Climatic Change (2007) 80:73–90
DOI 10.1007/s10584-006-9167-6

Long-term modeling of soil C erosion and sequestration at the small watershed scale

R. C. Izaurralde · J. R. Williams · W. M. Post · A. M. Thomson · W. B. McGill · L. B. Owens · R. Lal

Received: 27 July 2005 / Accepted: 18 May 2006 / Published online: 21 December 2006

Abstract The soil C balance is determined by the difference between inputs (e.g., plant litter, organic amendments, depositional C) and outputs (e.g., soil respiration, dissolved organic C leaching, and eroded C). There is a need to improve our understanding of whether soil erosion is a sink or a source of atmospheric CO_2. The objective of this paper is to discover the long-term influence of soil erosion on the C cycle of managed watersheds near Coshocton, OH. We hypothesize that the amount of eroded C that is deposited in or out of a watershed compares in magnitude to the soil C changes induced *via* microbial respiration. We applied the erosion productivity impact calculator (EPIC) model to evaluate the role of erosion–deposition processes on the C balance of three small watersheds (~1 ha). Experimental records from the USDA North Appalachian Experimental Watershed facility north of Coshocton, OH were used in the study. Soils are predominantly silt loam and have

R. C. Izaurralde (✉) · A. M. Thomson
The Joint Global Change Research Institute,
8400 Baltimore Avenue, Suite 201, College Park, MD 20740-2496, USA
e-mail: Cesar.Izaurralde@pnl.gov

J. R. Williams
Blacklands Research Center, Texas A&M University,
808 East Blacklands Road, Temple, TX 76502, USA

W. M. Post
Oak Ridge National Laboratory, Building 1509, Bethel Valley Road,
PO Box 2008 MS6335, Oak Ridge, TN 537831-6335, USA

W. B. McGill
College of Science and Management, University of Northern British Columbia,
3333 University Way, Prince George, BC V2N 4Z9, Canada

L. B. Owens
North Appalachian Experimental Watershed, USDA-Agricultural Research Station,
28850 SR 621, Coshocton, OH 43812-0488, USA

R. Lal
School of Natural Resources Food, Agricultural and Environmental Sciences,
The Ohio State University, 422B Kottman Hall, 2021 Coffey Road, Columbus, OH 43210, USA

developed from loess-like deposits over residual bedrock. Management practices in the three watersheds have changed over time. Currently, watershed 118 (W118) is under a corn (*Zea mays* L.)–soybean (*Glycine max* [L.] Merr.) no till rotation, W128 is under conventional till continuous corn, and W188 is under no till continuous corn. Simulations of a comprehensive set of ecosystem processes including plant growth, runoff, and water erosion were used to quantify sediment C yields. A simulated sediment C yield of 43 ± 22 kg C ha^{-1} $year^{-1}$ compared favorably against the observed 31 ± 12 kg C ha^{-1} $year^{-1}$ in W118. EPIC overestimated the soil C stock in the top 30-cm soil depth in W118 by 21% of the measured value (36.8 Mg C ha^{-1}). Simulations of soil C stocks in the other two watersheds (42.3 Mg C ha^{-1} in W128 and 50.4 Mg C ha^{-1} in W188) were off by <1 Mg C ha^{-1}. Simulated eroded C re-deposited inside (30–212 kg C ha^{-1} $year^{-1}$) or outside (73–179 kg C ha^{-1} $year^{-1}$) watershed boundaries compared in magnitude to a simulated soil C sequestration rate of 225 kg C ha^{-1} $year^{-1}$ and to literature values. An analysis of net ecosystem carbon balance revealed that the watershed currently under a plow till system (W128) was a source of C to the atmosphere while the watersheds currently under a no till system (W118 and W188) behaved as C sinks of atmospheric CO_2. Our results demonstrate a clear need for documenting and modeling the proportion of eroded soil C that is transported outside watershed boundaries and the proportion that evolves as CO_2 to the atmosphere.

1 Introduction

The soil (organic) carbon (C) balance is determined by the difference between C inputs (e.g., plant litter, crop residues, decaying roots, organic amendments, depositional C) and C outputs (e.g., soil respiration, dissolved organic C leaching and eroded C). While most of these C transfers occur vertically, erosion and deposition are lateral processes; thus, their account in the soil C balance depends on the scale of the system under consideration (i.e., field, landscape, region, etc.). The rates of C eroded from or deposited on the landscape are usually not considered in mass balance determinations due to difficulties in their estimation. Furthermore, most long-term experiments that yield information on the soil C balance have been conducted on relatively flat terrain thereby removing or significantly attenuating the influence of erosion and deposition on the soil C balance (Paul et al. 1997; Powlson et al. 1996). Consequently, most of the changes in the soil C balance reported in these experiments have been attributed to the difference between C inputs from net primary productivity (NPP) and C outputs from soil respiration. Rare is the case when lateral transfers of C due to erosion are discussed (Izaurralde et al. 2001) or estimated (Campbell et al. 2000; Gregorich et al. 1998; Monreal et al. 1997).

Cole et al. (1997) used estimates of C losses summarized by Davidson and Ackerman (1993) from 18 field experiments (mostly from North America) to project a global loss of 55 Pg C due to the cultivation of mineral soils since ~1800. This type of C loss estimate, however, does not consider lateral transfers of C from eroding to depositional sites and therefore disregards the possible C flux associated with these soil and C mass transfers. De Jong and Kachanoski (1988), for example, estimated that about half of the total SOC losses in Canadian prairie soils could be ascribed to soil erosion.

Lal (1995) argued that because soil aggregates break down during erosion, physically-protected C becomes available for decomposition and, thus, subject to loss as CO_2. He estimated that water erosion could induce an annual C flux of 1.14 Pg C from soil to the atmosphere. This calculation was derived on the basis that rivers transport annually to oceans ~19 Pg of soil sediments. Assuming a 10% delivery ratio, Lal (1995) estimated that ~190 Pg of soil could be in motion every year due to water erosion. This would represent 5.7 Pg C, assuming an average soil C content of 30 g kg^{-1}. The 1.14 Pg C flux from soil to the atmosphere was estimated assuming that 20% of the soil C displaced is oxidized by microbial activity. Lal (2003) further expanded this hypothesis to include wind erosion and the possible increased emissions of other greenhouse gases such as N_2O and CH_4.

Stallard (1998) arrived at a different conclusion when trying to link terrestrial sedimentation to the C cycle. He hypothesized that a significant amount of C eroded from fields becomes buried in depressional areas and is thus sequestered and unavailable for decomposition. With time, the C eroded from agricultural lands is replaced by new C fixed by plants growing on both eroding and depositional sites. Stallard (1998) estimated that up to 1.5 Pg C $year^{-1}$ could be sequestered globally by these processes. Results of a latitudinal model across 864 scenarios (that included conditions for wetlands, alluviation + colluviation, eutrophication, soil C replacement, wetland net ecosystem productivity, and CH_4 fluxes) suggested a human-induced C sink of 0.6–1.5 Pg C $year^{-1}$. Using a constrained budget approach across the conterminous US and then expanding the estimates to the global scale, Smith et al. (2001) derived a C sink (~1 Pg C $year^{-1}$), which was similar to that estimated by Stallard (1998). While Lal's and Stallard's hypotheses agree on the dimension of the problem, they diametrically oppose in sign. Is erosion then a sink or a source of C to the atmosphere?

Stallard's hypothesis prompted further studies trying to link erosion–sedimentation processes to the C cycle. Harden et al. (1999) sampled disturbed and undisturbed loess soils in Mississippi and then used soil C and N data to parameterize the Century model (Parton et al. 1987, 1993, 1994) for different erosion and tillage histories. Their results revealed that erosion processes could generate a significant sink for C in sediments where it is protected from decomposition. The simulation results suggested all soil C to be lost during a 127-year period and 30% of the C lost to be replaced after 1950. Later, Liu et al. (2003) used the framework of the Century model to develop the Erosion–Deposition-Carbon-Model (EDCM) in order to simulate the effects of rainfall erosion and deposition on soil organic C at several landscape positions in the Nelson Farm watershed in Mississippi. The simulation results suggested that soil erosion and deposition reduced CO_2 emissions from the soil to the atmosphere by the process of C sequestration at eroding sites and C burial at depositional sites. Because EDCM does not explicitly model rainfall erosion, Liu et al. (2003) input the estimated rates of erosion calculated by Harden et al. (1999).

