ISSN 1020-4555

Integrated Crop Management Vol. 11–2010

Grassland carbon sequestration: management, policy and economics

Proceedings of the Workshop on the role of grassland carbon sequestration in the mitigation of climate change

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Rome, April 2009

Prepared for the
Plant Production and Protection Division
Food and Agriculture Organization of the United Nations (FAO)

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FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS
Rome, 2010

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ISBN 978-92-5-106695-9

FOREWORD

Grasslands play a unique role as they link agriculture and environment and offer tangible solutions ranging from their contribution to mitigation of and adaptation to climate change, to improvement of land and ecosystem health and resilience, biological diversity and water cycles while serving as a basis of agricultural productivity and economic growth.

They are a major ecosystem and a form of land use giving us not only a range of useful products (meat, milk, hides, fur, etc.) but also 'ecosystem services'. The latter include the important role of grasslands in biodiversity, provision of clean water, flood prevention and, the focus of this book, carbon (C) sequestration. Soil carbon is important as a key aspect of soil quality but the sequestration or 'locking up' of carbon in the soil has acquired new importance in recent years in the context of climate change. Clearly, a central aspect of global environmental change is the build up of carbon dioxide (and other greenhouse gases) in the atmosphere. Therefore, to put it simply, the extent to which C can be taken out of the atmosphere by plants and stored in the soil is important in mitigating the impact of increased emissions. It seems logical that grassland farmers around the world should be encouraged to undertake management changes leading to enhanced sequestration and that policy to incentivize this process should be developed.

However, this apparent simplicity is deceptive. Much of this book is focused on the complexities of quantifying and monitoring C sequestration in grassland soils, in developing proxy indicators of likely changes in sequestration over time with different managements and in understanding the socio-economic framework within which policies can be successfully developed. These are important tasks not only with respect to climate change mitigation but also in the light of the other benefits that increased soil C can bring and the broader needs of developing mechanisms to enhance sustainable development for the many smallholders and pastoralists dependent on healthy grasslands for their livelihoods.

This book profiles 13 contributions by some of the world's best scientists on the subjects of measuring soil C in grassland systems and sustainable grassland management practices. While many different aspects of C sequestration in grasslands are provided as far as possible, many gaps in our knowledge are also revealed and, in line with the role of the Food and Agriculture Organization of the United Nations (FAO) of disseminating available information, it is hoped that this book will promote discussion, prompt further research, help develop global and national grassland strategies, and contribute to sustainable production intensification.

The major contribution of Mr Michael Abberton, Ph.D., Leader of Crop Genetics, Genomics & Breeding Research Division, Aberystwyth University, in the overall organization of the workshop and editing is much appreciated by FAO as is the contribution to the editing made by Mr Rich Conant, Ph.D., Ecosystem Ecologist, University of Colorado. Thanks are particularly due to Caterina Batello, Senior Officer; Constance Neely, Senior Rangeland Consultant; Eva Moller, Administrative Assistant and Suzanne Redfern, Consultant, Plant Production and Protection Division (FAO), for ensuring that the proceedings were brought to publication.

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CONTENTS

ABBREVIATIONS

AFOLU	Agriculture, Forestry and Land Use
AGRA	Alliance for a Green Revolution in Africa
BAU	business as usual
bST	Bovine somatotropin
C	Carbon
CAD	central anaerobic digestion
CARB	California Air Resources Board
CATIE	Centro Agronómico Tropical de Investigación y Enseñanza
CBD	Convention on Biological Diversity
CBP	Carbon Benefits Project
CCS	capture and storage
CDM	Clean Development Mechanism
CGIAR	Consultative Group on International Agricultural Research
CH_4	Methane
CIPAV	Centro para la Investigación en Sistemas Sostenibles de Producción Agropecuaria
CO	Carbon monoxide
CO_2	Carbon dioxide
CoC	command and control
CO_2eq	carbon dioxide equivalent
CPRS	Carbon Pollution Reduction Scheme
CRP	Conservation Reserve Program
CSU	Colorado State University
CTs	condensed tannins
DfID	Department for International Development (UK)
DNDC	DeNitrification-DeComposition (ecosystem model)
EC	eddy covariance
EQIP	Environmental Quality Incentives Program
ESD	Ecological Site Description
ETS	European Trading Scheme
FAPAR	Fraction of Absorbed Photosynthetically Active Radiation
FP	fast pyrolysis
GCWG	Grassland Carbon Working Group
GDP	Gross Domestic Product
GEF	Global Environmental Facility
GHG	greenhouse gas
GIS	Geographic Information System
GLADA	Global Assessment of Land Degradation and Improvement
GLASOD	Global Assessment of Soil Degradation
GLC	Global Land Cover
GPS	Global Positioning System
Gt	giga tonnes = 10^9g
GWP	global warming potential
ha	hectare
HC	high carbon price
HWSD	Harmonized World Soil Database
ICARDA	International Centre for Agricultural Research in the Dry Areas
IGBP	International Global Biosphere Programme
IPCC	Intergovernmental Panel on Climate Change

