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Review article

Carbon emission from farm operations

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Abstract

This manuscript is a synthesis of the available information on energy use in farm operations, and its conversion into carbon equivalent (CE). A principal advantage of expressing energy use in terms of carbon (C) emission as kg CE lies in its direct relation to the rate of enrichment of atmospheric concentration of CO_2. Synthesis of the data shows that estimates of emissions in kg CE/ha are 2–20 for different tillage operations, 1–1.4 for spraying chemicals, 2–4 for drilling or seeding and 6–12 for combine harvesting. Similarly, estimates of C emissions in kg CE/kg for different fertilizer nutrients are 0.9–1.8 for N, 0.1–0.3 for P_2O_5, 0.1–0.2 for K_20 and 0.03–0.23 for lime. Estimates of C emission in kg CE/kg of active ingredient (a.i.) of different pesticides are 6.3 for herbicides, 5.1 for insecticides and 3.9 for fungicides. Irrigation, lifting water from deep wells and using sprinkling systems, emits 129 ± 98 kg CE for applying 25 cm of water and 258 ± 195 for 50 cm of water. Emission for different tillage methods are 35.3 kg CE/ha for conventional till, 7.9 kg CE/ha for chisel till or minimum till, and 5.8 kg CE/ha for no-till method of seedbed preparation. In view of the high C costs of major inputs, sustainable management of agricultural ecosystems implies that an output/input ratio, expressed either as gross or net output of C, must be >1 and has an increasing trend over time.

Keywords: Carbon budgeting; Tillage systems; Sustainability; Greenhouse effect; Energy use

1. Introduction

Adoption of recommended management practices (RMPs) for agriculture involves off-farm or external input which are carbon (C)-based operations and products (Pimentel, 1992; Marland et al., 2003). Production, formulation, storage, distribution of these inputs and application with tractorized equipment lead to combustion of fossil fuel, and use of energy from alternate sources, which also emits CO_2 and other greenhouse gases (GHGs) into the atmosphere. Thus, an understanding of the emissions expressed in kilograms of carbon equivalent (kg CE) for different tillage operations, fertilizers and pesticides use, supplemental irrigation practices, harvesting and residue management is essential to identifying C-efficient alternatives such as biofuels and renewable energy sources for seedbed preparation, soil fertility management, pest control and other farm operations.

Sustainable use of soil, water and other non-renewable resources implies: (i) an efficient use of all off-farm input, (ii) minimal leakage or losses through leaching, volatilization and erosion, (iii) maintenance or enhancement of soil quality and (iv) minimal risks of environmental degradation such as pollution of water and emission of GHGs into the atmosphere. Land use and land cover change and agricultural practices contribute about 20% of the global annual emission of carbon dioxide (CO_2) (IPCC, 2001). A significant part of the emission due to agricultural practices can be reduced by the worldwide adoption of RMPs.

With reference to C emissions, agricultural practices may be grouped into primary, secondary and tertiary sources (Gifford, 1984). Primary sources of C emissions are either due to mobile operations (e.g., tillage, sowing, harvesting and transport) or stationary operations (e.g., pumping water, grain drying). Secondary sources of C emission comprise manufacturing, packaging and storing fertilizers and pesticides. Tertiary sources of C emission include acquisition of raw materials and fabrication of equipment and farm buildings, etc. Therefore, reducing emissions implies enhancing use efficiency of all these inputs by decreasing losses, and using other C-efficient alternatives.

The data available in literature on energy use for these practices are reported in diverse units such as volume

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(gallons or liters) of diesel, weight (kg, Mg) of coal, calories (kcal, Mcal), joules (MJ, GJ) and other units of energy (BTU) and energy or electricity (kW h). Such diverse units make it extremely difficult to compare the C cost of these practices. Therefore, it would be useful to convert these diverse units into kg CE for different farm operations to assess the real C cost of production systems and to develop and identify C-efficient technologies. Thus, the objectives of this manuscript are to: (i) collate and synthesize the available information in the literature on energy use for direct and indirect input involved in agricultural practices, (ii) convert energy use into kg CE and (iii) assess the sustainability of specific management systems in terms of the long-term changes in C output/input ratio.

2. Conversion coefficients for fuel sources and energy units

The data reported in the literature were converted into kg CE using emission coefficients for a wide range of fuel sources (Table 1). Although conversion coefficients vary within a fuel source (e.g., different types of coal have different conversion coefficients), an average value was used for simplification. Similarly, diverse energy units used in the literature were converted to kg CE using the conversion factors outlined in Table 1. A significant advantage of using kg CE rather than other energy units lies in its direct application to the rate of enrichment of atmospheric CO_2, which is a major global issue at the dawn of the 21st century. The data presented in the following sections for kg CE for different farm operations are organized into primary, secondary and tertiary sources.

2.1. Carbon emission from primary operations

Tillage and irrigation are among the most important primary sources of CO_2 emission.