Erosion–deposition processes, however, may have non-linear interactions with the C cycle. Using ^{137}Cs methods at a small watershed in Maryland, McCarthy and Ritchie (2002) discovered that upland agricultural activities increased C storage within a narrow streamside forest (riparian or wetland buffer) not only because of increased sediment deposition but also due to enhanced NPP.

Clearly, there is need to advance our understanding of the links between erosion–deposition processes and the C cycle in order to improve the accuracy of C budgets constructed at local, regional, and global scales. There are two types of uncertainties concerning the relationship between erosion–deposition processes and the C cycle. The first uncertainty refers to the link existing between erosion/deposition and NPP. At eroding sites, soil C removed by wind, water, or even tillage (van Oost et al. 2004) is replaced with new photosynthetically fixed C (Harden et al. 1999; Liu et al. 2003; Smith et al. 2001; Stallard 1998). At depositional sites, eroded C may be buried (Harden et al. 1999; Liu et al. 2003; Smith et al. 2001; Stallard 1998) and C content may increase even more due to enhanced photosynthetic activity (McCarty and Ritchie 2002). The assumption that eroded sites replace lost C at rates similar to non-eroded sites may not hold true. At two sites in Alberta (Canada), Izaurralde et al. (1998) measured drastic reductions in C inputs in artificially-eroded soils. The fraction of total C added to soil during a five-year period ranged from 0.17 to 0.33 in unfertilized eroded soils relative to unfertilized non-eroded soils. The fraction for fertilized treatments ranged from 0.55 to 0.82.

The second type of uncertainty concerns the fraction of the eroded or deposited C that evolves as CO_2. This fraction has been estimated as essentially being 0.0 (Smith et al. 2001; Stallard 1998), 0.2 (Lal 1995, 2003), or even 1.0 (Schlesinger 1995). The assumption that eroded C essentially undergoes no oxidation when dislodged and transported to a new location contradicts what is known about the physics of soil erosion (Troch et al. 1991) and the physical mechanisms of C protection in soil aggregates (Jastrow and Miller 1998).

Clearly, there is a need for an improved understanding of the role of soil erosion on the C cycle. The objective of this paper is to discover the long-term influence of soil erosion on the C cycle of three managed watersheds. We hypothesize that the amount of eroded C that is deposited in or out of a watershed compares in magnitude to the soil C changes induced *via* management of NPP and heterotrophic decomposition processes. Were the hypothesis confirmed, it would demonstrate the need to include the C fluxes associated with erosion in ecosystem C balance calculations. To test the hypothesis, we use the EPIC model (Williams 1995) to attempt to reproduce historical crop yields, runoff, soil sediment yield, soil sediment C yield, and soil C dynamics documented during several decades at three watersheds near Coshocton, OH. We use the model EPIC because of its ability to simulate soil C dynamics including those associated with wind and water erosion (Izaurralde et al. 2006). An improved understanding of how erosion–deposition processes control the soil C balance at the local scale will support further analysis of these phenomena at larger spatial scales and thus contribute to understanding whether they behave as sources or sinks of atmospheric CO_2.

2 Materials and methods

2.1 Description of experiments

The North Appalachian Experimental Watershed (NAEW; 40°22′N, 81°48′W) is a US Department of Agriculture research station established in 1938 near Coshocton, OH to study runoff and water erosion processes from agricultural land (Fig. 1) (Kelley et al. 1975). Because it is located on a local high ranging in elevation from 300 to 600 m, the 424 ha research site does not receive runoff water from surrounding areas. Berms have been used to divide the watershed into sub watersheds of different size (0.4–1,400 ha) to study the influence of specific management treatments on runoff, erosion, and water quality. These treatments have varied with time according to experimental objectives and also to reflect the evolution of farming practices in the region. Specific treatments included in the modeling experiment are described in Section 2.2.2.

Annual long-term precipitation averages 973 mm. Maximum daily air temperature averages 16.0 °C and minimum daily air temperature averages 3.4 °C. Natural vegetation at NAEW has been classified as mixed oak forest with white oak (*Quercus alba*), black oak (*Q. velutina*) and hickory (*Carya* spp.) being predominant species. The soils at NAEW are non-glaciated and developed from residuum and colluvium parent materials derived from sedimentary bedrock such as coarse-grain sandstone, shale, and limestone (Kelley et al. 1975). Dominant soil types are silt loam in texture and classify as Hapludalfs.

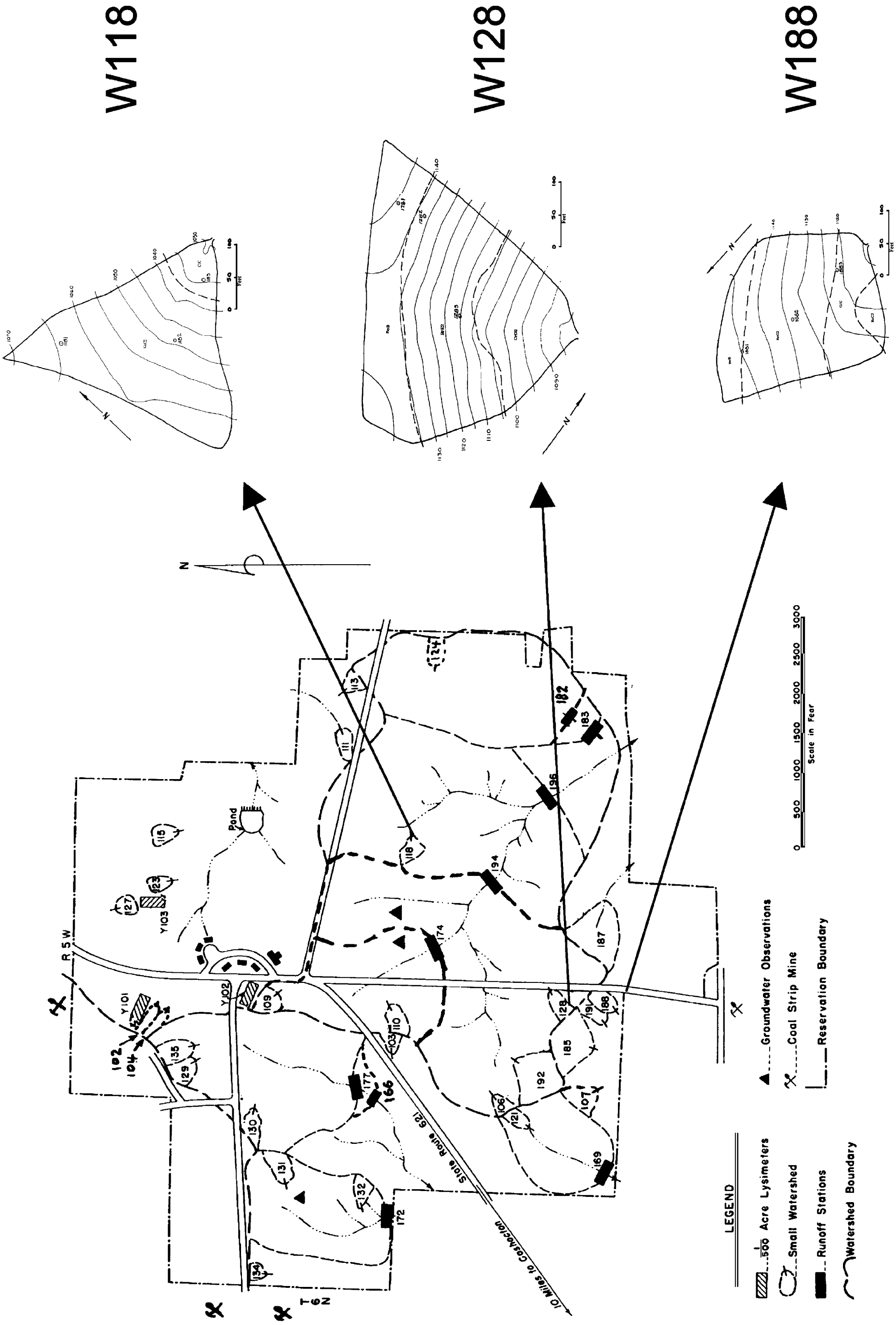

Fig. 1 Map of the North Appalachian Experimental Watershed showing the locations of the watersheds, runoff stations and groundwater observations

