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MODELING SOIL CARBON TRANSPORTED BY WATER EROSION PROCESSES

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ABSTRACT

Long-term monitoring is needed for direct assessment of soil organic carbon (SOC), soil, and nutrient loss by water erosion on a watershed scale. However, labor and capital requirements preclude implementation of such monitoring at many locations representing principal soils and ecoregions. These considerations warrant the development of diagnostic models to assess erosional SOC loss from more readily obtained data. The same factors affect transport of SOC and mineral soil fraction, suggesting that given the gain or loss of soil minerals, it may be possible to estimate the SOC flux from the data on erosion and deposition. One possible approach to parameterization is the use of the revised universal soil loss equation (RUSLE) to predict soil loss and this multiplied by the per cent of SOC in the near-surface soil and an enrichment factor to obtain SOC loss. The data obtained from two watersheds in Ohio indicate that a power law relationship between soil loss and SOC loss may be more appropriate. When measured SOC loss from individual events over a 12-year period was plotted against measured soil loss the data were logarithmically linear ($R^2 = 0{\cdot}75$) with a slope (or exponent in the power law) slightly less than would be expected for a RUSLE type model. The stable aggregate size distribution in runoff from a plot scale may be used to estimate the fate of size pools of SOC by comparing size distributions in the runoff plot scale and river watershed scales. Based upon this comparison, a minimum of 73 per cent of material from runoff plots is deposited on the landscape and the most stable carbon pool is lost from watershed soils to aquatic ecosystems and atmospheric carbon dioxide. Implicit in these models is the supposition that water stable soil aggregates and primary particles can be viewed as a tracer for SOC.

KEY WORDS: RUSLE; soil organic carbon; water erosion; diagnostic modeling; calibration data; soil loss; greenhouse effect

INTRODUCTION

The soil surface is subjected to vast inputs of energy from rainfall, runoff, wind, and solar radiation as well as a wide range of human and biotic inputs. Some of this energy is intercepted and absorbed by plants that use solar energy, soil nutrients, and atmospheric carbon for photosynthesis. By contrast with plants, soil is incapable of absorbing the large energy fluxes in a constructive manner and when the soil surface is exposed, the results can be highly degrading. Particularly in the case of energy from rainfall and runoff causing water erosion, the operative processes are destructive, both to soil structure, and to its capacity to sustain biomass growth (Lal, 1998). Rainfall impact and the shearing forces of runoff disintegrate soil aggregates (Yoder, 1936; Le Bissonnais and Arrouays, 1997) and transport fertile topsoil along with plant nutrients and organic matter away from eroded soil landscapes (Rogers, 1941; Massey and Jackson, 1952; Lal, 1980; Zobish, *et al.*, 1995). Some of the soil, nutrients, and organic carbon are redistributed across the landscape and some are transferred to aquatic ecosystems (Lal, 1995; Stallard, 1998), where they contribute to eutrophication (Vezjak, *et al.*, 1998; Frielinghaus and Vahrson, 1998), anoxia (Vezjak, *et al.*, 1998), turbidity (Wass, *et al.*,

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1997; Riley, 1998), greenhouse gas emissions (Lal, 1995), and general water quality degradation before a part of them are eventually stored in sediments.

The role of soil organic carbon (SOC) in stabilizing aggregates (Tisdall and Oades, 1982; Elliott, 1986; Kay, 1998) and thereby reducing the susceptibility to erosion (Yoder, 1936; Piccolo, *et al*., 1997) is fairly well established. Improved aggregate stability is but one of the many positive aspects of sequestering carbon in soil that also include reduced greenhouse gasses in the atmosphere and improved soil quality (Lal, 1997). These qualities have led to serious consideration of managing arable lands for soil carbon sequestration (Sharpenseel, 1997; Lal, *et al*., 1998a, b). Given the intense interest of both scientists and policy makers in SOC sequestration, it is important to improve the cursory and qualitative understanding of the principal factors and processes that affect its rate and magnitude within soil and terrestrial/aquatic ecosystems. An example of such a process with poorly understood implications for the pools and fluxes of carbon is accelerated soil erosion.