Table 1
Carbon emission coefficients for different fuel sources and the energy conversion units (Boustead and Hancock, 1979; Fluck, 1992)

Fuel source/energy units	Equivalent carbon emission (kg CE)
(a) One kg of fuel	
Diesel	0.94
Coal	0.59
Gasoline	0.85
Oil	1.01
LPG	0.63
Natural gas	0.85
(b) Units	
Million calories (mcal)	93.5×10^{-3}
Gigajoule (GJ)	20.15
BTU	23.6×10^{-6}
Kilowatt hour (kW h)	7.25×10^{-2}
Horsepower	5.41×10^{-2}

2.1.1. Tillage

Tillage, all operations involving mechanical soil disturbance for seedbed preparation, affects emission directly and indirectly. Direct emissions are due to the fuel use for tillage, which depends on numerous factors including soil properties, tractor size, implement used and depth of tillage. The fuel requirement increases with increase in depth of plowing and tractor speed (Collins et al., 1976), and also differs among the type of equipment used. The direct fuel consumption is also more for heavy than light-textured soils, and increases with increase in soil's cone index (Collins et al., 1976). In Nebraska, USA, Shelton et al. (1980) reported diesel consumption of 17.5 l/ha for moldboard plow, 9.3 l/ha for chisel plow and 7.4 l/ha for disk plow. Lockeretz (1983) reported fuel requirements ranging from 18.0 to 46.0 l/ha for moldboard plow, from 12.4 to 20.2 l/ha for chisel plow and from 5.6 to 11.2 l/ha for tandem disk. Schrock et al. (1985) estimated fuel use in Kansas at 18.9 l/ha for moldboard plow, 10.2 l/ha for chisel plow and 8.01 l/ha for disk. Using 81 kW tractor in North Carolina, Bowers (1989) estimated fuel requirements at 15.1–25.1 l/ha for moldboard plow, 9.5–15.9 l/ha for chisel plow, 7.2–11.7 l/ha for offset disk and 4.8–9.5 l/ha for tandem disk. Stout (1984) reported that diesel fuel requirements ranged from 14.7 to 21.6 l/ha for moldboard plowing, 6.6 to 14.3 l/ha for chisel plowing, 11.3 to 14.9 l/ha for sub-soiling, 6.7 to 12.2 l/ha for disking, 2.8 to 4.5 l/ha for using cultivator and 1.5 to 2.4 l/ha for using a rotary hoe.

Köller (1996) reported that the diesel fuel consumption was 49.4 l/ha for moldboard plow, 31.3 l/ha for chisel plow, 28.4 l/ha for disk plow, 25.2 l/ha for ridge plant and 13.4 l/ha for no-till system of seedbed preparation. Thus, reduction in fuel consumption in comparison with plow-based tillage system was 37% for chisel plow, 43% for disk plow, 49% for ridge plant and 73% for no-till. Lobb (1989) estimated that the fuel use (l/ha) and energy value (MJ/ha), respectively, for different tillage operations were 12.4 and 557 for moldboard plow (11.2 kg CE/ha), 9.2 and 416 for chisel plow (8.4 kg CE/ha), 6.5 and 294 for disking (5.9 kg CE/ha), 4.0 and 182 for cultivator (3.7 kg CE/ha), 3.6 and 162 for interrow cultivator (3.3 kg CE/ha) and 2.9 and 131 for rotary hoe (2.6 kg CE/ha).

The data in Table 2 show that the average C emission is 15.2 kg CE/ha for moldboard plowing, 11.3 kg CE/ha for sub-soiling, 8.3 kg CE/ha for heavy tandem disking, 7.9 kg CE/ha for chiseling, 5.8 kg CE for standard disking, 4.0 kg CE/ha for cultivation and 2.0 kg CE/ha for rotary hoeing. Therefore, conversion of conventional till (based on moldboard plowing) to reduced till (disking or chisel till) or no-till can lead to drastic reductions in C emissions. For example, carbon emission is 35.3 kg CE/ha for complete tillage (involving plowing, two disking, field cultivation and rotary hoeing), 20.1 kg CE/ha by elimination of moldboard plowing, and merely 5.8 kg K CE/ha by elimination of moldboard plowing, disking, cultivation and hoeing. In

Table 2
Estimates of equivalent carbon emissions for a range of tillage operations

Tillage operation	Equivalent carbon emission (kg CE/ha)	
	Range	Mean ± S.D.
Moldboard plowing	13.4–20.1	15.2 ± 4.1
Chisel plowing	4.5–11.1	7.9 ± 2.3
Heavy tandem disking	4.6–11.2	8.3 ± 2.5
Standard tandem disking	4.0–7.1	5.8 ± 1.7
Sub-soiler	8.5–14.1	11.3 ± 2.8
Field cultivation	3.0–8.6	4.0 ± 1.9
Rotary hoeing	1.2–2.9	2.0 ± 0.9

The data on fuel consumption (mostly reported as gals of diesel/acre) were obtained from FEA/USDA (1977), Stout (1984), Frye and Phillips (1981), Poincelot (1986), Bowers (1992), Swanton et al. (1996) and Borin et al. (1997).

contrast, seeding after chiseling would reduce the emission from 35.3 to 7.9 kg CE/ha.

