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Environment International 32 (2006) 781 – 796

ENVIRONMENT INTERNATIONAL

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Ecosystem carbon budgeting and soil carbon sequestration in reclaimed mine soil

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Received 15 August 2005; accepted 1 May 2006

Abstract

Global warming risks from emissions of green house gases (GHGs) by anthropogenic activities, and possible mitigation strategies of terrestrial carbon (C) sequestration have increased the need for the identification of ecosystems with high C sink capacity. Depleted soil organic C (SOC) pools of reclaimed mine soil (RMS) ecosystems can be restored through conversion to an appropriate land use and adoption of recommended management practices (RMPs). The objectives of this paper are to (1) synthesize available information on carbon dioxide (CO_2) emissions from coal mining and combustion activities, (2) understand mechanisms of SOC sequestration and its protection, (3) identify factors affecting C sequestration potential in RMSs, (4) review available methods for the estimation of ecosystem C budget (ECB), and (5) identify knowledge gaps to enhance C sink capacity of RMS ecosystems and prioritize research issues. The drastic perturbations of soil by mining activities can accentuate CO_2 emission through mineralization, erosion, leaching, changes in soil moisture and temperature regimes, and reduction in biomass returned to the soil. The reclamation of drastically disturbed soils leads to improvement in soil quality and development of soil pedogenic processes accruing the benefit of SOC sequestration and additional income from trading SOC credits. The SOC sequestration potential in RMS depends on amount of biomass production and return to soil, and mechanisms of C protection. The rate of SOC sequestration ranges from 0.1 to 3.1 Mg ha^{-1} yr^{-1} and 0.7 to 4 Mg ha^{-1} yr^{-1} in grass and forest RMS ecosystem, respectively. Proper land restoration alone could off-set 16 Tg CO_2 in the U.S. annually. However, the factors affecting C sequestration and protection in RMS leading to increase in microbial activity, nutrient availability, soil aggregation, C build up, and soil profile development must be better understood in order to formulate guidelines for development of an holistic approach to sustainable management of these ecosystems. The ECBs of RMS ecosystems are not well understood. An ecosystem method of evaluating ECB of RMS ecosystems is proposed.

Keywords: Ecosystem carbon budget; Carbon sequestration; Reclaimed mine soil; Disturbed soil

1. Introduction

Terrestrial ecosystems play a major role in moderating the global C cycle. The human perturbations of the C cycle directly affect ecosystem function. The CO_2 release to the atmosphere from fossil fuel combustion and other anthropogenic activities are well documented and accepted, as the principal cause of the projected climate change. Mining is an anthropogenic activity, which causes drastic soil disturbances. Drastically disturbed soils include coal and other mine soils, highway shoulders, ash disposal sites, and tailings. There are about 3.2 Mha of land disturbed by coal mining in the U.S. of which only 68,500 ha (2%) are reclaimed and bond released (Table 1). These disturbed soils have potential for C sequestration through restoration by using combustion by-products, and other organic amendments. However, the fluxes and inputs affecting C balance of these restorative measures are not well documented. If the disturbed soils are to be managed for C sequestration, and reduction in CO_2 emission into the atmosphere, the processes affecting the C cycle of the RMS ecosystems must be understood and documented. In this regard, the periods immediately following disturbance and restoration of mine soils are important. Reclamation and revegetation of drastically disturbed soils, including the creation of productive soil and biological resources for C sequestration and the development of profitable and environmentally sound

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doi:10.1016/j.envint.2006.05.001

Table 1
Number and size of coal mines in the U.S. and Ohio

Region	Number of mines for 2003[1]	Total area approved for mining (10^3 ha)[2]	Bond release for phase I to III after reclamation (10^3 ha)[2]
United States	1316	3200.0	67.5
Ohio	54	40.6	4.2

Source: [1]EIA (Energy Information administration), 2003.
[2]OSM, 2003.

land uses depend on a sound understanding of the chemical, physical and biological properties of the soil under the new ecosystems.

Estimation of net ecosystem productivity (NEP), also known as ecosystem C budget or balance (ECB), is important for quantifying changes in the C cycle by anthropogenic activities. The ECB includes both processes that store C in biomass and soil, and release C into the atmosphere. The difference in these two processes in an ecosystem refers to the ECB. The ECBs in the past have often been calculated considering only the balance between gross primary production and ecosystem respiration. The past approaches, however, have generally ignored other C fluxes from an ecosystem (e.g., leaching of dissolved C and losses associated with disturbance), which must also be considered.

The natural processes involved in C sequestration of disturbed soil are highly complex, and research results are relatively scanty. Therefore, the objectives of this review are to synthesize the available information on CO_2 emissions from coal mining activities, identify mechanism of SOC sequestration and its protection, assess the SOC sequestration potential through effective reclamation practices, evaluate ECB of RMSs, and identify and prioritize future research and development needs of RMS ecosystems.

2. Terrestrial carbon dioxide emission

The global average concentration of CO_2 in the atmosphere has increased by 35% since the beginning of the industrial revolution from 280 ppmv in 1800 to 353 ppmv in 1990 and 378 ppmv in 2005 (Anon, 2005; BBC, 2005). In the 2004 World Energy Outlook, the IEA Reference Scenario projected that CO_2 emissions will increase by 63% between 2004 and 2030, which is 90% higher than the 1990 emissions (World Coal Institute, 2005). From 1989 to 98, human activities contributed an average of 7.9 Gt CO_2- C yr^{-1}. Much of that emission was reabsorbed into oceans (2.3 Gt C) and terrestrial systems (2.3 Gt C), leaving a net global increment into the atmosphere of about 3.3 Gt C/year (Wood et al., 2000).

Total and per capita CO_2 emission are the highest for the United States (Fig. 1). However, the rate of per capita increase is high for Australia and USSR (Fig. 1B). The burning of fossil fuel, changes in land uses, mining and construction have caused an increase in the concentration of CO_2 in the atmosphere. Such increases have the potential to cause regional and global climate and related environmental changes like increase in global temperature, change in precipitation amount and pattern, rise in sea level, and increase in frequency and severity of extreme weather events (Easterling et al., 2000). These projections have encouraged scientists to consider options for minimizing future increase in global CO_2 concentrations. A potential approach to mitigating the rising CO_2 concentration is to enhance sequestration of C in terrestrial ecosystems (Paustian et al., 1998). This can be achieved by enhancing the biological processes like photosynthesis that assimilate CO_2 increasing biomass productivity, and allocating the assimilated C into long-lived plant and soil organic matter (SOM) pools resistant to microbial decomposition. This indicates the importance of plant- and soil-based C sequestration strategies, which can be successfully implemented to reduce the net CO_2 emission into the atmosphere. Although a key objective in C management research is to enhance the natural capacity of plants and soils to sequester C, the functionality of C storage in terrestrial ecosystem as a whole especially in RMS is poorly understood.

3. Drastically disturbed soils

Drastically disturbed soils are those where native vegetation and animal communities have been removed and most of the topsoil lost, altered or buried (Box, 1978). These soils may not completely regenerate themselves through normal ecological successional processes at least within a generation. Therefore, the natural processes must be facilitated by creation of conducive environment for the regeneration of vegetation.

