POTENTIAL OF DESERTIFICATION CONTROL TO SEQUESTER CARBON AND MITIGATE THE GREENHOUSE EFFECT

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Abstract. There is a strong link between desertification of the drylands and emission of CO_2 from soil and vegetation to the atmosphere. Thus, there is a strong need to revisit the desertification process so that its reversal can lead to C sequestration and mitigation of the accelerated greenhouse effect. Drylands of the world occupy 6.31 billion ha (Bha) or 47% of the earth's land area distributed among four climates: hyper-arid (1.0 Bha), arid (1.62 Bha), semi-arid (2.37 Bha) and dry sub-humid (1.32 Bha). Principal soils of drylands are Aridisols (1.66 Bha), Entisols (1.92 Bha), Alfisols (0.38 Bha), Vertisols (0.21 Bha) and others (1.27 Bha). Drylands occur in all continents covering 2.01 Bha in Africa, 2.00 Bha in Asia, 0.68 Bha in Australia, 1.32 Bha in the Americas and 0.30 Bha in Europe. Desertification, degradation of soil and vegetation in drylands resulting from climatic and anthropogenic factors, affects about 1.137 Bha of soils and an additional 2.576 Bha of rangeland vegetation. The rate of desertification is estimated at 5.8 million hectares (Mha) per year. Desertification is a biophysical process (soil, climate and vegetation) driven by socio-economic and political factors. The principal biophysical processes involved, accelerated soil erosion by water and wind and salinization, reduce soil quality and effective rooting depth, decrease vegetal cover, reduce biomass productivity, and accentuate vagaries of climate especially low and variable rainfall. Major consequences of desertification include reduction in the total soil C pool and transfer of C from soil to the atmosphere. Total historic loss of C due to desertification may be 19 to 29 Pg. The rate of C emission from drylands due to accelerated soil erosion is estimated at 0.227 to 0.292 Pg C y^{-1}. Therefore, desertification control and restoration of degraded soils and ecosystems would improve soil quality, increase the pool of C in soil and biomass, and induce formation of secondary carbonates leading to a reduction of C emissions to the atmosphere. Desertification control and soil restoration are affected by establishing vegetative cover with appropriate species, improving water use efficiency, using supplemental irrigation including water harvesting, developing a strategy of integrated nutrient management for soil fertility enhancement, and adopting improved farming systems. Adoption of these improved practices also have hidden carbon costs, especially those due to production and application of herbicides and nitrogen fertilizers, pumping irrigation water etc. Restoration of eroded and salt-affected soils is important to C sequestration. Total potential of C sequestration in drylands through adoption of these measures is 0.9 to 1.9 Pg C y^{-1} for a 25- to 50-year period beyond which the rate of sequestration is often too low to be important. In addition to enhancing productivity and food security, C sequestration in soils and ecosystem has numerous ancillary benefits. Therefore, identification and implementation of policies is important to facilitate adoption of recommended practices and for commodification of carbon.

1. Introduction

Increase in atmospheric concentration of CO_2 from 280 ppm in pre-industrial era to 365 ppm in 1995 (IPCC, 1996) is attributed to fossil fuel combustion and land

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Table I
Principal climates of arid lands (recalculated from Meigs, 1952; UNEP, 1992)

Climate	Land area (Bha)	Mean temperature (°C)	
		Coldest month	Warmest month
Hot	2.71	10–30	>30
Mild winter	1.14	10–20	10–30
Cool winter	0.95	0–10	10–30
Cold winter	1.51	<0	10–30
Total	6.31		

Bha = 10^9 ha.

use change. From 1850 to 1998, approximately 270 (±30) Pg C has been emitted as CO_2 into the atmosphere from fossil fuel burning and cement production. About 136 (±55) Pg C has been emitted as a result of land use change (IPCC, 2000). Land use change and soil degradation have played an important role in atmospheric enrichment of CO_2 (Lal, 1999). Soil degradation is especially important in drylands of the world where desertification is a serious problem (UNEP, 1992), and food insecurity is a major concern. Reversal of degradative trends in the world's drylands could enhance food security and resequester some of the historic C lost.

The world's drylands, 6.31 billion hectares (Bha) or 47% of the earth's land area, are found in a wide range of climates spanning from hot to cold (Table I). On the basis of rainfall amount and distribution (FAO, 1993), drylands comprise four ecoregions covering land area of 1.0 Bha in hyper-arid, 1.62 Bha in arid, 2.37 Bha in semi-arid and 1.32 Bha in dry sub-humid climates (Table II). Drylands occur in four continents and cover 2.0 Bha each in Africa and Asia, 0.68 Bha in Australasia, 0.76 Bha in North America, 0.56 Bha in South America, but only 0.3 Bha in Europe (UNEP, 1992). Soils of the drylands also vary widely, but are mostly Aridisols (2.12 Bha) and Entisols (2.33 Bha). Dryland soils also include Alfisols (0.38 Bha), Mollisols (0.80 Bha), Vertisols (0.21 Bha) and others (0.47 Bha) (Dregne, 1976; Noin and Clark, 1997). Soils of the dryland regions are characterized by frequent drought stress, low organic matter content, low nutrient reserves, and especially low N content (Skujins, 1991). The Alfisols, Vertisols and Mollisols with a capacity to produce large amounts of biomass under optimal conditions are rather rare in these regions. Drought stress, desertification, low germination and high seedling mortality, and low water and nutrient use efficiencies are among principal constraints to high biomass production in soils of the dryland regions. The world's drylands have been studied extensively (Heathcote, 1983; Dick-Peddie, 1991; Thomas, 1997a,b). Yet, the impact of desertification on the global C cycle and of desertification control on C sequestration in dryland ecosystems have not been widely studied. In this paper I collate and synthesize the available literature

Table II

The extent of global drylands (recalculated from UNEP, 1992). The climatic classification is based on FAO (1993)

Classification	Bha					
	Dry sub-humid	Semi-arid	Arid	Hyper-arid	Total	% of global area
Köppen (1931)	–	1.91	1.61	–	3.52	26.3
Thornthwaite (1948)	–	2.05	2.05	–	4.10	30.6
Meigs (1953)	–	2.11	2.17	0.58	4.86	36.3
Shantz (1956)	–	0.70	3.32	0.63	4.65	34.8
UN (1977)	–	1.78	1.83	0.78	4.39	32.8
UNEP (1992)	1.32	2.37	1.62	1.00	6.31	47.2

Hyper-arid = <200 mm precipitation annually.
Arid = <200 mm of winter rainfall or <400 mm of summer rainfall.
Semi-arid = 200 to 500 mm of winter rainfall or 400 to 600 mm of summer rainfall.
Dry sub-humid = 500 to 700 mm of winter rainfall or 600 to 800 mm of summer rainfall.
Bha = 10^9 ha.

on the impacts of desertification on soil carbon (C) pool and fluxes, and assess the potential of desertification control to sequester C in the soil and diminish the emissions of CO_2 that can lead to greenhouse warming. The objective is to highlight specific processes and provide a few examples in relation to soil C dynamics rather than to compile a comprehensive review on desertification and its control.

2. Extent and Rate of Desertification

The process of desertification has been studied with regards to its impact on production, income and well being of people (Mendoza, 1990; Blaikie, 1989). There is little, if any, information about the impact of desertification on emission of C to the atmosphere. It is in this context that the process of desertification and its control need to be revisited, and critically appraised.

Desertification is defined as 'the diminution or destruction of the biological potential of land which can lead ultimately to desert-like conditions' (UNEP, 1977). While the term can be vague and all encompassing (Verstraete, 1986), a practical or functional definition of desertification implies 'land degradation in arid, semi-arid and dry sub-humid areas resulting from various factors including climatic variations and human activities' (UNEP, 1990; UNCED, 1992). In this context, the term 'land' includes whole ecosystems comprising soil, water, vegetation, crops and animals. The term 'degradation' implies reduction of resource potential by one or a combination of degradative processes including erosion by water and wind and the attendant sedimentation, long-term reduction in the amount and diversity of natural vegetation and animals, and salinization. However, the process of deser-

Table III

GLASOD estimates of desertification (e.g., land degradation in dry areas excluding hyper-arid areas)

UNEP (1991)		Oldeman and Van Lynden (1998)	
Land type	Area (Bha)	Type of soil degradation	Area (Bha)
Degraded irrigated lands	0.043	Water erosion	0.478
Degraded rainfed cropland	0.216	Wind erosion	0.513
Degraded rangelands		Chemical degradation	0.111
(soil and vegetation)	0.757	Physical degradation	0.035
Sub-total	1.016	Total	1.137
Degraded rangeland			
(vegetation alone)	2.576		
Total	3.592	Light	0.489
Total land area		Moderate	0.509
(excluding hyper-arid		Severe and extreme	0.139
regions)	5.172		
% degraded	69.5	Total	1.137

The estimate by Oldeman and Van Lynden does not include the vegetation degradation on rangeland.
Bha = 10^9 ha.

tification is not confined to the drylands of the tropics or economically developing regions alone. It also occurs in developed countries (U.S.A.), high latitude humid ecoregions (Iceland) and even humid regions (tropical rainforest). Desertification in humid areas results mainly from land misuse and soil mismanagement.