2.1.2. Irrigation

Irrigation is important to achieving high yields in arid and semi-aid regions. On a global scale, 17% of irrigated cropland leads to 40% of the total production (Postel, 1999). Yet, irrigation is a very C-intensive practice. Sloggett (1979; 1992) estimated that 23% of the on-farm energy use for crop production in the US was for on-farm pumping. The energy required to pump water depends on numerous factors including total dynamic head (based on water lift, pipe friction, system pressure), the water flow rate and the pumping system efficiency (Whiffen, 1991). The energy use depends on the water table depth or the lift height (Plate 1). Batty and Keller (1980) estimated pumping energy needed for different lift heights, and reported that energy required for surface irrigation (MJ/ha m) was 3184 for 0 m lift, 56,250 for 50 m lift and 109,317 for 100 m lift. The energy required was high for hand moved, side roll and center-pivot sprinkle system (Plate 2). In comparison, energy required was low for the trickle system (Plate 3), and was estimated (MJ/ha m) at 20,637 for 0 m lift, 50,118 for 50 m lift and 79,599 for 100 m lift (Batty and Keller, 1980).

Plate 1. Tubewells are commonly used for irrigation in Punjab, India. Energy use depends on the water table depth, which in some areas is falling at the rate of 0.5 m/year.

Plate 2. A central pivot system of sprinkler irrigation used in the western US.

Lacewell and Collins (1986) noted that energy required per acre-foot of water for pumping at 45 psi for 250 m of lift was equivalent to 8.5 million cubic feet (mcf) of natural gas, or 56 gallons of diesel or 653 kW h of electricity. The energy required differed with pumping pressure and the lift height. Sloggett (1986) estimated the energy requirement per acre-foot per psi at 4.3876 kW h of electricity, 0.533 gallon of diesel, 0.5417 gallon of gasoline, 0.0677 mcf of natural gas and 0.6771 gallon of LPG. In addition to water application, there are also installation costs ranging from 9.4 to 121.3 kg CE/ha (Table 3).

The supplemental irrigation used for crop production ranges from 250 to 500 mm per season (Franzluebbers and Francis, 1995). Dvoskin et al. (1976) assessed fuel consumption for lifting irrigation water in several regions of the western US. The C emission ranged from 7.2 to 425.1 kg CE/ha (128.9 ± 97.6 kg CE/ha) for 25 cm of irrigation and from 53.0 to 850.2 kg CE/ha (257.8 ± 195.1 kg CE/ha) for 50 cm of irrigation. Schlesinger (1999) estimated C emission from irrigation at 220–830 kg CE/ha/year. Follett (2001) estimated C emission by pump irrigation at 150–200 kg CE/ha/year depending on the source of energy. West and

Plate 3. A trickle system is a water-efficient system of irrigation.

Table 3
Equivalent C emission for installation of irrigation systems (recalculated from Batty and Keller, 1980)

System	Installation energy (kg CE/ha/year)
Surface without IRRS	9.4
Surface with IRRS	24.6
Solid set sprinkle	121.3
Permanent sprinkle	35.5
Hand moved sprinkle	16.3
Solid roll sprinkle	23.3
Center-pivot sprinkle	21.6
Traveler sprinkle	16.9
Trickle	84.9

IRRS = irrigation runoff return system.

Marland (2002) estimated emission by irrigation at 125–285 kg CE/ha/year. Some industries estimate C emission at the rate of 395 kg CE/ha for furrow irrigation and 216 kg CE/ha for drip irrigation (ITRC, 1994). In comparison, irrigation of winter wheat in Punjab, India, by tubewell was estimated to emit 3–25 kg CE/ha (Singh et al., 1999).

Similar to fertilizer and pesticide use, enhancing water use efficiency (WUE) is important to decreasing emissions. Strategies to improve WUE include eliminating flood and furrow irrigation (Plates 4 and 5) in favor of sprinkler irrigation, for most upland crops (although rice requires flooding), using drip and sub-irrigation, adopting conservation tillage with residue mulch to reduce evaporation losses, and using supplemental irrigation only at critical stages of crop growth. Flood irrigation, the most primitive and wasteful use of water, is widely practiced especially in South Asia, North Africa and China. This wasteful practice can also lead to salinization, and alternative methods must be encouraged.

2.1.3. Sowing, spraying, harvesting and transport

The data on kg CE/ha for harvesting, spraying, fertilizer application and other farm operations are presented in Table 4. Most C-intensive operations include harvesting corn for silage, forage harvesting, knife-down ammonia, combine harvesting corn and soybean, fertilizer spreading, planting potato and spreading/incorporating fertilizers or lime. Windrowing and baling hay are also C-intensive operations (Table 4). There is a strong need to enhance efficiency of these operations and reduce CO_2–C emissions.

