Biophysical Aspects of Carbon Sequestration in Drylands

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Introduction

The process of carbon sequestration, or flux of carbon, into soils forms part of the global carbon cycle. Movement of carbon between the soil and the above ground environment is bi-directional and consequently carbon storage in soils reflects the balance between the opposing processes of accumulation and loss. This reservoir of soil carbon is truly dynamic, not only is carbon continually entering and leaving the soil, the soil carbon itself is partitioned between several pools, the residence times of which, span several orders of magnitude. Neither is soil carbon an inert reservoir, the organic matter with which it is associated is vital for maintaining soil fertility and it plays a part in such varied phenomena as nutrient cycling and gaseous emissions. A detailed description and analysis of soil carbon and organic matter can be found elsewhere (Schnitzer, 1991; FAO, 2001).

The quantity of carbon stored in soils is highly significant; soils contain approximately three times more carbon than vegetation and twice as much as that which is present in the atmosphere (Batjes & Sombroek, 1997). Of concern here, is the fact that various environmental factors, not least human dependent land management, can significantly affect the dynamic equilibrium that controls the size of the soil carbon pool.

The quantity of carbon within any soil has been described by (Reicosky, 1997) as a simple mass balance relationship:

carbon input – carbon output = net carbon accumulation

Many of the factors determining carbon input and output are influenced, intentionally or unintentionally, by land management practices. There is consequently much scope for agro-ecological processes to influence carbon input and output from soils. To promote carbon sequestration human activity needs to maximise the inputs and minimise the outputs. However, if carbon sequestration in soils is to be used to offset gaseous emissions of CO_2, it is necessary to look beyond the fluxes of carbon directly into and out of the soil. The carbon budget for the whole land management system must be considered. Where this system boundary is drawn may be as contentious as it is difficult to define. From a purely carbon accounting point of view, the system should envelope all aspects that are involved in soil management.

Specific Features of Drylands

Dryland environments are characterised by a unique set of features and these will impact on particular aspects of carbon sequestration. By definition, lack of water is the key aspect, and this is accompanied in many dryland regions by the occurrence of higher temperatures at some time in the diurnal or annual cycle. The deficiency of water in drylands severely constrains plant productivity that provides the ultimate source of soil carbon. Additionally, the size of soil organic matter pools in natural ecosystems decreases exponentially with temperature (Lal, 2002a) and consequently most dryland soils contain less than 1%, and frequently less than 0.5%, carbon (Lal, 2002b). Furthermore, although agricultural practices have reduced soil carbon levels globally, dryland landscapes are particularly prone to degradation and desertification. The quantity of carbon in dryland soils has therefore been reduced from an initially low base. However, there are some positive aspects relating to carbon storage in dryland soils. Drier soil *per se* is less likely to lose carbon (Glenn *et al*, 1993) and consequently the residence time of carbon in dryland soils is much longer than, for example, forest soils (Gifford *et al*, 1992). The organic matter itself and the processes operating on it are not believed to be any different in dry or hot regions compared to temperate zones where carbon levels are generally higher and where most research has been conducted (Batjes & Sombroek, 1997; Ayanaba & Jenkinson, 1990; Syers, 1997).

In spite of the generally low carbon content of dryland soils, the fact that drylands cover approximately 40% of the global land area (FAO, 2000), together with the fact that many of these soils have been degraded, means that drylands may well have the greatest potential to sequester carbon (Scurlock & Hall, 1998; Rosenberg *et al*, 1999).

Factors Controlling Soil Carbon Sequestration

The carbon sequestration potential of a soil depends on its capacity to store resistant plant components together with protecting, and accumulating, humic substances. The quantity of soil carbon present is controlled by a complex interaction of processes determined by carbon inputs and decomposition rates. Factors controlling the quantity of organic matter in soil include temperature, moisture, oxygen, pH, nutrient supply, clay content and mineralogy. Accumulation of carbon will be favoured by conditions that do not promote decomposition, i.e. low temperature, acid parent materials and anaerobic conditions. Himes (1998) estimates that it takes 833 kg N, 200 kg P and 143 kg S to sequester 10 tonnes of carbon in humus. Soil fertility is therefore an important aspect of carbon sequestration yet many dryland soils are now low in nutrients. Rasmussen & Parton (1994) suggest that carbon levels in dryland soils rise between 10 - 25% of the rate that carbon is added.

The ultimate source of soil carbon is atmospheric CO_2 that is captured by plants in the process of photosynthesis. Primary production therefore sets the upper limit on the amount of carbon that can be stored in soil. The biomass produced by net primary production will ultimately be available for decomposition and incorporation into the soil either directly as dead plant material or as organic matter that has passed through the animal food chain.

The most efficient method of accumulating carbon in soils must be by direct decomposition of plant material. If carbon passes through the heterotrophe chain some will be lost directly to the atmosphere as CO_2 as a consequence of respiratory activity and through digestive and assimilatory inefficiencies. This is illustrated in Figure 1 where the Rothamsted soil carbon model has been used to show the effect of adding plant residue and cattle manure to a vertisol soil in semi-arid Andhra Pradesh. The system is in steady state when, in 1989, 5 t C ha^{-1} in the form of plant residue or manure is added for 5 consecutive years. The soil organic carbon content rises continually during the period of application and then declines over subsequent years. Assuming a 50% carbon content of the plant material (Schlesinger, 2000), 10 t of plant residue would actually be required to provide each 5 t application of plant carbon. For the manure application, if the digestive efficiency of livestock is 60% (Schlesinger, 2000), 10 t of plant residue would only produce 4 t of manure. If cattle manure contains 25% carbon, only 1 t of carbon would actually be available for incorporating into the soil, Figure 1. Clearly, much less carbon is available for sequestering into the soil when it passes through the heterotrophic chain.

