover, heating of soil layers alters their physico-chemical properties (Neary et al., 1999) and affects soil nutrients.

During a fire, soil nutrients are either volatilised or transformed into ashes and, consequently, after a fire there is generally a net loss of nutrients from the ecosystem due to export through leaching, wind blow or erosion. However, in the short term soil fertility usually increases after fire due to a higher nitrogen and phosphorous availability in soil solution. Due to the increased pH and concentration of base cations soil respiration is temporally enhanced, indicating higher nitrogen mineralization and nitrification.

The magnitude of the nutrient flush depends on the temperature and duration of the fire and the amount of organic matter burned. Fire indeed also alters net assimilation rates of the standing vegetation (i.e. leaf damage or loss) causing a decrease in the supply of organic matter and root-derived substrates to the soil for quite some time (Duguy et al., 2006). In particular, in organic soils fires can smoulder underground for long periods leading to considerable C losses (Page *et al*., 2002).

Losses of organic C and nitrogen caused by volatilisation are proportional to fuel consumption or fire intensity or both. Moreover larger fuel loads result usually in more intense fire and thus larger losses of organic C and nitrogen. The heat transfer into mineral soil during fire is strongly limited by the insulating effect of the soil (Duguy et al., 2007). For this reason the main short-term effects of fire are expected to occur in the upper 0 to 2 cm layers of mineral

soil (Debano and Corrad, 1978). Duguy et al., (2007) reported a significant 25% decrease in soil organic matter C 4.8% to 3.6% due to recurrent fires in a shrubland ecosystem in eastern Spain.

Finally, fire in the forest floor and superficial soil layers always decreases C contents, whereas losses from the mineral layers depend on fire intensity, frequency and insulating layers.

2.1. The soil carbon pools

The soil consisting of a mixture of organic and mineral particles, soil solution and air, resulting from the interaction between biotic and abiotic factors; it is the medium in which plants acquire water and nutrients through their root systems. Soil organic compounds include all the organic matter present in the soil: living organisms (bacteria, algae, fungi, soil fauna and plant roots), the organic products derived from their activity (faeces, bacterial and fungal synthesized substances and root exudates) and plant and animal remains at various stage of decomposition, passing through several soil horizons and eventually entering various humus stages. Excluding living organisms (including living roots), we can define SOM as: "the plant and animal remains at different stages of decomposition and the substances derived from the biological activity of the soil-living population" (Rodeghiero 2006).

SOM consists of many classes of compounds with different biochemical properties and varying degrees of association with the mineral matrix which differentially contribute to the overall dynamics of total SOM.

Most model of SOM dynamics divide SOM into several kinetic compartments (e.g active, slow and passive, according to the Century model) based on assumed turnover times, and define their dynamics in terms of transfers form one compartment to another (Parton et al., 1987).

When the organic material enter the soil is subject to biological decay by various types of soil organisms including macro-and megafauna (e.g. earthworms, snails and larvae), mesofauna (e.g. springtails and mites) or microfauna (e.g. nematodes and protozoa), as well as bacteria and fungi. Abiotic chemical oxidation is only of minor importance (Von Lützow *et al*., 2006). The time organic matter remains in the soil depends on a number of site specific parameters of which soil temperature, moisture and substrate quality and supply can be seen as most important, as they together determine microbial activity. While probably all SOC can potentially be degraded by microbes, the rate at which it is mineralised also depends on the chemical nature of the organic material itself, its protection against microbial attack and its variability (e.g. litter mixtures might decompose faster; Briones and Ineson, 1996).

Biological fractionation separate SOM into labile and recalcitrant portion through microbial mineralization. The underlying assumption is that the microbial biomass will mineralize the labile portion first, leaving the recalcitrant portion behind.(Kutsh et al., 2009).

We can distinguish between primary and secondary chemical recalcitrance of organic matter (Von Lützow *et al*., 2006): *(1) primary recalcitrant* refers to organic molecules of plant origin, such as lignin, waxes, cutin or suberin, which are less easily degradable by soil organisms than starch or cellulose (Bertrand *et al*., 2006); *(2) secondary recalcitrant*, on the other hand, includes products and residues of soil fauna and microbes (such as murein, chitin, lipids and melanin), charcoal and humic polymers. If chemical recalcitrance were the main factor determining SOM accumulation, this would lead to a selective preservation of those molecules in old C fractions. Instead, a number of studies have shown that apparently old fraction of SOM still contain large amounts of labile organic matter, indicating that other stabilization mechanisms are also involved (Schoning and Kogel-Knabner, 2006), for example sorptive preservation of organic matter in association with certain minerals (such as allophone, poorly crystalline Al and Fe oxides and hydroxides) and metals (Kleber *et al*., 2005; Wiseman and Puttmann, 2005).

Occlusion of organic matter by soil aggregates (in particular in clay soils) can also be of considerable relevance, while intercalation within soil minerals does not seem to be quantitatively important (Von Lützow *et al*., 2006).

In soil C model (e.g. Century) such decomposition processes are represented in three arbitrary SOM pools, which only vaguely relate to measurable SOC fraction, with specific turnover rates and an additional clay decomposition modification factor (Rodeghiero, 2006).

In the soil are distinguished three pools of organic matter. The easily decomposable active soil organic matter and the more stable slow passive humic substances, also often called recalcitrant pool. These pool receive fresh organic matter via litter input but also from microbial turnover and other events such as fire. Litter and the resulting humic substances are decomposed, mainly by the bulk soil micro-organisms that comprise bacteria, fungi and soil meso- and macro-fauna, resulting in respirations and subsequent soil CO_2 efflux and changes in the chemical compositions of SOM. This biological activity is the driving force for the conversion of litter into stable humus and is also related to bioturbation (e. g. earthworm activity), which causes both aeration and SOM incorporation into clay minerals or deeper soil layers. Importantly, this biological activity, although crucial in determining SOM turnover and soil C storage capacity, is not yet adequately included in most modelling frameworks. As simple as measuring soil CO_2 efflux or determining the soil C stocks of a might appear, this basic research on soil C dynamics provides many challenges. In contrast to above-ground C relations, the world of soil C turnover and decomposition discovered within the 'hidden half' of terrestrial

ecosystems. It involves thousands of species of soil organisms, ranging from metre or centimetre to less than micrometre scale. It is surprising how little we know globally about those biological components that drive turnover rates of litter and organic matter. Many of the decomposer bacteria, for example, are still unknown, but their role is important in aerobic, and even more so in anaerobic, decomposition, Were are not yet to name most of them at species level or to group them into functional types as is commonly done when modelling plant C dynamics at the larger scales. About another important group, the soil fungi, we do not yet fully understand their life cycles, environmental responses or specific role in the C cycle.

Only recently, the advance of molecular techniques enabled us to realize lived of complexity of processes involved in the decomposition of organic matter, and the combination of these novel molecular techniques with stable isotope probing will hope full improve our understanding in the near future. In addition, the activity of below-ground decomposer organisms is influenced by many factors such as the amount and chemical quality of litter and soil organic matter, soil texture and porosity, pH value, input via litter fall and root turnover, root biomass and root activity. Also past and recent disturbances, such as fire or erosion as well ploughing, agro chemicals or acid deposition, can affect the soil C turnover substantially. Therefore, site inter-comparisons even for the same soil type are difficult to interpret. Moreover, our basic ability to adequately separate autotrophic and heterotrophic flux components . and their respective responses to en environmental changes is still limited. However, the proper link between above-and below -ground C flow in the plant-soil C flux continuum needs to be established, as can be seen in the continued debate on the existence and role of soil priming, which may alter turnover of older organic matter as a result of fresh plant-derived soil C input.

Finally, all these novel aspects need to be considered by the modelling community, for which the incorporation of basic soil biology still seems a major obstacle (Kutsch *et al*., 2009).

3. Soil organic carbon stock changes

All of the factors of climate change (raised atmospheric CO_2 concentration, temperature, precipitation) affect soil C, with the effect on soils of CO_2 being indirect (through photosynthesis) and the effects of weather factors being both direct and indirect. Climate change affects soil C pools by affecting each of the processes in the C-cycle: photosynthetic C-assimilation, litter fall, decomposition, surface erosion, hydrological transport. Due to the relatively large gross exchange of CO_2 between atmosphere and soils and the significant stocks of C in soils, relatively small changes in these large but opposing fluxes of CO_2 may have significant impact on our climate and on soil quality. Therefore, managing these fluxes (through proper soil management) can help mitigate climate change considerably. Can be used two hypotheses to predict the potential dynamics of soil C stocks in such a changing environment (Garten, 2004).

Based on the first hypothesis, the decomposition of SOM is more sensitive to temperature change than NPP consequently increasing temperatures will result in a net transfer of soil C to the atmosphere as SOM decomposition is stimulated more than NPP (Kirschbaum, 2000).

This would lead to a positive feedback mechanism: the release of soil C further increases atmospheric CO_2 concentrations, leading to even higher surface temperatures and consequently more soil C loss through accelerated decomposition. However, there is also a potential inhibitor to this positive feedback. Increased decomposition rates may stimulate greater soil nitrogen availability, leading to higher NPP, which potentially increases C inputs into the soil through litter fall and rhizodeposition that would somewhat offset the increased soil C loss. In this first hypothesis soil C dynamics depend on the soil C balance changes. Currently is not known in which direction this soil C balance will move (Davidson *et al*., 2006) but indications are for a net loss (Bellamy *et al*., 2005).

In addition it is inaccurate to suppose that an increase in net primary productivity will translate simply into additional below-ground storage. In fact they found that a doubled litter addition in an old growth forest in Oregon resulted in a much higher soil respiration than expected based on the additional C added (Rodeghiero 2006).

The second hypothesis further considers the additional effect of human produced nitrogen (N) fertilization from agricultural fertilizer usage and the combustion of fossil fuel the

former producing predominantly NH_3 (and NO_3), the latter NO_X ($NO+NO_2$), eventually leading to dry and wet N deposition on vegetation and soil (Cowling and Galloway, 2002).

Soil C stocks might then gain more C than they lose due to rising temperatures in a nitrogen-rich environment. Firstly, increased soil N availability would increase NPP and subsequently litter and thus soil C inputs (as seen in the first hypothesis). Secondly, although higher N content might stimulate initial litter decomposition it seems to suppress humus decay in later stages, thus leading to stabilization of SOM in mineral-associated fractions.

There are a strong correlation between the soil C stocks and temperature and in the certain regions temperature explains a large part of the variation in soil C stocks. Clearly other factors co-exist and also impact on soil C stocks. However, temperature seems to be one of the most important environmental factors (Rodeghiero 2006), and interestingly, for the UK analysis, this seems to be supported by measured losses in soil C stocks (Bellamy *et al*., 2005).

3.1. Methods for the soil carbon determination

The knowledge of temporal and spatial heterogeneity of soil properties, general environmental conditions, and management history are essential when designing methods for monitoring and projecting changes in soil C stocks. Measuring changes in soil C storage is complicated because a substantial fraction of it appears relatively inactive in the short term (i.e., few years). When short-term changes in soil C often occur in the relatively small light fraction, known for its temporal and spatial variability due to environmental conditions and management histories (Post *et al*., 2001).

The measurement of soil C stocks relies on fallowing a precise sample protocol (including backup samples for possible laboratory inter-comparison or later cross-validation) but there are several question with sampling of soil for analysis (i.e. sample depth, pooling of samples, bulk density and stone content). In general the best advice is to follow (inter)nationally accepted soil sample protocols, but specific attention should be paied to site- specific issues such as stoniness or SOM distribution within the soil profile (Rodeghiero 2006).

There are many methods available to analyze SOM and SOC concentration, each with advantages and disadvantages (MacDicken, 1997; Palmer, 2002, Bisutti *et al*., 2004). Because of many available methods, all with their specific advantages and limitations, it is vital to know how a sample has been processed when comparing datasets analyzed by different laboratories or inventories. In fact, applying the same protocols at different laboratories can deliver quite different results, exhibiting instruments- specific bias. This enables correction of laboratory-specific differences and is especially important if soil analyses of later soil inventories cannot be performed in the same laboratory or with the same instruments. Though this might sound meticulous, we have to consider the relatively small changes we intend to measure in large existing C stocks, which justifies avoiding any bias in sample analyses. Generally the obtained SOM values are then converted into SOC into SOC value using conversion factors. However, some concerns remain over how to convert SOM into SOC content accurately and it is good practice to include an internal laboratory standard in every sample batch analysis. Otherwise, the comparison of different techniques to determine soil C content can often only be done after a stratification of the samples at least by bedrock and texture. Therefore, it is good practice to store dried soil samples to enable analytical comparisons to be carried out at a later stage (as done by Bellamy *et al*., 2005) (Rodeghiero 2006).

3.2. Monitoring soil C at local, regional and global level.

Several approaches and tools will be required to develop reliable estimates of changes in soil C at scales ranging from the individual experimental plot (local level) to whole regional and national inventories.

At *local level* (i.e. where the mean change over time in a small area is target quantity) the spatial heterogeneity of soil organic C and its dynamic nature prevent direct detection of changes on annual or smaller time scales; stratified sampling (e.g. by soil or vegetation type) is one way to increase the precision of the estimates of mean change (Post et al., 2001).

At *regional level* differences in geology or topography can affect the variation in C content. A national or regional soil sampling project should be planned in detail from the beginning and, in particular, the objectives should be agreed and an acceptable level of confidence in the result defined. Following the methodology of designing sampling schemes and choosing the number and position of the sampling plots in De Gruijter *et al*. (2006) the choice is between a design-based and a model-based approach.

The design-based approach requires probability sampling (e.g. stratified random sampling) and should be used where the required result is an estimate of the frequency distribution of change in organic C or a parameter of this distribution (e.g. the mean change in organic C stock and its standard deviation).

The model-based approach, on the other hand, requires systematic sampling (e.g. grid-based sampling) and should be used when prediction of values at individual points or maps are required.

Regional estimates of C stocks can be obtained by extrapolation of data obtained at the plot scale or field scale. Usually the landscape is divided into homogeneous areas (i.e. stratification) with regard to management and/or soil environmental conditions. Estimates for each patch are then multiplied by its area and the sum of all the areas leads to the regional estimate. Each patch can be defined at various level based, for example, on soil, climatic or land-use maps, or other data of which there are spatial estimates. The more homogeneous the patches are, the less will be the variation in the estimate of the mean values for each patch. Possible factors for stratification could be: climate, aspect, elevation, soil type and texture. Specifically, significant aggregation errors might be introduced with varying soil texture inside

the investigated area. Indeed, silt and clay particles play important roles in physically protecting labile organic matter from decomposition (Post *et al*., 2001). After a suitable stratification, application of a geographical information system (GIS) can then help significantly with the extrapolation of data to the regional, national or even global scale using many layers related to topography, micro-climate, geology, vegetation and factors that could influence soil formation.

There two most important examples are presented as follows: Guo et al. (2006) created maps of soil organic C and inorganic C generated from the State Soil Geographical database (STATSGO) and overlain with land-cover, topography (elevation and slope), mean annual precipitation, and mean annual temperature, while Jones et al, (2005) used a methodology for obtained estimating organic C contents (%) in topsoils across Europe. The information presented in map form provides policy-makers with estimates of current topsoil organic C contents for developing strategies for soil protection at regional level.

GIS models can also be used together with digital elevation models (DEMs) in order to account for topography effects on surface area (Garnett *et al*., 2001). Also satellite images and remote sensing data can help with stratification and land-use change assessments.

Regional soil C investigations tend to consider only the soil to a depth of usually 30 cm (i.e. topsoil, equal to arable ploughing depth). The choice of sampling depth is determined by two considerations: (1) at a global level around 30 to 70% of the organic C is stored in the topsoil (Batjes, 1996) and (2) the topsoil is more frequently subject to anthropogenic and climatic perturbations and land-use change (Sun *et al*., 2004); moreover the organic matter stored in deeper soil layers has turnover times that are much longer than the life span of the above –starting crops or forests. However, some studies demonstrated that soil depth even below 30 cm can have a major impact on the estimates of C stock (e.g. Harrison *et al*., 2003). Therefore, sampling down the entire soil profile is more accurate, in particular for peat soils (i.e. to the mineral soil), as shrinking or expansion of peat due to water table changes alters soil C content per depth increment. However, even the mineral soil beneath a peat layer can store large quantities of organic C of up to 8 Kg C m^{-2} (Moore and Turunen, 2004).

At *global level*, since soils are very dynamic systems, spatial changes have to be considered when estimating C balance, as uncertainty increases with up-scaling and the necessary spatial and soil-depth related aggregations. Uncertainties are already large at regional to national level, and even larger at global scale. More importantly, up-scaled soil C estimates per biome vary considerably between different datasets (i.e. due to different soil depth and

vegetation map data). There is certainly scope for an improved global soil dataset using more data and geostatistical approaches. In practice, soil monitoring schemes tend to be based on soil inventories, designed to estimate regional SOC stock and not its change over time. Thus, when designing a new soil inventory scheme it is important to consider the possibility of repeating the sampling to enable estimation of SOC stocks changes.

Many soil inventories have been carried out in the past and it can be useful to use the data from these inventories when looking for changes in C stocks. However, it is very important that the protocols for these historical inventories are well recorded to ensure that estimates change are possible (e.g. when using different SOM/SOC methods and conversion factors). Saby *et al*. (2008) have investigated the ability of national soil inventories across Europe to detect changes in organic C if they were repeated. They estimated that, with a reasonably dense number of sites, changes in soil C should be able to be detected after ten years across the majority of countries in Europe.

Bellamy et al. (2005) determined the C stocks change in UK, in the period 1978-2003 at a depth of 15 cm using the data from the national soil inventory of England and Wales. This inventory was based on a sampling grid of 5 Km, for a total of 6000 points covering all possible land uses. At each sampling site 25 cores were collected in a systematic way and bulked. Forty per cent of the points were sampled again after between 12-25 years. This study showed a loss of 13 million t of C per years (about 0.6% of the UK soil C stocks), corresponding to about 9 % of UK's CO_2 emission reductions achieved between 1990 and 2002 (12.7 million tonnes of C per year). The greatest losses were observed in the organic, C–rich soils; moreover, no-relationship was found between the rate of loss of soil C and land use-type. This study gives a first indication of a real positive feedback effect of climate change on soil C stocks, as during the study period average air temperature in the UK increased by 0.5 °C (Shulze and Freibauer, 2005). However, no bulk density data were collected and so these data could not be expressed as C densities (i.e. ignoring peat shrinking and expanding effects on measured organic C contents per depth increment) and further SOM analysis differed between analyses.

In Belgium Lettens *et al*., (2005) investigated SOC changes in the whole of Belgium (30599 km^2), which was divided into 567 landscape units obtained by intersecting maps of land use and soil type. The SOC stock for 1960 was derived from data of C concentration, stone content, soil horizon depth and bulk density (using pedotransfer functions). Soil sampling campaigns were performed between 1950 and 1970; they were then repeated in the 1980s and again between 1997 and 2002. Soil C stock changes were derived from the differences between

sampling periods, e.g. 1960, 1990 and 2002. In 17% of the landscape units, a significant increment of the topsoil (30 cm) C stock was observed, whereas 16% showed a significant decrease, both were related to land use. Between 1960 and 2000 the largest and smallest increments were recorded in the mixed and broadleaves forests (+29 Mg C ha^{-1}) and grassland (+9 Mg C ha^{-1}), respectively. On the whole, the topsoil C stock of rural Belgium increased from 140 Mt C in 1960 to 157 Mt C in 2000. The average stock per unit surface increased from 58 Mg C ha^{-1} to 65 Mg C ha^{-1} (Kutsch et al., 2009).

Jones et al (2005) was to produce a continuous pan-European cover of quantitative OC content in the topsoil, taken as 0–30 cm depth. This study uses a novel approach combining a rule-based system with detailed thematic spatial data layers to arrive at a much-improved result over either method, using advanced methods for spatial data processing. The rule-based system is provided by the pedo-transfer rules, which were developed for use with the European Soil Database. The strong effects of vegetation and land use on SOC have been taken into account in the calculations, and the influence of temperature on organic C contents has been considered in the form of a heuristic function. Processing of all thematic data was performed on harmonized spatial data layers in raster format with a 1 km * 1 km grid spacing. This resolution is regarded as appropriate for planning effective soil protection measures at the European level. The results demonstrate that the methodology utilized represents a realistic alternative to approaches based on direct extrapolation of point observations, either by assigning measured data from a small number of points to polygons delineated on a soil map that represent much larger areas with no measured values, or by employing a spatial extrapolation procedure of values derived from point data. Even with the apparently large number of ground data points (> 12 000 values available to the study) some soils with limited spatial representation are hardly included in the sample data.

In the last twenty years increasing attention to the issue of soil C has led to the realization of several projects at international, European and national level. Generally these projects emphasise the common need of developing a robust and clear C accounting and verification systems that combine direct and indirect measurements techniques that are repeatable and can be performed quickly and cheaply.

Following are presented some examples of project:

RSTCB (The Role of Soil in Terrestrial C Balance, Werner Kutsch chair, Max-Planck-Institute for Biogeochemistry-Jena) is a European project developed over 5 years, from mid 2002 to mid 2007. The main objectives of programme were: to increase confidence in soil C flux and stock change estimates to generate datasets that are reliable and consistent; to develop a new generation of models describing soil C dynamics; to investigate the effects of perturbation on soil C balance and the potential for mitigation of C emissions.

For these tasks, the programme included scientists from out with the soil community, for example agricultural economists and land use planners, so as to generate two-way dialogues that inform the wider global change debate with soil-related scientific knowledge as well as adding relevance to scientific work by placing it in a broader context.

GCTE SOMNET (A Global Network and Database of Soil Organic Matter Models and Long-Term Experimental Datasets, Pete Falloon administrator, Soil Science Department, Rothamsted Research, Harpenden) is a global network of soil organic matter models and long-term experiments established during 1995 that includes several participants from south and north America, Australia, Africa Asia and Europe with a total of 120 experiments and 31 models C soil utilised. SOMNET has rapidly become internationally recognised as an important scientific initiative in the following ways: SOMNET has been adopted by the IGBP's GCTE programme as a Core Project of its focus on Soil Organic Matter; SOMNET has been invited to participate in the IPCC's Joint Working Group on Methodologies for Establishing National CO_2 Inventories; long- term datasets and models selected from SOMNET have been used to complete the most comprehensive evaluation of soil organic matter models undertaken to date, in a process begun at a NATO-funded Advanced Research Workshop held at Rothamsted. In this exercise, nine leading SOM models were compared for performance in simulating twelve datasets representing different land-uses (arable, grassland, forestry), climatic zones and management practices. Only four models were able to simulate all land-uses (RothC, NCSOIL, CENTURY and SOMM) and a group of six models (RothC, CENTURY, DAISY, CANDY, NCSOIL and DNDC) performed significantly better than did three others. In addition to the model comparison exercise we have also used European datasets from SOMNET (see Smith *et al*., 1996c) to estimate the potential for C sequestration in agricultural soils in the UK the European Union and the wider Europe. These studies shown that agronomically realistic scenarios could sequester up to 10% of the anthropogenic CO_2 produced in Europe each year or

up to 2% of that produced globally. The data were used to estimate the potential consequences of land-use change following the British BSE crisis.

CARBOEUROPE (Assessment of the European Terrestrial C Balance, 2004-2008, Ernst-Detlef Schulze coordinator) has the aim to enhance the understanding and the methodologies for the observation, quantification and prediction of the terrestrial C Cycle of Europe in order to fulfill the political need for accurate estimates of regional C sinks and sources. CarboEurope provide critical knowledge to adequately trace the C fluxes and identify the ecosystem compartments in which C is ultimately immobilized. CarboEurope attempts to have this observation system in place 5 years before the First Commitment Period under the Kyoto Protocol (2008-2012), and to operate it for an equivalent time period as the Commitment Period. It thus may act as a test run for the feasibility of detecting the biospheric C sources and sinks and changes in fossil fuel emissions over this politically important time span of 5 years.

The content of C sequestration in soils needs to be known at different scales of resolution: field, regional, national, and global. Estimations of soil C stocks have been made at national and global levels (Post et al., 2001; Batjes, 1996), but while there are soil quality programs that include soil C monitoring, there is no internationally agreed-upon method to verify these changes. There is, therefore, an urgent need to develop robust, science-based, flexible, and practical protocols for monitoring and verifying temporal changes in soil C (Post *et al.,* 2001; Rodeghiero 2006).

4 Soil organic C models

Large uncertainties are involved in predicted soil C change estimates, comparisons of predictions between different countries, estimates prepared with different methodologies, and in comparisons to empirical data. Therefore, uncertainties of predicted soil C change estimates should be always assessed. Process-based modeling of soil C dynamics can be used to produce soil C change estimates for countries with different resources and levels of input data, since a wide range of models with differing input data requirements exists. Whenever possible, models and other methods should be developed in parallel (Peltoniemi *et al*., 2007).

SOM models can be used to assess the impact of global change on SOM and subsequent feedback effects. A model simulation of future events obviously cannot be compared to measured data to verify its validity. We can, however, get some measure of performance by testing a model's ability to simulate long-term SOM changes using existing datasets (Smith *et al*., 1997). The need for models of soil organic matter (SOM) turnover is similar to the need for many models of environmental processes; extrapolate or interpolate experimental results in time, space and to different environmental conditions; and to investigate scenarios and hypotheses that are beyond the realm of experimental works (Kutsch et al., 2009).

The aim of process models is to reproduce soil C dynamics in form of equations. These models in combination with GIS and remotely sensed data can be useful to predict changes in soil C stocks in long term. Soil organic matter models are now used, more than ever, to extrapolate the understanding of SOM dynamics both temporally (into the future) and spatially (to assess C fluxes from whole regions or continents). Another increasing application of SOM models is in agronomy; many SOM models are now being used to improve agronomic efficiency and environmental quality through incorporation into decision support systems (Post *et al*., 2001).

Among the models the most common and widely used models are RothC (Rothamsted C Model; Coleman and Jenkinson, 1996); CENTURY (Parton *et al*., 1987); DNDC (Li *et al*., 1994) and SOCRATES (Grace and Ladd, 1995). These models describe processes, such as transformation, protection and mineralization of SOM, at various levels of detail and with different dependencies on environmental conditions. Consequently, each model has its own power and limitations, and their results should always be validated with experimental data.

4.1. Modelling carbon accumulation in soil

There are several approaches to modelling SOM turnover including process-based multi-compartment models, models that consider each fresh addition of plant debris as a separate cohort that decays in a continuous way, and models that account for C and nitrogen transfers through various trophic levels in a soil food web.

The understanding and interpretation of SOM processes vary widely in current SOM models. Generally soil organic matter models can be classified as

1) single homogeneous compartment;
2) two compartment;
3) non compartmental decay
4) multi-compartmental.

Models in group 1 and 2 are principally but not exclusively static (where the current model status depends on the status in the previous time step, e.g. time varying environmental conditions), while the models in group 3 and 4 are mostly dynamic (where the current model status depends on the status in the previous time step, e.g. varying environmental conditions) the dynamics models can be further split into organism-oriented and process-oriented models (Kutsch et al., 2009).

The SOM model can provide tool for understanding spatial and temporal soil organic C dynamics and may simulate SOM as a whole, or SOC, nitrogen or other nutrients.

Most models are process based, that is they focus on the processes mediating the movements and transformations of matter or energy and usually assume first-order rate kinetics. Early models simulated the SOM as one homogeneous compartment but some years later Jenkinson (1977) proposed two-compartment models, and as computers became more accessible, multi-compartment models were developed. Of the 33 SOM models currently represented within the GCTE-SOMNET database 30 are multi-compartment, process based-models"(Smith *et a*l., 1997).

The decomposition of organic matter in soil is a complex process involving multiple biological interactions due to enzyme activities necessary for the decomposition itself. At the same time is simple to make a conceptual division of the different fractions of organic matter in the soil depends on the speed with which they are degraded. For example, the stubble degrade quickly once incorporated into the soil as humus, which gives the intrinsic fertility soil, it

degrades slowly. We can assume that other fractions characterized from different degradation rates are present in the ground and between them exchange of matter occur. The abundance and reactivity of each fraction determines its contribution to total mineralization rate; fractions large and relatively stable with low reactivity, leading to a slow mineralization while the incorporation of small amounts of substrates in the soil characterized by high degradability, begin of the mineralization peaks. The main concept of the models that simulate the mineralization of organic matter is that the process of C mineralization occurs in different fractions (pool), each characterize in terms of quality and quantity; the speed with which this process occurs depends on the their chemical composition which determines, for example, the degradability of the same organic matter. Thus generally all the simulation models represent the dynamics of C by the subdivision of organic matter into different pools.

A pool is defined as "a compartment that contains materials which are chemically indistinguishable and equally accessible to plants or soil microbial population" (Smith *et a*l., 1997). Thus the soil organic matter that the addition through crop residues or organic fertilizers is assigned to one or more pools each of which has specific properties (C / N ratio, speed decomposition, etc.) that allow the dynamic simulation decomposition. In the models each pool is represented by a state variable and corresponding rates of change are calculated according to the specific characteristics of the pool and the relationships between them. Moreover each pool can be represented in different ways: for example it is possible replicate a pool at various depths to take into account the effects of treatment or incorporation of organic materials in soil. Then, it is possible to realistically divide soil organic matter in different fractions each characterized by a level of specific and constant activity, in the model simulation assumes that every fraction affected by the process of decay according to a first-order kinetics. Each compartment or SOM pool within a model is characterized by its position in the model's structure and its decay rate. Decay rates are usually expressed by first-order kinetics with respect to the concentration (C) of the pool:

$$\frac{dC}{dt} = -kC$$

where *t* is the time. The rate constant k of first-order kinetics is related to the time required to reduce by half the concentration of the pool *when there is not input*. The pool's half-life *(h = (ln2)/k)*, or its turnover time (τ= 1/k) are sometimes used instead of k to characterize a

pool's dynamics: the lower the decay rate constant, the higher the half-life, the turnover time and the stability of the organic pool.

Solving this equation simulation models allow to estimate the C content in different pools for each instant of time (t). Resolution of the first equation we get:

$$\frac{dC}{C} = -k\ dt$$

$$\int \frac{dC}{C} = \int k\ dt$$

$$ln\ C = -k\ t + c$$

The final equation allows to calculate the C content for a given pool and at time *t*, while the first equation indicates a reduction C content of the passage of time since the rate mineralization is negative, showing a loss of C due to biochemical processes that occur in the soil. The use of one or two different pools to represent fractions of the substance organic soil is usually insufficient to estimate the turnover of organic matter in soil even if ultimately the general approach by who develops the simulation models is to create simplified models that can satisfactorily describe the changes in content soil organic matter in the long term (Jenkinson 1990; Kutsch et al., 2009).

Generally in the models there are three or four pools and at least one of these includes microbial biomass is characterized by high activity. This allocation of C and nitrogen in different pools can better represent the reality as homogenous materials are divided into different compartments each having specific characteristics.

The flows of C within most models represents a sequence of C going from plant and animal debris to the microbial biomass, then the soil organic pools of increasing stability. Some models also use feedback loops to account for catabolic and anabolic processes, and microbial successions. The output flow from an organic pool is usually split. It is directed to a microbial biomass pool, another organic pool and, under aerobic conditions, to CO_2. This split simulates the simultaneous anabolic and catabolic activities and growth of a microbial population feeding on one substrate. Two parameters are required to quantify the split flow. They are often defined by a microbial (utilization) efficiency and stabilization (humification) factor, which control the

flow of decayed C to the biomass and humus pools respectively. The sum of the efficiency and humification factors must be inferior to one to account for the released CO_2.

Another approach to modelling SOM turnover is to treat each fresh addition of plant debris into the soil as a cohort. Such model consider one SOM pool that decays with a feedback loop into itself. Q-SOIL, for example, is represented by a single rate equation. The SOM pool is divided into an infinite number of components, each characterised by its ”quality” with respect to degradability as well as impact on the physiology of the decomposers. The rate equation for the model Q-SOIL represented the dynamics of each SOM components of quality q and is quality dependent (Bosatta and Agren, 1996).

Exact solutions to the rate equations are obtained analytically (e.g. Bosatta and Agren, 1995). Another type of model simulated C and nitrogen transfers through a food web of soil organisms (Smith *et al*., 1997).

Some models have been developed in order to combine an explicit description of the soil biota with a process-based approaches. Food web models require a detailed knowledge of the biology of the system to be simulated (mostly unknown) and are usually parameterized for application at the specific sites (Kutsch et al., 2009).

4.2. A quick view on some models

There are several SOM model, below we will describe some of them:

- CANDY (C-Nitrogen-Dynamics) (Franko, 1996) is a modular system of simulation models and a data base system for model parameters, measurement values, initial values, weather data and soil management data. It simulates dynamics of soil N, temperature and water in order to provide information about N uptake by crops, leaching and water quality. CANDY uses a semi-cohort system to track litter decay, and calculates a biologically active time to allow comparisons among sites. The inert organic matter (IOM) component is calculated from the proportion of soil particles < 6~m (Franko 1996).
- CENTURY (Parton et al. 1989) was developed to simulate long-term (decades to centuries) SOM dynamics, plant growth and cycling of N, P and S. It was originally developed for grasslands but has since been extended to agricultural crops, forests and savannah systems. It uses a monthly time step with monthly average maximum and minimum temperatures and monthly precipitation data. It comprises two forms of litter: Metabolic and Structural, and three SOM compartments: Active, Slow, and Passive. C leaving the Active organic matter compartment is partitioned into either CO_2 or Slow forms with the split determined by soil texture. Soil texture also regulates the rate of transfer between Slow and Passive forms. CENTURY has been used to simulate C accumulation during soil formation, and changes in soil C storage following climate change scenarios.
- DAISY (Muller et al., 1996) simulates crop production, and dynamics of soil water and nitrogen under diverse agricultural management systems. It was developed as a field management tool as well as for regional administrative purposes, and has been applied to catchment areas, farmland areas and specific sites. DAISY contains a hydrological model with a soil water submodel, a soil nitrogen model with a soil organic matter submodel, and a crop model with a nitrogen uptake submodel. Rate constants are modified by soil clay content, a semi-cohort accounting system is used for litter decay and soil microbial biomass is a dependent variable although concepts of zymogenous and autochthonous microorganisms are included (Smith et al., 1997).
- DNDC (DeNitrification and DeComposition Li et al., 1994), couples denitrification and decomposition processes as influenced by the soil environment to predict emission of CO 2, N20 and N 2 from agricultural soils. DNDC contains four interacting submodels: soil climate, decomposition, denitrification, and plant growth. The plant growth

submodel includes subroutines for cropping practices such as fertilization, irrigation, tillage, crop rotation and manure addition to simulate SOM turnover in arable lands. Clay adsorption of humads allows some soil-specificity; decomposition is first order, such that biomass formed during decomposition is a dependent variable (Smith et al., 1997).

- The Hurley Pasture Model and the ITE (Institute of Terrestrial Ecology 1989-Edinburgh) Forest Model (Thornley and Verbene, 1989) share a common soil submodel, hereafter referred to as ITE. The pasture model aims to simulate N cycling in a grazed soil-plant system, and comprises three submodels. They are: (1) a grazing animal-intake model in which the C fluxes and N content of faeces and urine are sensitive to the N content of ingested plant material; (2) a vegetative grass growth model which responds to light, temperature and N; and (3) a soil organic matter submodel that responds to faeces, urine, and decaying plant residues. A physiologically based treatment is used for the plant component, but a functional compartment-based treatment is used for SOM. All decomposition rates are a function of the quantity of microbial biomass. Although some components are mineralized to CO_2 or NH_3 without passing through the biomass, the rate of such mineralization is a function of quantity of biomass. This model has also been combined with a transport-resistance model to describe C and N compartments and fluxes in a plantation forest soil system (Smith et al., 1997).
- NCSOIL (Molina et al., 1983) simulates N and C flow through soil microbes and organic components. It comprises four organic compartments: plant residues, microbial biomass (Pool I), humads (Pool II) and stable organic matter (Pool Ill). The original version did not include stable organic matter. Flows of C and N are interconnected, and increasing stability of organic matter results from metabolism and not from sorption mechanisms that would be sensitive to clay content. Microbial succession is simulated on residues, and although decomposition rate is independent of microbial biomass, microbial succession leads to more stable materials. In addition to ^{12}C and taN it simulates ^{14}C and tSN dynamics. NCSOIL has been incorporated into a deterministic model (NCSWAP) of the soil-plant system that simulates interactions of N and C dynamics with crop growth and soil water. The model has been simplified to avoid having to determine too many initial variables and parameters. To do so, microbial succession was collapsed into one microbial component with the dynamics determined by the rate of C flow through populations that consume microbes (Smith et al., 1997).
- ROTHC is the Rothamsted C model in which the turnover of C in aerobic soil is sensitive to soil type, temperature, moisture and plant cover (Jenkinson, 1990; Coleman and

Jenkinson, 1999). Nitrogen and C dynamics are not interconnected in ROTHC, the IOM component is quantified using C-dating, and starting values are obtained by running the model to steady-state. Unlike other models examined, ROTHC has been used for calculating organic matter inputs to soil and net primary productivity using soil organic matter and radioC measurements. It is formulated as a discrete sums-of-exponentials that can be transformed into a continuous form which can yield analytical results that may provide additional useful insights into soil organic matter turnover. It uses primarily monthly input data, and shares several other basic ideas with CENTURY (Coleman et al., 1997).

This is the model I chose to develop the work of phD thesis for simulate the C stocks in Sardinia soils, so is therefore best described more ahead.

