OFFSETTING CHINA'S CO_2 EMISSIONS BY SOIL CARBON SEQUESTRATION*

R. LAL

Carbon Management and Sequestration Center, The Ohio State University, 2021 Coffey Road, Columbus, OH 43210, U.S.A.
E-mail: lal.1@osu.edu

Abstract. Fossil fuel emissions of carbon (C) in China in 2000 was about 1 Pg/yr, which may surpass that of the U.S. (1.84 Pg C) by 2020. Terrestrial C pool of China comprises about 35 to 60 Pg in the forest and 120 to 186 Pg in soils. Soil degradation is a major issue affecting 145 Mha by different degradative processes, of which 126 Mha are prone to accelerated soil erosion. Similar to world soils, agricultural soils of China have also lost 30 to 50% or more of the antecedent soil organic carbon (SOC) pool. Some of the depleted SOC pool can be re-sequestered through restoration of degraded soils, and adoption of recommended management practices. The latter include conversion of upland crops to multiple cropping and rice paddies, adoption of integrated nutrient management (INM) strategies, incorporation of cover crops in the rotations cycle and adoption of conservation-effective systems including conservation tillage. A crude estimated potential of soil C sequestration in China is 119 to 226 Tg C/y of SOC and 7 to 138 Tg C/y for soil inorganic carbon (SIC) up to 50 years. The total potential of soil C sequestration is about 12 Pg, and this potential can offset about 25% of the annual fossil fuel emissions in China.

1. Introduction

Fossil fuel emissions of carbon(C) in China are about 1 Pg/y, which may surpass that of the U.S. (1.84 Pg C/y) by 2020 (National Environment Trust, 1998). In comparison to China, global C emissions are estimated at 6.3 ± 0.4 Pg/y from fossil fuel and 1.6 ± 0.8 Pg/y from land use change and tropical deforestation (Battelle, 2000; IPCC, 2001). Emissions of C in China have been steadily increasing since 1950, with increase of more than 30% during the 1990s (World Watch Institute, 2002).

The land use in China comprises 124 million hectares (Mha) of cultivated cropland (12.9%), 11.2 Mha of permanent crops (1.2%), 400 Mha of pastures (41.7%) and 133.7 Mha of forest and woodland (1.9%) (Table I). The area under forest and woodland and nature reserves has increased since the 1960s. The area under the forest in China was 85.5 Mha in 1962 and 133.7 Mha in 1993. The protected areas in China have increased from 0 Mha in 1978 to 87.4 Mha in 1999 (World Bank, 2001). With conversion to a judicious land use and adoption of recommended man-

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Table I
Land use in China (FAO, 1999; Dept. of Forest Resources and Management, 1996)

Land use	Area (Mha)
Total area	959.7
Land area	932.6
Crop land	124.1
Permanent crops	11.2
Pasture/grasslands	400.0
Forest and wetland	133.7
Miscellaneous	263.6

agement practices (RMPs), there is a large potential of terrestrial C sequestration. Realization of this potential, however, depends on adoption of appropriate soil, crop, pasture and forest management practices for principal soils and ecoregions. The objective of this report is to assess the potential of soil C sequestration in China's agricultural and forest soils. This report illustrates how such potential could be derived from information in the literature, assuming that soil quality could be restored through land use conversion and adoption of RMPs. The objective is to show that potential of soil C sequestration exists rather than to predict that potential precisely and accurately.

2. Terrestrial Carbon Pool in China

The terrestrial pool comprises C stocks in the biota, wetlands, and soil. The biotic C pool, contained largely in above ground and below ground biomass in the forest (live and dead), is a labile pool depending on the use of the forest products. China's forestland range from tropical forest in the south to boreal forest in the north (Fang et al., 2001). Estimates of the total vegetation C pool in all forest ecosystems range from 35.2 to 57.9 Pg (Ni, 2001; Peng and Apps, 1997). In addition, grasslands in China contain 3.06 Pg C (Ni, 2002).