Plate 4. Flood irrigation is the most wasteful irrigation system, and except for rice, must be avoided.

Plate 5. Furrow irrigation is comparatively more efficient than flood irrigation system.

Spraying chemicals and sowing/drilling crops have relatively low C costs. As expected, no-till seeding has more C cost than drilling in a plowed field. Lobb (1989) reported that energy use in field spraying operations was 91.1 MJ/ha or 1.8 kg CE/ha. West and Marland (2002) reported that post-production C cost of applying pesticides is about 0.35 kg CE/kg of active ingredient (a.i.). These values are lower than those reported in Table 4. Despite the range of values reported in the literature, the need for improving the efficiency of all farm operations cannot be overemphasized.

Table 4
Estimates of equivalent carbon emissions for other miscellaneous farm operations

Farm operation	Equivalent carbon emission (kg CE/ha)	
	Range	Mean ± S.D.
Knife-down ammonia	10.1	10.1
Spray herbicide	0.7–2.2	1.4 ± 1.3
Plant/sow/drill	2.2–3.9	3.2 ± 0.8
No-till planting	3.7–3.9	3.8 ± 0.1
Chemical incorporation	3.6–7.8	5.7 ± 2.1
Fertilizer spraying	0.5–1.3	0.9 ± 0.4
Fertilizer spreading	5.1–10.1	7.6 ± 2.5
Potato planter	5.6–8.2	6.9 ± 1.3
Windrower	4.1–5.5	4.8 ± 0.7
Rake	1.0–2.4	1.7 ± 0.7
Baler (rectangle)	1.6–5.0	3.3 ± 1.7
Baler (large round)	2.8–8.8	5.8 ± 3.0
Corn silage	13.2–26.0	19.6 ± 6.4
Shred corn stalk	3.5–5.3	4.4 ± 0.9
Soybean harvesting combine	6.2–8.6	7.4 ± 1.2
Corn harvesting combine	8.5–11.5	10.0 ± 1.5
Forage harvesting	9.2–18.0	13.6 ± 4.4

The data on fuel consumption (gallon of diesel/acre) are obtained from Frye and Phillips (1981), Poincelot (1986), Swanton et al. (1996) and Bowers (1992).

2.2. Carbon emissions from secondary sources

Fertilizers and pesticides are among the most important secondary sources of emission.

2.2.1. Fertilizers

Chemical fertilizers were first introduced during the 19th century, and their use is an important input in all modern/commercial cropping/farming systems. Although the efficiency of the 1913 discovery of the Haber–Bosch process has been greatly improved, use of nitrogenous fertilizer is a principal source of CO_2 and N_2O emissions. Therefore, enhancing fertilizer use efficiency and finding alternatives is important to reducing emission of GHGs.

Most industries use a figure of 24,600 BTU of energy per pound of N fertilizer (ITRC, 1994). Southwell and Rothwell (1977) reported that the energy requirement in MJ/kg of N was 78 for anhydrous ammonia, 80 for aqueous ammonia, 90 for ammonium nitrate, 101 for urea and 116 for diammonium phosphate. The energy requirement is estimated at 15 MJ/kg of P_2O_5 for superphosphate, and 8 MJ/kg of K_2O for muriate of potash. Energy requirement for mining and manufacture of liming material ranges from 315 to 2400 Mcal/kg (Terhune, 1980). Lewis (1982) estimated the energy use (MJ/kg of N) at 65.1 for ammonium nitrate, 77.8 for urea, 70.1 for 15:15:21, 66.1 for 22:11:11, 73.4 for 9:24:24 and 68.4 for 17:17:17 compound fertilizer. The energy required ranges from 17.8 to 18.7 MJ/kg P_2O_5 and 7.9 to 8.2 MJ/kg of K_2O for different compound fertilizers. Stout (1990) estimated the energy input at 55–65 MJ/kg of N for ammonia, 11–18 MJ/kg of P_2O_5 and 7–9 MJ/kg of K_2O.

The data in Table 5 show C emission in relation to production, packaging, storage and distribution of fertilizers. Estimates of emission are 0.9–1.8 kg CE/kg N, 0.1–0.3 kg CE/kg P_2O_5, 0.1–0.2 kg CE/kg K_2O and 0.03–0.23 kg CE/kg of $CaCO_3$. Some studies (Lal et al., 1998) have shown that N fertilizer manufacture results in about 0.82 kg CE/kg N. West and Marland (2002) reported that emissions for the production of fertilizers are 0.81, 0.101, 0.08 and 0.007 kg CE/kg of N, P_2O_5, K_2O and lime, respectively. Izaurralde et al. (1997) reported a value of 1.23 kg CE/kg of N, which also included application of fertilizer N.