2.2 Modeling

2.2.1 The EPIC model

EPIC was originally designed to quantify the effects of wind and water erosion on soil productivity (Williams et al. 1989). Over the last 20 years, EPIC has developed into a terrestrial ecosystem model capable of simulating NPP, crop yields, hydrologic balance, heat balance, nutrient cycling, soil erosion, and a complete suite of environmental controls on plant growth such as irrigation, fertilization, and liming (Williams et al. 1995). Recently, Izaurralde et al. (2006) added and tested new C and N modules to simulate soil C dynamics. One of the major objectives of this work was to examine the interactive effects of soil erosion on the C cycle.

Daily gains in plant biomass in EPIC are proportional to the daily photosynthetically active radiation intercepted by the plant canopy. These daily gains in aboveground plant biomass are affected by vapor pressure deficits, atmospheric CO_2 concentration (Stockle et al. 1992a,b), and other physiological stresses caused by environmental factors such as water, temperature, N, P and aeration. Similarly, daily root growth may be affected by non-optimal values of soil strength, temperature, and aluminum content (Jones et al. 1991). Currently, EPIC can simulate about 100 plant species including crops, native grasses, and trees. Plant competition is simulated as in the ALMANAC model (Kiniry et al. 1992). Daily weather can be estimated from precipitation, air temperature, solar radiation, wind, and relative humidity parameters or it can be input from historical records. Soil information on layer depth, texture, bulk density, and C concentration is needed to drive EPIC.

EPIC is optimized to compute components of the hydrological cycle at the small watershed scale (1–100 ha). Components calculated include snowmelt, surface runoff, infiltration, soil water content, percolation, lateral flow, water table dynamics, and evapotranspiration. Runoff can be calculated directly with the curve number method (US Department of Agriculture (USDA)-Soil Conservation Service 1972) or indirectly by calculating infiltration with an iterative solution of the Green–Ampt equation (Green and Amp 1911). Wind erosion is calculated on a daily time step based on wind speed distribution and adjusted according to soil properties, surface roughness, vegetative cover, and distance across wind path (Potter et al. 1998). Water erosion is computed as a function of the energy in rainfall and runoff. Six equations based on the Universal Soil Loss Equation (USLE) (Wischmeier and Smith 1965, 1978) are available to the user (Williams 1995). The general equation to calculate water erosion is:

$$Y = X \times K \times LS \times C \times P \times RF_f \tag{1}$$

where Y, in the case of the USLE equation, is soil erosion in Mg ha^{-1}, X is the erosivity factor, K is the soil erodibility factor (Mg MJ^{-1}), LS is the slope length and steepness factor (dimensionless, calculated relative to a slope 22.1 m long and with a gradient of 0.09 m m^{-1}), C is the cover-management factor (dimensionless), P is the conservation practice factor (dimensionless), and RF_f is the coarse fragment factor (dimensionless). In the USLE equation, the factor X is the rainfall erosivity index EI_{30} (MJ ha^{-1} year^{-1}), which represents how the energy of falling water and maximum rainfall intensity combine (in a statistical way) to detach and transport soil particles. EPIC also includes a modification of the

USLE equation (Williams 1975) in which X is a function of runoff volume (Q, mm), peak runoff rate (q_p, mm h^{-1}), and watershed area (A, ha):

$$X = 1.586(Q \times q_p)^{0.56} \times A^{0.12} \tag{2}$$

The modified USLE equation, called MUSLE, allows for the calculation of erosion and sediment yield as a function of runoff. Four other variations of USLE and MUSLE are available to the user (Williams 1995; Williams and Izaurralde 2005). In all of these variations, Y is sediment yield (Mg ha^{-1}) because these equations implicitly account for depositional processes that occur within the watershed. The difference between soil erosion (estimated with the USLE equation) and sediment yield (e.g., with the MUSLE equation) should give an estimate of soil deposition within the watershed.

The MUSS (Modified Universal Soil [Loss Equation] Small [Watershed]) equation [Equation (3)], a particular variation of the MUSLE equation, was used in this modeling experiment because it is particularly adapted for small watersheds without channel erosion (William and Izaurralde 2005).

$$X = 0.79(Q \times q_p)^{0.65} \times A^{0.009} \tag{3}$$

EPIC uses a similar approach to the Century model (Parton et al. 1994) to distribute C and N into several pools across up to 15 soil layers: metabolic litter, structural litter, active, slow, and passive. Soil C losses from the watershed occur vertically *via* CO_2 generation during microbial respiration and horizontally *via* transport of sediment and soluble C outside watershed boundaries. Currently, EPIC does not calculate the fraction of the eroded C that, either retained in or transported from the watershed, evolves as CO_2 to the

Table 1 Selected initial soil properties for three small watersheds at the North Appalachian Experimental Watershed in Coshocton, OH (Kelley et al. 1975)

Watershed, soil name	Layer	Layer depth (M)	Bulk density (Mg m^{-3})	pH	Sand	Silt	Clay (g kg^{-1})	Organic C
W118, Coshocton, silt loam	1	0–10	1.47	5.8	123	704	173	111
	2	10–20	1.47	5.7	115	697	187	95
	3	20–30	1.49	4.9	96	643	260	27
	4	30–36	1.52	4.6	113	602	285	21
	5	36–44	1.65	4.5	193	507	300	21
	6	44–69	1.64	4.5	152	523	325	20
	7	69–117	1.54	4.5	278	493	229	20
	8	117–147	1.60	4.5	138	582	280	20
W128, W188, Rayne, silt loam	1	0–5	1.46	5.6	232	604	164	133
	2	5–10	1.46	5.6	232	604	164	133
	3	10–20	1.46	5.7	278	554	168	121
	4	20–30	1.50	5.9	462	354	184	81
	5	30–40	1.51	5.7	458	361	181	69
	6	40–50	1.57	4.7	444	388	168	21
	7	50–60	1.57	4.7	444	388	168	21
	8	60–70	1.57	4.7	432	393	176	21
	9	70–150	1.58	4.6	413	400	187	20

Table 2 Historical management of three sub watersheds at the North Appalachian Experimental Watershed research facility near Coshocton, OH

Watershed number	Area (ha)	Period[a]	Crop rotation[b]	Tillage[c]	References[d]
118	0.79	1951–1970	C–W–M–M	CT	1, 2
		1971–1975	C	CT	
		1976–1983	M, W in 1983		
		1984–2001	C–S	NT	
128	1.08	1966–1973	C–W–M–M	CT	1, 2, 3
		1974–1978	C	NT	
		1979–1983	M		
		1984–2001	C	CT	
188	0.83	1966–1970	C–W–M–M	CT	3
		1971–2001	C	NT	

[a] The historical period extends back to 1939 but only the period of simulation is included here.

[b] *C* corn (*Zea mays* L.); *W* wheat (*Triticum aestivum* L.); *S* soybean (*Glycine max* [L.] Merr); *M* meadow (timothy [*Phleum pretense* L.], red clover [*Trifolium pretense* L.], and alsike clover [*Trifolium hybridum* L.]) sown with wheat in fall. Meadow crops were mowed for hay in late spring and fall.

[c] *CT* Plow till, *NT* no till.

[d] 1 Hao et al. (2001); 2 Hao et al. (2002); and 3 Puget et al. (2005).

atmosphere. The model also simulates transport and losses of dissolved C below the soil profile. For a complete description of the model see Izaurralde et al. (2006).