Soils of China comprise a large C reservoir consisting of both soil organic carbon (SOC) and soil inorganic carbon (SIC) pools. In comparison with the biotic pools, soils of China comprise 81.8 to 185.7 Pg C, or about 12% of the global SOC pool. A large part of China's land area is in semi-arid and arid climates. Carbonate-rich soils cover an area of 344 Mha containing 4.9 Pg of SIC (Feng et al., 2002). These carbonate-rich soils of arid and semi-arid regions also form secondary carbonates or pedogenic carbonates through bio/pedogenic processes.

Table II
Soil degradation in China (adapted from Lindert, 2000)

Year	Salinization (Mha)	Erosion (Mha)	Inundation (Mha)	Drought-prone (Mha)	Total (Mha)
1980	6.0	109.8	4.5	9.5	129.8
1985	6.2	118.8	4.7	8.3	138.0
1990	6.1	114.8	5.0	6.1	132.0
1995	6.2	125.5	4.6	8.3	144.6

3. Soil Degradation

Soil degradation exacerbates depletion of SOC because of reduction in biomass production and low amounts of residues returned to the soil. Soil degradation in China comprises 144.6 Mha or 7.4% of the world total (Table II). Of the 144.6 Mha of the total degraded soil, 125.5 Mha (86.8%) is by accelerated erosion, 6.2 Mha (4.3%) by salinization, 4.6 Mha (3.2%) by inundation, and 8.3 Mha (5.7%) by drought (Lindert, 2000). Total soil area affected by different degradative processes in China increased from 129.8 Mha in 1980 to 144.6 Mha in 1995, at an average rate of 11.4%/y. In addition, desertification or degradation of soil and vegetation is also a major problem in China. Desertification is caused by anthropogenic activities which decrease effective precipitation by increasing runoff and evaporation and reducing available water capacity of the soil (UNEP, 1992). Accelerated erosion is one of the processes which exacerbates desertification. There are varying estimates of the extent of desertification in China. Duan Zhenghu et al. (2001) estimated that desertified land in China covers 177 Mha comprising 81 Mha of slight, 61 Mha of moderate and 35 Mha of severe desertification. Dregne and Chou (1992) estimated that land area in China prone to desertification (moderate+) comprises 8.1 Mha of irrigated cropland, 11.4 Mha of rainfed cropland, and 364 Mha of rangeland. Huang (2000) and State Environment Protection Administration (1998) estimated that 331 Mha, about one-third of China, is prone to desertification. In addition, 82 to 100 Mha is affected by salinization, including 7 to 8 Mha of cultivated land (Huang, 2000; State Environment Protection Administration, 1998). Of the 400 Mha of grazing land, 137 Mha are moderately to severely degraded (World Bank, 2001). The risks of soil degradation may increase with the projected climate change (IPCC, 2000).

Accelerated erosion is the principal soil degradative process in China where both water and wind erosion are severe hazards. In addition to erosion by water and wind, tillage erosion also causes soil movement with adverse impact on soil quality (Quine et al., 1999a,b). Water erosion has been shown to cause a considerable loss of SOC in soils of the Loess Plateau (Zhu, 1984). Yuanda et al. (2002) simulated

Table III

Trends in soil organic carbon content in China (adapted from Lindert, 2000)

Region	1930s (%)	1950s (%)	1980s (%)
All north (except desert steppe)	1.06 (24)	0.83 (124)	0.65 (989)
All south China	0.92 (32)	1.29 (56)	1.37 (1129)

Number in parenthesis refers to samples analyzed.

the impact of erosion and observed severe losses in SOC pool of the eroded soils. Susceptibility to erosion has decreased SOC concentration of soils in north China over the 50-year period ending in 1980. However, SOC concentration increased slightly over the same period in south China (Table III). The decline in SOC by erosion *on site* is caused by: (i) disruption of aggregates by tillage and erosional processes, and an attendant release of C which becomes prone to microbial breakdown, (ii) transport of C by water and wind and tillage-induced redistribution over the landscape, and (iii) susceptibility of C displaced to oxidation/mineralization with attendant release of CO_2 to the atmosphere. In comparison, depositional sites may gain SOC. An erosion-induced decline in SOC pool has a severe adverse impact on soil quality (Li and Linkdstrom, 2001).