Table 5
Estimates of carbon emission for production, transportation, storage and transfer of agricultural chemicals

Fertilizer	Equivalent carbon emission (kg CE/kg)	
	Range	Mean ± S.D.
(A) Fertilizers		
Nitrogen	0.9–1.8	1.3 ± 0.3
Phosphorus	0.1–0.3	0.2 ± 0.06
Potassium	0.1–0.2	0.15 ± 0.06
Lime	0.03–0.23	0.16 ± 0.11
(B) Pesticides		
Herbicides	1.7–12.6	6.3 ± 2.7
Insecticides	1.2–8.1	5.1 ± 3.0
Fungicides	1.2–8.0	3.9 ± 2.2

The data in kcal/kg were obtained from Lockeretz (1980), Terhune (1980), Pimentel (1980), Bonnie (1987), Green (1987), Helsel (1992) and Spugnoli et al. (1993).

In contrast to chemical fertilizers, energy input is much less for nutrients from animal manure (Stout, 1990). The CE of fresh manure is estimated at 7–8 g/kg manure. The nutrient composition of manure varies widely, and may contain 0.484 kg N, 0.286 kg P_2O_5 and 0.616 kg K_2O per 100 kg of fresh manure (Stout, 1990).

Being a very C-intensive input, it is prudent to enhance use efficiency of N (by minimizing losses caused by erosion, leaching and volatilization) and also identifying alternate sources through integrated nutrient management (INM) strategies including biological nitrogen fixation, animal manure and other biosolids, and recycling nutrients contained in crop residue.

2.2.2. Pesticides

Pesticides are also extremely C-intensive, and their use is increasing rapidly worldwide, but especially in India, China, Brazil and other emerging economies. Improper use can be a major environmental hazard and a principal source of pollution. Pimentel (1980) estimated that energy required for production, formulation, packaging and transport of various pesticides (Mcal/kg of the active ingredient) ranged from 63 to 100 for fungicides, 61 to 87 for insecticides and 28 to 65 for herbicides. The average energy required for production of pesticides was 67 Mcal/kg of a.i. and was in the order wettable powder < granules = dust < miscible oil. Stout (1990) estimated that energy (MJ/kg a.i.) required for production of herbicides was 203 for 2, 4-D, 238 for atrazine, 374 for trifluralin, 396 for alachlor and 414 for paraquat.

Herbicides (phenoxies) were first introduced in 1945 (McDougall and Phillips, 2003). Subsequently, triazines, thiocarbametes and bipyridyls were introduced in the 1950s, and acetamides, hydroxybenzonitriles, carbonates, pyridines, dinitroanilines, pyridazines and chloracetanilides in the 1960s. Aminoacidis, diphyenyl ethers and cyclohexandiones were introduced during the 1970s; and aryloxyphenoxypropinates, sulfonyl ureas, imidazolinones and sulfonamides during the 1980s. Similar to herbicides, insecticides were also introduced during the 1940s (organochlorines, organiophosphates) and 1950s (carbonates). Several insecticides were introduced during the 1970s (benzoyl ureas, pyrethroids, nereistoxins), 1980s (avermectins) and 1990s (neonicotinoids and hydrazides) (McDougall and Phillips, 2003). While fungicides were initially used during the 19th century, new fungicides have been introduced during the 1940s (dithiocarbamates), 1950s (phthalimides and organophosphates), 1960s (guanidines, benzimidazoles, arboxamides, pyrimidines, morpholines), 1970s (azoles, dicarboxamides, triazoles, pyrroles, henylamides, carbamates), 1980s (quinolines) and 1990s (anilinopyrimidines and strobilurins) (McDougall and Phillips, 2003). The energy use efficiency in production, formulation and packaging of all these compounds is progressively improving.

Estimates of emission range from 1.7 to 12.6 kg CE/kg a.i. for herbicides (with a mean value of 6.3 ± 2.7 kg CE/kg a.i.), from 1.2 to 8.1 kg CE/kg a.i. for insecticides (5.1 ± 3.0 kg CE/kg a.i.) and from 1.2 to 8.0 kg CE/kg a.i. for fungicides (3.9 ± 2.2 kg CE/kg a.i.) (Table 5). West and Marland (2002) estimated 4.4, 4.6 and 4.8 kg CE/kg a.i. for production, packaging and transport of herbicides, insecticides and fungicides. The equivalent C emissions for commonly used herbicides are listed in Table 6, and for fungicides and insecticides in Table 7. Additional energy (0.4 kg CE/kg a.i.) is required for formulations (Green, 1987).

Similar to fertilizers, identifying strategies of integrated pest management (IPM) is important to reducing C emissions from pesticide use. Herbicide use may be reduced by banding rather than broadcast application (Butler and Bode, 1987), applying only during the critical periods of weed growth, and by using genetically modified (GM) crops (e.g., round up ready corn or soybeans). There are several options of reducing herbicide use in combination with conservation tillage (Eadie et al., 1992; Swanton and Weise, 1991; Swanton et al., 1993). Clemens et al. (1995) calculated energy input for weed control on 12 farms in Ontario, Canada. For plow-based systems, emissions were estimated at 18.5–26.4 kg CE/ha, of which 70–100% was due to primary and secondary tillage. For conservation tillage, emissions were estimated at 9.6–28.4 kg CE/ha, of which 5–100% were due to herbicide use. Introduction of transgenic crop varieties, possessing herbicide tolerance and/or insect resistance, has a drastic impact on chemical use and the attendant effect on C emission (McDougall and Phillips, 2003). From the management perspective, there is more potential for reducing emissions by fertility management (using INM and enhancing efficiency) than in weed management (Clemens et al., 1995).