2.2.2 Description of simulation experiments

We obtained a 47-year weather record (1956–2002) of daily precipitation and air temperature (maximum and minimum) for Coshocton, OH from the National Climatic Data Center (NCDC; Asheville, NC). These daily data were read by the model for selected periods of the simulation. The rest of the weather variables needed to run EPIC (solar radiation, relative humidity, and wind speed were simulated by the model from weather parameters developed for New Philadelphia, OH (latitude 40°21′N, longitude 81°47′W).

Soil layer properties (e.g., layer depth, texture, water retention capacity, organic C, etc.; Table 1) and landscape characteristics (watershed area, slope length and gradient; Fig. 1) for each of the watersheds were obtained from Kelley et al. (1975) and complemented with unpublished records.

We selected three sub watersheds for this modeling study: W118, W128 and W188 (Fig. 1), which had been studied in detail by Hao et al. (2001, 2002) and Puget et al. (2005). Hao et al. (2001) selected W118 to study the impact of historical management on soil and C erosion during a 49-year period (1951–1999). A rotation of plow-till corn (*Zea mays* L.), wheat (*Triticum aestivum* L.), and two years of meadow (timothy [*Phleum pretense* L.], red clover [*Trifolium pretense* L.], and alsike clover [*Trifolium hybridum* L.]) cut for hay was simulated during the first 20 years (Table 2). This was followed by five years of continuous plow till corn, eight years of meadow and 18 years of a corn–soybean (*Glycine max* [L.] Merr.) no till rotation. For W118, EPIC was run from 1951 to 1999 to cover the period of recorded runoff and sediment yield (Hao et al. 2001).

Puget et al. (2005) selected five sub watersheds, including W128 and W188, to assess the impacts of contrasting land use and management on soil C stocks. Watersheds W128 and W188 were under a plow till corn–wheat-meadow-meadow rotation for eight and five years,

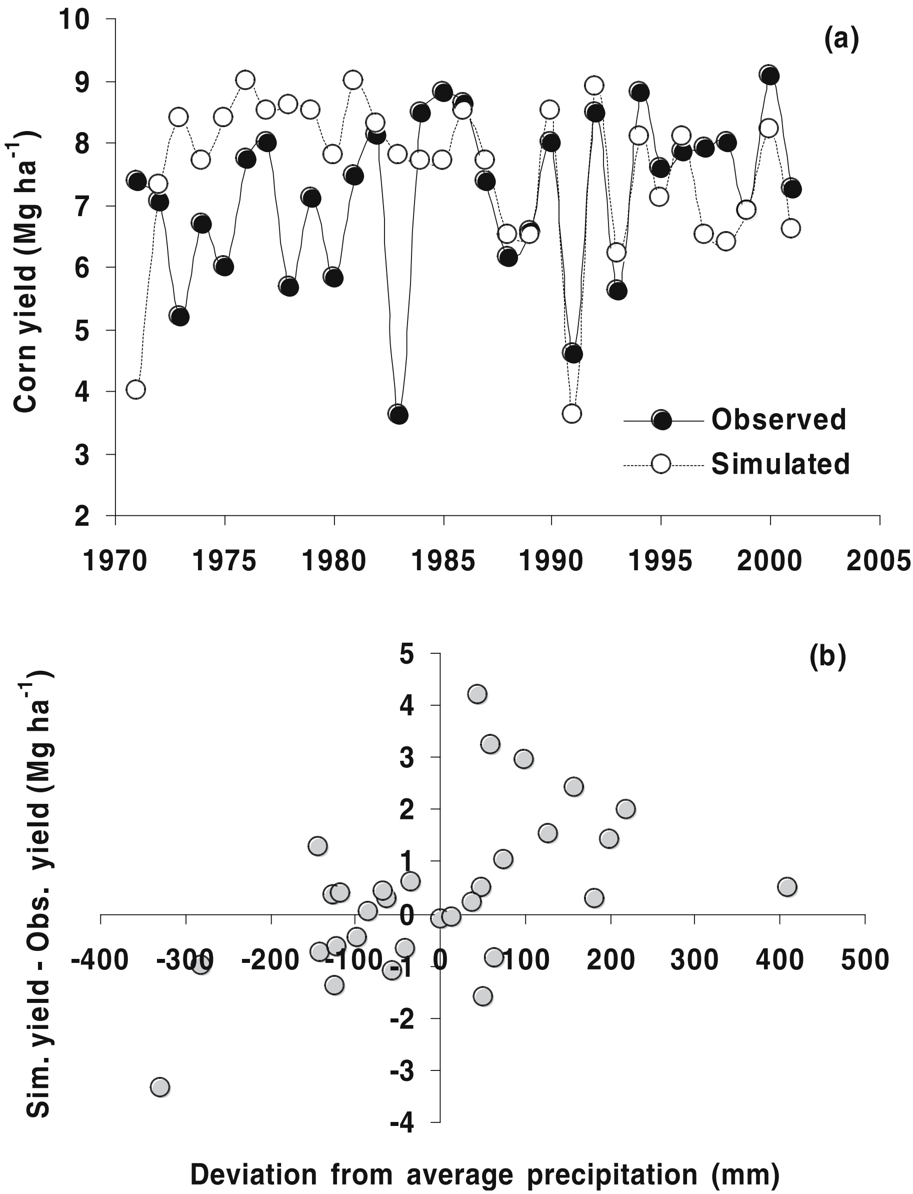

Fig. 2 (a) Observed and simulated dry corn yields in W188 from 1971 to 2001. (b) Simulated/observed corn yield differences in relation to deviations from average precipitation

respectively (Table 2). For W128, continuous no till corn was simulated for five years followed by five years of meadow and then plow till corn until the end of the simulation in 2001. Continuous no till was simulated for W188 from 1971 to 2001. For these last two watersheds, the model was run from 1966 to 2001 to coincide approximately with soil organic C determinations made in 1965 (Kelley et al. 1975) and with soil organic C measurements conducted in 2001 by Puget et al. (2005). Data and outputs from W128 and W188 were used to calculate soil C sequestration rates as a result of adoption of no tillage practices.

3 Results and discussion

3.1 Crop productivity

The EPIC crop growth module has been tested against experimental data many times since its development (e.g., (Bryant et al. 1992; Cavero et al. 1999; Izaurralde et al. 2006; Roloff et al. 1998; Touré et al. 1994; Williams et al. 1989). In general, EPIC has been found to reproduce accurately crop mean yields and interannual variability although inaccuracies in capturing interannual yield trends have also been reported (Roloff et al. 1998; Warner et al. 1997).

Since plant-C input is a main driver of soil C dynamics, it is important to confirm that the EPIC crop growth module approximates correctly the annual inputs of C to soil. Corn yield data were available only for W118 and W188. In W118, corn yields in the corn–soybean no till rotation during 1951–1999 averaged 5.59 ± 1.81 Mg ha^{-1} (n = 18) while EPIC simulated 7.77 ± 1.61 ± Mg ha^{-1}. Also in W118, observed soybean yields in the corn–soybean rotation averaged 1.79 ± 0.49 Mg ha^{-1} (n = 8) while simulated yields averaged 2.67 ± 0.54 Mg ha^{-1}. In W188, simulated no till corn yields averaged 7.52 ± 0.23 Mg ha^{-1} while observed yields averaged 7.16 ± 0.21 Mg ha^{-1} (n = 31). The 1971–2001 no till corn yield record from W188 was used to examine the ability of EPIC to reproduce interannual yield variability (Fig. 2a). EPIC consistently overpredicted yields during the first half of the simulation period (Fig. 2a). After 1985, however, EPIC captured well both yields and yield trends except in the late 1990s when the simulated yields were lower than the observed.