Estimates of the extent of desertification range widely and are highly subjective. UNEP estimated 3.97 Bha in 1977, 3.48 Bha in 1984 and 3.59 Bha in 1992 (UNEP, 1977, 1984, 1992). Land area affected by desertification was estimated at 3.25 Bha by Dregne (1983) and 2.0 Bha by Mabbutt (1984). According to the GLASOD methodology (Oldeman and Van Lynden, 1998), land area affected by desertification due to soil degradation is estimated at 1.14 Bha (Table III). These estimates are similar to those by UNEP (1991) with reference to degradation of soil and vegetation. In addition, UNEP's (1991) estimates include 2.58 Bha of degraded vegetation on rangelands (Table III).

As with the area affected, estimates of the current rates of desertification also vary widely. The annual rate of desertification is estimated at 5.8 million hectares (Mha) or 0.13% of the dryland in mid latitudes (Table IV).

Desertification is a biophysical process driven by socio-economic and political factors (Mortimore, 1994; Mainguet and Da Silva, 1998). Two principal biophysical processes leading to desertification are erosion and salinization. Accelerated soil erosion by wind and water are severe in semi-arid and arid regions (Balba,

Table IV
Estimate of annual rate of land degradation in mid latitude drylands (calculated from Mainguet, 1991; UNEP, 1991)

Land use	Total land area (Mha)	Rate of desertification	
		Mha y^{-1}	% of total y^{-1}
Irrigated land	131	0.125	0.095
Rangeland	3700	3.200	0.086
Rainfed cropland	570	2.500	0.439
Total	4401	5.825	0.132

Mha = 10^6 ha.

1995; Baird, 1997), especially those in the Mediterranean climates (Brandt and Thornes, 1996; Conacher and Sala, 1998a,b). Secondary salinization is a major problem on irrigated lands. The irrigated land area in the world has increased 50 fold during the last three centuries which was 5 Mha in 1700, 8 Mha in 1800, 48 Mha in 1900, and 255 Mha in 2000 (Table V). Risks of secondary salinization are exacerbated by use of poor quality water, poor drainage and excessive irrigation, leakage of water due to a defective delivery system, impeded or slow soil drainage, and other causes. Salinization is a severe problem in China, India, Pakistan, and in countries of Central Asia (Babaev, 1999). The extent of land area salinized is 89% in Turkmenistan, 51% in Uzbekistan, 15% in Tadjikstan, 12% in Kyrgyzstan and 49% of the entire region (Pankova and Solovjev, 1995; Esenov and Redjepbaev, 1999). Salinization is also a problem in southwestern U.S.A., northern Mexico and dry regions of Canada (Balba, 1995).

3. Desertification Effects on Soil Quality and the Greenhouse Effect

Soil degradation impacts the global C cycle through its effect on land use change and reduction in vegetation cover that adversely affect top soil depth, and soil quality. There exists a strong link between soil quality, soil organic C content, and desertification. Decline in soil quality leads in reduction in soil organic C pool, and increase in risk, extent and severity of desertification. Further, these adverse effects of decline in soil quality are more severe in hot and dry than in cold and moist environments (Stewart et al., 1990), and are exacerbated by land misuse and soil mismanagement. Decline in soil structure, exacerbated by desertification, leads to emission of C from soil to the atmosphere, and desertification leads to decline in soil structure and reduction in aggregation. For example, in the Mediterranean region, Lopez-Bermudez et al. (1996) observed that aggregate stability of a soil was $53.8 \pm 4.27\%$ under vegetation, $18.4 \pm 13.7\%$ under bare soil and $10.2 \pm 4.2\%$ under cropland. For soil moisture content at pF 1.0, aggregate stability under these

Table V
World irrigated land area (Rozanov et al., 1990; FAO, 1996, 1998; Meyer, 1996; Postel, 1999)

Year	Irrigated area (Mha)
1700	5
1800	8
1900	48
1949	92
1950	100
1959	149
1980	200
1981	213
1984	220
1990	241
1995	255
1997	268

Mha = 10^6 ha.

field conditions was $79.4 \pm 62.1\%$, $74.5 \pm 37.2\%$ and $14.5 \pm 11.2\%$, respectively. Decline in aggregation leads to formation of surface crust, reduction in water infiltration rate, decline in available water reserves in soil, and reduction in biomass production. For example, water infiltration in a southwestern Niger soil was 360 to 600 mm/h for non-crusted soils compared with 2 mm/h for crusted soils (Hammer, 1994; Bleich and Hammer, 1996). Further, crusted soils are prone to water runoff and erosion. These soils have low productivity because of poor stands of plants and stunted growth, low soil-water and nutrient reserves, and shallow effective rooting depth. In Spain, Martinez–Cortizas (1988) observed that soil's available water capacity (AWC) was strongly correlated with % soil organic carbon (SOC) and % of particles <0.02 mm. Desertification decreases AWC due to loss in SOC and the fine-soils fractions (silt, clay). Decline in soil quality adversely impacts agronomic productivity. In Greece, Kosmas et al. (1993) observed that biomass production of rainfed wheat (*Triticum aestivum*) decreased exponentially with reduction in effective soil depth, primarily due to reduction in nutrient and water reserves. Adverse effects of desertification on soil quality include the following (Mainguet and Da Silva, 1998): (i) loss of soil aggregation, (ii) decrease in topsoil infiltration capacity, (iii) reduction in soil-water storage, (iv) loss of resistance to climatic erosivity, and (v) low threshold of runoff initiation. To these must be added: (vi) depletion of soil organic matter content, (vii) difficulty in seed germination and vegetation re-establishment and shift in climax vegetation, (viii) disruption in biogeochemical cycles of C, N, P, S and other elements, (ix) alterations in water and energy balance,

Table VI
Dust deposition rates in the Sahel in different regions and during different time periods (Stahr and Herrmann, 1996)

Region	Sampling time	Deposition ($Mg\ km^{-2}\ y^{-1}$)	Reference
Chad	1966–1967	109	Maley (1980)
Northern Nigeria	1976–1979	137–181	McTainsh and Walker (1982)
Southwest Niger	1987–1989	164–212	Drees et al. (1993)
Southwest Niger	1992–1994	62–186	Herrmann et al. (1994)

and (x) loss of soil resilience. All these effects accentuate emission of C from soil to the atmosphere.

In addition to the impact on soil quality, the quality of biomass produced is also adversely impacted by desertification. In general, in the desertification process productive grasses are replaced by scrub vegetation which increases patchiness and accentuates variability in soil quality (Pickup et al., 1994; Aronsen et al., 1995; Imeson et al., 1996). In Jornada Experimental Range in New Mexico, Schlesinger et al. (1990) observed that the coefficient of variation in soil properties (e.g., pH, % base saturation, total N, soil moisture) was 2 to 12% for grass cover, 4 to 40% for creosote shrub cover, and 5 to 42% for mesquite vegetation. Patchiness and decline in vegetative cover exacerbate susceptibility to inter-rill erosion (Abrahams et al., 1995). In a Mediterranean climate, Lavee et al. (1998) observed that soil organic matter content and structural stability decreases with decrease in rainfall amount and its effectiveness.

Desertification also affects air quality that can adversely affect human and animal health. A decline in soil structure, low soil water reserves and AWC, and low biomass productivity makes the soil more vulnerable to wind erosion. Dust storms are a common phenomenon in drylands regions. The data in Table VI show that annual rate of dust deposition at different locations in sub-Saharan Africa may be 60 to 200 Mg ha^{-1}. The observed rates of dust deposition are 5 to 10 Mg km^{-2} y^{-1} in Australia, 10 to 100 Mg km^{-2} y^{-1} for the Mediterranean region and 13 to 110 Mg km^{-2} y^{-1} for the Middle East (Skujins, 1991; Goudie, 1995; Middleton, 1997). Such dust storms can cause severe damage to crops and infrastructure. At Sadore, Niger, Eltrop et al. (1996a) observed that 40% of the 11-day old seedlings of pearl millet (*Pennisetum glancum*) were completely covered by a severe dust storm.