IUCN	International Union for Conservation of Nature
K	Potassium
Kt	kilo tonnes
LADA	Land Degradation Assessment in Drylands
LAI	Leaf Area Index
LC	low carbon price
LCA	Life Cycle Analysis
LIBS	Laser-Induced Breakdown Spectroscopy
IEM	Integrated ecosystem management
KAPSLM	Kenya Agricultural Productivity and Sustainable Land Management
LIBS	Laser-Induced Breakdown Spectroscopy
MACC	marginal abatement cost curve
MBI	market-based instruments
MDG	Millennium Development Goals
meq	milliequivalents
Mg	mega grams = 10^6g
MIRS	Mid-InfraRed Spectroscopy
MMV	measurement, monitoring and verification
MRT	mean residence time
MRV	monitoring, reporting and verification
Mt	Mega tonnes= 10^6t
$MTCO_2e$	metric tonnes of carbon dioxide equivalent
N	Nitrogen
N_2	Nitrogen gas
NAMA	Nationally Appropriate Mitigation Actions
NAPA	National Adaptation Programmes of Action (of UNFCCC)
NDVI	normalized difference vegetation index
NIRS	Near InfraRed Spectroscopy
N_2O	Nitrous Oxide
NPP	Net Primary Productivity
NRCS	Natural Resources Conservation Service
NRM	natural resource management
OAD	Overseas Development Assistance
OFAD	on-farm anaerobic digestion
P	Phosphorous
PASS	Programme for Africa's Seed System
PES	Payment for Environmental Services
Pg	Peta grams = 10^{15}g
PRSP	Poverty Reduction Strategy Papers
REDD	Reducing Emissions from Deforestation and forest Degradation
SIC	soil inorganic carbon
SLM	sustainable land management
SMU	soil map unit
SOC	Soil Organic Carbon
SOFESCA	Soil Fertility Consortium for Southern Africa
SOM	Soil Organic Matter
SP	slow pyrolysis
SPC	shadow price of carbon
SPS	Silvopastoral systems
SSURGO	Soil Survey Geographic database
STATSGO	State Soil Geographic database
STM	state and transition model
t	tonnes
TA	tropical America
Tg	Tera grams = 10^{12}g
UNCBD	United Nations Convention on Biological Diversity

UNCCD	United Nations Convention to Combat Desertification
UNFCCC	United Nations Framework Convention on Climate Change
USDA	United States Department of Agriculture
US-EPA	United States Environmental Protection Agency
VCS	Voluntary Carbon Standard
WCI	Western Climate Initiative
WDPA	World Database on Protected Area
WOCAT	World Overview of Conservation Approaches and Technologies
WWF	World Wide Fund for Nature

Michael B. Jones

CHAPTER I

Potential for carbon sequestration in temperate grassland soils

ABSTRACT

Soil carbon (C) sequestration in grasslands may mitigate rising levels of atmospheric carbon dioxide (CO_2) but there is still great uncertainty about the size, distribution and activity of this "sink". Carbon accumulation in grassland ecosystems occurs mainly below ground where soil organic matter (SOM) is located in discrete pools, the characteristics of which have now been described in some detail. Carbon sequestration can be determined directly by measuring changes in C stocks or by simulation modelling. Both methods have many limitations but long-term estimates rely almost exclusively on modelling. Management practices and climate strongly influence C sequestration rates, which, in temperate grasslands across Europe, range from 4.5 g C/m^2/year (a C source) to 40 g C/m^2/year (a C sink). Because of uncertainties in location of sinks and their activity, we currently only have enough information to infer the order of magnitude of soil C sequestration rates in temperate grasslands.