3.1. Types of drastically disturbed soils

Soil degradation is caused by several anthropogenic activities resulting in drastically disturbed soils. The most common

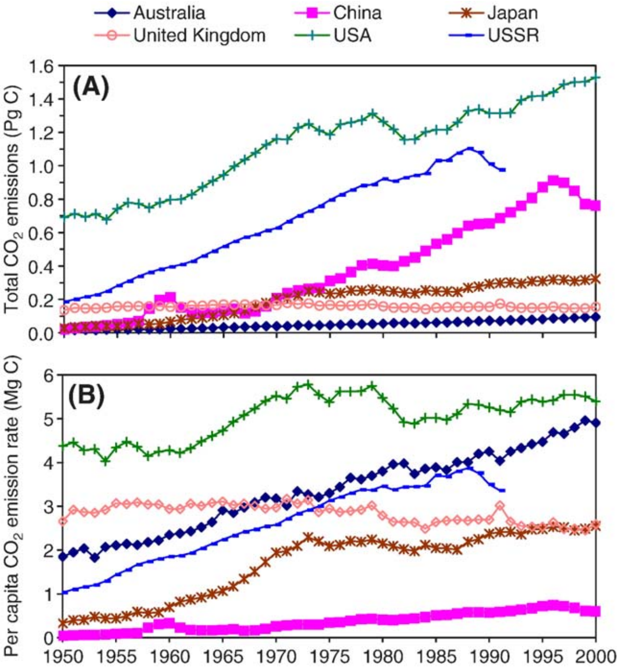

Fig. 1. (A) Total and (B) per capita CO_2 emissions from fossil fuel in selected countries (data from Marland et al., 2003).

disturbed soils are construction related such as those in urban centers, roadways and highways, fills and shoulders. The second most common disturbed soils are those from mining of coal, uranium, phosphates and others. The third types of drastically disturbed soils are severely eroded farmland, rangeland, mountains and river basins due to overuse or misuse. In case of naturally disturbed soils, the rate of disturbance is low. In contrast, disturbance by mining is rapid and can cause most drastic perturbation to the landscape. This disturbance changes landscape totally and drastically, and it has severe environmental consequences. Reclamation of these soils is necessary to minimize risks of environmental pollution.

Mining is defined as a process or activity aimed at removing the desired minerals from its natural placement in the Earth (Paone et al., 1978). In short term, some minerals (e.g., coal, gypsum, uranium and other metals) are of more economic value than the soil that covers them. These minerals are removed by two methods: surface or strip mining and underground mining. The proximity of the minerals to the earth's surface is a principal factor in determining which method is the most suitable. The surface mining process permits the mineral to be removed through excavations. Surface mining: (1) eliminates vegetation, (2) changes topography permanently, (3) alters soil and subsurface geological structure permanently and drastically, and (4) disrupts surface and subsurface hydrologic regimes. The magnitude of surface mining impact depends on the mining technology employed, extent of the disturbance, chemical and physical composition of the minerals present and its overburden, surface and subsurface hydrologic patterns, and method of reclamation.

Coal, one of the important minerals being surface mined, is an important resource for power, heat, industry and transport (World Coal Institute, 1997). It accounts for 51% of the total electricity supply and approximately 95% of fossil energy reserves of USA (NMA, 2004a,b,c; OSM, 2003). Despite its economic importance, mining operations completely remove and stockpile soil materials resulting in drastic landscape perturbations. This causes major damage to whole ecosystem, drastically disturbing soil properties, and adversely affecting or impairing nutrient cycling (Table 2) (Insam and Domsch, 1988; Harris et al., 1993; Anderson et al., 2004). Mined soils are often characterized by high bulk density, low pH, low nutrient availability, poor structure, low water holding capacity, and low biomass productivity (Indorante et al., 1981; Boerner et al., 1998; Hearing et al., 2000; Palumbo et al., 2004; Seybold et al., 2004; Sinclair et al., 2004). Drastic pH changes can adversely affect biotic or living component of the soil such as the fungi, which has symbiotic relationship with plant roots helping in nutrient absorption from the soil. Adverse physical, chemical and biological conditions often limit restoration of surface mine reclamation sites (Coyne et al., 1998).

Mining activities can accentuate CO_2 emissions from mineralization of SOM by soil disturbances and fluxes of C from fell biomass decomposition (Indorante et al., 1981; Smith et al., 1994). The contribution of coal mining to emitting C in the U.S. increased from 250 Tg in 1950 to 550 Tg C in 2001 (Blasing et al., 2004). Coal contributes about 19% of enhanced GHGs emission worldwide (World Coal Institute, 2003).

Table 2
The major problems of mine soil ecosystems and their short and long-term solutions (modified from Bradshaw, 1983)

Limiting factor	Variable	Problem	Immediate treatment	Long-term treatment
Physical	Structure	Too compact	Rip or scarify	Vegetation
		Too loose	Compaction	Vegetation
	Stability	Unstable	Stabilizer, mulch	Regrade or vegetation
	Moisture	Too wet	Drain	Drain
		Too dry	Organic mulch	Tolerant species
Nutritional	Macronutrients	Nitrogen deficiency	Fertilizer	N-fixing plants like leguminous trees or shrubs
		Other nutrient deficiencies	Fertilizer	Application of organic manure or tolerant species
	Micronutrients	Deficiency	Fertilizer	Application of organic manure or tolerant species
Toxicity	pH	Too high	Pyritic waste or organic matter	Weathering or tolerant species
		Too low	Lime	Tolerant species
	Heavy metals	Too high	Organic matter or tolerant cultivar	Inert covering or tolerant cultivar
	Salinity	Too high	Gypsum, irrigation	Weathering or tolerant species

In 1977, "the Surface Mining Control and Reclamation Act" became public law 95-87 (SMCRA, 1977; USDA, 1983). According to this, reclamation is defined as creating a site, which will support organisms in approximately the same percentage and number after the reclamation process is completed, as it did before mining began (Box, 1978; Brown and Hallman, 1984). Reclamation after coal mining reduces off-site impacts, mitigates aesthetic damage to disturbed soil, reconstructs topography and hydrologic patterns, and develops soil profiles over time. The reclamation of drastically disturbed soils leads to improvements in soil quality and developments in soil pedogenic process over time (Table 3 and Fig. 2) making it similar to its pre-mining morphology (Barnhisel and Gray, 2000), providing additional economic benefits from potential of trading C credits, and creating ancillary environmental benefits of C sequestration (Palumbo et al., 2004). Land uses in RMSs include grassland, forest, cropland, rangeland, wildlife habitat, and recreational land. Thus, enhancing C sequestration in these land use of RMSs is an important strategy.

The reclamation of RMSs, represents an opportunity of coupling C sequestration while improving soil quality. The goal of surface mine reclamation is to restore the ecological integrity. The success depends on multivariate interaction of factors, which include strategies adopted by mining companies for reclamation, soil physical and chemical properties of RMS, vegetation establishment, and soil fertility management techniques.

There are well-recognized problems of the mine ecosystems, which require specific immediate and long-term care for the

Table 3
Changes in soil properties with mine soil reclamation

Soil properties	Soil depth (cm)	Undisturbed soil	Reclaimed soil	Year after reclamation	Land use	Location	References
BD (Mg m^{-3})	0–15	1.03	1.39	11		Mercer County, ND	Potter et al. (1988))
	0–15	1.19	1.53	10	Forest	Eastern Ohio	Akala and Lal (2001)
	0–15	1.04	1.19	15	Forest	Eastern Ohio	Akala and Lal (2000)
	0–15	1.42	1.67	10	Pasture	Eastern Ohio	Akala and Lal (2001)
	0–15	1.41	1.67	10	Pasture	Eastern Ohio	Akala and Lal (2000)
	0–15	1.33	1.39				Indorante et al. (1981)
		1.51	1.41	16		Wise County, VA	Bendfeldt et al. (2001)
pH	0–15	7.3	7.7	11		Mercer County, ND	Potter et al. (1988)
	0–15	6–8	8.1	10	Forest	Eastern Ohio	Akala and Lal (2001)
	0–15	7.2	8.1	10	Pasture	Eastern Ohio	Akala and Lal (2001)
	0–15	5.4	5.6	2–5		Louisiana	Feagley et al. (1994)
SAR	0–15	0.2	1.2	11		Mercer County, ND	Potter et al. (1988)
Carbon (%)	0–5	6[a]	2[a]	6	Grass	Montana	Filcheva et al. (2000)
	0–5	6[a]	3.5[a]	50	Grass	Montana	Filcheva et al. (2000)
	0–30	66.3[b]	62.7[b]	25	Pasture	Ohio	Akala and Lal (1999)
	0–30	56.6[b]	58.9[b]	25	Forest	Ohio	Akala and Lal (1999)
C/N	0–15	9.3	11.2	10	Pasture	Eastern Ohio	Akala and Lal (2001)
	0–15	11.3	15.2	10	Forest	Eastern Ohio	Akala and Lal (2001)
WSA(g kg^{-1})	0–10	867	711	24	Grass	Jackson and Vinton county, OH	Shukla et al. (2004)
	0–10	520	610*	16		Wise County, VA	Bendfeldt et al. (2001)
		630	560	23	Forest	Logan County, West Verginia	Thomas et al. (2000)
EC (dS m^{-1})	0–10	0.19	0.26	24	Grass	Jackson and Vinton county, OH	Shukla et al. (2004)

WSA–water soluble aggregates, SAR–sodium adsorption ratio, C/N–carbon nitrogen ratio, BD–bulk density.