The impact of erosion on net primary productivity can lead to serious reduction in agronomic yields, particularly in severely eroded soils that are not adequately fertilized (Lal, 1998). The reduction in living plant biomass that accompanies accelerated soil erosion is implicated as a causative factor in reduced soil C levels through reduced inputs of litter and root biomass to the SOC pools (Gregorich, *et al*., 1998). Although SOC moves along with soil particles and is directly transported away from eroded landscapes, it is difficult to assess both the direct SOC flux magnitudes and the residual or indirect effects of these fluxes on SOC pools. It has been estimated (Lal, 1995) that 5·7 Pg of the world's SOC is translocated by water erosion annually where the majority (perhaps 90 per cent) of the translocated SOC is deposited across the landscape. The deposited SOC may lose some of its physical protection and become exposed to advanced oxidative processes such that some of it goes through stages of accelerated decomposition. This, hypothetically, may lead to erosion induced release of carbon dioxide from the translocated SOC pool to the atmosphere at an estimated 20 per cent greater annual rate than if the erosion had not occurred (Lal, 1995).

Stallard (1998) estimated that enough SOC is deposited annually in aquatic sediments to account for a significant portion of the 'missing sink' (1·5–2·0 Pg C/yr) of atmospheric carbon dioxide. The hypothesis (Stallard, 1998) that SOC transferred to sediments account for a massive atmospheric carbon dioxide sink assumes that the sediment-borne SOC is replaced in the landscape from atmospheric carbon dioxide. In contrast to Stallard's hypothesis, a study of the long-term cultivation effects on a Canadian grassland showed that the decline in SOC concentrations following cultivation did not level off and reach equilibrium, but continued to decline for 60–90 years which was attributed to erosion (Tiessen, *et al*., 1982). This long-term decline in SOC contents caused by erosion was predicted by Voroney, *et al*. (1981) in their model of erosion and SOC and may not be easy to reverse. In a Canadian study, Geng and Coote (1991) showed that soil loss by erosion caused reductions in SOC, N, P, and aggregate stability that were not replenished even after 17 years of continuous grass cover in a coarse- and medium-textured soil. However, in the same study but a different site and time-sequence, the SOC level in previously eroded fine-textured soil was restored after 27 years of continuous grass cover.

Runoff from eroding landscapes is enriched in clay sized particles (Ongley, *et al*., 1981; Pert and Walling, 1982), particulate organic carbon (POC), and dissolved organic carbon (DOC) (Lal, 1995). Suspended sediments in rivers will, in majority, be less than 0·062 mm diameter (Pert and Walling, 1981). Ongley, *et al*. (1981) found that 60–90 per cent of suspended sediments were $<0{\cdot}002$ mm in size in Wilton Creek, Ontario. The smallest size SOC fraction (less than about 0·005 mm) is thought to be the most stable size fraction in soil (Anderson, *et al*., 1981; Paul, *et al*., 1995) because of the physical protection afforded in soil aggregates (Kay, 1998) and the recalcitrance of humic materials (Paul, *et al*., 1995). Physical protection in aggregates is undermined by processes such as tillage and water erosion (Lal, 1997). Erosion is one of the only soil processes that can remove stable SOC in large quantities so its effects may be dramatic. The data presented by Mitchell, *et al*. (1998) on modeling carbon storage in soil showed that erosion by water is the most significant factor affecting the SOC balance in the north central USA with erosion by wind being the second most significant factor.

Little is known of the properties and fate of SOC that is translocated from erosion to deposition points on the landscape. The physics of particle settling would tend to suggest that the extent of erosional translocation is negatively correlated with the size and density of detached soil aggregates and primary particles. Thus, soil erosion is a highly selective process that preferentially removes the smallest and lowest density components of soil and transports them great distances (Lal, 1995).

It is important to develop diagnostic models that will improve our understanding of the underlying mechanisms and processes affecting erosion induced losses of SOC. Available data on erosion and deposition of SOC are very limited, so our approaches to diagnostic modeling are based on the information on the transport of soil and drawing inferences concerning the closely related transport of SOC.

Long-term monitoring for direct assessment of SOC, soil, and nutrient loss by water erosion on a watershed scale is time consuming and expensive. On large watershed or continental scales, determination of SOC loss by water erosion or gain by deposition is fraught with uncertainties. These considerations warrant the development of diagnostic models to assess erosional SOC loss from more readily obtained data. The objective of this study was to identify and describe approaches for assessing erosional impacts on SOC dynamics using diagnostic models. Hypothetically, because the same factors affect transport of both SOC and mineral soil fraction, it may be possible to estimate the SOC flux from the data on soil erosion and deposition. Implicit in this hypothesis is the supposition that water-stable soil aggregates and primary particles can be viewed as a tracer for SOC of an equivalent settling velocity.

MODELING STRATEGIES

This paper involves three approaches to predicting SOC dynamics in relation to erosional processes.