We advance at least two reasons to explain the lack of agreement between simulated and observed yields during the first half of the experiment. First, inadequate weed control (e.g., resistance to triazine herbicides) or a change in the nutrient-cycling regime (e.g., reduced N mineralization) during the first years of no till could have contributed to reduce yields. A slight, albeit non-significant, upward trend in measured yields

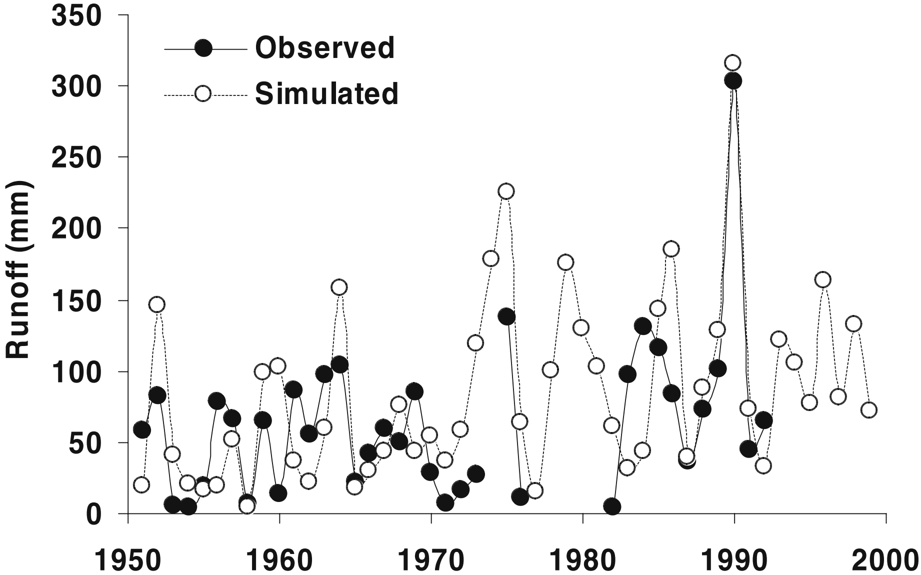

Fig. 3 Temporal dynamics of surface runoff in W118 from 1951 to 1999

Table 3 Observed and simulated runoff, sediment yield, and sediment C yield in W118 averaged by period and crop type

Period	Crop	Precipitation (mm)	Runoff (mm)		Sediment yield (Mg ha^{-1})		Sediment C yield (kg C ha^{-1})	
			Observed	Simulated	Observed	Simulated	Observed	Simulated
1951–1970	Corn	839	59.6	47.2	1.55	1.31	39.9	42.6
1951–1970	Wheat	982	65.1	99.7	0.35	0.24	9.0	7.4
1951–1970	Meadow	897	39.9	31.8	0.00	0.00	0.0	0.3
1971–1975	Corn	930	37.2	122.8	7.36	6.87	191.0	325.7
1976–1983	Meadow	1034	2.2	92.0	0.03	0.01	0.8	0.4
1984–1999	Corn	840	41.5	92.2	0.23	0.08	5.9	7.9
1984–1999	Soybean	996	77.4	132.0	0.95	0.25	24.5	22.0
Average[a]		932	46.4	84.4	1.18	0.92	30.6	43.3
Standard error		21	7.9	9.1	0.47	0.49	12.1	21.8

[a] Average calculated with yearly values, not with averages by period.

suggests improvement in management, including the use of improved cultivars (Fig. 2a). Second, there could be a propensity in EPIC to overpredict yields. Warner et al. (1997) also observed a tendency in EPIC to overpredict corn yields grown under different N rates in Connecticut. In this experiment, the proclivity of EPIC to overpredict yields was evidenced during years wetter than normal (Fig. 2b). Using data from a long-term experiment at Breton, Alberta, Izaurralde et al. (2006) found that while EPIC explained 69% of the yield variability it could explain 89% of the crop residue and root C added to soil.

3.2 Surface runoff, sediment yield, and sediment carbon yield

Prediction of C losses in sediment yield requires a correct approximation of surface runoff and sediment yield. We used a 49-year record of surface runoff, sediment yield, and sediment C yield available from W118 to test the ability of EPIC to simulate these processes. Annual runoff simulated by EPIC followed, in general, the patterns of observed runoff (Fig. 3). There were periods when simulated runoff matched almost exactly in trend and quantity those of observed runoff (e.g., 1987–1992). The previous 5-year period, however, produced obvious reversals. When averaged by crop type within a period, the values of observed runoff did not necessarily follow the expected pattern of higher runoff in annual crops than in meadows (Table 3). Direct comparisons between simulated and observed runoff could not be made during certain periods of the study due to lack of runoff measurements (e.g., only three runoff values were available during 1976–1983). Although EPIC generally simulated annual runoff volumes greater than observed (Table 3), it realistically captured the evolution of runoff during the 49-year period of measurement under changing weather, soil, and management conditions (Fig. 3).

Accompanying comparisons between observed and simulated sediment yield and sediment C yield (Fig. 4 and Table 3) show an improved match. In 1975, during a year of conventional till corn, high runoff values (Fig. 3) corresponded well with high values of sediment and sediment-C yields (Fig. 4). In 1990, however, high runoff values (Fig. 3) did not translate into high values of sediment and sediment-C yields (Fig. 4) likely due to the

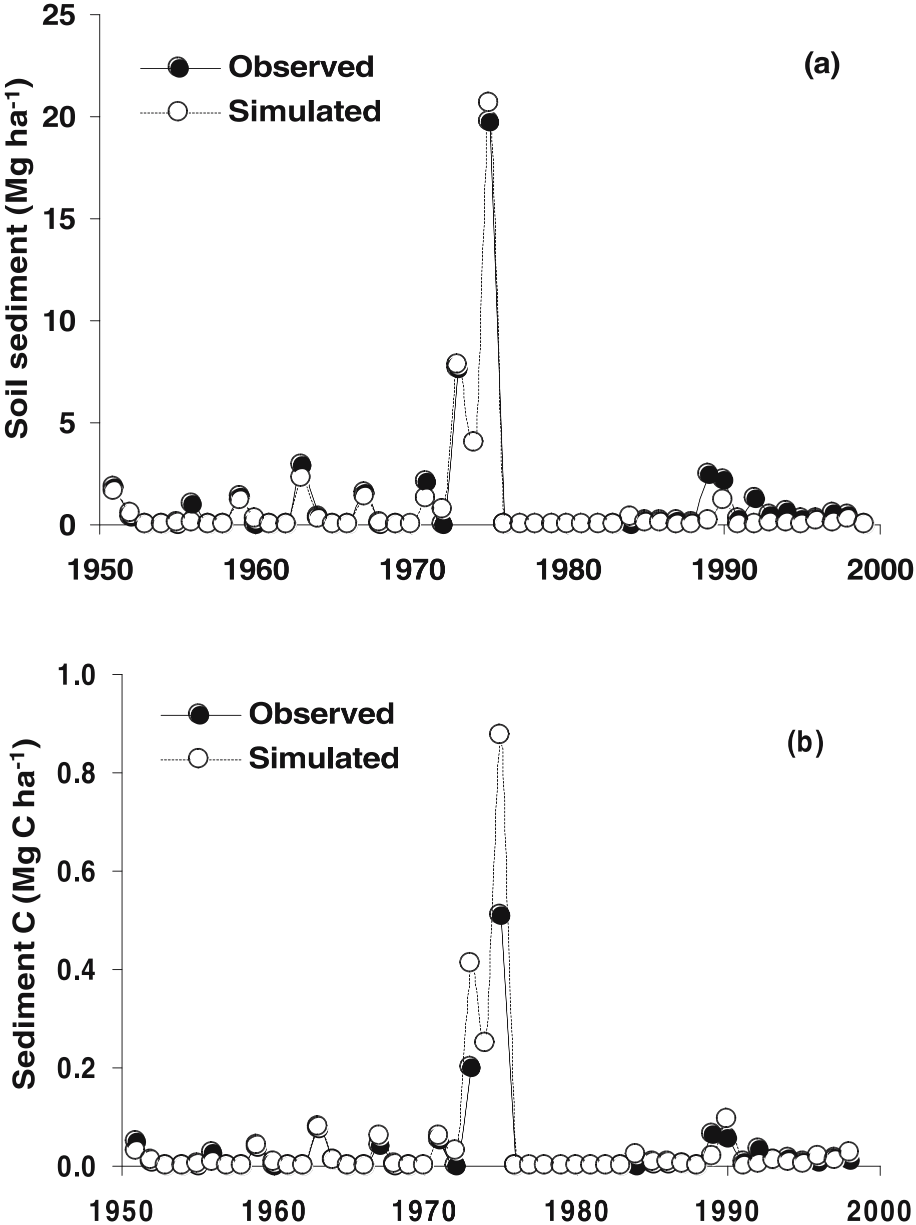

Fig. 4 Temporal dynamics of observed and simulated (a) soil sediment and (b) sediment C in W118 from 1951 to 1999

no till management imposed on the corn–soybean rotation. Observed and simulated sediment yields were highest under conventional till corn followed by conventional till wheat, no till corn and soybean, and meadow. Similar trends could be observed in sediment C yields (Fig. 4) while the largest discrepancy between observed and simulated values occurred in 1975 under conventional till corn.