INTRODUCTION

Carbon (C) sequestration by terrestrial ecosystems is responsible for a partial mitigation of the increase in atmospheric carbon dioxide (CO_2) but the exact size and distribution of this sink for C remain uncertain (Janssens *et al.*, 2003). Carbon sequestration is the process of removing CO_2 from the atmosphere and storing it in C pools of varying lifetimes. The amount of C sequestration is the overall balance between photosynthetic gain of CO_2-C and losses in ecosystem respiration as well as lateral flows of C, particularly as dissolved organic and inorganic C (Chapin *et al.*, 2006). As about 32 percent of the Earth's natural vegetation is temperate grassland (Adams *et al.*, 1990), these ecosystems make a significant contribution to the global C cycle. It has been estimated that soil organic matter (SOM) in temperate grasslands averages 3.31 x 10^4 g m^2 and that grasslands contain 12 percent of the Earth's SOM. The relatively stable soil environment is conducive to accumulation of organic matter because of the slower turnover of C below ground. Consequently, grassland soils contain large stocks of C in the form of SOM that has accumulated during the lifetime of the grassland community.

The main factors that influence the accumulation and sequestration of C are past and current land-use changes; agricultural management, including the horizontal transfer of hay/silage and manure deposition and application, soil texture, vegetation composition and climate. The amount of organic matter in the soil at a given moment is the net result of additions from plant and animal residues and the losses through decomposition. The C in the soil is present in a complex association with the soil particles and it is the nature of this relationship that ultimately determines how long the C remains in the soil and therefore the C sequestration potential of the soil. Research on the quantification of C sequestration is based on the three associated approaches of monitoring C stocks, experimental manipulations and modelling. Each has its uncertainties and knowledge gaps but it is through the joint use of these that a clearer picture emerges.

CARBON IN GRASSLAND SOILS

Classical descriptions of SOM have normally combined chemical extractions with the identification of specific chemical compounds, but this has unfortunately contributed little to a functional understanding of soil processes (Jones and Donnelly, 2004). More recently, the approach has been to identify different fractions of plant residues at different stages of decomposition or to group together the various organic matter components into categories with similar breakdown characteristics (Six *et al*., 2004). The turnover rate of SOM is an important property of different types of organic matter and many SOM models use a compartmentation approach with pools represented as fast, intermediate and slow organic matter turnover.

Most organic matter enters the soil as readily recognizable plant litter and is mineralized within months (Christensen, 1996). A small portion, however, may be stabilized to form aggregates through interactions with mineral surfaces. These aggregates are formed initially by root exudates and fungal and plant debris. Decomposition reduces the size of these aggregates, which subsequently become encased in clay particles. As these particles form barriers to microbes the C becomes physically protected and more recalcitrant so that they are stabilized for periods up to thousands of years (Six *et al.*, 2004; Lehmann, Kinyangi and Solomon, 2007). In many soils, such as mollisols and alfisols, strong feedbacks exist between SOM stabilization and aggregate turnover (Jastrow and Miller, 1998; Six *et al.*, 2004). In these soils, the deposition and transformation of SOM are dominant aggregate stabilizing mechanisms. Soil aggregate structure is usually hierarchic (Tisdall and Oades, 1982; Oades and Waters, 1991) with primary particles and silt-sized aggregates (<50 µm diameter) bound together to form micro-aggregates (50–250 µm diameter) and these primary and secondary structures, in turn, bound into macro-aggregates (>250 µm diameter). Current evidence suggests that micro-aggregates are formed inside macro-aggregates, and that factors increasing macro-aggregate turnover decrease the formation and stabilization of micro-aggregates (Angers, Recous and Aita, 1997; Gale, Cambardella and Bailey, 2000; Six, Elliott and Paustian, 2000; Six *et al.*, 2004). However, micro-aggregates, and smaller aggregated units, are generally more stable and less susceptible to disturbance than macro-aggregates (Tisdall and Oades, 1982; Dexter, 1988; McCarthy *et al.*, 2008). Soil C storage following land-use change and other management changes has previously been attributed to changing C contents of micro-aggregates within macro-aggregates (Six *et al.*, 2004). For these reasons, soil physical fractionation forms a useful tool to evaluate changes in soil C and SOM dynamics.