[a] Concentration in %.

[b] Stock in Mg ha^{-1}.

restoration of the ecosystem to increase biomass productivity and sequester C in soil (Table 2). However, certain extreme soil conditions (nutrient deficiencies or toxicity) may occur which inhibit plant growth. It is important that these constraints are identified first; otherwise the whole restoration process may fail after a few years. With a careful planning, ecosystem restoration of RMSs can be achieved and become self-sustaining in the long-term.

3.2. Causes of soil carbon loss

The soil C pool has two fractions, SOC and soil inorganic C (SIC). The SOC fractions, predominant form of C in soils of humid and sub-humid regions, have humic and non-humic substances. The SIC fractions, predominant form of C in soils of arid and semi-arid regions, have carbonates and bicarbonates. Both fractions can release C from soil in the form of CO_2, while the SOC fraction can also emit CH_4. The loss of C pool in disturbed soil usually occurs by mineralization, erosion and leaching (Izaurralde et al., 2000), changes in soil moisture and temperature regimes, and reduction in the amount of biomass returned to the soil. Increase in soil temperature, increases the rate of mineralization of the SOC pool. The exposed subsoil rich in calciferous materials is subject to climatic factors leading to dissolution of carbonates and emission of CO_2 to the atmosphere. However, the depleted SOC pool can be restored through conversion to an appropriate land use, and adoption of RMPs (Lal et al., 2003; Lal, 2004a). This is possible by increasing growth of biomass and returning biomass to the soil at a rate in excess of the mineralization capacity.

3.3. Carbon sequestration in reclaimed mine soil

Carbon sequestration is essentially the process of transforming atmospheric CO_2 into biomass through photosynthesis, and incorporation of biomass into the soil as humus. Soils contain approximately 75% of the terrestrial C pool—three times more than the amount stored in living plants (Houghton et al., 1985; Schlesinger, 1986). Therefore, soils play a major role in the global C cycle.

3.3.1. Mechanism of C protection in soil

Protection of soil C is an integral part of C sequestration. Different mechanisms are identified for the protection of SOC (Greenland et al., 1992) including biological, chemical, physical, physicochemical, biochemical and biophysical (Fig. 3). Chemical protection is through bonding between minerals, and formation of recalcitrance compounds (Bayer et al., 2001), which can limit accessibility of decomposers to the organic inputs. The physical protection is through formation of aggregates (Six et al., 2002; Blanco-Canqui and Lal, 2004; Pulleman and Marinissen, 2004). Soil microaggregates are particularly crucial to long-term sequestration because they protect C against decomposition, resulting in much longer residence time for C. Soil aggregation results from the rearrangement, flocculation and cementation of the soil particles, which is mediated by SOC, biota, ionic bridging, clay, and carbonates (Bronick and Lal, 2005). Aggregation can be enhanced by encouraging root growth, fungal hyphae, and SOC (Materechera et al., 1992; Tisdall et al., 1997; Denef et al., 2002). This can enmesh primary soil particles together while realigning them and releasing organic compounds, which can bind particles together (Bronick and Lal, 2005).

The protected SOC is mostly located in 0.2–5 mm aggregates (Chevallier et al., 2004). The physico-chemical protection is due to sorption of organic substrates onto clay mineral grains (Christensen, 1996; Arnarson and Keil, 2005). The biological protection is through involvement of microbes and earthworms in protecting the soil by initiating the formation of microaggregates within large macroaggregates leading to long-term stabilization of SOM against microbial decay (Bossuyt et al., 2005; Pulleman et al., 2005). Microbes and soil animals promote aggregation by excreting binding agents and forming fecal pellets (Lynch and Bragg, 1985) and root exudates flocculate colloids to bind or stabilize aggregates (Glinski and Lipiec, 1990).

3.3.2. Carbon sequestration potential

Enhancing SOC pools of degraded soils can improve soil quality and ecosystem productivity in addition to improving the environment. The C content of spoil material is typically very low compared to undisturbed surface soils. Therefore, the potential to increase the C capital of RMSs is significant. Degraded mine soils accumulate CO_2 through development of soil horizons. The disruption of natural ecosystems during mining and other activities with an attendant decline in ecosystem productivity is part of the impetus for reclamation act that helps restore soil functions as quickly as possible to sustain different ecosystem processes interlinked with soil, water and plant. However, stability of a disturbed ecosystem, a function of SOM accumulation, transformations, soil water and gas exchange processes, may require long period of 30 years or more to attain an equilibrium state (Sopper, 1992) unlike in undisturbed ecosystem. The low SOC concentration in RMSs can be enhanced by: (a) proper reclamation, (b) adoption of RMPs, (c) improvement in soil fertility using integrated nutrient management (INM) technologies, (d) nutrient cycling by returning biomass to the soil, and (e) growing leguminous annuals or tree plants with potential of biological N_2-fixation (BNF). The balances between C

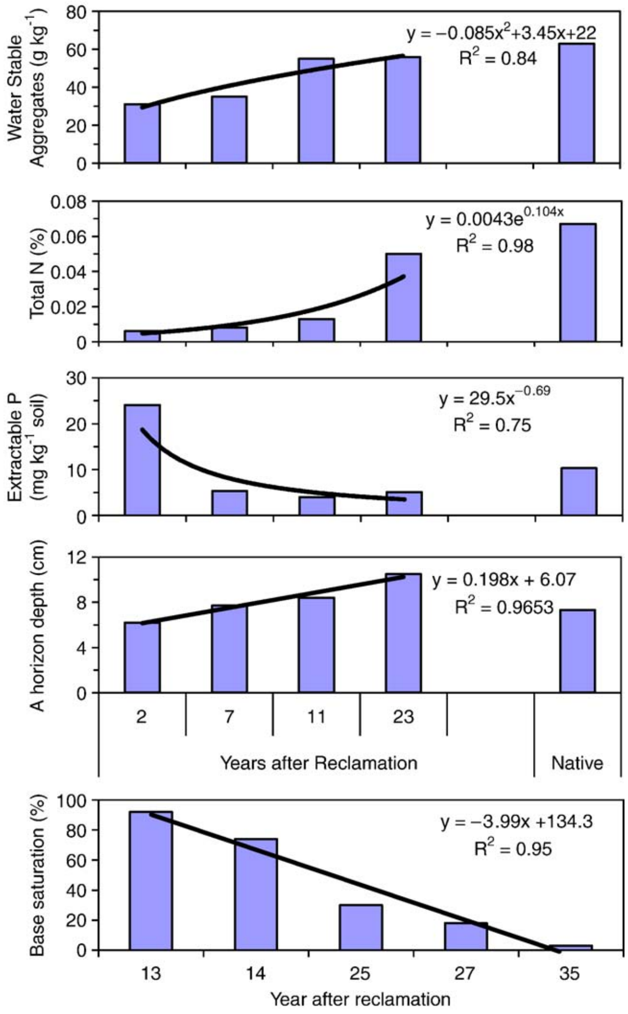

Fig. 2. Effect of mine reclamation age on water stable aggregates, soil nitrogen, phosphorus, base saturation and development of A horizon (data from Ziemkiewicz and Takyi, 1990; Thomas et al., 2000; 2001).