Table 4 Observed and simulated soil C after 36 years of conventional and no till

Watershed, Reference[a]	Period	Sediment C yield ($Mg\ C\ ha^{-1}$)		Depth (cm)	Final soil C ($Mg\ C\ ha^{-1}$)		
		Observed	Simulated		Observed	Standard Error	Simulated
W118, (1)	1951–1999	1.53	1.92	0–10	18.1	2.65	20.7
				10–20	12.3	2.55	16.9
				20–30	6.5	3.00	7.0
				0–30	36.8		44.6
W128, (2)	1966–2001	–	2.96	0–5	7.4	0.46	11.1
				5–10	8.9	0.53	8.6
				10–20	17.4	0.77	13.3
				20–30	7.5	1.07	9.4
				0–30	41.3		42.3
W188, (2)	1966–2001	–	4.71	0–5	17.4	1.31	12.6
				5–10	11.1	1.08	10.4
				10–20	13.8	0.93	17.8
				20–30	9.1	1.05	9.6
				0–30	51.8		50.4

[a] 1 Hao et al. (2001); 2 Puget et al. (2005).

3.3 Carbon balance

In this section, we analyze the long-term impacts of environmental and management factors on the C balance of the three watersheds of this study (Table 4). The 1.53 $Mg\ C\ ha^{-1}$ of sediment C removed from W118 (currently under a corn–soybean no till rotation) during the 49-year period is comparable to the 1.92 $Mg\ C\ ha^{-1}$ loss simulated by EPIC. Larger sediment C yields were simulated for W128 (currently under continuous conventional till corn) and W188 (currently under continuous no till corn; Table 4) but these results cannot be verified due to lack of observations. Intriguing is the finding that simulated sediment C yield in W188 was 59% larger than in W128 (Table 4). These two simulations are identical except for differences in management. The main reason for these results may be that, in W128, low runoff during meadow years may have offset high runoff during conventional till corn years.

For each simulation, the last simulation year was chosen so as to be able to compare the simulated soil C stocks in the top 30 cm with those reported by Hao et al. (2002) for W118 and by Puget et al. (2005) for W128 and W188 (Table 4). EPIC captured the distribution of soil C stocks with depth as well as the soil C stocks in the top 30 cm observed in watersheds W128 and W188 (Table 4). The exception was W118 where EPIC overpredicted the soil C stock by 21%.

The simulated C balance of the three watersheds presented in Table 5 provides an opportunity to examine the major processes controlling the C balance in these managed ecosystems. Soil C stocks to a ~1.5-m depth increased by 28% in W118 and by 15% in W188 but decreased by 10% in W128 (Table 5). Watershed 118 received the largest additions of plant and manure C albeit over a period 13-year longer than that of the other two watersheds. Soil respiration accounted for ≥96% of the simulated C losses from the watersheds. Simulated losses of sediment C and soluble C in runoff were small but nevertheless significant in the three watersheds. This is not surprising since observed and simulated sediment yields in all three watersheds were relatively small (<2 $Mg\ ha^{-1}\ year^{-1}$)

Table 5 Simulated C balance of three watersheds as affected by topographic, soil, and management conditions

	W118 – 49 years	W128 – 36 years	W188 – 36 years
	Soil C and total C added or subtracted during period (Mg C ha^{-1})		
Initial soil C	71.7	94.7	94.7
Added C	290.6	211.5	212.5
Plant C	279.4	209.1	211.2
Manure C	11.2	2.5	1.2
Subtracted C	270.5	220.7	198.3
Respired C	265.5	215.3	190.8
Sediment C in runoff	1.9	3.0	4.7
Soluble C in runoff	1.7	1.7	1.7
Leached C	1.3	0.8	1.1
Final soil C	91.8	85.3	108.8
	Annual C fluxes (kg C ha^{-1} $year^{-1}$)		
Plant C	5,702	5,807	5,868
Manure C	228	69	34
Respired C	5,419	5,979	5,300
Sediment C in runoff	39	82	131
Soluble C in runoff	34	46	48
Leached C	27	22	31
	Net ecosystem carbon balance[b] (Mg C ha^{-1})		
Sediment excluded[a]	20.1	−9.4	14.1
Sediment included[b]			
0.0	23.7	−4.7	20.5
0.2	23.0	−5.6	19.2
1.0	20.1	−9.4	14.1

[a] Net ecosystem carbon balance (NECB) is calculated as the difference in soil C storage between the final and initial year. For W118, NECB = 91.8−71.7 = 20.1 Mg C ha^{-1}.

[b] Net ecosystem carbon balance is calculated as the difference in soil C storage between the final and initial year plus one minus the fraction of sediment and soluble C in runoff that is assumed to evolve as CO_2. This fraction (f) has been estimated as 0.0 (Smith et al. 2001; Stallard 1998), 0.2 (Lal 1995, 2003), or 1.0 (Schlesinger 1995). For W118 at $f=0.0$, NECB = 20.1 + (1.0 − 0.0)(1.9 + 1.7) = 23.7 Mg C ha^{-1}; for W118 at $f=0.2$, NECB = 20.1 + (1.0 − 0.2)(1.9 + 1.7) = 23.0 Mg C ha^{-1}; and for W118 at $f=1.0$, NECB = 20.1 + (1.0 − 1.0)(1.9 + 1.7) = 20.1 MgC ha^{-1}.

compared to other values reported in the literature. An increase, not necessarily proportional, in simulated C losses with runoff would be expected under conditions of increased erosion.

On an annual basis, simulated C losses through soil respiration were lower in W188 and W118 than in W128 (Table 5). This is supported by the relatively similar amounts of annual additions of plant C to soil. An extended period without soil disturbance (i.e., no tillage) in W188 allowed for an enhanced stabilization of the added C within the various soil C pools. Conversely, tillage operations in W128 may have exacerbated the loss of C through the breakdown of soil aggregates and enhanced C mineralization (Izaurralde et al. 2006). Runoff transport of solid and soluble C outside watershed boundaries was relatively small (73–179 kg C ha^{-1} $year^{-1}$) compared to soil respiration losses (Table 5). However, these rates are comparable to the simulated soil C sequestration rates calculated from the difference between W188 and W128 (225 kg C ha^{-1} $year^{-1}$ for the top 30-cm soil depth and 653 kg C ha^{-1} year^{-1} for the entire soil profile depth). The magnitude of C transported in runoff is similar to average soil C sequestration rates under no till reported by West and Post (2002) on a global basis (570 kg C ha^{-1} year^{-1}) and by Franzluebbers (2005) for the southeastern USA (420 kg C ha^{-1} year^{-1}). This strongly suggests that the amount of C removed from agricultural fields by erosion compares in magnitude to C sequestered through the application of no till practices. It also emphasizes the need to include erosion and deposition processes when calculating rates of soil C sequestration and, more generally, in C cycle studies.