In a conceptual model of soil C dynamics, Six *et al.* (2002) distinguished the SOM that is protected either physically or biochemically against decomposition from that which is unprotected. They identified four measured pools as follows: (i) an unprotected C pool; (ii) a biochemically protected C pool; (iii) a silt and clay-protected C pool; and (iv) a micro-aggregate-protected C pool. The unprotected SOM pool consists of the light fraction (LF) or particulate organic matter (POM) fraction, which are considered conceptually to be identical pools by Six *et al.* (2002). The origin of both LF and POM, which are highly labile, is mainly plant residues but they may also contain microbial debris.

Protected SOM is stabilized by three main mechanisms. First, chemical stabilization is the result of chemical binding between soil minerals (clay and silt particles) and SOM. Second, biochemical stabilization is a result of the chemical complexing processes between substrates such as lignins and polyphenols and soil particles. Finally, physical aggregates form physical barriers between microbes and enzymes and their substrates. Organic matter can be protected against decomposition when it is positioned in pores that are too small for bacteria or fungi to penetrate or it can be inside large aggregates that become partially anaerobic because of slow O_2 diffusion through the small intra-aggregate pores (Marinissen and Hillenaar, 1997). Soil aggregates are held together by microbial debris and by fungal hyphae, roots and polysaccharides, so that increased amounts of any of these agents will promote aggregation. Earthworms (*Lumbricidae*) are often the dominant soil-ingesting animals that mix plant residues and mineral soil, thus promoting aggregate stability. Marinissen (1994) found a strong correlation between macro-aggregate stability and earthworm numbers.

SOIL ORGANIC MATTER MODELS

Although models are inevitably simplifications of reality, they are crucially important because they are able to assess the impacts of combinations of environmental factors that are difficult or impossible to establish in experimental treatments. In fact, models are frequently the only available tool to study climate change related issues and other long-term effects. The general approach in modelling is to simplify nature by distinguishing only a small number of C pools, with different levels of stability and therefore different turnover rates (Smith *et al.*, 1997). The turnover rates are generally considered to be controlled by substrate supply, temperature and water but the degree of control exerted by these factors is assumed to differ between the pools.

Widely used models include: CENTURY (Parton *et al.*, 1987); DNDC (Giltrap *et al.*, 1992); Roth C (Coleman *et al.*, 1997); and LPJ-DGVM (Zaehle *et al.*, 2007). They are all process-based or mechanistic models that use an understanding of ecological processes and the factors influencing these processes to forecast C stocks and changes under different management or environmental scenarios. They can also scale to larger spatial scales than direct measurements (Smith *et al.*, 2005; Janssens *et al.*, 2005; Zaehle *et al.*, 2007) and at continental and global scale, models such as C Emission and Sequestration by Agricultural land use (CESAR) (Vleeshouwers and Verhagen, 2002) have been developed that can run with the very limited data available at this scale. When CESAR was run for the European continent to evaluate the effects of different CO_2 mitigation measures on soil organic carbon (SOC) and was parameterized for several arable crops and grassland, Vleeshouwers and Verhagen (2002) found considerable regional differences in the sequestration of European grasslands resulting from the interaction between soil and climate. Average C fluxes under a business-as-usual scenario in the 2008–12

Kyoto commitment period was 52 g C/m^2/year and conversion of arable land to grassland yielded a flux of 144 g C/m^2/year. Application of farmyard manure increased C sequestration by 150 g C/m^2/year.

Models are therefore an essential tool to assess the impacts of climate change as well as land-use change, although the outcomes should still be evaluated with care as there is still insufficient understanding of the underlying processes. Although many experimental studies have demonstrated the complexity of the C cycle and the large number of interactions between the environmental variables, at present only a fraction of the complexity is represented in the models.

EFFECTS OF MANAGEMENT ON CARBON SEQUESTRATION

When vegetation and soil management practices change they can have a wide range of effects on the processes that determine the direction and rate of change in SOC content (Conant, Paustian and Elliott, 2001; Chen *et al.*, 2009). Among the most important for increasing SOC storage are increasing the input rates of organic matter, changing the decomposability of organic matter, placing organic matter deeper in the soil and enhancing the physical protection of the soil fractions (Post and Kwon, 2000). Increased management intensity associated with higher nitrogen (N) inputs and frequent cutting applied to temperate grasslands in Switzerland has also been demonstrated to stimulate C sequestration (Ammann *et al.*, 2007) and this appears to be the consequence of reduced rates of SOM loss through mineralization under more intensive management.