However, the depositional material can also be a source of nutrients, especially basic cations. Stahr and Herman (1996) reported that Harmattan dust (fine dust blowing from Sahara) contains 0.7 to 5% Ca^{+2} and 0.4 to 1.6% Mg^{+2}. The rate of elemental addition through deposition of dust may be 3.8 to 25.8 kg ha^{-1} y^{-1} for Ca^{+2} and 1.1 to 5.2 kg ha^{-1} y^{-1} for Mg^{+2}. Nutrient addition from outside the ecosystem may have positive effects on productivity and carbon sequestration as secondary carbonates. The net effect of desertification is decline in soil quality, reduction in quantity and quality of biomass produced, decline in air quality, and emission of greenhouse gases and particulate matter into the atmosphere.

4. Depletion of Soil Organic Carbon by Desertification

The SOC pool is usually low in dryland soils. It declines with cultivation and even more so with desertification. Decline in soil quality caused by desertification leads to severe reductions in the SOC pool. In northwestern Nigeria, Raji et al. (1996) observed that soils of the stabilized sand dunes are extremely low in SOC content often in the range of 1 to 2 g kg^{-1}. In East Africa, Swift et al. (1994) reported that continuous cultivation for 14 years without recommended inputs of fertilizers and manures decreased SOC content by half from 2% to 1%. The SOC content was maintained at the antecedent level with application of fertilizers at the recommended rate, use of farmyard manure, and return of crop residue to the soil surface. This drastic decline in SOC content, although not a desertification *per se*, can exacerbate the risks of the on-set of the degradative trends leading to desertification. Similar conclusions were made by Pieri (1991) on the basis of several long-term experiments conducted in sub-Saharan Africa. He showed that continuous cropping without application of fertilizers and/or manure leads to rapid decline in SOC content. The rate of depletion of SOC content is accentuated by soil erosion, because of the preferential removal of the finer soil fractions comprised of clay and organic matter. The SOC is often bound with the clay fraction (Quiroga et al., 1996, 1998), which is preferentially removed by erosion. The C enrichment ratio of the wind-blown sediments in Southeastern Australia was 16 (Leys and McTanish, 1994) and 5 to 10 in Texas, U.S.A. (Zobeck and Fryrear, 1986; Zobeck et al., 1989). In Southwest Niger, Sterk et al. (1996) reported that the wind-blown material trapped at 2-m high above the original soil contained 32 times more C (5.36%) than the topsoil (0.15%).

Adoption of inappropriate land use and practices based on mining soil fertility depletes SOC content, degrades soil structure and sets-in-motion the degradative trends. These trends, if unchecked, accentuate the process of desertification. Therefore, assuming that land degradation around the world has led to an SOC loss of 8 to 12 Mg C ha^{-1} (Swift et al., 1994) on land area of 1.02 Bha (UNEP, 1991), the total historic C loss would be 8 to 12 Pg C. Similarly, if vegetation degradation has led to a C loss of 4 to 6 Mg C/ha on 2.6 Bha, the historic C loss would total 10 to 16 Pg C. Therefore, the total historic C loss due to desertification may be 18 to 28 Pg C. These estimates of historic loss of C are similar to those of Ojima et al. (1993) who estimated that grasslands and drylands of the world have lost 13.1 to 23.6 Pg C due to desertification. Assuming that two-thirds of the C lost (18–28 Pg) can be resequestered (IPCC, 1996) through soil and vegetation restoration, the potential of C sequestration through desertification control is 12 to 18 Pg C. This potential may be realized over a 25- to 50-year period. These estimates provide a reference point with regard to the historic C loss and potential for C sequestration through restoration of soil and biotic ecosystems in desertified lands.

5. Soil Erosion and C Emission in Desertified Lands

Accelerated soil erosion affects the C pool and fluxes because of breakdown of soil aggregates, exposure of C to climatic elements, mineralization of organic matter in disrupted aggregates and redistributed soil, transport of sediments rich in SOC downslope into protected areas of the landscape, and sequestration of C with sediments in depositional sites and aquatic ecosystems. In general, C content of water- and wind-borne sediments is higher than that of the contributing soil. The data in Table VII show estimates of the impact of wind and water erosion on C dynamics. Assuming that 20% of the C displaced is emitted to the atmosphere (Lal, 1995; Lal et al., 1998), erosion (e.g., light, moderate, severe and extreme forms) leads to emission of 0.206 to 0.262 Pg C y^{-1}. Erosion also leads to exposure of the sub-soil rich in calciferous materials. These areas, severely affected by strong and extreme wind erosion, are estimated at about 103.6 Mha. If 10% of these areas have calciferous horizons exposed at the soil surface, about 10 Mha are subject to the impact of anthropogenic perturbations and environmental factors (e.g., plowing, application of fertilizers, root exudates, acid rain, etc.). These factors may lead to dissolution of carbonates and emission of CO_2. If this exposed layer containing high amounts of carbonates and bicarbonates leads to emissions of C at the rate of 0.2 to 0.4 Kg C ha^{-1} yr^{-1}, the annual rate of emissions of C from SIC is 2 to 4×10^6 Kg C y^{-1}. Therefore, total C emission due to soil erosion and exposure of calciferous horizon is 0.21 to 0.26 Pg C y^{-1} (Table VII).

6. Strategies for Desertification Control to Sequester Carbon

Biomass productivity in drylands is limited by lack of water and plant nutrients. Therefore, an important strategy lies in growing xerophytic plants and adopting techniques that enhance water and nutrient use efficiencies and improve biomass productivity.

6.1. VEGETATIVE COVER

Removal of vegetative cover exacerbates the soil erosion problem (Castillo et al., 1997). Thus, establishing a vegetative cover is the key to controlling soil erosion,

Table VII
Estimates of C emission by soil erosion in desertified lands

Degree	Total area affected by wind and water erosion (Mha)	Presumed rate of soil erosion[a]		Total amount of sediments displaced ($Pg\ y^{-1}$)[b]	Total C displaced ($Pg\ y^{-1}$)[c]	C emission to the atmosphere ($Pg\ C\ y^{-1}$)[d]
		Multiple of T value	Rate ($Mg\ ha^{-1}\ y^{-1}$)			
Slight	372.3	1.25	14	52.1	0.52	0.08–0.10
Moderate	423.9	1.5	17	72.1	0.72	0.11–0.14
Strong[e]	97.0	2.0	22	21.3	0.10	0.015–0.02
Extreme[e]	6.6	3.0	34	2.2	0.01	0.0015–0.002
Sub-total						0.206–0.262
Total						0.21–0.26

Assumptions
[a] T value is 11.2 $Mg\ ha^{-1}\ y^{-1}$ (T refers to tolerable soil loss, estimated to be 12.5 $Mg\ ha^{-1}\ y^{-1}$).
[b] A delivery ratio of 10%.
[c] SOC content of 1% for slight and moderate erosion and 1.5% in sediments of strong and extreme erosion.
[d] C emission at 15–20% of SOC displaced.
[e] It is possible that strong and extreme erosion lead to erosion of sub-soil with low SOC content. In such cases, the loss of C by erosion would be less.

Table VIII

Differences in SOC content and CEC of soil under shrubs and the inter-shrub area in semi-arid woodlands of Australia (Tongway and Ludwig, 1990)

Depth (cm)	Soil organic C content (%)		CEC (cmol(+)/kg)	
	Shrub	Inter-shrub	Shrub	Inter-shrub
0–1	1.97	0.71	10.5	7.4
1–3	0.95	0.43	9.4	6.6
3–5	0.71	0.40	9.2	8.4
25–50	0.30	0.23	7.9	9.2

improving soil quality and increasing SOC contents. C_4 and CAM (Crassulacean Acid Metabolism) plants provide advantages in this regard (Lal et al., 1999). The vegetation or ground cover in drylands comprises two well-defined zones: the area under the shrub and the inter-shrub or the open space without any vegetative cover. Vegetative cover leads to improvement of soil quality that is generally superior under the shrub than under bare inter-shrub areas. The bare inter-shrub area is characterized with sealed surfaces that are quite impermeable and absorb very little rain (Dunkerley and Brown, 1997). In contrast, soil under vegetated cover is generally more porous, more organically rich, with a high infiltration rate. Therefore, with 50:50 surface area covered by vegetation, the effective rainfall received in the vegetated areas is twice the normal rain. Consequently, soils have different leaching rates, SOC content, salinity, and structural features within meters of each other.

In the Mexican Chihuahuan Desert, Montaña et al. (1988) reported that the mean SOC content was 1.50% under vegetation patches but only 0.46% in the intervening bare spaces. These differences were observed over spatial scales of 10–100 m. In western Australia, Mabbutt and Fanning (1987) observed that beneath vegetative bands 10–20 m wide, a siliceous hard pan was typically located more deeply within the soil beneath shrubs than beneath inter-shrub areas. The depression of the hardpan acted as a ‘trench’ beneath the shrubs and trapped the infiltrating water.