The MUSS equation simulates sediment yield by implicit estimation of the amount of eroded soil that is re-deposited within the watershed boundary. We calculated the amount of soil C deposition within the watershed boundary by multiplying the difference between the USLE estimate of soil erosion and the MUSS estimate of sediment yield times the C concentration of sediment C yield. For W118, the calculation, on an annual basis, is:

$$C_{\mathrm{DWWB}} = (\mathrm{USLE}_{\mathrm{Erosion}} - \mathrm{MUSS}_{\mathrm{SedYield}}) \times \frac{C_{\mathrm{SedYield}}}{\mathrm{MUSS}_{\mathrm{SedYield}}} \tag{4}$$

Where C_{DWWB} (kg C ha^{-1}) is C eroded and deposited within the watershed boundary, $\mathrm{USLE}_{\mathrm{Erosion}}$ (kg ha^{-1}) is soil erosion estimated with the USLE equation, $\mathrm{MUSS}_{\mathrm{SedYield}}$ (kg ha^{-1}) is sediment yield estimated with the MUSS equation, and C_{SedYield} (kg ha^{-1}). Thus:

$$C_{\mathrm{DWWB}} = (1210 - 860) \times \frac{73}{860} \tag{5}$$

$$C_{\mathrm{DWWB}} = 30 \text{ kg C ha}^{-1} \text{ year}^{-1} \tag{6}$$

For W128, the annual rate of soil C re-deposition within the watershed was estimated at 212 kg C ha^{-1} year^{-1}) while for W188 the estimate was 178 kg C ha^{-1} year^{-1}. Currently, EPIC does not estimate the fraction of the re-deposited C that evolves to the atmosphere as CO_2. As discussed previously, estimates of this fraction range from 0.0 (Smith et al. 2001; Stallard 1998) to ≥ 0.2 (Lal 1995, 2003; Schlesinger 1995). Our findings support a clear need to document and model the transformations and fate of eroded soil C that is deposited either within or outside watershed boundaries. Research progress in this area would, in turn, lead to an improved understanding of the C cycle at the landscape scale and, eventually, to elucidation of the role of erosion and deposition processes as sources or sinks of atmospheric CO_2.

Simulated leached C ranged from 22 to 31 kg C ha^{-1} year^{-1} (Table 5). These values are larger than the 4.5 kg C ha^{-1} year^{-1} (range: 1.5–12.4 kg C ha^{-1} year^{-1}) leached, on average, below 2.4-m depth from a weighing lysimeter cropped to a corn–soybean rotation during 1989–1998 (Owens et al. 2002).

We used the difference in soil C stocks at the end and beginning of the simulation period to estimate the net ecosystem carbon balance (NECB) (Randerson et al. 2002) (Table 5) and thus assess the extent to which each watershed had been a source or a sink of C to the atmosphere. The two watersheds that contained significant periods under no till (W118 and W188) behaved as C sinks while the one that had been under plow till during the last 18 years (W128) behaved as a C source to the atmosphere (Table 5). These results are consistent with those reported by Puget et al. (2005).

We also estimated the influence of sediment C and its fate on NECB (Table 5). We assumed that if none of the sediment C evolves as CO_2 (fraction 0.0 in Table 5), then NECB would be enhanced. Conversely, there would be a reduction in NECB if a fraction or all of sediment C were to evolve as CO_2 to the atmosphere. Relative to non-eroding conditions and depending on the decomposition described above and in Table 5, NECB would have increased or decreased by 18% for W118, by 50% for W128, and by 45% for W188. This demonstrates the importance or erosion and sediment decomposition on NECB.

4 Conclusions

This simulation study linked the interactions of erosion–deposition processes to the C cycle of three small watersheds near Coshocton, OH. A simulated sediment C yield of 443 ± 22 kg C ha^{-1} year^{-1} during 1951–1999 compared well against the 31 ± 12 kg C ha^{-1} year^{-1} observed in W118. These estimates resulted from realistic simulation of runoff and erosion processes. EPIC overestimated the soil C stock in the top 30-cm soil depth in W118 by 21% of the measured value (36.8 Mg C ha^{-1}). Simulations of soil C stocks in the other two watersheds (42.3 Mg C ha^{-1} in W128 and 50.4 Mg C ha^{-1} in W188) were off by <1 Mg C ha^{-1}. Simulated eroded C re-deposited inside (30–212 kg C ha^{-1} year^{-1}) or outside (73–179 kg C ha^{-1} year^{-1}) watershed boundaries compared in magnitude to a simulated soil C sequestration rate of 225 kg C ha^{-1} year^{-1} and to literature values. An analysis of net ecosystem carbon balance revealed that the watershed currently under a plow till system (W128) was a source of C to the atmosphere while the watersheds currently under a no till system (W118 and W188) behaved as C sinks of atmospheric CO_2. Our results demonstrate a clear need for documenting and modeling the proportion of eroded soil C that is transported outside watershed boundaries and the proportion that evolves as CO_2 to the atmosphere. In future work, we will use the APEX model (the landscape version of EPIC) (Williams and Izaurralde 2005) to study the role of erosion and deposition as sources or sinks of atmospheric CO_2 at the large watershed scale.

Acknowledgments Research conducted within the CSiTE (Consortium for Research on Enhanced Carbon Sequestration in Terrestrial Ecosystems) and CASMGS (Consortium for Agricultural Soils Mitigation of Greenhouse GaSes) programs. CSiTE is supported by the US Dept. of Energy, Office of Science, as part of the Terrestrial Carbon Sequestration program in the Office of Biological & Environmental Research. CASMGS is supported by the US Dept. of Agriculture and the US Environmental Protection Agency. We gratefully acknowledge A. Rappaport and three anonymous reviewers for their constructive criticisms and insightful comments.

References

Bryant KJ, Benson VW, Kiniry JR, Williams JR, Lacewell RD (1992) Simulating corn yield response to irrigation timings: Validation of the EPIC model. J Prod Agric 5:237–242

Campbell CA, Zentner RP, Liang BC, Roloff G, Gregorich EC, Blomert B (2000) Organic C accumulation in soil over 30 years in semiarid southwestern Saskatchewan – Effect of crop rotations and fertilizers. Can J Soil Sci 80:179–192

Cavero J, Plant RE, Shennan C, Friedman DB, Williams JR, Kiniry JR, Benson VW (1999) Modeling nitrogen cycling in tomato–safflower and tomato–wheat rotations. Agric Syst 60:123–135

Cole CV, Duxbury J, Freney J, Heinemeyer O, Minami K, Mosier A, Paustian K, Rosenberg N, Sampson N, Sauerbeck D, Zhao Q (1997) Global estimates of potential mitigation of greenhouse gas emissions by agriculture. Nutr Cycl Agroecosyst 49:221–228

Davidson EA, Ackerman IL (1993) Changes in soil carbon following cultivation of previously untilled soils. Biogeochemistry 20:161–164

de Jong E, Kachanoski RG (1988) The importance of erosion in the carbon balance of prairie soils. Can J Soil Sci 68:111–119

Franzluebbers AJ (2005) Soil organic carbon sequestration and agricultural greenhouse gas emissions in the southeastern USA. Soil Tillage Res 83:120–147

Green WH, Ampt GA (1911) Studies in soil physics. I. The flow of air and water through soils. J Agric Sci 4:1–24

Gregorich EG, Greer KJ, Anderson DW, Liang BC (1998) Carbon distribution and losses: erosion and deposition effects. Soil Tillage Res 47:291–302

Hao YL, Lal R, Izaurralde RC, Ritchie JC, Owens LB, Hothem DL (2001) Historic assessment of agricultural impacts on soil organic carbon erosion in an Ohio watershed. Soil Sci 166:116–126

Hao YL, Lal R, Owens LB, Izaurralde RC, Post WM, Hothem DL (2002) Effect of cropland management and slope position on soil organic carbon pool at the North Appalachian Experimental Watersheds. Soil Tillage Res 68:133–142

Harden JW, Sharpe JM, Parton WJ, Ojima DS, Fries TL, Huntington TG, Dabney SM (1999) Dynamic replacement and loss of soil carbon on eroding cropland. Glob Biogeochem Cycles 13:885–901