Table 6
Equivalent carbon emissions for common herbicides

Herbicides	Equivalent C emissions (kg CE/kg a.i.)
2, 4-D	1.7
2, 4, 5-T	2.7
Alachlor	5.6
Atrazine	3.8
Bentazon	8.7
Butlylate	2.8
Chloramben	3.4
Chlorsulfuron	7.3
Cyanazine	4.0
Dicamba	5.9
Dinoseb	1.6
Diquat	8.0
Diuron	5.4
EPTC	3.2
Fluazifop-butyl	10.4
Fluometuron	7.1
Glyphosate	9.1
Linuron	5.8
MCPA	2.6
Metolachlor	5.5
Paraquat	9.2
Propachlor	5.8
Trifluralin	3.0

The data in MJ/kg obtained from Green (1987), Green and McCulloch (1974), Helsel (1992) and Clemens et al. (1995).

Table 7
Carbon equivalent for production of different fungicides and insecticides (recalculated from Helsel, 1992; Green, 1987; Green and Mc Culloch, 1974) Green and McCulloch (1974)

Pesticides	Equivalent C emission (kg CE/kg a.i.)
(I) Fungicides	
Ferbam	1.2
Maneb	2.0
Captan	2.3
Benomyl	8.0
(II) Insecticides	
Methyl parathion	3.2
Phorate	4.2
Carbofuran	9.1
Carbaryl	3.1
Taxaphene	1.2
Cypermethrin	11.7
Chlorodimeform	5.0
Lindane	1.2
Malathion	4.6
Parthion	2.8
Methoxychlor	1.4

2.3. Soil erosion and carbon emission

Plowing and other tillage operations also exacerbate soil erosion. Accelerated erosion, either by water or wind, leads to a preferential removal of soil organic carbon (SOC), because it has lower density than the mineral fraction and it is concentrated in the vicinity of the soil surface. In some soils and ecosystems, accelerated erosion may account for more loss of SOC than mineralization (Lucas and Vitosh, 1978; Slater and Carleton, 1938). Soon after conversion from natural to agricultural ecosystems, the loss of SOC due to mineralization may be more than that due to erosion (Gregorich and Anderson, 1985). Subsequently, however, the progressive decline in soil structure and reduction in aggregation may drastically increase erosion-induced loss in SOC (de Jong and Kachanoski, 1988). Consequently, eroded soils are characterized by lower SOC pool than uneroded soils (Lal, 2000, 2003). Rhoton and Tyler (1990) observed that SOC pool to 1-m depth in a fragipan soil in southern Mississippi was 60 Mg/ha in an uneroded phase, 35 Mg/ha in a slightly eroded phase and only 19 Mg/ha in a severely eroded phase. Lal (2000) reported that the magnitude of SOC loss due to historic erosion may be 3–30 Mg C/ha.

The fate of erosion-displaced C is a subject of much debate. Some argue that all of the SOC translocated by erosional processes is mineralized and released into the atmosphere as CO_2 (Schlesinger, 1997). Others argue that all of the SOC displaced by erosional processes is transported to depressional sites and/or aquatic ecosystems and is buried or taken out of circulation (Stallard, 1998; Smith et al., 2001). Lal (1995, 2003) and Jacinthe and Lal (2001) estimated that 20–30% of the SOC displaced is mineralized, some is redistributed over the landscape and only a small part of it is buried in depressional sites and aquatic ecosystems. Lal (1995, 1999) estimated that 1.14 Pg C/year displaced by erosional processes is mineralized and released into the atmosphere. In further analyses, involving the database on sediment transport in world rivers, Lal (2003) estimated that 0.8–1.2 Pg C/year is emitted into the atmosphere by erosional processes. Nonetheless, 0.4–0.6 Pg C/year may be buried in depressional sites and aquatic ecosystems.

There is a strong need for soil/ecoregional specific research on determining the pathways and fate of SOC displaced by erosional processes. Such research needs to be conducted on nested watersheds, which provide information on delivery ratio of sediments including SOC. With appropriate sampling, research data obtained on nested watersheds would yield credible information on the fate of SOC as it is translocated over the landscape.