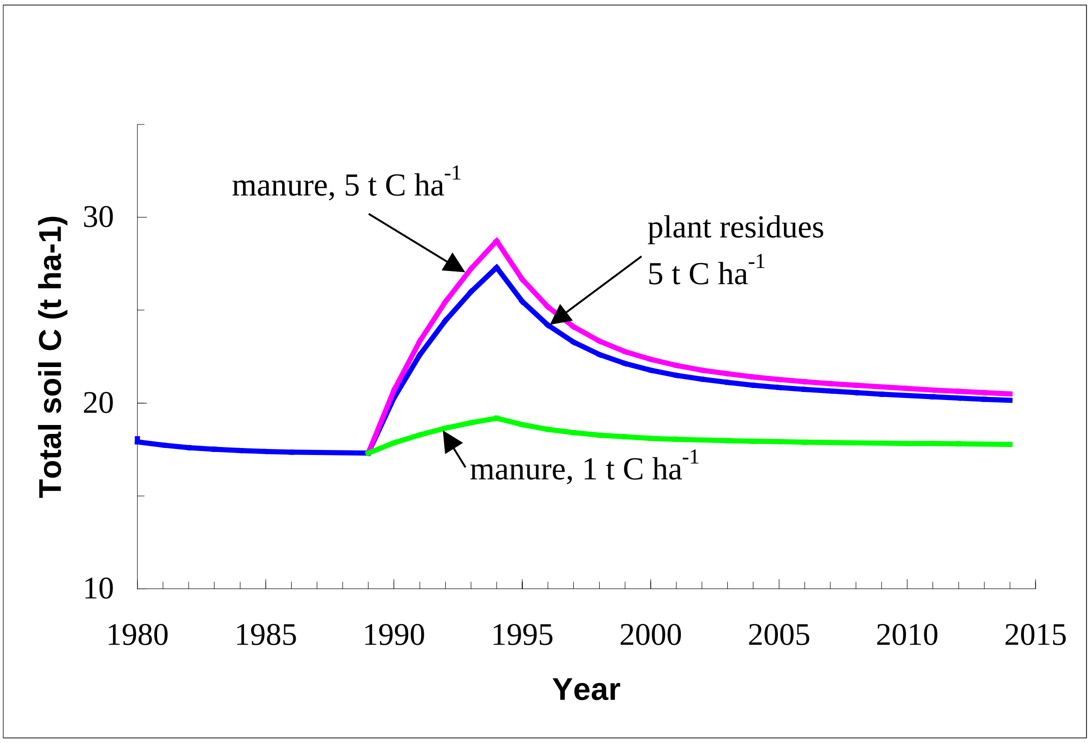

Figure 1 *Change in organic soil carbon for a vertisol in semi-arid Andhra Pradesh, India modelled with the RothC soil carbon model. Five tonnes of plant residue carbon and five tonnes of cattle manure carbon were applied in 1989 for five consecutive years. Application of one tonne of carbon in cattle manure represents the amount of manure carbon that would be available for incorporation into the soil if the plant residues had been directly fed to the cattle.*

Long-term field experiments have shown that there is a direct linear relationship between the quantity of carbon added to soil as organic matter and the amount of carbon accumulated in the soil, other factors remaining constant (Cole *et al*, 1993; Duiker & Lal, 1999). However, the dynamics of soil organic matter are complex and the factors controlling the flux of carbon will interact uniquely at each site. Predictions of how future carbon pools will change must be made with caution. This is particularly so as effects of land management practice on soil carbon may not be measurable for twenty years (Rasmussen *et al*, 1998). Eventually, if conditions are maintained the system will reach a new steady state, commonly within 50-100 years (Swift, 2001). The crucial point of consideration with regard to soil carbon sequestration is the dynamic nature of the process; even if soils are acting as a sink, they will be simultaneously losing some carbon and many factors can act at any time to shift the balance. All carbon in soils has the ability to completely decompose.

Other than erosion, the major route for carbon loss from soil is through CO_2 efflux, commonly referred to as soil respiration. This loss is enhanced in warmer environments because the flux of CO_2 emanating from microbial activity increases with temperature, although the relationship is non-linear, and best described by an Arrhenius model (Fang & Moncrieff, 2001; Qi *et al*, 2002). This is a primary factor acting against carbon sequestration in hot, arid climates. However, decomposition also requires moisture and in drylands the availability of water is, by definition, frequently limiting. The interaction between temperature and moisture in controlling decomposition is complex and dependent on site characteristics. For example, Mielick and Dugas (2000) found that CO_2 efflux from the soil in a tall-grass prairie depended primarily on temperature and secondly on water. Conversely, in semi-arid Alberta, moisture was shown to be the main factor controlling soil respiration during the growing season (Akinremi *et al*, 1999). A more complex situation was discovered in Central Texas by Franzluebbers *et al* (1995). They discovered that depending on crop sequence, tillage regime and season, between 65-98% of the variation in CO_2 flux could be accounted for by a combination of soil temperature, moisture and time of year.

Bursts of microbial activity following episodic rainfall are common in drylands. Such seasonality of climate can make the timing of some land management practices, such as plant residue and manure applications, critical in terms of their effectiveness, not only for soil fertility, but also for maximum carbon sequestration.

Plants – Capturing Carbon for Soils

Carbon in soils ultimately comes from CO_2 in the atmosphere, which is captured by plants using the process of photosynthesis. Globally, productivity is strongly correlated with water availability. Consequently drylands are disadvantaged by the fact that carbon fixation is lower than in many other ecosystems because of the limitation imposed by water shortage. From a purely carbon sequestration point of view, plants that are grown in drylands should be selected on the basis of having the best adapted features for these conditions. The growing of vegetation on badly degraded dryland soils not suited to conventional agriculture has been suggested as a way of increasing soil carbon stocks (Lal *et al*, 1999; Pretty *et al*, 2002).

In the most arid areas true xerophytic species are required which are able to produce biomass with zero or minimal inputs from man. Plant species that are best adapted to restricted water supply use either the C_4 or CAM (crassulacean acid metabolism) photosynthetic mechanism. The latter are able to open their stomatal pores at night when the transpiratory water loss is least. They fix CO_2 into four carbon acids that can be used as an internal reservoir of CO_2 to supply photosynthesis during daylight. C_4 species have a CO_2 concentrating mechanism, the net result of this is to reduce the flux of gases into and out of their leaves, and consequently enhance water-use-efficiency. This is accompanied by a higher nitrogen-use-efficiency, which reduces the need for fertiliser application. These species also lack photorespiration, a consequence of which is a higher optimum temperature for photosynthesis and an ability to increase CO_2 uptake to maximum irradiance in the tropics. Desert plants were considered to have slow rates of growth but research with *Agave* has shown that this CAM utilising species, which is adapted to severe arid conditions, can exhibit very high rates of productivity (Nobel *et al*, 1992).

Whilst above ground biomass provides soils with organic matter following senescence, below ground biomass is an important source of carbon, not only through root death but also from root exudates. However, quantifying root exudation is a difficult task. Xerophytes typically have deep root systems that enable them to tap below ground water. These deep root systems are ideal for taking carbon deeper into the soil where it is less susceptible to oxidation. Fisher (1994) has emphasised the importance of introduced deep-rooted grasses for increasing carbon storage in South American savannas, while Gifford (1992), has estimated that thirty percent of total primary production enters long-term carbon storage in Australian soils from roots.