In NSW Australia, Tongway and Ludwig (1990) observed that SOC content and cation exchange capacity (CEC) of soil in the upper 0.05 m layer were higher under the shrub than in the inter-shrub area (Table VIII). Consequently, experiments with simulated rains showed that runoff began after only 7 minutes of rain at 29 mm h^{-1} in the inter-shrub area, while the mulga grove soils showed no runoff. High infiltration rate was related to high SOC content under shrub than inter-shrub areas (Table VIII). In Argentina, Bravo et al. (1995) observed 29% higher mean weight diameter of soil aggregates and 29% more SOC content under grass cover than cropland prone to desertification. At the Jornada Experimental Range in New

Table IX
Effect of grazing on herbaceous plant biomass in desertified area of Chios (recalculated from Margaris et al., 1996)

Treatment	Biomass (kg ha^{-1})	Plant species
With grazing	290	22
Without grazing for:		
3 years	390	24
6 years	650	24
9 years	900	29
12 years	1090	37

Mexico, Herrick et al. (1999) observed that infiltration capacity was higher under grass than in bare soil. In southwestern U.S.A., Bedunah and Sosebee (1986) observed that eradicating mesquite (*Prosopis gladulosa*) increased herbaceous cover of useful klein grass (*Panicum coloratum*). Other important herbaceous species that are proven to improve vegetative cover and enhance soil quality in southwestern U.S.A. are black grama (*Bouteloua eriopoda*), blue grama (*B. gracilis*), and tabosa grass (*Hilaria mutica*). In addition to grasses, there are several multipurpose trees which are useful for establishing shelter belts (*Neem* or *Acacia* spp), reinforcing river banks (*Eucalyptus* or *Populus*), animal fodder (*Leucaena, Acacia, Dalbergia*) and fuel wood (*Prosopis* spp.).

6.2. CONTROLLED GRAZING

Excessive stocking rate and uncontrolled grazing are important factors that accentuate risks of desertification. In arid and semi-arid regions of Botswana and Zimbabwe, Abel and Blaikie (1989) observed that low stocking rate and controlled grazing improved biomass productivity and enhanced floral biodiversity. Experiments conducted by Margaris et al. (1996) in the Mediterranean region showed that both biomass and number of species diversity increased with elimination of grazing (Table IX). In the Kalahari Desert, Wiggs et al. (1994) observed that denudation by over-grazing, burning or drought led to a 3-fold increase in dune movement and wind erosion. In the Rajsthan desert of India, Kumar and Bhandari (1992, 1993) observed a severe decline in vegetal cover in the grazed sand dune areas.

6.3. WATER CONSERVATION BY RESIDUE MANAGEMENT AND MULCHING

Decreasing water losses by runoff and evaporation is critical to enhancing biomass productivity. Beneficial effects of establishing stone bunds on the contour in decreasing losses by runoff have been documented in sub-Saharan Africa (Lamachère and Serpantie, 1991), and in Algeria by the use of appropriate crop rotations and sylvo-pastoral systems (Arabi and Roose, 1992; Roose, 1996).

Residue management and choice of appropriate tillage methods are also important to increasing water use efficiency. In Niger, Eltrop et al. (1996b) reported that application of 2 Mg ha^{-1} of crop residue mulch decreased soil erosion by about 50%. Also, in Niger, Michels et al. (1995) observed that surface application of millet residue mulch at 2 Mg ha^{-1} significantly reduced wind erosion and increased SOC content and CEC of the surface layer. An additional benefit of erosion control by mulching is decreased loss of water by runoff and evaporation. Residue mulch decreases soil temperature which contributes to a reduction in evaporation. In Niger, Buerkert et al. (1996a,b) showed that crop residue mulch decreased maximum daily temperature at the 0.1 m depth by 8 °C. The temperature at this depth in unmulched bare soil was 50 °C. Beneficial effects of crop residue mulch on the soil quality of drylands have been reported from Burkina Faso (Mando, 1997), Niger (Sterk et al., 1996), southern India (Badanur et al., 1990) and China (Li et al., 1994). Spreading crop residue and dead wood etc. also stimulates termite activity which improves soil structure, and increases SOC and SIC contents (Lal, 1987).

6.4. SUPPLEMENTAL IRRIGATION

While there may be little potential for expanding irrigation in Asia, the potential for expansion of irrigable cropland area in sub-Saharan Africa may be 39 Mha (Hillel, 1997). This potential needs to be realized through development of small-scale irrigation projects involving use of ground water, runoff storage through water harvesting, micro-catchment farming, and other cost-effective and simple watershed management techniques (Essiet, 1990). There is great potential to improve irrigation efficiency in drylands. Wasteful flood irrigation systems should be replaced by what Hillel (1997) calls the 'HELPFUL' system (high frequency, efficient, low volume, partial area, farm unit, and low cost). A principal objective is to improve the water use efficiency, defined as the amount of vegetative dry matter produced per unit volume of water taken up by the crop from the soil (Viets, 1962). It is reciprocal of the transpiration ratio, or the amount of water transpired per unit mass of dry matter produced.

6.5. SOIL FERTILITY MANAGEMENT

Soil fertility improvement is essential to enhancing biomass productivity, increasing water use efficiency, and improving soil quality. Several long-term experiments in drylands have demonstrated the importance of judicious use of fertilizer, compost, and nutrient management (Fuller, 1991; Traore and Harris, 1995; Singh and Goma, 1995; Pieri, 1995; Miglierina et al, 1996; Laryea et al., 1995). Diaz et al. (1997) monitored the impact of urban wastes (biosolids) on soil quality in semi-arid areas of Spain. Plant cover and biomass increased substantially. The urban biosolids proved effective as an amendment to regenerate the plant cover on degraded soils. Nambiar (1995) summarized data from several long-term soil fertility management experiments conducted in India. For sandy soils of Ludhiana, Punjab, the SOC content with manuring increased from 0.20% in 1971 to 0.25% in 1989. With application of N, P, K, S, and manure, the SOC content doubled to 0.40%. For a clayey soil at Jabalpur, central India, the SOC content increased from about 0.6% to 1.1% with recommended soil fertility management. Jangir et al. (1997) observed that yield of pearl millet in an arid region of Jodhpur, Rajsthan, increased with application of N fertilizer up to 40 Kg N ha^{-1} under rainfed conditions and 80 Kg N ha^{-1} with supplemental irrigation. In Gurdaspur and Hissar, Mishra et al. (1974) reported that application of manure at the rate of 9–30 Mg ha^{-1} y^{-1} caused significant increase in SOC content. Similar observations were made by Ruhal and Singh (1982). Chaudhary et al. (1981) reported that SOC content increased by 0.033%, 0.042% and 0.143% by applications of 13 Kg of P, 26 Kg P, and 15 Mg ha^{-1} of manure to a pearl millet-wheat (*Triticum aestivum*) rotation. Muthuvel et al. (1989) reported that application of manure to a cotton (*Gossipium hirsutum*)-pearl millet rotation in dryland Vertisol increased SOC content and crop yield. For Vertisols in the Ethiopian Highlands, Wakeel and Astartke (1996) recommend adoption of improved agricultural practices (e.g., fertilizer use, water conservation, new varieties and cropping systems) to minimize risks of soil degradation.

The beneficial impact of improved soil fertility on SOC content has been observed elsewhere in dry areas prone to desertification. In the Kalahari Desert of Botswana, Perkins and Thomas (1993) reported that SOC content of soil near a water-well where animals congregate ranged from 4.8 to 7.6% compared with 0.4 to 0.6% about 1000 m away. Given the input of organic matter, therefore, SOC content can be increased by an order of magnitude even in harsh environments.

It is also important to recognize that nitrogenous and other fertilizers are based on C-input (Schlesinger, 1999). Some management practices have low N use efficiency and can lead to emission of N_2O and NO (Sahrawat et al., 1985). Factors affecting N_2O emission include soil pH, organic matter content, soil temperature, and soil moisture regimes. There are soil, water, crop residues, and fertilizer management practices that can decrease the emission of N_2O and NO.

6.6. IMPROVED FARMING SYSTEMS

Unproductive and inefficient farming systems must be replaced by efficient and productive systems if soils are to be protected against deforestation. Important components of farming systems are crop rotations, fallowing, agroforestry, and grazing management.