Nitrogen fertilization increases productivity in N-limited grasslands and if this is greater than any associated increase in decomposition rate, it will lead to an overall increase in net ecosystem production (NEP) (Conant, Paustian and Elliott, 2001). As a result of an expert assessment of temperate grasslands in France, Soussana *et al.* (2004) concluded that moderate applications of nitrogenous fertilizer increase C input to the soil more than they increase soil C mineralization. However, intensive fertilizer use accelerates mineralization and enhances decomposition of SOM, resulting in reduced soil C stocks. Jones *et al.* (2006) investigated how different organic and mineral fertilizer treatments influenced C sequestration in a temperate grassland in Scotland, United Kingdom. They assessed the effect of additions of sewage sludge, poultry manure, cattle slurry and two different mineral fertilizers (NH_4, NO_3 and urea). The addition of organic manures resulted in increased C storage through sequestration with most C being retained following additions of poultry manure, and least following additions of sewage sludge. However, the manure input also enhanced the emission of nitrous oxide N_2O and, when expressed in terms of global warming potential, the benefits of increased C sequestration were far outweighed by the additional loss of N_2O. In this particular study, mineral fertilizer had only a small impact on C sequestration (Jones *et al*., 2006). However, the addition of N is also very likely to stimulate N_2O emissions, thereby offsetting some of the benefits of C sequestration (Conant *et al*., 2008). Furthermore, on organic soils, because of the relatively large pool of organic matter available for decomposition, N fertilization may trigger large C losses (Soussana *et al*., 2007). In summary, practices that enhance C stocks appear to be those that reduce intensification of highly fertilized grasslands and stimulate a more moderate intensification of nutrient-poor grasslands.

Most livestock systems on grasslands generate large amounts of manure that is returned to the fields, including in mixed farming systems land for arable crops. When spread on grassland, these C-rich farm manures help to maintain or increase the soil C stocks. However, Smith *et al*. (2007) have proposed that the residence time of organic C is greater in arable soils than in grasslands, with the consequence that farm manures have a greater C sequestration potential when applied to arable land. Soussana *et al*. (2004) have argued that few experimental data support this proposition.

Grazers significantly impact on the C balance of grasslands through effects on vegetation type, organic matter inputs to the soil microbial community and soil structure through trampling. The intensity and timing of grazing influence the removal of vegetation and C allocation to roots as well as the grassland flora. All these influence the amount of C accumulating in the soil. Because of the many types of grazing practices and the diversity of plant species, soils and climates, the effects of grazing are inconsistent. Grazing animals emit methane (CH_4) which offset the gains from C sequestration when a full greenhouse budget is calculated (Soussana *et al*., 2007). For nine contrasted grassland sites covering a major climatic gradient over Europe the emissions of N_2O and CH_4 resulted in a 19 percent offset of the net ecosystem exchange of CO_2 sink activity (Soussana *et al*., 2007). Based on modelling of an upland semi-natural grassland site at Laqueille in the Massif Central, France, Soussana *et al*. (2004) concluded that the CO_2 sink would be greatest, and CH_4 sources associated with the grazing cattle smallest, at low stocking densities.

Introducing grass species with high productivity, or C allocation to deeper roots, has the potential to increase soil C, although there is some uncertainty about effectiveness of this in practice (Conant, Paustian and Elliott, 2001). However, the introduction of legumes into grasslands has been clearly demonstrated to promote soil C storage through enhanced productivity from the associated N inputs (Soussana *et al*., 2004). There is also evidence from experiments that have manipulated biodiversity on former arable fields that an increase in plant species richness has a positive effect on the buildup of new C in the soil (Steinbeiss *et al*., 2008).

Finally, the land management option of converting tilled land to permanent grassland has been demonstrated worldwide to increase soil C content and net soil C storage (Post and Kwon, 2000; Conant, Paustian and Elliott, 2001; McLauchlan, Hobbie and Post, 2006). The rates of C sequestration observed or estimated in these newly established grasslands are some of the highest recorded. For example, 144 g C/m^2/year for grasslands in Europe (Vleeshouwers and Verhagen, 2002) and 62 g C/m^2/year in the mid-western United States (McLauchlan, Hobbie and Post, 2006).