Plants of even the most arid regions have, therefore, a good potential to assimilate carbon that is then available for sequestering into the soil. Not only is there a variety of plant species available for growing in dryland environments, a number of these can be harvested for useful products (Lal *et al*, 1999). Harvesting will of course decrease the amount of plant residues available for incorporation into the soil, but the potential for carbon sequestration does exist.

Halophytes

A special feature of many dryland soils is salinity, either through natural occurrence or increasingly as a result of irrigation. Saline soils affect large parts of the drylands (Glenn *et al*, 1993; Lal, 2000b). Such lands are often abandoned but halophytic plants are especially adapted to these conditions and offer potential for sequestering carbon in this inhospitable environment. It has been estimated that 130 million hectares are suitable for growing halophytes and these can be used for forage, feed and oilseed (Glenn *et al*, 1993). Glenn estimates that 0.6–1.2 Gt C y^{-1} could be assimilated annually by halophytes and using evidence from decomposition experiments suggests, perhaps optimistically, that 30-50% of this carbon might enter long-term storage in soil. Although irrigation would be required to achieve these figures, a complete carbon budget still suggests a carbon accrual rate of 22.5-30%.

Grasslands

Grasslands form the natural biome in many drylands, partly because rainfall is insufficient to support trees, and partly because of prevailing livestock management. However, grassland productivity and carbon sequestration have been controversial. The productivity of tropical grasslands is now known to be much higher than was previously thought, and consequently they sequester much more carbon (Scurlock & Hall, 1998). Estimates for carbon stored under grassland are 70 t ha^{-1}, which is not unlike values for forest soils. Unfortunately many of the grassland areas in drylands are poorly managed and degraded but as a consequence offer potential for carbon sequestration.

The average annual input of organic matter into grassland is approximately double the 1-2 t ha^{-1} that is contributed to cropped soils (Jenkinson & Rayner, 1977). This fact is borne out by the results of studies from varied locations. The data has shown that grassland, even if subject to controlled grazing, generally has higher soil carbon levels than cropland. Hence, Chan and Bowman (1995) found that 50 years of cropping soils in semi-arid New South Wales has on average reduced soil carbon by 32% relative to pasture. The reduction was linearly related to the number of years of cropping. Similarly, soils of tall-grass pasture subjected to controlled grazing had greater soil carbon than adjacent cropland subject to conservation tillage (Franzluebbers *et al*, 2000).

The key factor responsible for enhanced carbon storage in grassland sites is the high carbon input derived from plant roots. It is this high root production that provides the potential to increase soil organic matter in pastures and vegetated fallows compared to cropped systems. Root debris tend to be less decomposable than shoot material because of their higher lignin content (Woomer *et al*, 1994). Consequently, the key to maintaining and increasing carbon sequestration in grassland systems is to maximise grass productivity and root inputs (Trumbmore *et al*, 1995). Grasses have also been shown to sequester more carbon than leguminous cover crops (Lal *et al*, 1999). Grasses also have the potential to sequester carbon on previously degraded land. Garten and Wullschleger (2000) used a modelling approach that estimated a 12% increase in soil carbon could be obtained under switchgrass (*Panicum virgatum* L.) over ten years degraded land.

Grazing is a feature of many grasslands, natural or managed. This might be expected to decrease the availability of residues that can be used to sequester carbon, especially as the quantity of carbon returned in manure is less than that consumed (Figure 1). However, provided there is careful grazing management many investigations have found a positive effect of grazing on the stock of soil carbon. This was found to be the case for a composite pasture (alfalfa and perennial grasses) in the semi-arid pampas (Diaz-Zorita *et al*, 2002). Even under harsher conditions in Syria, grazing was found to have no detrimental impact on soil carbon (Jenkinson *et al*, 1999). Schuman *et al* (2002) have calculated that with proper grazing management, US rangelands can increase soil carbon storage by 0.1 – 0.3 t ha^{-1} y^{-1} and for new grasslands this can rise to 0.6 t ha^{-1} y^{-1}.

The positive effect of grazing appears to result from the effect that it has on species composition and litter accumulation. Willms *et al* (2002) found that when prairie was protected from grazing there was little effect on production but there was an increase in the quantity litter. Reeder *et al*, (2002) also found that there was an accumulation of litter in an un-grazed semi-arid system and that soil carbon levels were higher in the grazed lands. The litter acted as a store of immobilised carbon. The un-grazed grassland also experienced an increase in species which lacked the fibrous rooting system that is conducive to soil organic matter formation and accumulation.

Grasslands can therefore play a vital role in sequestering carbon, but careful grazing management is essential. The historical record shows how very susceptible semi-arid grasslands are to overgrazing, soil degradation and carbon loss.

Burning

Fires form part of the natural cycle in many biomes and are especially prevalent in grassland ecosystems. However, humans have also used fire to clear areas for agriculture and to clear crop residues. The action of fire would seem to be counter to carbon sequestration: it returns carbon fixed by vegetation directly to the atmosphere thereby preventing its incorporation into the soil. The effect of fire is difficult to generalise because it will depend upon its characteristics; namely its intensity and speed. These factors will obviously be influenced by the state of the vegetation, i.e. its maturity and woody component, accumulation of litter and climatic factors such as moisture level. Clearly carbon in all the above ground material that is fully combusted will be lost from the system. However, in grassland ecosystems carbon lost to fires can be replaced quickly by increased photosynthesis and vegetative growth (Knapp, 1985; Svejcar & Browning, 1988). Even in savannah systems that contain woody species it has been shown that carbon lost through combustion can be replaced during the following growing season (Ansley *et* al, 2002). Regarding the soil, the intensity and speed of the fire will govern the depth that is affected. In one study where burning was used to clear forests, 4 t C ha^{-1} was lost in the top 3 cm of soil but that this was replaced within one year under a pasture system (Chone *et al*, 1991).

Not all plant material is fully combusted by fire and a variable amount of charcoal will be produced. Charcoal is extremely resistant to decomposition; it is not cycled like most organic matter, and has a mean residence time of 10,000 years (Swift, 2001). Consequently in severely degraded soils that have been affected by several fires, charcoal and charred material can form a substantial proportion of the remaining organic carbon. It is not know however whether charcoal and other charred material has any protective effect on the native organic matter. Thus although fires release CO_2 back into the atmosphere the simultaneous production of charcoal can be regarded as a sequestration process that results in a substantial amount of carbon being accumulated within the soil for a long period of time (Swift, 2001).