Soil degradation and desertification lead to emission of CO_2 and other gases from soils to the atmosphere because of the depletion of the C pool in soils and vegetation. Yuanda et al. (2002) observed that over the 10-year period, SOC concentration decreased by 1.25 g/kg in sloping land, 0.5 g/kg in terraced land and 0.14 g/kg in sloping land orchards. Yuanda and colleagues calculated that SOC loss from tilled cropland was 20 to 25 Mg/ha over a 10-year period. These and other data support the assumption that 125.5 Mha of eroded soils have lost 20 to 30 Mg/ha of SOC, 6.2 Mha of salinized soils have lost 5 to 10 Mg/ha of SOC, 8.3 Mha of drought-prone soils have lost 10 to 20 Mg/ha of SOC, and 186 Mha (331 Mg–144.6 Mha) of desertified soils have lost 10 to 20 Mg/ha of C in the vegetation. Thus, a crude estimate of the total C lost from degraded and desertified ecosystems in China may be 8 to 14 Pg C due to historic land misuse and soil mismanagement. Even if one-half to two-thirds of this C can be resequestered through land use conversion and restoration of degraded soils and ecosystems, the total potential of C sequestration in restoring degraded/desertified soils and biota may be 4 to 9 Pg over 50 years between 2000 and 2050.

Table IV
Fertilizer use in China (FAO, 1998)

Year	Total fertilizer use (10^6 Mg)	Nitrogen fertilizer (10^6 Mg)
1990	27.4	4.4
1996	35.8	4.9
1998	36.0	3.0

4. Soil Carbon Sequestration

Presently the low level of SOC concentration in soils of China can be enhanced by: (i) restoration of degraded soils, (ii) conversion of agriculturally marginal soils to pastures or forest lands, and (iii) adoption of RMPs on cropland. Relevant RMPs for cropland include: (i) returning crop residues and other biosolids to soil, (ii) adopting nutrient recycling technologies, (iii) improving soil fertility through integrated nutrient management (INM) technologies based on biological nitrogen fixation (BNF) and using compost and farmyard manure, (iv) growing legume-based crop rotations with frequent incorporation of cover crops in the rotation cycle, and (v) adopting conservation-effective farming systems (e.g., conservation tillage) that reduce risks of soil erosion.

Fertilizer consumption in China increased by 31% over the 8-year period ending in 1998 (FAO, 1998), of which principal increase has been in the nitrogenous fertilizer use. Whereas the judicious use of fertilizers is an important component of agricultural intensification and increasing C sequestration, enhancing use efficiency of fertilizers is extremely important. With regard to SOC sequestration, an important strategy is to improve nutrient cycling and adopt the INM techniques.

Zhu and Wen (1992) and Cai (1996) reported that SOC concentration in paddy soils (which remain inundated during most of the growing cycle) are usually higher than those in upland soils. The high SOC concentration in paddy soils of the flood plains is due to deposition of eroded materials from uplands, multiple cropping, favorable moisture regime, low decomposition rate and high input of roots and other biomass. Erda et al. (1997) assessed C budget for agriculture (cropland and grazing land) in China. They estimated that annual photosynthetic uptake is 652 Tg CO_2-C by agroecosystems involving both primary and secondary production. Of this, 584.9 Tg C is re-emitted, 8.5 Tg C is transferred out of the ecosystem and 59.5 Tg is fixed within the soil and biota. Erda and colleagues concluded that Chinese agriculture is a 'sink' for atmospheric CO_2.