Parameterization

One possible approach to parameterization is the use of the revised universal soil loss equation (RUSLE) (Renard, *et al.*, 1997) or some other model or measurement (e.g. ^{137}CS method, depth to benchmark horizons, direct determination from runoff) to estimate soil loss. Soil loss can be multiplied by the per cent of SOC in the near-surface soil and an enrichment ratio (ER) to obtain SOC loss [Equation (1)].

$$\text{SOC loss} = (\text{soil loss})(\text{SOC content})(\text{ER}) \tag{1}$$

As an example, total SOC loss is plotted (Figure 1) against soil loss (Starr, Lal, Owens, Hothem, Malone and Kimble, unpublished manuscript method) for two conservation tillage watersheds (Kelly, *et al.*, 1975) at the North Appalachian Experimental Watersheds in Coshocton, OH [no-till and chisel till corn (*Zea mays*)/ soybean (*Glycine max*) rotation]. The SOC content was calculated by averaging per cent SOC measurements from the top 2·5 cm of soil at several locations in the watershed (identifiable plant residues not included) before and after the 12-year period (1984–96) of runoff sampling. A value of 2·1 for ER was calculated from average values of SOC and soil loss yielding a reasonably good fit (solid line in Figure 1) between model and measurement over much of the range of plotted data. However, the parametric model tends to underestimate SOC loss in small events and overestimate them in the critical large events.

The linear Equation (1) has the same coefficients (SOC content and ER) whether it is applied for an average of many events or for individual events and this may be a limitation because both coefficients may change with time, soil, and rainfall conditions (Massey and Jackson, 1952; Lal, 1975). The relationship between soil loss and ER has been observed to be logarithmically linear with ER inversely proportional to soil loss (Massey and Jackson, 1952). Inclusion of a logarithmic function to express ER causes the modeling to be empirical rather than strictly parametric as discussed in the next section.

Empirical Models

The data obtained from two watersheds in Ohio suggest a power law relationship between soil loss and SOC loss (Figure 1). Measured SOC loss in individual events over a 12-year period was plotted against measured