EFFECTS OF CLIMATE CHANGE ON CARBON SEQUESTRATION

It is now well established that the observed increase in atmospheric CO_2 and other greenhouse gases (GHG) since the Industrial Revolution will continue into the future and is leading to climate change that is manifested primarily through increased global temperatures and changed patterns of rainfall (IPCC, 2007). Climate change has impacts on two crucial stages of the C cycle: decomposition and net primary productivity (NPP). Furthermore, the increasing CO_2 concentration in the atmosphere is anticipated to have direct effects on the C cycle in grasslands through increasing primary productivity that may also impact on C sequestration (Jones and Donnelly, 2004).

Elevated temperatures have been shown in many experimental studies to increase the rate of soil respiration associated with decomposition that leads to a loss of soil C. It is furthermore expected that increasing temperature will affect decomposition more than primary productivity and the consequence of this is a net loss of soil C and a positive feedback to the climate system in the long term. The loss is expected to be greatest at higher latitudes where the current decomposition processes are limited by temperature, although experimental studies have not always supported this hypothesis. Warming experiments have shown an "acclimation" of soil respiration whereby the magnitude of the response declines over time, most likely because of a limitation of readily available substrate supply (Kirschbaum, 2006). Furthermore, changes in microbial composition over time may result in a transition to communities that are more tolerant of high temperatures (Zhang *et al*., 2005). The result may be lower soil C loss than anticipated at elevated temperatures. However, there is still no agreement on how temperature sensitivity varies with the lability of organic matter substrate, although Conant *et al*. (2008) have recently presented evidence for an increase in the temperature sensitivity of SOM decomposition as SOM lability decreases. These results therefore suggest that future losses of soil C may be even greater than previously supposed under global warming, and may actually increase the positive feedback on the climate.

The other climate variable that will be influenced by climate change is rainfall. It is anticipated from global climate models that the changing patterns of precipitation in temperate climates will probably mean drier summers and wetter winters (IPPC, 2007). The increased frequency and severity of droughts in summer will reduce net primary productivity, and therefore the supply of organic matter to the soil, as well as decrease the rate of decomposition. However, because higher temperatures are likely to be experienced at the same time, it is difficult to separate the single and interactive effects of drought and temperature.

Elevated CO_2 concentrations have a direct positive effect on NPP but there are strong interactions with nutrient and water availability. Although it has been hypothesized that higher CO_2 concentrations may increase net C sequestration, this can only be sustained if soil mineralization lags behind the increase in soil C input. There is conflicting evidence on the impact on decomposition so that while some studies suggest that the additional C may accelerate decomposition (Fontaine *et al*., 2007), others have found that additional litter will form coarse particulate organic matter that initiates aggregation formation (Six *et al*., 1998). In general, evidence from single factor studies suggests that impacts on decomposition are relatively small as it is the soil properties that determine turnover rates and most of the new C does not enter the long-lived pools (Hagedorn, Spinnler and Siegwolf, 2003). The use of labelled CO_2 in elevated atmospheric CO_2 treatments in an open field experiment allowed the tracing of the long-term dynamics of C in a pasture system (van Kessel *et al*., 2006). It was concluded that elevated CO_2 did not lead to an increase in soil C and it was suggested that the potential use of fertilized and regularly cut pastures as net soil sinks under long-term elevated CO_2 appears to be limited (van Kessel *et al*., 2006). Experimental evidence from multifactoral experiments is limited, but Shaw *et al*. (2002) also found no overall increase in C sequestration in a grassland system.

Furthermore, van Groenigen *et al*. (2006), using meta-analysis, have shown that soil C sequestration under elevated CO_2 is constrained both directly by N availability and indirectly by nutrients needed to support N_2 fixation.

Smith *et al*. (2005) have used the process-based SOC model (RothC) to make a pan-European assessment of future changes in grassland SOC stocks for the period 1990–2080 under climate change as well as land-use and technology change. They find that while climate change will be a key driver of change in soil C over the twenty-first century, changes in technology and land use are also predicted to have very significant effects. When incorporating all factors, grasslands showed a small increase of 3–6 x 10^2 g C/m^2 but when the greatly reduced area of grassland is accounted for, total European grassland stocks decline in three out of four climate scenarios used. Zaehle *et al*. (2007), in another modelling exercise, showed that C losses resulting from climate warming reduce or even offset C sequestration resulting from increased NPP, while Jones *et al*. (2005) suggest that the magnitude of the projected positive feedback between the climate and C cycle is dependent on the structure of the soil C model. Scenario studies carried out with models indicate that climate change is likely to accelerate decomposition and as a result decrease soil C stocks. However, these effects are partly or wholly reversed by increasing NPP, changes in land use and soil management technologies. In order to use the models to best effect there is a requirement for more detailed information on a large number of processes and drivers (Jones *et al*., 2005)

LIMITS TO THE SIZE OF THE CARBON POOLS

As the capacity of soils to sequester C is finite, when a change in management or climate stimulates the process of C sequestration then this process will continue until a new equilibrium is achieved. At this point, the C input is equal to the C released by the mineralization of organic matter (Post and Kwon, 2000). The accumulation of C over time is a non-linear process and it normally takes between 20 and 100 years to reach a new equilibrium (Freibauer *et al*., 2004; Soussana *et al*., 2004). Therefore, soil C sequestration does not have an unlimited potential to mitigate CO_2 emissions and benefits offered by grasslands sequestering grasslands probably do not go beyond a 20–25 year time frame (Skinner, 2008).

The final level at which the soil C stabilizes depends on the ability of the soil to stabilize C. This is related to the soil structure and composition, the prevailing climate determining soil moisture and temperature, the quality of the C added to the soil and the balance between the C input to the soil and the C lost through respiration (Post and Kwon, 2000). Grasslands in general store more C than arable soils because a greater part of the SOM input from root turnover and rhizodeposition is physically protected as POM and a greater part of this is chemically stabilized (Soussana *et al*., 2004).

McLauchlan, Hobbie and Post (2006) have shown that former agricultural lands of the northern Great Plains that were depleted in SOM by decades of cultivation accumulate soil C linearly for at least the first 40 years after conversion from agricultural land to grassland. Furthermore, the recalcitrant C formed in former agricultural soil can function as an immediate and persistent sink because of the formation of stable microbial products. However, these soils do not continue to accumulate C beyond about 75 years from the cessation of agriculture.

MONITORING CHANGES IN SOIL CARBON

The evaluation of the confidence with which changes in SOC content can be detected is important for the implementation of national and international directives, national treaties, emission trading schemes and *a posteriori* validation of predicted changes using modelling. The inherent spatial variability of SOC content will strongly influence the ability to detect changes (Conant and Paustian, 2002).

Methods to estimate changes in soil C pools involve soil sampling by: (i) repeated measurements in time or from chronosequences where simultaneous measurements are made at sites with different histories of change; (ii) modelling; or (iii) a combination of monitoring and modelling and measurements of CO_2 fluxes. While measurements of CO_2 fluxes using soil respiration chambers or eddy covariance methods provide important information on processes on time scales from hours to years (Flanagan, Wever and Carlson, 2002; Li *et al*., 2005; Novick *et al*., 2004), they are less suitable for monitoring because of difficulties in separating plant respiration from decomposition of dead SOM, and insufficient geographical coverage of these measurements.

The most established form of direct measurement is to extract and analyse soil core samples. The sample is combusted in the laboratory and analysed for C content. This process does not differentiate between organic and inorganic C so that inorganic C is normally removed before analysis by digestion with acid. Monitoring by sampling requires large numbers of spatially distributed soil C pool measurements. This is time-consuming and therefore costly. Sampling costs can be reduced by stratification. Stratification is a means of improving the efficiency of sampling by subdividing the area to be measured into regions (strata) that are relatively homogeneous in characteristics that affect stocks and fluxes of C. Stratification allows optimal allocation of sampling effort to the different strata to minimize the cost for a given level of precision. The amount of work can be reduced by combining modelling with sampling even though there are concerns about the current reliability of the results from models.

Several studies have assessed the feasibility of verifying the effects of changes in land use or management practice on SOC (Conant and Paustian, 2002; Smith, 2004; Saby *et al.*, 2008) both at the field and regional scale. At the regional scale, Saby *et al.* (2008) found that the minimum detectable changes in SOC concentration differ among the national soil-monitoring networks in Europe and that considerable effort would be necessary for some countries to reach acceptable levels of minimum detectable changes in C concentration. They concluded that, in Europe, national soil monitoring networks are not able to detect annual changes in SOC stocks but they would allow longer-term assessments over about ten years. Negra *et al.* (2008) have recently described the characteristics of indicators of C storage in ecosystems in the United States. They make it clear that in order to facilitate detection of meaningful patterns in C storage it is important to measure both changes in C stocks over time as well as total C stocks. However, they acknowledge that these measurements are constrained by serious technical limitations that are largely a result of spatial heterogeneity.

CARBON STOCKS AND STORAGE RATES IN TEMPERATE GRASSLANDS

Under existing management most grasslands in temperate regions are considered to be C sinks. Post and Kwon (2000) estimated that the land-use change from arable cropping to grassland results in an increase of soil C of 30 g C/m^2/year direct measurements of soil C suggest a C sequestration of 45–80 g C/m^2/year and Janssens *et al.* (2005) estimated average accumulation of 67 g C/m^2/year. In France, meta analysis has shown that on average, for a 0–30 cm soil depth, C sequestration reached 44 g C/m^2/year over 20 years (Soussana *et al.*, 2004). This is approximately half the rate (95 g C/m^2/s/year) at which C is lost over a 20–year period following conversion of permanent grassland to an annual crop (Soussana *et al.*, 2004).

Skinner (2008) proposed that as temperate pastures in the northeast United States are highly productive they could potentially act as significant C sinks. However, these pastures are subject to relatively high biomass removal as hay or through consumption by grazing animals. Consequently, for the first eight years after conversion from ploughed fields to pasture they were a small net sink for C at 19 g C/m^2/year but, when biomass removal and manure deposition were included to calculate net biome productivity, the pasture was a net source of 81 g C/m^2/year. The conclusion from Skinner (2008) is that heavy use of biomass produced on grasslands prevents them from becoming C sinks. Ogle, Conant and Paustian (2004) have derived grassland management factors that can be used to calculate C sequestration potential for managed grasslands in the United States and found that, over a 20–year period, changing management could sequester from 10–90 g C/m^2/year depending on the level of change.

Modelled estimates of C sequestration for the 2008–12 commitment period of the Kyoto Protocol of the United Nations Framework Convention on Climate Change for Europe (Vleeshhouwers and Verhagen, 2002) were 52 g C/m^2/year for established grassland and 144 g C/m^2/year for conversion of arable land to grassland. Country estimates varied from a source of 4.5 g C/m^2/year for Portugal to sequestration of 40.1 g C/m^2/year for Switzerland (Janssens *et al.* 2005). Bellamy *et al.* (2005) suggest a link to climate change to explain an observed mean loss of SOC of 0.6 percent/year between 1978 and 2003 in England and Wales, although Smith *et al.* (2004) have subsequently shown that, at most, 10–20 percent of the loss is attributable to climate change.

Levy *et al.* (2007) have shown, using the DNDC model to estimate the full GHG balance for grasslands across Europe, that most grassland areas are net sources for GHGs in terms of their total global warming potential because the beneficial effect of sequestering C in soils is outweighed by the emissions of N_2O from soils and CH_4 from livestock. Direct flux measurements for nine sites covering a major climatic gradient over Europe concluded that the attributed GHG balance (i.e. including off-site emissions of CO_2 and CH_4 as a result of the digestion and enteric fermentation by cattle of the cut herbage) was on average not significantly different from zero (Soussana *et al.*, 2007; Soussana, Klumpp and Tallec, 2009).

CONCLUSIONS

Assessing the potential for C sequestration requires understanding of, and quantifiable information on, the various processes and their drivers in the terrestrial C cycle. Currently, there are many gaps in our knowledge and a paucity of data available to determine precisely the amount of C accumulating from the field to the regional and global scales. At present, we hardly have enough information to infer the order of magnitude of the soil C sequestration rate, so there is still a need for more long-term experiments that follow SOC dynamics when land is either converted to permanent grassland or its management changes in order to improve our predictive capability over short- and long-term scales.

Insufficient understanding of the underlying processes limits the utility of SOM models. Therefore, concentrated efforts need to be made to acquire measured information on the critical processes of the C cycle in soils. With respect to monitoring, there is a requirement to refine methodologies for measuring both C stocks and fluxes. In experimentation, outputs from multiple-factor treatments and their interactions are required to test outputs for models. In models there is a need to reduce uncertainties to ensure that modelling also provides an essential complement to soil sampling.

In conclusion, opportunities for increasing C sequestration in temperate grasslands include: (i) moderately intensifying nutrient-poor temperate grasslands; (ii) reducing N-fertilizer inputs in intensively managed grasslands; (iii) lengthening the duration of grass leys; (iv) converting arable land to long-term or permanent pastures; and (v) converting low-diversity grasslands to high-diversity mixed grass-legume swards. However, these opportunities are unlikely to be realized until we have a more detailed understanding of the processes involved.

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