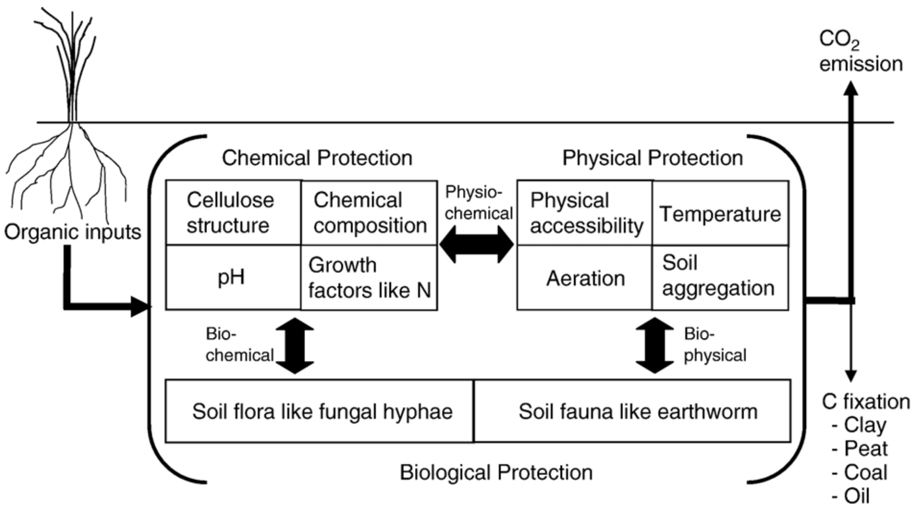

Fig. 3. Factors affecting organic carbon protection and release (modified from Cambardella and Elliot, 1993; Christensen, 1996; Van Noordwijk et al., 1997; Palumbo et al., 2004; Bossuyt et al., 2005).

Table 4
Effect of organic manure in improving soil and biomass productivity of reclaimed mine soil

Soil properties	Source and amount of organic manure	Soil depth (cm)	Amended mine soil	Un-amended mine soil	References
BD (Mg m^{-3})	25 Mg poultry manure and 40 Mg sawdust ha^{-1}	0–10	1.4	1.5	Coyne et al. (1998)
	280 Mg ha^{-1} flue gas desulfurization by-products	0–10	1.0	1.4	Shukla et al. (2005)
Water holding capacity (kg kg^{-1})	25 Mg poultry manure and 40 Mg sawdust ha^{-1}	0–10	0.21	0.20	Coyne et al. (1998)
Total carbon (g kg^{-1})	25 Mg poultry manure and 40 Mg sawdust ha^{-1}	0–10	9.6	4	Coyne et al. (1998)
	155 Mg biosolids ha^{-1}	NA[1]	18.5	2	Vinson et al. (1999)
	112 Mg ha^{-1} of limestone incorporated into the graded spoil and was covered with 20 cm of graded borrowed topsoil	0–10	30[a]	11[a]	Shukla et al. (2005)
Total nitrogen (mg kg^{-1})	25 Mg poultry manure and 40 Mg sawdust ha^{-1}	0–10	250	210	Coyne et al. (1998)
	22 Mg ha^{-1} sludge	0–10	1000	500	Bendfeldt et al. (2001)
	67 Mg ha^{-1}, biosolid	0–30	700	200	Thompson et al. (2001)
	112 Mg ha^{-1} of limestone incorporated into the graded spoil and was covered with 20 cm of graded borrowed topsoil	0–10	1.9	1.4	Shukla et al. (2005)
Mineral N (mg kg^{-1})	25 Mg poultry manure and 40 Mg sawdust ha^{-1}	0–10	2.34[b]	1.74[b]	Coyne et al. (1998)
	Biosolid, 200 Mg ha^{-1}	0–30	400[c,d]	50[c,d]	Thompson et al. (2001)
	Biosolid, Mg ha^{-1}	15	250[e]	12[e]	Rogers et al. (1998)
Aggregate stability (g kg^{-1})	22 Mg ha^{-1} sludge	0–10	610	570	Bendfeldt et al. (2001)
	112 Mg ha^{-1} of limestone incorporated into the graded spoil and was covered with 20 cm of graded borrowed topsoil	0–10	575	200	Shukla et al. (2005)
pH	Poultry manure, 25 Mg ha^{-1} and sawdust, 40 Mg ha^{-1}	0–10	7.2	6.8	Coyne et al. (1998)
	Limestone slurry by-product (lime cake)	NA	6.0	3.5	Yang et al. (2004)
	Papermill sludge, 112 Mg ha^{-1}	NA	7.6	6.6	Li and Daniels (1997)
	280 Mg ha^{-1} Flue gas desulfurization by-products	0–10	7.5	3.4	Shukla et al. (2005)
Biomass (Mg ha^{-1})	Sewage sludge 92 Mg ha^{-1}	NA	4.51[f]	2.09[f]	Daniels and Haering, 1994
	Papermill sludge, 112 Mg ha^{-1}	NA	2.27	1.73	Li and Daniels (1997)
	Biosolid, 9.9 Mg ha^{-1}	NA	79.9[g]	31.4[g]	Cook et al. (2000)
	Biosolid, 52 dry Mg ha^{-1},	NA	0.67	0.32	Vinson et al. (1999)
	Dry biosolids, 67 Kg ha^{-1}	NA	2100	100	Thompson et al. (2001)
	Green shredded plant material, 220 Mg ha^{-1}	NA	800[h]	100[h]	Thompson et al. (2001)
	Sewage sludge, 200–500 Mg ha^{-1}	NA	769	294	Moreno-Penaranda et al. (2004)

[1] NA = not available.

[a] Soil organic C in Mg ha^{-1}.

[b] NH_4+NO_3− N, average of three observations made in June, July and Nov 1993.

[c] Green shredded materials applied at the rate of 220 Mg ha^{-1}.

[d] Nitrate–N.

[e] Approximately estimated data from figure.

[f] Forage biomass averages of 5 years.

[g] Dry grass biomass.

[h] Dry aboveground biomass.

addition via decomposition of living photosynthetic plants and organisms, and losses from microbial respiration determine the amount of C accumulated in soil.

Although mining is an age old and an ongoing activity, importance of SOC sequestration in RMS is only recently recognized (Akala and Lal, 2000, 2001; Jacinthe et al., 2003; Burger, 2004; Sourkova et al., 2005; Shukla et al., 2005). The research interest in SOC sequestration of RMS ecosystem is focused on enhancing natural capacity of ecosystems to increase rates of organic matter input into soil in a form with a long residence time (Post et al., 2004). However, the functionality of C fluxes and pools in RMS ecosystems, as a whole is still a poorly understood process.

Soil disturbances lead to loss of C and other important properties of an ecosystem. However, these soils are the ones with the high potential to sequester SOC for a long enough time to off-set fossil fuel emissions for the region (Lal et al., 1995, 1998; Akala and Lal, 2001; Jacinthe et al., 2004a). There are about 3.2 Mha of mine soil in the U.S. which have the potential of C sequestration at the rate of 0.5 to 1 Mg C ha^{-1} yr^{-1} through land restoration (Lal, 2000; OSM, 2003; Lal, 2004b) thereby sequestering 1.6 to 3.2 Tg C yr^{-1} into soil, off-setting 5.8 to 11.7 Tg CO_2 yr^{-1} emitted by coal activities.

Enhancing soil fertility is one of the major goals of restoring ecosystems functions of drastically disturbed soils (Whitford, 1988). Adding organic materials to soil stimulates microbial activity, promotes N transformation and nutrient cycling, and accelerates ecosystem recovery. The use of natural organic soil materials like mucks and peat, mulches, soil stabilizers and amendments reduces erosion, improves soil health and thereby increases biomass productivity and C sequestration (Norland, 2000) (Table 4). Several studies have documented that application of biosolids, sludge and manure decreases soil bulk density, improves water holding capacity, increases SOM along with total and mineral N, and changes pH with the attendant increase in biomass productivity (Table 4).

Soil aggregation is an important factor enhancing C sequestration in RMS (Hearing et al., 1993; Malik and Scullion, 1998). Other factors enhancing C sequestration are root growth and biomass productivity. Increase in biomass productivity can be achieved by growing species adaptable to adverse conditions, planting healthy tree seedlings or seeding quality seeds of grasses, seeding at recommended rate and time, mulching, in addition to soil management to assure vegetative establishment (Barnhisel and Hower, 1997). The reclamation of disturbed soil depends on the rate of formation of surface horizon rich in SOC. These rates are affected by cultural practices, land use and plant species (Table 5).