2.4. Sustainability of different production systems

There are numerous ways to assess sustainability of a production system. Economists use productivity or total factor productivity (Herdt and Steiner, 1995), soil scientists use soil quality (Doran and Parkin, 1994; Bezdicek et al., 1996; Carter et al., 1997), ecologists use energy coefficients (Odum, 1998; Ulgati and Brown, 1998) and engineers assess the energy use efficiency (Lockeretz, 1983; Stout, 1984). In the context of the global climate change and anthropogenic emissions of GHGs into the atmosphere, however, sustainability of a system can be assessed by evaluating temporal changes in the output/input or (output − input)/input ratios of C using a holistic approach (Eqs. (1), (2) and (3)):

$$I_s = \left(\frac{C_O}{C_I}\right)_t \quad (1)$$

$$I_s = \left[\frac{C_O - C_I}{C_I}\right]_t \quad (2)$$

$$I_s = \left[\frac{C_O - C_I - C_{OR}}{C_I - C_{IR}}\right]_t \quad (3)$$

where I_s is the index of sustainability, C_O is the sum of all outputs expressed in C equivalent C_I is the sum of all inputs expressed in C equivalent, C_{OR} is the output in the reference treatment, C_{IR} is the input in the reference treatment, and t is the time in years, which may range as a multiple of 25 years corresponding to one human generation.

The term C_O comprises all output including grains, stover/straw, root biomass and exudates. Similarly, the term C_I can be comprehensive and include direct input and indirect losses in the terrestrial/soil C pool. For example, losses of C due to erosion caused by plow-based tillage must also be included in the C_I term. Tertiary sources of C emission (e.g., manufacture of farm machinery) may also be accounted in the C_I term.

Clemens et al. (1995) conducted energy analysis of tillage and herbicide inputs for alternative weed management system in Ontario. The data in Table 8 show total C input and net C gains vis-à-vis the zero herbicide input system. The index of sustainability computed by using Eq. (3) ranges from a low of 1.8 for high weed control input corn to a high of 26.6 for low weed control input wheat. In general, the I_s value was greater for the low than the high weed control input system (Table 8). Franzluebbers and Francis (1995) reported that output: input ratio, based on energy use, ranged from 4.1 ± 0.5 in fully irrigated, broad-

Table 8
Carbon budget of cropping systems in relation to herbicide use in Ontario, Canada (recalculated from Clemens et al., 1995)

	Carbon emissions (kg CE/ha)											
	Corn			Soybeans			Wheat			Rotation		
	H	L	O	H	L	O	H	L	O	H	L	O
Input												
Weed control	18.9	11.0	5.9	17.5	14.5	5.9	9.1	4.0	4.0	20.0	13.2	5.2
Seedbed preparation	26.6	23.8	23.8	17.9	17.9	17.9	14.2	0	14.2	19.6	13.9	18.7
Operations	21.9	21.9	23.3	13.9	13.9	12.8	10.2	10.2	24.4	15.3	15.3	17.2
Fertilizer manufacture	204.3	151.3	25.2	27.8	20.2	0	127.1	63.6	24.0	119.7	78.3	16.4
Total input	271.8	208.0	78.2	77.1	66.5	36.6	160.6	77.8	66.6	174.6	120.7	57.5
ΔInput	193.6	129.8	–	40.5	29.9	–	94.0	11.2	–	117.1	63.2	–
Output												
ΔYield	543.6	561.2	–	128.6	209.8	–	517.4	309.3	–	416.0	370.8	–
Net C gain	350.1	431.4	–	88.1	179.9	–	423.4	298.3	–	299.0	307.5	–
I_s (Eq. (3))	1.8	3.3	–	2.2	6.0	–	4.5	26.6	–	2.6	4.9	–

H = high; L = low; O = zero.

cast herbicide, traditional tillage systems with cereal as previous crop and no N fertilizer to 11.6 ± 2.5 in dryland, broadcast herbicide, traditional tillage systems with legume as previous crop and no N fertilizer. The energy output/input ratio decreased with addition of N fertilizer in all management systems.

Swanton et al. (1996) assessed the energy use efficiency of agriculture in Canada. They defined energy efficiency as energy used (GJ) per ton of crop produced. They reported that energy efficiency improved over the period between 1975 and 1991 because of improved crop varieties (more stress tolerance, genetic gains). Indeed, the energy use per hectare decreased by about 40% for corn and 20% for soybean, respectively. The data in Table 9 show the energy use for corn and soybean production for different levels of herbicide input. For corn, C emission in kg CE/ha was 32.3 for high input, 27.8 for low input and 23.0 for minimum input. A considerable amount of C emission can be avoided by substituting some herbicides. The C saved by substitution of herbicides was 15.3 kg CE/ha in high input and 5.2 kg CE/ha for low input systems (Table 9). In contrast to corn, C emission in kg CE/ha for soybean was 14.6 for high, 20.7 for low and 23.0 for minimal input. Use of inter-row cultivation and rotary hoeing increased C emissions in low and minimal input systems.