Afforestation

Forestry is recognised as major sink for carbon but as well as accumulating carbon above ground, it can also make significant contributions to soil carbon even in drylands. A number of suitable species are available that make for viable afforestation in dryland environments (Srivastava *et al*, 1993; Silver *et al*, 2000; Kumar *et al*, 2001; Niles, *et al*, 2002). In particular, nitrogen-fixing trees generally lead to increased accumulation of soil carbon. For example, *Prosopis* and *Acacia* are adapted to subtropical semi-arid lands and are reported to increase the level of soil carbon by 2 t ha^{-1} (Geesing, 2000).

Some tree varieties are particularly suitable for growing on degraded lands, their deep rooting systems tapping resources unavailable to shallow-rooted crops. *Prosopis juliflora* has been grown on salt-affected soils in north-west India and increased the soil organic carbon pool from 10 t ha^{-1} to 45 t ha^{-1} over an eight year period (Garg, 1998). Even where large-scale forestry is not appropriate there is often scope for introducing trees around farmers' fields. Such is the case in semi-arid India where *Prosopis cineraria* has improved soil fertility as well as sequestering extra carbon (Nagarajan *et al*, 2000).

However natural systems are complex and trees do not guarantee improved carbon sequestration. Jackson *et al* (2002) found that soil carbon decreased when woody vegetation invaded grasslands. Although there was an increase in above and below ground biomass these gains were offset by losses in soil carbon.

Residues

Plant residues provide a renewable resource for incorporation into the soil organic matter. Production of plant residues in an ecosystem at steady-state will be balanced by return of dead plant material to the soil. In a native prairie about 40% of plant production is accumulated in the soil organic matter (Batjes & Sombroek, 1997). However, in agricultural systems, because plants are harvested, only about 20% of production will on average be accumulated into the soil organic fraction (Batjes & Sombroek, 1997). Furthermore, in some farming systems all above ground production may be harvested, leaving only the root biomass. Of the plant residue retuned to the soil, about 15% can be expected to be converted to passive soil organic carbon (Lal, 1997). Schlesinger (1990) is more pessimistic, suggesting that only 1% of plant production will contribute to carbon sequestration in soil. The actual quantities of residue returned to the soil will depend on the crop, growing conditions and agricultural practices. So, for a soybean-wheat system in sub-tropical central India the annual contribution of carbon from above ground biomass is ca. 22% for soybean and 32%

for wheat, (Kundu *et al*, 2001). This resulted in 18% of annual gross carbon input being incorporated into the soil organic matter. In semi-arid Canada the conversion of residue carbon to soil organic carbon was reported to be 9% in frequently fallowed systems, increasing to 29% for continuously cropped systems (Campbell *et al*, 2000).

All belowground production, unless a root crop is being harvested, is available for incorporation into the soil organic matter. Roots are believed to be the major constituent of particulate organic matter although tillage substantially reduces the net accumulation of carbon from roots (Hussain *et al*, 1999; see section Tillage). In cool climates below ground carbon inputs from roots alone can generally maintain soil carbon levels, but this is not the case in warmer or semi-arid regions where residues are oxidized much more readily, providing sufficient moisture is available (Rasmussen *et al*, 1998). Consequently, when continuous cropping is practised in drylands, failure to return above ground plant residue will invariably lead to a reduction in soil carbon. Many African soils demonstrate this phenomenon: years of continuous cropping without the recommended inputs has often halved their carbon content (Woomer *et* al, 1997; Ringius, 2002).

Both the quality and quantity of plant residues are important factors for determining the amount of carbon that is stored in soil. The quantity is highly dependent on the environmental conditions and agricultural practices and so will be very varied. Differences between crops can be most marked. A crop such as corn will return nearly twice as much residue to the soil compared to soybean and consequently will result in a higher rate of soil organic matter increase (Reicosky, 1995). The advantage that cereals have over legumes for achieving maximum carbon sequestration rates has also been demonstrated by Curtin *et al* ,(2000). They have shown that whilst black lentil fallow in semi-arid Canada added between 1.4 and 1.8 t C ha^{-1}, a wheat crop would add 2-3 times this amount of carbon annually. Similarly in Argentina, soybean, which produced 1.2 t ha residue, resulted in a net loss of soil C while corn, with 3.0 t residue, significantly lessened the loss of soil carbon from the system (Studddert & Echeverria, 2000).

Even within one crop group large differences in organic matter production occur. Abdurahman *et al*, (1998) compared dry leaf production from pigeon pea and cowpea. Whilst the former yielded 3 t ha^{-1}, cowpea only produced 0.14 t ha^{-1}. These examples serve to illustrate the fact that choice of crop can have a major influence on how much carbon can be sequestered by an agricultural system.

The importance of roots relative to shoots for providing soil carbon is another factor illustrated in an experiment to compare the fate of shoot and root derived carbon (Puget & Drinkwater, 2001). In this study with leguminous green manure (hairy vetch), nearly half of the root derived carbon was still present in the soil after one growing season whereas only 13% of shoot carbon remained. This implies they shoot residues are rapidly broken down (higher N content, Woomer *et al*, 1994) and serve as a nitrogen source for the following crop.

Chemical composition of plant residues affects their decomposability. On average crop residues contain about 40-50% C but N is a much more variable component. A high concentration of lignin and other structural carbohydrates together with a high C:N ratio will decrease decomposability. For example measurement of CO_2 evolution from tree leaves of African browse species and goat manure showed a significant correlation with initial nitrogen content and a negative correlation with lignin content (Mafongoya *et al*, 2000). Legume residues such as soybean are generally of high quality (low C:N ratio) and so decompose rapidly (Woomer *et al*, 1994). Although the chemical composition of the plant residue affects its rate of decomposition there is little effect on the resulting soil organic matter (Gregorich *et al*, 1998).

When residues accumulate on the soil surface their physical presence will affect the soil. Mulches reduce water loss and soil temperature (Duiker & Lal, 2000); both important factors for drylands, especially if the soil temperature is above the optimum for plant growth. Ability of soil to assimilate organic matter is not clear-cut. A linear relationship between application and accumulation of soil organic matter is often quoted. However measurement of CO_2 flux following addition of wheat straw in central Ohio showed this to increase with application of straw (0, 8 & 16 t ha^{-1}) but after 4 years soil organic carbon was 19.6, 25.6 and 26.5 t ha^{1}, suggesting that carbon sequestration was reaching saturation (Jacinthe *et al*, 2002).

Care must be taken when applying residues as large losses of carbon can still occur under certain conditions. One example in western Kenya 70-90% of the added carbon was lost within 40 days when green manure from agroforestry trees was applied during the rainy season (Nyberg *et al*, 2002). In Niger, west Africa, addition of millet residue and fertiliser for 5 years had no significant effect on the carbon level in these sandy soils (Geiger *et al*, 1992). It is believed that termite activity may have contributed to the low levels of soil carbon here because these insects can consume entire surface mulches within one year.

The potential amount of residues available for applying to soils can be large. Gaur (1992), estimated that in India about 235 million tonnes of straw/stover is produced annually from five major cereals (not only from drylands). Even if half of this was used for feeding livestock there would be over one million tonnes available for adding to the soil. However, availability of sufficient plant residues is often a problem, especially if they are required for livestock feed. This conflict of requirement frequently occurs in many dryland farming systems.

In West Africa, crop residues are either removed or burnt; consequently the amount of soil carbon has steadily declined. Continuous cultivation and manure application can raise soil carbon levels by 40% but this frequently involves mining carbon from neighbouring areas to support the livestock (Ringius, 2002). Where plant, and animal, residues are in short supply possibilities may exist for alternative organic inputs to the soil. For example, in India, waste products from a plant processing coir dust have been successfully incorporated into soil (Selvaraju *et al*, 1999) and other experiments with industrial glue waste (Dahiya *et al*, 2001) have successfully increased soil carbon levels.

With regard to the complete carbon budget, if plant residues accumulate *in situ* there is no extra carbon cost involved, and consequently the carbon fixed by plants in photosynthesis is available as a net gain to the soil. The situation becomes more complex if machinery is required to separate the residue from the harvestable components – can this energy/carbon usage be ascribed solely to the harvesting process or should some of the cost be subtracted from the residue addition? If plant residues are transported between fields the energy cost would need to be included.

Perhaps the biggest issue regarding residue application and carbon accounting is the question of whether carbon is simply being transferred form one place to another rather than being truly sequestered. If organic material from industrial processes or other sources is to be used for incorporation into the soil then any carbon used in transport would have to be accounted for. However, if the material is truly a waste product there should be no need to consider the carbon used in its production. In this case the carbon available for sequestration should be viewed no more differently than the carbon dioxide emitted from a fossil fuel source that is subsequently fixed by plants in photosynthesis and returned to the soil via crop residues.

Applied Manures

Application of farmyard manure has long been treated as a valuable source of organic matter that aids soil fertility. One of the key characteristics of manure application is that it promotes the formation and stabilisation of soil macroagregates (Whalen & Chang, 2002) and particulate organic matter (Kapkiyai *et al*, 1999). Manure is more resistant to microbial decomposition than plant residues: consequently for the same carbon input, carbon storage is higher with manure application than with plant residues (Jenkinson, 1990; Feng & Li 2001). This is shown in Figure 1 where equal additions of carbon as plant residues a) and manure b) are compared. Following five years of application, soil receiving the manure had 1.18 t ha^{-1} more carbon present than soil receiving the plant residues. Even after 15 years there was a still a difference of 0.37 t C ha^{-1} as calculated by the RothC soil carbon model. This can similarly be demonstrated in the field: Gregorich *et al* (1998) found that manured soils had large quantities of soluble carbon with a slower turnover rate than in control or fertilised plots.

The composition and therefore decomposition of manure varies between species from which it originates and also within species according to their diet (Somda & Powell, 1998). Many field trials have found that manure is the best means to incorporate organic matter into soils and promote carbon storage. For example, Li *et al*, (1994) found that over a range of soils and climatic conditions manure yielded the largest amount of carbon sequestered, although soil texture was important, and the greatest rate of sequestration occurred when there was the highest clay content. Traditionally, however, many agro-systems add manure in combination with fertilisers; reviewed by Haynes and Naidu (1998).

Depending on the system, application of even relatively high amounts of farmyard manure does not guarantee an increase in soil carbon. A long-term study in Kenya has shown that even when manure is applied, and maize residue returned, soil organic matter declined (Kapkiyai *et al*, 1999). It has been estimated that to maintain soil carbon in this system would require 35 t ha^{-1} of manure or 17 t ha^{-1} manure with 16 t ha^{-1} stover annually (Woomer *et al*, 1997). Consequently, in this case there is a carbon shortfall but it is difficult to see how the present system could provide for it. Additionally, high application rates of manure can sometimes cause problems in the soil through the accumulation of K^+, Na^+ and $NH4^+$ and the production of water–repellent substances by decomposer fungi (Haynes & Naidu, 1998). An additional problem in drylands that restricts the quantity of manure that can be applied is ‘burning’ of the crop when insufficient moisture is available at the time of application. Consequently farmers will often wait until the rains have come before an application is made especially as precipitation is often erratic in arid regions.

Production of sufficient manure for application to fields is a real problem for many smallholder farming systems. In Nigeria, direct manure input onto land from dry season grazing is approximately 111 kg ha^{-1}, dry matter, (Powell, 1986). This quantity will have little effect on the soil. More useful is the practice of night-parking cattle. Manure production is usually greatest at dawn and dusk and when 50 cattle were penned in an area of 0.04 hectares for 5 nights they produced the equivalent of 6.875 t ha^{-1} manure (Harris, 2000).

Normally cattle will be penned in fields for 2-3 nights and in northern Nigeria this can supply manure at a rate of 5.5 t ha^{-1} (Harris, 2000). Alternatively, in densely populated parts such as the kano-closed-settlement-zone, cattle and crop production are totally integrated. Cattle are permanently kept in pens and fed using feed grown on neighbouring fields. Their manure is collected and spread onto the croplands. Although an efficient system, some carbon will be lost as a consequence of respiratory and growth requirements of the cattle. An additional problem associated with cattle rearing is that ruminants produce significant quantities of methane, which itself, is a very potent greenhouse gas, (see section Trace Gasses).

There has been some debate as to the usefulness that animal manure can make to carbon sequestration. Schlesinger (1999, 2000) has calculated that to provide 13.4 t ha^{-1} would require 3 hectares of cropland in order to produce sufficient cattle feed. This means that manure production requires mining of carbon from neighbouring lands. Although this is a generalisation, it is argued that the 3:1 differential makes it unlikely that manure production *per se* could be used as a means of providing a net carbon sink in soils. However, in many dryland smallholder cropping systems manure application rates are much less than this, (see section - case studies), although fodder production will also be less efficient. Smith and Powlson (2000) have made the case that keeping cattle is part of many agricultural systems and so manure should be considered as a byproduct that can be added to arable land without necessarily including the carbon cost of its production. Part of the disagreement with Schlesinger on the usefulness of manure depends on where the system boundary for carbon accounting is drawn. However, when such a carbon audit is conducted it is essential to remember that the purpose of agriculture is to feed people; offsetting greenhouse emissions can only ever been evaluated as a secondary activity.

Inorganic Fertilizers and Irrigation

Fertilisation and irrigation are primary means used to increase plant productivity and crop yield. Any increase in biomass also offers increased scope for carbon sequestration by soil. Consequently irrigation and fertilisation have been recommended, and proved to be successful methods of increasing carbon sequestration (Lal *et al*, 1999). Rasmussen and Rohde (1988) have shown a direct linear relationship between long-term nitrogen addition and accumulation of organic carbon in some semiarid soils of Oregon. However, these technologies provide no additional organic matter themselves but do carry a carbon cost. Schlesinger (1999, 2000) has pointed out that pumping water requires energy and that the process of fertiliser manufacture, storage and transport is very energy intensive. Consequently, Schlesinger (2000) has estimated that the gains in carbon stored using either fertilisation or irrigation are offset by losses elsewhere in the system. Irrigation can also lead to the release of inorganic carbon from the soil, (see section Inorganic Carbon).

Izaurralde *et al*, (2000) have argued that the calculations used by Schlesinger (1999) are based on very high rates of fertiliser application. Certainly, for many dryland agriculture systems in the developing world, farmers do not have sufficient funds to apply large quantities of fertiliser, even if it were available. With regard to the energy costs incurred in pumping water, solar powered systems are being developed (Sinha *et al*, 2002) and dryland environments with frequently clear skies would be able to make the best use of this solar radiation.

These examples serve to illustrate how important it is to consider the whole system when carbon sequestration is being considered to offset carbon dioxide emissions. The exact carbon cost of irrigation and fertilisation would require calculating for each system, but the carbon deficits associated with both technologies makes their incorporation into net carbon sequestration systems difficult to achieve. Water conservation, growing of legumes and careful nutrient cycling are more likely to yield a positive carbon balance.

Tillage

Tillage is generally recognised as one of the major factors responsible for decreasing carbon stocks in agricultural soils (Pretty *et al*, 2002). Research and experimentation with reduced tillage practices are most prevalent in the Americas. The mouldboard plough and disc harrow are the biggest contributors to the loss of soil carbon through their destruction of soil aggregates and acceleration of decomposition by mixing of plant residues, oxygen and microbial biomass. Soil aggregates are vital for carbon sequestration (Six *et al*, 2000), a process that is maximal at intermediate aggregate turnover (Plante & McGill, 2002), and of the organic matter fraction, the particulate organic matter is the most tillage sensitive (Hussain *et al*, 1999).

It is difficult to quantify the effects of tillage on soil carbon because this very much depends on the site, e.g. coarse textured soils are likely to be more affected by cultivation than fine ones (Buschiazzo *et al*, 2001). However, reducing tillage should be most effective in hot, dry environments (Batjes & Sombroek, 1997).

Reicosky (1997) conducted an elegant experiment that used measurements of CO_2 efflux to investigate tillage induced carbon loss from soil. The flux of CO_2 was monitored for nineteen days following different forms of tillage practice on a The mouldboard plough buried most of the crop residue and produced the maximum CO_2 efflux. The carbon released by the different treatments as a percentage of carbon in the crop residue was mouldboard plough 134%, mouldboard plough and disc harrow 70%, disc harrow 58%, chisel plough 54% and no-till 27%. This clearly demonstrates the correlation between CO_2 loss and tillage intensity, and demonstrates why farming systems that use mouldboard ploughing inevitably lose soil carbon. Very large amounts of organic matter would be required to replace the loss incurred by such heavy tillage. Reicosky *et al* (1995) estimate that 15-25 t ha^{-1} manure plus crop residue annually would be needed in north America to offset these loses.

The flux of CO_2 from soil generated directly by the tillage process may not always reflect the overall release of CO_2 and hence carbon storage of the system. This is illustrated by a comparison of conventional disc tillage and no-tillage in Central Texas by Franzluebbers *et* al, (1995). Here, seasonal evolution of CO_2 was up to 12% greater in the no-till system after 10 years. This was despite the fact that more carbon was sequestered by the no-till system. They suggest that a change in the dynamics of carbon sequestration and mineralisation have occurred under the no-till system. Similarly, Costantini *et al* (1996) found that more CO_2 was released from zero-till or reduced-till compared to conventional tillage despite there being increased levels of soil carbon. They ascribe this difference to an increase in the microbial biomass.

Rates of carbon loss through tillage depend very much on the site and cropping system. When Ellert & Janzen (1999) measured the flux of CO_2 following the passage of a heavy cultivator on a semi-arid Chernozerm soil in the Candian prairies they found that although tillage increased rates of CO_2 loss by 2 to 4 times, values returned to normal after 24 hours. They calculated that even with ten passes of the cultivator, only 5% of crop residue production would be released from this cropping system. In another situation, ploughing of a wheat-fallow cropping system near Sydney Australia reduced soil carbon by 32% after 12 years. Elimination of ploughing and adopting a no-till approach was unable to prevent a decrease in the carbon stock, although the loss was reduced to just 12% (Doran *et al*, 1998). The authors suggest that at this site a fallow period would be required to halt the decline in soil carbon.

There are many different types of tillage system, reviewed Unger (1990): conservation tillage covers a range of practices - no-till, ridge-till, mulch-tillage. In the latter higher levels of residue cover are maintained. With mulches, only a small fraction of the residue is in contact with the soil surface and the microbes it contains. Decomposition is slow, especially as oxygen availability will be limited. The physical presence of crop residues on the soil surface also alters the microclimate of the upper soil layer, which tends to be cooler and wetter compared with conventional tillage (Doran *et al*, 1998).

Accumulation of residues also reduces the loss of CO_2 from the soil surface. Alvarez *et al*, (1998) reported an increase in labile forms of organic matter under no-till in the Argentine rolling pampa, indicating a decrease in the minerliastion of the organic fraction. This study also noted that although organic carbon was increased by 42%-50% under no-till compared to ploughing and chisel tillage. There was also a marked stratification in the distribution of carbon under the no-till regime that was not evident in the ploughed system.

Stratification of organic carbon is common with reduced or zero tillage. Zibilske *et al*, (2002) in semi-arid Texas demonstarted that the organic carbon concentration was 50% greater in the top 4cm of soil of a no-till experiment compared to ploughing, but the difference dropped to just 15% in the 4-8cm depth zone. This is typical of organic carbon gains observed with conservation tillage in hot climates. Bayer *et al* (2000), working on a sandy clay loam Acrisol also found that increase in total organic carbon was restricted to the soil surface layers under no-till but that the actual quantity depended on the cropping system. An oat/vetch – maize/cowpea no-till system produced the greatest quantity of crop residues and sequestered the most carbon, 1.33 t C ha^{-1} y^{-1} over 9 years.

Reicosky (1995) has compared the results from many no-till trials. The data emphases the effect that crop rotation and quantity of crop residue has on organic matter accumulation. Overall, rates of organic matter accumulation can be expected to be lower in the hotter climates. Never the less, even in the very sandy soils of northern Syria it has been possible to make modest increases in soil organic matter with zero-till (Ryan, 1997) and in western Nigeria no-tillage combined with mulch application had a dramatic effect, increasing soil carbon from 15 to 32.3 t ha^{-1} over the course of just 4 years (Ringius, 2002).

Although no-tillage systems are an excellent tool for combating the carbon losses associated with conventional cultivation, they do have their own special problems. In temperate lands the reduction in soil temperature commonly associated with plant residue accumulation on the soil surface can retard germination, but in drylands, where soil temperatures are frequently above the optimum for germination and plant establishment, such cooling is likely to be beneficial (Phillips *et al*, 1980). No-tillage systems frequently suffer from an increased incidence of pests and diseases; the mouldboard plough and disc harrow are efficient weed controllers (Reicosky, 1995). Consequently, no-till systems generally rely upon extra herbicides and pesticides. These not only have an economic price (Phillips *et al*,

1980) but they also incur a carbon cost. However, in many dryland farming systems of the developing world, purchasing such products is not feasible and quite often there is plentiful labour available for weeding. Application of nitrogen fertiliser can also be problematic when used on undisturbed, no-till soils (Phillips *et al*, 1980). If the soil is poorly drained denitrification can occur and the reduced rate of evaporation increases the risk of nitrate leaching. Additionally the native soil nitrogen has a lower rate of mineralisation in undisturbed soil.

Not all soils are suited to a reduced tillage approach. Some soils in the Argentine pampa may actually lose more carbon under no-till (0.7 to 1.5 t C ha^{1} y^{1}) compared to conventional ploughing (Alvarez *et al*, 1995) and periodic ploughing is required to avoid soil compaction (Taboada *et al*, 1998). Where no-tillage is used on the pampas the physical status of the soil when the system is first introduced is a critical factor for the success of the system (Diaz-Zorita *et al*, 2002). Similarly, in the West African Sahel, the highest crop yields are obtained with deep ploughing, which is required to prevent crusting and alleviate compaction (world bank paper). In general, the success of reduced tillage systems is often dependent on soil texture (Needelman *et al*, 1999).

A particular advantage of the no-till system is that it favours multi-cropping; harvesting can be immediately followed by planting, (Phillips *et al*, 1980). Any cropping system that allows for continuous, or near continuous plant growth should yield the maximum capacity to produce plant biomass and consequently has the potential to provide the greatest amount of organic mater for inclusion into the soil.

Considering the overall carbon budget, no-till systems have a lower energy requirement because tillage is very energy intensive. Phillips *et al* (1980) have calculated that in North America no-till systems reduce the energy input into maize and soybean production by 7% and 18% respectively. Better water use efficiency means that the energy, and hence carbon, cost of irrigation are reduced but the impact of energy savings is frequently offset by additional herbicide requirements (Phillips *et al*, 1980). Kern & Johnson (1993) estimated that manufacture and application of herbicides to no-till systems of the Great Plains is equivalent to 0.02 t C ha^{-1}.

Reduced-tillage systems were originally adopted to help combat soil degradation. They were not intended as a means of sequestering carbon, which is a fortunate side effect. Although the effectiveness of no-till at sequestering carbon will depend on the specific agricultural system to which it is introduced, there is no doubt that as the intensity of tillage decreases the balance between carbon loss and gain swings toward the latter.

Rotations

The importance of rotation in agricultural systems has long been known and the procedure now forms an intricate part of many conservation tillage practices. The inclusion of rotations has many benefits such as countering the build up of crop specific pests and thereby lessening the need for 'carbon costly' pesticides and herbicides. Different crop species have a variety of rooting depths and this aids in distributing organic matter throughout the soil profile. In particular, deep-rooting plants are especially useful for increasing carbon storage at depth, where it should be most secure. The inclusion of nitrogen-fixing varieties in a rotation increases soil nitrogen without the need for energy intensive production of nitrogen fertilisers.

The beneficial effects that rotations have for carbon sequestration have been proven in many long-term field experiments. For example, Gregorich *et al* (2001) made a comparison of continuous maize cultivation with a legume-based rotation. The rotation had a greater effect on soil carbon than did fertiliser. The difference between monocultered maize and the rotation was 20 t C ha^{-1} whilst fertilisation only had a 6 t C ha^{-1} effect after 35 years. Additionally the soil organic matter present below the ploughed layer in the legume-based rotation appeared to be in a more biologically resistant state. This shows that soils under legume-based rotations tend to preserve residue carbon. A positive effect on soil organic carbon (increase of 2-4 t ha^{-1}) was also found with legumes and alternate cattle grazing in semiarid Argentina (Miglierina *et al* 2000).

Rotations, especially legume-based are now generally regarded as extremely valuable for maintaining soil fertility and have a very good potential for sequestering carbon in dryland systems. Drinkwater *et al* (1998) estimate that their use in the maize/soybean growing region of the USA would increase soil carbon sequestration by 0.01-0.03 Pg C y^{-1}. The effectiveness of rotations for sequestering carbon is likely to be greatest when they are combined with conservation tillage practices.

Fallows

The role that fallows play in carbon sequestration is varied. If cropping is not taking place it is important that vegetation cover is preserved. This is especially so in drylands where exposed soil is most likely to suffer from erosion and degradation. In addition to protecting the soil, cover crops will utilise solar energy that would otherwise be wasted. The CO_2 fixed is then available for sequestering into the soil as the plants senesce. The importance of vegetation cover can be illustrated with the results from an experiment conducted at a semi-arid site in Mediterranean Spain (Albaladejo *et al*, 1998). Four and a half years after the vegetation cover was removed from one site the soil organic carbon had decreased by 35% compared to the control plots.

The type of fallow is important. In Nigeria forest clearance caused a decline in soil carbon from 25 to 13.5 t ha^{-1} in 7 years but 12 –13 years of bush fallow restored the carbon content (Juo *et al*, 1995). Conversely, pigeon pea fallow was unable to sequester sufficient carbon on account of its low biomass production and rapid degradation.

In many situations, however, fallows can have a negative effect on carbon storage. The frequency of summer fallows in semi-arid regions has been suggested to be one of the major factors influencing the level of soil carbon in agricultural systems (Rasmussen *et al*, 1998). Reducing the summer fallow in the semi-arid north west USA is reported to have had a greater effect on soil carbon than decreasing tillage intensity: the loss of carbon in this region is believed to reflect the high rates of biological oxidation that occur here and this can only be offset by very large applications of manure (Rasmussen *et al*, 1998). Consequently yearly cropping and the associated organic additions is the recommended practice. Miglierina *et al* (2000) also found that reducing summer fallow increased soil carbon, a consequence of the additional crop residue that was added. Using the CENTURY agro-ecosystem model Smith *et al* (2001) predicted that reducing summer fallow in wheat cropping systems (wheat-fallow to wheat-wheat-fallow) in the semi arid chernozems of western Canada would reduce carbon losses by 0.03 t ha^{-1}.

Elimination of fallows can be highly beneficial for soil carbon simply because most fallows are associated with small inputs of plant residue. The significance of fallows for carbon sequestration in a given system will depend on whether or not the cropping cycle adds significant quantities of organic matter to the soil: if it does, then the presence of fallows is unlikely to enhance carbon storage within the system. Conversely, if the cultivation practice is poor and little or no organic matter added, fallow periods will serve to counter this situation.

Soil Inorganic Carbon

Not all soil carbon is associated with organic material; there is also an inorganic carbon component in soils. This is of particular relevance to drylands because calcification and formation of secondary carbonates is an important process in arid and semi-arid regions. Consequently the largest accumulations of carbonate occur in the soils of arid and semi-arid areas (Batjes & Sombroek, 1997). Dynamics of the inorganic carbon pool are poorly understood although it is normally quite stable. Sequestration of inorganic carbon occurs via movement of HCO^-_3 into ground water and closed systems. According to Schlesinger (1997) accumulation of calcium carbonate is quite low at between 0.0012 – 0.006 t ha^{-1}. However, Lal *et al* (1999) believe that sequestration of secondary carbonates can contribute 0.0069 - 0.2659 Pg C y^{-1} in arid and semi-arid lands.

Although soil inorganic carbon is relatively stable it will release CO_2 if the carbonates become exposed through erosion (Lal *et al*, 1999). Additionally, irrigation can cause inorganic carbon to become unstable if acidification takes place through nitrogen and sulphur inputs. The release of CO_2 through precipitation of carbonate is seen as a major problem if irrigation waters are used in any system that is trying to store carbon (Schlesinger, 2000). Further more, Schelsinger (2000) has pointed out that ground water of arid lands often contains up to 1% Ca and CO_2. Since this concentration is much higher than that which occurs in the atmosphere, when these waters are applied to arid lands CO_2 is released to the atmosphere and $CaCO_3$ precipitates. Schlesinger's calculations actually suggest that irrigation of some cropping systems would yield a net transfer of CO_2 from the soil to the atmosphere.

Trace Gases

An important aspect of agricultural systems in relation to the global carbon balance is the production of trace gases, particularly methane and nitrous oxide. When carbon sequestration by soils is being considered as a mechanism for offsetting greenhouse gas emissions it is necessary to consider all the interacting factors that can influence global warming. Both methane and nitrous oxide are radiatively active gases and, like carbon dioxide, contribute to the greenhouse effect. Although they are present in the atmosphere at much lower concentrations than carbon dioxide they are much more potent. Methane is twenty one times more effective at producing global warming than carbon dioxide and nitrous oxide is over three hundred times more active.

Our knowledge of trace gas fluxes is still incomplete; suffice it to say that ruminants, composting, biomass burning and water-logging produce methane whilst nitrous oxide is released from soils when nitrogen fertiliser or manure is applied (Vanamstel & Swart, 1994). Manure usage is in fact considered to be the major problem with regard to trace gas emissions in agriculture. This is a potentially serious problem because the application of manure is a major tool for increasing soil carbon in drylands. Smith *et al* (2001) calculated that for European soils the effect of trace gases is sufficient to reduce the carbon dioxide mitigation potential of some no-till and manure management practices by up to a half. Additionally, climate change is likely to amplify the problem as increased temperature is predicted to promote nitrous oxide emissions (Li & Harriss, 1996).

Climate Change

Climate is a major factor involved in soil formation; consequently future climate change will undoubtedly influence soils. Photosynthesis and decomposition will be affected directly and this will impact on soils. Whether soil carbon levels increase or decrease will depend on the balance of effect between primary production and decomposition, (Kirschbaum, 1995). Overall, productivity is predicted to increase as a consequence of rising CO_2 concentration and temperature and this will lead to increased amounts of residue available for incorporation into the soil. However, higher temperatures can be expected to increase mineralisation of soil organic matter because this process is more sensitive to temperature increase than primary production. Kirschbaum (1995) predicts that soil organic carbon stocks will decline overall with global warming, yet Goldewijk *et al* (1994) suggest that the effects of temperature and water availability on soil respiration will be smaller than those attributable to the CO_2 fertilisation effect. What is clear is that the direction of change is far from certain and the balance of change will most likely operate at the regional level or less.

Agricultural systems, it has been argued, are to some extent buffered from environmental effects, whilst decomposition is not protected (Cole *et al*, 1993). Hence increased rates of mineralistion might be more significant than any enhancement of production. However, the quality of plant organic matter is expected to decrease under elevated CO_2 owing to an increase in the C:N ratio. This would slow the rate of degradation (Batjes & Sombroek, 1997). Globally the drylands are expected to become moister (Glenn *et al*, 1993) and this should lead to an increase in productivity and decomposition. However shifts in climate zones are dependent upon a complex array of variables. Predictions based on the CENTURY agro-ecosystem model suggest that overall, grasslands will lose soil carbon except tropical savannahs, which should show a small increase (Parton *et al*, 1995). Experiments at elevated CO_2 have also shown that changes in soil carbon in agro-ecosystems are particularly dependent upon the crop species being grown (Rice *et al*, 1994).

The full extent of the global rise in temperature associated with climate change may not be felt in many of the drylands because warming is predicted to be greatest at higher latitudes. With regard to the vegetation, some of the best-adapted plants for dryland regions use the C_4 photosynthetic mechanism (section Plants – Capturing Carbon for Soils). Unfortunately, because these species already have a CO_2 concentrating mechanism they show little or no increase in productivity at elevated CO_2. However, they are still likely to receive some benefit from the increased water use efficiency that that accompanies the rise in CO_2 concentration.

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