Table 5
Carbon sequestration in reclaimed mine soils

Land use	Depth (cm)	Period (Years)	Rate of C sequestration Mg ha^{-1} yr^{-1}	Reference
Grass	0–15	11	3.1	Akala and Lal (2000)
	0–15	25	0.5–3.1	Akala and Lal (2001)
	0–15	47	0.53	Shukla and Lal unpublished
	–	45	0.13	Wali (1999)
Forest	0–15	14	2.6	Akala and Lal (2000)
	0–15	21	0.7 to 2.3	Akala and Lal (2001)
	–	–	4.0	Burger (2004)
	0–10	40	0.58	Sourkova et al. (2005)

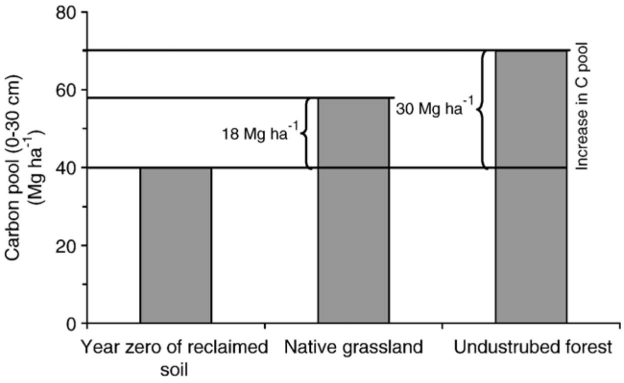

Fig. 4. Carbon sequestration potential of RMS of eastern Ohio as indicated by the differences in SOC pools of different ecosystem (Jacinthe et al., 2004a).

In Ohio, the cumulative SOC sequestration potential of these RMS is about 18 Mg C ha^{-1} for grassland and 30 Mg C ha^{-1} for forestland (30-cm depth), at an average rate of 0.7 to 3 Mg C ha^{-1} yr^{-1} (Akala and Lal, 2000; Jacinthe et al., 2004a) (Fig. 4). The potential of SOC sequestration in mine soils of the U.S. is estimated to be 1.28 Tg C yr^{-1} (Ussiri and Lal, 2005).

Jacinthe et al. (2004a) observed higher SOC concentration in mine reclaimed hardwood plantation than reclaimed grassland (81 vs 71 Mg C ha^{-1}). Among grassland, SOC stocks were higher in reclaimed hay field (79 Mg C ha^{-1}) compared to reclaimed grassland (72 Mg C ha^{-1}), reclaimed meadow (68 Mg C ha^{-1}) and reclaimed pasture (65 Mg C ha^{-1}). Jacinthe et al. (2004b) also reported that after 15 years of reclamation, biomass C was the highest in standard topsoil application technique of reclamation (35 Mg C ha^{-1}) followed by that in the ripped topsoil after reclamation (30 Mg C ha^{-1}), which alleviated compaction. The SOC pool was the least in graded cast overburden (18 Mg C ha^{-1}). The focus on C sequestration in RMS can ensure the long-term success of soil reclamation, as the potential of C accumulation with time in reclaimed forest and pasture is large (Fig. 5).

Land use is a major factor affecting SOC storage (Giuffre et al., 2003). However, the effects of land use changes on ECB are poorly understood (Brye et al., 2002). Factors which affect ecosystem C stocks and fluxes include NPP, biomass decomposition, physical disturbance, movement of soil, introduction of exotic plant and animal species, atmospheric deposition of various materials, and toxic effects on soil decomposers and primary producers (Carreiro et al., 1999). Management practices, which optimize C accumulation, may not only enhance overall soil quality but also help to mitigate CO_2 emitted to the atmosphere by coal mining or combustion and benefit from C credit market.

4. Factors affecting ecosystem productivity and C sequestration in reclaimed mine soils

Several factors like aboveground and belowground biomass, presence of N-fixers, spoil depth, soil compaction level, and

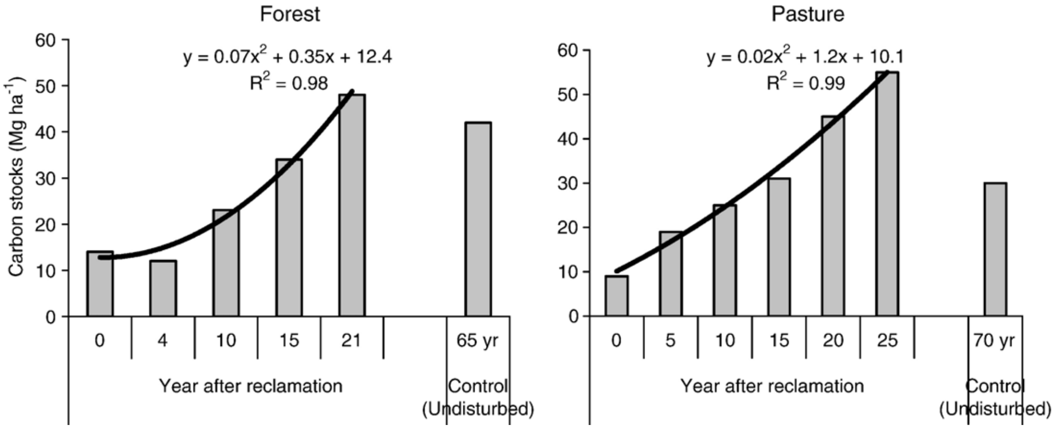

Fig. 5. Carbon stocks in 0–15 cm soil depth in reclaimed mine soils and undisturbed soil of Ohio (drawn using data from Akala and Lal, 2001).

type or source of spoil affect C sequestration. Identification of the factors controlling plant communities in RMSs is useful in identifying reclamation strategies for self-sustaining ecosystem. The SOC is the key indicator of soil quality in RMS ecosystem. However, factors affecting soil properties, SOC addition and protection leading to increase in microbial activity, nutrient availability, soil aggregation, and soil profile development must be better understood to target possible interventions for good biomass production and potential sequestration of atmospheric C into RMSs (Fig. 6).

4.1. Species selection

The reclamation act SMCRA, 1977 requires vegetation to be reestablished on mined soils and that revegetation become self-sustaining. Restoration of vegetation on RMSs is difficult because these soils contain toxic materials, low in nutrient contents and plant available water reserves. Selecting right species of grasses or trees suitable to the soil and environment of RMS ecosystem is the first requirement for the success of the revegetation program. Many studies have been conducted to determine adoption of plant species for revegetation of disturbed soils in the U.S. (Gardiner, 1993; Wali, 1999), France (Hery et al., 2005), South Africa (Blignaut and Milton, 2005), India (Dutta and Agrawal, 2003; Praveen-Kumar Kumar et al., 2005), Australia (Krauss and Koch, 2004), Portugal (Bleeker et al., 2002), China (Ye et al., 2000) and other countries. For example, ryegrass (*Lolium perenne* L.) is tolerant to Cu toxicity and is suitable for metal mine tailings (Hao et al., 2004). On the basis of biomass and primary productivity, hybrid *Eucalyptus* and *Acacia auriculiformis* are suitable for plantation on coal mine spoil land (Dutta and Agrawal, 2003).

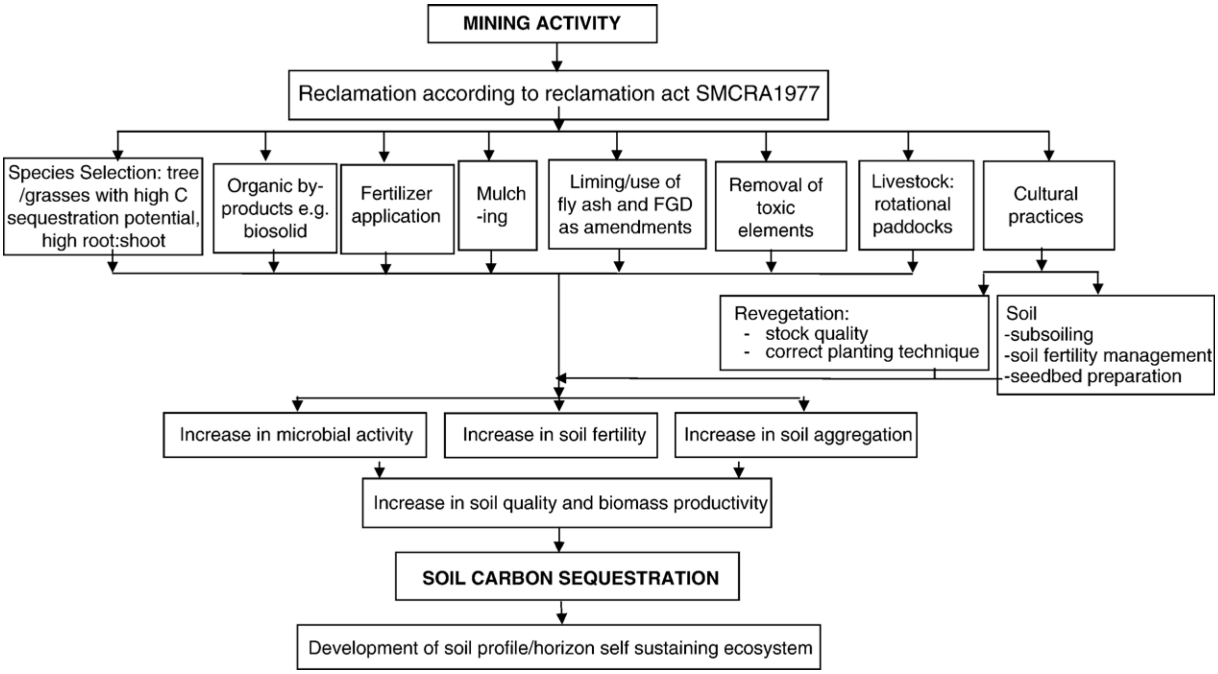

Fig. 6. Factors affecting biomass productivity and C sequestration leading to self-sustaining ecosystem in drastically disturbed soil.

4.2. Application of amendments

The addition of organic amendments (coal combustion by-products, biosolids, swine or poultry manure, sewage or paper mill sludge, sawdust or wood residue, limestone slurry by-product) can ameliorate the drastically disturbed soils. These amendments can alleviate the adverse conditions of the degraded soils through varieties of mechanisms, such as capturing organic C in long-lived soil C pools, improving soil fertility and enhancing plant growth (Table 4). These organic amendments can decrease soil bulk density (Coyne et al., 1998), increase water-holding capacity (Coyne et al., 1998), improve aggregate stability (Bendfeldt et al., 1999) and enhance availability of plant nutrients (Coyne et al., 1998; Vinson et al., 1999; Bendfeldt et al., 2001) (Table 4). Improvements in chemical and physical properties of the soil, improve fertility of degraded soil for crop establishment (Bendfeldt et al., 1999), increase biomass productivity (Daniels and Haering, 1994; Li and Daniels, 1997; Cook et al., 2000), and enhance sequestration of C into soil (Coyne et al., 1998; Vinson et al., 1999). Webber (1978) observed that use of sewage sludge decreased bulk density and increased the percentage of 1–2 mm water stable aggregates.

Inorganic by-products from coal combustion (e.g., fly ash and flue-gas desulfurization material or FGD) in combination with organic amendments (e.g., mulch, residue, biosolids pulp and sludge) can enhance C sequestration (American Coal Ash Association, 1997; Matsi and Keramidas, 1999; Hearing et al., 2000). Application of fly ash can increase population of phosphate solubilizing and N_2-fixing bacteria in soil; improve physical properties (by changing soil structure, increasing porosity and amount of plant available water); decrease cohesiveness of soil particles; and increase water holding capacity, pH, EC and CEC (Gaind and Gaur, 2004; Siddiqui et al., 2004; Lu and Zhu, 2004).

Organic amendments are excellent sources of nutrients for degraded soils but these readily available organic amendments are less likely to be retained in the soil for long-term, contributing little to long-term SOC sequestration. However, these readily available organic amendments, when applied in combination with recalcitrant, lignin rich materials (e.g., paper mill sludge and woody residues), are converted to recalcitrant organic C by microbial oxidation of lignin to polyphenols (Senesi and Loffredo, 1999), thereby contributing to long-term C sequestration.

4.3. Removal of toxicity to self-sustain ecosystem

Although application of organic and inorganic amendments improves soil fertility and increases biomass productivity of RMSs, there are potential risks of contamination of the soil–plant system by heavy metals. For example, fly ash can be a significant carrier of arsenic contamination (Camm et al., 2004). Use of different biological agents to reduce toxic effect of mining can improve soil health and increase biomass productivity. Use of organic C source like molasses to support sulfate-reducing bacteria can remove sulfate from iron mine tailings (Eger et al., 2004); and phytoremediation using canola (*Brassica napus*) and broccoli (*Brassica oleracea*) as biological tools can reduce soluble Se in soils and waters (Bañuelos, 2004). Use of organic amendments can bind metals like Ni, Pb and Cd present in fly ash and FGD by-products (Wong, 1995; Chu and Poon, 1999). Therefore, strategy of increasing SOM concentration of RMSs can also stabilize toxic metals, reduce their percolation into groundwater, and decrease plant uptake.

4.4. Mulching

Mulches can both protect RMSs and enhance revegetation. The soil is protected by shielding it from raindrop impact, and reducing runoff and soil movement. Mulching also increases water infiltration capacity, and protects soils against erosion by water and wind. Agricultural crop residues (e.g., straw or hay), and wood residues can be used as mulching material on disturbed soils. Mulching of RMS affects soil surface temperature and moisture regimes, and SOC stock. Application of mulch in coal mine overburden increases available nutrient status (N, P, K), CEC and exchangeable cations in the soils. Mulching of RMSs improves plant growth, biomass production and nutrient uptake. Straw mulch applied at the rate of 1120 kg ha^{-1} to the surface after sowing grasses and legumes can increase plant stand ratings of several species including *Astragalus cicer*, *Bromus biebersteinii*, *B. inermis*, *B. marginatus*, *Elymus hispidus*, *E. hispidus* subsp. *barbulatus*, *Medicago falcata*, *M. sativa* and *Phleum pratense* grown in a RMSs in Colorado (McGinnies, 1987).

4.5. Livestock management

Controlled grazing at a low stocking rate can aid the reclamation process. Use of livestock for the reclamation, stability, and ecological productivity of copper tailings slopes was awarded the Annual Reclamation Award by Arizona State Mine Inspector (Bengson, 1999). In this system, the livestock are concentrated on relatively small areas for a very short duration and are fed hay. An abundance of organic matter is incorporated into the tailings by the hoof action of the animals. As the organic matter builds up in the sterile tailings, a soil-like medium is developed which enhances the reclamation of the tailing site. As plant communities develop, a self-sustaining ecosystem is established (Bengson, 1999).

5. Methods for the estimation of ecosystem carbon budget in reclaimed mine soils

The ECB in RMSs is defined as the net ecosystem production (NEP), which is net C exchange between the RMS ecosystems and the atmosphere, and includes C pools and fluxes for the ecosystem. There are no published data on ECB in RMSs ecosystems. However, few studies have been conducted on nitrogen, phosphorus and water balances in RMSs (Robertson et al., 1996; Gast et al., 2001; Knappe et al., 2004). Different methods like meteorological (eddy covariance), ecological, model using data or inventory, and tree ring chronologies used in other ecosystems (Table 6) can also be adopted for RMS ecosystems. However, these methods must be tested and validated. The NEP is determined by the difference of