- SOMM (Chertov and Komarov 1996) is described as the submodel of a single plant ecosystem model (SPECOM) developed for forested ecosystems. SOMM is a simulation model of soil organic matter mineralization, humification and nutrient release. The model takes into account a rate of the processes in dependence of litter fall nitrogen and ash content, soil temperature and moisture. The functioning of main complexes of soil destructors being reflected in the model. The model represents a system of linear differential equations with variable coefficients. The result of simulation shows SOMM applicability for wide range of environmental conditions from tundra to tropical rain forest. The models has supposed being used for modelling soil system and ecosystem dynamics on site, landscape and regional levels. Rates of processes are regulated by N and ash content of litter fall and it uses temperature and moisture as environmental variables. It treats the soil litter layers L, F, and H explicitly. C flow from L to H are governed by biological activity and progressive humification of the residual material. Activities of soil animals are implicit regulators of C flow; distinctions among humus forms such as mull in contrast to mor reflect proportions of earthworms in contrast to micro-arthropods etc. This is the only model of the group examined that simulates C accumulation in soil organic horizons explicitly.
- The Verberne/Van Veen model (Verbene et al., 1990) aims to simulate N and water balance in a grassland soil-plant system in order to predict yield, N uptake, N leaching, N mineralization and accumulation of soil organic N. The soil is treated as a multilayer mineral soil system, and is simulated using three submodels: (1) soil water submodel, (2) soil organic matter submodel; and (3) soil N submodel. Transformations follow first-order kinetics, with physical protection caused by soil clay resulting in protected and nonprotected biomass and organic matter. Specific decomposition rates are modified by soil moisture and temperature, but

microbial biomass does not influence decay rate. Clay protection reduces microbial biomass turnover by a factor of 100. Plant residues are partitioned into three compartments: 'Decomposable', 'Structural', and 'Resistant'; there is a 'Stabilized' organic matter component in addition to 'Protected' and 'Nonprotected' active organic matter. This model emphasizes the influence of clay on protection of microorganisms and soil organic components more than do other models (Smith 1997).

- SOCRATES (Soil Organic C Reserves And Transformations in agro-EcoSystems, Grace and Ladd, 1995) is a process based simulation model designed to estimate changes in topsoil SOC with a minimum dataset set of soil, climate and biological inputs. It uses a weekly time step, however, the minimum driving variables are annual precipitation (mm), mean annual temperature (°C), soil clay content (%) or CEC (cation exchange capacity mmol kg^{-1}), initial soil organic C (%) and bulk density (g cm^3). A value of 1.3 g cm3 is assumed for bulk density if that information is not available. Monthly climate values can be input if desired. The C model consists of five compartments which undergo first-order decomposition in response to temperature and moisture. All plant material can be divided into decomposable (DPM) and resistant (RPM) components based on the conceptual fractions initially described by Jenkinson (1990). The respective DPM/RPM ratios for the litter produced from a terrestrial ecosystems are the same as those used by Jenkinson (1990). The soil components consist of microbial biomass and humus, with the microbial fraction differentiated into a transient unprotected fraction (which is only involved in the initial stage of crop residue decomposition) and a protected fraction that is actively involved in the decomposition of native humus and subsequent microbial metabolites. The decomposition process in SOCRATES produces humus, microbial materials and C dioxide in proportions which are dependent on the CEC of the soil (Grace *et al*., 1995; Smith et al., 1997).

Table 2.: Main features of some models used in GCTESOMNET database

MODEL	Time step	Inputs Meteorology	Inputs Soil and Plant	Inputs-Management	Factors affecting decay rate costant	Outputs
CANDY (Franko 1996)	day	Precipitation, air temperature, irradiation	Soil description, depth of impermeable layer, soil water characteristics	Rotation, tillage practice, inorganic fertilizer applications, organic manure applications, residue management, irrigation, atmospheric N inputs	Temperature, water, nitrogen, clay	Soil carbon content, soil nitrogen content, soil water characteristic, soil temperature, gas (CO_2, NO_2, N_2)
CENTURY (Parton et al., 1996)	month	Precipitation, air temperature	Soil water characteristics, clay content, organic matter content, pH, soil N content, soil N content	Rotation, tillage practice, inorganic fertilizer applications, organic manure applications, residue management, irrigation, atmospheric N inputs	Temperature, water, nitrogen, clay	Soil C content, biomass carbon, ^{13}C ^{14}C dynamics, soil N content, soil water characteristic, soil temperature, gas
DAISY (Müller et al., 1996)	hour, day	Precipitation, air temperature, irradiation evaporation over bare soil	Soil layers, clay content, soil C content, soil N content, plant growth characteristics, plant species composition	Rotation, tillage practice, inorganic fertilizer applications, organic manure applications, residue management, irrigation, atmospheric N inputs	Temperature, water, nitrogen, clay	Soil carbon content, biomass carbon, soil nitrogen content, soil water characteristic, soil temperature, gas
DNDC (Li et al., 1994)	hour, day, month	Precipitation, air temperature	Soil layers, clay content, organic matter content, pH, soil bulk density	Rotation, tillage practice, inorg. fert. applic., organic manure applic., residue management, irrigation, atmospheric N inputs	Temperature, water, nitrogen, clay, tillage	Soil C content, biomass carbon, soil N content, soil water characteristic, soil temperature, gas

Table 2.: Continued

MODEL	Time step	Inputs Meteorology	Inputs Soil and Plant	Inputs-Management	Factors affecting decay rate costant	Outputs
HPM (Thornley and Verbene 1989)	day	Precipitation, air temperature, irradiation, wind speed	Soil water characteristics, clay content, plant species composition	Rotation, tillage practice, inorganic fertilizer applications, irrigation, atmospheric N inputs	Temperature, water, nitrogen,	Soil carbon content, biomass carbon, soil nitrogen content, soil water characteristic, gas
NCSOIL (Molina et al., 1998)	day	Soil temperature, air temperature	Soil water characteristics, organic matter content, soil C content, soil N	inorganic fertilizer applications, organic manure applications,residue management	Temperature, water, nitrogen, clay, pH, tillage	Soil carbon content, carbon, 13C dynamics, soil nitrogen content, 15 N dynamics,
RothC (Coleman et al, 1999)	month	Precipitation, air temperature, evaporation over water	Clay content, soil C content, soil inert C content	organic manure applications,residue management, irrigation	Temperature, water, clay, cover crop	Soil carbon content, biomassCarbon, gas,14C dynamics
Socrates (Grace and Ladd 1995)	week	Precipitation, air temperature	Cation exchange capacity, yield	Rotation, tillage practice, inorganic fertilizer applications residue management	Temperature, water, nitrogen, cover crop, cation exchange capacity	Soil carbon content, biomass,carbon gas
SOMM (Chertov and Komarov 1996)	day	Precipitation, soil temperature	organic matter content soil N content, ash content of litter, N content of litter	organic manure applications	Temperature, water, nitrogen,	Soil carbon content soil nitrogen content gas
Verbene (Verbene et al., 1990)	day	Precipitation, air temperature, irradiation, wind speed, evaporation over bare soil	Soil description, soil water characteristics, clay content, plant species composition, organic matter content, soil C content, soil N content	organic manure applications, atmospheric N inputs	Temperature, water, nitrogen, clay	Soil carbon content, biomass carbon, soil nitrogen content, soil water characteristic

4.3. Key environmental factors affecting SOM decomposition

The environmental factors affecting SOM turnover considered by models comprise temperature, water, pH, nitrogen, oxygen, clay content, cation exchange capacity, type of crop/plant cover and tillage. Below some of this factors will be describe;

Many studies show the effect of temperature on microbially mediated transformations in soil, either expressed as a reduction factor or Arrhenious equation, as shown below

$$k_2 = k_1 Q^{10} (T_2 - T_1)/10$$

In this equation, k_2 and k_1 are rate constants at two observed temperatures T_2 and T_1 . stated that this relationship lacks any theoretical justification and could be hard to apply to a system such as a population of soil organisms, where the total activity is determined by a whole range of different organisms with quite different responses to temperature. A further difficulty is that Q_{10} values also change with temperature and between systems (Kutsch et al., 2009). There is evidence to suggest that there is little change in the rate constant, but a change in the utilization of substrate pool and shift in function and composition of microbial communities. More recently Giardina and Ryan (2000) challenged the assumption that SOM decomposition is temperature dependent by showing that old SOM in forest soils from warmer climates than in soils from colder regions.

In all reviewed models, temperature affects a rate modifier that multiplies the decomposition rates of one or several compartments. Main differences among the models arise from use of air or soil temperature as the input data, exact formulation of the temperature models, and their time steps. DNDC uses daily minimum and maximum air temperature as inputs and implements an O'Neill response function to describe the effect of temperature on decomposition. This function yields an exponential increase in decomposition at lower temperatures ranges, but it has a distinct temperature optimum and a sharp decrease at higher temperatures. The effects of moisture and temperature on decomposition are combined by multiplication in many decomposition models response functions, including DNDC). Moreover, in the DNDC model, the advantage of using a multiplier for moisture and temperature effects on decomposition (i.e., continuity) is combined in a framework to evaluate the most limiting factor for microbial activity (Liebig´s law). This is achieved by adding reciprocal values of the respective response functions for moisture and temperature. Using this approach, the factor most limiting for decomposition (temperature or soil moisture) dominates the relationship.

In RothC (Coleman and Jenkinson 1999), mean monthly air temperature (ºC) is used. Temperature effects are represented as a multiplicative rate modifier on the decomposition of active compartments. The relationship is close to linear between temperatures 10 and 35ºC.

In CENTURY, soil temperature is modeled as a function of daily maximum and minimum air temperatures, along with the influence of canopy cover on the radiation budget. Temperature effects on decomposition vary across a range of temperatures according to empirically-derived relationships from decomposition studies. The influence of temperature on decomposition is combined with moisture into a single multiplicative factor (Peltoniemi et al., 2007).

Water have a major impact on the microbial physiology. The effect of soil moisture on decomposition in the models is closely related or similar to the effect of temperature on decomposition.

DNDC uses total daily rainfall as input to calculate soil moisture values. The modelled amount of rainfall reaching the soil surface is influenced by interception (depending on LAI and rainfall history). Furthermore, winter storage of precipitation as snow is considered. Newer versions also account for surface runoff during heavy rainfall events. Actual soil moisture is calculated using a cascade model with determination of evapotranspiration losses. The agricultural version of DNDC uses a linear function to describe the effect of soil moisture on decomposition, but the Forest-DNDC version uses a Weibull function, which is allowed to vary from 0–1. These values are combined with a temperature factor to modify decomposition.

In RothC total monthly rainfall (mm) and monthly open pan evaporation (mm) (or PET, mm) (Smith et al. 1996) are used to calculate topsoil moisture deficit (TSMD), as it is easier to obtain rainfall and pan evaporation data, from which the TSMD is calculated, than monthly measurements of the actual topsoil water deficit. Decomposition constants of active compartments are multiplied by a rate modifier that depends on cumulative monthly TSMD using a linear function within the typical range of TSMD. Outside the typical range, fixed minimum and maximum values are used.

In Century, soil moisture availability is simulated based on input data for rainfall, snow and irrigation, after adjusting for interception within the canopy, and taking into account water losses from simulated storm runoff, groundwater flow, and evapotranspiration.

Decomposition rates in Century are influenced by moisture availability according to a calculation based on the moisture left over from the previous monthly time step and current

rainfall divided by PET. This estimation is combined with the temperature influence on decomposition to form a single multiplicative factor (Peltoniemi et al., 2007).

While some models simulate O_2 concentration in soil explicitly, many define the extent of anaerobiosis on the basis of soil pore space filled with water.

Soil clay content and total SOM are correlated. Various schemes simulate the effect of clay on rate equations to obtain SOM accumulation. Soil texture has two main impacts on decomposition in the RothC, DNDC and CENTURY models. Texture influences decomposition both by affecting soil moisture through soil water holding capacity and by affecting stabilization of soil organic matter at higher clay contents.

Nitrogen is an essential element for microbial growth, which will be maximal when enough N is assimilated to maintain the microbial C:N ratio. The models require nitrogen input data such as atmospheric N deposition and N additions in fertilizers by contrast, nitrogen is ignored RothC.

DNDC decomposition of SOM leads to the formation of inorganic ammonium, which can be processed by microbial activity into various reactive forms (e.g., NO_3, NO_2, N_2O, NO) and also to N_2, volatilized as NH_3 (depended on soil pH), chemo-denitrified (NO_2) (depended on soil pH) or leached (depending on soil water flux, texture and plant N uptake).

In Century, C uptake is simulated through plant production, and this process is limited by nitrogen availability according to a maximum C:N ratio. Thus, nitrogen availability affects the amount of nitrogen taken up by the plant and its subsequent litter quality, as well as decomposition. In turn, nitrogen availability is affected by the rate of decomposition, the amount of nitrogen immobilized, and the external inputs of nitrogen to the model, including atmospheric deposition, biological nitrogen fixation and fertilization events. Nitrogen is omitted in RothC, but is included in the Rothamsted Nitrogen Model, which has been further developed into the SUNDIAL model and currently being extended for use in organic soils as the ECOSSE model (Peltoniemi et al., 2007).

4.4. Comparison of the SOM model performance

Model performance, besides availability of input data, can affect model selection. Evaluation data are used to test a model's performance, which along with an assessment of composition and sensitivity determines the adequacy of a model for reporting C stock changes; the performance of soil models is evaluated by comparing simulation results to field measurements of stocks or fluxes of C (Smith et al. 1997).

The performance of model shows how well a model can be expected to simulate in a given situation. It can provide tool to improve the understanding of the system, provide confidence in the model's ability to predict future changes in SOM. Model can be evaluated at individual process level, at the level of a sub-set of processes, or the model's overall outputs (e.g. changes in total SOM over time) can be tested against measured laboratory and field data. Models can also be evaluated for their applicability in different situations, e.g. for scaling up simulate net C storage from a site-specific to a regional level (Kutsch et al., 2009).

At the most basic level, comparing the performance of SOM models involves comparing predicted changes in SOM from a number of models. If more complex model packages including plant growth and soil climate modules are being compared then comparison of more variables and model sensitivity to them may be necessary to identify the reasons for differences between models. However, the differences between the central SOM decomposition modules in compartmental SOM models are generally small and may give similar results when driven with equivalent input data. Smith *et al*. (1997) completed the most comprehensive evaluation of SOM models to date. Nine models where tested against twelve datasets from seven long-term experiments representing arable rotations, managed and unmanaged grassland, forest plantations and natural woodland regeneration. The results showed that six models had significantly lower overall errors (RMSE) than another group of three models. The poorer performance of three models was related to failures in other parts of the ecosystem models, thus providing erroneous inputs into the SOM module (Smith et al., 1997).

The difficulty in accurately measuring SOC presents both problems and opportunities for SOM modelers. Challenges in measuring SOC include: obtaining representative undisturbed soil cores, obtaining samples for different layer depths accurately, using adequate replicates, conversion of SOC concentration to mass through accurate bulk density measurements, the high random spatial variation in SOC and changing methodology –thus unbiased and high-precision time-series measurements of SOC are rare. On the other hand, eddy covariance flux tower

measurements of total net ecosystem exchange flux tower measurements are often unreplicated and make measurements at time scales incompatible with most SOM models and dependent on many assumptions and correction factors.

Detection of relatively small differences in SOC compared to a large background value may also present problems. In reality, this means that there is rarely enough adequate data for model calibration and validation and the absence of replicated SOC measurements for many experiments makes it difficult to critically assess model fit to measurements, although this also implies that models may be of equal (or even greater) use than measurements in assessing SOC changes particularly over large areas (Kutsch et al., 2009).

There may be many weaknesses and limitations of SOM models, since most were parameterized under particular management or climatic regions. Ideally SOM model should account for all major SOM controlling factors, such as parent material, time, climate, litter quality (decomposed) biota management. These factors may have complex interactions, and separate analysis of controls could limit predictions of their effects on SOM.

Many models have been evaluated under different climatic and management conditions, but rarely compared using common datasets. Few models simulate aggregation processes, which may be important in the stabilization of plant residues, microbial biomass and humic substances. Differences in drainage are often not accounted for in SOM models. However, SOM accumulation in grassland and forest soils may be partly attributable to reduced drainage. Soil C matter models also need better integration with landscape processes, and better description of plant root development, layering, dissolved organic C and deep soil processes, which are an important part of the SOM system. In terms of soils, RothC and CENTURY may be limited by failing to account for pH effects on soil C turnover. This may be important in soil with low litter quality and fungal decomposition and thus pronounced litter layer development, or grassland with constant mineral fertilizer applications (Smith *et al*., 1997).

Soil organic matter models generally predicted faster C turnover than observed in very acid soils, since decomposition is up two-thirds slower especially in early stages, under acid conditions, although overall model errors may be small. Model improvement needs information on both the short-and long-term effects of pH on SOM, in particular the effects on the microbial biomass. Few models are able to predict SOC change in variable charge and allophanic soils. Some authors have used “effective” clay contents, based on % clay, % ferrihydrite and % allophone to get model agreement with measured data; others have suggested using surface area measurements or accounting for mineralogy. Most models are also unable to simulate SOC

changes in sub-soils, permanently waterlogged and consequent CH_2 fluxes, very dry, highly organic and recent volcanic soils. However, recently ECOSSE has been developed from RothC and SUNDIAL for organic soils and RothC has been extended to sub-soils. Several authors have suggested that current SOM models mat be limited in their applicability to tropical systems, and few models have been tested under arid conditions (Smith *et al*., 1997).

Possible reasons could include differences in soil fauna, the much faster turnover time of slow and passive SOM pools in the tropics, different temperature and moisture relationships with metabolic rates, differences in mineralogy and solution chemistry in tropical soils. Inadequate description of nutrients (N,P and K) and inability to account for aluminum (Al) toxicity may also limit SOM model predictions in tropical soils. Finally, the compartments of SOM models (i.e. C pools) are usually theoretical without measurable counterparts making it difficult to initialize the models and validate model-calculated results for the individual pools (Kutsch et al., 2009).

5. The RothC model

The Rothamsted C model (RothC, Coleman and Jenkinson, 1996) is a one of a very few models currently used world-wide to study global C dynamics and to report in national inventories of C stocks for the United Nations Framework Convention on Climate Change. The dynamic model has been extensively tested using long term SOC data from a wide range of soil types, land uses and environments and also it needs few inputs, which are easily obtainable. It is solely concerned with soil processes: it does not contain a sub model for plant production, nor it does attempt to compute annual returns of plant C to the soil from above-ground yields. RothC should not be applied to soils that are permanently waterlogged, and it is designed to run in two mode; if an input of organic C is known, for example, that ploughed in as straw, the model can be run in *forward mode*, to calculate how this input will decay in a particular soil under a particular climate.

If this input is continued for many years, the model will calculate the resulting change in soil organic C. However, a large part of the annual return of organic matter to soil comes from roots decaying throughout the year, from inputs of above ground detritus, again received at various times throughout the year, from root exudates, from free-living autotrophic microorganisms, etc., and this part is never known with any accuracy. RothC, runs also in *reverse mode*, calculating what the annual input must have been to give the measured amount of organic C in a particular soil, at a particular time (Coleman et al., 1997).

RothC is a model that allows for the effects of soil type, temperature, moisture content and plant cover on the turnover process. It uses a monthly time step to calculate total organic C (t ha^{-1}), microbial biomass C (t ha^{-1}) and $\Delta^{14}C$ (from which the equivalent radioC age of the soil can be calculated) on a years to centuries timescale. It was originally developed and parameterized to model the turnover of organic C in arable top soils from the Rothamsted Long Term Field Experiment, hence the name. Later, it was extended to model turnover in grassland and in woodland and to operate in different soils and under different climates.

It should be used cautiously on sub soils, soils developed on recent volcanic ash, soils from the tundra and taiga and not at all on soils that are permanently waterlogged.

The RothC model has been incorporated into a variety of application C accounting models such as CAMag, CAMfor and FullCAM as the soil C module by the Australian Greenhouse Office, and integrated with national scale data of soil properties, land use and climate to estimate soil C fluxes from mineral soils caused by changes in climate. RothC

computes the changes in organic C as it is partitioned into five basic compartments (*fig. 9*): the incoming plant residues is split into the decomposable plant material (DPM) and resistant plant material (RPM); these both decompose to form microbial biomass (BIO), humified organic matter (HUM) and evolved CO_2. The model also includes an inert pool of organic matter (IOM). With the exception of IOM, each compartment decomposes by first-order kinetics, and each has an intrinsic maximum decomposition rate. The actual rate of decomposition is determined using modifiers for soil moisture, temperature and plant cover, operating on the maximum rate. The clay content of the soil affects the apportioning of SOM between the evolved CO_2, the BIO and the HUM pools (Coleman & Jenkinson, 1999). RothC does not include a plant growth sub-module, so the plant C input rate to soil must be estimated either independently, provided by a plant production model, or by inverse modeling (iteratively changing C inputs to fit measured SOC values).

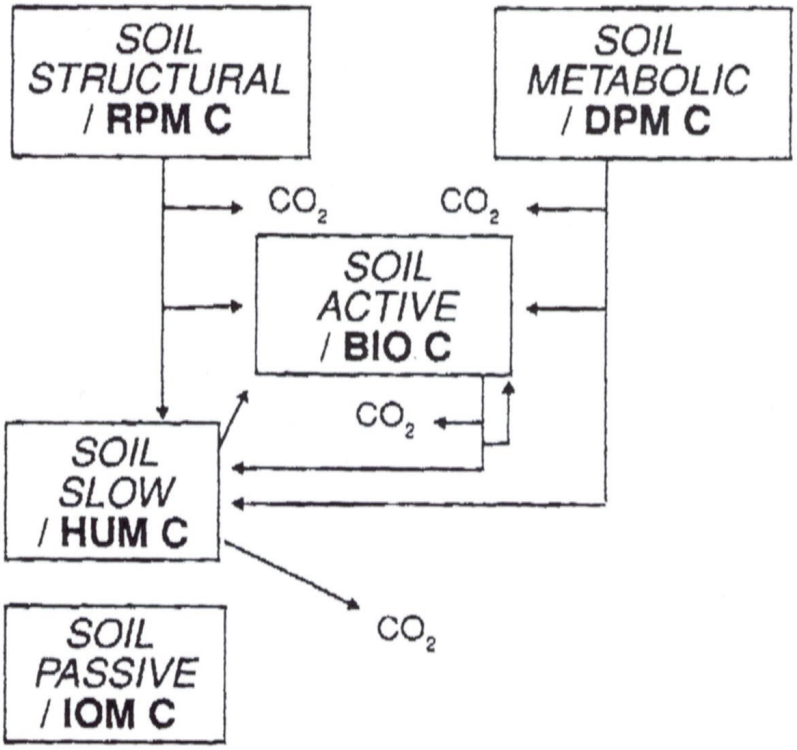

Fig. 9 Structure of the RothC model (from Kutsch, 2009).

Incoming plant C is split between DPM and RPM, depending on the DPM/RPM ratio of the particular incoming plant material. For most agricultural crops and improved grassland, we use a DPM/RPM ratio of 1.44, i.e. 59% of the plant material is DPM and 41% is RPM. For unimproved grassland and scrub (including Savanna) a ratio of 0.67 is used. For a deciduous or tropical woodland a DPM/RPM ratio of 0.25 is used, so 20% is DPM and 80% is RPM. All incoming plant material passes through these two compartments once, but only once. Both DPM and RPM decompose to form CO_2, BIO and HUM. The proportion that goes to CO_2 and to BIO + HUM is determined by the clay content of the soil. The BIO + HUM is then split into 46% BIO and 54% HUM. BIO and HUM both decompose to form more CO_2, BIO and HUM.

FYM (monthly input of farmyard manure) is assumed to be more decomposed than normal crop plant material. It is split in the following way : DPM 49%, RPM 49% and HUM 2%. In each active compartment is a process of decomposition, governed by a specific decomposition rate for each compartment and by other rates modifying factor for temperature, moisture and soil cover.

5.1. Model process

RothC model was chosen and applied to estimate actual and future C pools in Sardinia soil. The model splits the C in four active compartments decomposable plant material (DPM), resistant plant material(RPM); these both decompose to form microbial biomass (BIO), humified organic matter (HUM) and evolved CO_2.

Each active compartment (expressed as $Y\ t\ C\ ha^{-1}$) declines to $Y\ e^{-abckt}\ t\ C\ ha^{-1}$ at the end of the month. In this equation

- *a* is the rate modifying factor for temperature
- *b* is the rate modifying factor for moisture
- *c* is the soil cover rate modifying factor
- *k* is the decomposition rate constant for that compartment
- *t* is 1 / 12, since *k* is based on a yearly decomposition rate.

So $Y\ (\ 1\ -\ e^{-abckt}\ )$ is the amount of the material in a compartment that decomposes in a particular month. The decomposition rate constants (*k*), in years^{-1}, are set as for each compartment DPM : 10.0 RPM : 0.3 BIO : 0.66 HUM : 0.02

A fifth compartment assumed in the model is the inert organic matter (IOM). The IOM C fraction has a turnover time of 50.000 years, plays no active part in soil / atmosphere CO_2 exchange, and is unlikely to have any plant-derived ^{14} C. The plant derived ^{14} C content of this fraction is assumed in this study to be zero. In addition a regional scale the IOM is a small C fraction of total organic C (TOC) ($IOM=0.049TOC^{1.139}$).

The data required to run the model are:

- Monthly rainfall (mm).
- Monthly potential evapotranspiration (mm
- Average monthly mean air temperature (^{o}C). Air temperature is used rather than soil temperature because it is more easily obtainable for most sites
- Clay content of the soil (as a percentage). Clay content is used to calculate how much plant available water the topsoil can hold; it also affects the way organic matter decomposes.
- An estimate of the decomposability of the incoming plant material the DPM/RPM ratio.
- Soil cover. Is the soil bare or vegetated in a particular month. It is necessary to indicate whether or not the soil is vegetated because decomposition has been found to be faster in fallow soil than in cropped soil, even when the cropped soil is not allowed to dry out.
- Monthly input of plant residues (t C ha^{-1}). The plant residue input is the amount of C that is put into the soil per month (t C ha^{-1}), including C released from roots during crop growth. As this input is rarely known, the model is most often run in 'inverse' mode, generating input from known soil, site and weather data.
- Monthly input of farmyard manure (FYM) (t C ha^{-1}), if any. The amount of FYM (t C ha^{-1}) put on the soil, if any, is inputted separately, because FYM is treated slightly differently from inputs of fresh plant residues.
- Depth of soil layer sampled (cm) (Coleman and Jenkinson 1999)

5.2. Calculation of the rate modifying factors

The rate modifying factor (a) for temperature is given by:

$$a = 47.9 / 1 + e^{(106/ T+18.3)}$$

where T is the average monthly air temperature (^{o}C).

The topsoil moisture deficit (TSMD) rate modifying factor (b) is calculated in the following way: The maximum TSMD for the 0-23 cm layer of a particular soil is first calculated from

$$\text{Maximum TSMD} = -(20.0 + 1.3\ (\%\text{clay}) - 0.01\ (\%\text{clay})^2)$$

So for Rothamsted (%clay = 23.4), the maximum TSMD = - 44.94. For a soil layer of different thickness, the maximum TSMD thus calculated is divided by 23 and multiplied by the actual thickness, in cm. Next, the accumulated TSMD for the specified layer of soil is calculated from the first month when 0.75*(open pan evaporation) exceeds rainfall until it reaches the max. TSMD, where it stays until the rainfall starts to exceed 0.75*(open pan evaporation) and the soil wets up again.

The maximum TSMD obtained above is that under actively growing vegetation : if the soil is bare during a particular month, this maximum is divided by 1.8 is give BareSMD, to allow for the reduced evaporation from a bare soil. When the soil is bare it is not allowed to dry out further than BareSMD, unless the accumulated TSMD is already less than BareSMD in which case it cannot dry out any further.

Finally, the rate modifying factor (*b*) used each month is calculated from:

If acc. TSMD < 0.444 max TSMD,

b=1.0

otherwise

*b = 0.2 + (1.0 -0.2) * (max. TSMD – acc. TSMD) / (max. TSMD – 0.444 max. TSMD)*

Note that the calculation in the above table starts from the 1st January, when the soil is assumed to be at field capacity. For situations where this is not so, the weather data input should be displaced by a whole number of months, so that the soil is at field capacity at the start of the model run.

Thus, in the Southern Hemisphere, the weather data file should start in July when the soil is wet, so that July will appear as January in the output.

The soil cover factor (c) slows decomposition if growing plants are present. In earlier version of the model this factor is called the 'retainment factor'

if soil is vegetated c=0.6

if soil is bare c=1.0

The model adjusts for soil texture by altering the partitioning between CO_2 evolved and (BIO+HUM) formed during decomposition, rather than by using a rate modifying factor, such as that used for temperature. The ratio CO_2 / (BIO + HUM) is calculated from the clay content of the soil using the following equation:

$x = 1.67 (1.85 + 1.60 \exp(-0.0786\%clay))$

where x / (x+1) is evolved as CO_2

and 1 / (x+1) is formed as BIO+HUM

The scaling factor 1.67 is used to set the CO_2 / (BIO+HUM) ratio in Rothamsted soils (23.4% clay) to 3.51: the same scaling factor is used for all soils (Coleman and Jenkinson 1999).

5.3. The Roth C applications

There are several paper about applications of RothC, these can be classified according to time scale (short or long term) and extension of the application (small or large scale):

Short term and small scale: There are relatively few application of RothC to investigate short term (decadal time scales or less) aspects of the soil C cycle, and the observed data available for such studies have largely been used in calibrating the models themselves. Since the original development of RothC applications have also tended to focus on longer term issues (annual to decadal time scales or greater). Datasets describing the decomposition of uniformly ^{14}C-labelled plant materials under field conditions have been widely used to parameterize RothC, covering time scales of one to ten years and a variety of litter types and environmental. These datasets confirm the ability of RothC to mimic short-term soil C dynamics.

Exercises such as these were used to fit model parameters such as DPM/RPM ratio for different vegetation types: Shirato and Yokozawa (2006) using various plant materials for identified two conceptual pools of plant litter, decomposable plant material (DPM) and resistant plant material (RPM), in the RothC, by comparing the default proportions of DPM and RPM in the RothC and proportions in plant material fractions as determined by two-step acid hydrolysis with H_2SO_4. In Japan the plant samples are collected from several species at six different sites (arable land, grassland, forest). C in the plant materials was divided into three pools by acid hydrolysis: Labile Pool I, Labile Pool II and Recalcitrant Pool (RP), the unhydrolyzed residue. The average proportion of Labile Pool I in crops and grasses was 59%, which was the same as the proportion of DPM defined in the RothC as the default value for crops and grasses. The remaining 41% (23% Labile Pool II +18% RP) was consequently the same as the RPM proportion defined in the RothC. These results indicate that DPM in the RothC can be identified as Labile Pool I from the acid hydrolysis analysis and RPM as Labile Pool II + RP.

Long term and small scale: The vast majority of model applications have focused on evaluating the model against measured datasets of changes in SOC over decadal to century time scales. Smith et al (1997) shows a comparison of RothC with other eight model evaluated using twelve datasets from seven long-term experiments. Datasets represented three different land-uses (grassland, arable cropping and woodland) and a range of climatic conditions within the temperate region. Different treatments (inorganic fertilizer, organic manures and different

rotations) at the same site allowed the effects of differing land management to be explored. Model simulations were evaluated against the measured data and the performance of the models was compared both qualitatively and quantitatively. The plots also demonstrate the different pool concepts of the model, the fast (BIO and active) pools are smallest and most responsive, while the intermediate pools reflect the behavior or total SOC most closely; RothC has a very small inert pool of C while Century uses a larger, very stable (but not inert) pool of C. Falloon and Smith (2002) proves a comparison of Century, RothC and RothC driven by Century C inputs for the Rothamsted Woburn Ley Arable Experiment. Three treatments were simulated on this sandy soil , an all arable rotation (arable roots), an arable rotation with one year in six under hay (arable hay) and an arable ley rotation with three years in six under grass (grazed-ley) treatments. Both models reproduced measured trends in SOC, which highlights their ability to simulate the effects of management on SOC in the long term.

Large scale: there are several number of regional SOM models studies. *Fig 10* shows results from the application of RothC and Century to investigate land management scenarios for C sequestration in a region of Central Hungary (Falloon *et al*., 1997). This approach, linking detailed spatial databases to dynamic SOM models, is a powerful method for studying regional –scale SOM dynamics, integrating detailed knowledge on soils, climate and land use with state-of-the-art SOM models, and allowing identification of "hot spots".

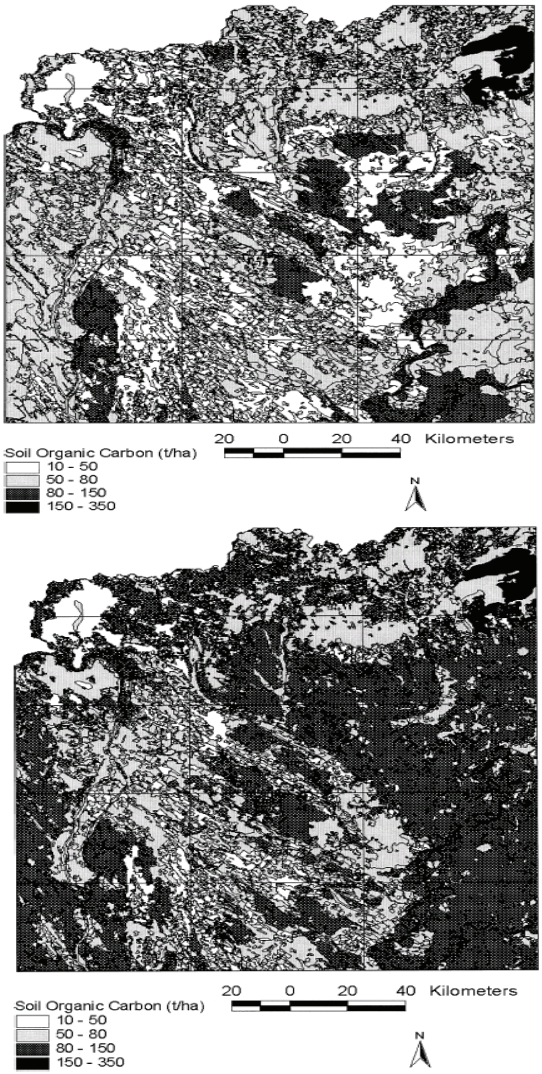

Fig. 10 Soil organic C before land use change scenario (1997) and after land use change scenario (2097) (from Fallon et al, 1997)

For C sequestration RothC has also been applied to 1 Km level databases of soil and land use, and coarser resolution databases of climate in the UK. Most of Great Britain was shown to lose soil C, which clearly has an impact on soil quality issues, as well as being a potential positive feedback mechanism to climate change via the enhanced release of CO_2.

By investigating C sequestration scenarios either alone or in combination with climate change scenarios, RothC also showed that C mitigation potential could be weakened in the future by climate change. It is also important to note that modeled impact "climate scenario" + "C sequestration scenario" does not equal the impact of "climate scenario" + "C sequestration

scenario": models predict interactions between management, soil and climate that are non linear and non additive.

Cerri et al (2007) predict soil C stocks and changes in the Brazilian Amazon during the period between 2000 and 2030, using the GEFSOC soil C modeling system. The current and future land use scenarios were integrated with spatially explicit soils data (SOTER database), climate, potential natural vegetation and land management units using the recently developed GEFSOC soil C modeling system. Results are presented in map for the entire Brazilian Amazon for the current situation (1990 and 2000) and the future (2015 and 2030). Results include soil organic C (SOC) stocks and SOC stock change rates estimated by three methods: the Century ecosystem model, the RothC model and finally the intergovernmental panel on climate change (IPCC) method for assessing soil C at regional scale. Values from Century and RothC (30,430 and 25,000 Tg for the 0–20 cm layer for the Brazilian Amazon region were higher than those obtained from the IPCC system (23,400 Tg in the 0–30 cm layer). The results can help to understand the major biogeochemical cycles that influence soil fertility and help devise management strategies that enhance the sustainability of these areas and thus slow further deforestation.

OBJECTIVE

As previous mentioned soil organic C plays a vital role in ecosystem function, determining soil fertility, water holding capacity and susceptibility to land degradation. In addition, SOC is related to atmospheric CO_2 levels with soils having the potential for C release or sequestration, depending on land use, land management and climate. Estimates of soil organic C stocks and changes under different climate scenario and land use systems can help determine vulnerability to land degradation.

Simulation models are widely used to assess the impacts of management and environmental variables on SOM dynamics, to address questions on ecosystem sustainability and C cycling under global change. The aim of process models is reproduce the soil C dynamics in the form of equations.

These models, in combination with Geographic Information Systems (GIS) and remotely sensed data, can be useful to predict changes in soil C stocks in long term; GIS linked modeling is a useful tool for large scale C cycle studies, allowing current estimates of regional C sequestration to be refined.

In this context fits my work and the main objectives of my research are to:

- Evaluate the Roth C model performance by comparing modeled and measured results.
- Using RothC model link to GIS-database to map existing stocks of C in Sardinia soils
- Assess future SOC change using GIS and RothC model under climate change scenarios.

The combination of GIS with SOM model can provide tool for understanding spatial and temporal soil organic C dynamics. By linking GIS that contain detailed information on soils, land use and climate to dynamic simulation models (RothC) for the turnover of organic C, it is possible to estimate actual and future C stock in Sardinia soil.

MATERIALS AND METHODS

1. The case study

The study area is represented by Sardinia Island (Italy), the second largest island in the Mediterranean Basin, located between 38° 51' 52" and 41° 15' 42 " latitude degrees and 8° 8' and 9° 50' longitude degrees (*fig 11*).

The whole extension (24.090 km^2) and the heterogeneity of the Sardinian territory led several difficulties in collecting and managing input data. It is well known that the soils of Sardinia have genesis, characteristics and properties very different, depending on the type of substrate, morphology, vegetation cover and land use.