6.6.1. *Crop Rotations*

Importance of crop rotations in reversing soil degradative trends is even more prominent in harsh arid and semi-arid environments than in humid ecoregions. Incorporating legumes in the rotation cycle, especially those with a deep and prolific root system and a high capacity to fix nitrogen is an important strategy to enhance soil quality. Choice of an appropriate rotation is also critical to adoption of a conservation tillage system, whose effectiveness in soil and water conservation in arid and semi-arid regions depend on the amount of surface area covered by crop residue mulch. In semi-arid regions of Argentina, Galantini and Rosell (1997) reported that rotations of mixed pasture (5.5 years) and annual crops (4.5 years) maintained 17.3 Mg ha^{-1} of SOC compared with 11.2 Mg ha^{-1} in continuous cultivation with a wheat-sunflower (*Helianthus annuus*) rotation. In another study, Miglierina et al. (1993, 1996) observed that SOC content was high in wheat-grassland and wheat-alfalfa (*Medicago sativa*) rotations, especially with a conservation tillage system. In eastern Bolivia, Barber (1994) observed that sub-soiling and incorporation of cover crops in rotation enhanced soil quality. In Saudi Arabia, Shahin et al. (1998) observed that introducing alfalfa in rotation with wheat grown on a sandy soil decreased salinity and increased SOC content three fold as compared with continuous wheat. In Maharashtra, India, Lomte et al. (1993) reported that intercropping sorghum (*Sorghum bicolor*) with legumes and application of manure increased SOC content and aggregation.

6.6.2. *Planted Fallows*

Taking land out of agricultural production and permitting natural vegetation to grow leads to restoration of degraded soils. In contrast, summer fallowing that has no vegetal cover can cause severe erosion and accentuate the desertification process. Bush fallowing is most widely practiced in tropical and sub-tropical agriculture (Nye and Greenland, 1962). In northern Nigeria, Abubakar (1996) monitored the impact of duration of fallowing on changes in SOC content. The mean SOC content of the surface soil was 0.94% for 2-year fallow, 1.13% for 5-year fallow, 1.42% for 10-year fallow and 1.44% for 15-year fallow. SOC content of the sub-soil also increased. Data from a 30-year fallowing experiment conducted in eastern Spain showed that SOC content stayed more or less constant at a low level for several years. Subsequently, SOC content in the top 10-cm layer increased significantly. A slight increase also occurred in 20–30 cm depth. The increase in SOC content was notable after 20 years of fallowing. Although natural regeneration can enhance soil quality (Ruecker et al., 1998), fallowing efficiency can be greatly improved by the use of appropriate cover crops (Barber and Navarro, 1994).

6.6.3. *Forestry and Agroforestry Measures*

Widespread deforestation for fuel wood and other domestic uses accentuates the impact of harsh dry environments (Boahene, 1998). Lack of household energy for cooking is a major constraint among rural communities in dry areas of developing countries. Deforestation for fuel wood has led to denudation of landscape and exacerbated risks of erosion and desertification. Therefore, afforestation is an important strategy to restore vegetal cover, restore degraded ecosystems and grow household fuel. There are several multi-purpose trees which can grow under the harsh environments of drylands, improve soil quality (albeit slowly), sequester C in the soil and biomass, and produce fuel wood. Kair (*Capparis decidua*) is one such tree adaptable to the drylands of northwest India (Gupta et al., 1989). Growing mesquite (*Prosopis* spp) has been useful in reclaiming salt-affected soils in India. Some promising species for fuel wood production, soil quality improvement and desertification control include *Tamarix, Eucalyptus, Leucaena, Cupressus, Casuarina, Capparis, Prosopis, Azadirachta, Acacia, Tectona, Cassia, Dalbergia, Khaya, Albizia, Parkia, Terminalia, Pongamia, Sesbania, Morus* and *Populus* (Le Houerou, 1975; Gupta et al., 1989; Lattore, 1990; Mainguet, 1991; Singh et al., 1994; Alstad and Vetaas, 1994; Singh and Singh, 1995; Patil et al., 1996). In Nigeria, Jaiyeoba (1998) monitored changes in soil properties related to conversion of savannah woodland into pine (*Pinus oocarpa*) and eucalyptus (*Eucalyptus camaldulensis*) plantations. The SOC content of 0–0.15 m depth showed a declining trend during initial stages of tree establishment, then an increase and finally a steady equilibrium value. The equilibrium value was attained in about 16 years. The initial decline was apparently due to an essential lack of ground cover, and low biomass production. Growing grass or cover crop in association with trees is a useful strategy. Garg (1998) monitored changes in properties of a sodic soil under four different tree species in north-central India. Results showed a marked improvement in soil fertility in general but SOC content in particular. The SOC pool increased from less than 10 to about 45 Mg ha^{-1} over an 8-year period. Dry soil bulk density of the 0–0.15 m layer of the unplanted site was 1.8 Mg m^{-3} compared with 1.61 Mg m^{-3} under *Acacia nilotica*, 1.50 Mg m^{-3} under *Dalbergia sissoo*, 1.43 Mg m^{-3} under *Prosopis juliflora*, and 1.55 Mg m^{-3} under *Terminalia arjuna*.

6.6.4. *Grazing Management*

Excessive and uncontrolled grazing are a major cause of the acceleration of the desertification process. Pluhar et al. (1987) conducted grazing experiments at the Texas Experimental Ranch near Throckmorton, Texas. They observed that the water infiltration capacity increased as vegetal cover increased, soil bulk density decreased, and soil organic matter content increased. Grazing caused a significant decline in infiltration capacity by reducing the protective vegetal cover and increasing the surface area of the bare ground. Thurow et al. (1988) also observed that infiltration capacity decreased and inter-rill erosion increased in the heavily stocked pastures. In Alice, Texas, Weltz and Blackburn (1995) observed that the saturated hydraulic conductivity was the least for the bare soil. Biomass burning also affects soil hydrological properties. Experiments conducted at the Edwards Plateau, Texas, by Hester et al. (1997) showed that fire reduced water infiltration capacity in case of the oak and juniper vegetation types. Burning increased hydrophobic properties of the soil. Therefore, controlled grazing, fire management, and planting improved species are important considerations of enhancing biomass production and improving soil quality.

7. Assumptions Made in Calculating Potential of Soil C Sequestration

Estimates of soil C sequestration presented in this report are based on numerous assumptions, most of which are supported by published literature. Principal assumptions are the following:

1. Restoration of degraded soils and ecosystems improves soil organic carbon content through increase in net primary productivity and biomass returned to the soil. This assumption is validated through some long-term experiments conducted in different dry regions of the world (Gupta and Rao, 1994; Velayutham et al., 2000; Pal et al., 2000; Swarup et al., 2000; Pan and Guo, 2000; Bationo et al., 2000; Dalal and Carter, 2000).
2. Rate of SOC sequestration under recommended agricultural practices differ among soils, cropping systems, and ecoregional characteristics (Table X).
3. Experimental data in support of these rates are scarce and not available for numerous soils, cropping systems and ecoregions. Even with the limited data available, an appropriate strategy is to extrapolate on the basis of soil mapping units using GIS techniques. However, the estimates computed in this report are based on the use of average numbers applied to large areas on a global basis.
4. The rates of soil C sequestration computed in this report are gross rates, and the hidden costs of C sequestration (Schlesinger, 1999) are not computed. Improved management practices (e.g., fertilizer and herbicide use, irrigation, liming, plowing) are based on use of C-based input. Further, some of these practices (e.g., use of nitrogenous fertilizer) can lead to emission of non-CO_2 greenhouse gases (e.g., N_2O, NO_x). The net global warming potential (GWP) may be computed on the basis of all outputs and inputs as shown in Equation (1) and (2).

$$\text{Net GWP of a practice} = \{(\text{soil C gain}) - (\text{input of C}) - (\text{GHG emissions})\}/\text{time}\ldots. \quad (1)$$

$$\text{Net GWP of a practice} = \{(?\text{SOC} + ?\text{SIC}) - (\text{carbon used in fertilizer} + \text{lime} + \text{herbicides} + \text{fuel} + \text{irrigation}) - (\text{C equivalents of } N_2O + CH_4)\}/\text{time}\ldots. \quad (2)$$

In these equations, changes in SOC and SIC pools need to be monitored to 2-m depth over a time period of 5 to 10 years.

Table X

Rates of soil organic carbon (SOC) sequestration for adoption of recommended agricultural practices and restoration of desertified soils and vegetation

Management system		Change in C pool (Mg/ha/yr)			References
		SOC	SIC	Vegetation	
A.	*Restoration of desertified lands*				
1.	Erosion control on strongly and extremely degraded areas	0.04–0.06	–	2–3	Dalal and Carter (2000); Lal et al. (1999); Bhojvaid and Timmer (1998)
2.	Adoption of recommended agricultural practices on moderately eroded lands	0.08–0.12	–	–	Swarup et al. (2000); Bationo et al. (2000); Dalal (1989)
3.	Physical and chemical degradation	0.04–0.06	–	–	Lal et al. (1998); Batjes and Sombroek (1997); Fullen and Mitchell (1994)
4.	Soil salinity control	0.2–0.4	–	3–4	Bhojvaid and Timmer (1998); Singh (1998); Singh et al. (1994, 1997); Garg (1998); Farrington and Salama (1996); More (1994); Batra et al. (1997)
B.	*Fossil fuel offset through biofuel production*	–	–	2–3	Bhojvaid and Timmer (1998); Marland and Turhollow (1991); Paustian et al. (1998)
C.	*Secondary carbonates*	–	0.11–0.12	–	Monger and Gallegos (2000); Nordt et al. (2000); Wilding (1999)

5. It is assumed that C sequestered in soils and ecosystems has a long residence time. The latter depends on continuous use of recommended agricultural practices, some of which may be carbon neutral (Robertson et al., 2000) because of C-based inputs and emission of N_2O and CH_4.
6. It is assumed that afforestation, establishment of fast growing trees on restored/reclaimed lands, would enhance C sequestration in soil and biomass (Bhojvaid and Timmer, 1998). It is possible, however, that in some ecoregions, afforestation may not lead to net C sequestration (Schulze et al., 2000).
7. An important assumption is that recommended agricultural practices will be adopted, inputs are available to implement land restorative measures, and incentives are provided to facilitate adoption of improved practices. While the potential of C sequestration in soil and biomass is large, the challenge to implement the recommended practices is also the greatest.
8. The socioeconomic and political impacts (Shiva, 1991) and ecological concerns (Ehrlich, 1993) of agricultural intensification are well documented. Risks of irrigation are also known (Postel, 1999). Yet, degraded and desertified soils and ecosystems have to be restored to enhance biomass productivity and minimize adverse impacts on the environment.
9. The economics of desertification control is another issue. It is evident that C sequestration is not economical in all ecoregions. It is important to identify bright spots (i.e., economically feasible, socially acceptable and politically permissible) of C sequestration.
10. The process of C sequestration in soil and ecosystems has both positive and negative feedbacks. Positive feedbacks are those that relate to win-win scenarios, leading to enhancement in productivity and improvement in environment quality. The strategy is to enhance positive feedbacks and minimize negative feedbacks.

There is also a question of tradeoffs between irrigation and use of non-native species with traditional practices. However, an important underlying assumption in restoring desertified lands and adoption of recommended agricultural practices is the overriding need for enhancing food production in these ecoregions. Restoring desertified lands and adopting recommended agricultural practices would improve food production, and an important effect of these is C sequestration in soil and ecosystems.

8. Potential of Improved Cropping Systems and Agricultural Intensification for C Sequestration

Adoption of improved/recommended practices on agricultural and pastoral lands is an important strategy for agricultural intensification. Recommended practices include those for soil-water conservation and management, irrigation management, soil fertility enhancement including inorganic fertilizers and organic amendments, residue management and tillage methods, improved varieties and associated cropping systems, improved systems of grazing land management, and integrated pest management (IPM) including effective weed control measures.

Programs of agricultural intensification can be applied to prime agricultural land, the land with slight or no degradation, and on restored lands that have been reclaimed. Adoption of intensive agricultural practices is a viable option only on lands whose soil quality responds favorably to input. Further, it is important to avoid double accounting and not consider restored degraded lands twice. Long-term experiments conducted on Vertisols in southern India showed that practices leading to agricultural intensification increased SOC content in the 0.2 m layer and improved yield of groundnut (*Arachis hypogea*). The SOC pool in that layer increased from 13.1 Mg ha^{-1} for the control to 15.4 Mg ha^{-1} with 30 Mg ha^{-1} of manure applied (Ismail et al., 1998).

The global land area with slight or no degradation is estimated at 427.3 Mha. Although these lands are better left alone, high demographic pressure and scarcity of agricultural land necessitates use of those lands for enhancing production. Assuming that adoption of improved land use and farming/cropping systems may lead to an increase in SOC content in these soils at the rate of 30 to 50 kg ha^{-1} y^{-1} (Galantini and Rosell, 1997; Miglierina et al., 1993, 1996; Shahin et al., 1998), the total potential of C sequestration is 12.8 to 21.4 $\times$ 106 Mg C y^{-1} (0.013 to 0.021 Pg C y^{-1}).

9. Restoration of Desertified Lands and C Sequestration

Principal categories of degraded soils with potential for C sequestration through restoration and soil quality enhancement include: (i) eroded soils, (ii) physically and chemically degraded soils, and (iii) salt-affected soils. There are also dry lands disturbed by mining. Restorative measures for such lands are described elsewhere (Allen, 1988).

9.1. POTENTIAL OF EROSION CONTROL MEASURES IN DRY AREAS FOR CARBON SEQUESTRATION

The land area affected by strong and extreme soil erosion is 104 Mha and an additional 424 Mha is affected by moderate levels of soil erosion. Strongly and extremely degraded areas (104 Mha) need to be taken out of agricultural/pastoral land uses and put under restorative measures. Trees, shrubs and perennial grasses can be grown on these lands for preventing soil erosion, and for generating biofuel. Trees may grow in humid, sub-humid and semi-arid regions. However, many arid and semi-arid ecoregions may not support trees. Where feasible, establishing suitable tree species would have three distinct benefits: (a) providing the much needed ground cover and root system for protecting the soil against erosive forces of wind and water, (b) producing biomass that can be used as fuel, and (c) enhancing SOC content and sequestering C in soil. Species with deep root systems anchor the soil and protect the seedlings against drought stress.

Once established (2 to 3 years after planting/sowing), the above-ground biomass production potential in semi-arid (500–750 mm annual rainfall) and sub-humid (750–1000 mm annual rainfall) environments is 2 to 3 Mg ha^{-1} y^{-1} with a total production of 208 to 312 million Mg y^{-1}. With a fuel efficiency of 0.7 for direct use (relative to 1 for fossil fuel), the biofuel C offset for these lands is 0.14 to 0.21 Pg C y^{-1}. In addition, some of the below-ground biomass produced is converted to humus or SOC content. Assuming a low rate of increase of SOC content at 40–60 kg C ha y^{-1}, the potential of C sequestration in soil is 0.004 to 0.006 Pg C y^{-1}.

There are 424 Mha of moderately eroded lands (Oldeman, 1994) in dry areas. Recommended agricultural management practices should be adopted on these lands for soil erosion control and soil quality enhancement. Such practices include installation of engineering devices (e.g., diversion channels, drop structures, infiltration ditches, stone filters or gabiens, terraces) and adoption of appropriate soil (tillage methods, soil fertility management, residue management, soil amendments) and crop management systems (e.g., rotations, agroforestry techniques, integrated pest management etc.). Some simple and proven technologies include contour plowing, strip cropping and windbreaks. In Greece, Kosmas et al. (1997) observed that runoff and sediment losses were greatly reduced by contour plowing compared with up and down-hill farming. To be effective, however, contour farming has to be used in conjunction with other practices (e.g., strip cropping, conservation tillage). In Chile, Raggi (1994) reported higher yields of wheat and lentils (*Lens esculenta*) and higher profits with minimum tillage and direct seeding than with conventional tillage. In Calleguas Creek, California, USDA (1995) recommends use of several practices for erosion control including conservation tillage, contour farming, cover crops, critical area planting, crop residue use, diversion channels, filter strips, grade stabilization structures, grass waterways and others. Adoption of improved practices controls soil erosion, improves crop yield, enhances soil quality and sequesters C through improvement in SOC content. However, enhancing SOC content in dry climates is a challenging task, and the rate of increase in SOC may be low even under the ideal conditions. The rate of SOC sequestration is likely to be low in hot and dry regions and relatively high in moist and cool ecoregions. Further, the effects of climate on SOC sequestration is also modified by soil profile characteristics and the landscape position. For the same rainfall regime, the SOC sequestration potential is more for heavy-textured than coarse-textured soils, and for foot slope than shoulder or side slope positions. Because of a wide range of climate, soils, and landscape conditions, only average rates are used in these calculations. Therefore, assuming the rate of increase in

Table XI
Potential of C sequestration through restoration of degraded soils

Technological options	C sequestration potential ($Pg\ C\ y^{-1}$)
1. Restoration of eroded soils	
(a) Erosion control (50–75% of emission reduction)[a]	0.13–0.20
(b) Restorative plantation on strongly/extremely eroded soils	0.004–0.006
(c) Adoption of recommended practices on slightly/ moderately eroded soils	0.06–0.10
2. Fossil fuel off-set	0.14–0.21
Total	0.33–0.52

[a] Adoption of conservation-effective measures would lead to reduction in emission of CO_2 due to erosion caused displacement of soil carbon and its enhancement mineralization.

SOC content to be 80 to 120 Kg C ha^{-1} y^{-1}, the total potential of C sequestration in moderately eroded soils is 0.03 to 0.05 Pg C y^{-1}.

The total potential of C sequestration through soil erosion management is shown in Table XI. Adoption of erosion control measures and recommended management practices would decrease C emissions from erosion-displaced sediments by as much as 50 to 75% of the estimated emission (0.13–0.20 Pg) (Lal, 1995). Converting strong and extremely eroded lands to restorative plantation can lead to fossil fuel off-set at 0.14 to 0.21 Pg C y^{-1}, and SOC sequestration of 0.004 to 0.006 Pg C y^{-1}. Because the hidden C costs of restorative measures are important, the rates of C sequestration used must reflect these costs. Adoption of recommended agricultural practices on slightly and moderately eroded lands (Oldeman, 1994) has a potential to sequester 0.06 to 0.10 Pg C y^{-1}. Thus the total potential of C sequestration through soil erosion management in dry areas is about 0.33 to 0.52 Pg C y^{-1}.

9.2. PHYSICAL AND CHEMICAL DEGRADATION

In addition to erosion, drylands also are prone to physical and chemical forms of soil degradation. Soil physical degradation involves decline in soil structure leading to crusting, compaction, hard-setting and exposure of plinthite that eventually hardens into a laterite (Lal et al., 1989). Soil chemical degradation includes fertility depletion, nutrient imbalance (toxicity and deficiency), and acidification. The land area affected by strong and extreme forms of physical and chemical degradative processes is 34 Mha (UNEP, 1991). Similar to severely eroded lands, these lands may also be taken out of production and converted to restorative land use through planting appropriate trees, shrubs, and perennial grasses while increasing some hidden C costs of adopting such measures. In addition to production of biofuel, soil restoration would also improve SOC content, albeit at a low rate of about 40 to 60 kg ha^{-1} y^{-1}. Therefore, the potential of SOC sequestration in these soils is 0.001 to 0.002 Pg y^{-1}.

In addition, there are 46 Mha of moderately degraded soils. Assuming that adoption of recommended practices on these soils may lead to C sequestration at the rate of 80 to 120 kg ha^{-1} y^{-1}, the potential of SOC sequestration on these lands is 0.004 to 0.006 Pg C y^{-1}. Therefore, total potential of C sequestration through restoration of physically and chemically degraded soils is 0.005 to 0.008 Pg C y^{-1}.

9.3. SOIL SALINITY CONTROL

There are about 930 Mha of salt-affected soils in the world (Sumner et al., 1998; Sumner and Naidu, 1998). Therefore, reclamation of salt-affected soils is an important aspect of soil quality enhancement and increasing C sequestration in above-ground biomass and in the SOC pool. There are numerous proven technologies for reclaiming salt-affected soils (Gupta and Abrol, 1990). Experiments conducted at the Central Soil Salinity Research Institute, Karnal, India, have demonstrated the importance of applying gypsum and organic manures and of leaching and growing appropriate plants for salinity control. The strategy is to enhance SOC content, improve soil structure and infiltration rate, and replace Na^+ adsorbed on clay minerals with Ca^{2+} and Mg^{2+}. Wilding (1999) observed that a major mechanism of sequestration of inorganic carbon is via movement of HCO_3 into ground water or closed systems that have limited exchange with ambient environments. There is a total of 19 Mha of irrigated land in the U.S.A. (Solley et al., 1993), 50 Mha in China, 48 Mha in India, 17 Mha in Pakistan and 9 Mha in Iran (FAO, 1998; Suarez, 2000). Pan and Guo (2000) observed that the importance of irrigated agriculture and of secondary carbonates in C sequestration in China is high, and can be enhanced by adoption of recommended practices. Secondary salinization is an extremely serious problem in central Asia. Saline soils account for 89% of total irrigated area of 1.22 Mha in Turkmenistan, 51% of 4.14 Mha in Uzbekistan, 15% of 0.7 Mha in Tadjikistan and 11.5% of 1.0 Mha in Kyrgyzstan (Esenov and Redjepbaev, 1999). Reclamation of these soils would enhance productivity, improve soil quality and sequester carbon. In Australia, 15 Mha of cropland is threatened by soil salinity (Barrett–Lennard, 2000). Moezel et al. (1988) observed that several tree species are tolerant to high salinity and waterlogging. These species have potential to sequester C in otherwise unproductive soils.

Growing salt-tolerant (halophytic) plants can improve above- and below-ground biomass production and increase SOC content. Singh (1989) observed that among several fuelwood species evaluated, *Prosopis juliflora* was most adapted to alkaline soils and produced the most biomass. *Sesbania sesban* and *Tamarix dioca* also exhibited good adaptability. Singh et al. (1994) observed that growing salt-tolerant woody species improved soil quality. Among these species *Prosopis juliflora,* *Acacia nilotica, Casurina equisetifolia, Tamarix articulata, Leptochloa fusca*, and some other species caused a notable increase in SOC content. The original SOC content of this soil was 0.24%. SOC content in the 0–0.15 and 0.15–0.30 m layers was, respectively, 0.85% and 0.55% for *Acacia nilotica*, 0.66% and 0.33% for *Eucalyptus tereticornis*, 0.93% and 0.58% for *Prosopis juliflora*, 0.86% and 0.58% for *Terminalia arjuna*, and 0.62% and 0.47% for *Albizzia labbek*. In addition to production of fuel wood, there are also suitable fruit trees that are adaptable to highly alkaline soils. Adaptable fruit trees for northwestern India include jamun (*Syzygrum cuminii*), tamarind (*Tamarindus indica*), ber (*Zizyphus mauritiana*), and guava (*Psidium guajava*) (Singh et al., 1997). Improvements in SOC content can set-in-motion restorative trends leading to increase in soil microbial activity and biomass carbon (Ragab et al., 1990). In north central India, Garg (1998) observed significant improvements in SOC content of a sodic soil through planting salt-tolerant tree species over an 8-year period. The SOC pool in the top 0.6 m layer increased with duration after tree planting. The rate of increase was low for the first 2 to 4 years, high (exponential) between 2 and 6 years, and stabilized at a low rate of further increase between 6 and 8 years.

Salt-tolerant trees and forage species have also proven useful in reclaiming sodic soils of Pakistan (Qadir et al., 1996). In addition to improving structural properties, trees also affect salt balance by increasing the depth of the water table leading to a net downward leaching. In southwestern Australia, Farrington and Salama (1996) recommended that revegetation by trees and shrubs be used to control dryland salinity.

Application of manure and gypsum is also important to improving soil structure and reclaiming sodic soils. Use of manure and compost is facilitated by integrating livestock with the cropping system (Chaudhary et al., 1981; Haque et al., 1995; Harris, 1995; Pieri, 1995). In Maharashtra state in India, More (1994) reported that application of farm by-products and organic manures improved quality of sodic Vertisols, enhanced SOC content and increased crop yields. Batra et al. (1997) observed the impact of growing karnal grass (*Leptochloa fusca*) and of applying gypsum in reclamation of an alkaline soil. There were significant improvements in soil quality (e.g., infiltration rate; SOC content; aggregation) within 3 years of applying these treatments. Singh et al. (1988, 1989) and Singh and Singh (1995, 1996) observed that application of gypsum and farmyard manures enhanced the survival rate of mesquite (*Prosopis juliflora*) on a highly alkaline soil. In southern Australia, Emerson (1995) observed that soil-water retention at low suction increased almost linearly with SOC content and independently of clay content. Creating and maintaining preferential flow paths are important to leaching soluble salts out of the root zone. Heavily grazed areas are prone to compaction and a decline in the infiltration rates. In the Pampa region of Argentina, Dreccer and Lavado (1993) observed that preferential flow paths of a soil with clayey natric horizon were decreased by the trampling effect of cattle.

Adoption of reclamative measures on 930 Mha of salt-affected soils can lead to increases in above- and below-ground biomass production that lead to an increase in SOC content. Assuming the rate of SOC increase through adoption of reclamative measures is 200 to 400 kg C ha^{-1} y^{-1} (the rate was as high as 3 to 4 Mg C ha^{-1} y^{-1} in some soils of north central India; Bhojvaid and Timmer, 1998), the potential for C sequestration is 0.186 to 0.372 Pg C y^{-1}. In addition to some hidden C costs, there may also be some financial input needed to restore these soils, especially in relation to installing drainage outlet, application of amendments (e.g., gypsum, biosolids) and fertilizers. National, regional and global soil policy may need to be identified and implemented to restore large areas of degraded lands.

10. Fossil Fuel Offset through Biofuel Production by Desertification Control

Strongly and extremely degraded soils can be taken out of agricultural and pastoral land use, and planted to specific trees, shrubs or grass species that can be harvested for biofuel. The biomass can be used as direct fuel for cooking, heating, power generation and/or as a substrate for conversion to liquid fuels. Total land area of strongly and extremely degraded lands (erosion, physical and chemical degradation) in dry areas is 138 Mha (Oldeman, 1994). Growing appropriate biofuel crops on such lands is an important strategy. The effectiveness of biofuels in generating fossil fuel offset is based on numerous assumptions: (i) the amount of C equivalent released by biomass burning (including CH_4 and N_2O) is equal to that produced by photosynthesis, (ii) solid fuel can be used in the power plants, and (iii) the infrastructure exists to supply the needed inputs and to harvest and transport the wood. Once established (2 to 3 years after planting/sowing), the above-ground biomass production potential in these environments can be 2 to 3 Mg ha^{-1} y^{-1} with a total production of 0.275–0.413 Mg C y^{-1}. With a fuel efficiency of 0.7 for direct use (Marland and Turhollow, 1991; Paustian et al., 1998), the biofuel C offset for these lands is 0.19 to 0.29 Pg C y^{-1}.

11. Potential for Carbon Sequestration in Secondary Carbonates in Drylands

The role of soil inorganic carbon (SIC) in C sequestration is less well understood. Depending on the site-specific conditions, the SIC may act as a sink or source or have no effect upon C sequestration. In systems of partial or complete soil leaching, the major mechanism for sequestered SIC is via movement of HCO_3 into ground waters or closed-systems with limited exchange with ambient environments. Based on reconstruction of carbonate fluxes in soils formed in strongly calcareous parent materials over the geological time periods, this could account for upwards of 1 Mg of SIC ha^{-1} y^{-1}. While this mechanism may appear to be of limited impact for

Table XII
Estimates of C sequestration through formation of secondary carbonates

Ecoregion	Land area (Bha)	Potential rate of C sequestration [a] (kg ha^{-1} y^{-1})	Total sequestration potential (Pg C y^{-1})
Arid	2.55	0–1	0–0.0026
Semi-arid	2.31	3–114	0.0069–0.2633
Sub-humid	1.30	1–124	0.0013–0.1599
Total	6.16		0.0082–0.4258

[a] Potential rates of C sequestration made available by the courtesy of Dr. L. P. Wilding of Texas A&M University, College Station, TX

degraded drylands, certainly it has implications when ground waters undersaturated with respect to Ca $(HCO_3)_2$ are used for irrigation. Enhanced biomass primary productivity and salinity control strategies (e.g., gypsum amendments, organic wastes, etc.) can result in increased leaching of Ca $(HCO_3)_2$ via irrigation if the irrigation waters are not already saturated with respect to bicarbonates (Wilding, 1999; Nordt et al., 2000).

Dissolution of exposed carbonates in soil systems by acid rain, nitrogenous fertilizers, oxidation of iron sulfides, or organic acids may result in a source of greenhouse gases if subsequent precipitation of pedogenic carbonates occurs. However, if the dissolved bicarbonates are either leached through the soil or removed by overland flow without subsequent precipitation of carbonates, then this mechanism is a transient sink of intermediate to long residency rather than a source of atmospheric CO_2. The rate of C sequestration through formation of secondary carbonates is the subject of debate. Some researchers argue that the rate is slow (3–5 g C m^{-2} y^{-1}) and of little significance (Schlesinger, 1997). Others, however, support the idea that rate of sequestration of atmospheric C may be much higher with a maximum rate of 114 to 124 Kg C ha^{-1} y^{-1} (Table XII). For example, formation of secondary carbonates is accentuated by biotic activity because of high concentration of CO_2 in the soil air (e.g., root growth, termites) (Monger and Gallegos, 2000).

Data in Table XII show the potential of C sequestration through formation of secondary carbonates to range from 0.008 to 0.426 Pg C y^{-1}. The wide range is the result of both actual variability and measurement indicative of the high variation and large uncertainty due to differences in soil profile characteristics, moisture and temperature regimes, land uses and ecoregional characteristics.

Table XIII
Potential of desertification control and land restoration to sequester C (Pg C y^{-1})

Process	Range	Mean	% of total potential
Emission reduction through erosion control	0.2–0.3	0.25	18
Restoration of eroded lands	0.2–0.3	0.25	18
Restoration of physically and chemically degraded soils	<0.01	<0.01	–
Reclamation of salt-affected soils	0.2–0.4	0.3	21
Agricultural intensification on undegraded soils	0.01–0.02	0.015	–
Fossil fuel C offset through biofuel production	0.3–0.5	0.4	29
Sequestration as secondary carbonates	0.01–0.4	0.2	14
Total	0.9–1.9	1.4	100

These estimates have large uncertainties, the potentials of different strategies may not be additive, and adoption of recommended measures at global scale is a major challenge to humanity.

12. Potential of Desertification Control to Sequester C and Mitigate the Greenhouse Effect

The total potential of degraded lands restoration and desertification control to sequester C is shown in Table XIII. The total potential is 0.9 to 1.9 Pg C y^{-1}, with a mean of about 1.4 Pg C y^{-1}. The options with high C sequestration potential include erosion control (36%), biofuel C offsets (29%), reclamation of salt affected soils (21%), and secondary carbonates formation (14%). It is apparent that restoration of eroded and salt-affected soils and erosion control is important strategies. Squire et al. (1995) estimated that management of drylands through desertification control has an overall C sequestration potential of 1.0 Pg C y^{-1}. These high estimates are in contrast to the overall low C storage potential of world soils estimated by Schlesinger (1990). Estimates presented in Table XIII are crude, tentative, and merely suggestive of the high potential that exists if judicious land use measures are adopted in the drylands. Uncertainties are high and may be 30 to 50%, as is evidenced by a wide range in the rate of C sequestration in soil and biomass. Further, estimates of potential for different strategies are not additive, and the data need to be used with due consideration of site-specific conditions.

The potential of C sequestration in the ecosystem is computed for a 50-year period. Although C sequestration in an ecosystem can continue for up to 150 years (Akala and Lal, 2000), the rate and cumulative amount of sequestration are high only for up to 50 years. Upon conversion to restorative or improved systems, rate of C sequestration may peak within 10 to 15 years. Therefore, for practical purposes, 50 years is an adequate period to estimate the potential (Lal et al., 1998).

An important consideration to realization of this biophysical potential is identification of policies that facilitate adoption of recommended practices, assessment of the societal value of soil carbon, and development of institutions that involve carbon trading through clean development mechanism. An important issue is C farming and its commodification. It is fair that farmers and land managers are justly compensated for adopting practices that benefit the world community. Important among the societal benefits of enhancing soil and ecosystem C, for which farmers must be compensated, include: (i) reduction in erosion and downstream sedimentation, (ii) decrease in non-point source pollution, (iii) biodegradation of pollutants, (iv) purification of natural waters, (v) enhancement of biodiversity (soil and vegetation), and (vi) reduction in risks of accelerated greenhouse effect.

13. Conclusions

Desertification, decline in quality of soil and vegetation and spread of desert-like environments, is a biophysical process driven by socio-economic and political factors. Important biophysical effects involve deterioration of aggregates and soil structure, soil erosion by water and wind, decline in topsoil depth and a reduction in available water capacity, salinization, nutrient depletion, and reduction in the soil C pool. These processes and effects lead to a decline in biomass production and net loss of C from soil and terrestrial ecosystems to the atmosphere. Total historic loss of C due to desertification is in the order of 19 to 29 Pg.

Desertification control implies re-establishing the vegetative cover, conserving soil and water, improving soil fertility, enhancing soil quality, and increasing biomass production. There are several proven technologies, which on successful implementation can set-in-motion processes leading to restoration of vegetative cover and soil quality. These include introduction of appropriate species, integrated nutrient management, water harvesting and supplementary irrigation, conservation tillage, improved farming systems etc. The strategy is to restore degraded soils and ecosystems. The global potential of C sequestration through these measures is 0.9 to 1.9 Pg C y^{-1} for 25 to 50 years.

Before this potential can be reached, research must be done to achieve the following:

(a) obtain credible estimates of the extent and rate of soil degradation by different processes at regional, national and global scales;
(b) understand soil C dynamics, pool and fluxes, as influenced by soil degradative and restorative processes, and soil management;
(c) identify site-specific practices for desertification control and restoring degraded soil;
(d) assess the role of SIC in C-sequestration processes, and evaluate the rate, dynamics and magnitude of SIC fluxes in relation to climate, land use and management;

(e) improve water (irrigation) and nutrient use efficiency in dryland ecosystems;
(f) understand processes that contribute resilience to soils in dryland ecosystems;
(g) provide incentives to farmers and land managers to adopt recommended agricultural practices, and
(h) develop mechanisms of C farming and its commodification.

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