5. Strategies of Carbon Sequestration in Soils of China

There are several strategies of soil C sequestration in China. Important among these are briefly described below:

(i) *Restoration of degraded soils:* Restoration of degraded/desertified soils is a high priority. In coastland of southeastern Fujian, China, Yusheng et al. (2002) observed a strong increase in SOC pool through restoration of eroded and degraded soils. The SOC concentration of the lateritic red soil increased from 0.18% to 0.82% over a 3-year period from 1991 to 1994. The degree of humification was 7.7 times more in the restored compare with unrestored soils. In the Loess Plateau region, Xiaoning et al. (2002) observed increases in SOC concentration upon conversion of agricultural soils to forestry. Ten years after conversion to forest, the SOC concentration in cultivated and forested soil, respectively, was 0.5 and 3.4% for 0 to 20 cm depth, 0.3 and 2.1% for 20 to 40 cm depth, 0.3 and 0.8% for 40 to 60 cm depth, 0.2 and 0.5% for 60 to 80 cm depth, and 0.2 and 0.3% for 80 to 100 cm depth. In accord, there were also increases in aggregation of the forested soil. Also in the Loess Plateau, Xiaoning et al. (2002) studied increase in SOC concentration and improvement in soil quality by converting eroded soils to seabuckthorn (*Hippophae rhamnoides*) plantations. The SOC in the seabuckthorn plantation was mainly concentrated in the surface horizon. The SOC concentration under the plantation at the Lingyou site was 2.6% in 0 to 10 cm, 0.8% in 20 to 30 cm and 0.5% in 40 to 50 cm depth. The SOC concentration under grassland was 2.4% in 0 to 10 cm, 0.9% in 20 to 30 cm and 0.5% in 40 to 50 cm depth. In comparison, SOC concentration in cropland was 0.5% in 0 to 10 cm, 0.22% in 20 to 30 cm depth and 0.2% in 40 to 50 cm depth. Similar trends were observed at the Qingyan site.

The survey of literature shows that adoption of conservation-effective farming systems (e.g., conservation tillage, no till farming or direct seeding with crop residue mulch) can minimize the risk of losing 32 to 64 Tg C/y through erosion-induced emissions (Lal, 1995, 2002). In addition, depleted SOC stock can be restored at a modest rate of 100 to 200 kg/ha/y through adoption of soil restorative measures. Restorative measures include afforestation of severely eroded lands, application of biosolids to enhance microbial activity, land forming and application of compost and other amendments on drastically disturbed (e.g., mined) lands, leaching of excess salts from salt-affected soils etc. An example of increase in SOC pool through change in land use is shown by the data in Table V which indicate differences in SOC 80 Mg/ha to 1-m depth over a 10-year period. It is assumed that SOC sequestration is less rapid than the depletion. Thus, restoration of degraded soils with an area of 144.6 Mha can lead to soil C sequestration at the rate of 14 to 28 Tg C/y. In addition, there is a potential of SOC sequestration in desertified soils. While the total desertified land in China is estimated at 331 Mha, deducting degraded land area of 144.6 Mha, gives the remaining area of 186 Mha. Assuming SOC sequestration rate of 75 to 150 kg/ha/y, the potential of desertification control

Table V

Soil organic carbon pool in a forested and cultivated soil (10 yrs) of the Loess Plateau region of China (recalculated from Xiubin et al., 2002)

Depth (cm)	Soil organic carbon (%)		Bulk density (Mg/m^3)		Soil organic carbon pool (Mg/ha)	
	Forest	Cultivated	Forest	Cultivated	Forest	Cultivated
0–20	3.40	0.48	0.65	1.04	44.2	10.0
20–40	2.10	0.33	0.86	1.24	36.1	8.2
40–60	0.83	0.29	1.01	1.26	16.8	7.3
60–80	0.51	0.23	1.21	1.29	12.3	5.9
80–100	0.32	0.23	1.26	1.29	8.1	5.9
Total					117.5	37.3

SOC pool (Mg/ha $= \frac{\text{SOC concentration}}{100} \times \frac{10^4\ \text{m}^2}{\text{ha}} \times$ depth (m) $\times$ bulk density $\frac{(\text{Mg})}{\text{m}^3}$.

is 14 to 28 Tg C/y. Therefore, total SOC sequestration potential through erosion management and restoration of degraded soils is about 46 to 92 Tg C/y.