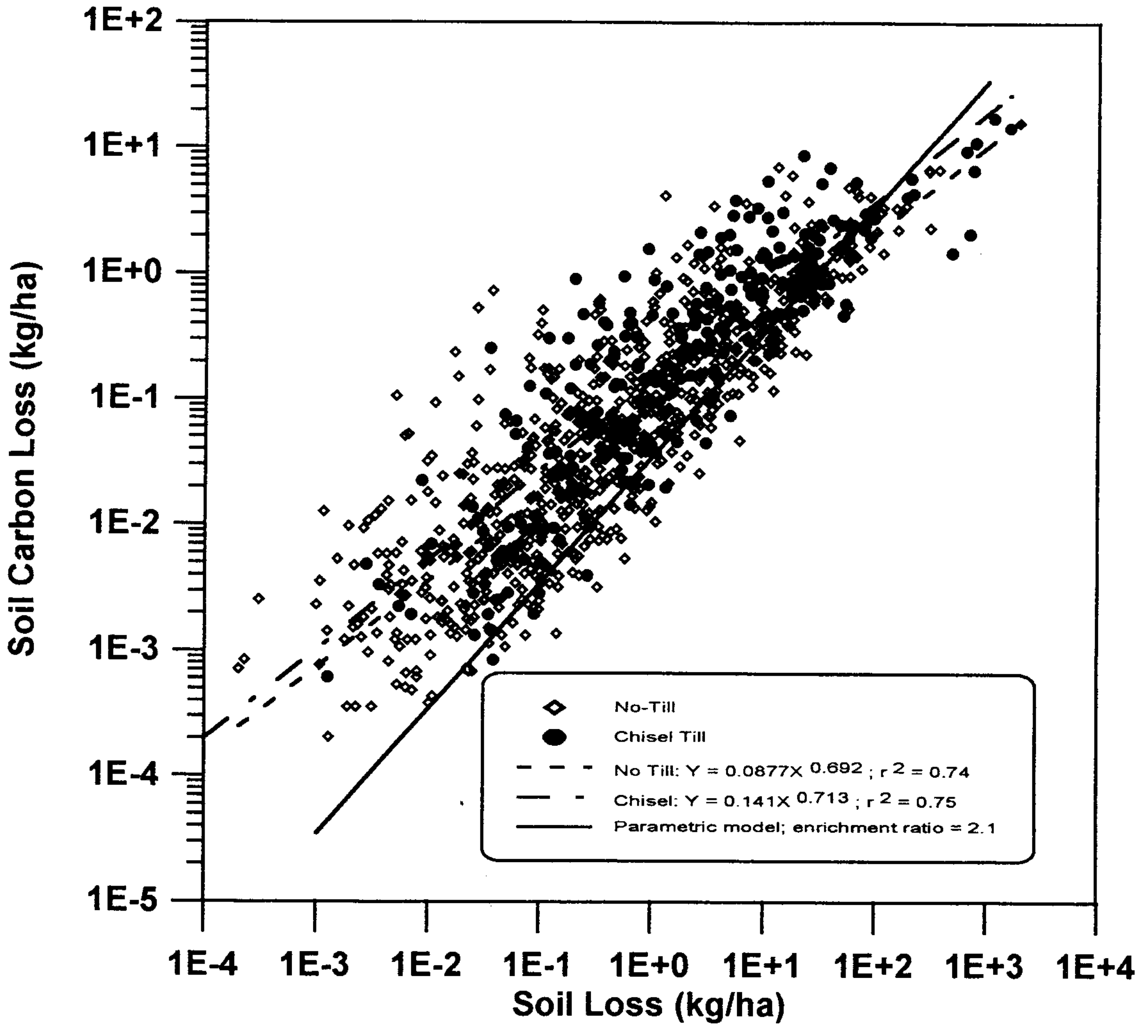

Figure 1. Soil and carbon loss in watershed runoff events.

soil loss and the data were logarithmically linear ($R^2 = 0{\cdot}75$). The slope (or exponent in the power law) was considerably less than would be expected for a constant ER parametric model (compare solid and dashed lines). The empirical model fits the observed data from individual events better than the linear parametric model, particularly in the region of the graph showing critical large events. Because the empirical relationship is a power law, it is not as easy to draw broader inferences from the results as with the linear parametric model. For instance, calculation of cumulative C losses from data on cumulative soil losses is not trivial. The general logarithmic linearity seen in these data are, essentially, the same functional forms as the parametric model discussed above with a log-linear relation between enrichment ratio and soil loss remarkably similar to what has been observed by previous researchers (Massey and Jackson, 1952). The proposed empirical relationship is of the form:

$$\text{SOC loss} = a(\text{soil loss})^b \tag{2}$$

where a and b are statistical constants. This empirical formulation does not require an explicit expression for ER, an advantage for watershed scales where source materials of different SOC contents contribute to a

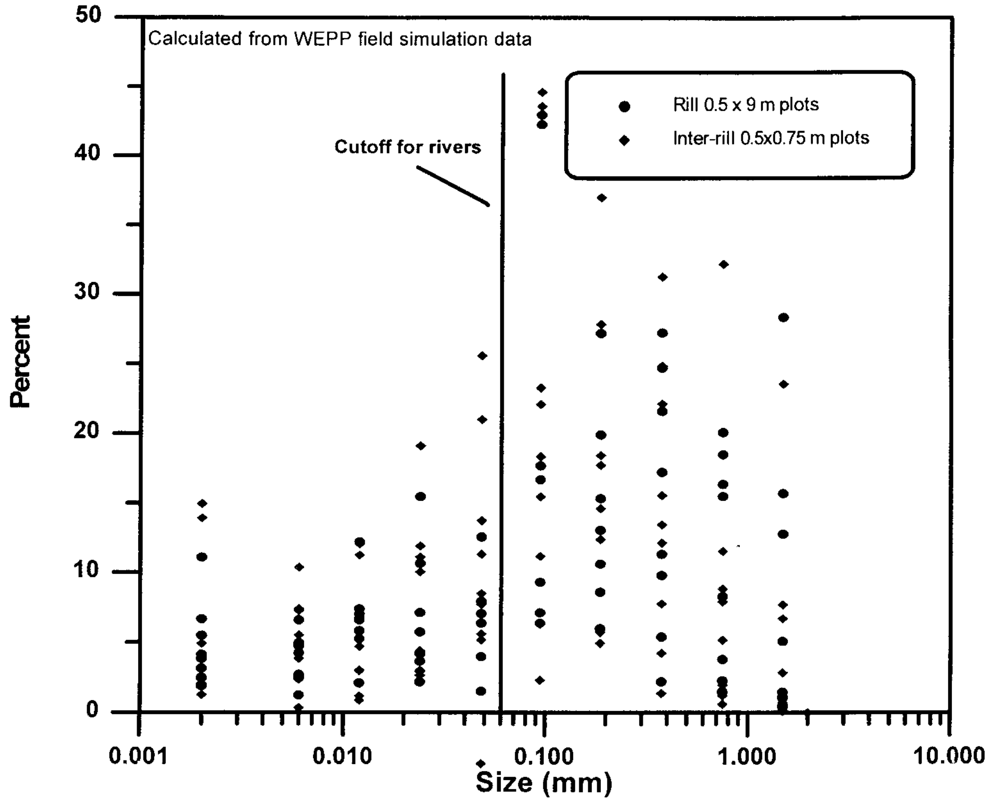

Figure 2. Aggregate sizes in runoff from plots.