Izaurralde RC, Nyborg M, Solberg ED, Janzen HH, Arshad MA, Malhi SS, Molina-Ayala M (1998) Carbon storage in eroded soils after five years of reclamation techniques. In: Lal R, Kimble J, Follett R, Stewart BA (eds) Soil processes and the carbon cycle. CRC Press, Boca Raton, Florida, pp 369–386

Izaurralde RC, McGill WB, Robertson JA, Juma NG, Thurston JJ (2001) Carbon balance of the Breton classical plots over half a century. Soil Sci Soc Am J 65:431–441

Izaurralde RC, Williams JR, McGill WB, Rosenberg NJ, Quiroga Jakas MC (2006) Simulating soil C dynamics with EPIC: model description and testing against long-term data. Ecol Model 192:362–384

Jastrow JD, Miller RM (1998) Soil aggregate stabilization and carbon sequestration: feedbacks through organo–mineral associations. In: Lal R, Kimble J, Follett R, Stewart BA (eds) Soil processes and the carbon cycle. CRC Press, Boca Raton, FL, pp 205–224

Jones CA, Bland WL, Ritchie JT, Williams JR (1991) Simulation of root growth. In: Hanks J, Ritchie T (eds) Modeling plant and soil systems, Agronomy no. 31. ASA, CSSA, SSSA Publ., Madison, WI, pp 91–123

Kelley GE, Edwards WM, Harold LL, McGuinness JL (1975) Soils of the North Appalachian Experimental Watershed. USDA-ARS Misc. Public. 1296. U.S. Gov. Print. Office, Washington, District of Columbia

Kiniry JR, Williams JR, Gassman PW, Debaeke P (1992) A general, process-oriented model for 2 competing plant-species. Trans ASAE 35:801–810

Lal R (1995) Global soil erosion by water and carbon dynamics. In: Lal R, Kimble JM, Levine E, Stewart BA (eds) Soils and global change. CRC Press, Boca Raton, FL, pp 131–142

Lal R (2003) Soil erosion and the global carbon budget. Environ Int 29:437–450

Liu SG, Bliss N, Sundquist E, Huntington TG (2003) Modeling carbon dynamics in vegetation and soil under the impact of soil erosion and deposition. Glob Biogeochem Cycles 17:1074 doi:10.1029/2002GB002010

McCarty GW, Ritchie JC (2002) Impact of soil movement on carbon sequestration in agricultural ecosystems. Environ Pollut 116:423–430

Monreal CM, Zentner RP, Robertson JA (1997) An analysis of soil organic matter dynamics in relation to management, erosion and yield of wheat in long-term crop rotation plots. Can J Soil Sci 77:553–563

Owens LB, Starr GC, Lightell DL (2002) Total organic carbon losses in subsurface flow under two management practices. J Soil Water Conserv 57:74–81

Parton WJ, Schimel DS, Cole CV, Ojima DS (1987) Analysis of factors controlling soil organic matter levels in Great Plains grasslands. Soil Sci Soc Am J 51:1173–1179

Parton WJ, Scurlock JMO, Ojima DS, Gilmanov TG, Scholes RJ, Schimel DS, Kirchner T, Menaut J-C, Seastedt T, Garcia Moya E, Kamnalrut A, Kinyamario JI (1993) Observations and modelling of biomass and soil organic matter dynamics for the grassland biome worldwide. Glob Biogeochem Cycles 7:785–809

Parton WJ, Ojima DS, Cole CV, Schimel DS (1994) A general model for soil organic matter dynamics: sensitivity to litter chemistry, texture and management. In: Quantitative modeling of soil forming processes. Soil Science Society of America Journal Special Publication No. 39, Madison, Wisconsin, pp 147–167

Paul EA, Paustian K, Elliott ET, Cole CV (eds) (1997) Soil organic matter in temperate agroecosystems: long-term experiments in North America. Lewis Publishers, CRC Press, Boca Raton, Florida

Potter KN, Williams JR, Larney FJ, Bullock MS (1998) Evaluation of EPIC's wind erosion submodel using data from Southern Alberta. Can J Soil Sci 78:485–492

Powlson DS, Smith P, Smith JU (eds) (1996) Evaluation of soil organic matter models using existing long-term datasets NATO ASI Series I, Vol. 38. Springer, Berlin, Heidelberg, New York
Puget P, Lal R, Izaurralde C, Post M, Owens L (2005) Stock and distribution of total and corn-derived soil organic carbon in aggregate and primary particle fractions for different land use and soil management practices. Soil Sci 170:256–279
Randerson JT, Chapin FS, Harden JW, Neff JC, Harmon ME (2002) Net ecosystem production: a comprehensive measure of net carbon accumulation by ecosystems. Ecol Appl 12:937–947
Roloff G, de Jong R, Zentner RP, Campbell CA, Benson VW (1998) Estimating spring wheat yield variability with EPIC. Can J Plant Sci 78:541–549
Schlesinger WH (1995) Soil respiration and changes in soil carbon stocks. In: Woodwell GM, Mackenzie GM (eds) Biotic feedbacks in the global climatic system: will the warming feed the warming? Oxford Univ. Press, New York, pp 159–168
Smith SV, Renwick WH, Buddemeier RW, Crossland CJ (2001) Budgets of soil erosion and deposition for sediments and sedimentary organic carbon across the conterminous United States. Glob Biogeochem Cycles 15:697–707
Stallard RF (1998) Terrestrial sedimentation and the carbon cycle: coupling weathering and erosion to carbon burial. Glob Biogeochem Cycles 12:231–257
Stockle CO, Williams JR, Rosenberg NJ, Jones CA (1992a) A method for estimating direct and climatic effects of rising atmospheric carbon dioxide on growth and yield of crops: Part I – Modifications of the EPIC model for climate change analysis. Agric Syst 38:225–238
Stockle CO, Dyke PT, Williams JR, Jones CA, Rosenberg NJ (1992b) A method for estimating direct and climatic effects of rising atmospheric carbon dioxide on growth and yield of crops: Part II – Sensitivity analysis at three sites in the Midwestern USA. Agric Syst 38:239–256
Touré A, Major DJ, Lindwall CW (1994) Comparison of five wheat simulation models in Southern Alberta. Can J Plant Sci 75:61–68
Troeh FR, Hobbs JA, Donahue RL (1991) Soil and water conservation, 2nd edn. Prentice Hall, Englewood Cliffs, NJ, pp 530
US Department of Agriculture (USDA)-Soil Conservation Service (1972) National Engineering Handbook. Hydrology Section 4, Chapters 4–10. Washington, District of Columbia
van Oost K, Govers G, Quine TA, Heckrath G (2004) Comment on “managing soil carbon” (I). Science 305:1567b
Warner GS, Stake JD, Guillard K, Neafsey J (1997) Evaluation of EPIC for a shallow New England soil: I. Maize yield and nitrogen uptake. Trans ASAE 40:575–583
West TO, Post WM (2002) Soil organic carbon sequestration rates by tillage and crop rotation: a global data analysis. Soil Sci Soc Am J 66:1930–1946
Williams JR (1975) Sediment-yield prediction with universal equation using runoff energy factor. In: Present and prospective technology for predicting sediment yield and sources, USDA-ARS.S-40, US Gov. Print. Office, Washington, District of Columbia, pp 244–252
Williams JR, Jones CA, Kiniry JR, Spanel DA (1989) The EPIC crop growth-model. Trans ASAE 32:497–511
Williams JR (1995) The EPIC model. In: Singh VP (ed) Computer models of watershed hydrology. Water Resources Publications, Highlands Ranch, CO, pp 909–1000
Williams JR, Izaurralde RC (2005) The APEX model. In: Singh VP, Frevert DK (eds) Watershed models. CRC Press, Boca Raton, FL
Wischmeier WC, Smith DD (1965) Predicting rainfall erosion losses from cropland east of the Rocky Mountains, Agric. Handbook No. 282. US Gov. Printing Office, Washington, District of Columbia
Wischmeier WC, Smith DD (1978) Predicting rainfall erosion losses – A guide to conservation planning, Agric. Handbook No. 537. US Gov. Printing Office, Washington, District of Columbia