(ii) *Recommended management practices:* Adoption of RMPs may enhance SOC sequestration at the rate of 200 to 300 kg/ha/y in cropland, 100 to 200 kg/ha/y in forest land, and 50 to 100 kg/ha/y in grazing land in the U.S. (Lal et al., 1998; IPCC, 2000; Lal, 2001; Follett et al., 2001; Heath et al., 2002). These rates are extremely conservative, and much higher rates have been observed in China and elsewhere. For example, the data in Table VI show that land use change can cause a substantial change in the SOC pool. For a soil on the Loess Plateau, the SOC pool to 0.5 m depth was 50.8 Mg/ha for forest land, 70.6 Mg/ha for grassland and 17.4 Mg/ha for cropland for the Lingyou site. The SOC pool for the Qingan site was 44.5 Mg/ha for forestland, 22.5 Mg/ha for grassland and 32.8 Mg/ha for cropland. Thus, conversion of cropland to grassland or forestland at the Lingyou site, and that of cropland and grassland to forestland would enhance SOC pool over time. Therefore, land use change and improved management of these soils has a potential to enhance SOC pool and sequester C in the soil. Assuming that these rates are applicable over large areas of China, the potential of soil C sequestration in China is as follows:

- cropland: 25 to 37 Tg C/y,
- forest and plantations: 14 to 29 Tg C/y,
- grazing land: 20 to 40 Tg C/y.

Thus agricultural intensification and adoption of RMPs on cropland, forestland and grazing land has a potential to sequester 49 to 106 Tg C/y.

(iii) *Secondary carbonates:* In addition to SOC, there is also a potential of SIC sequestration. The rate of formation of secondary carbonates also depends

Table VI

The soil organic carbon dynamics in three land use systems for Lingyou and Qingan sites in the Loess Plateau region (recalculated from Xiaoning et al., 2002; Xiubin et al., 2002)

Location	Depth (m)	Soil organic carbon (%)			Soil bulk density (Mg/m^3)			Soil organic carbon pool (Mg/ha)		
		Forest	Grassland	Cropland	Forest	Grassland	Cropland	Forest	Grassland	Cropland
Lingyou	0–0.1	2.55	2.37	0.45	0.65	0.85	1.04	33.2	40.3	9.4 (0.2 m)
	0.2–0.3	0.80	0.94	0.22	0.86	1.05	1.24	6.9	19.7	2.7 (0.1 m)
	0.4–0.5	0.53	0.46	0.21	1.01	1.15	1.26	10.7	10.6	5.3 (0.2 m)
	Total							50.8	70.6	17.4
Qingan	0–0.1	1.79	0.77	0.58	0.65	0.85	1.04	23.3	13.1	12.1 (0.2 m)
	0.2–0.3	0.99	0.30	0.51	0.86	1.05	1.24	8.5	3.2	6.3 (0.1 m)
	0.4–0.5	0.63	0.27	0.57	1.01	1.15	1.26	12.7	6.2	14.4 (0.2 m)
	Total (0.5 m)							44.5	22.5	32.8

Soil bulk density are taken from those of the Loess soil reported by Xiabin et al. (2002).

Table VII

Measured and predicted rates of secondary carbonates in different soils of China (recalculated from Pan and Guo, 2000)

Soil	Region	Rate of formation of secondary carbonates (kg/ha/yr)
Hap-Cryic Aridisols	Northern Tibet	100
Cal-Orthic Aridisols	Qinghai	360–400
Cal-Orthic Aridisols	Gansu	100–200
Cal-Orthic Aridisols	Lanzhou	100–150
Cal-Ustic Isohumisols	Inner Mangolia	30–110
Cal-Orthic Cambisols	Loess Plateau	320
Hap-Ustic Cambisols	Central Shangdong	20
Hap-Aquic Vertisols	Gaomi, Shangdong	120
Ust-Alluvic Entisols	Northern Anhui	40

on the rainfall and quality of irrigation water. The rate of formation of secondary carbonates in China has been reported to range from 30 to 400 kg/ha/y (Table VII), with a mean value of 100 kg/ha/y. These rates reported for China are much larger than those reported for other dry regions (Schlesinger, 1997), and can be achieved with recommended management practices. Assuming these rates apply on a large area of 344 Mha, the total potential of SIC sequestration is 7 to 138 Tg C/y.