Borin et al. (1997) assessed the output/input and (output − input)/input ratio for three tillage methods expressed in units of energy for three crops grown in northeastern Italy. The output/input ratio was 4.1 for conventional till, 4.2 for ridge till and 4.6 for no-till. In comparison, the (output − input)/input ratio was 2.9 for conventional till, 3.0 for ridge till and 3.6 for no-till system. Borin et al. also computed the SOC pool for three tillage systems. They observed that saving in C (fuel and soil C) was 637 kg C/ha/year upon conversion from plow till to ridge till and 832 kg C/ha/year upon conversion to no-till. Of this, saving in fuel alone was equivalent to 44 kg C/ha/year for ridge till and 62 kg C/ha/year for no-till (Borin et al., 1997).

Table 9
Estimates of C emission for alternative weed management options in corn and soybean production in Ontario, Canada (recalculated from Swanton et al., 1996)

Input source	Equivalent C emission (kg CE/ha)		
	High	Low	Minimum
(a) Corn			
(i) Experiment			
Glyphosate	10.4	10.4	10.4
2, 4-D	3.5	3.5	3.5
Metolachlor + linuron	18.5	7.4	–
Inter-row cultivation	–	6.5	6.5
Rotary hoeing	–	–	2.6
Total	32.3	27.8	23.0
(ii) Assumed case of replacing metolachlor + linuron			
Nicosulfuron/rimsulfuron + bromoxynil	3.1	2.2	–
Revised total	17.0	22.6	23.0
C saved	15.3	5.2	0
(b) Soybean			
Glyphosate	10.4	10.4	10.4
2, 4-D	3.5	3.5	3.5
Imazethepyr	0.7	0.3	–
Inter-row cultivation	–	6.5	6.5
Rotary hoeing	–	–	2.6
Total	14.6	20.7	23.0

Energy analyses for sugar beet production under traditional and intensive farming systems in Morocco was assessed by Mrini et al. (2002). Total energy involved, computed as C equivalent, was 522 kg C/ha in small farms and 1078 kg C/ha in large farms. Direct input (fuel and electricity) represented 43.2% in small farm operations and 70.7% in large farm operations. The most important indirect C input was in the form of nitrogenous fertilizer, which represented 30.2% for small farms and 21.1% for large farms. In comparison, machinery represented 11.5% of the C input for small farms compared with 5.8% for large farms. Transport and seedbed preparation, respectively, accounted for 22% and 20% of C input on small farms. In large farms, C input involved 33% for irrigation, 27% for fertilizers, 18% for seedbed preparation and 12% for transport. Total energy outputs were 3263 kg C/ha for small farm and 4472 kg C/ha for large farms. Using these data, the I_s computed by Eqs. (1) and (2) are as follows:

(a) Small farms

$$I_s = C_O/C_I = \frac{3263 \text{ kg C/ha}}{522 \text{ kg C/ha}} = 6.3$$

$$I_s = (C_O - C_I)/C_I = \frac{3263 \text{ kg C/ha} - 522 \text{ kg C/ha}}{522 \text{ kg C/ha}} = 5.3$$

(b) Large farms

$$I_s = C_O/C_I = \frac{4472 \text{ kg C/ha}}{1078 \text{ kg C/ha}} = 4.1$$

$$I_s = (C_O - C_I)/C_I = \frac{4472 \text{ kg C/ha} - 1078 \text{ kg C/ha}}{1078 \text{ kg C/ha}} = 3.1$$

Thus, C use efficiency is greater for small compared with large farm systems.

Dyer and Desjardins (2003) assessed the impact of farm machinery management on emission of GHGs from Canadian agriculture. They concluded that substantial reductions in the fossil fuel related GHG emissions from Canadian agriculture were possible by adopting the following:

(i) eliminating or reducing summer fallow as a method of weed control,
(ii) converting conventional plowing to minimum or no-till system,

(iii) substituting tillage implements such as the chisel plow for the traditional moldboard plow and
(iv) converting cropland to pasture on marginal agricultural land.

Dyer and Desjardins concluded that eliminating primary tillage had the largest potential reduction in emission of GHGs.

3. Conclusions

Any criteria used to assess sustainability of land use and management system must address the issues of the time. At the dawn of the 21st century, principal global issues include the accelerated greenhouse effect, emission of CO_2 and other GHGs from agricultural practices and food security in relation to soil and environmental degradation. There are several agricultural practices that are C-intensive because of the fossil fuel and energy involved in their use. Important among these are plowing, fertilizers and pesticides, and irrigation. A careful assessment is needed to reduce their use, and to enhance use efficiency of these practices. Conversion of plow till to no-till, using integrated nutrient management and integrated pest management practices, and enhancing water use efficiency by adopting drip irrigation and sub-irrigation practices can save C emission and at the same time increase soil C pool. Adopting a holistic approach to management of soil and water resources, which decreases losses, improves efficiency and enhances agronomic productivity per unit consumption of C-based input is an important strategy. Sustainability of a production system can be assessed by evaluating temporal changes in C output to C input ratio or the net C output to C input ratio. The objective of sustainable management is to enhance the ecosystem C pool by increasing output, improving use efficiency of C-based input and decreasing losses.

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