Table 6
Carbon budget of grassland, forest and agricultural ecosystem

Type of ecosystem	Location	Method of budget estimation	C budget^1 (g C m^{-2} yr^{-1})	References
Grassland ecosystem				
Miscanthus sinensis	Nagano, Japan	Ecological method	−100 to −56	Yazaki et al. (2004)
Pasture	New Zealand	Mass balance and modelling	−414	Tate et al. (2000)
Grassland	Cork, Ireland	Eddy covariance	+236	Leahy et al. (2004)
Grass (200 kg N ha^{-1})	Uppsala, Sweden	Ecological method	+140	Paustin et al. (1990)
Tall-grass prairie	Texas, USA	Bowen ratio/energy balance	+50 to +80	Dugas et al. (1999)
	Oklahoma, USA	Eddy covariance	−8	Suyker and Verma (2001)
	Wisconsin, USA	Difference method^2	−410 to +70	Brye et al. (2002)
Mixed-grass prairie	North Dakota, USA	Bowen ratio/energy balance (soil flux)	+31	Frank and Dugas (2001)
Moist-mixed prairie	Alberta, Canada	Eddy covariance	−18 to +21	Flanagan et al. (2002)
Meadow	Moscow, Russia	Ecological method	+387	Larionova et al. (1998)
Forest ecosystem				
Aspen-lime-birch	Moscow, Russia	Ecological method	+135	Larionova et al. (1998)
Scots pine forest, 40 y old (*Pinus sylvestris*)	Southern Finland	Eddy covariance	+228	Kolari et al. (2004)
French pine forest (*Pinus pinaster*)	Les Landes, France	Eddy covariance	−200 to −340	Kowalski et al. (2003)
Boreal and temperate forest of Ontario	Ontario, Canada	Model: CBM-CFS2	−40	Liu et al. (2002)
Ontario's forest ecosystem	Ontario, Canada	Model: CBM-CFS2	−43	Peng et al. (2000)
Indigenous forest	New Zealand	Mass balance and modelling	−136	Tate et al. (2000)
Agricultural ecosystem				
Mix agricultural crops	Denmark	Eddy covariance	−31	Soegaard et al. (2003)
Barley–no fertilizer	Uppsala, Sweden	Difference method^2	−20	Paustin et al. (1990)
–120 kg N	Uppsala, Sweden	Difference method^2	+10	Paustin et al. (1990)
Corn–continuous	Ohio, USA	Cropland ecosystem model C (CEM)	+26	Evrendilek and Wali (2004)
–chisel plowed, fertilized	Wisconsin, USA	Difference method^2	−90 to +590	Brye et al. (2002)
–no till, fertilized	Wisconsin, USA	Difference method^2	−210 to +430	Brye et al. (2002)
No till corn–soybean	North Central USA	Eddy covariance	+90	Hollinger et al., 2005

^{1}− source,+ sink.
^{2}Difference of input and output.

C absorption by plants and release by soils in RMS ecosystems. In other words, it determines whether a mine ecosystem is a C sink (net positive value) or a source (net negative value). The ECB of drastically disturbed ecosystems like RMS is needed for improving the scientific understanding and better restoration of understudied disturbed ecosystem, and for policy makers to reduce antecedent impact on the environment. The following methods of estimating ECB can be useful for RMS with some modification based on the climate, age of RMSs, vegetation, etc.

5.1. Terrestrial or regional or national scale

5.1.1. Carbon balance based on modelling

The technique of modelling C balance is widely used for terrestrial or regional C balance (Smith and Heath, 2004; Ito, 2005). A terrestrial ecosystem model (Sim-CYCLE) has been used in Japan to estimate NEP using three gross fluxes (Ito, 2005):

$$\mathrm{NEP} = \mathrm{GPP}-\mathrm{AR}-\mathrm{HR} \tag{1}$$

where, GPP, AR, and HR are gross primary productivity, autotrophic plant respiration and heterotrophic soil microbial respiration, respectively.

A positive value of NEP indicates net C uptake in reclaimed mine ecosystems. The forest C budget simulation model "FORCARB2", used in the U.S., estimates and projects forest C budget based on inventory data which includes different pools of C like live tree, dead tree, harvested wood, down dead wood, forest floor and soil C (Heath et al., 2002; Smith and Heath, 2004). However, it does not consider decomposition of SOM (heterotrophic respiration). The CENTURY model is also used for simulating C dynamics for different ecosystems (Parton et al., 1993). It includes a decomposition sub-model, a water budget sub-model, and two plant production sub-models (grassland and forest).

Terrestrial C budget of a forest ecosystem in Canada has been estimated by using an Integrated Terrestrial Ecosystem C-budget (InTEC) model (Chen et al., 2000) and C Budget Model for the Canadian forest sector (CBM-CFS2) (Apps and Kurz, 1991; Li et al., 2003). This model is designed to investigate C budget using forest inventory data, but it does not adequately represent the effects of future climate change (Liu et al., 2002).

5.1.2. Eddy covariance

Eddy covariance measurements provide continuous observations of ecosystem level exchanges of CO_2, water, and energy between terrestrial ecosystems and the atmosphere.

FLUXNET is a global network of micrometeorological tower sites, which use eddy covariance methods (Baldocchi et al., 2001). The parts of Global network are AMERIFLUX in the U.S. (Suyker and Verma, 2001), CARBOEUROPE in Germany, AsiaFlux in Japan, KoFlux for Korea, Fluxnet-Canada in Canada, OZFLUX in Australia and ChinaFlux in China. The combination of instrumentation used in this system allows high precision flux measurements. Although eddy covariance can provide reliable estimates of the photosynthesis and respiration at ecosystem or regional scale (Jarvis et al., 1997), there is still uncertainty whether it can provide accurate estimate of net C budget.

5.1.3. Carbon balance based on tree ring chronologies

A method of estimating volume balance of the world's boreal forests has been proposed by Auclair and Bedford (1997) Eq. (2):

$$\begin{aligned}&\text{forest volume balance}\\&\quad=\text{total of forest depletions}-\text{total of forest accruals}\end{aligned} \tag{2}$$

$$\begin{aligned}\text{where, depletions} &= \text{wildfire, harvest, damage by insect and pest}\\ \text{accruals} &= \text{increased tree growth and regrowth}\end{aligned}$$

Estimated forest volume balance is converted to C balance of forest ecosystem by a factor in a range from 4 to 7 (i.e. 4–7 Mg C storage in biomass and soil of boreal and mixed forest ecosystems per Mg of C in volume balance). The C balance of terrestrial ecosystems is uncertain due to methodological problems resulting in incomplete accounting (Houghton, 2003) and uncertainty in the behavior of soil C stocks (Malhi et al., 1999).

5.2. Ecosystem or farm scale

An ECB of a farm or an ecosystem is useful in determining different components affecting pools and fluxes of C, and analyzing whether the system or farm understudy is a C sink (net positive value) or source (net negative value). Brye et al. (2002) estimated ECB by the difference between NEP and heterotrophic respiration (HR) in prairie and agroecosystem. In other words, difference between the sum of C inputs (aboveground net primary productivity and belowground net primary productivity) and outputs (heterotrophic soil respiration, leaching, and export from harvest and burning) is expressed as Eq. (3):

$$\text{NEP} = \text{NPP}-\text{HR}-\text{leaching}-\text{ export from harvest and burning} \tag{3}$$

where, NPP = above and belowground net primary productivity

The heterotrophic soil respiration is estimated by assuming mean annual heterotrophic soil respiration to total soil surface CO_2 flux ratio of 0.25 for grassland ecosystem. Yazaki et al. (2004) proposed a method similar to that of Brye et al. for estimating ECB from the difference between NPP, HR and removal by mowing (RM). The difference in two methods is that, the heterotrophic respiration is calculated by the difference between soil respiration (SR) and root respiration (RR), and the leaching is not included. Yazaki et al. (2004) estimated annual NPP from monthly NPP using the following Eq. (4):

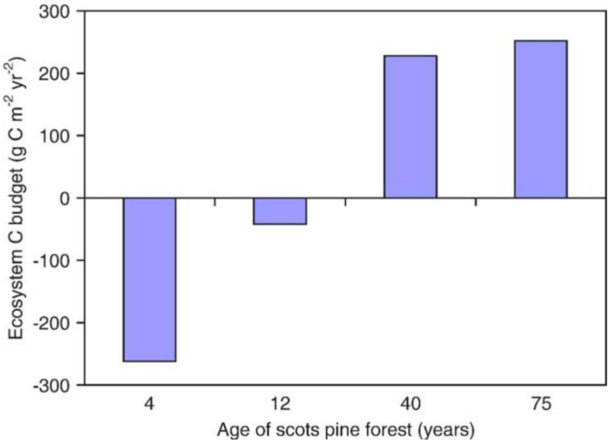

Fig. 7. Ecosystem C budget of different aged Scots pine forests (*P. sylvestris*) in Southern Finland (data used from Kolari et al., 2004).

$$\text{NPP} = \Delta B + L + G \tag{4}$$

where, ΔB is the monthly increment in live plant biomass, L is the monthly increment in dead plant biomass, and G is the monthly grazing loss. The annual NPP is estimated from the sum of monthly NPP. Therefore, the equation of Yazaki et al. (2004) can be expressed as Eq. (5):

$$\text{NEP} = \text{NPP}-\text{HR}-\text{RM} = \text{NPP}-(\text{SR}-\text{RR})-\text{RM}. \tag{5}$$

The ECB is affected by the balance between C uptake through photosynthesis and loss by respiration. Schulze et al. (2000) observed a steady state level in ECB. However, Buchmann and Schulze (1999) reported that a mature ecosystem does not reach a steady state and can continue to act as a net C sink. The ability of an ecosystem to continue to act as net C sink may be attributed to continuous addition of C from above and belowground biomass contributions from plants, thereby leading to an increase in the long-term SOC pool (Law et al., 2001).

These approaches show that *Miscanthus sinensis* ecosystem of Japan and grassland of New Zealand are C source. However, grasslands of Ireland, mixed-grass prairie of USA and meadows of Russia are C sink (Table 6). The available data show that NEP for Scots pine forests (*Pinus sylvestris*) ecosystem at the initial establishment period of 15 years is negative. It becomes positive as ecosystem becomes older (Fig. 7).

6. Ecosystem carbon budget for reclaimed mine ecosystems

There is a long history of research on the reclamation of degraded and disturbed soil ecosystems (Kohnke, 1950; Vogel and Berg, 1968) and recent interest on potential for RMS ecosystem to be a C sink (Akala and Lal, 2000; Bendfeldt et

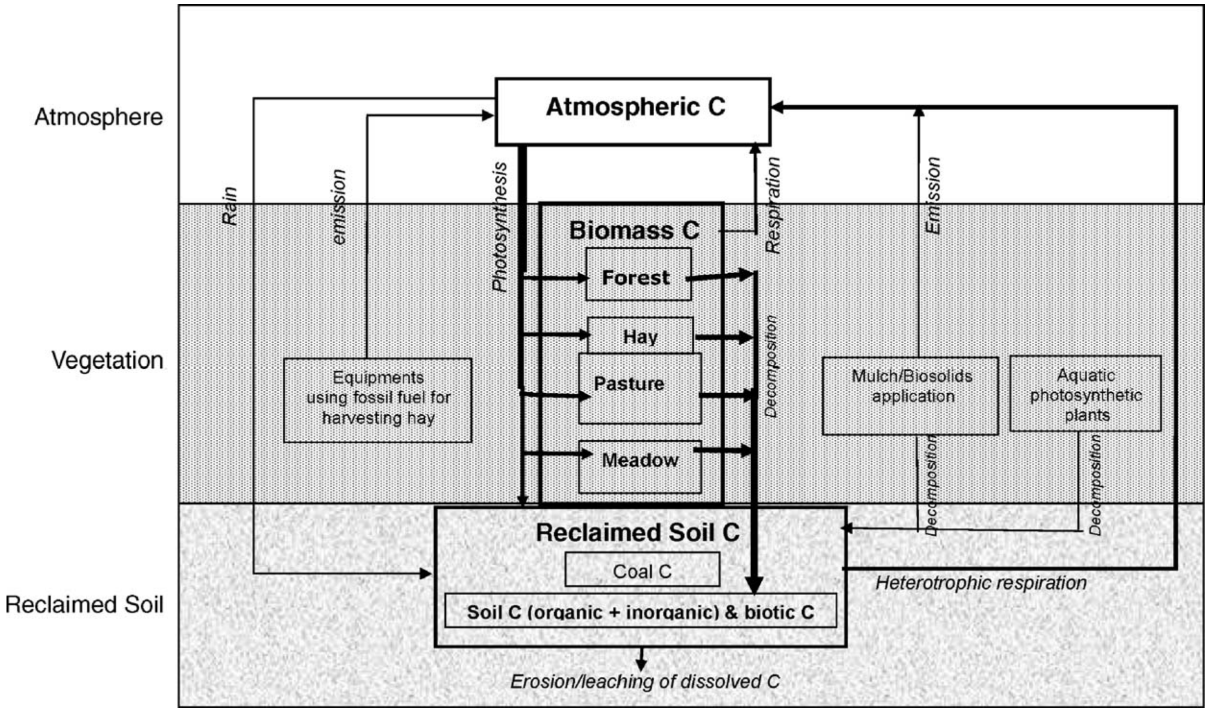

Fig. 8. A flow chart showing principal pools and fluxes of C in a RMS ecosystem.

al., 2001). However, ECB of drastically disturbed ecosystems are poorly understood. The data on ECB for reclaimed grass, forest and agricultural ecosystem are not available unlike in grassland, forest and agricultural ecosystem (Table 6). Therefore, inventories are needed to understand and estimate the ECB of RMSs.

The different components of pools and fluxes of C in RMS ecosystems are outlined in Fig. 8. Understanding and estimating these pools and fluxes provide inventories for the estimation of ECB for RMS ecosystem. The ECB for different reclamation practices can be estimated by the modified method of Paustin et al. (1990) using data collected on C input and output of the system understudy. The proposed equation for ECB calculation is as follows (Eq. (6)):

$$\text{ECB} = \sum \text{C input} - \sum \text{C output} \tag{6}$$

Carbon input of a RMS ecosystem includes aboveground and belowground biomass, C from amendments (like manure, mulch, etc.), and precipitation. Similarly, C output includes C loss from the ecosystem by respiration, erosion and leaching (Fig. 8).

7. Conclusions

The enforcement of Surface Mining Control and Reclamation Act of 1977 has effectively reduced the off-site impacts, improved aesthetic quality, reconstructed topography and hydrologic patterns, and led to the development of soil pedogenic processes over time. The quality of RMSs has been improved for alternative land uses. The C sequestration potential in reclaimed mine soil is as high as 4 Mg ha^{-1} yr^{-1}. Thus, reclamation of mined soil has a high potential to sequester SOC for a long enough time to off-set C emissions by mining activities in the region. However, the functionality of C fluxes and pools, and assessment of ECB in RMSs are rarely studied. There is a need to use research techniques (e.g. eddy covariance, modeling, and ecological) to estimate ECB for RMSs. Post-reclamation assessment of ECB is important to understanding C sink capacity and identifying practices with high ECB. Carbon sequestration in RMSs can be achieved by adoption of: (a) recommended reclamation techniques, (b) recommended soil and vegetation management practices, (c) INM technologies for enhancing soil fertility, (d) land use practices which improve nutrient cycling by returning biomass into the soil, and (e) cover crops and agroforestry system with potential for BNF. Carbon sequestration can provide additional benefits through trading C credits in addition to environmental benefits. Therefore, careful and well-planned research is needed for obtaining inventories of fluxes and pools of C for the estimation of NEP. This review has identified numerous researchable priorities to enhance C pools of RMSs ecosystem. Some of these are briefly described below.

- The literature survey has shown that plant species, spoil depth, and spoil materials influence biomass productivity and litter quality in RMSs. However, none of the studies have assessed the relationship of these parameters with C sequestration.
- A large number of studies indicated beneficial effects of using organic amendments in restoring RMSs. However, information is still lacking on source, quality and amount of organic amendments for different reclamation practices and their effects on soil characteristics.

- Some published studies have focused on assessing physical, chemical and biological properties of the RMSs. However, information is still lacking on the effect of post-reclamation land use on soil properties and the rate of profile development.
- Plant establishment and growth are poor on compacted RMSs, leading to poor C sequestration potential. These soil-related constraints are indication of the importance of methods of reducing sub-soil compaction on C sequestration and NEP.
- Mine soils are sources of GHGs. Quantification of year-round emissions from RMSs is necessary to identify practices with the least risks of GHG emissions.
- Recent studies have shown large potential of C sequestration in RMSs. However, residence time of the sequestered C for long-term stability, and net ecosystem productivity of diverse reclamation practices are still unknown.
- The long-term effects of reclamation practices on ecosystem carbon budget are unknown. Studies on long-term productivity of RMSs are needed to identify sustainable management options.

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