composite mixture in runoff. A recent study (Collins, *et al.*, 1997) showed the complexity of relating eroded sediment to source material in intermediate sized watersheds (10 km scale). In this study ER ranged from 0·5 to greater than 5 depending on stream bank, forest, pasture, or cultivated soil source material.

This empirical model only predicts C losses from these watersheds for events with given soil loss. The two watersheds do, however, exhibit nearly identical equations and it would be an interesting course of future research to add more data to this graph from different areas and different scales in an effort to understand the underlying processes affecting the relationship between soil and SOC loss.

Size Distribution Comparison

The stable aggregate size distribution in runoff (data calculated from Elliott, *et al.*, 1989) on a plot scale (Figure 2) may be used to estimate the fate of size pools of SOC by comparing size distributions in the runoff plot scale and river watershed scales. Using a cutoff of 0·062 mm for transport to rivers (Pert and Walling, 1982), it is possible to draw conceptual inferences concerning the fate of material running off the plot scale that is redistributed across the landscape. The spline-smoothed curve depicting the percentage of total soil carbon loss as a function of aggregate size (assuming the same carbon content on a mass basis is present in various aggregate sizes) from WEPP runoff plots (Elliott, *et al.*, 1989) (Figure 3) has been divided into three regions. About 73 per cent of material from runoff plots is contained in aggregates greater than 0·062 mm and is likely to be deposited on the landscape. There is an intermediate range of primarily silt-sized particles that may be deposited or transferred to rivers and if the lower size boundary for this range is taken, arbitrarily, an order of magnitude lower than the cutoff for rivers, then the intermediate range constitutes 19 per cent of the

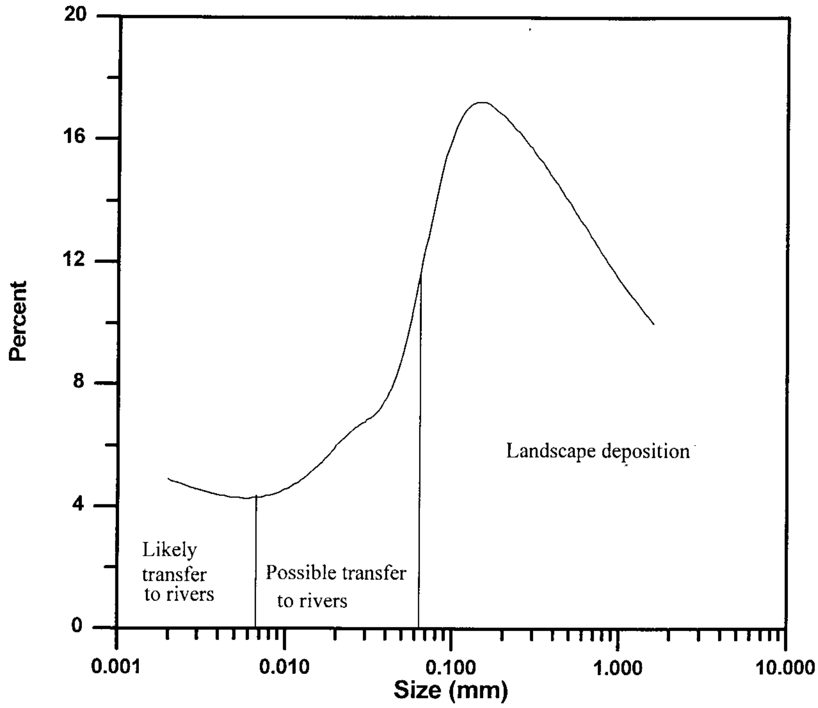

Figure 3. Fate of transported soil carbon as a function of stable aggregate size.

total. The most stable carbon pool (fine silt and clay domain), representing only 8 per cent of the total, has the highest likelihood to be lost from watershed soils to aquatic ecosystems.