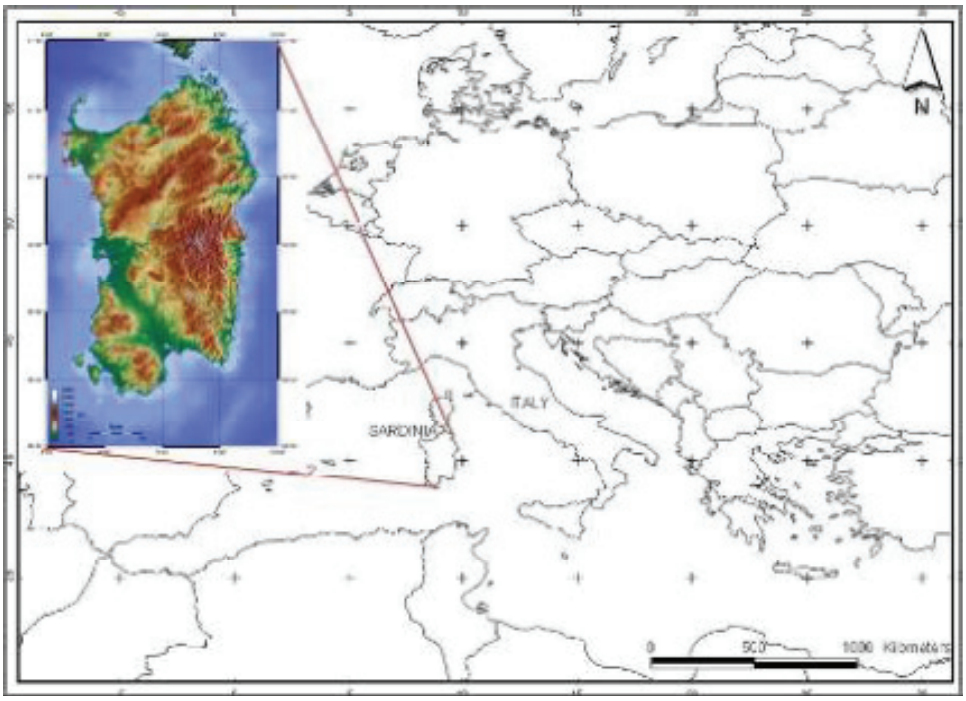

Figure 11 Geographical map of occidental Mediterranean basin and location of Sardinia.

The climate is Mediterranean, sub-arid with a warm summer, mild winter, and with a remarkable water deficit from May to September, and most annual rainfall (approximately 700 mm) occurs in fall and winter. Mean annual temperature is 15°C, mean minimum temperature is 7°C, and mean maximum temperature is 28°C. Annual mean thermal excursion is of 14°C (10°C in January and 24°C in August) (Chessa and Delitala, 1997).

Concerning the orography the Sardinian territory is composed by mountains (18.5%), hills (67.9%) and relatively plane areas (13.6%).

Almost 28% of the total area of the island has an association formed by rock outcrops and Leptosols (Eutric, Dystric e Lithic) (FAO-Unesco, 1988), spread to all territory and especially on hard rock (metamorphic, intrusive, effusive, dolomitic and limestone), mostly in steep, areas with irregular shape and free of tree cover and shrub. In these areas the main activity is extensive grazing, especially by sheeps and goats. Only 18% of the Sardinian soils are susceptible to irrigation: 3% are soils highly susceptible (Eutric, Calcaric e Mollic Fluvisols; Eutric, Calcaric e Vertic Cambisols; Eutric e Calcic Vertisols; Haplic e Calcic Luvisols), 6% are moderately susceptible soil (Vertic, Eutric Cambisols and Calcaric; Calcic and Haplic Luvisols; Calcaric Arenosols), and 9% are marginal susceptible soils (Haplic Nitisols; Chromic Luvisols, Eutric and Dystric Cambisols; Calcaric Regosols).

These soils are located in the interior plains and the coastal areas, mostly on the alluvial deposits of Pleistocene and Holocene, and support a predominantly intensive farming, irrigation or not. In the remaining 54% of the territory are mainly Cambisols, and Leptosols Regosols, in relation to different substrates, the morphological conditions and the degree and type of vegetation cover. These areas of the island are marginal for intensive agriculture, but have great importance for particular crops, for grazing and forestry activities.

The vegetation of trees and shrubs are an expression of environmental conditions and often indicates the climax vegetation of the area.

The climax of Sardinian is *Phytocoenoses*, mainly represented by *Quercion-ilicis* and *Oleo-Ceratonion* that occur with different aspects of ilex and thermophilous Mediterranean shrubland. However, there are in higher mountain areas aspects relating to the climax of the prostrate dwarf shrubs.

In the dynamics of the Sardinian vegetation, the paleogeographic and paleoclimatic events have had a very important effect on the island, influencing the composition and distribution of the original climaxes and contributing to determine the current vegetation cover. The main land use of the Island is agriculture (*fig.12*). It is interesting that the percentage of the areas suitable to the agriculture, including crops, sown field and orchards, accounts for 38% of the whole area, while similarly shrubs and herbaceous associations contribute to 34%. Forest cover 21% of island surface.

Climatic factors, with summer drought, rainfall concentrated in short periods for most of winter sunshine and strong winds, have a determining influence on the vegetation and its dynamics. All this highlights the heterogeneity of the island vegetation cover (Camarda and Valsecchi, 1992).

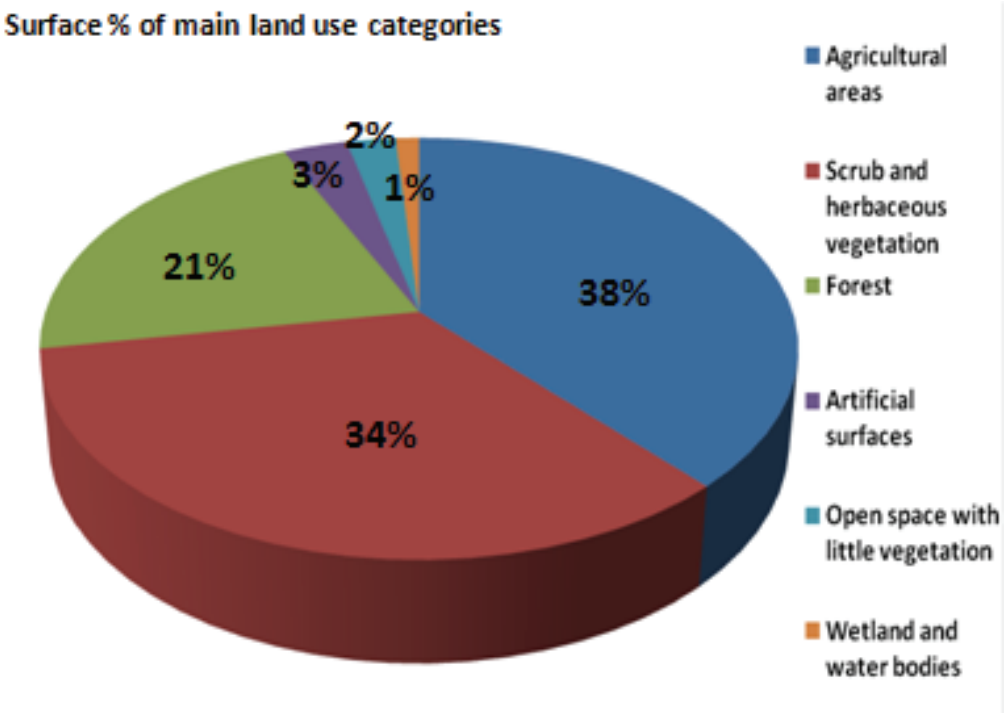

Figure 12. Surface % for main land use categories from Corine Land Use 2008

2. Methodology

The methodology developed in the present study is schematized in fig 13.

After data collection, we overlap land use map (Corine Land Use 2008), soil map (Regione autonoma della Sardegna1998) and actual climate layers (Dipartimento Specialistico Regionale Idrometeoclimatico-Arpas) in a GIS environment (ArcGis 9.3, ESRI Inc., Redlands, CA, USA), with the aim to obtain homogeneous areas (336888 polygons) by characteristics and input (fig 14). This first step allowed us to create the necessary first layer inputs to run the RothC model, in order to obtain the raw C soil map.

In order to automate and to speed up the process, the RothC model equations were written in a spreadsheet (Excel) and subsequently it was linked to GIS database.

The model was run with the rough input data for 1500 years to reach the equilibrium and obtain the current soil C stock with the current climate and land use.

These preliminary calculations showed how coupling a detailed GIS database with a dynamic simulation model can refine estimates of regional SOC stocks.

The raw C soil map allowed the model calibration by using the ground data; quantification of organic C in soil through modeling approach always requires a comparison with ground data in order to calibrate the model. For this reason, about 160 samples of soil were collected across the region (*fig. 15*).

Ground data were mostly collected from thesis or studies of Department of Soil Science, (University of Sassari) carried out in the northwest of the island. About the 37% of the ground data were collected from soils with nature grassland cover, about the 17% from soils with sclerophyllous vegetation cover and about the 16% by arable lands. 80 samples out of the 160 samples of soil analysis were used for model calibration, while the remaining 80 for validation.

Unfortunally, it was not possible to found soil sampling representing all the classes of Corine Land Use, but only for 8 of these. The 8 classes calibrated with ground data represented 79% of the island surface. The remaining land use classes were then calibrated accordingly to the 8 classes for which we found the data (*tab. 4*)[e. g Class 313 (Mixed woods of conifers and hardwoods) was calibrated based on the average results obtained from the classes 311(Broad-leaved forest) and 312 (Coniferous forest)].

Once calibrated, the model was run with correct inputs for the definitive actual C map and the data obtained were validated with another dataset of ground data. Finally, using future climatic scenario, (A1b scenario CMCC), projections about the Sardinia future C stock were obtained.

The system has been studies and developed with the aim to provide a flexible and powerful way to assess how different scenarios climatic change can affect C dynamics at the regional scale.

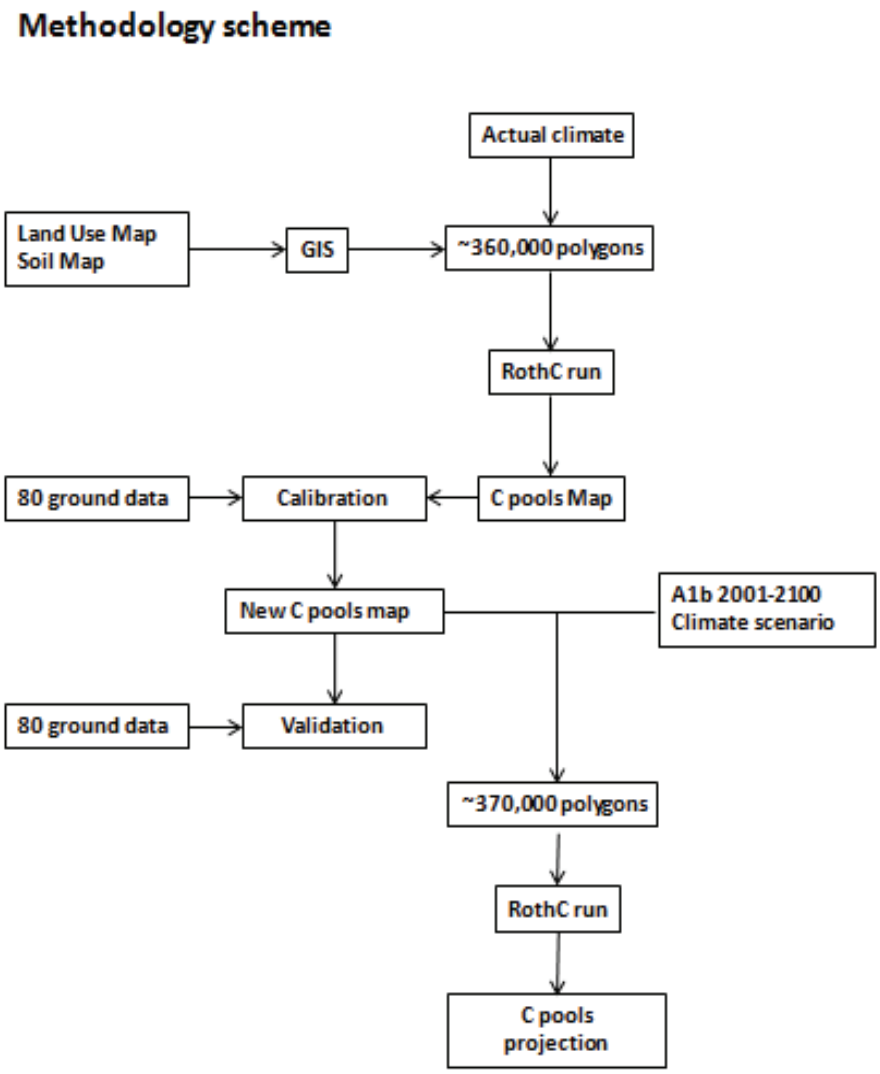

Figure 13. The methodology.

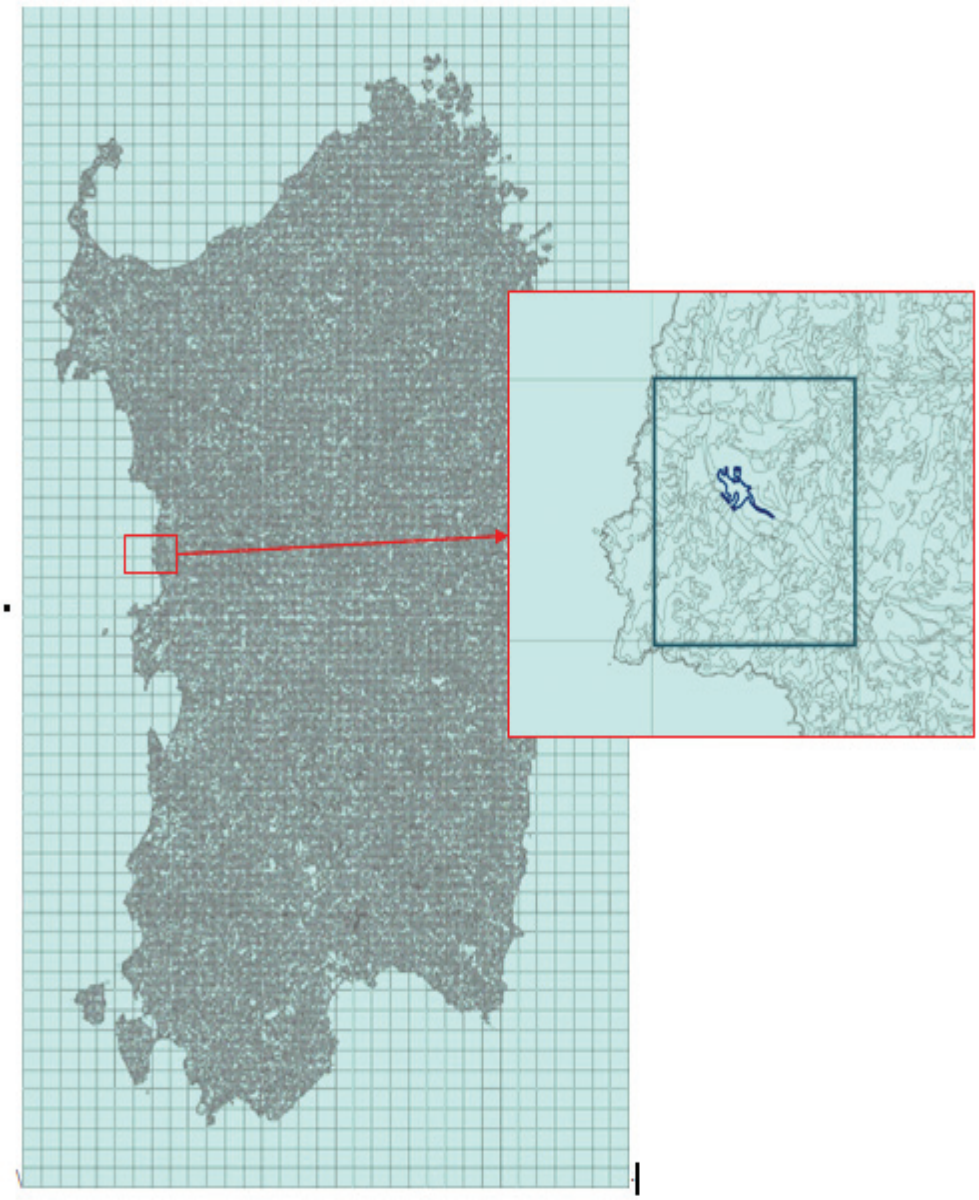

Figure 14. A particular of GIS elaboration: The overlap of land use + soil map with meteorological grid layer. In particular all polygons that are in one meteorological cell (e. g. blue square evidenced) have the same meteorological input. While each polygon (e.g. the blue polygon evidenced) is characterized by specific land use+ soil information.

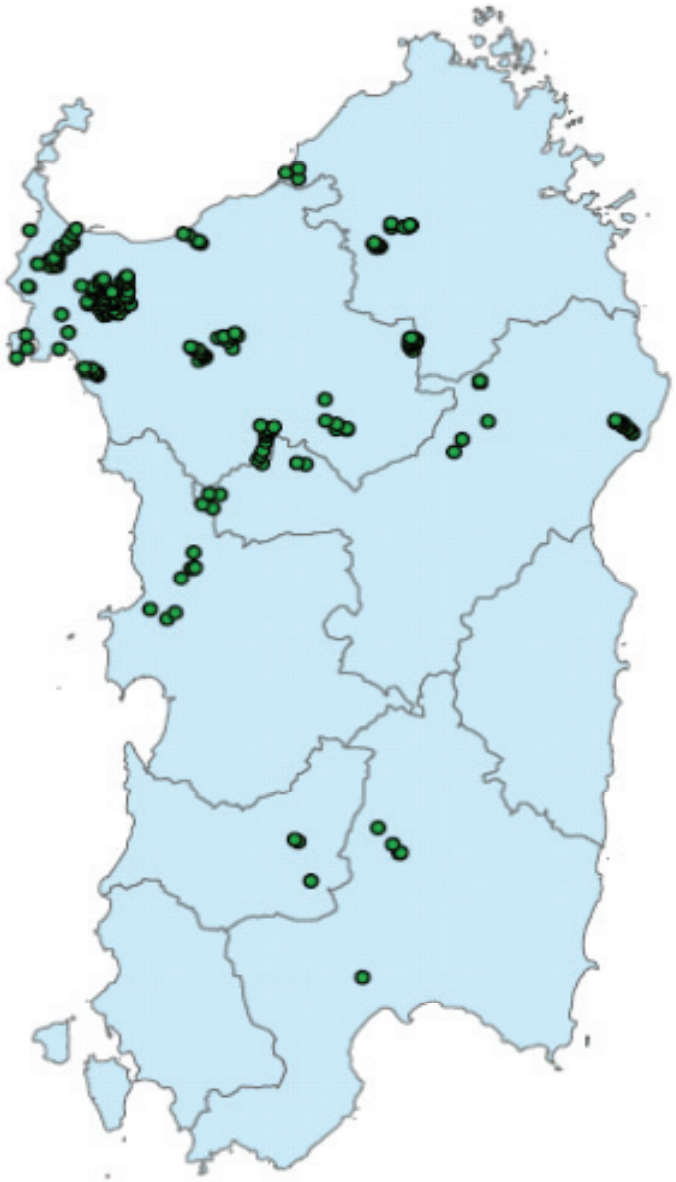

Fig. 15 Sampling areas (160 total ground data)

3.Input data

The RothC model requires few inputs, but the extension of the study area led to a significant difficulty in collecting and managing data (*tab. 3*).

Below we analyzed separately the various inputs sources:

Table 3. The data required to run RothC model (from Coleman and Jenkinson, 1999)

Data required to model run
• DPM/RPM ratio: an estimate of the decomposability of the incoming plant material
• Soil cover.
• Monthly input of plant residues (t C ha^{-1}).
• Monthly input of farmyard manure (FYM) (t C ha^{-1}), if any.
• Clay content of the soil (as a percentage).
• Monthly rainfall (mm).
• Monthly potential evapotranspiration (mm)
• Average monthly mean air temperature (^{o}C).
• Deph of soil layer sampled (cm)

Land Use Map: the model inputs (Decomposable Plant Material/Resistant Plant Material- DPM/RPM, plant residue and soil cover) were associated to Sardinia Land Use Map (2008 shapefile format ESRI). 24 land use classes were selected among the most representative (*fig. 16 and 17*), and the 70 original land use codes were rationalized into 24 codes (*tab. 4*). Each classes was associated a value of DPM/RPM, soil cover and plant residue input.

The Decomposable Plant Material/Resistant Plant Material (DPM/RPM) is an index of decomposability of the income plant material. Unfortunately, it is difficult to find data in literature that could fit Mediterranean ecosystem. So that , each land use class was associated with one of the three indices confirmed by various long term experiment at Rothamsted (0.25, 0.67, 1.44), these values have been used successfully in different ecosystems types (Coleman and Jenkinson, 1999).

The plant residues input is the amount of C that is put into the soil per month (t C ha^{-1}), including C released from roots during crop growth. An extensive literature research was focused mostly on works conducted in the Mediterranean Basin (Italy, Spain and Greece) was

made to order to collect all the existing data on monthly litter in Mediterranean Ecosystem (*tab.5*).

For that concerns land cover, we just associated *fallow* (0) or *covered* (1) values to each class for each month.

Monthly input of farmyard manure (FYM) (t C ha^{-1}) have been omitted because in Sardinia is little practiced fertilization, and regarding this data are conflicting.

Soil Map: the soil map (Regione Autonoma della Sardegna, 1998 shapefile format ESRI coordinate system Monte Mario, *fig 18*) provides several information about Sardinian soils (morphology, cover vegetation, taxonomy, profile description, texture, depth and limitations). The soil input data needed by RothC is the clay content expressed in percentage, and it is one of the most variable components in the soil. It can vary even within a few meters because many factors influence its amount. The Sardinia Soil Map allowed to detected an average rate for each texture class based on the USDA triangle (*tab 6 and fig.19*); more specifically to each homogeneous area was associated an average clay value.

The soil depth utilized for the run model is 23 cm.

Climate data input: the ARPAS (Dipartimento Specialistico Regionale Idrometeoclimatico) provided for the climatological data map of Sardinia region, on a number of years (1968-2000) having the following input: monthly rainfall (mm), monthly evapotranspiration (mm), average monthly mean air temperature (°C) (*fig 19*). Thirty-six files data, provided in grid format ESRI (resolution of 250 m coordinate system UTM ED50, *fig 21*) were used to have mean values of twelve months for each variable (*fig. 20*). The resolution, being too fine , was reduced to 5000 m.

Scenario climate data input: the CMCC (Centro Mediterraneo per i Cambiamenti Climatici) provided for the climatological Sardinia scenario data (A1b), on a number of years (2000-2100). 1200 files data, provided in grid format ESRI (resolution of 8 Km) having the following input: monthly rainfall (mm), monthly evapotranspiration (mm), average monthly mean air temperature (°C) for each years The trend shows a average temperature increase of about 2.5 °C expected for 2100, while the evapotranspiration expects an increase of approximately 50 mm and a decrease in rainfall of about 50 mm (*fig 22*).

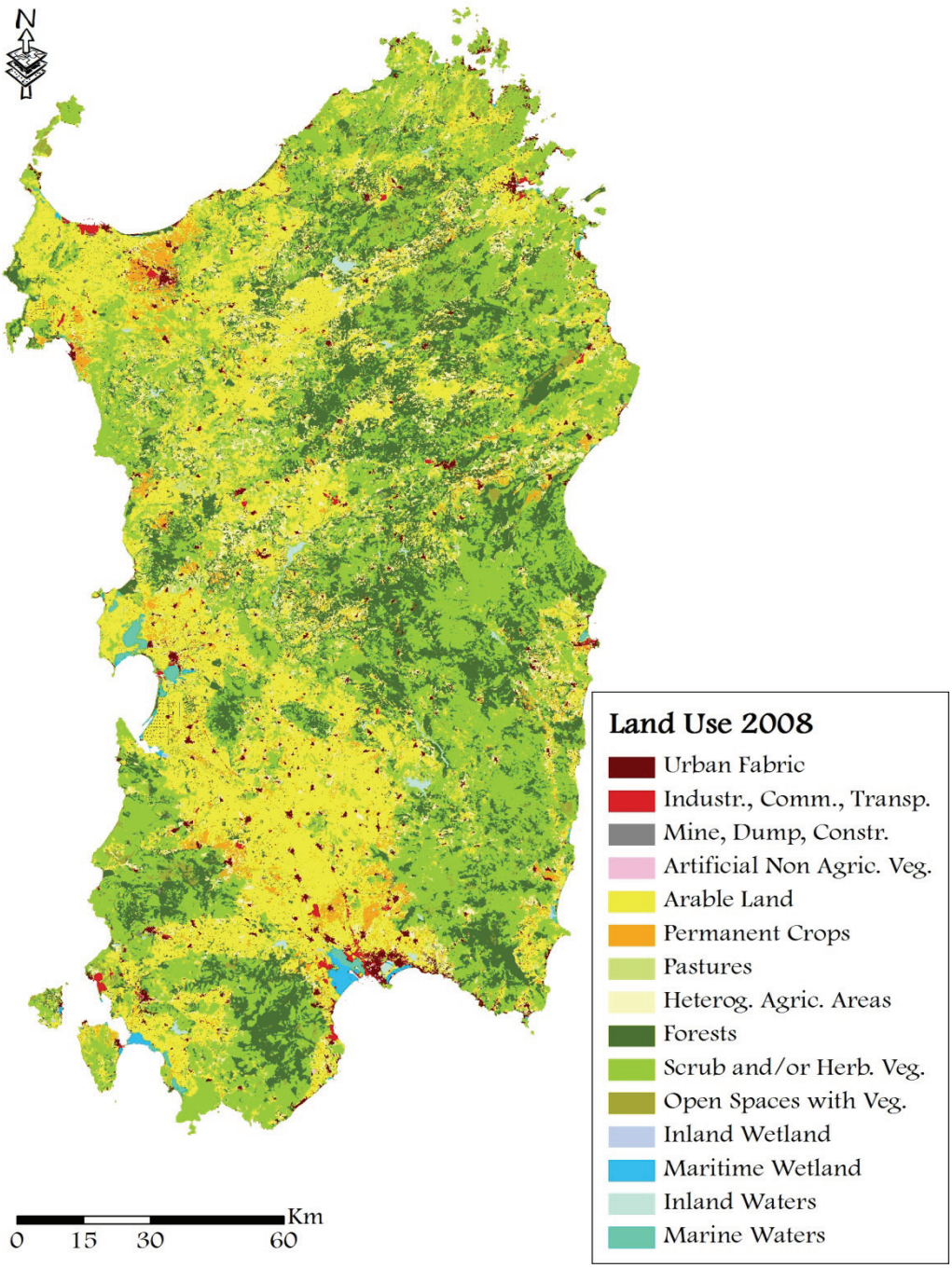

Figure 16 Corine Land Use 2008 with classes of 2 label

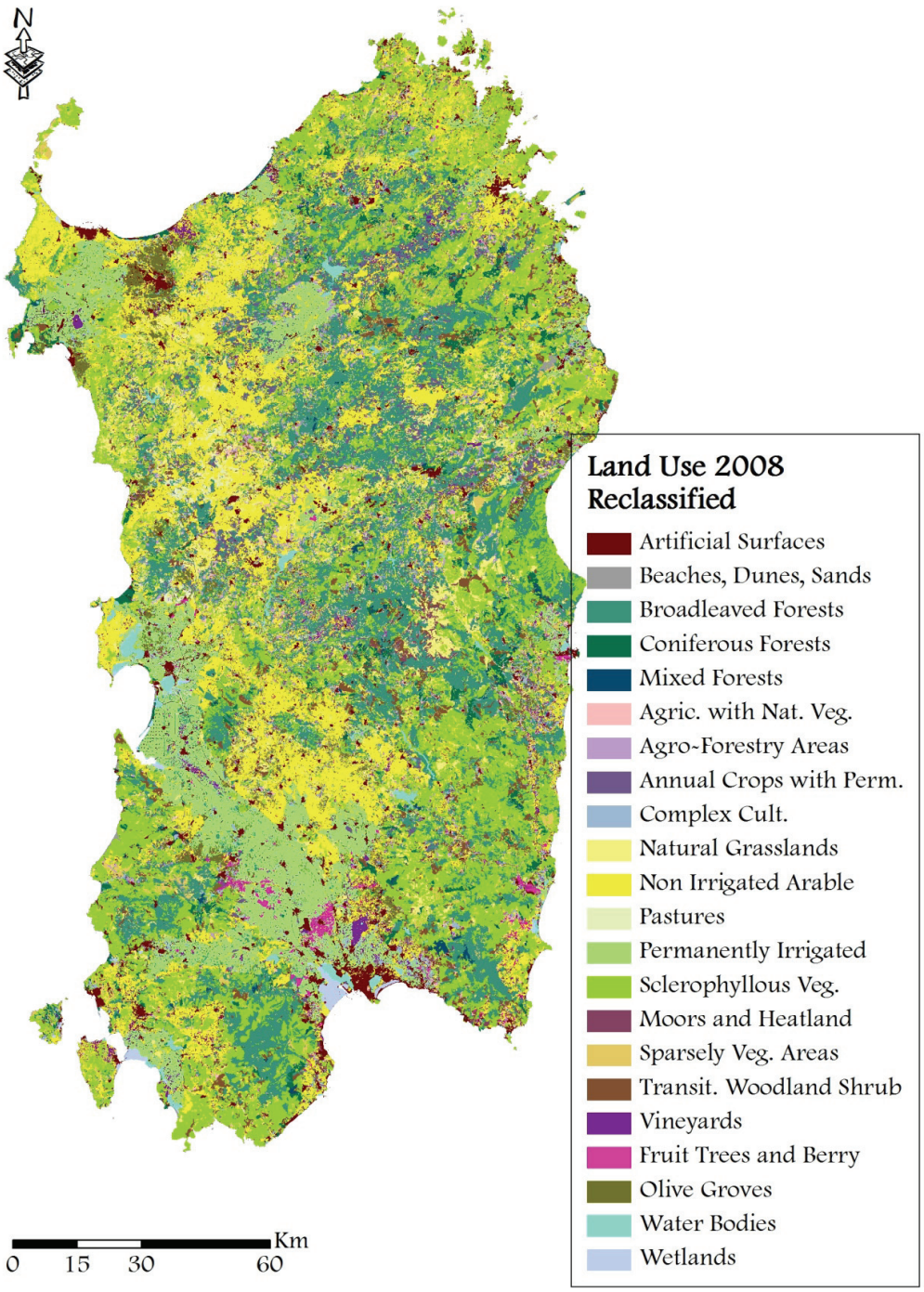

Figure 17 Corine land use with the 24 reclassified classes.

Table 4. The 24 classes selected by Land Use 2008, with surface value (% and ha) associated.

code	Land use categories reclassified	surface (%)	surface (ha)
1	Artificial surfaces	3.1	74343
141	Artificial surfaces	0.0	431
142	Artificial surfaces	0.1	3107
211	Permanently irrigated	17.3	415801
212	Irrigated arable lands	8.8	212525
221	Vineyards	1.0	24693
222	Fruit trees and berry plantation	0.5	11908
223	Olive groves	2.0	48779
231	Pastures	0.4	10316
241	Annual crops associated with permanent crops	2.9	70078
242	Complex cultivation	1.8	42214
243	Agriculture with natural vegatation	1.2	29287
244	Agro-forestry areas	2.4	57430
311	Broad-leaved forest	18.8	452356
312	Coniferous forests	1.6	38716
313	Mixed forest	0.5	12350
321	Natural grassland	6.0	144793
322	Moors and heathland	0.7	17608
323	Sclerophyllous vegetation	23.3	561021
324	Transitional woodland-shrub	3.8	92693
333	Sparsely vegetated areas	2.1	49572
3	Beaches, dune and sand	0.4	10187
4	Wetlands	0.4	9219
5	Water bodies	0.8	19182
		Total	2409000

Table 5 The data of plant residues (t ha-¹) collected in the bibliography.

Land use code	Author/ Source	Country	Typology vegetation	Plant residues (t/ha y s.s)
211-212	Di Blasi et al 1996	Italy	Soft Wheat	2.50
211-212			Hard Wheat	1.60
211-212			Barley	2.70
211-212			Oats	1.40
211-212			Corn Cobs	1.40
211-212			Sunflowers	4.00
211-212			Grain legumes	2.00
211-212			Soya	5.20
211-212			Rice	3.80
211-212	Enea 2005	Italy	Cereals	0.50
221	Di Blasi *et al* 1996	Italy	Vineyard	2.80
				2.90
223	Di Blasi *et al* 1997	Italy	Olive trees	1.60
				2.20
231	Hoepli Manuale di agricoltura	Italy	Trifolium subterraneum	4.40
231			Dactylis glomerata	5.25
231			Lolium perenne	4.35
231			Festuca arundinacea	5.25
231			Medicago sativa	4.80
231	F. Chiarini,*Veneto Agricoltura*	Italy	Trifolium incarnatum	5.00
231	Bullita *et al* 1989	Italy	Trifolium nigrescens	2.50
311	Rapp *et al* 1999	Spain	Quercus ilex	3.50
311			Quercus pyrenaica	3.14
311			Quercus lanuginosa	3.10
311	Zimmermann *et al* 2001	Southern Switzerland	Castanea sativa	7.60
311	Caritat *et al* 1996	Spain	Quercus suber	4.25
311	Leonardi *et al 1995*	Italy	Castanea sativa	4.52
311	Kavvadias *et al* 2000	Greece	Fagus sylvatyca	4.00
311	Peressotti *et al* 2006	Italy	Mediterranean forest	3.60
312	Kavvadias *et al* 2000	Greece	Pinus pinaster	1.40
312				4.00
312	Santa Regina *et al* 2001	Spain	Pinus sylvestris	5.80
312	Roig *et al* 2004	Spain	Pinus pinaster	1.52
312			Pinus pinaster	5.70
321	Hoepli Manuale di agricoltura	Italy	Pasture	0.10
321	Bullita 1976	italy	Pasture	1.80
321				2.30
323	Fioretto *et al* 2003	Italy	Mediterranean shrubland	5.80
323	Nunez-Olivera *et al* 1993	Spain	Mediterranean shrubland	2.90
323	Arianoutsou *et al* 1989	Greece	Mediterranean shrubland	1.40

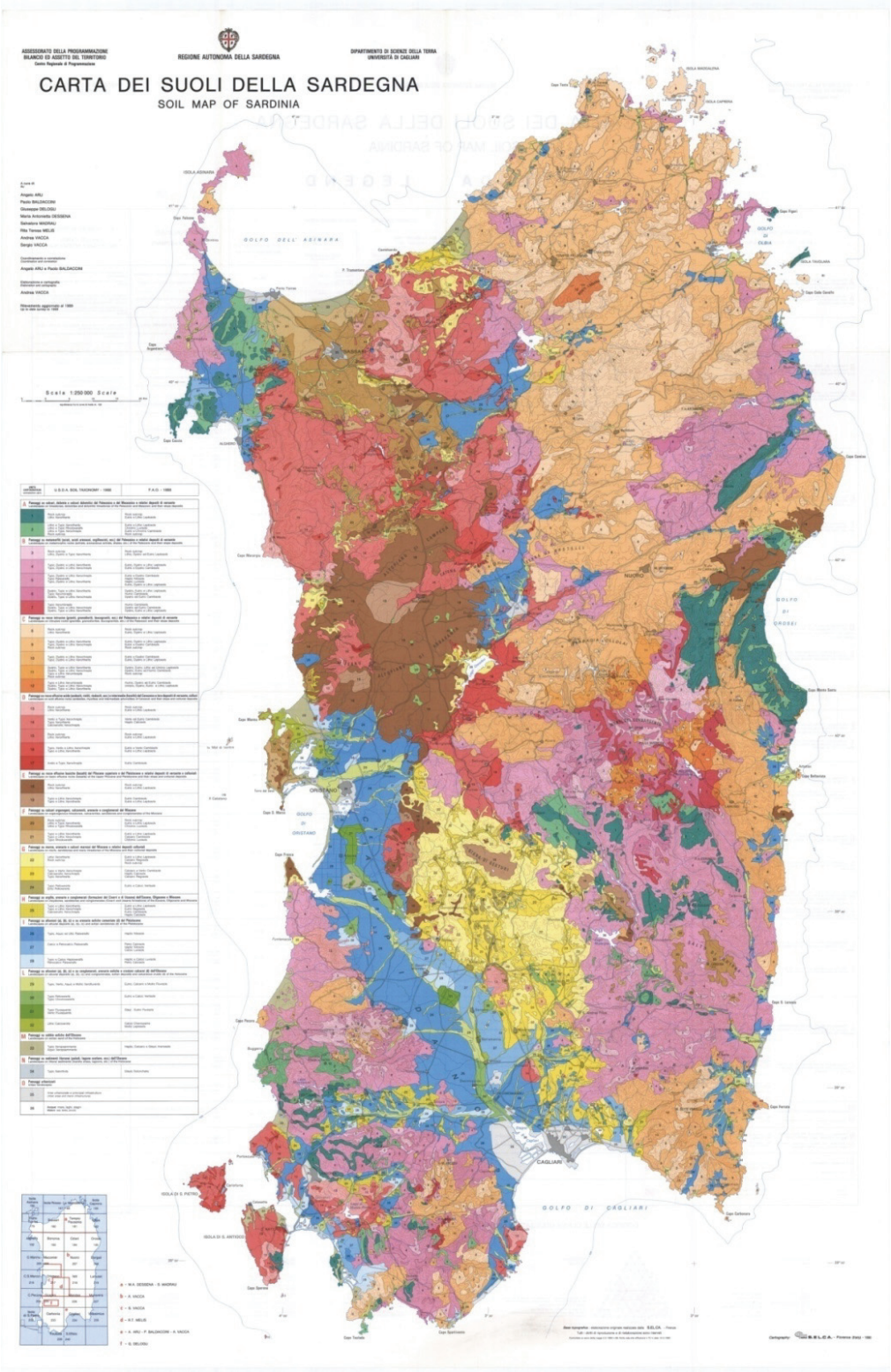

Figure 18 The Sardinia Soil Map 1998 (from Regione Autonoma della Sardegna)

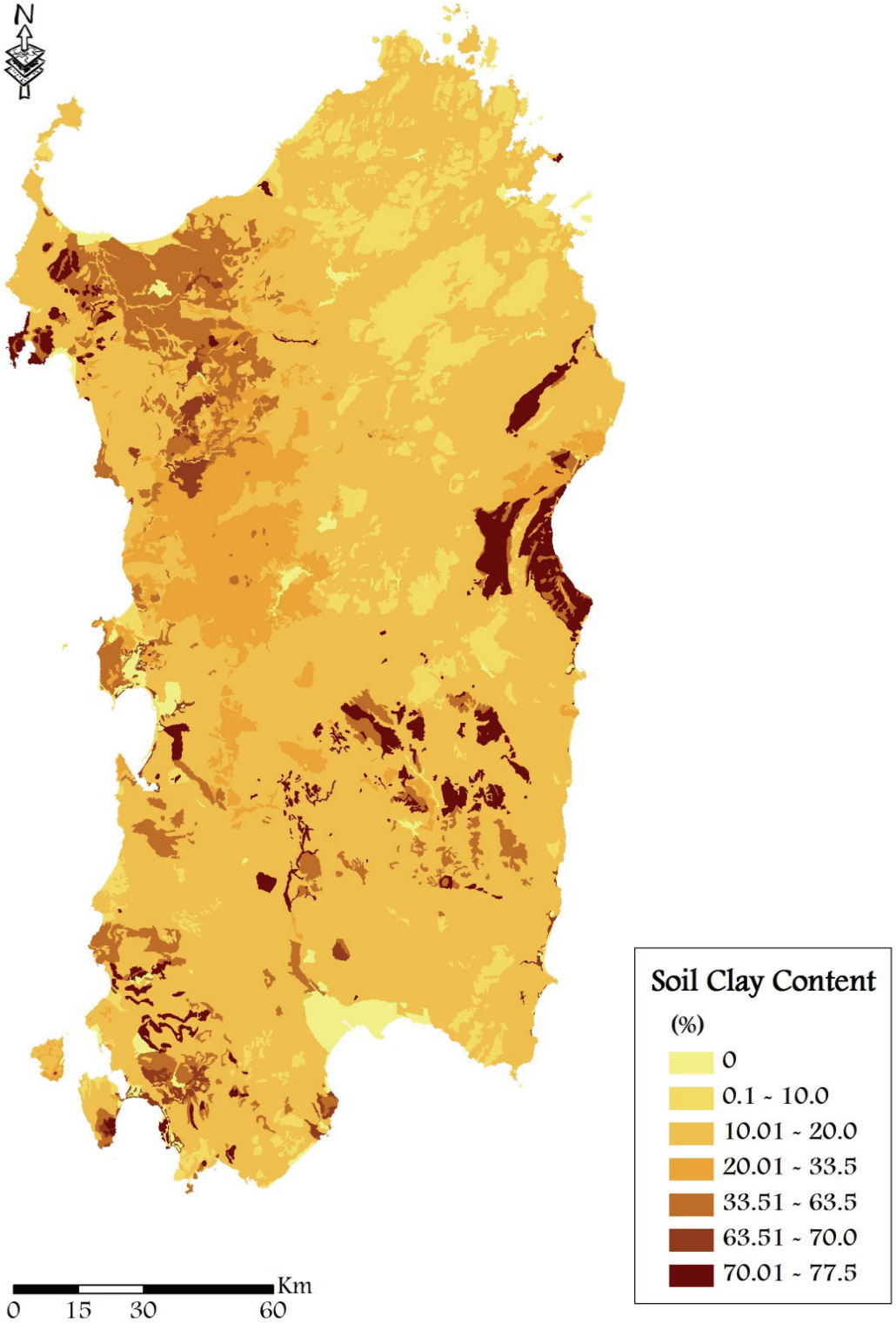

Figure 19 The map of clay content % associated at each texture class.

Table 6: The texture classes of Soil Map 1998, for each classes there is a value % clay content associated.

Soil Texture Classes	Clay %
from sandy to sandy loam	7.5
from sandy loam to loam sandy	10.0
from loam sandy to sandy clay	17.5
from loam sandy to sandy loam in surface, from loam sandy clay to clay in depth	17.5
from sandy loam to loamy sandy clay	17.5
from sandy loam and clay loam	20.0
from sandy and clay loam in the surface, from loam sandy clay to clay in depth	20.0
from sandy loam to clay loam	20.0
from sandy loam to loam clay with a varied and plenty content of skeleton	20.0
from sandy loam to clay sandy	27.5
sandy clay loam	27.5
from clay loam to sandy and clay loam	30.0
clay loam	33.5
from sandy loam to clayey	60.0
from clay to clay loam	63.5
from clay loam to clay	63.5
from clay sandy to clay	67.5
from clay to silty clay	70.0
clay	77.5

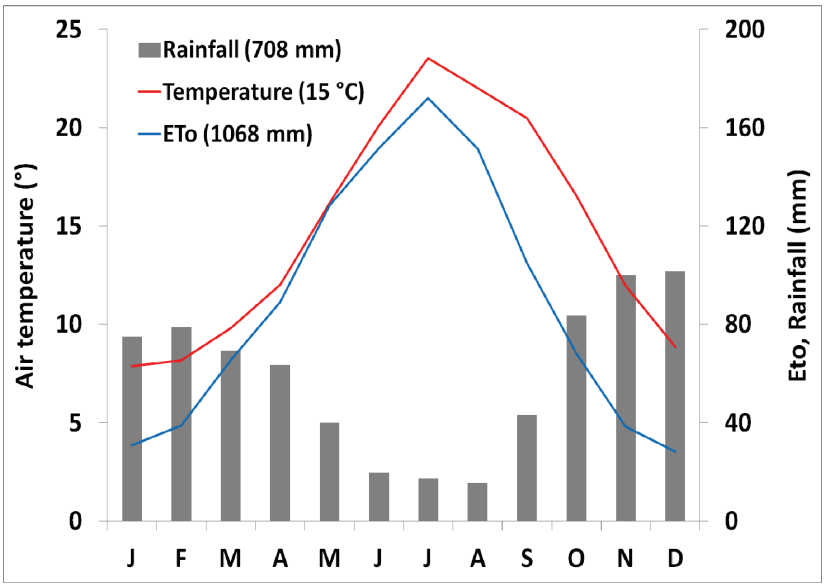

Figure 20 Monthly average of actual climate data input .

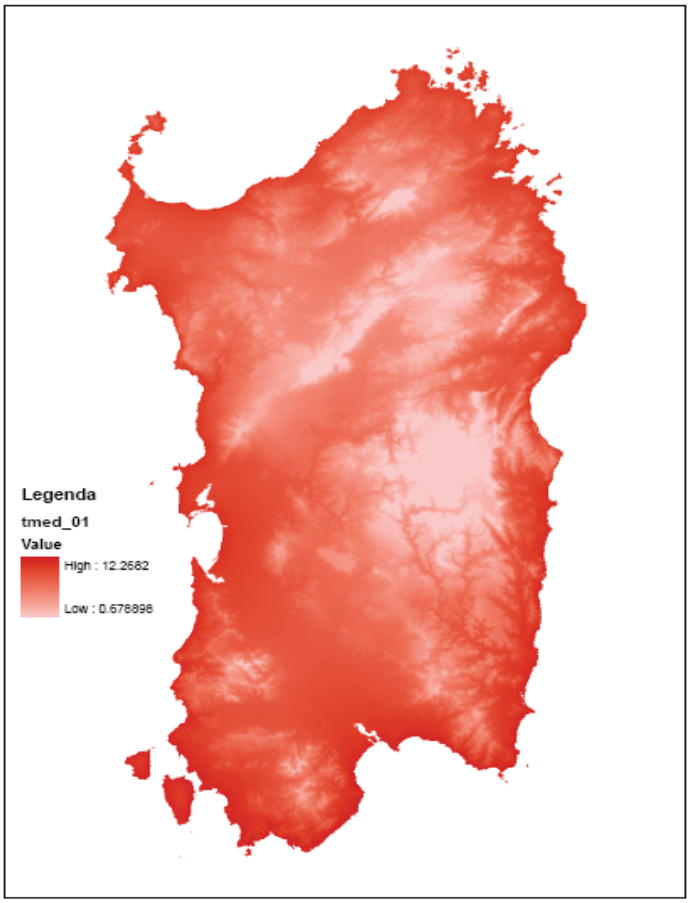

Fig. 21 Example of meteorological file grid format ESRI (250m resolution). The map showed the value of average monthly mean air temperature (°C) of January. Values range from a minimum of 0.68 °C to a maximum of 12.25 °C (from Arpas).

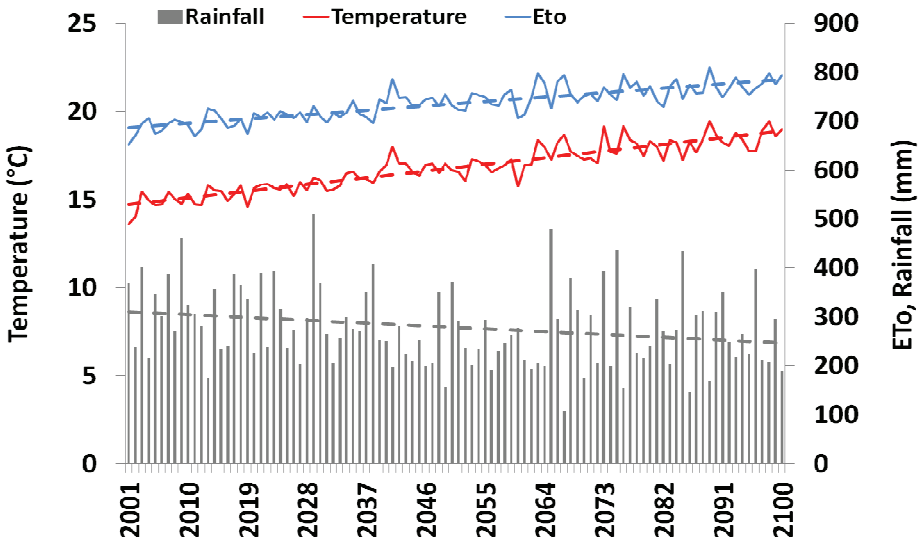

Figure 22 Climatic trend of A1b scenario (from CMCC)

4. Statistical analysis

Model evaluation, in its simple form, is a comparison between simulated and observed values. Through the calibration, based on comparisons between experimental results and the results provided by the model projections, the parameters of the simulation model were evaluated in order to improve its ability to describe properly system functioning.). With the validation, the calibrated model is used in real situations to verify the reliability of predictions (correspondence between the results of simulations and field results).

The performance of model was determined using several indexes mainly based on the calculation of correlation and differences between estimate and measured values. Results obtained from data used for the calibration and validation of plant residue input for each UDS classes were analyzed calculating the correlation coefficient (*r*) and its square, the coefficient of determination (R^2), the root mean squared error (RMSE), general standard deviation or relative root mean squared error (GSD), modelling efficiency index (EF), mean bias error (MBE) and mean absolute error (MAE).

The Pearson correlation coefficient (*r*) gives a measure of the degree of association between simulated and measured result; it is the correlation coefficient between measured and calculated values defined as:

$$r = \frac{\sum_{i=1}^{n}(E_i - \overline{E}) \cdot (M_i - \overline{M})}{\sqrt{\sum_{i=1}^{n}(E_i - \overline{E})^2 \cdot \sum_{i=1}^{n}(M_i - \overline{M})^2}}$$

The range of *r* is $-1 \leq r \leq 1$. A value of $r=1$ indicates that there exists a perfect linear relationship between simulated and observed values. However this does not necessarily imply that the model is perfect.

The RMSE provides a percentage term for the total difference between simulated and measured results, it was used to test the accuracy of the model, which is defined as the variation, expressed in the same unit as the data, between simulated and measured values (Loague and Green, 1991):

$$RMSE = \sqrt{\frac{\sum_{i=1}^{n}(E_i - M_i)^2}{n}}$$

where Ei and Mi indicate the simulated and measured values and n the number of values. RMSE represents the typical size of model error, with values equalling or near zero indicating perfect or near perfect estimates. The RMSE was also expressed as a coefficient of variation (GSD) by dividing it by the mean of the measured yield or modelled values ($\overline{M}$):

$$GDS = \sqrt{\frac{\sum_{i=1}^{n}(E_i - M_i)^2}{n}} \frac{100}{\overline{M}} = RMSE \frac{100}{\overline{M}}$$

In addition, the accuracy of the model was evaluated using another index based on squared differences, the modelling efficiency index (EF):

$$EF = 1 - \frac{\sum_{i=1}^{n}(E_i - M_i)^2}{\sum_{i=1}^{n}(M_i - \overline{M})^2}$$

EF values greater than 0 indicate that the model estimates are better predictors than the average measured value, with negative values indicating the opposite.

A EF value equal or near 1 means a perfect or near perfect estimates.

Finally, the mean bias error (MBE) and the mean absolute error (MAE) were used to measure the tendency of the model to overestimate or underestimate the measured value:

$$MBE = \sum_{i=1}^{n} \frac{E_i - M_i}{n}$$

$$MAE = \sum_{i=1}^{n} \frac{|E_i - M_i|}{n}$$

A positive bias error indicates a tendency to over predict a variable while a negative bias error implies a tendency to under predict a variable. In this case the MBE is negative). MAE values near or equal to zero indicate a better match along the 1:1 line comparison of estimated and observed values (Rasse *et al.,* 2000).

RESULTS

1. Calibration and validation of RothC model in Sardinia soils

This section reports the results obtained from the first RothC data, using as litterfall input data coming from the literature. Figure 23 shows the map of total C pool in the soils calculated from this preliminary run, while the figure 24 shows the CO_2 pool map before model calibration.

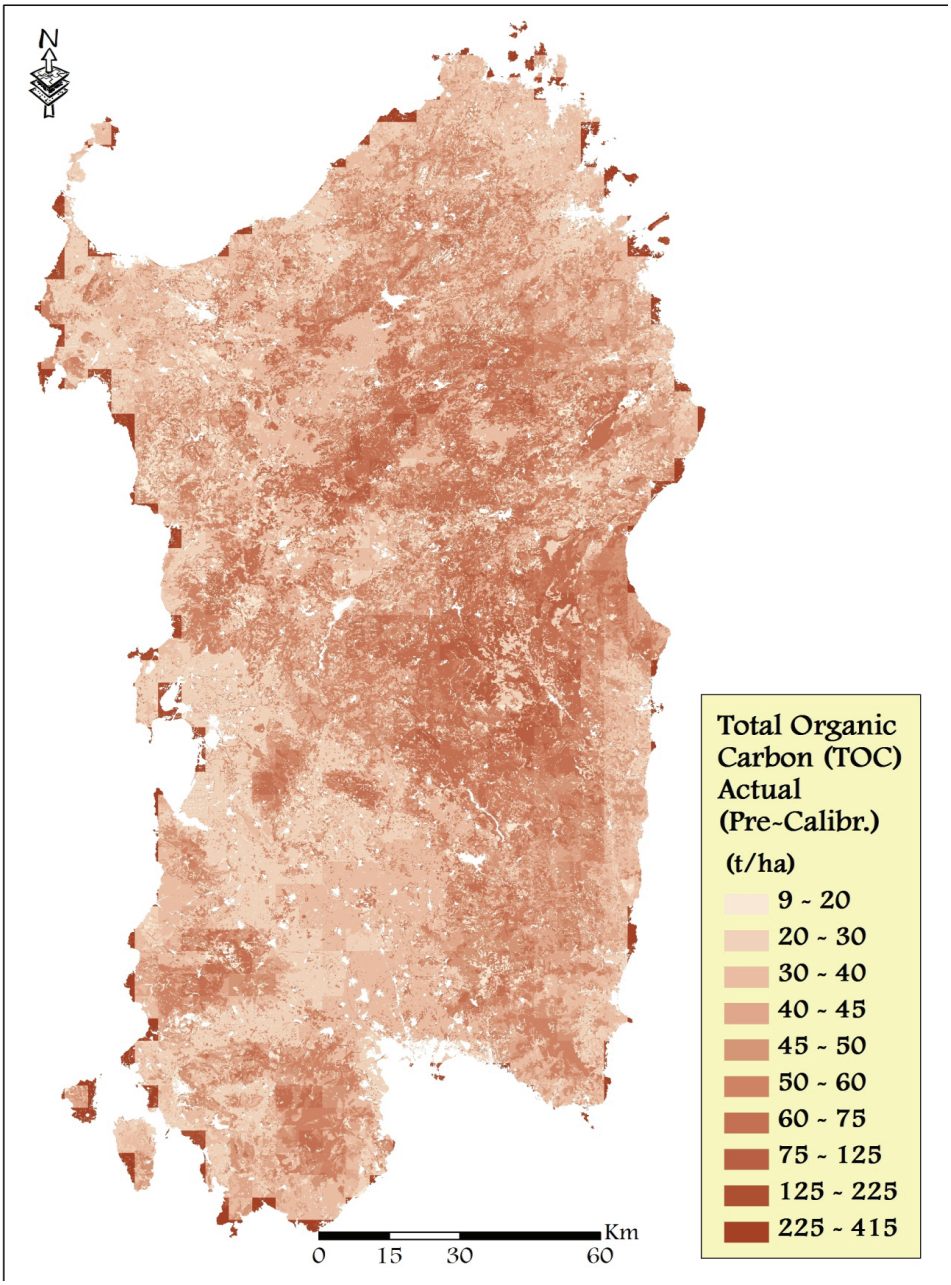

Figure 23. Map of total C pool in Sardinia obtained before the model calibration.

The total organic C amount in the soil is estimated as 390.4 Mt C ha^{-1} (46.1 t C ha^{-1}, on average), and the CO_2 released is 5479.1 Mt C (2.39 t C*1000 ha^{-1}, on average). Table 7 reports the C pool amounts for the LU classes used in the run.

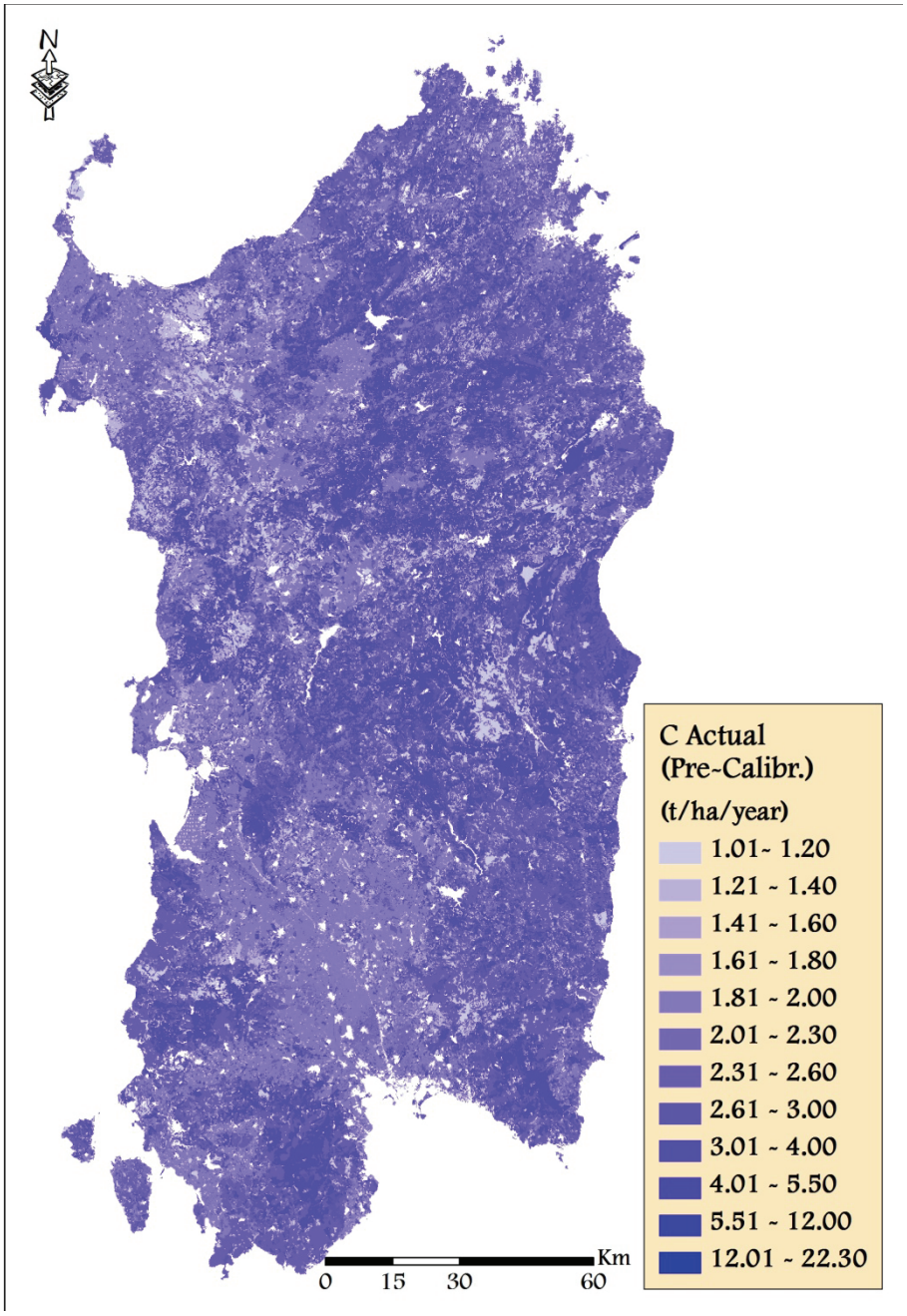

Figure 24. Map of CO_2 pool in Sardinia obtained before the model calibration.

Table 7. Values of decomposable plant material (DPM), resistant plant material (RPM), microbial biomass (BPM), humified organic matter (HUM), total organic C (SUM) and evolved CO_2 (CO_2) pools estimated in Sardinia before the calibration.

UDS CLASS	ha	DPM C (t*1000)	RPM C (t*1000)	BIO C (t*1000)	HUM C (t*1000)	SUM C (t*1000)	CO2 C (t*1000)
211	415801	92.4	2074.2	300.0	11296.8	13763.4	783.2
212	212525	47.3	1033.9	146.7	5514.2	6742.1	400.3
221	24693	5.0	236.2	16.0	601.6	858.9	52.8
222	11908	2.4	106.4	7.1	267.9	383.8	25.5
223	48779	0.5	331.1	25.7	973.5	1330.8	68.3
231	10316	4.2	94.6	14.1	529.2	642.2	34.9
241	70078	13.7	416.9	60.2	2298.2	2789.0	167.6
242	42214	18.9	418.5	59.0	2199.3	2695.7	142.8
243	29287	6.6	363.9	35.5	1329.3	1735.3	92.3
244	57430	12.2	684.8	68.0	2566.7	3331.7	181.1
311	452356	36.3	7318.2	534.8	20235.5	28124.8	1424.8
312	38716	4.5	681.6	47.6	1762.1	2495.7	104.1
313	12350	1.1	200.5	14.0	526.2	742.0	36.1
321	144793	11.8	413.5	60.5	2274.4	2760.2	153.5
322	17608	3.6	199.1	18.9	709.9	931.6	45.8
323	561021	116.3	6429.4	616.5	22895.6	30057.9	1457.8
324	92693	8.6	1395.6	101.1	3793.8	5299.0	255.8
333	49572	4.8	158.4	21.3	792.9	977.5	52.5
Total	**2292139**	**390.4**	**22556.8**	**2147.1**	**80567.3**	**105661.6**	**5479.1**
t C ha^{-1}		0.17	9.84	0.94	35.15	46.10	2.39

The data obtained from the first RothC run (the total amount of C) were compared with the ground data (80 points). The figure 25 shows the regression obtained comparing modeled and measured C data in those points.

For the calibration, the input data related to litterfall were adjusted (*table 8 a,b*). This way was chosen because that input data is affected more than the other by a certain uncertainty, while the other factors are already well tested in previous works for different situations.

Even the RothC model, as the same Authors suggest in the manual, is designed to run in two modes: "forward" in which known inputs are used to calculate changes in soil organic matter and "inverse", when inputs are calculated from know changes in soil organic matter.

Litterfall data were then adjusted and runs of RothC were performed to obtain the best R^2 coefficient between measured and modeled data.

Tables 9, 10,11,12,13, 14 shows the results obtained in the statistical analysis before and after calibration. Generally after calibration all statistical indexes are improved.

In particular before calibration the Pearson coefficient is 0.9 while after calibration is 1. Also modeling efficiency value is enhanced after calibration, proving a near perfect estimates.

The tables 13 and 14 summarize the statistical results obtained comparing modeled and measured C data in the model validation, generally even in this case there are goods results. The figure 26 shows the regression obtained comparing modeled and measured C data in model validation (R^2 0.9).

Table 8a. Value of DPM/RPM, plant residue (pr) and soil cover (sc) associated each LU class before (a) and after (b) calibration.

UDS	DPM/ RPM	Lf , sc Jan.	Lf , sc Feb.	Lf , sc Mar.	Lf , sc Apr.	Lf , sc May	Lf , sc June	Lf , sc July	Lf , sc Aug.	Lf , sc Sept.	Lf , sc Oct.	Lf , sc Nov.	Lf , sc Dec.
211	**1.44**	0.157 , 1	0.157 , 1	0.157 , 1	0.157 , 1	0.157 , 1	0.157 , 1	0.157 , 1	0.157 , 1	0.157 , 1	0.157 , 1	0.157 , 1	0.157 , 1
212	**1.44**	0.157 , 1	0.157 , 1	0.157 , 1	0.157 , 1	0.157 , 1	0.157 , 1	0.157 , 1	0.157 , 1	0.157 , 1	0.157 , 1	0.157 , 1	0.157 , 1
221	**0.25**	0.000 , 0	0.000 , 0	0.000 , 0	0.000 , 1	0.000 , 1	0.000 , 1	0.000 , 1	0.000 , 1	0.535 , 1	0.535 , 1	0.535 , 1	0.535 , 0
222	**0.25**	0.000 , 0	0.000 , 0	0.000 , 0	0.000 , 1	0.000 , 1	0.000 , 1	0.000 , 1	0.000 , 1	0.535 , 1	0.535 , 1	0.535 , 1	0.535 , 0
223	**0.25**	0.280 , 1	0.280 , 1	0.280 , 1	0.280 , 1	0.280 , 1	0.000 , 1	0.000 , 1	0.000 , 1	0.000 , 1	0.000 , 1	0.000 , 1	0.000 , 1
231	**1,44**	0.282 , 1	0.282 , 1	0.282 , 1	0.282 , 1	0.282 , 1	0.282 , 1	0.282 , 1	0.282 , 1	0.282 , 1	0.282 , 1	0.282 , 1	0.282 , 1
241	**1,44**	0.281 , 1	0.281 , 1	0.281 , 1	0.281 , 1	0.281 , 1	0.281 , 1	0.141 , 1	0.141 , 1	0.141 , 1	0.141 , 1	0.141 , 1	0.141 , 1
242	**1,44**	0.282 , 1	0.282 , 1	0.282 , 1	0.282 , 1	0.282 , 1	0.282 , 1	0.282 , 1	0.282 , 1	0.282 , 1	0.282 , 1	0.282 , 1	0.282 , 1
243	**0,67**	0.196 , 1	0.196 , 1	0.196 , 1	0.259 , 1	0.279 , 1	0.341 , 1	0.366 , 1	0.416 , 1	0.316 , 1	0.196 , 1	0.196 , 1	0.196 , 1
244	**0,67**	0.196 , 1	0.196 , 1	0.196 , 1	0.259 , 1	0.279 , 1	0.341 , 1	0.366 , 1	0.416 , 1	0.316 , 1	0.196 , 1	0.196 , 1	0.196 , 1
311	**0.25**	0.100 , 1	0.100 , 1	0.100 , 1	0.350 , 1	0.350 , 1	0.500 , 1	0.500 , 1	0.550 , 1	0.300 , 1	0.100 , 1	0.100 , 1	0.100 , 1
312	**0.25**	0.120 , 1	0.120 , 1	0.120 , 1	0.120 , 1	0.200 , 1	0.300 , 1	0.400 , 1	0.550 , 1	0.400 , 1	0.120 , 1	0.120 , 1	0.120 , 1
313	**0,25**	0.110 , 1	0.110 , 1	0.110 , 1	0.235 , 1	0.275 , 1	0.400 , 1	0.450 , 1	0.550 , 1	0.350 , 1	0.110 , 1	0.110 , 1	0.110 , 1
321	**1,44**	0.020 , 1	0.020 , 1	0.050 , 1	0.250 , 1	0.200 , 1	0.100 , 1	0.050 , 1	0.050 , 1	0.100 , 1	0.150 , 1	0.050 , 1	0.020 , 1
322	**0,67**	0.010 , 1	0.050 , 1	0.100 , 1	0.100 , 1	0.200 , 1	0.600 , 1	0.500 , 1	0.500 , 1	0.100 , 1	0.100 , 1	0.100 , 1	0.200 , 1
323	**0,67**	0.010 , 1	0.050 , 1	0.100 , 1	0.100 , 1	0.200 , 1	0.600 , 1	0.500 , 1	0.500 , 1	0.100 , 1	0.100 , 1	0.100 , 1	0.200 , 1
324	**0,25**	0.105 , 1	0.080 , 1	0.105 , 1	0.168 , 1	0.238 , 1	0.500 , 1	0.475 , 1	0.525 , 1	0.225 , 1	0.105 , 1	0.080 , 1	0.155 , 1
333	**1,44**	0.020 , 1	0.020 , 1	0.050 , 1	0.250 , 1	0.200 , 1	0.100 , 1	0.050 , 1	0.050 , 1	0.100 , 1	0.150 , 1	0.050 , 1	0.020 , 1

Table 8b

UDS	DPM/ RPM	pr , sc Jan.	pr , sc Feb.	pr , sc Mar.	pr , sc Apr.	pr , sc May	pr , sc June	pr , sc July	pr , sc Aug.	pr , sc Sept.	pr , sc Oct.	pr , sc Nov.	pr , sc Dec.
211	**1.44**	0.152 , 1	0.152 , 1	0.152 , 1	0.152 , 1	0.152 , 1	0.152 , 1	0.152 , 1	0.152 , 1	0.152 , 1	0.152 , 1	0.152 , 1	0.152 , 1
212	**1.44**	0.152 , 1	0.152 , 1	0.152 , 1	0.152 , 1	0.152 , 1	0.152 , 1	0.152 , 1	0.152 , 1	0.152 , 1	0.152 , 1	0.152 , 1	0.152 , 1
221	**0.25**	0.000 , 0	0.000 , 0	0.000 , 0	0.000 , 1	0.000 , 1	0.000 , 1	0.000 , 1	0.000 , 1	0.520 , 1	0.520 , 1	0.520 , 1	0.520 , 0
222	**0.25**	0.000 , 0	0.000 , 0	0.000 , 0	0.000 , 1	0.000 , 1	0.000 , 1	0.000 , 1	0.000 , 1	0.520 , 1	0.520 , 1	0.520 , 1	0.520 , 0
223	**0.25**	0.330 , 1	0.330 , 1	0.330 , 1	0.330 , 1	0.330 , 1	0.000 , 1	0.000 , 1	0.000 , 1	0.000 , 1	0.000 , 1	0.000 , 1	0.000 , 1
231	**1,44**	0.231 , 1	0.231 , 1	0.231 , 1	0.231 , 1	0.231 , 1	0.231 , 1	0.231 , 1	0.231 , 1	0.231 , 1	0.231 , 1	0.231 , 1	0.231 , 1
241	**1,44**	0.281 , 1	0.281 , 1	0.281 , 1	0.281 , 1	0.281 , 1	0.116 , 1	0.116 , 1	0.116 , 1	0.116 , 1	0.116 , 1	0.116 , 1	0.116 , 1
242	**1,44**	0.231 , 1	0.231 , 1	0.231 , 1	0.231 , 1	0.231 , 1	0.231 , 1	0.231 , 1	0.231 , 1	0.231 , 1	0.231 , 1	0.231 , 1	0.231 , 1
243	**0,67**	0.183 , 1	0.183 , 1	0.183 , 1	0.246 , 1	0.258 , 1	0.331 , 1	0.356 , 1	0.403 , 1	0.291 , 1	0.183 , 1	0.183 , 1	0.183 , 1
244	**0,67**	0.183 , 1	0.183 , 1	0.183 , 1	0.246 , 1	0.258 , 1	0.331 , 1	0.356 , 1	0.403 , 1	0.291 , 1	0.183 , 1	0.183 , 1	0.183 , 1
311	**0.25**	0.110 , 1	0.110 , 1	0.110 , 1	0.360 , 1	0.360 , 1	0.550 , 1	0.550 , 1	0.600 , 1	0.300 , 1	0.110 , 1	0.110 , 1	0.110 , 1
312	**0.25**	0.160 , 1	0.160 , 1	0.160 , 1	0.160 , 1	0.210 , 1	0.310 , 1	0.410 , 1	0.550 , 1	0.400 , 1	0.160 , 1	0.160 , 1	0.160 , 1
313	**0,25**	0.135 , 1	0.135 , 1	0.135 , 1	0.260 , 1	0.285 , 1	0.430 , 1	0.480 , 1	0.575 , 1	0.350 , 1	0.135 , 1	0.135 , 1	0.135 , 1
321	**1,44**	0.050 , 1	0.050 , 1	0.140 , 1	0.340 , 1	0.340 , 1	0.140 , 1	0.050 , 1	0.050 , 1	0.140 , 1	0.240 , 1	0.140 , 1	0.050 , 1
322	**0,67**	0.050 , 1	0.100 , 1	0.100 , 1	0.100 , 1	0.200 , 1	0.500 , 1	0.400 , 1	0.400 , 1	0.100 , 1	0.100 , 1	0.100 , 1	0.050 , 1
323	**0,67**	0.050 , 1	0.100 , 1	0.100 , 1	0.100 , 1	0.200 , 1	0.500 , 1	0.400 , 1	0.400 , 1	0.100 , 1	0.100 , 1	0.100 , 1	0.050 , 1
324	**0,25**	0.093 , 1	0.118 , 1	0.118 , 1	0.180 , 1	0.243 , 1	0.465 , 1	0.440 , 1	0.488 , 1	0.225 , 1	0.118 , 1	0.118 , 1	0.093 , 1
333	**1,44**	0.050 , 1	0.050 , 1	0.140 , 1	0.340 , 1	0.340 , 1	0.140 , 1	0.050 , 1	0.050 , 1	0.140 , 1	0.240 , 1	0.140 , 1	0.050 , 1

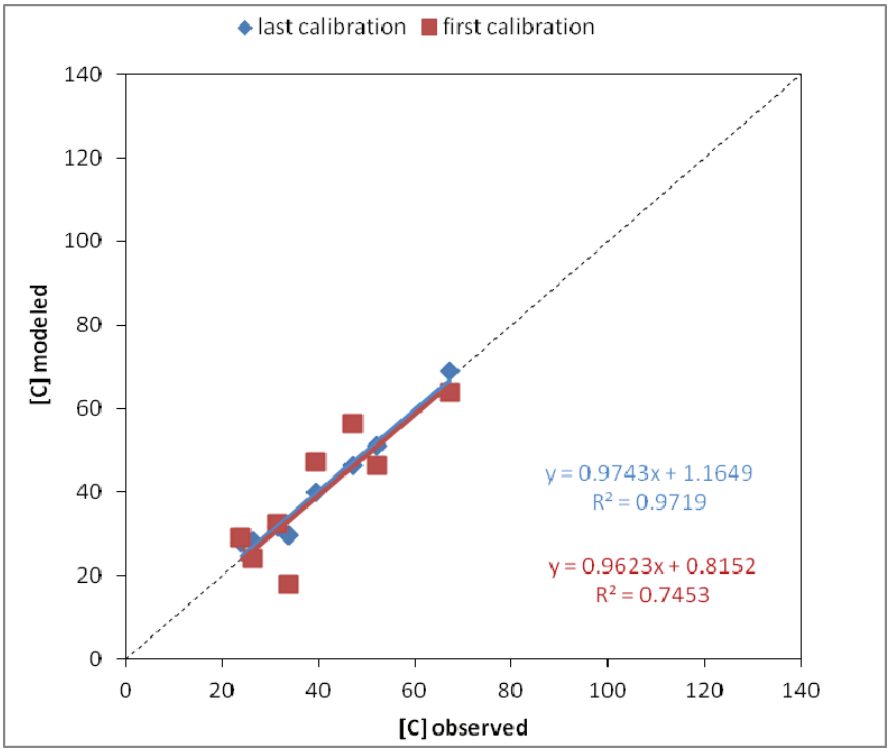

Figure 25. Linear regression between measured and modeled C data before and after the calibration.

Table 9. Observed and simulated data(t C ha^{-1}) for each LU class before calibration.

uds code	observed data	modeled data
211-212	31.7	32.3
221	24.0	29.1
223	26.5	24.0
231	47.2	56.4
311	67.3	63.8
312	52.2	46.2
321	33.8	17.8
323	39.6	47.2

Table 10. Statistical analysis before calibration

Statistical analisys		Observed data	Model data
Mean		40.3	39.6
Standard Deviation		14.6	16.3
Maximun		67.3	63.8
Minimun		24.0	17.8
Numbers of samples	8.0		
Pearson coefficient	0.9		
Coefficient of determination	0.7		
Root Mean Square Error	7.7		
General Standard Deviation %	7.0		
Modeling Efficiency	0.7		
Mean Bias Error	-0.7		
Mean Absolute Error	6.3		

Table 11. Observed and simulated data(t C ha^{-1}) for each LU class after calibration.

uds code	observed data	modeled data
211-212	31.7	31.7
221	24.0	28.0
223	26.5	28.2
231	47.2	46.2
311	67.3	68.9
312	52.2	51.0
321	33.8	29.5
323	39.6	39.9

Table 12. Statistical analysis after calibration

Statistical analisys			Observed data	Model data
Mean			40.3	38.8
Standard Deviation			14.6	13.9
Maximun			67.3	68.9
Minimun			24.0	28.0
Numbers of samples		8.0		
Pearson coefficient	r	1.0		
Coefficient of determination		0.9		
Root Mean Square Error		2.3		
General Standard Deviation	%	7.0		
Modeling Efficiency		1.0		
Mean Bias Error		0.1		
Mean Absolute Error		1.8		

Table 13. Observed and simulated data for each LU class in the model validation.

uds code	observed data	modeled data
211-212	30.5	30.3
221	24.2	29.9
223	24.5	26.3
231	49.0	50.5
311	74.0	69.7
312	51.5	58.5
321	34.9	28.9
323	43.0	41.0

Table 14. Statistical analysis in the model validation

Statistical analisys			Observed data	Model data
Mean			41.5	41.9
Standard Deviation			16.8	16.1
Maximun			74.0	69.7
Minimun			24.2	26.3
Numbers of samples		8.0		
Pearson coefficient	r	1.0		
Coefficient of determination		0.9		
Root Mean Square Error		4.3		
General Standard Deviation	%	10.0		
Modeling Efficiency		0.9		
Mean Bias Error		0.4		
Mean Absolute Error		3.6		

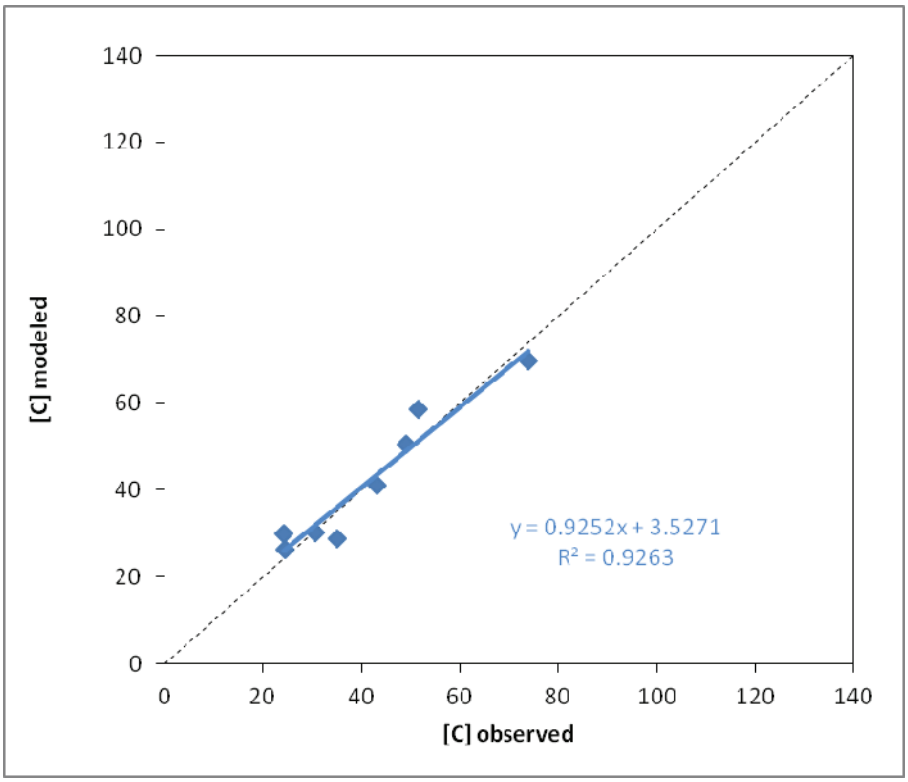

Figure. 26 Relationship between observed and simulated data in the model validation

The figure 27 and 28 shows respectively the total organic C pool map and CO_2 map after calibration.

The calibration affects the several C pools in different percentage.

As expected, the biggest difference between uncalibrated and calibrated C pool differences was found for the DPM. This is due to the fact that this C pool is directly affected by the litterfall rates.

The Figures 29a-b-c-d-e-f show the C pool amounts before and after the calibration. In general, the most affected LU classes after the calibration were the 321 (natural pasture), the 333 (areas with sparse vegetation), the 322 (bushes and shrubs) and the 323 (maquis and garrigue). The Table 15 reports the % values of the variation for the C pool amounts after the calibration for each LU class.

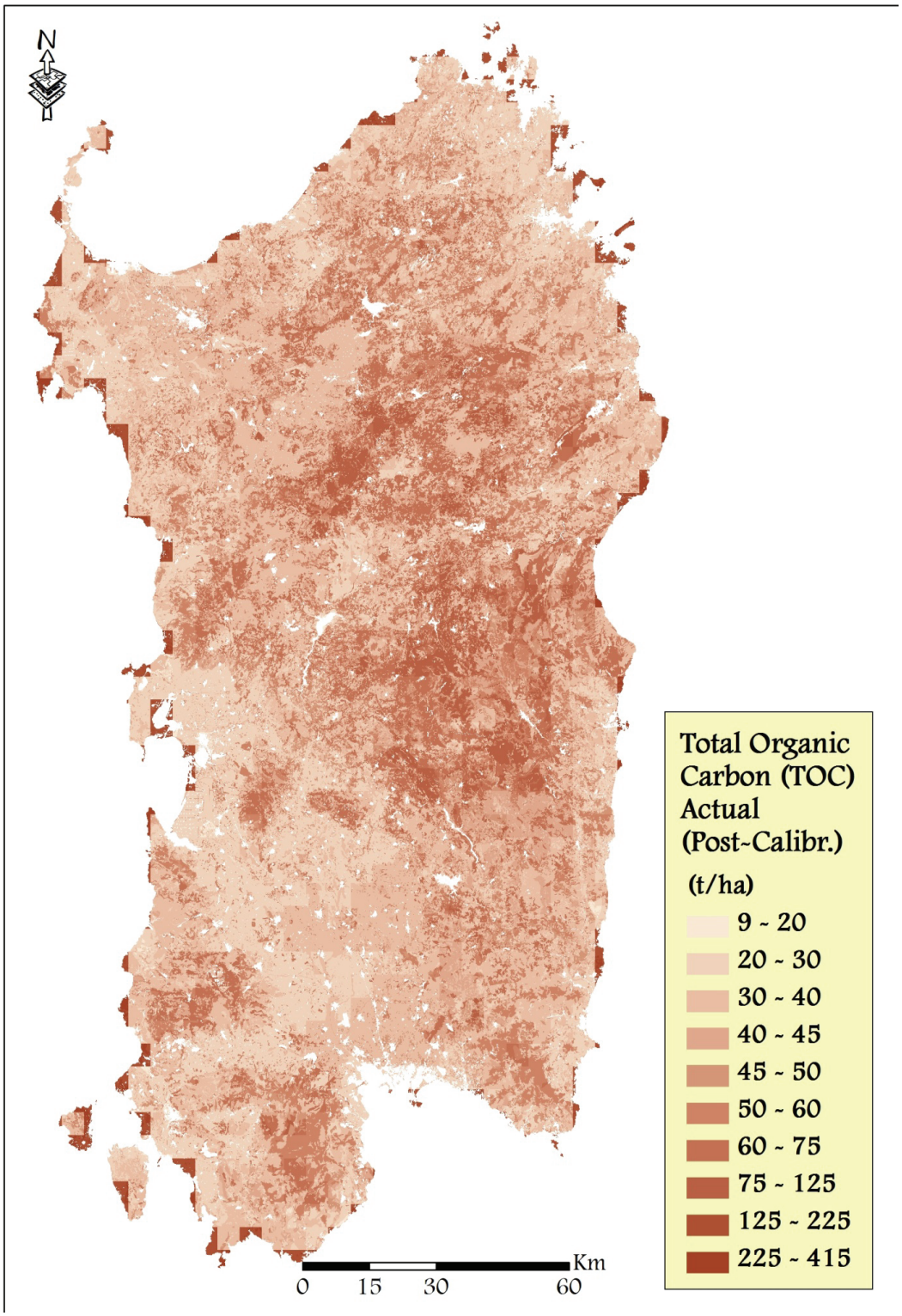

Figure 27 The total organic C (TOC) map post calibration

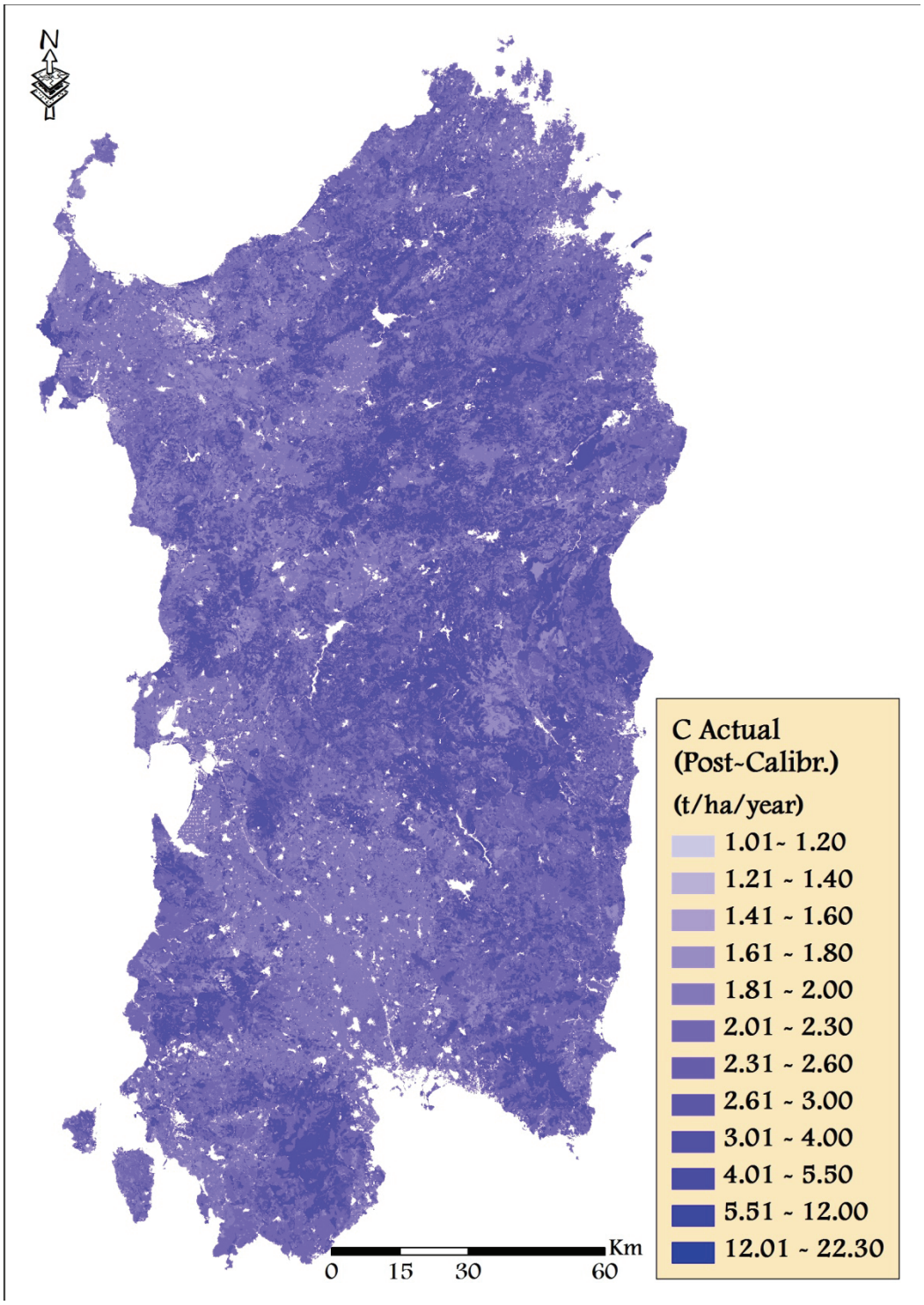

Figure 28 The CO_2 pool map post calibration

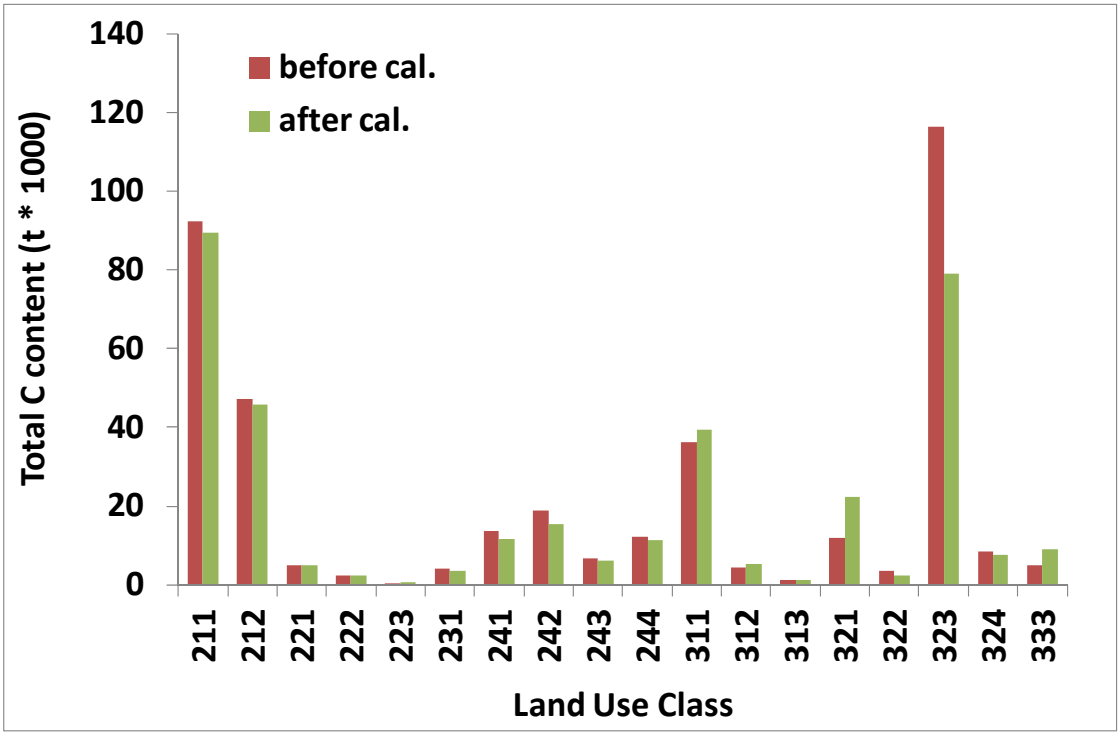

Figure 29a. Decomposable plant matherial (DPM) amounts in Sardinia before and after the calibration for each Land Use Class.

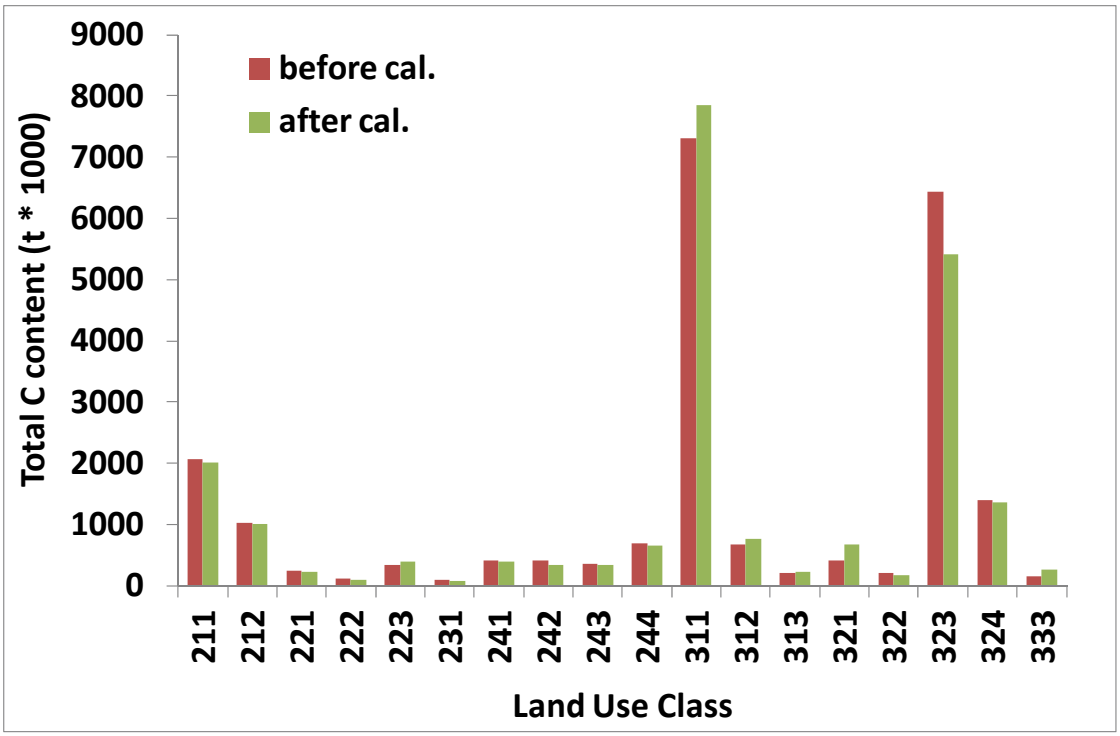

Figure 29b. Resistant plant matherial (RPM) amounts in Sardinia before and after the calibration for each Land Use Class.

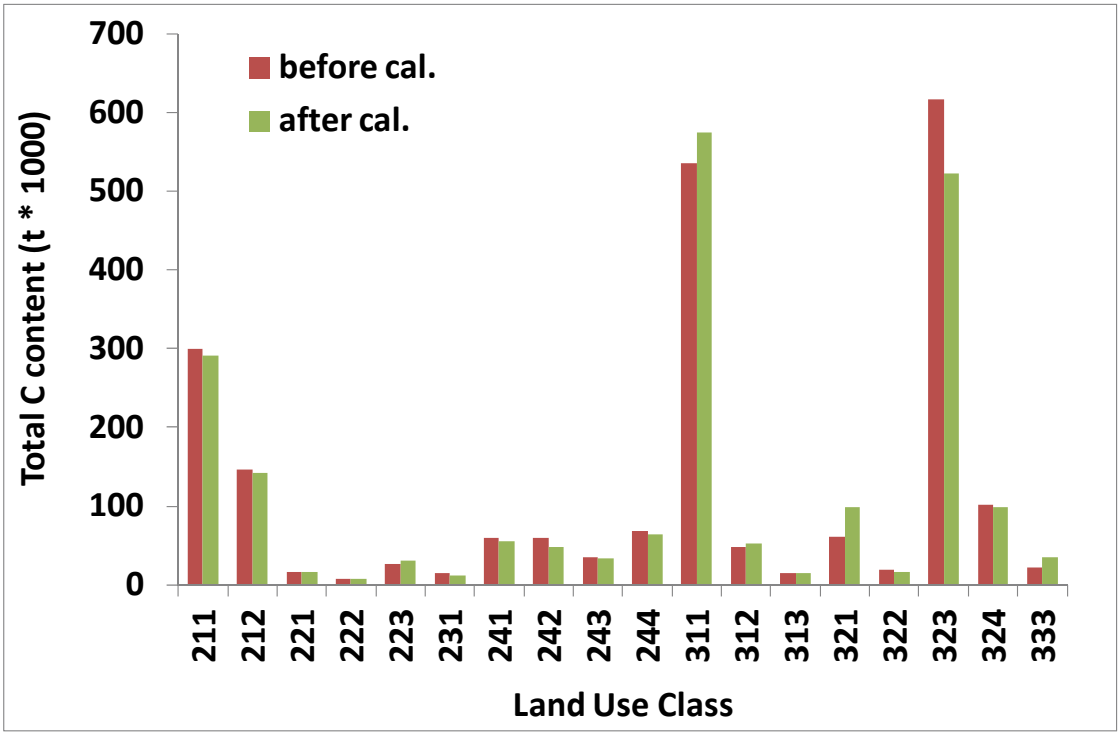

Figure 29c. Microbial biomass (BIO) amounts in Sardinia before and after the calibration for each Land Use Class.

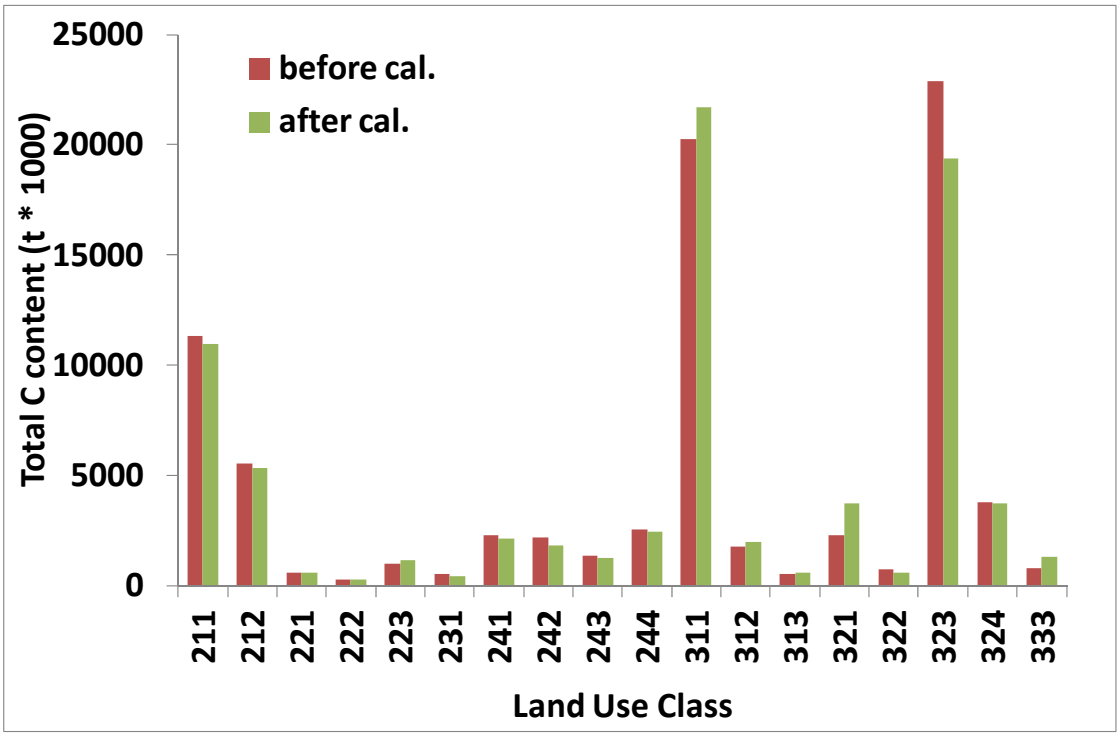

Figure 29d. Humified organic matter (HUM) amounts in Sardinia before and after the calibration for each Land Use Class.

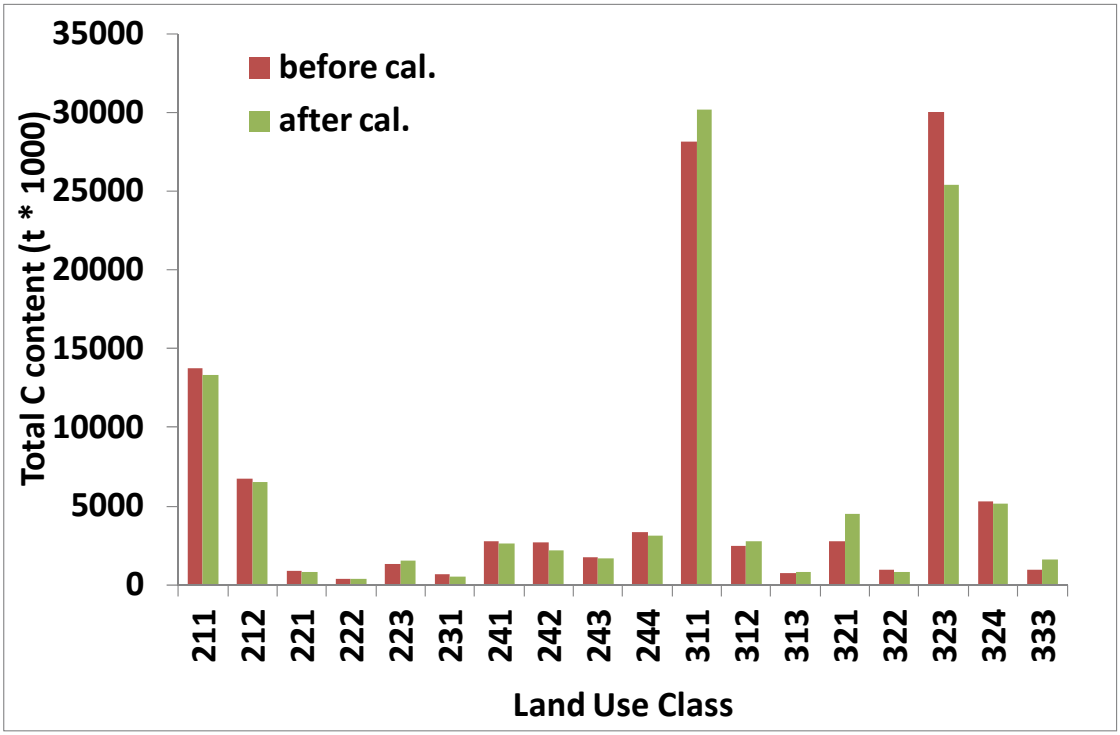

Figure 29e. Total organic C pool (SUM) amounts in Sardinia before and after the calibration for each Land Use Class.

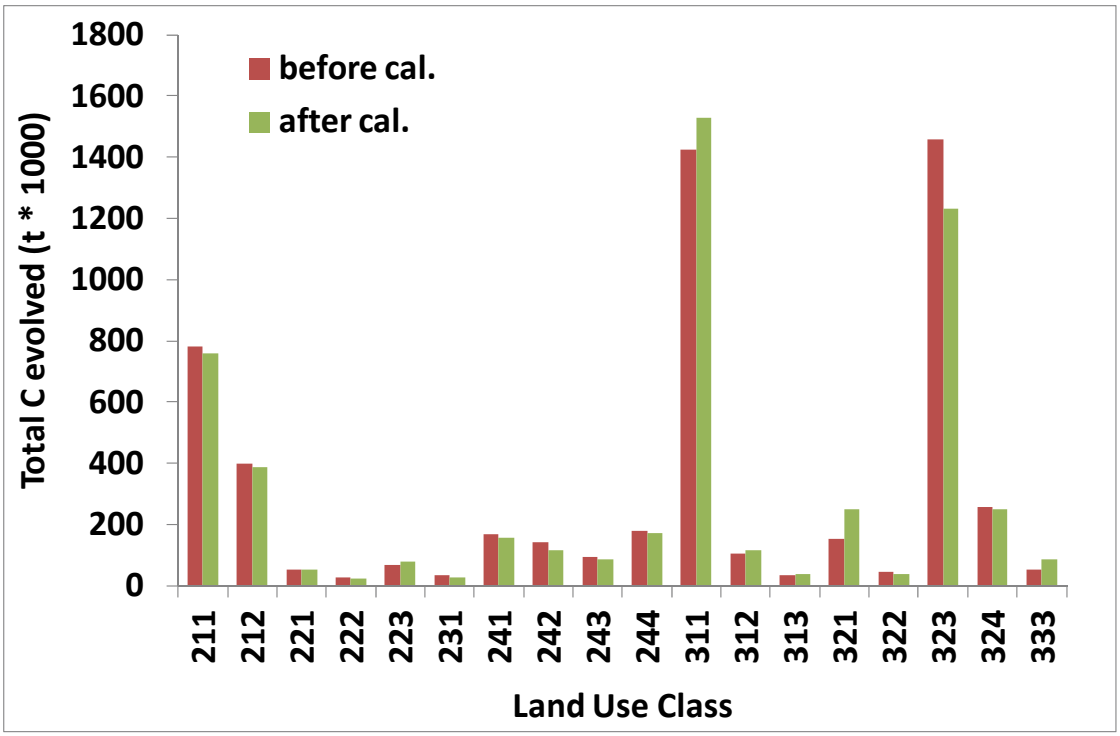

Figure 29f. C evolved as CO_2 amounts in Sardinia before and after the calibration for each Land Use Class.

Table 15. Relative variation for the C compartments in the soil before the calibration.

UDS CLASS	DPM %	RPM %	BIO %	HUM %	SUM %	CO2 %
211	-3.2	-3.2	-3.2	-3.2	-3.2	-3.2
212	-3.2	-3.2	-3.2	-3.2	-3.2	-3.2
221	-2.8	-2.8	-2.8	-2.8	-2.8	-2.8
222	-2.8	-2.8	-2.8	-2.8	-2.8	-2.8
223	17.9	17.9	17.9	17.9	17.9	17.9
231	-18.1	-18.1	-18.1	-18.1	-18.1	-18.1
241	-16.1	-7.6	-7.5	-7.3	-7.4	-7.3
242	-18.1	-18.1	-18.1	-18.1	-18.1	-18.1
243	-6.0	-5.4	-5.4	-5.4	-5.4	-5.4
244	-6.0	-5.4	-5.4	-5.4	-5.4	-5.4
311	8.1	7.3	7.3	7.3	7.3	7.3
312	15.8	11.5	11.4	11.5	11.5	11.5
313	12.6	9.2	9.2	9.2	9.2	9.2
321	89.2	63.3	62.7	63.2	63.3	63.2
322	-32.1	-15.7	-15.2	-15.4	-15.5	-15.4
323	-32.1	-15.7	-15.2	-15.4	-15.5	-15.4
324	-10.7	-2.4	-2.2	-2.2	-2.3	-2.2
333	84.4	63.3	62.8	63.2	63.3	63.2
average	0.16	9.71	0.92	34.58	45.4	2.36

2. Organic C pools in Sardinia with actual climate

The total C amount in Sardinia after the calibration, calculated for the 95.2% of the whole island surface, is estimated, at the year 2000 as 103979.1 t C*1000. This means, on average, an amount of 45.4 t C ha^{-1}.

Most of this C is stored as humified organic matter HUM (76.3%) and resistant plant material RPM (21.4%), that are the more stables C pools (*fig. 30*).

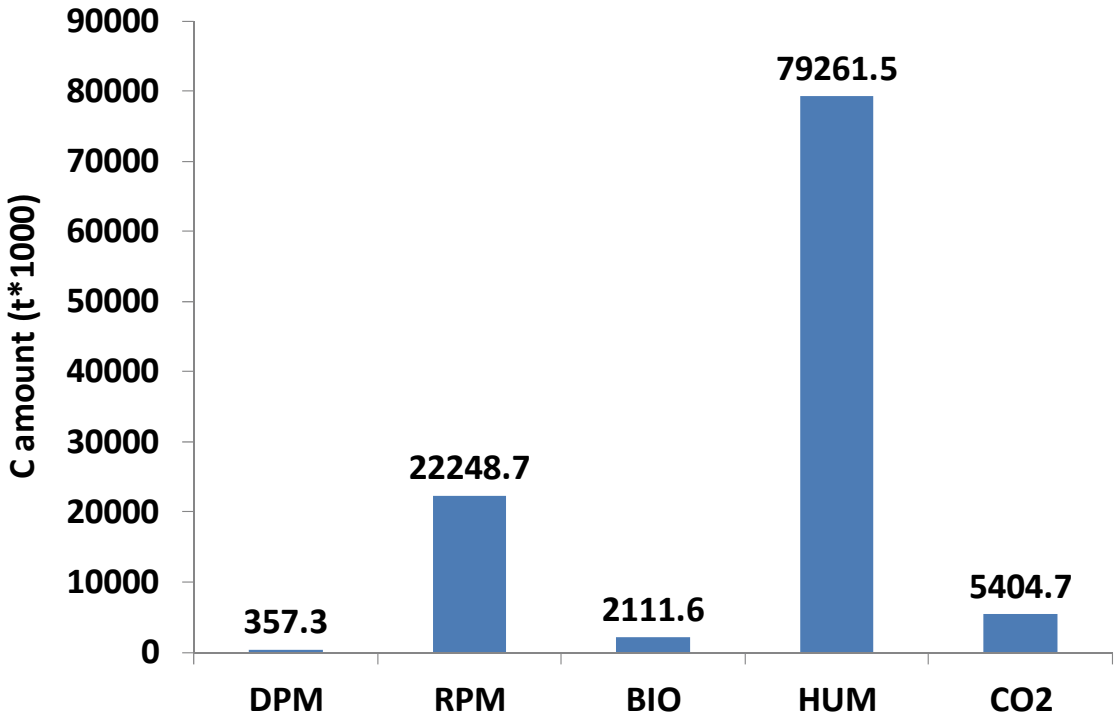

Figure 30. Soil C pools in Sardinia estimated at the year 2000 for each compartment.

In the Table 16, the amount of each C pool after the calibration for the several Land Use classes is showed.

Table 16. Values of decomposable plant matherial (DPM), resistant plant matherial (RPM), microbial biomass (BPM), humified organic matter (HUM), total organic C (SUM) and evolved CO_2 (CO_2) pools after the calibration (Year 2000).

UDS CLASS	DPM C (t*1000)	RPM C (t*1000)	BIO C (t*1000)	HUM C (t*1000)	SUM C (t*1000)	CO_2 C (t*1000)
211	92.4	2074.2	300.0	11296.8	13763.4	783.2
212	47.3	1033.9	146.7	5514.2	6742.1	400.3
221	5.0	236.2	16.0	601.6	858.9	52.8
222	2.4	106.4	7.1	267.9	383.8	25.5
223	0.5	331.1	25.7	973.5	1330.8	68.3
231	4.2	94.6	14.1	529.2	642.2	34.9
241	13.7	416.9	60.2	2298.2	2789.0	167.6
242	18.9	418.5	59.0	2199.3	2695.7	142.8
243	6.6	363.9	35.5	1329.3	1735.3	92.3
244	12.2	684.8	68.0	2566.7	3331.7	181.1
311	36.3	7318.2	534.8	20235.5	28124.8	1424.8
312	4.5	681.6	47.6	1762.1	2495.7	104.1
313	1.1	200.5	14.0	526.2	742.0	36.1
321	11.8	413.5	60.5	2274.4	2760.2	153.5
322	3.6	199.1	18.9	709.9	931.6	45.8
323	116.3	6429.4	616.5	22895.6	30057.9	1457.8
324	8.6	1395.6	101.1	3793.8	5299.0	255.8
333	4.8	158.4	21.3	792.9	977.5	52.5
Total	**357.3**	**22248.7**	**2111.6**	**79261.5**	**103979.1**	**5404.7**
t C ha^{-1}	0.16	9.71	0.92	34.58	45.36	2.36

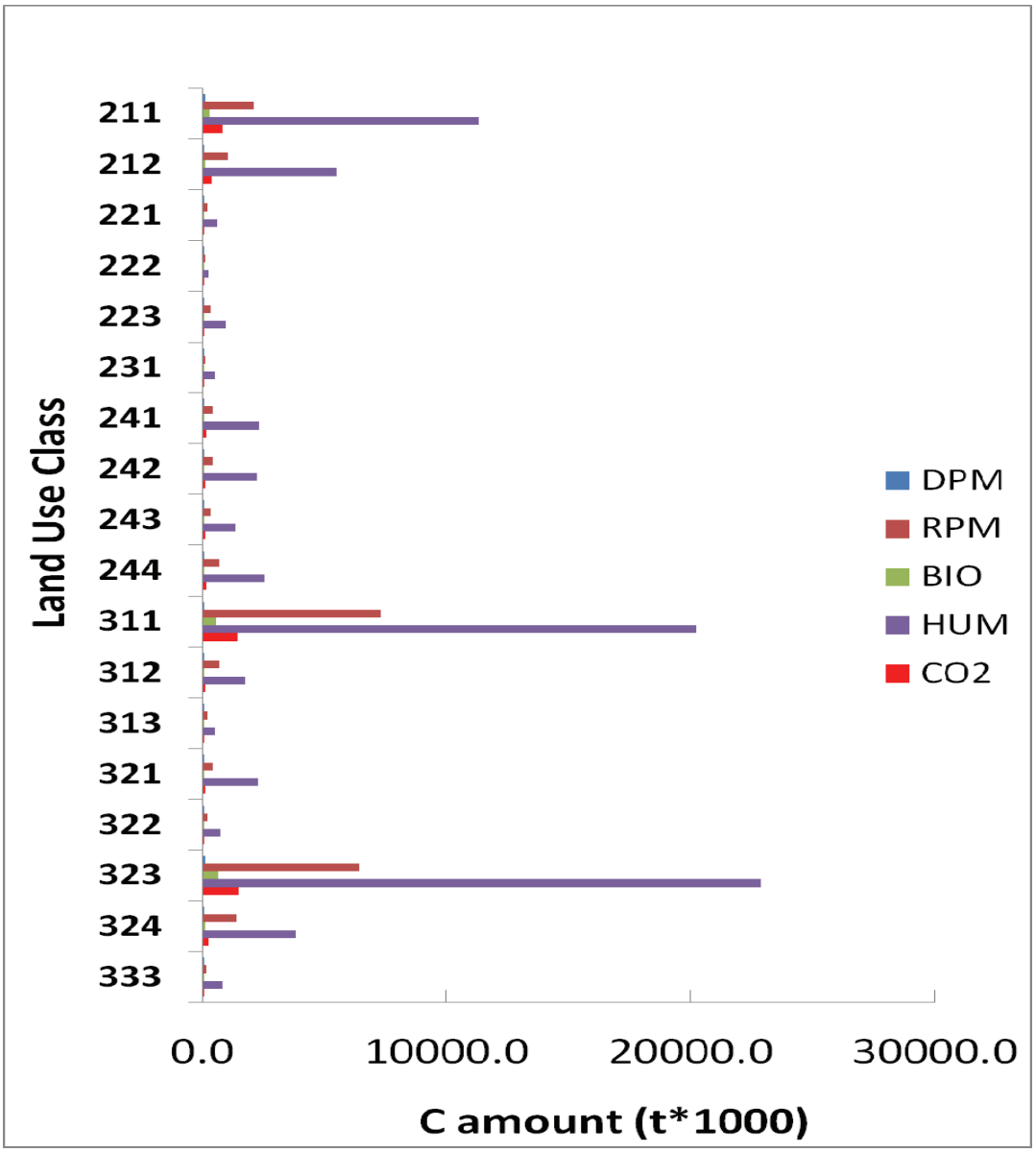

Figura 31 Estimates at the year 2000 of C stored in the 4 compartments and evolved as CO_2 for each Land Use class.

With respect to the Land Use typology, natural areas cover almost the 60% of the total area where C pools were estimated. On average, the C stored in the natural areas is higher than that stored in the agricultural areas (quantities per ha). The higher natural surfaces and the higher C amount per ha of these areas cause that, in Sardinia, these areas contain the 68.5% of the total C stored in soils (*tab. 17 and fig.31*).

With respect to the evolved CO_2 per ha, soils of natural areas release more CO_2 than agricultural areas (56.2% versus 43.8%, respectively). Rapporting these rates to the total natural and agricultural surfaces of Sardinia, data indicates that the 65.6% of the evolved CO_2 for the year 2000 comes from the natural areas (*Tab. 17*).

Table 17. C pool distribution estimation within agricultural and natural areas in Sardinia in the year 2000.

			Organic C				CO_2			
	ha	%	Mt C	%	t C ha^{-1}	%	Mt C	%	t C ha^{-1}	%
Agricultural areas	923030	40.3	32738.9	31.5	35.5	40.5	1861.9	34.4	2.02	43.8
Natural areas	1369109	59.7	71240.2	68.5	52.0	59.5	3542.8	65.6	2.59	56.2
Total	2292139		103979.1				5404.7			

3. C pools projections with the A1b climatic scenario

For a general view of the C pool projections, Figure 32 shows the general trend of DPM, BIO, HUM, RPM and the total C pool for the 21th century. The C projections for the 21th century show a general decrease of C stored in the soils for all compartments, due to a general enhanced respiration caused by a rising temperature levels. It is important to say that the projections showed in this work did not account for the feedbacks of climatic changes expected in the Land Use changes (LUC) and in the Net Primary Production (NPP). The exercise showed so far is only the first methodological step to estimate, in Sardinia, the climate change feedback in the soil C pools and dynamics, and LUC and NEP variations (these last directly connected to litterfall variations) should be accounted in the next step of the work.

The figure 33 and 34 (a, b, c, d, e)showed the maps of TOC and CO_2 pools projections.

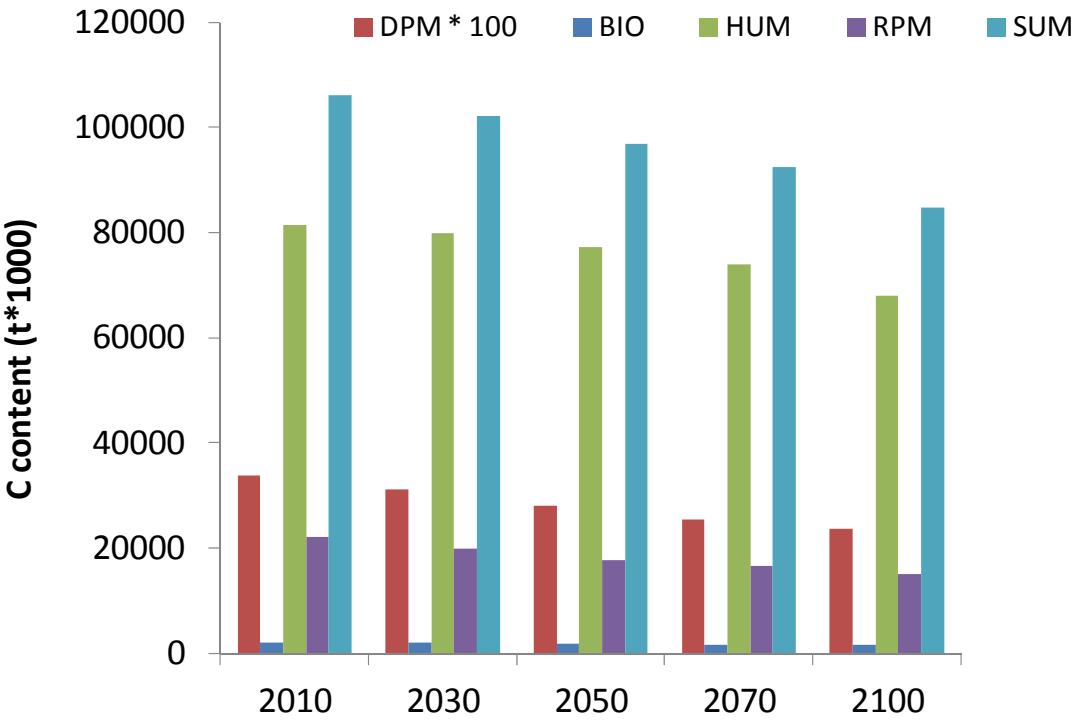

Figure 32. Estimated values of DPM, BIO, HUM, RPM, and total organic C (SUM) in Sardinia for the 21th century with the A1b climatic scenario (data area averaged for decade, e.g. 2010 = mean data of 2001-2010 period).

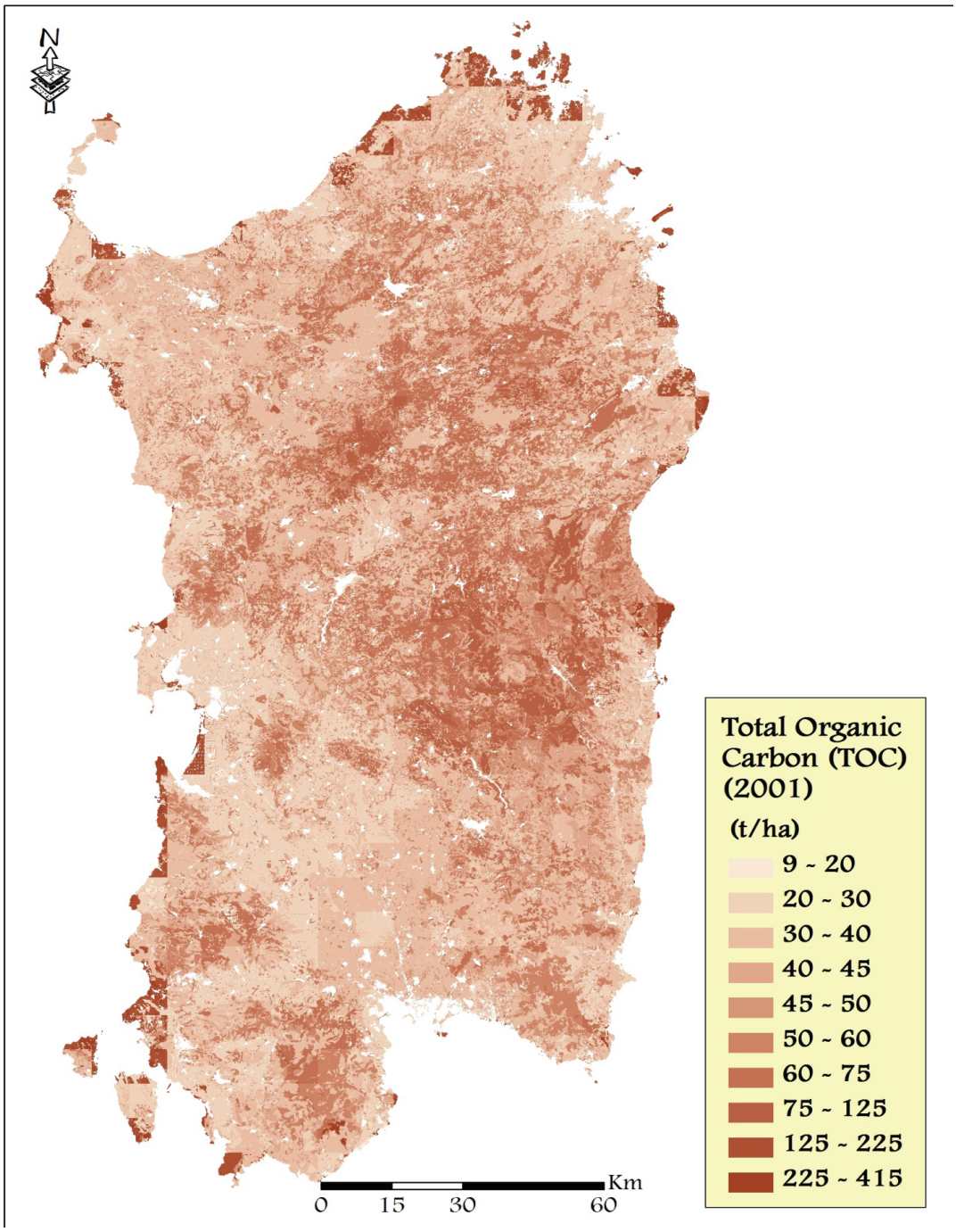

Figure 33a Map of total organic C (TOC) projection (2001).

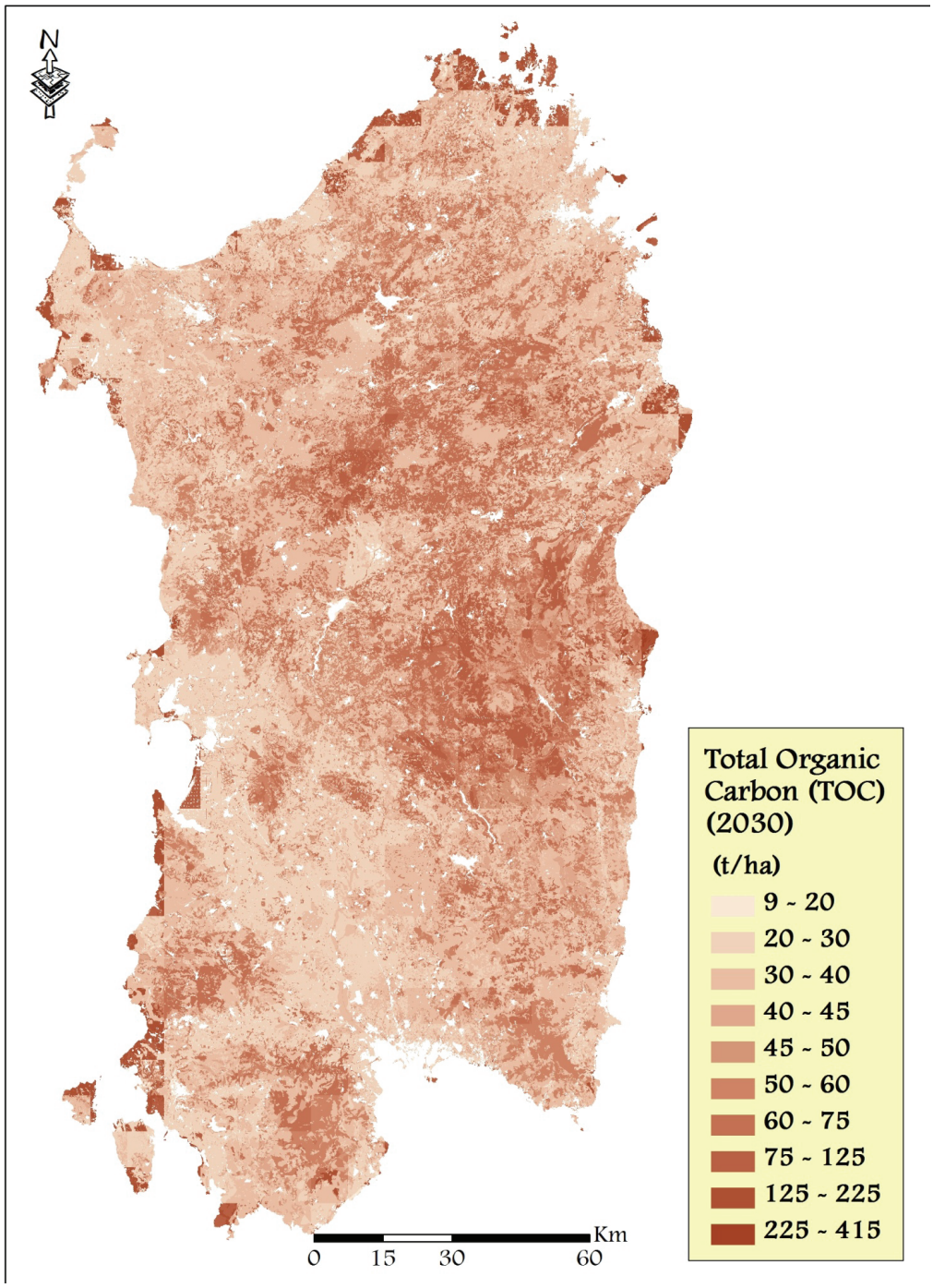

Figure 33b Map of total organic C (TOC) projection (2030).

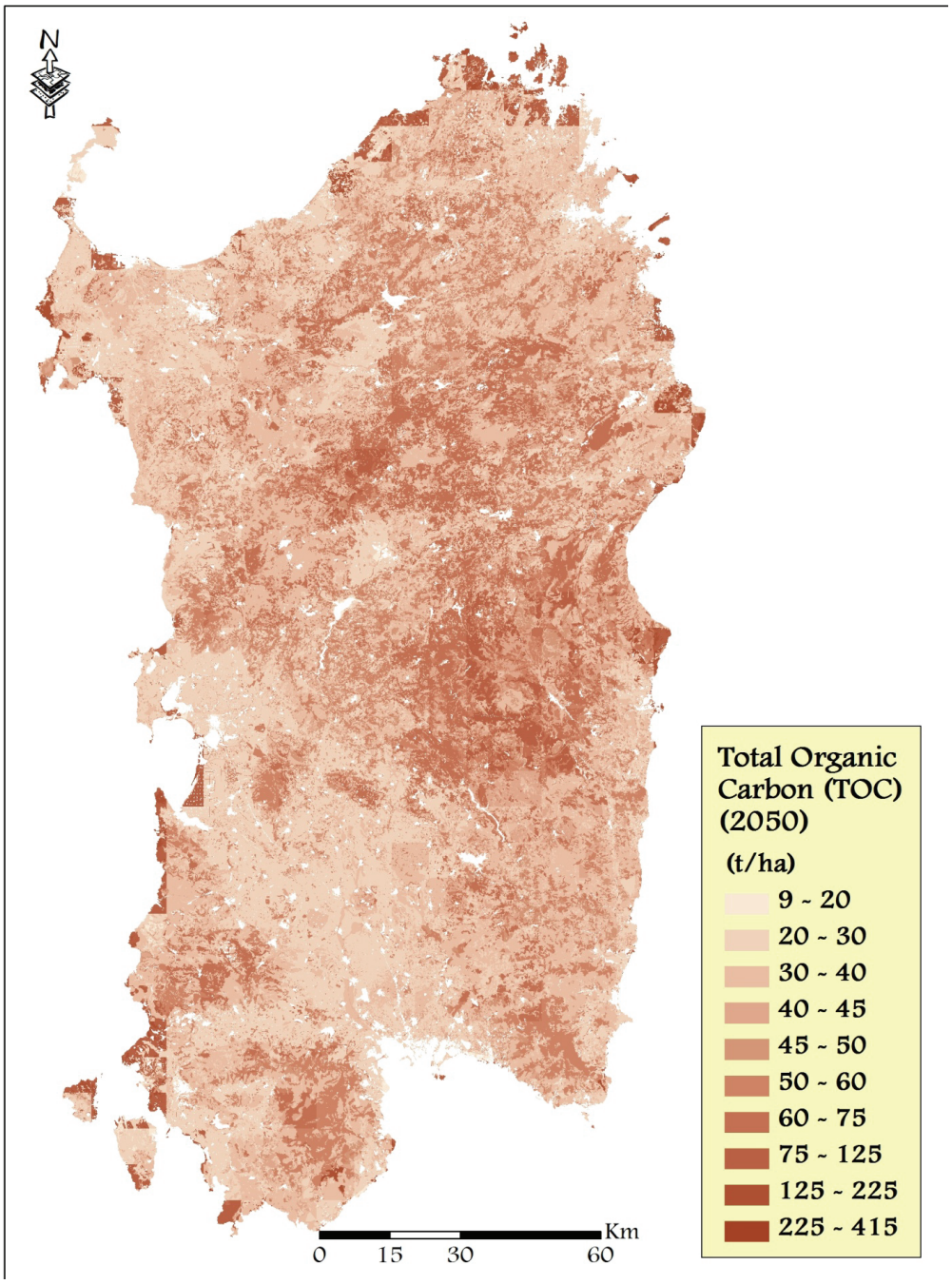

Figure 33c Map of total organic C (TOC) projection (2050).

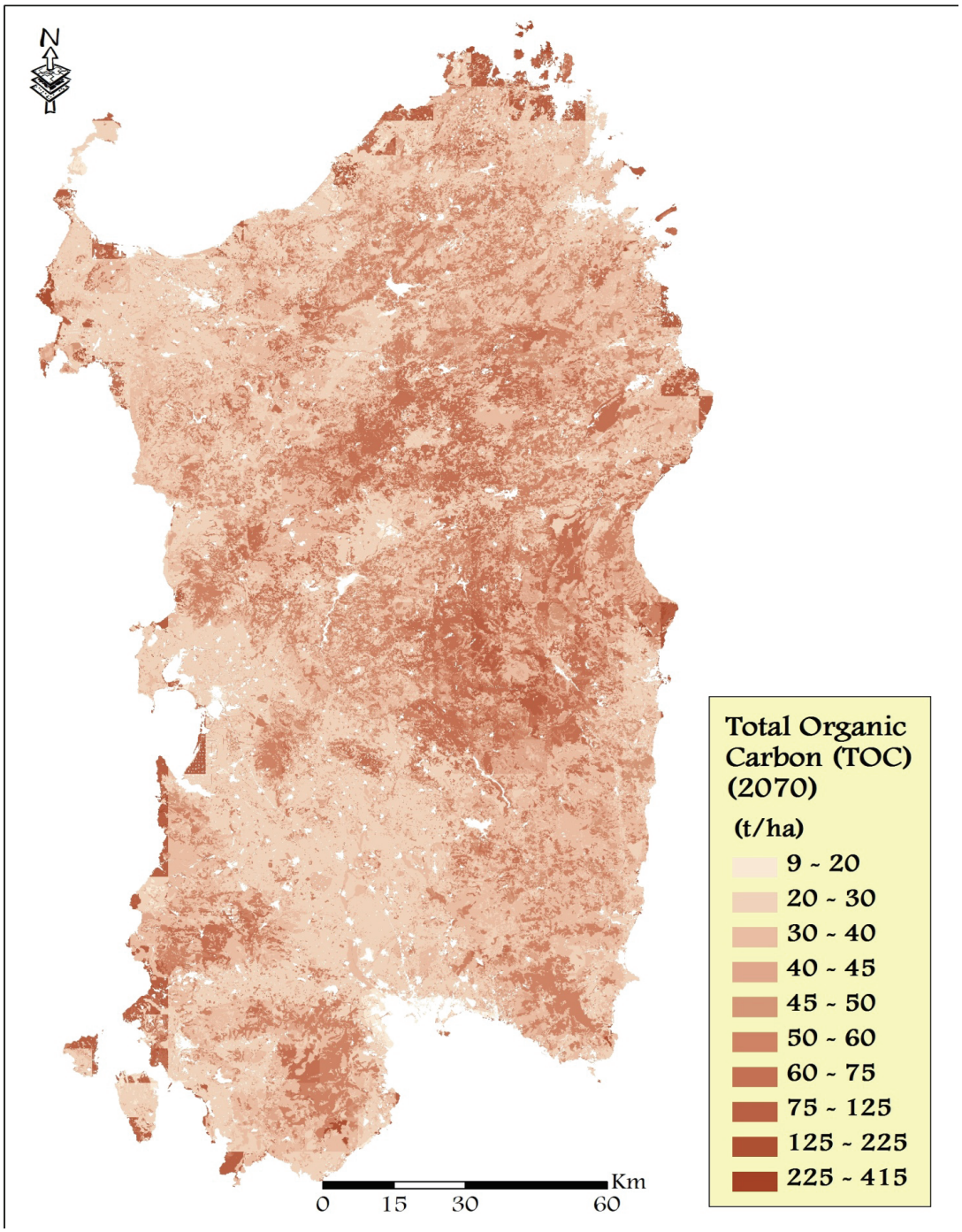

Figure 33d Map of total organic C (TOC) projection (2070).

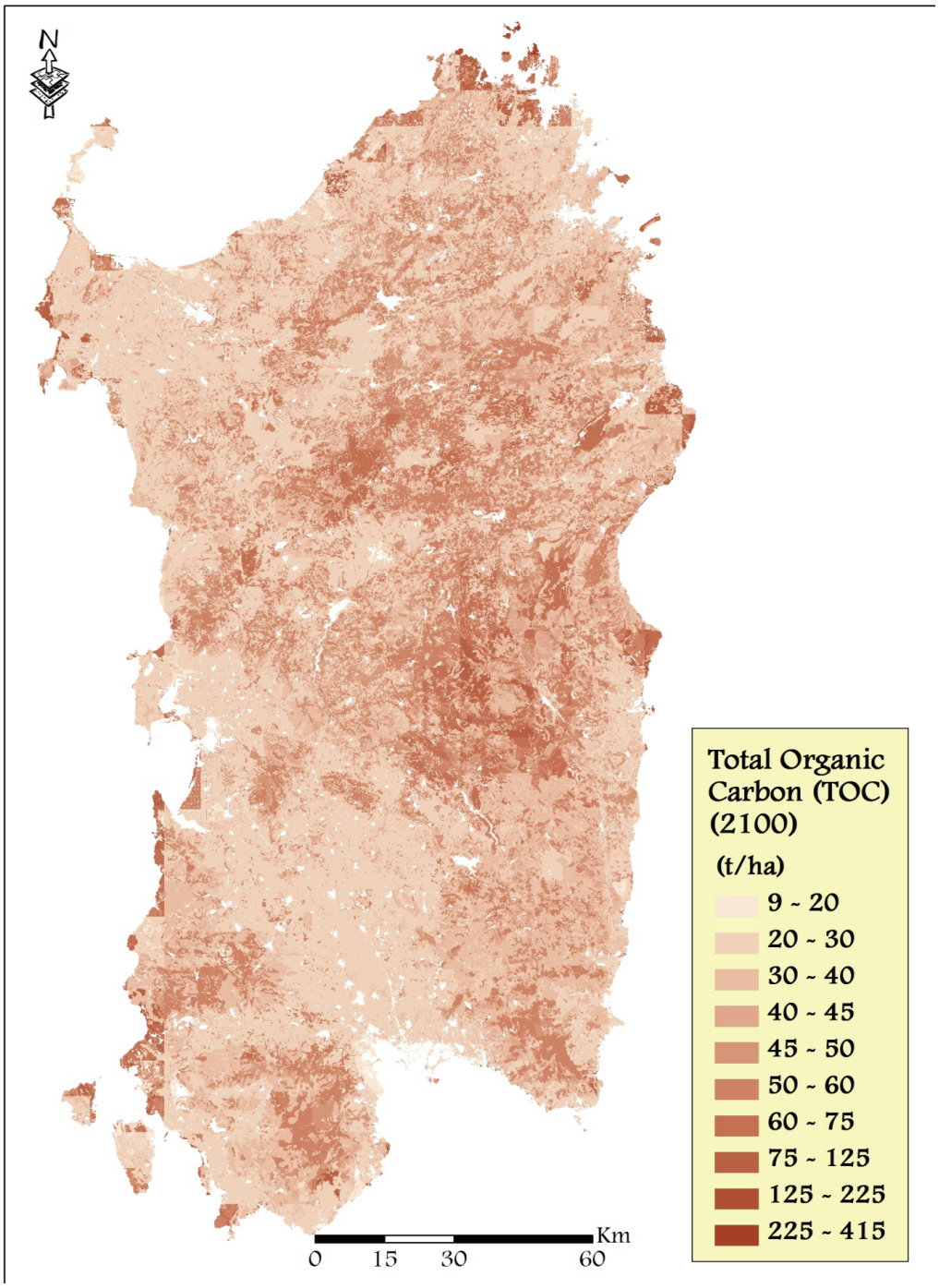

Figure 33e Map of total organic C (TOC) projection (2100).

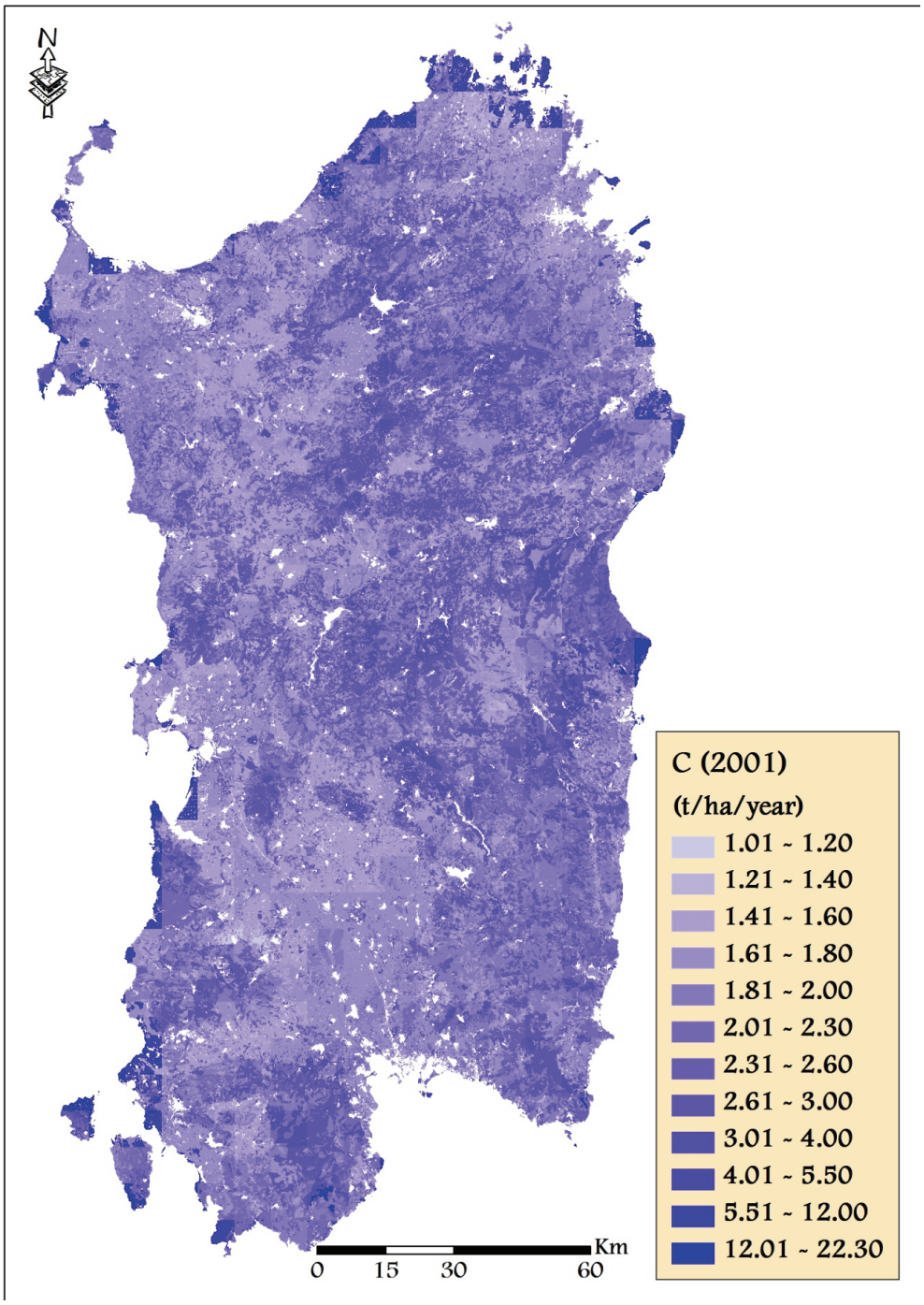

Figure 34a Map of CO_2 pool projection (2100).

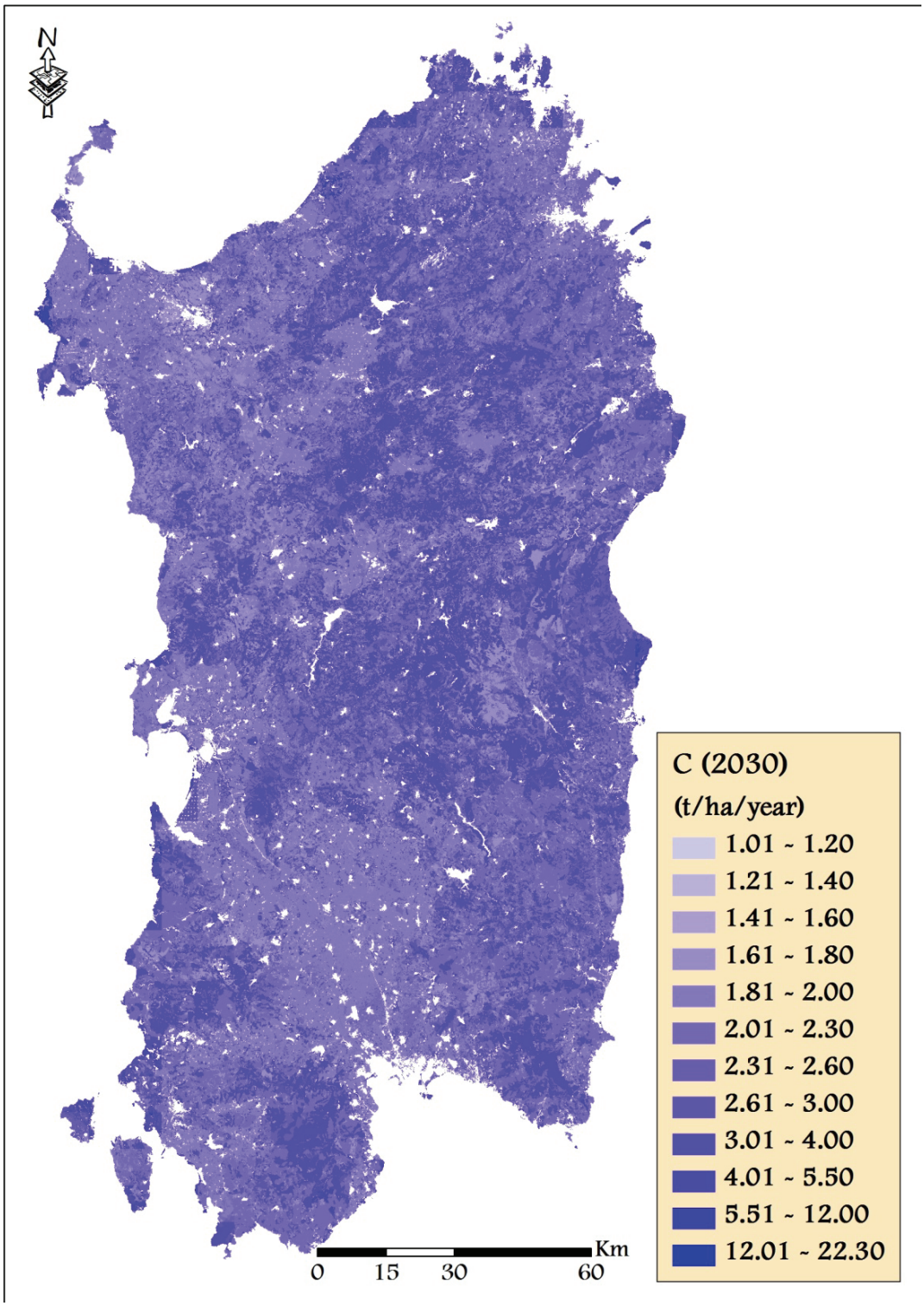

Figure 34b Map of CO_2 pool projection (2100).

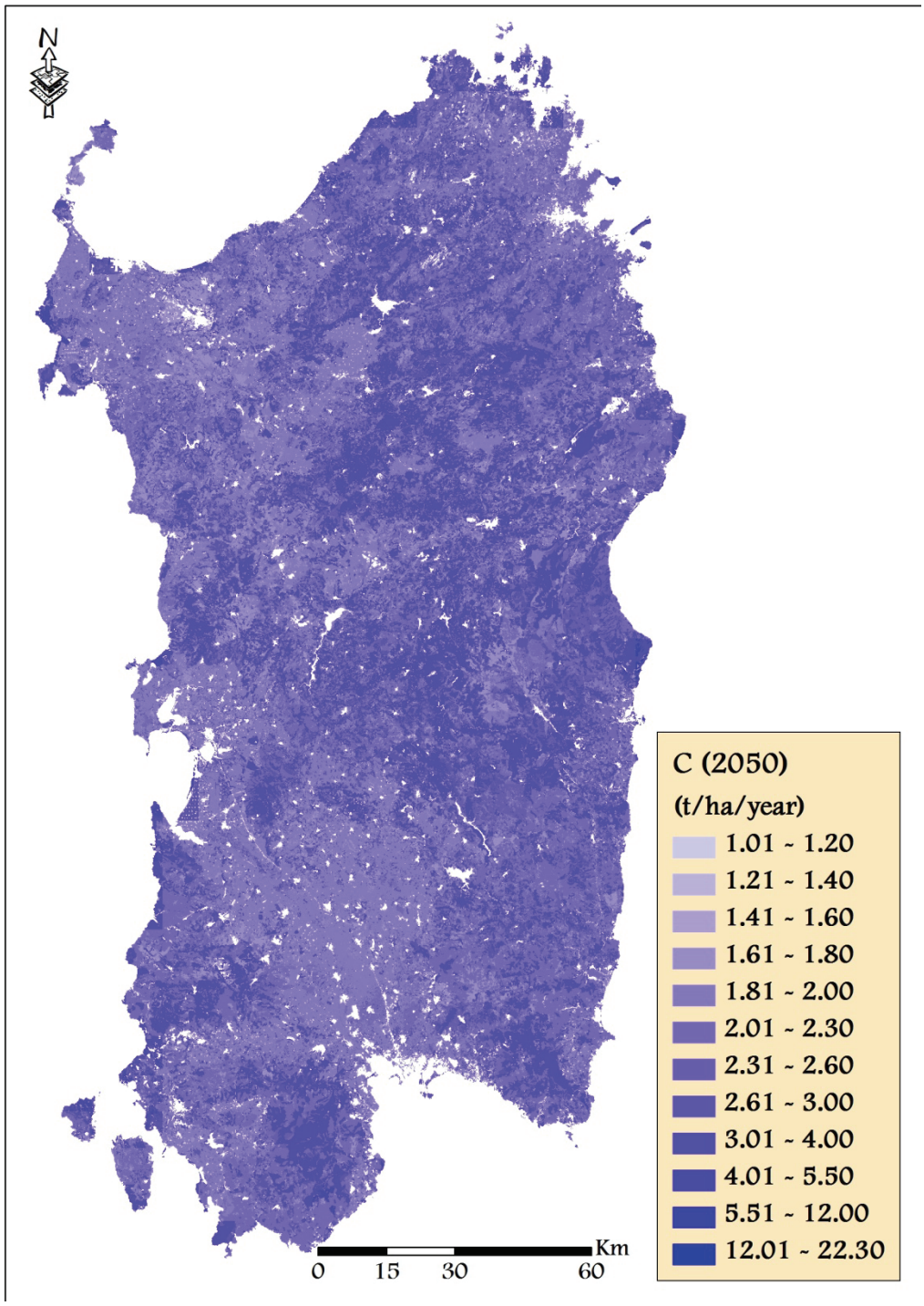

Figure 34c Map of CO_2 pool projection (2100).

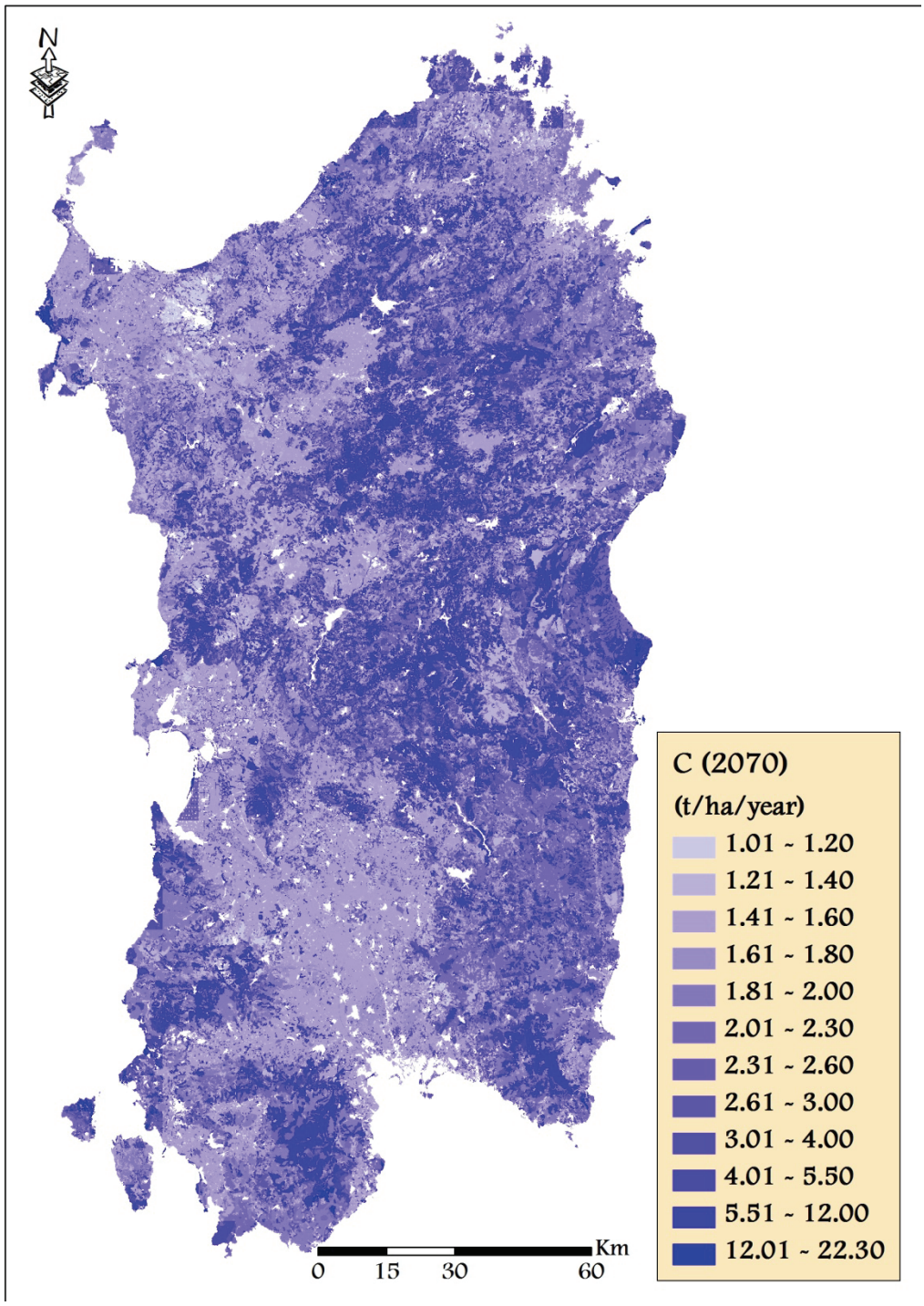

Figure 34d Map of CO_2 pool projection (2100).

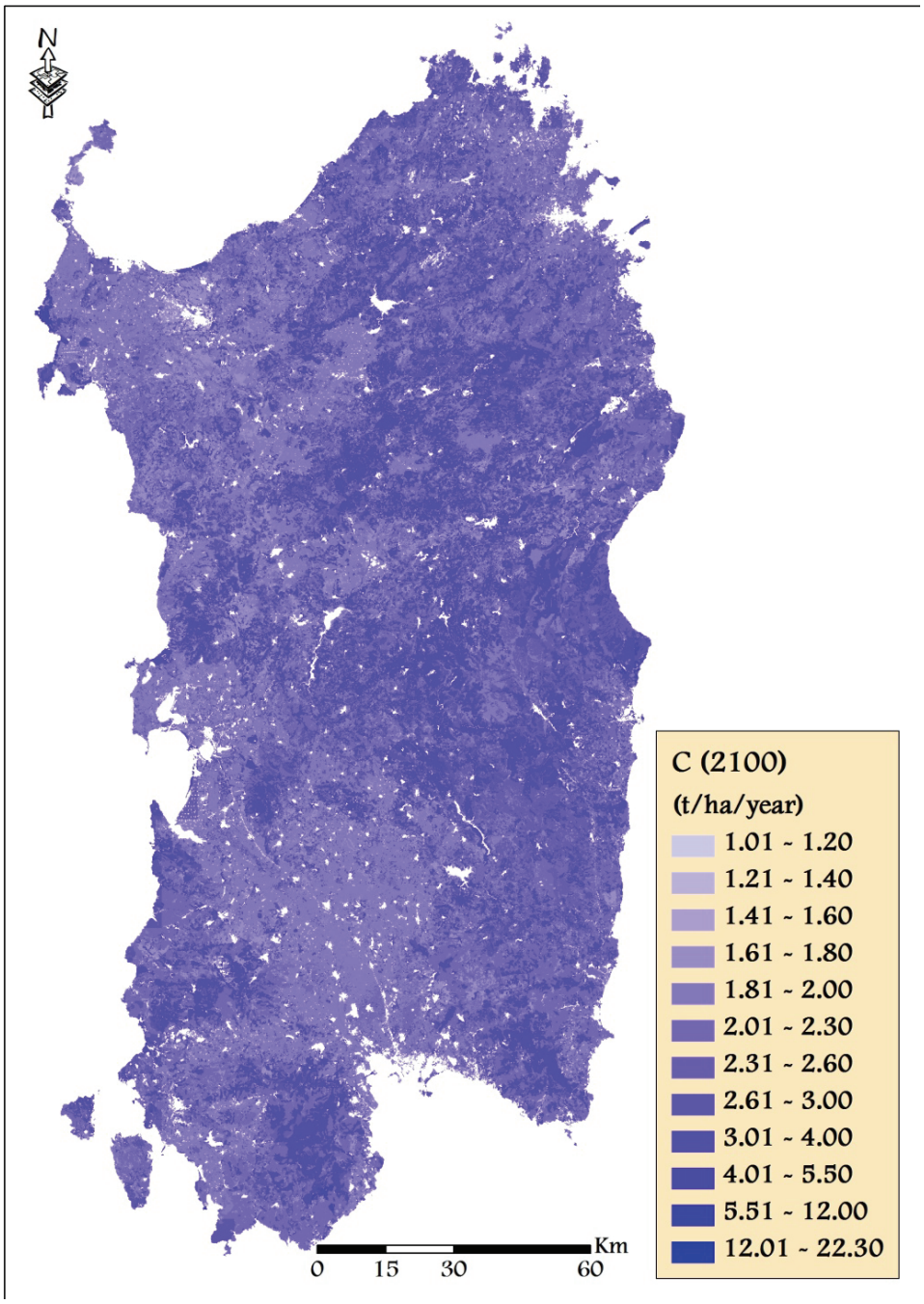

Figure 34e Map of CO_2 pool projection (2100).

The relative variation of each compartment during the 21th century was different (Table 18). The reduction for the whole organic C pool (SUM) in the year 2100 is estimated to be the 18.4% with respect to the year 2000. The pool that showed the highest C loss in the period 2001-2100 is the DPM (-33.7%), while the HUM pool exhibited the lowest reduction (-14.1%). Complexively, the total C loss estimated in the 2090-2100 period with respect to the year 2000 is equal to 19,165 ktons C.

The Figures 35a-b-c-d-e shows the trend of each C compartment for the period 2001-2100.

The variations of each compartment were different even within each LU class. Figure 36 shows how the total C pool varies in each LU class. With respect to the total organic C, the loss of C was a little less in the agricultural areas than in the natural areas. The Table 20 reports the variations of C in each compartment for the respective LU class.

With respect to the evolved CO_2, the quantities rising in the period 2001-2100. At the end of the 21th century the soils of Sardinia release +7.1% of the CO_2 than the year 2000. The trend of the CO_2 fluxes variation is showed in the Figure 37. The Figure 38 and the table 21 shows the variations, at the end of the 21th century, of the CO_2 fluxes

Table 18. Relative variation of the organic C pools and the evolved CO_2 during the 21th century with A1b climatic scenario. Data area averaged for the respective last decade.

	2010	2030	2050	2070	2100
	%	%	%	%	%
BIO	-0.02	-8.50	-17.93	-22.07	-29.49
DPM	-5.16	-12.72	-21.39	-28.94	-33.74
HUM	2.73	0.80	-2.66	-6.70	-14.12
RPM	-0.32	-10.11	-20.77	-25.12	-32.48
SUM	1.99	-1.77	-6.91	-11.03	-18.43
CO_2	4.31	3.77	4.96	6.50	4.29

Table 19. C pools projections for the 21th century in Sardinia. Data (excluded the 2000) are averaged for the respective last decade.

	2000	2010	2030	2050	2070	2100
			C (t*1000)			
BIO	2111.6	2111.2	1932.2	1733.0	1645.7	1488.8
DPM	357.3	338.9	311.9	280.9	253.9	236.8
HUM	79261.5	81425.0	79895.0	77153.6	73953.9	68067.1
RPM	22248.7	22178.4	19998.8	17628.4	16660.5	15022.1
SUM	103979.1	106053.4	102137.9	96795.8	92513.9	84814.7
CO_2	5404.7	5637.9	5608.6	5672.7	5756.0	5636.8

Table 20. Variations of C pools in the 21th century. Data area averaged for the respective last decade.

	2000	2010	2030	2050	2070	2100
			C (t*1000)			
BIO	0	-0.4	-179.4	-378.6	-465.9	-622.8
DPM	0	-18.4	-45.4	-76.4	-103.4	-120.6
HUM	0	2163.5	633.5	-2107.9	-5307.6	-11194.4
RPM	0	-70.3	-2249.9	-4620.3	-5588.3	-7226.7
SUM	0	2074.3	-1841.2	-7183.3	-11465.2	-19164.5
CO_2	0	233.1	203.9	268.0	351.2	232.1

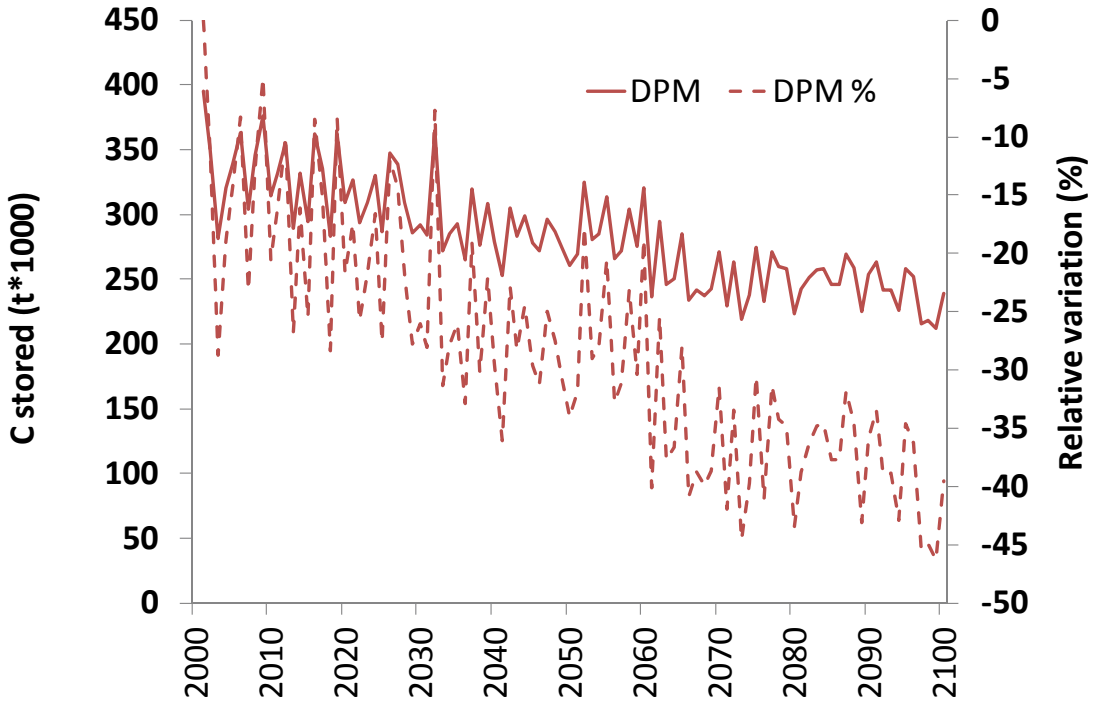

Figure 35a. Absolute and relative variation of the DPM during the 21th century.

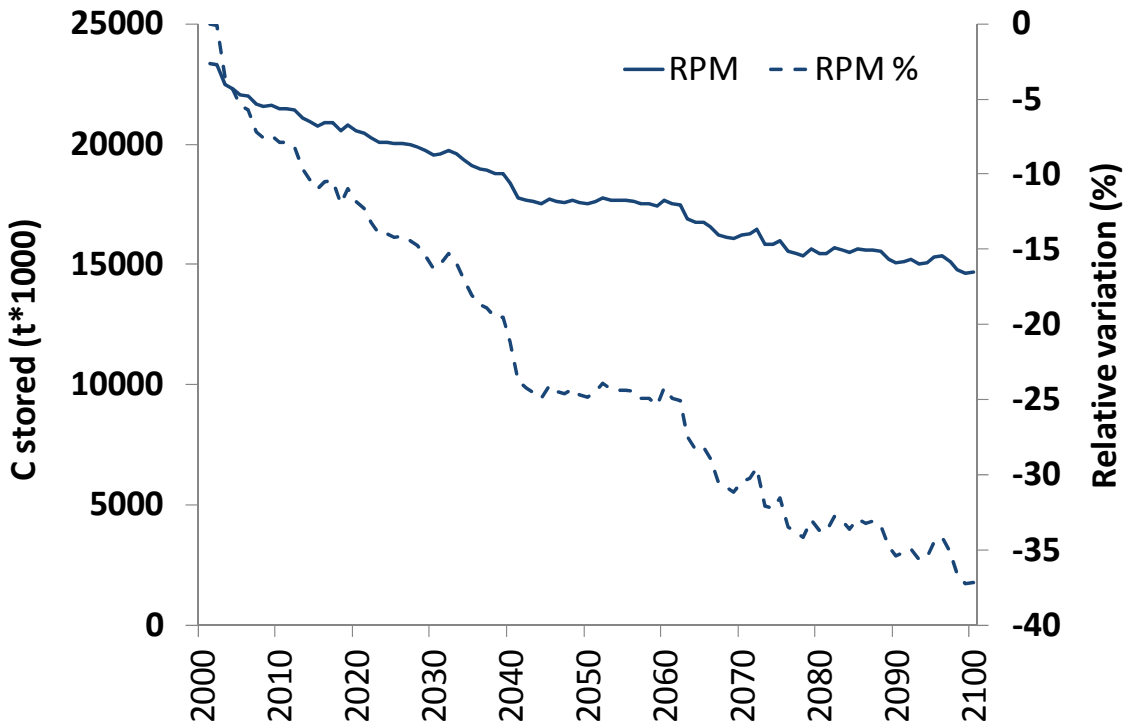

Figure 35b. Absolute and relative variation of the RPM during the 21th century.

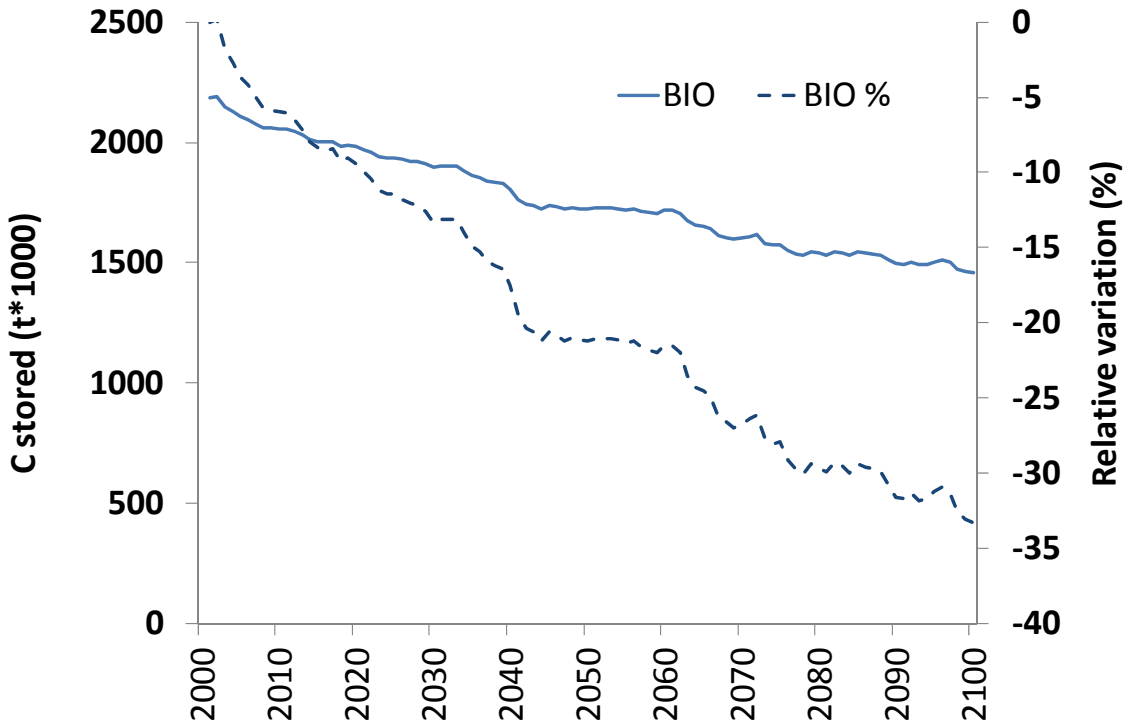

Figure 35c. Absolute and relative variation of the BIO during the 21th century.

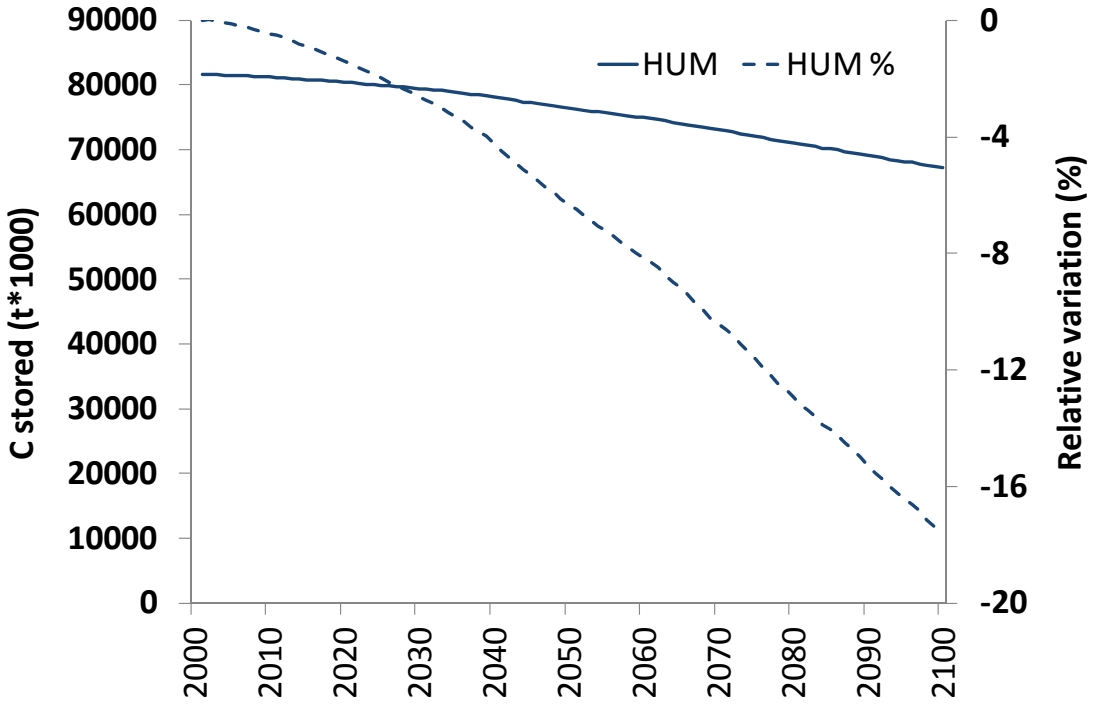

Figure 35d. Absolute and relative variation of the HUM during the 21th century.

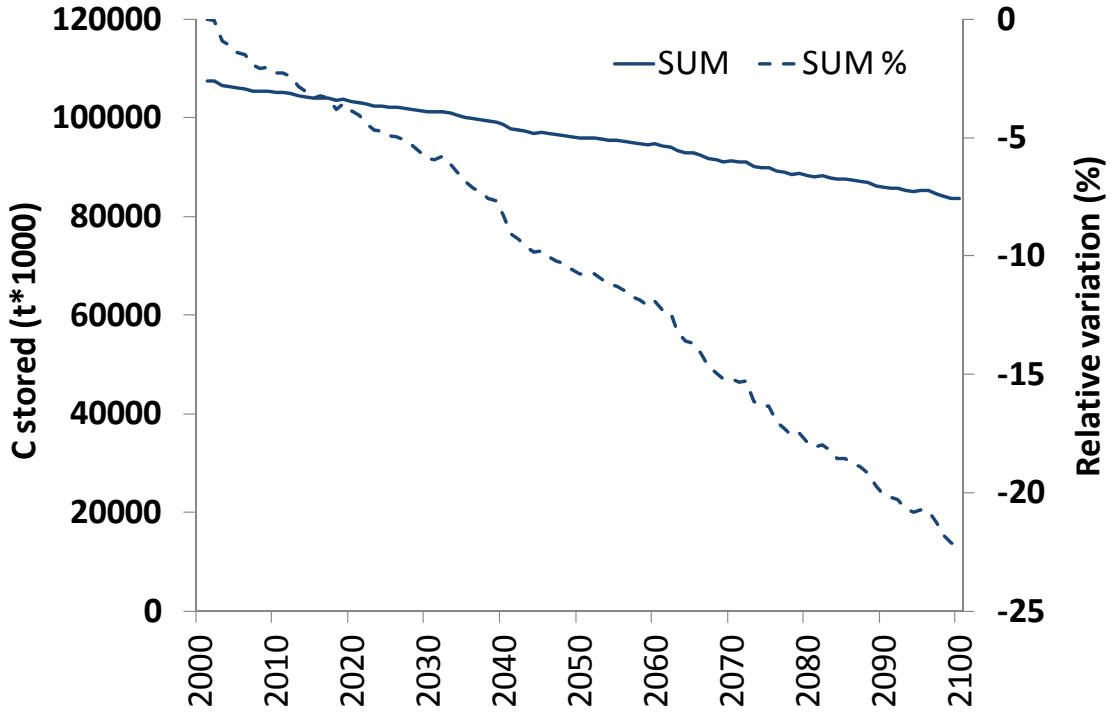

Figure 35e. Absolute and relative variation of the total organic C (SUM) during the 21th century.

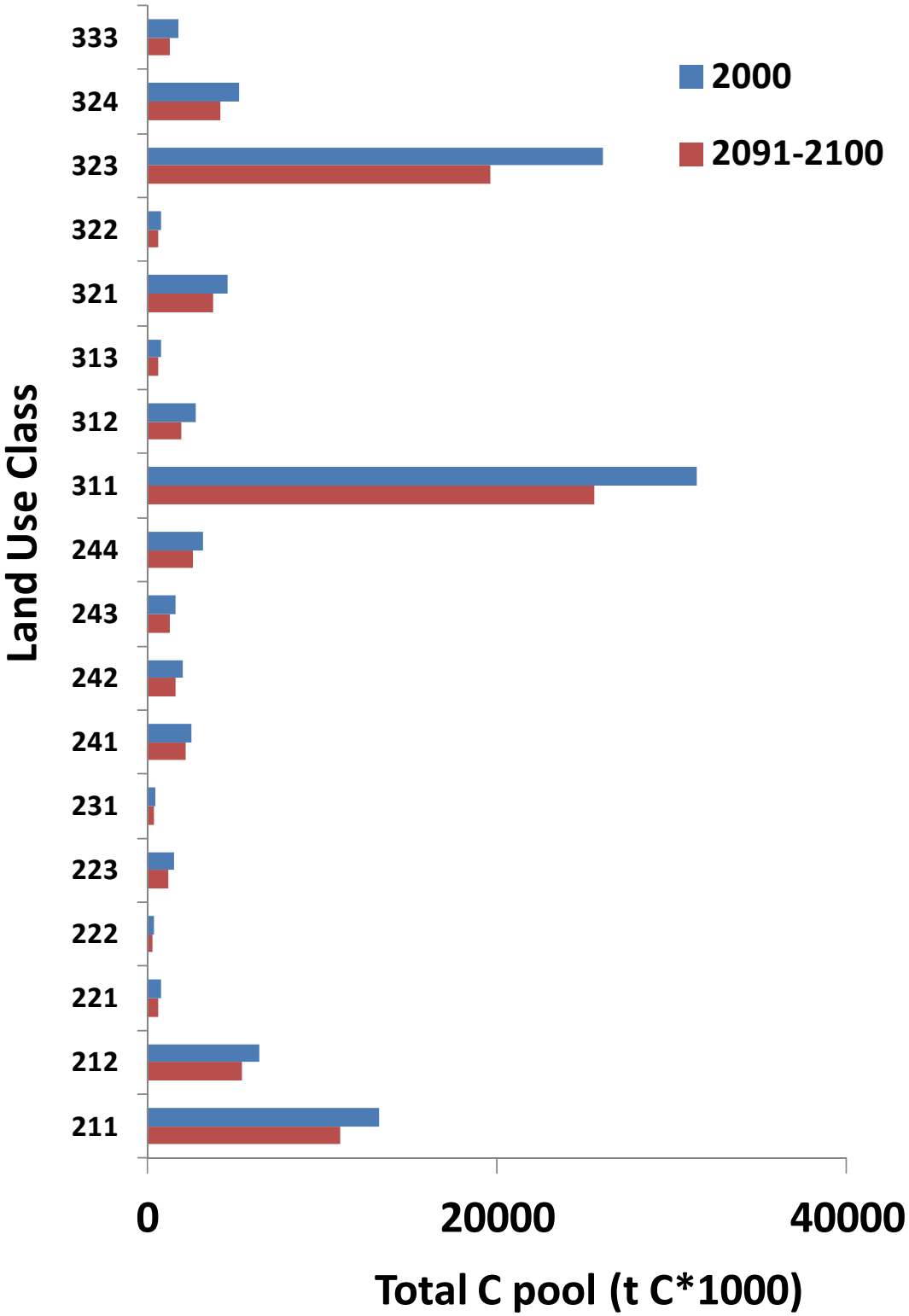

Figure 36. Variations of the total C pool in the LU classes during the 21th century.

Table 21. Variations (period 2091-2100 vs year 2000) of the organic C pools for each Land Use class.

	DPM %	RPM %	BIO %	HUM %	SUM %
211	-25.69	-31.15	-28.49	-14.32	-17.24
212	-30.04	-29.82	-27.22	-14.10	-16.90
221	-16.93	-28.99	-28.05	-16.76	-20.33
222	-18.43	-20.51	-18.67	-8.61	-12.15
223	-86.55	-34.04	-29.79	-14.62	-19.77
231	-28.99	-35.25	-32.22	-20.35	-22.86
241	-20.94	-25.99	-23.57	-10.25	-12.94
242	-35.34	-38.65	-36.00	-21.73	-24.77
243	-29.51	-31.42	-28.41	-14.68	-18.52
244	-24.21	-27.95	-25.31	-11.49	-15.21
311	-31.08	-28.32	-25.22	-10.22	-15.25
312	-47.17	-45.53	-42.01	-23.25	-29.73
313	-37.53	-34.98	-31.75	-15.47	-21.08
321	-30.70	-30.00	-26.97	-11.96	-15.08
322	-41.65	-31.73	-28.25	-12.86	-17.29
323	-49.33	-38.40	-34.92	-17.82	-22.67
324	-38.30	-33.14	-29.85	-13.91	-19.31
333	-41.06	-37.38	-33.71	-13.10	-17.64
211	-33.74	-32.48	-29.49	-14.12	-18.43
avg	-33.74	-32.48	-29.49	-14.12	-18.43
Agricultural areas	-27.19	-30.63	-28.07	-14.32	-17.44
Natural areas	-41.28	-33.09	-30.20	-14.03	-18.88

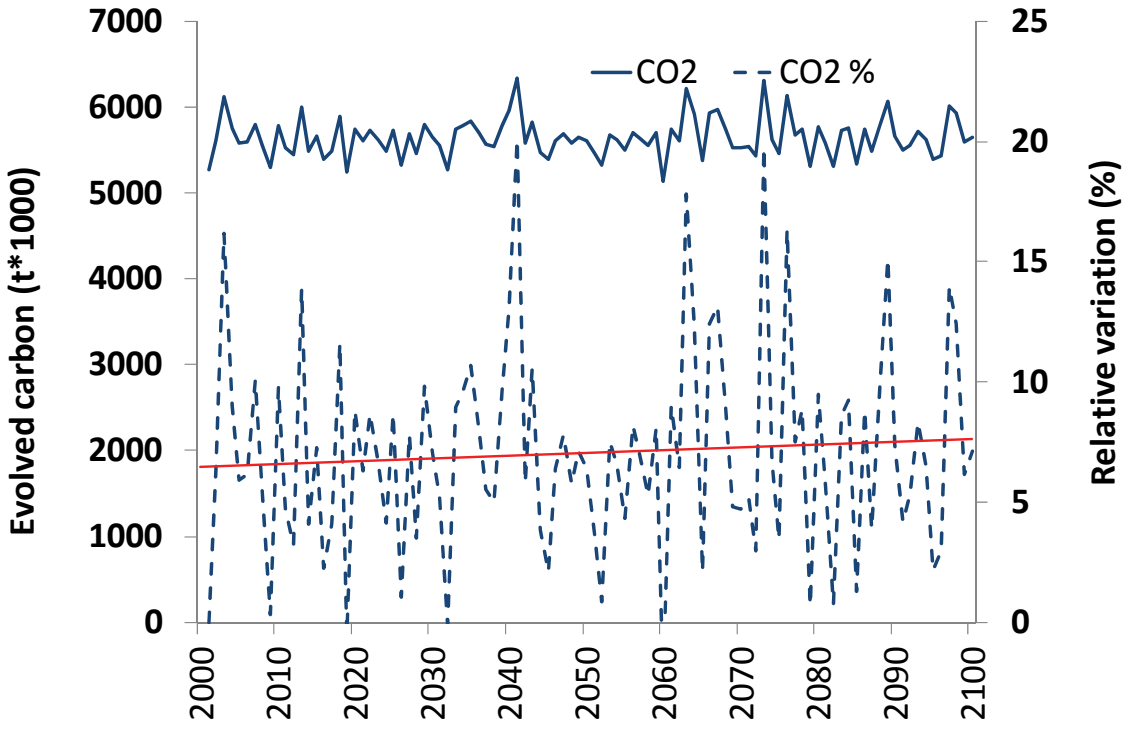

Figure 37. Absolute and relative variation of the evolved CO_2 during the 21th century.

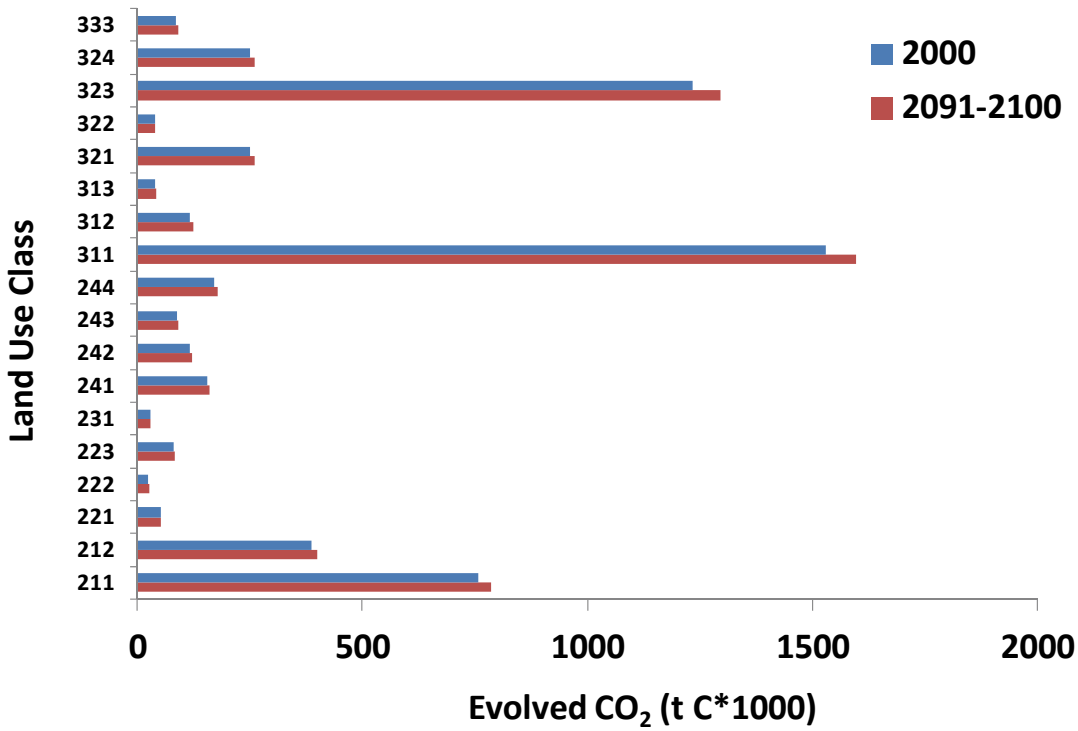

Figure 38. Variations of the CO_2 fluxes during the 21th century.

4. Model sensitivity analysis

Good practice in model application requires knowledge of the of the model to changes in input variables over a range of conditions. For univariate testing, the model was run with a fixed (default) set of variables, except for the particular single variable that was allowed to vary.

The range of variation depended somewhat on the nature of the particular variables.

The variables considered are: the clay content, the monthly plant residual input and the climate variables.

Percent clay in the soil

Table 22 Variation of C (t ha^{-1}) due to change clay content in the soil for each UDS class

Clay %	211	221	223	231	311	312	321	323
5	-11,3	-8,7	-9,3	-15,7	-18	-15,8	-10,1	-13,2
15	-5,3	-4	-4,3	-7,2	-8,3	-7,1	-4,6	-6,1
30	0	0	0	0	0	0	0	0
45	1,9	1,5	1,5	2,7	3,1	2,7	1,8	2,2
60	2	1,6	1,6	2,9	3,2	2,8	1,9	2,3
75	2,4	1,9	1,9	3,4	3,8	3,3	2,2	2,8
90	2,8	2,2	2,4	4	4,6	4	2,6	3,3

For univariate analysis, the initial 30% clay value was varied from 5% to 90%. It was considered that these values covered the expected range of the variable.

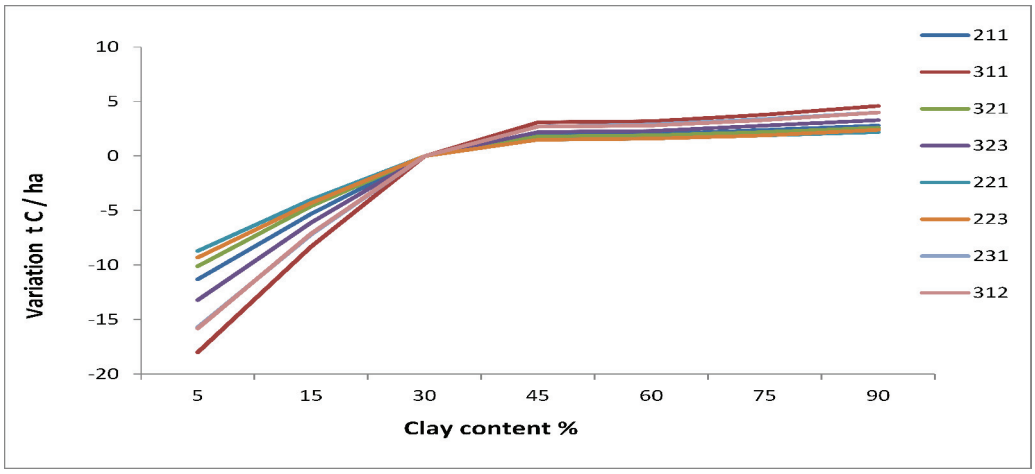

Figure 39. Variation of C (t ha^{-1}) due to change clay content in the soil.

Low values of clay (around 5-30%) affect substantially negatively the C soil content (-20/-5 t C ha^{-1}), while if the clay content increases the C content varies little. All 8 UDS classes have roughly the same trend but the class 312 reacts slightly more sensitive to the clay variation (*tab.22 and fig 39*).

Monthly plant residual

The monthly of plant residual is the variable that most influences the C soil content. All 8 UDS classes have roughly the same trend, with an increase/decrease of 2,5 t C ha^{-1} about for 0,01 t C ha^{-1} of input variation *(tab.23 and fig 40)* .

Table 23 Variation of C (t ha^{-1}) due to plant residual change.

Plant Residual inputs (t C/ha	211	221	223	231	311	312	321	323
>0,03	7,3	6,5	8,2	7,3	8,2	8,2	7,3	7,8
>0,01	2,4	2,2	2,7	2,4	2,8	2,7	2,4	2,6
0	0	0	0	0	0	0	0	0
<0,01	-2,4	-2,2	-3	-2	-2,8	-3	-2,4	-2,6
<0,03	-7,3	-6,6	-8,3	-7,3	-8,2	-8,2	-7,3	-7,8

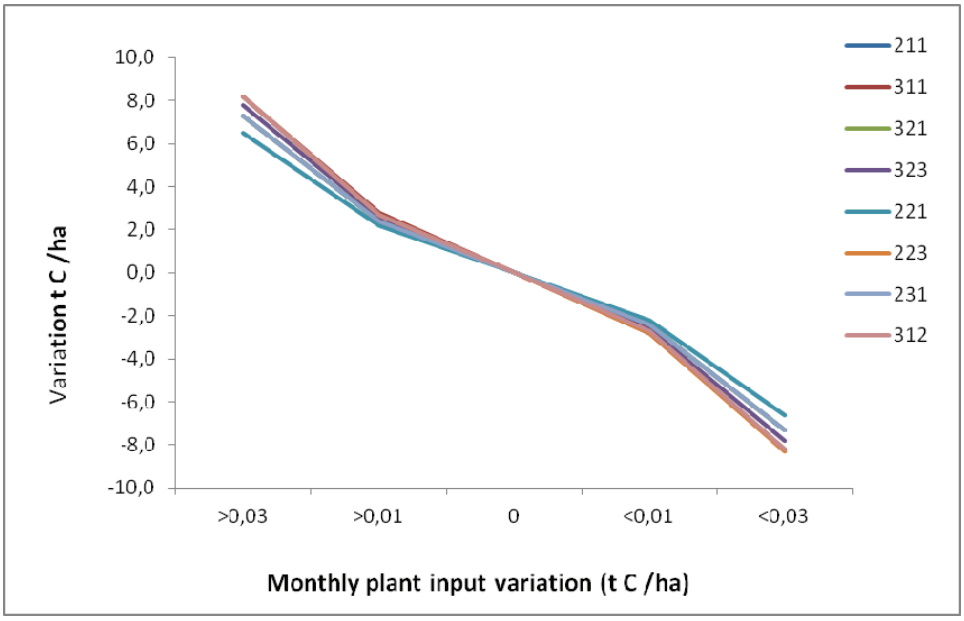

Figure 40. Variation of C (t ha^{-1}) due to plant residual change.

Climate variables

The observed average monthly pan evaporation had no effect on the estimate of soil C. A small (-0,9/-2,3 t C ha^{-1}) response was detected (fig 20) at -15 mm $month^{-1}$ pan evaporation *(tab.24 and fig 41)*

Table 24 Variation of C (t ha^{-1}) due to ETP change.

ETP (mm)	211	221	223	231	311	312	321	323
>15	0	0	0	0	0	0	0	0
>5	0	0	0	0	0	0	0	0
0	0	0	0	0	0	0	0	0
<5	0	0	0	0	0	0	0	0
<15	-1,3	-0,9	-1,2	-1,8	-2,3	-2	-1,1	-1,6

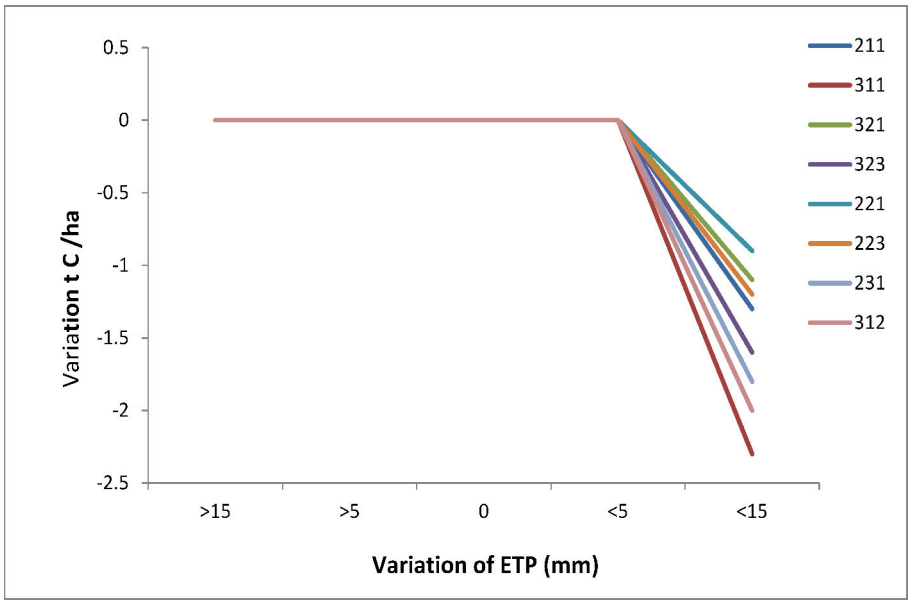

Figure 41 Variation of C (t ha^{-1}) due to ETP change.

Also the temperature strongly influences the C soil variation. An increase in one degree celsius (°C) would decrease the amount of soil C by about 4 t C ha^{-1} until 8 t C ha^{-1} (*tab.25 and fig 42*). Temperature may be a critical variable because large changes in temperature can occur within the same region

Table 25 Variation of C (t ha-1) due to temperature change

TEMP (°C)	211	221	223	231	311	312	321	323
>2,5	-9,3	-8,7	-8,8	-13,2	-17	-15	-8,3	-11,6
>1	-4,3	-3,9	-4	-5,9	-7,7	-6,7	-3,7	-5,3
0	0	0	0	0	0	0	0	0
<1	5	4,6	4,7	7	9,1	8	4,5	6,2
<2,5	14,4	13,6	13,5	20,3	26,3	23,1	13	17,9

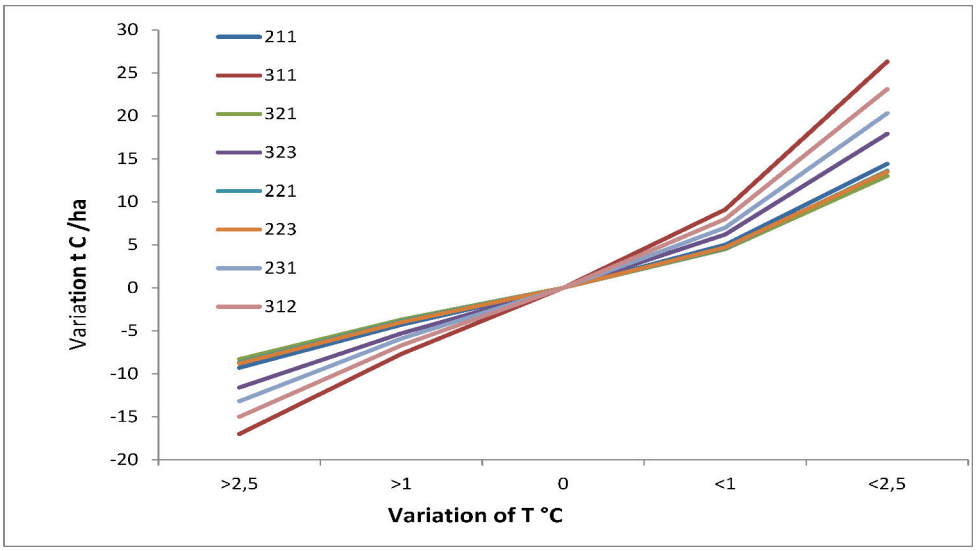

Fig. 42 Variation of C (t ha^{-1}) due to temperature change.

A rain increase of 15 mm per month had a significant effect, decreasing from 1,6 to about -4 t C ha^{-1}. However, it would be expected that increased monthly rainfall would correlate with increased biomass production, there by lessening the real observed impact (*tab.26 and fig 43*).

Table 26 Variation of C (t ha^{-1}) due to rain change.

RAIN (mm)	211	221	223	231	311	312	321	323
>15	-2,3	-1,6	-2,1	-3,1	-4	-3,5	-1,9	-2,7
>5	0	0	0	0	0	0	0	0
0	0	0	0	0	0	0	0	0
<5	0	0	0	0	0	0	0	0
<15	0	0	0	0	0	0	0	0

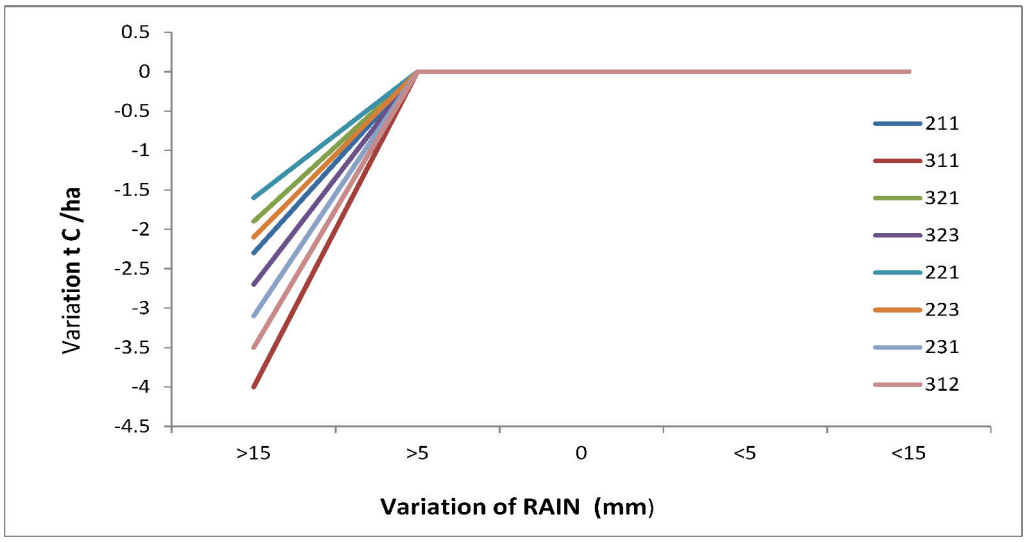

Fig. 43 Variation of C (t ha^{-1}) due to rain change.

Further complications can arise due to interactions between variables. For example, climatic conditions and soil type may have a profound effect on crop growth, which in turn affects the amount of C entering the soil. There is often also a correlation between temperature, rainfall and evaporation characteristics affecting the moisture content of the soil, with an additional correlation with clay content (Janik et al., 2002). Although it is acknowledged that these correlations may have significant influence on the calculated sensitivities, they are too complex to address in this work.

Conclusions

In this preliminary study, an analysis of organic carbon pools in a representative Mediterranean area was carried out. A detailed GIS database, coupled with climate data and a dynamic simulation model was used to estimate the regional SOC stocks and the sequestration potential.

The results show the total organic C amount in the soil is estimated as about 104,000,000 t C (45 t C ha^{-1}, on average), and the CO_2 released is 5479.1 kt C (2.39 t C ha^{-1}) on average. In general, the naturals areas contain an higher amount of organic carbon. Most of this carbon is stored as humified carbon. With respect to the evolved CO_2, natural areas release more C than agricultural areas (56.2% vs 43.8%, respectively).

The projections through the 21th century in a climate change scenario (A1b) showed a general reduction of the organic carbon pool in the soil (-18.4% in the 2100). The C pool decrement rate was different between the compartments and, on average, the natural areas lose more C than the agricultural areas.

The rate of the released CO_2 from soil increases in the natural areas. On average, the annual CO_2 flux at the end of the 21th century increased of 4.3%.

This work represents a preliminary exercise that shows how climate change can affect the organic carbon decomposition in the soil of Mediterranean areas. But other processes, coupled to climate change, first of all the Land Use change, and the changes in the annual inputs of plant debris (changing in a context of climatic change) should be accounted in the next steps to give more realistic estimations of C pool dynamics.

Acknowledgments

This research work has been possible thanks to the contribution of AGRIS (Agenzia della Regione Sardegna per la ricerca scientifica, la sperimentazione e l'innovazione tecnologica nei settori agricolo, agroindustriale e forestale).

I wish to thank Professor Sandro Dettori, Chair of the Department of Economics and Woody Plant Ecosystems, University of Sassari.

I'm grateful to my supervisors Professor Donatella Spano and Dr Costantino Sirca for having me offered chances for professional growth during the course of my PhD studies and for having supported every step of this work.

I want to thank to Dr. Kevin Coleman (Rothamsted Research) who developed the RothC model and advised and directed me to model use.

I can't forget to thanks Dr. Fulvio Trebini for helping me with the ground data collection from theses.

A sincere thank to Dr. Michele Salis for the great help in the GIS elaboration, and to Dr. Valentina Bacciu for the assistance of reading my draft ad giving me several advices.

I want to give special thanks to all my friends and collegues from the PhD program, especially Dr AnnaPaola Chergia he was a constant supporter during these years and Dr. Stefania Pisanu, who joined and helped me in the field surveys. A sincere thank goes to Dr.Pasquangela Muresu, Dr Ana Sofia Oliveira, Dr. Valentina Mereu, Dr. Serena Marras, Dr Noemi Mancosu, Dr. Matteo Cabras, Dr Giuseppe Dettori and Dr. Francesco Masala,, I am grateful to my friends Domenica , Rosanna, uncle Gino and aunt Domenica for the constant support.

Finally, and must important, all my gratitude is for my husband Angelo and my family for having encouraged and supported me especially in difficult times.

References

Aalde H., Gonzalez P., Gytarsky M., (2006) *Generic methodologies applicable to multiple land use categories* IPCC 2006 Guidelines for National Greenhouse Gas Inventories.

Al-Adamat R.,. Rawajfih Z , Easter M, Paustian K., Coleman K, Milne E, Falloon P, Powlson D.S, Batjes N.H (2007) *Predicted soil organic C stocks and changes in Jordan between 2000 and 2030 made using the GEFSOC Modelling System* Agriculture, Ecosystems and Environment **122,** 35–45

Ågren G. I., Bosatta E. (1996) *Theoretical ecosystem ecology. Understanding element cycles*, Cambridge University Press.

Arianoutsou M.(1989) *Timing of litter production in a maquis ecosystem of North-Eastern Greece* Acta ecologica **10** n 4 p 371-378

Bahn M., Knapp M., Garajova Z., Pfahringer N., Cernusca A. (2006) *Root respiration in temperate mountain grasslands differing in land use*. Global Change Biology, **12**, 995-1006.

Baldocchi D. , (2003) *Assessing the Eddy Covariance technique for evaluating C dioxide exchange rates of ecosystems: past, present and future* .Global change biology **9**, 1-14

Batjes, N. H.(1996) *Total C and nitrogen in the soil of the world*. European Journal of Soil Science, **47**, 151-163.

Bellamy P. H, Loveland P. J., Bradley R. I., Lark M. and Kirk J. D. (2005) *C losses from all soils across England and Wales 1978–2003* R. Nature, **437**, 245-248

Bertrand I., Chabbert B., Kurek B., Recous S. (2006) *Can the biochemical features and histology of wheat residues explain their decomposition in soil?* Plant and Soil, **281**,291–307

Bisutti I., Hilke I., Raessler M. (2004) *Determination of total organic C – an overview of current methods* Elsevier Trends in Analytical Chemistry, Vol. **23**, No. 10–11,

Bond-Lamberty B., Wang C. K., Gower S. T. (2004) *A global relationship between the heterotrophic and autotrophic components of soil respiration?* Global Change Biology, **10**, 1756-1766.

Bosatta E., Agren G&AN (1996)*Theoretical analyses of C and nutrient dynamics in soil profiles* Soil Biol.Biochem. Vol. **28**, pp. 1523-153

Bhattacharyya A, Pal D.K., Easter M,. Batjes N.H, Milne E., Powlson D.S., (2007) *Modelled soil organic C stocks and changes in the Indo-Gangetic Plains, India from 1980 to 2030* Agriculture, Ecosystems and Environment **122**, 84–94

Briones M., Ineson P., (1996) *Decomposition of eucalyptus leaves in litter mixtures.* Soil boil. Biochem. Vol 28, 1381-1388

Di Blasi C., Tanzi V., Lanzetta M., (1997) *A study of the agricultural residues in Italy,* Biomass and Bioenergy vol 12 n 5 pp 321-33112

Brown T. (2006) *RothC UK a dynamic modeling system for estimating changes in soil C from mineral soils at 1-km resolution in the UK* Soil Use and Management, **22**, 274–288

Burton A. J., Pregitzer K. S., Zogg G. P., Zak, D. R. (1998) *Drought reduces root respiration in sugar maple forests*. Ecological Applications, **8**, 771-778.

Calfapietra C., Gielen B.,, Galema A., Lukac M., De angelis P., Moscatelli M., Ceulemans R., Scarascia-Mugnozza G., (2003) *Free-air CO2 enrichment (FACE) enhances biomass production in a short-rotation poplar plantation* Tree Physiology **23**, 805–814

Camarda i., Valsecchi F. (1992) L*a flora della Sardegna Piccoli arbusti liane e suffrutici spontanei della Sardegna* Università di Sassari Carlo Delfino editore

Caritat A., Bertoni G., Molinas M., Oliva M., Dominguez-Planella A (1996) *Litterfalll and mineral return in two cork-oak forest in Northeast Spain,* Elselvier **53**, 1049-1058

Cerri E.P.,. Easter M., Paustian K.,. Killian K.,. Coleman K., Cerri C. C (2007) *Simulating SOC changes in 11 land use change chronosequences from the Brazilian Amazon with RothC and Century model* Agriculture, Ecosystems and Environment **122** 46–57

Coleman K. and Jenkinson D. S., Crocher G., Richter D.(1997) *Simulating trends in soil organic C in long term experiment using RothC-26.3* Geoderma **81**, 29-44

Coleman K. and Jenkinson D. S. (1999) *RothC-26.3 A model for the turnover of C in soil* Rothamsted Research Harpenden.

Conant R. T., Smith G. R., Paustian K. (2003) *Spatial variability of soil C in forested and cultivated sites: Implications for change detection*. Journal of Environmental Quality, **32**, 278-286.

Conant R.T., Paustian K (2002) *Spatial variability of soil organic C in grasslands: implications for detecting change at different scale* Environmental Pollution, **116**, 127–135.

Conen F., Yakutin, M. V., Sambuu A. D. (2003) *Potential for detecting changes in soil organic C concentrations resulting from climate change*. Global Change Biology, **9**, 1515-1520.

Conen F., Zerva A., Arrouays D., Jolivet C. *et al* (2004) *The C balance of forest soils: detectability of changes in soil C stocks in temperate and Boreal* forests, The C Balance of Forest Biomes ch. 11,p. 233

Conference of the parties (COP4) *Report of the conference of the parties on its fourth session, held at buenos aires from 2 to 14 november* 1998 Framework convention on climate change.

Chertov O., Komarov A., (1996) *SOMM a model of soil organic matter and nitrogen dynamics in terrestrial ecosystems.* In *Evaluation of soil organic matter models using existing long term data sets* ed. Powlson P., Smith P., Smith J.,NATO ASI 138 Berlin., Springer pp 231-6

Chessa P.A., Delitala A., (1997). *Il Clima della Sardegna. Servizio AgrometeorologicoRegionale (SAR) per la Sardegna*, nota tecnica n° 2 http://www.sar.sardegna.it/pubblicazioni/notetecniche/nota2/index.asp.

Davidson E. A. and Janssens I. A (2006) *Temperature sensitivity of soil C decomposition and feedbacks to climate change* Nature Vol. **440**,9

Debano L. F. Conrad C. (1978) *The effect of fire on nutrients in a chaparral ecosystem* . Ecology **59**, 489-497.

De Dato G, Pellizzaro G, Cesaraccio C, Sirca C, De Angelis P, Duce P, Spano D, Scarascia Mugnozza (2008) *Effects of warmer and drier climate conditions on plant composition and biomass production in a Mediterranean shrubland community* Forest **1**: 39-48

Denman K. L., Brausser G., Chidthaisong A., (2007) *Couplings Between Changes in the Climate System and Biogeochemistry.* Climate change 2007 Cambridge University press.

Duguy A., Rovira P. B. and Vallejo R. (2007) *Land-use history and fire effects on soil fertility in eastern* Spain European Journal of Soil Science, **58**, 83–91

Ellert B.H Janzen H.H and McConkey B. G. (2001) *Measuring and comparing soil C storage*. Chapter 10: *Assessment methods for soil C* Lal R. CRC press 131-136.

Epron D., Farque L., Lucot E., Badot P. M. (1999) *Soil CO_2 efflux in a beech forest: dependence on soil temperature and soil water content*. Annals of Forest Science, **56**, 221-226.

Eriksson C. P, Holmgren P. (1996) *Estimating stone and boulder content in forest soils evaluating the potential of surface penetration Methods* Elsevier Science, Catena, **28**, 121-134.

European Commission (2010) *Soil- a key resource for the EU*. Soil Environment

Fahey T. , Siccama T. G. Driscoll G. E., *et al* (2005) *The biogeochemistry of C at Hubbard Brook* Biogeochemistry ,**75,**109–176.

Falloon P.D., Smith P., Smith J.U., Szabó J., Coleman K., Marshall S (1998) *Regional estimates of C sequestration potential: linking the Rothamsted C Model to GIS databases*Biol Fertil Soils **27** :236–241

Falloon P., Smith P. (2003) *Accounting for changes in soil C under the Kyoto Protocol: need for improved long-term data sets to reduce uncertainty in model projections* Soil Use and Management, **19**, 265-269.

Falloon P., Smith P., Bradley R. I., Milne R., Tomlinson R., Viner D., Livermore M. Fang, C., Moncrieff, J. B. (1999) *A model for soil CO_2 production and transport 1: Model development* Agricultural and Forest Meteorology, **95**, 225-236.

Falloon P., Smith P (2002) *Simulating SOC change in long-term experiments with RothC and Century: model evaluation for a regional scale application*. Soil use and management, **18**, 101-11

Falloon P. D., Jones C. D., Ades M., Paul K. (2008) *Soil moisture controls of future global soil C changes: a unconsidered source of uncertainty*. Global Biochemical cycle

FAO Conference Paper (2010) *Towards a Global Soil Partnership for Food Security and Climate Change Mitigation and Adaptation.*

Ferran A., Delitti W. Trabaud L. and. Vallejo V. R (2005) *Effects of fire recurrence in Quercus coccifera L. shrublands of the Valencia Region (Spain): I. plant composition and productivity* Plant Ecology **177**, 57–70

Fioretto A., Papa S. Fuggi A. (2003) *Litter-fall and litter decomposition in a low Mediterranean shrubland* Biol Fertil Soils **39**:37–44

Flechard C. R., Neftel A., Jocher M., Ammann C., Leifeld J., Fuhrer J. (2007) *Temporal changes in soil pore space CO_2 concentration and storage under permanent grassland*. Agricultural and Forest Meteorology, **142**, 66-84.

Franko U. (1996) Modeling approaches of soil organic matter turnover within the CANDY system. In Evaluation of soil organic matter models using existing long term data sets Powlson and Smith NATO ASI 138 Berlin: Springer pp 247-54

Freeman C., Fenner N., Ostle N. J., Kang H., Dowrick D. J., Hudson J. et al,(2004) *Export of dissolved organic C from peatlands under elevated C dioxide levels* Nature vol **430** 195-198

Freibauer A., Schulze D. E.(2005) *Environmental science : C unlocked from soil*. Nature, **437**, 205-206

Garnett M., Ineson P., Stevenson A., (2001) *Terrestrial organic C storage in a British moorland* Global Change Biology 7, 375-388

Garten C.T. (2004) *Soil C Dynamics Along an Elevation Gradient in the Southern Appalachian Mountains* Environmental Sciences Division

Garten C.T., Post W. M., Hanson P. J. Cooper L. (1999) *Forest soil C inventories dynamic along an elevation gradient in the southern Appalachian Mountains*. Biochemistry **45**, 115-145

Giardina C, Ryan M., (2000) *Evidence that decomposition rates of organic C in mineral soil do not vary with temperature* Nature, **404**, 858-61

Giardina C., Coleman M., Binkley D. et al (2005) *The response to of belowground C allocation in forest to global change* In Trees species effect on soils: implications for global change ch.**10** pp 120-154

Guo Y., Gong P., Amundson R., Qian Yu (2006) *Analysis of Factors Controlling Soil C in the Conterminous United States* Soil Science Society of America **70**:601–612

Grace P. R., Ladd J. N., (1995) *Socrates v. 2 user manual* PMB 2 Glen Osmond, Australia, Cooperative research centre for soil and land management

Gruijter J, Brus D., Bierkens M., and Knotters M. (2006) *Sampling for Natural Resource Monitoring* Math Geosci **40**, 349–351

Hanson P.J., Edwards N.T., Garten C., Andrews J.(2000) *Separating root and soil microbial contributions to soil respiration: A review of methods and observations* Biogeochemistry, **48**, 115–146.

Harrison R., Adams A., Licata C., (2003) *Quantifying deep-soil and coarse soil fractions: avoiding sampling bias.* Soil science society of America journal, **67**, 1062-6

Högberg P., Nordgren A., Buchmann N., Taylor A. F. S., Read D. J. (2001) *Large-scale forest girdling shows that current photosynthesis drives soil respiration*. Nature, **411**, 789-792.

Houghton R. A., Hackler J.L., Lawrence K.T. (1999) *The US C Budget: contributions from land-use change*. Science, **285**, 574-8

Huggins D. R., Buyanovsky G. A., Wagner G. H., Brown J. R. (1998) *Soil organic C in the tallgrass prairie-derived region of the corn belt - effects of long-term crop management.* Soil and Tillage Research, **47**, 219-234.

IPPC (2000a) *Special report on emission scenarios* Cambridge University press.

IPPC (2000b) *Land Use, Land Use Change and Forestry. A special report of the IPCC*: Cambridge University press.

IPPC (2001) *Climate Change 2001 The scientific basis* Cambridge University press.

IPPC (2007) *Climate Change: The Physical Science Basis* Cambridge University Press.

Irvine J ., Law B. E. Hibbardw K. A. (2007) *Post fire C pools and fluxes in semiarid ponderosa pine in Central Oregon* Global Change Biology **13**, 1748–1760.

Jacinthe P.A., Lal R., Kimble J.M (2002) *C budget and seasonal C dioxide emission from a central Ohio Luvisol as influenced by wheat residue amendment* Soil & Tillage Research,**67,** 147–157.

Jankauskas B., Slepetiene A., Jankauskiene G., Fullen M. A., Booth C. A. (2006) *A comparative study of analytical methodologies to determine the soil organic matter content of Lithuanian Eutric Albeluvisols* Geoderma **136** 763–773.

Janik. Janik L, Spouncer L., Correll R. Skjemstad J.,(2002) *Sensitivity analysis of the Roth-c soil C model (ver. 26.3 excel).* National C Accounting System Technical Report No. 30

Janssens I. A., Freibauer A., Ciais P., (2003) *Europe's Terrestrial Biosphere Absorbs 7 to 12% of European Anthropogenic CO_2 Emissions.* Science, **300**, 1538-1543.

Janssens I. A., Freibauer A., Schlamadinger B.*et al*, (2005) *The C budget of terrestrial ecosystems at country-scale a European case study* Biogeosciences, **2**, 15–26.

Janzen H.H (2004) *C cycling in earth systems—a soil science perspective* Agriculture, Ecosystems and Environment, **104,** 399–417.

Janzen H.H (2006) *The soil C dilemma: Shall we hoard it or use it?* Soil Biology & Biochemistry,**38,** 419–424.

Jenkinson D. S. Coleman K. (2008) *The turnover of organic C in subsoils. Part 2. Modelling C turnover* European Journal of Soil Science, **59**, 400–413.

Jenkinson D. S., Johnston A. E. (1977) *Soil organic matter in the Hoosfield continuous barley experiment Rothamsted Experimental station*. Report n. part 2

Jenkinson D. S (1990) *The turnover of organic C and nitrogen in soil* Biological Sciences, 329, 361-8

Jones A. Hiederer R., Rusco E., Montanarella L.,(2005) *Estimating organic C in the soils of Europe for policy support* European Journal of Soil Science, **56**, 655–671

Kavvadias A., Alifragis D., Tsiontsis A., Brofas G, Stamatelos G. (2001) *Litterfall, litter accumulation and litter decomposition rates in four forest ecosystems in northern Greece* Forest Ecology and Management **144** 113-127

Kimball B. A., Lemon E. R. (1971) *Air Turbulence Effects Upon Soil Gas Exchange*. Soil Science Society of America Proceedings, **35**, 16-21.

Kinchloe S. *Soil defined* Efficient fertilizer use manual . Mosaic

Kirschbaum M. (2000) *Will changes in soil organic C act as a positive or negative feedback on global warming?* Biogeochemistry ,**48**, 21–51.

Kleber M., Mikutta A., Torn B., Jahn R. (2005) *Poorly crystalline mineral phases protect organic matter in acid subsoil horizons* European Journal of Soil Science, **56**, 717–725.

Kögel-Knabner I., Guggenberger G., Kleber M, K., Leinweber P. (2008) *Organo-mineral associations in temperate soils: Integrating biology, mineralogy, and organic matter chemistry*. Journal of Plant Nutrition and Soil Science, **171**, 61-82.

Kutsch W. L., Staack A., Wötzel J., Middelhoff U., Kappen L. (2001) *Field measurements of root respiration and total soil respiration in an alder forest*. New Phytol, **150**, 157-168.

Laiho R., Penttilä T. and Laine J. (2004) *Variation in Soil Nutrient Concentrations and Bulk Density within Peatland Forest Sites* Silva Fennica **38**(1), 29–41.

Lal R. (2004b) *Soil C Sequestration Impacts on Global Climate Change and Food Security* Science, **304**, 1623-1627

Lal R. (2004a) *Soil C sequestration to mitigate climate change.* Geoderma, **123**, 1-22

Lambers H., Chapin F. S. I., Pons T. L. (1998) *Plant Physiologial Ecology*. New York: Springer.

Leonardi S., Santa Regina I., Rapp M., Gallego H. A., Rico M.,(1996) *Biomass, litterfall and nutrient content in Castanea Sativa coppice stands of outhern Europe* Elsevier **53** 1071-1081

Lettens S., Orshoven J., Wasemal B., Muys B., Perry D., (2005) *Soil organic C change in landscape units of Belgium between 1960 and 2000 with reference to 1990*. Global change biology **11**, 2128-40

Lettens S., De Vos B., Quataert P.,Van Wesemael B., Van Orshoven J. (2007) *Variable C recovery of Walkley-Black analysis and implications for national soil organic C accounting* European Journal of Soil Science, **58**, 1244–1253.

Li C., Frolking S., Harriss R. (1994) *Modeling C biogeochemistry in agricultural soils*. Global Biogeochemical Cycles **8**, 237-54

Loague K., Green R.E. (1991), *Statistical and graphical methods for evaluating solute transport models: overview and application,* J. Contam. Hydrol. **7**, pp. 51–73.

MacDicken K.G. (1997) *A Guide to Monitoring C Storage in Forestry and Agroforestry Projects* Forest C Monitoring Program

Majdi H. (2001) *Changes in fine root production and longevity in relation to water and nutrient availability in a Norway spruce stand in northern Sweden* Tree Physiology **21**, 1057–1061

Marland G., McCarl B., Schneider U. (2001) *Soil C: Policy and Economics* Climatic Change **51**: 101–117.

Martinez A. C., Valladares F., Camarero J., Lopez Arias M., Serrano M., Antonio J.Rodrıguez (2007), *The uncoupling of secondary growth, cone and litter production by*

intradecadal climatic variability in a mediterranean scots pine forest Forest Ecology and Management **253** 19–29

Miltner A., Kopinke F. D., Kindler R., Selesi D., Hartmann A. and Kästner M. (2005) *Non-phototrophic CO_2 fixation by soil microorganisms* Plant and Soil, **269**, 193–203.

Molina J., Clapp C., Shaffer M., Larson (1983) *NCSOIL, a model of nitrogen and C transformation in soil: description calibration and behavior* Soil science society of America journal, **47**, 85-91

Moore T., Turunen J., (2004) *C accumulation and storage in mineral subsoil beneath peat*. Soil science society of America journal, **68**, 690-6

Moyano F. E., Kutsch W. L., Schulze E. (2007) *Response of mycorrhizal, rhizosphere and soil basal respiration to temperature and photosynthesis in a barley field*. Soil Biology and Biochemistry, **39**, 843-853.

Muller T., Jensen L., Hansen S., Nielsen (1996) *Simulating soil C and nitrogen dynamics with the soil-plant atmospheresystem model DAISY* In Evaluation of soil organic matter models using existing long term data sets Powlson and Smith NATO ASI 138 Berlin: Springer pp 275-81

Neary D. G., Klopatek C., Leonard F. Folliot P. F. (1999) *Fire effects on belowground sustainability: a review and synthesis* Forest Ecology and Management, **122** , 51-71.

Nadelhoffer K., Raich, J., (1992) *Fine roots productions estimates and belowground C allocation in forest ecosystem* Ecology vol **73** issue 4, 1139-1147

Odum, E. P. (1969) *The strategy of ecosystem development. Science*, **164**, 262-270.

Page S. E., Siegert F., Rieley J.O (2002) *The amount of C released from peat and forest fires in Indonesia during 1997* Nature Vol. **420** / 7

Palmer C.J., Smith W.D, Conkling B.L., (2002) *Development of a protocol for monitoring status and trends in forest soil C at a national level* Elsevier Science, Environmental Pollution, **116**, 209–219.

Parton W., Schimel D., Cole C., (1987) *Analysis of factors controlling soil organic matter levels in Great-Plains grassland*. Soil science society of America journal , **51**, 1173-9

Parton W., Sanford R., Sanchez P., (1989) *Modeling soil organic matter dynamics in tropical soils*. In Dynamics of soil organic matter dynamics in tropical ecosystems Coleman D. C. Oades M., NifTAL project University of Hawaii Press pp 69-95

Paustian K., Cole C.V., Sauerbeck D., Sampson N., (1998) *CO_2 mitigation by agriculture: an overview*. Climate Change, **40**, 135-162.

Peltoniemi M., Thürig E., Ogle S., Palosuo T., Schrumpf M., Wutzler T., Smith P. Mäkipää, R. (2007) *Models in Country Scale C Accounting of Forest Soils*. Silva Fennica **41** (3) review articles.

Peressotti A., Cotrufo M.F., Miglietta F.,Valentini R., Inglima I., Pecchiari M., Alberti G., Arriga N., Piermatteo D., Zaldei A., (2006) *Mediterranean forest ecosystem under changing precipitation* regimes Open Science Conference on the GHG Cycle in the Northern Hemisphere

Post W. M. and K. C. Kwon (2000) *Soil C Sequestration and Land-Use Change: Processes and Potential* Global Change Biology **6**, 317–328

Post W. M., Izaurralde R.. Mann C, Bliss N. (2001) *Monitoring and verifying changes of organic C in soil* Climatic Change, **51**,73–99.

Powlson D. S., Smith P., Coleman K., Smith J. U. (1998) *A european network of long-term sites for studies on soil organic matter. Soil and Tillage Research*, **47**, 263-274.

Pregitzer K. S., Euskirchen E. S.(2004) *C cycling and storage in world forests: biome patterns related to forest age* Global Change Biology, **10**, 1–26.

Raich W., Potter S., Bhagawati D.(2002) *Interannual variability in global soil respiration (1980-1994)* Global Change Biology,**8**, 800-812

Rapp M., Santa Regina I., Rico M., Antonio Gallego H. (1999) *Biomass, nutrient content, litterfall and nutrient return to the soil in Mediterranean oak forests* Forest Ecology and Management **119** 39-49

Rasse D.P, Ritchie J.T, Wilhelm W.W., Wei J.. Martin E.C, (2000), *Simulating Inbred–Maize yields with CERES-IM,* Agron. J. **92** pp. 672–678.

Robertson, G. P. (2004) *Abatement of nitrous oxide, methane, and the other non-CO_2 greenhouse gases gases: The need for a systems approach*. Pages 493-506 in C. B. Field and M. R. Raupach, eds. The Global C Cycle. Island Press, Washington, DC, USA.

Rodeghiero M., Cescatti A. (2005) *Main determinants of forest soil respiration along an elevation/temperature gradient in the Italian Alps* Global Change Biology **11**, 1024–1041.

Rodeghiero M., (2006) *Metodologie per la stima del Cio nei suoli forestali (Assessing forest C stock and temporal changes: methods and regulations*) Atti del 42 Corso di Cultura in Ecologia.

Rodeghiero M., Falloon P. and Kutsch W. L. (2009) *The role of soils in the Kyoto Protocoll.* Soil C dynamics ch. **13** Cambridge University Press

Rothamsted Research (2005) *Climate change and land management* Climate change brochure.

Saby P., Patricia A. Bellamy H. et al., (2008) *Will European soil monitoring networks be able to detect changes in topsoil organic C content?* Global Change Biology vol. **14** 2432 2442

Santa Regina I., Tarazona T. (2001) *Nutrient pools to the soil through organic matter and throughfall under a Scots pine plantation in the Sierra de la Demanda, Spain* Eur. J. Soil Biol. **37** 125−133

Schimel, D. S., House, J. I., Hibbard, K. A. et al (2001) *Recent patterns and mechanisms of C exchange by terrestrial ecosystems*. Nature, **414**, 169-72.

Schlesinger W.H., Andrews J. (2000) *Soil respiration and the global C cycle* Biogeochemistry, **48,** 7–20.

Schoning I. and Kogel-Knabner I. (2006) *Chemical composition of young and old C pools throughout Cambisol and Luvisol profiles under forests* Soil Biology & Biochemistry, **38,** 2411–2424.

Schulze E. D. and Freibauer A. (2005) *C unlocked from soils* Nature, Vol 437

Singh J. S., Gupta S. R. (1977) *Plant decomposition and soil respiration in terrestrial ecosystems*, Botanical Review, **43**, 449-528.

Six J., Ogle S. M. Breidt F. J. Conant R .T. Mosier A. R. Paustian K. (2004)*The potential to mitigate global warming with no-tillage management is only realized when practised in the long term* Global change biology, **10**, 212-218

Sun O., Campbell J., , Beverly E., (2004) *Dynamics of C stocks in soils and detritus across chronosequences of different forest types in the Pacific Northwest, USA* Global Change Biology **10**, 1470–1481

Shirato Y., Yokozawa M. (2006) *Acid hydrolysis to partition plant material into decomposable and resistant fractions for use in the Rothamsted C model* Soil Biology & Biochemistry **38** 812–816

Smith P. Smith J.U. Powlson D.S., McGill W.B. Arah J.R.M, Coleman K., et al, (1997) *A comparison of the performance of nine soil organic matter models using datasets from seven long-term experiments* Geoderma **81** 153-225

Smith P. (2004) *How long before a change in soil organic C can be detected?* Global Change Biology, **10**, 1878-1883.

Smith P. (2004a) *C sequestration in croplands: the potential in Europe and the global context* Europe . J. Agronomy ,**20**, 229–236.

Smith P. (2004b) *Soils as C sinks: the global context.* Soil Use and Management, **20**, 212-218.

Smith P. (2004c) *Monitoring and verification of soil C changes under Article 3.4 of the Kyoto Protocol* Soil Use and Management, **20**, 264-270.

Smith P. (2005) *An overview of the permanence of soil organic C stocks: influence of direct human-induced, indirect and natural effects* European Journal of Soil Science, **56**, 673–680.

Smith P., Goulding K. W., Smith K. A. (2001) *Enhancing the C sink in European agricultural soils: including trace gas fluxes in estimates of C mitigation potential*. Nutrient cycling in agroecosystems, **60**, 237-252.

Smith P., Powlson D. S., Smith, J. U., Glendining M. J. (1996) *The GCTE SOMNET. A global network and database of soil organic matter models and long-term datasets*. Soil Use and Management, **12**, 104.

Smith P., Falloon P. and Werner L. Kutsch (2009) *The role of soils in the Kyoto Protocoll.* Soil C dynamics ch. **13** Cambridge University Press

Staddon P. L., Heinemeyer A., Fitter A. H. (2002) *Mycorrhizas and global environmental change: research at different scales*. Plant Soil, **244**, 253-261.

Subke J. A., Inglima I., Cotrufo M. F. (2006) *Trends and methodological impacts in soil CO_2 efflux partitioning: A meta analytical review*. Global Change Biology, **12**, 921-943.

Tilman D., Reich P., Phillips H., (2000) *Fire suppression and ecosystem C storage* Ecological Society of America **81**, 2680–2685.

Thornley J., Verbene E., (1989) *A model of nitrogen flows in grassland* Plant cell and environment, **12**, 863-86

Trumbore S. (2006) *C respired by terrestrial ecosystems - recent progress and challenges,* Global Change Biology, **12**, 141-153.

Van Bavel C. H. M. (1952) *Gaseous diffusion and porosity in porous media*. Soil Science, **73**, 91–104.

Vance E. D. (2003*)Approaches and Technologies for Detecting Changes in Forest Soil C Pools.* Soil Science Society of America Journal, 67, 1582.

Vance E. D., Chapin F. S. (2001) *Substrate limitations to microbial activity in taiga forest floors. Soil* Biology and Biochemistry, **33**, 173-188.

Verbene E., Hassink J., Dewilligen P., Groot J., (1990) *Modeling organic matter dynamics in different soils* Netherlands journal of agricultural science, **38**, 221-38

Von Lützow M., Kogel-Knabener *et al* (2006) *Stabilization of organic matter in temperate soils: mechanisms and their relevance under different soil condition- a review* European Journal of Soil Science, **57**, 426–445.

Von Lützow M., Kogel-Knabener, Ekschmitt K., Flessac H. *et al*., (2007), *SOM fractionation methods: Relevance to functional pools and to stabilization mechanisms* Soil Biology & Biochemistry, 39, 2183–2207.

Kutsch W. L.., Bahn M., Heinemeyer A. (2009*) Soil C Dynamics*. Cambridge University press.

Wiseman C., Puttmann W., (2005) *Soil organic C and its sorptive preservation in central Germany* European Journal of Soil Science, **56**, 65–76

Zimmermann S., Braun S., Conedera M., Blaser P. (2002) *Macronutrients inputs by litterfall as opposed to atmospheric deposition into two contrasting chestnut forest stands in Southern Switzerland* Forest Ecology and Management **161** 289-302