6. Potential of Soil Carbon Sequestration in China

Total potential of soil C sequestration in China is 126 to 364 Tg C/y, with a mean of 245 Tg C/y (Table VIII). Having been cultivated for a long period of time (5000 years in some areas of China) and highly depleted of their SOC pool, soils of China may sequester C at the rates described above for at least 50 years. Thus, total or accumulative potential of C sequestration in soils of China is about 12 Pg by 2050. Of the SOC sequestration, 19.6% is by erosion management, 12.7% by cropland management, 8.8% by forestland management, and 17.1% by restoration of degraded soils and desertification control. Thus, erosion management and restoration of degraded/desertified soils comprise 36.7% of the total soil C sequestration potential in China.

Battelle (2000) computed regional CO_2 emissions for different scenarios. China's economy is rapidly growing with high-energy demands. Because of increasing energy demands, innovative options are needed to reduce or offset fossil fuel emissions. Battelle (2000) observed that soil C sequestration has an important role to play in stabilizing emissions during the first half of the 21st century. This

Table VIII
Potential of soil carbon sequestration in China

Strategy/land use	Area (Mha)	Potential (Tg C/yr)
A. Soil organic carbon		
Erosion management	125.5	32–64
Restoration of degraded soils	144.6	14–28
Desertification control	186.0	14–28
Cropland management	124.1	25–37
Forest land and management	133.7	14–29
Grazing land management	400.0	20–40
Sub-total		119–226
B. Soil inorganic carbon sequestration	344.0	7–138
Total potential		126–364

scenario is also in accord with the need for enhancing soil quality to meet the food demands of the country with the world's largest population. Soil C sequestration is, indeed, a win-win strategy.

7. Policy Considerations

The goal of the UN Framework Convention on Climate Change (UNFCCC) is to stabilize greenhouse gas concentration in the atmosphere at a level that will prevent dangerous anthropogenic interference with the climate system. Annex II or developing countries, such as China, have several options to meet their Kyoto Protocol commitments (Albrecht, 2002): (i) reduce domestic emissions by using renewable energy sources, (ii) adopt emission trading with developed or Annex I countries that allow China to sell or trade greenhouse gases under the 'Clean Development Mechanism', and (iii) use soil/terrestrial sink to sequester C in terrestrial ecosystems. It is in this regard that soil C sequestration offers a viable option to developing countries such as China to reduce net emissions and meet its commitments under the Kyoto Protocol.

In contrast to other options, soil C sequestration has numerous ancillary benefits, which cannot be over-emphasized. Having a population of 1.1 billion, food security is an important issue. With a limited cropland area and severe problem of soil degradation, enhancing soil quality through restoration of degraded soils is an important strategy. Therefore, afforestation of denuded hills and agriculturally marginal lands, and adopting other measures of soil restoration are important to achieving food security and mitigating the climate change.

Preliminary and crude as they may be, the estimates of soil C sequestration presented in this report indicate that the potential is substantial, and has numerous secondary benefits which are important to agricultural economy of China. Whereas implementation of options (i) and (ii) outlined above require market forces to be in place, implementation of option (iii) to use terrestrial C sinks provides the much needed flexibility and makes it a truly win-win strategy. Any trading of C (under joint implementation or clean development mechanisms) at international level must be supplemented by domestic action of which soil C sequestration is a promising option.

8. Summary

Adoption of conservation-effective measures of erosion management, restoration of degraded soils and ecosystems, and conversion of traditional farming to RMPs can reverse soil degradative trends, enhance soil quality, increase biomass productivity and sequester C in soil and the biomass. The total potential of C sequestration in soils of China is about 12 Pg over a 50-year period. Realizing this potential requires identification of policies that encourage adoption of RMPs and restoration of degraded soils and ecosystems. The potential of soil C sequestration in China can offset about 25% of the annual fossil fuel emissions.

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