Thus, the largest aggregates are only transferred a short distance in a given rainfall event. Large aggregates, deposited on the soil surface, will be exposed to repeated rainfall events, solar radiation, extreme variation in temperature, dramatic variation in water content, crushing forces from farm equipment, and other destructive processes that may lead to more advanced aggregate disruption than occurs in a single rainfall event. The combined influence of destructive surface processes may cause the physical protection afforded SOC in aggregates to be undermined and lead to increased emissions of carbon dioxide. A previous model (Van Veen and Paul, 1981; Varoney, *et al.*, 1981) considered decomposition rates two orders of magnitude lower for physically protected SOC than for exposed SOC. Therefore, the hypothesis of Lal (1995) (20 per cent erosion-induced annual loss of translocated SOC to atmospheric carbon dioxide) may be an underestimate. The smaller aggregates are transferred a greater distance and there is more opportunity for dispersion and further breakdown within the runoff waters and aquatic ecosystems. The smaller aggregates lost from a watershed scale will, hypothetically, approach a state of dispersion similar to maximum water dispersion, a state that is somewhat less dispersed than chemical or ultrasonic treatments, and thereby lose much of the physical protection afforded in soil.

DISCUSSION

When they are available, data from long-term studies on experimental watersheds representing principal soils and ecoregions are invaluable for assessing the erosion-induced losses of soil and SOC. However, labor and capital requirements preclude implementation of such monitoring at numerous locations. These considerations warrant the development of diagnostic models to assess erosional SOC losses using available information.

Table I. Data inputs and applicable scales of three modeling strategies

Factors	Parametric	Empirical	Size Distribution
Data Input	Soil loss, SOC percent, enrichment ratio	Soil loss, calibration data	Size distribution in runoff on comparative scales
Scale	Continental, large watershed, small watershed, plot	Small watershed, plot	Continental, large watershed, small watershed, plot

Table II. Pros, cons, and relevant references for various modeling strategies

Modeling strategy	Pros	Cons	Relevant references
Size distribution	• Broad applicability • SOC pools assessed • Range of scales • Conceptual inferences	• Data acquisition • Assumptions • Indirect impacts not treated	Tollner and Hayes, 1986 Yoder, 1936
Parametric	• Simple extension of soil loss models • May work with limited data sets • Easily generalized across a range of scales	• Accuracy • Assumptions for estimating input data • Range of source material may confound the results • Indirect impacts not treated	Mitchell, *et al*., 1998 Collins, *et al*., 1997 Lal, 1995 Varoney, *et al*., 1981 Massey and Jackson, 1952
Empirical	• Good for understanding specific watershed losses • Accurate once calibrated	• Need for calibration data • Not easily generalized to other locations • Requires single event input data • Indirect impacts not treated	Collins, *et al*., 1997 Massey and Jackson, 1952

The potential applicability of three modeling strategies identified in this study (parametric, empirical, and size distribution) can be summarized by their scales and input data (Table I). The parametric approach requires an estimate of soil loss, SOC content and enrichment ratio in the near-surface horizon. Enrichment ratio is frequently assumed to be one (Varoney, *et al*., 1981; Mitchell, *et al*., 1998) in plot scale studies but this could be an underestimate (Massey and Jackson, 1952). On larger scales, both enrichment ratio and near-surface SOC content must be estimated as a composite parameter because of the range of source materials contributing to runoff losses of soil (Collins, *et al*., 1997). The empirical modeling requires soil loss data in single events and calibration data. It is applicable to small watershed and plot scales. Size distribution comparisons require a measurement of aggregates and primary particles in runoff on comparative scales. The potential for studying size distributions on various scales ranging from continental to plot gives direction to future research.

Each of the modeling strategies has associated advantages and disadvantages (Table II) but these strengths and weaknesses generally complement one another. One disadvantage common to all these models is that they do not assess indirect effects such as reduction in SOC inputs because of crop productivity decline (Gregorich, *et al*., 1998) and increased carbon dioxide losses of translocated SOC (Lal, 1995). The size distribution strategy has broad applicability across a range of scales, is able to assess the relative impacts on specific size pools, and the conceptual inferences drawn from this model are useful for understanding the erosion and deposition processes with regard to SOC transformation and translocation. It is, however, laborious to obtain runoff size distribution data (Tollner and Hayes, 1986) and the assumption that aggregates remain stable as they are transported and deposited between scales may not be strictly true. The parametric modeling strategy represents a simple extension of soil loss models (Varoney, *et al*., 1981; Lal, 1995; