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Carbon cycling and sequestration opportunities in temperate grasslands

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Abstract. Temperate grasslands account for c. 20% of the land area in Europe. Carbon accumulation in grassland ecosystems occurs mostly below ground and changes in soil organic carbon stocks may result from land use changes (e.g. conversion of arable land to grassland) and grassland management. Grasslands also contribute to the biosphere–atmosphere exchange of non-CO_2 radiatively active trace gases, with fluxes intimately linked to management practices. In this article, we discuss the current knowledge on carbon cycling and carbon sequestration opportunities in temperate grasslands. First, from a simple two-parameter exponential model fitted to literature data, we assess soil organic carbon fluxes resulting from land use change (e.g. between arable and grassland) and from grassland management. Second, we discuss carbon fluxes within the context of farming systems, including crop–grass rotations and farm manure applications. Third, using a grassland ecosystem model (PaSim), we provide estimates of the greenhouse gas balance, in CO_2 equivalents, of pastures for a range of stocking rates and of N fertilizer applications. Finally, we consider carbon sequestration opportunities for France resulting from the restoration of grasslands and from the de-intensification of intensive livestock breeding systems. We emphasize major uncertainties concerning the magnitude and non-linearity of soil carbon stock changes in agricultural grasslands as well as the emissions of N_2O from soil and of CH_4 from grazing livestock.

Keywords: Carbon dioxide, nitrous oxide, methane, carbon cycle, greenhouse effect, climate change, grassland, carbon sequestration

INTRODUCTION

The actual global increase in atmospheric CO_2 levels has proved to be less than previously anticipated from CO_2 emission records and oceanic uptake, which has led scientists to postulate the existence of a carbon 'sink' in continental ecosystems. The demonstration of this sink has made it possible to envisage its development and use to sequester carbon and thus slow down the current rise in 'greenhouse' gases and associated greenhouse effect.

The Marrakech Accords allow biospheric carbon sinks and sources to be included in attempts to meet emission reduction targets for the first commitment period of the Kyoto Protocol. Signatory Annex I countries may take into account for their national greenhouse gas emissions any sequestration of carbon induced by 'additional human activities'. These activities principally target the storage of carbon in biomass and soil, pertaining either to afforestation and deforestation (Article 3.3 of the Protocol) or to agriculture and forestry management (Article 3.4). The carbon amounts deductible under the terms of the 'agriculture' section of Article 3.4 are not, in principle, limited; each country fixes the levels it undertakes to ensure, but their accounting is conditioned by the requirement to verify by an independent method the sequestration claimed (see P. Smith, this supplement).

The IPCC (2000a) has proposed a list of 'additional anthropogenic activities' that are liable to increase carbon stocks, and potentially eligible under the conditions of Article 3.4. These include notably the management of cultivated land, the control of erosion, the management of grasslands and of set-aside land.

Temperate grasslands account for c. 20% of land area in Europe (FAO land use statistics for year 2000). Carbon accumulation in grassland ecosystems occurs mostly below ground. Past and current land use changes (e.g. conversion of arable land to grassland) as well as the agricultural management of grasslands affect the below-ground carbon stocks. Moreover, grasslands contribute to the biosphere–atmosphere exchange of non-CO_2 radiatively active trace gases, with fluxes intimately linked to management

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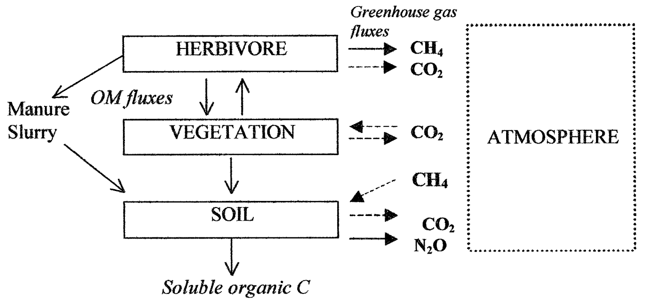

Figure 1. Schematic diagram of the greenhouse gas fluxes and main organic matter (OM) fluxes in grazed grassland. Greenhouse gas fluxes currently included (solid arrows), or not directly included (broken arrows), in the national inventories under Article 5.1 of the Kyoto Protocol (IPCC 1996a). Changes in soil carbon stocks, rather than CO_2 fluxes, are reported under Article 3.4. Methane exchanges with the soil are usually small in grassland.

practices. Of the three greenhouse gases that are exchanged by grasslands, CO_2 is exchanged with the soil and vegetation, N_2O is emitted by soils and CH_4 is emitted by livestock at grazing and can be exchanged with the soil (Figure 1). The magnitude of these greenhouse gas exchanges with the atmosphere may vary according to several factors: climate, soil, vegetation, management and global environment. Moreover, horizontal transfers of organic carbon to or from grassland plots may occur as a result of harvesting grass as silage or hay, on the one hand, and of farm manure applications on the other.

Grassland ecosystems are particularly complex and difficult to investigate because of the wide range of management and environmental conditions that they are exposed to. Currently, the net global warming potential (in terms of CO_2 equivalent) from the greenhouse gas exchanges with grasslands is not known. From Figure 1, it is clear that an integrated approach would be desirable, which would allow quantification of the fluxes from all three radiatively active trace gases (CO_2, CH_4, N_2O). However, current knowledge is scant. Management choices to reduce emissions involve important trade-offs: for example, preserving grasslands and adapting their management to improve carbon sequestration in the soil may actually increase N_2O and CH_4 emissions.

In this article, we discuss the current knowledge on carbon cycling and carbon sequestration opportunities in temperate grasslands and we point to the many uncertainties that future research will need to reduce. First, we assess changes in soil organic carbon stocks resulting from land use changes and from grassland management using a simple two-parameter model fitted to literature data. Second, we discuss the role of N_2O and CH_4 emissions from pastures and the need to account consistently for the net exchanges of CO_2 and of these two gases. Finally, we discuss carbon fluxes within the context of farming systems. We mention the main sources of uncertainty as well as the research needed to improve our understanding of carbon cycling and of carbon sequestration opportunities in temperate grasslands.

METHODS

Database for organic soil carbon stocks

A database has been assembled for assessing the carbon sequestration opportunities in French agricultural soils (INRA 2002), which consists of around 19 000 unpublished references concerning organic carbon stocks in French soils (Arrouays *et al.* 2001), and of c. 1000 additional references from the literature. This database has made it possible to produce an estimate of stocks by soil type and land-use class, then of national stocks and their regional distribution. All the data refer to the upper 30 cm of soil. This layer is supposed to account for 80–90% of the potential stock variations to be observed over decades.

Modelling organic matter stock changes

The Hénin–Dupuis model (1945) has been used to fit soil carbon stock changes calculated for 0–30 cm depth. This has a single carbon pool and two coefficients, one corresponding to the rate of conversion into humus of the organic matter added to the soil and the other to the rate of mineralization of this humus. Carbon storage by converting management A to management B is determined according to two parameters:

Δ the stock difference at equilibrium, $C_{eqB} - C_{eqA}$

k a time constant for carbon storage 'rate'.

The mean annual carbon storage flux, F, can be calculated for a duration T (in years):

$$F = \Delta[1 - \exp(-kT)] / T$$

Compared to a linear approximation, this exponential model has the following advantages:

- it is closer to the kinetics that are actually measured
- it does not create risks of overestimating the carbon storage fluxes by extrapolating the duration of the short-term fluxes for too long
- the asymmetry between two land-use changes can easily be quantified with this model.

Modelling soil carbon stock changes in a grass–crop rotation

The same simple statistical model can be applied with the following parameters:

C_0 initial carbon stock at t_0, the start of the simulation,
C_c in $\mathrm{t\,C\,ha^{-1}}$, equilibrium soil carbon stock under an annual crop monoculture,
C_g in $\mathrm{t\,C\,ha^{-1}}$, equilibrium soil carbon stock under a grassland monoculture ($C_g > C_c$).

The net accumulation rate, $C(t)$, of 'grassland' carbon after sowing a grassland (assuming a time constant k_s) can be calculated from:

$$C(t) = C_0 + (C_g - C_0)[(1 - \exp(-k_s t)] \quad (1)$$

The net decomposition rate of carbon after tilling an existing grassland and sowing an annual crop (assuming an exponential time constant k_c) can be calculated, assuming an initial carbon stock ($C_l > C_c$) at the end of the grassland phase ($t = t_l$):

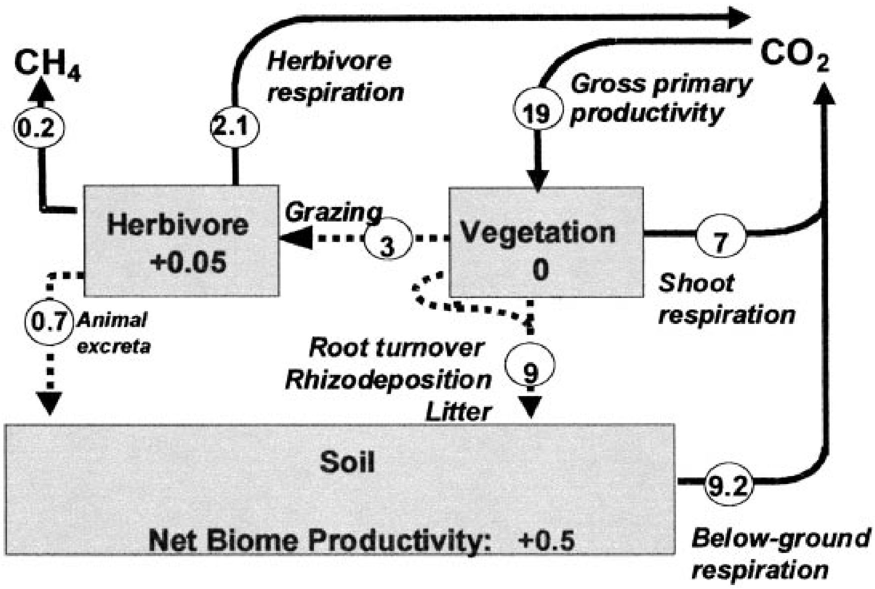

Figure 2. Carbon cycling in grazed grassland. The main carbon fluxes ($t\,C\,ha^{-1}\,yr^{-1}$) are illustrated for intensive grassland grazed continuously by cattle at an annual stocking rate of 2 livestock units per hectare.

$$C\,(t) = C_l - (C_c - C_l)\,[\exp(-k_c(t-t_l))] \tag{2}$$

Since there is no simple analytical solution to these equations when k_c differs from k_s, numerical calculations of the soil organic carbon dynamics have been performed with a one-year time step.

Modelling greenhouse gas fluxes in permanent grasslands

A mechanistic model of the grassland ecosystem (Riedo *et al.* 1998; Thornley 1998; Schmid *et al.* 2001) has been developed to predict trace gas fluxes in relationship to other important properties of grassland ecosystems. The model is fully dynamic and can be used to simulate above- and below-ground dry matter production of a perennial sward in relationship to fluxes of C, N, water, and energy. Optionally, the model can be applied for either grazing or cutting conditions. It consists of linked submodels: (1) the *plant submodel* is used to simulate shoot and root growth in relationship to C and N uptake, energy fluxes, and soil moisture conditions; (2) the *microclimate submodel* calculates canopy radiation interception and the energy balances of canopy and soil surface; (3) the *soil biology submodel* calculates plant available soil C and N, together with rates of nitrification, denitrification, mineralization and immobilization; (4) a *soil physics submodel* calculates the water and temperature of soil profiles. (5) The *animal submodel* is used to calculate intake by cattle, milk and meat production, and losses of carbon via respiration and as methane.

The model is driven by hourly data for temperature, precipitation, vapour pressure, radiation, and wind speed. Important model outputs are: above- and below-ground biomass, fluxes of CO_2, N_2O and CH_4, and total C and N in the system. The net greenhouse gas balance was calculated as equivalent CO_2-C, assuming global warming potentials over a 100-year horizon of 23 and 296 (per unit mass) for CH_4 and N_2O, respectively (IPCC 1996b). The global warming potential is defined as the cumulative radiative forcing between the present and some chosen later time 'horizon' caused by a unit mass of gas emitted now, expressed relative to the reference gas, that is, CO_2.

RESULTS AND DISCUSSION

Carbon stocks and carbon cycling in grassland soils

The annual net ecosystem production (NEP) of a temperate grassland is between 1 and 6 $t\,C\,ha^{-1}\,yr^{-1}$ according to the radiation, temperature and water regimes, as well as to the nutrient status and the age of the sward (IPCC 1996c). Nutrient and water supplies limit the potential NEP. For grasslands, the nature, frequency and intensity of disturbance plays a key role in the carbon balance. In a cutting regime, a large part of the primary production is exported from the plot as hay or silage, but part of these exports is compensated by farm manure and slurry application. Under intensive grazing (Figure 2), up to 60% of the above-ground dry matter production is ingested by domestic herbivores (Lemaire & Chapman 1996). However, this percentage can be much lower during extensive grazing. The largest part of the ingested carbon is digestible (up to 75% for highly digestible forages) and, hence, is respired shortly after intake. Only a small fraction of the ingested carbon is accumulated in the body of domestic herbivores or is exported as milk. Large herbivores, such as cows, respire approximately one tonne of carbon per year (Vermorel 1995). Additional carbon losses (c. 5% of the digestible carbon) occur through methane emissions from the enteric fermentation. The non-digestible carbon (25–40% of the intake according to the digestibility of the grazed herbage) is returned to the pasture in excreta (mainly as faeces)

(Figure 2). In most European husbandry systems, the herbage digestibility tends to be maximized by agricultural practices such as frequent grazing and use of highly digestible forage cultivars. Consequently, the primary factor that modifies the carbon flux returned to the soil by excreta is the grazing pressure, which varies with the annual stocking rate (i.e. mean number of livestock units per unit area). Secondary effects of grazing on the carbon cycle of a pasture include: (i) the role of excretal returns (concentrated in patches) in SOM mineralization and N cycling, especially in nutrient-poor grasslands; (ii) the role of defoliation by animals and of treading, both of which reduce the leaf area and the canopy photosynthesis.

Soil carbon stocks display a high spatial variability (coefficient of variation of 50%; Cannell *et al.* 1999) in grassland as compared to arable land, and c. 15% of this variability comes from sampling to different depths (Robles & Burke 1998; Chevallier *et al.* 2000; Bird *et al.* 2002). According to Conant *et al.* (2001), in a recent review of soil carbon changes below temperate and tropical grasslands, a major factor accounting for changes in SOM content is the climate, because it affects differently the net primary productivity and the soil N mineralization.

The initial soil carbon content also accounts for part of the variability by being negatively correlated to the carbon stock change (Conant *et al.* 2001). By contrast, the soil texture does not seem to explain the variability between the different values of soil carbon contents observed. This last point is unexpected since numerous studies have shown a strong positive relationship between the soil carbon stocks and the fraction of clay or of clay plus fine silt (0–20 μm) (Parton *et al.* 1987). Moreover, Reeder *et al.* (1998) have observed greater carbon storage after conversion from arable to grassland in a sandy soil compared with a clay soil. Therefore, more knowledge is needed concerning the role of texture in the response of soil carbon.

Organic matter is partly incorporated in grassland soils through rhizodeposition (Wood *et al.* 1991). This process favours carbon storage (Balesdent & Balabane 1996), because direct incorporation into the soil matrix allows a high degree of physical stabilization of the soil organic matter. Root turnover creates the largest organic carbon input to grassland soils and favours soil carbon storage, since root litter contains lignin and polyphenols which tend to be recalcitrant to degradation. Moreover, the soil organic matter is richer in aromatic compounds under a grassland than under a cereal monoculture, which confers on it a greater ability to resist degradation (Gregorich *et al.* 2001). After grassland establishment, roots and their associated microflora (bacteria and fungi) tend to stabilize the soil aggregates (Jastrow 1996). Therefore three reasons explain a greater carbon sequestration in grasslands than in arable soils: (i) a greater part of the soil organic matter input from root turnover and rhizodeposition is physically protected as particulate organic matter (POM); (ii) a greater part of this POM is chemically stabilized; and (iii) aggregates tend to protect the native soil organic matter from decomposition (Balesdent *et al.* 2000).

Table 1. Fitted values of the Hénin–Dupuis model for land use changes between arable and grassland (see also Figure 3).[a]

	Δ ($t\,C\,ha^{-1}$)	k	$\Delta\ k$	F_{20} ($t\,C\,ha^{-1}\,yr^{-1}$)
Grassland to arable	-25 ± 7	0.07 ± 0.01	-1.7 ± 0.8	-0.95 ± 0.3
Arable to grassland	25 ± 7	0.025 ± 0.01	0.63 ± 0.36	0.49 ± 0.26

[a]Carbon storage is determined according to two parameters: the magnitude Δ, which is the stock difference at equilibrium ($C_{eqB} - C_{eqA}$), and a time constant (k). F_{20} is the average carbon storage flux during a 20-year period after the start of the land use change. Uncertainties are 95% confidence intervals on the estimated value.

Average soil carbon stocks in grasslands compared to other land use types

Average soil (0–30 cm) organic carbon stocks range from 30 to 90 $t\,C\,ha^{-1}$ in France, and can be broken down into three groups according to land use (Arrouays *et al.* 2001):

- land under annual crops and perennial crops with bare soil, where stocks are less than 45 $t\,C\,ha^{-1}$;
- land under permanent grassland and forests (excluding litter) with average stocks of nearly 70 $t\,C\,ha^{-1}$;
- land under high altitude pastures and in wetlands, where average stocks reach 90 $t\,C\,ha^{-1}$ due to low temperatures and the effect of anoxia on carbon mineralization, respectively.

Based on this inventory, the average difference between cropland and pasture in lowlands is c. 25 $t\,C\,ha^{-1}$. This value represents mean carbon change involved in land use conversions.

Effects of land use changes to or from grasslands on soil carbon stocks

Grassland versus arable land. The conversion of grasslands to arable use has led to a 25–43% decline in soil carbon stocks in the uppermost 120 cm in the USA, as compared to native grassland (Potter *et al.* 2000). A well documented chronosequence in France has yielded similar results (Boiffin & Fleury 1974). The mean carbon change induced each year by converting a permanent grassland to an annual crop can reach $-0.95 \pm 0.3\ t\,C\,ha^{-1}\,yr^{-1}$ over a 20-year period (Table 1 and Figure 3).

A 3-year period of bare fallow induced mean soil carbon losses of 1.7, 2.8 and 3.2 $t\,C\,ha^{-1}\,yr^{-1}$ following an annual crop, a sown grassland and a permanent grassland, respectively (Loiseau *et al.* 1996). Hence, carbon losses tend to increase with the duration of the previous grassland phase.

According to IPCC (2000a), the conversion of arable land to grassland resulted in a 0.5 $t\,C\ ha^{-1}\,yr^{-1}$ average carbon storage over 50 years, with a range of 0.3–0.8 $t\,C\,ha^{-1}\,yr^{-1}$. Another meta-analysis (INRA 2002) showed that, on average, for a 0–30 cm soil depth, carbon storage reached $0.49 \pm 0.26\ t\,C\,ha^{-1}\,yr^{-1}$ over 20 years (Table 1).

This rate of increase of soil carbon after conversion to grassland is slow. After 50 years, the soil carbon content did not regain the level it had reached before the grass was first ploughed up (Burke *et al.* 1995; Rasmussen *et al.* 1998). Because of this slow accumulation, Franck (2002) considers that grassland of more than 20 years old no longer acts as a carbon sink. The average time constant of carbon storage

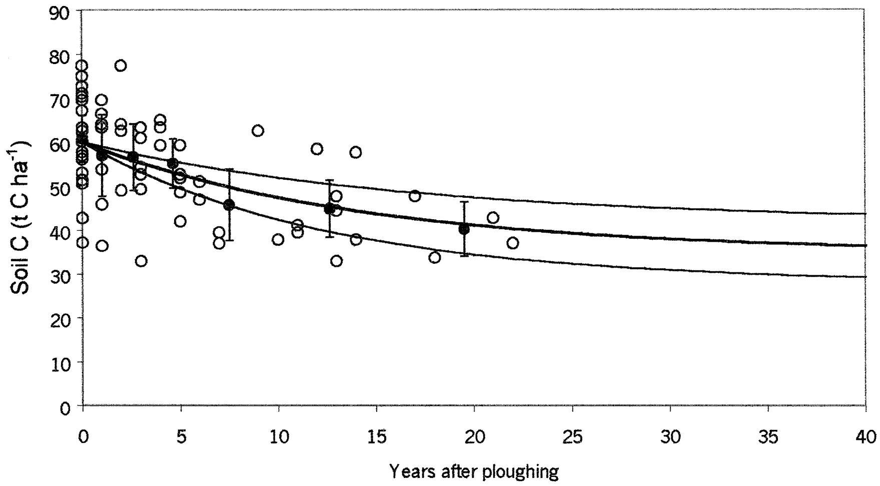

Figure 3. Carbon stocks in arable soils (t C ha^{-1} in the top soil, 0–22 cm) after ploughing up permanent grassland: ○, experimental data; ●, means and standard error. Fitted model (middle line) and confidence interval at 95%. Fitted values: $\Delta = -25$ t C ha^{-1}; $k = 0.07$ yr^{-1}.

(0.025 ± 0.1 yr^{-1}), according to the fitted model, is less than half that of the carbon release rate after ploughing (Table 1). Indeed, after 6 years of cultivation, Reeder *et al.* (1998) observed that soil carbon stocks had already reached the low values found after 60 years of cultivation. Hence carbon losses are much faster after returning grassland to arable use than the build-up of soil carbon when establishing grassland.

The increase in soil carbon content after a shift from arable to grassland is partly explained by a greater supply of carbon to the soil under grass, mainly from the roots but also from shoot litter, and partly by the increased residence time of carbon due to the absence of tillage. Carbon losses after tillage reduce the degree of physical protection of the organic matter, resulting in a decrease of the humified soil organic matter fraction (Post & Kwon 2000). An increase of the soil disturbance caused by tillage increases the turnover of aggregates and accelerates the decomposition of soil organic matter within aggregates (Paustian *et al.* 2000). After the establishment of grassland on an arable soil, a continuous vegetation cover, and hence continuous protection of the soil organic matter, contributes to increased soil carbon storage.

Grassland versus forest. Converting grassland to forest can lead to an accumulation or to a release of soil carbon, depending on the conditions (Post & Kwon 2000). In some favourable conditions, for example, clay or calcareous soils in a mountain climate, an average accumulation between 0.1 and 0.2 t C ha^{-1} yr^{-1} in 30 cm depth has been reported by Moares *et al.* (2001) following afforestation of 200-year-old grassland. However, this carbon storage can only be detected per unit mineral mass of the soil. In andosols, Ross *et al.* (1999) have also measured strong accumulation rates of 0.2 t C ha^{-1} yr^{-1} over 25 years and 0.12 t C ha^{-1} yr^{-1} over 200 years. However, under less favourable conditions, that is, warmer climate, sandy or acidic soils, a carbon loss has been measured after the conversion of grassland or moorland to forest (Aggangan *et al.* 1998; Compton & Boone 2000; Franzluebbers *et al.* 2000). This literature survey leads us to propose small average carbon storage rates of 0.1 ± 0.02 t C ha^{-1} yr^{-1} over 20 cm during 90 years (INRA 2002). There is very little information on the effects of deforestation followed by the establishment of pastures or grasslands for temperate zones. In New Zealand, the establishment of grasslands on old degraded forest soils has allowed an increase in the soil carbon stocks (Haynes & Williams 1993). Because of a lack of appropriate information, carbon stock changes after converting forest to pasture may be considered to be symmetric to the inverse change, but with a higher degree of uncertainty: –0.1 ± 0.1 t C ha^{-1} (INRA 2002).

Short duration leys (sown grasslands). As a result of periodic tillage and resowing, short duration grassland tends to have a potential for soil carbon storage intermediate between crops and permanent grassland. Part of the additional carbon stored in the soil during the grassland phase is released when the grassland is ploughed up. The mean carbon storage increases in line with prolonging the lifespan of plant cover, that is, less frequent ploughing.

Loiseau *et al.* (1996) studied carbon losses from sandy soils by comparing a permanent grassland, a crop–ley system (11 years and 9 years annual crop) and annual cropping systems. After 20 years, soil organic carbon stocks reached 24, 31 and 38 t C ha^{-1} for the arable, crop–ley and permanent grassland systems, respectively. Hence, introducing a ley into the rotation increased the soil carbon stock by 7 t C ha^{-1} after 20 years, which is approximately half the increase in soil carbon stock when arable is changed to permanent grassland (Loiseau *et al.* 1996). Establishing grassland for 3 years in a crop rotation leads to the additional storage of 3.5 t C ha^{-1} within 9 years (Lubet & Juste 1979). This carbon storage potential is, however, strongly affected by the type of grassland management (see below).

Grassland management and ley farming systems

To our knowledge, the effects of grassland management on carbon accumulation in soils have not yet been reviewed. To make some progress, based on an expert assessment for

France (INRA 2002), we try to identify some of the possible trends in soil carbon caused by changes in grassland management. For France, different types of grassland have been classified according to their mean organic carbon stock (INRA 2002). Grassland types (Table 2) were defined according to the vegetation type (A to D) and to the nitrogen status of the vegetation, estimated by the nitrogen nutrition index method of Lemaire & Gastal (1997). As sown grasslands usually have a greater nitrogen status than permanent grasslands, the most frequent types occur in the table along the diagonal. The following types have been described:

A3 – Short duration leys which are highly fertilized and used for cutting (>400 kg N ha^{-1} yr^{-1}) or grazing (>150 kg N ha^{-1} yr^{-1}) regimes. Losses of inorganic N are relatively large.

A2 - Short duration leys which are managed less intensively by cutting (<400 kg N ha^{-1} yr^{-1}) or grazing (<150 kg N ha^{-1} yr^{-1}) and which display a higher C/N ratio and, hence, less nitrogen losses than the A3 type.

B2 – Grass–legume mixtures or legume monocultures (e.g. lucerne) which are not supplied with N fertilizer. Nitrogen inputs to the soil tend to be self-regulated by ecological processes such as the inhibition of symbiotic nitrogen fixation, by soil inorganic N and the competitive decline of legumes under conditions of high soil N availability (Loiseau *et al.* 2002).

C2/D2 – Grasslands, both sown or permanent, which are intensively managed for silage cuts or by intensive grazing (>1.5 LSU ha^{-1} yr^{-1}).

C1/D1 – Species rich grasslands which have been extensively managed for hay production or for extensive grazing (<1.5 LSU ha^{-1} yr^{-1}).

D0 – Nutrient-poor grasslands and moorlands developed on organic soils at medium or high altitude. These grasslands are grazed with a low stocking rate (<0.8 LSU ha^{-1} yr^{-1}), without N fertilizer supply and are often invaded by shrubs or coniferous tree species.

Table 2. A typology of the most common grassland types in France, which has been developed to classify grasslands according to their mean soil carbon stock (INRA 2002). This typology is based on the type of vegetation cover (A to D) and on the mean nitrogen status of the grassland vegetation.

Type of grassland	Duration (yr)	Nitrogen nutrition index[a]			
		0.4–0.6	0.6–0.8	0.8–0.9	>0.9
A. Short duration grass leys	1–2	–	–	A2	A3
B. Legume based leys	3–6	–	–	B2	–
C. Sown intensive grasslands	3–15	–	C1	C2	–
D. Permanent grasslands	>15	D0[b]	D1	D2	–

[a]Values are between 0 and 1, calculated from the shoot nitrogen concentration and the standing biomass (Lemaire & Gastal 1997). [b]These permanent grasslands are found at medium to high altitude.

Carbon stocks cannot be estimated in the grass–ley system without taking into account the effects of the rotation between a crop and a sown grassland. Hence, mean soil carbon stocks were calculated at the end of the grassland and at the end of the crop stages for contrasting ley systems (Table 3). The values for time constants in Table 3 have been estimated from a previous assessment of humus types and soil organic matter inputs in contrasting grassland soils (INRA 2002).

According to these estimates, the grassland management strongly affects the soil carbon stocks. Moderate N fertilizer use increases the organic carbon input to the soil more than the soil C mineralization. Intensive N fertilizer use induces not only a rise in production, but also accelerates mineralization and enhances decomposition of soil organic matter (Loiseau & Soussana 1999) and, hence, reduces soil carbon stocks. Practices that enhance carbon stock involve a reduction in the intensification of highly fertilized grasslands and a moderate intensification of poor grasslands. However, mountain pastures and wetlands developed on organic soils should be excluded from the latter practice, because they are naturally endowed with high carbon levels which intensification could reduce by 1 t C ha^{-1} yr^{-1} (Loiseau *et al.* 1996). Under conditions prevailing in Europe, grazing, which stimulates primary production, often increases carbon accumulation compared with cutting.

Table 3. Simulated soil organic carbon stocks at the end of the grassland phase, at the end of the annual crop phase and on average for contrasted ley farming systems.

Grassland type	Ley system[a]	Carbon stocks (t C ha^{-1})			Time constant[b] (yr^{-1})	
		End of grassland	End of crop	Average	Start of grassland	End of grassland
A3	1/1	34.6	34.6	34.6	0.035	0.040
A2	2/1	44.3	44.9	44.6	0.035	0.040
B2	5/2	55.0	57.1	56.0	0.030	0.040
C2	5/2	53.4	55.3	54.3	0.025	0.040
C1	5/2	41.4	42.2	41.8	0.020	0.035
C2	10/2	59.0	61.2	60.1	0.025	0.040
C1	10/2	44.4	45.3	44.6	0.020	0.035
D2	PG	–	–	70	0.025	0.040
D1	PG	–	–	50	0.020	0.035
D0	PG	–	–	110	0.010	0.035

[a]Ley farming systems were constructed by assuming a variable duration (left number, col 2) of the grassland phase with a given grassland type (see Table 2), followed by a variable duration (right number, col 2) of the annual crop phase; PG, permanent grassland. [b]The time constant values were obtained by fitting the model to data of soil carbon stock changes (INRA 2002). Simulations were run using the simple statistical model described in the Methods section. The uncertainty in these estimates is evaluated at $\pm$ 0.25 t C ha^{-1} yr^{-1}. For other abbreviations, see Table 2.

Table 4. Soil carbon sequestration as affected by grassland management options.[a]

	Initial type	Final type	Mean SOC stock after 20 yr ($t\,C\,ha^{-1}$)	SOC change ($t\,C\,ha^{-1}$)	C stock rate of change ($t\,C\,ha^{-1}\,yr^{-1}$)
Reduction of N fertilizer input	A3	A2	41.4	6.4	0.3
Conversion to grass–legume mixtures	A3	B2	45.6	10.2	0.5
	A2	B2	50.3	6.3	0.3
Intensification of permanent grassland	D1	D2	53.9	3.9	0.2
Intensification of nutrient poor grassland (organic soils)	D0	D2	87.4	–22.6	–1.1
	D0	C2 (10/2)	91.3	–18.7	–0.9
Permanent grassland to medium duration leys	D2	C2 (10/2)	67,0	–3,0	–0,2
Increasing the duration of leys	C2 (5/2)	C2 (10/2)	58.1	3.9	0.2
	C1 (5/2)	C2 (10/2)	50.9	9.1	0.5
Short duration leys to permanent grassland	C2 (5/2)	D2	60	5.7	0.3
	C1 (5/2)	D2	80	8.2	0.4

[a]The effects on the soil carbon stocks after 20 years of changes in grassland management have been simulated with a 1-year time step using the values of soil carbon stocks and the time constants in Table 3 and the model described in the Methods section. SOC, soil organic carbon; other abbreviations as in Table 2.

Some of the possible soil carbon sequestration opportunities for grasslands have been calculated and compared (Table 4) for 20-year time periods, by using the simple model described in the Methods section with the time constants and carbon stocks values displayed in Table 3. According to these estimates, annual carbon storage rates between 0.2 and 0.5 $t\,C\,ha^{-1}\,yr^{-1}$ are obtained for a range of options, which seem compatible with gradual changes in the forage production systems, namely:

- reducing N fertilizer inputs in highly intensive grass leys;
- increasing the duration of grass leys;
- converting these leys to grass–legume mixtures or to permanent grasslands;
- moderately intensifying nutrient-poor permanent grasslands.

By contrast, the intensification of nutrient-poor grasslands developed on organic soils leads to large carbon losses, and the conversion of permanent grasslands to leys of medium duration is also conducive to the release of soil carbon. Nevertheless, the uncertainties concerning the estimated values of carbon storage or release after a change in grassland management are still very high (estimated at 0.25 $t\,C\,ha^{-1}\,yr^{-1}$). Clearly, further work will be needed to ascertain the direction and magnitude of the soil carbon stock changes resulting from changes in grassland management. Moreover, it should be underlined that the values calculated here are mean values for French soils and cannot be extrapolated without caution to other temperate grassland areas, which may have different soil and climate conditions.

Manures and livestock wastes

In contrast to arable farming, most livestock systems generate large amounts of manure, which are currently spread on both grassland and arable land. Most mixed farming systems, which produce farmyard manure, involve carbon transfer from grassland to annual crops, because farmyard manure spreading is considered to interfere with grazing. To a lesser extent, most of the pure grassland farming systems, which produce slurry, involve a carbon transfer from pastures (and from external inputs) to meadows cut for silage and hay. These C-rich farm manures vary a lot with the cropping system (straw production), the type of animal, the housing system and the waste management system. Farm manures are a large source of organic carbon, currently estimated at 22.7 million $t\,C\,yr^{-1}$ in France (Table 5). Spread on grassland, they help to maintain or to increase the soil carbon stocks. When manures are not returned to grassland, the negative consequences for the N cycle and for primary production are generally compensated by mineral fertilizer and by symbiotic N fixation.

Table 5. Annual carbon production from livestock farm wastes in France. (Data from ADEME 2001; IFEN 2002.)

	Annual production (10^3 t FW)	Dry weight (% FW)	Organic carbon (% DW)	Total organic carbon (10^3 t C)
Livestock farm wastes	275 000			
Of which 86% are manures	236 000	20 (18–22)	48	22 656

DW, dry weight; FW, fresh weight.

The yield in soil organic matter (K_1 coefficient in the Hénin–Dupuis model, being the ratio of the quantity of carbon entering the soil organic carbon cycle to the quantity of organic carbon applied) has often been evaluated for different organic matter inputs, soil types and for various locations in Europe. Apart from some of the older values obtained by Hénin & Dupuis (1945), the organic matter yields of manures seem to be between 0.2 and 0.4 (Table 6). However, when manures are already partly composted, the K_1 coefficient increases from 0.25 to about 0.5 (Rémy & Marin-Laflèche, 1976). During composting, the more degradable organic compounds are decomposed and the residual compounds, which tend to have a longer life span, increase in concentration. In one study, cumulative carbon losses during storage and after incubation in the soil accounted for 60 and 54% of carbon initially present in composted and anaerobically stored manure, respectively (Thomsen & Olesen 2000).

Table 6. Values of the K_1 coefficient for cattle manure in the Hénin-Dupuis model.

	Crop	K_1[a]	Location, soil type
Hénin & Dupuis 1945	Wheat–Barley	0.10 – 0.70	Rothamsted, UK; Versailles, France
Morel *et al.* 1984	Wheat–Sugar beet	0.25–0.30	Grignon, France clay-loam carbonate
Morel *et al.* 1981	Bare soil	0.2	Grignon, France clay-loam calcareous
Jenkinson & Rayner 1977	Barley	0.2	Hoosfield, Rothamsted, UK
Persson & Kirchmann 1994 Gerzabek *et al.* 1997	Cereals, Rapeseed	0.27	Uppsala, Sweden Clay loam, pH 6.6 Eutric Cambisol
Boiffin *et al.* 1986	Annual crops	0.30	Noyon, France, various soil types
Delas & Molot 1983	Maize, Potato	0.32	Bordeaux region, Grave, sandy soil, alkaline pH

[a]K_1 is the ratio of the amount of carbon entering the soil organic carbon single compartment to the amount of organic carbon applied. The storage (S) in the soil of carbon from manures per unit carbon applied (exponential calculation 20 years after application) can be calculated from the value of K_1 according to the following formula: $S = K_1 / (20\,k)\,[1 - \exp(-20\,k)]$, where $k = 0.025\ \mathrm{yr}^{-1}$.

By assuming a K_1 value of 0.30 for the manure production in France (P, 23 million tonnes) and a mineralization rate constant (k) for SOC of $0.025\ \mathrm{yr}^{-1}$, the annual carbon storage *(PS)* from livestock farm wastes can be calculated (Table 6) as:

$$PS = PK_1\ /\ (20\,k)\ [1 - \exp(-20\,k)] = 5.4\ \mathrm{Mt\,C\,yr}^{-1}$$

Since similar values of the K_1 coefficient have been found for different soil types and for different crops and grasslands, it is unlikely that the application of farm manures to arable soils can increase the soil carbon stock to a greater extent than application to grasslands. Smith *et al.* (2000a) have hypothesized that the residence time of organic carbon is greater in arable soils than in grassland and, hence, that manures offer a large carbon sequestration potential which can be realized by applying them to arable land. However, we are not aware of experimental data supporting this hypothesis.

Greenhouse gas balance of grasslands

In addition to CO_2, two other trace gases play an important role for the net greenhouse gas exchanges in grassland ecosystems. Nitrous oxide has a global warming potential which is 300 times greater than that of CO_2 over a 100-year horizon and, at a global scale, soils account for 65% of N_2O emissions (IPCC 1996b). This gas is produced during both denitrification and nitrification in soils (see Smith and Conen, this supplement). For given soil and climate conditions, N_2O emissions are likely to scale with the nitrogen fertilizer inputs. Therefore, the current IPCC (1996a) methodology assumes a default emission factor of c. 0.01 emitted as N_2O per unit nitrogen input N (0.0025 - 0.0225 kg N_2O-N kg N input^{-1}). The emission factor of nitrogen in grazed grassland is higher (0.031 kg N_2O-N kg N input^{-1}) than for a cut grassland supplied with mineral fertilizer (Skiba *et al.* 1996).

Methane has a global warming potential that is 23 times that of CO_2 over a 100-year horizon. The emission of CH_4 by ruminants due to enteric fermentation contributes between 16 and 23% of the global emissions of this gas (IPCC 1996b). The annual emissions of CH_4 originating from enteric fermentation are typically between 80 and 100 kg animal^{-1} yr^{-1} for dairy cattle in Europe (IPCC 1996a), leading to annual emissions equivalent to 0.67–0.84 t C per animal as CO_2 equivalent. These amounts depend upon the type, age and weight of the animal and the quantity and quality of the feed consumed (IPCC 1996a). Under grazing conditions, most of the variability in the enteric CH_4 production lies in the number of animals, and therefore the emissions per unit land area will primarily vary with the stocking rate. Although grassland soils are likely to be a small sink for CH_4, the sink strength may be dependent on management.

Currently, the net global warming potential (in terms of CO_2 equivalent) arising from the greenhouse gas exchanges with grassland is not known. An integrated approach is needed which would allow the fluxes of all three trace gases (CO_2, CH_4, N_2O) to be quantified (Figure 1). To make progress, we have used a simulation model to estimate this greenhouse gas budget under contrasting management conditions.

Modelling the greenhouse gas exchanges of a permanent grassland

The greenhouse gas (GHG) balance of a permanent grassland site in the French Massif Central (Laqueuille 1000 m a.s.l.) was simulated with the PaSim model for contrasted grazing and fertilizer management scenarios. The model was first brought near to equilibrium using soil properties and meteorological data from the site and a typical grassland management for the site, namely, four cuts per year (beginning of May, mid-June, mid-August and mid-October), 150 kg N ha^{-1} yr^{-1} of inorganic N (applied in three equal parts) and 50 kg N ha^{-1} (in March) of organic N. The same meteorological data were used in loop in order to avoid the effects of interannual climate variability on the output data. At the end of the equilibrium run, the GHG budget showed a carbon storage of 3.8 C ha^{-1} yr^{-1}.

Five years of simulation were then started from this equilibrium state. The grassland was assumed to be grazed continuously by cattle during the 5 months between DOY 141 and 292. Two scenarios were compared: 150 kg N ha^{-1} yr^{-1} split in three applications (mid-May, mid-July, end August) and no fertilizer N. Different mean annual stocking rates were also compared (from 0 to 2.4 livestock units [LSU] ha^{-1}, with a 0.4 step). However, the

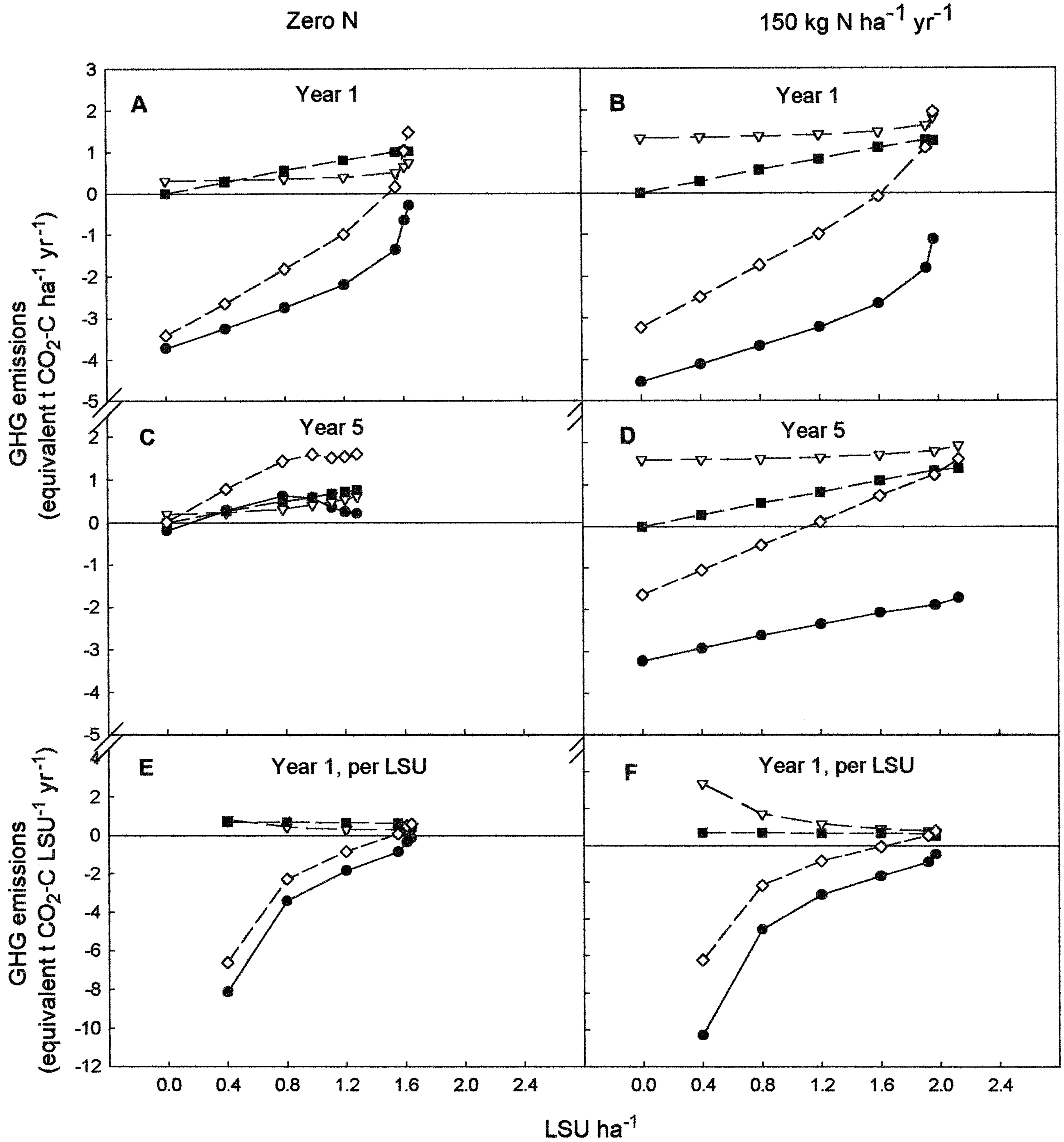

Figure 4. Simulated net greenhouse gas emissions and their balance in CO_2-C equivalents of permanent grassland continuously grazed by cattle at a range of mean annual stocking rates (in livestock units [LSU] per hectare) during the first (A, B, E, F) and the fifth (C, D) year after grazing started. Two fertilization scenarios were considered: no-fertilizer (left hand panels) and with 150 kg N ha^{-1} yr^{-1} (right hand panels). Emissions were expressed either on a grassland surface area basis (in t CO_2-C equivalents ha^{-1} yr^{-1}) (A, B, C, D) or on a livestock unit basis (in t CO_2-C equivalents LSU^{-1}) (E, F). A positive value denotes a greenhouse gas source to the atmosphere and a negative value denotes a sink. Key to symbols: CO_2 (●), N_2O (▽), CH_4 (■), greenhouse gas balance (◇).

simulated stocking density was reduced whenever the daily cattle intake dropped below a threshold (13.5 kg DM LSU^{-1} day^{-1}) as a result of insufficient herbage available. Thus, the simulated mean annual stocking rate was found to be less than 2.4 LSU ha^{-1} (Figure 4).

The daily output data from the PaSim model included: net canopy photosynthesis; soil, plant and animal respiration; methane emission by cattle; and N_2O emission from soil. The net annual emission of CO_2, N_2O and CH_4 and their balances were computed from these data in CO_2-C equivalents and expressed either per unit land area (Figure 4A–D) or per livestock unit (Figure 4E, F).

Irrespective of N input, the carbon balance (i.e. the net biome productivity) of the simulated grassland was negative during the first year of grazing, indicating a sink for atmospheric CO_2 (Figure 4A, B). However, the magnitude of this sink declined with the mean annual stocking rate, while the CH_4 emissions increased proportionally to stocking rate. The N_2O emissions were greater with N fertilizer than without it, but were little affected by the stocking rate. For mean annual stocking rates of less than 1.6 LSU ha^{-1}, the simulated net balance of the GHG emissions in CO_2 equivalents was always negative during the first year, denoting sink activity.

After 5 years of grazing, the simulated GHG emissions displayed the same trends as during the first year of grazing in the N fertilized grassland (Figure 4D). By contrast, the unfertilized grassland displayed a strong decline in herbage growth (data not shown) and a positive or nil carbon balance (Figure 4C). The simulated net balance of GHG emissions in CO_2 equivalents was negative (sink) for the fertilized grassland below 1.2 LSU ha^{-1}, but was always positive or nil for the unfertilized grassland.

GHG fluxes were also calculated for the first year per unit of livestock (Figure 4E, F). The simulated CH_4 emissions per livestock unit were approximately constant. By contrast, the magnitude of the CO_2 fluxes per livestock unit declined strongly as the stocking rate increased. Therefore, the GHG budget per unit livestock was close to zero at high stocking rates (Figure 4E, F).

A sustained GHG sink activity after 5 years was simulated for low stocking rates and N fertilizer inputs (Figure 4D). The past land management also affected the GHG fluxes, as can be seen from the difference in emissions between year 1 (after several years of cutting) and year 5 (after 4 years grazing). Finally, when expressed per livestock unit the GHG balance appears to be small but usually positive (source) at high stocking rates, while a negative GHG balance (sink) is obtained with low stocking rates (Figure 4E, F).

Such model predictions need to be validated with experimental data. Therefore, it is not possible at present to generalize from the conclusions of this modelling study.

Carbon sequestration opportunities in temperate grasslands: a case study in France

Farms raising domestic herbivores occupy two-thirds of farmland in France, and 60% of all commercial farms raise some herbivores (Bontron *et al.* 2001). However, grasslands, which occupied about 25% of the total land area in 1970, have declined markedly in favour of arable use, including growing of fodder crops such as maize, silage and fallow.

Restoration of permanent grasslands. Restoration over 20 years of half the amount of land under permanent grass lost since the 1970s would give a mean annual increase of 90 000 ha, which would lead to an estimated increase of 16 Mt of soil organic carbon during this period (INRA 2002). This is, however, equivalent to only 10% of the annual CO_2 emissions from fossil fuels in France (INRA 2002). Moreover, such an increase in the grassland area would imply major changes to livestock breeding systems and grassland management. In addition, the consequences on the emissions of other GHGs (CH_4 and N_2O) are still unknown.

Extensification of livestock production. Herds of domestic herbivores are still generally feeding on grasslands, which occupy more than 80% of all land put to fodder crops (15% for maize silage). However, the proportion of grass differs considerably as a function of the animal production system: it reaches nearly 95% in cattle farms which breed for meat production, but is much smaller in, for example, the 49 000 dairy farms which produce half of all French milk and 20% of French meat using much more intensive production systems (with 41% of maize silage and a stocking rate of 1.7 LSU ha^{-1}) (Bontron *et al.* 2001).

The extensification of intensive livestock production provides an interesting option to enhance soil carbon accumulation by increasing the proportion of grass in the diet. This would involve conversion of annual fodder or cereal crops into temporary grassland, and conversion of temporary grassland into permanent grassland. Such extensification could also reduce the CH_4 emissions per unit land area, because of a lower stocking rate, and the N_2O emissions through the limitation of nitrogen inputs. However, it is necessary to take account of the emission coefficient associated with symbiotic nitrogen fixation by leguminous crops, which current estimates (IPCC 1996a) equate with the coefficient for nitrogen fertilizer. From an agronomic point of view, several options can be proposed to increase and optimize the use of grass in livestock breeding systems in order to store more C: (i) an increase in the length of the grazing season, (ii) the use of deferred grazing, (iii) lowering the costs of production by using grass–legume mixtures or permanent grassland, (iv) more efficient use of livestock manures.

CONCLUSIONS

The kinetics of carbon accumulation following change in land use or in grassland management are:

1. Non-linear. They are more rapid during the early years after adopting a practice that enhances accumulation. If practices remain constant, the stocks tend to reach a new equilibrium, in which the carbon input is equal to the carbon released by the mineralization of organic matter.
2. Asymmetric. For example, over a period of 20 years, the accumulation arising from a conversion from arable to grassland is on average half that of the release induced by conversion from grassland to arable.

These characteristics have several practical consequences.

- Any estimate of carbon storage must be strictly referred both to the previous management and to the current management.
- Rates of carbon sequestration expressed in t C ha^{-1} yr^{-1}, are highly dependent upon the duration to which they apply.

- At equilibrium, accumulation of carbon no longer increases, but stock conservation requires maintenance of the practices which enabled its accumulation.
- The cessation or temporary interruption of stock-enhancing practices usually results in a rapid release. If interruptions prove necessary, the carbon stocks claimed should be revised downwards.
- Annual pool fluxes are not independent of the history of the plot. It is therefore not possible to evaluate carbon stocks that have accrued between two dates based solely on measuring the surface area concerned in changes in management occurring over that period.

Farm manures contribute to maintain or increase the soil carbon stocks of grassland and it seems unlikely that the magnitude of the soil carbon storage resulting from farm manure application is greater on arable soils than on grassland.

Simulation results show that the greenhouse gas balance of pastures may differ by several tonnes of CO_2-C equivalent per hectare from the carbon balance (i.e. from the net biome productivity). Moreover, it appears that grazing management and N fertilization may to a large extent alter the greenhouse gas balance of permanent grasslands. A sustained greenhouse gas sink activity over a 5-year period was simulated only for low stocking rates with moderate N fertilizer inputs.

Preliminary results show that the restoration of grasslands and the extensification of intensive herbivore breeding systems are likely to provide carbon sequestration opportunities. Nevertheless, to quantify potential carbon sequestration by grasslands at a national or regional level, a global assessment of the 'grasslands–livestock farming' sector is necessary. This should in particular include emissions of CH_4 by ruminants, which make an important contribution to agricultural emissions of greenhouse gases, and emissions of N_2O associated with grazing and with symbiotic nitrogen fixation. This assessment will require specific research efforts.

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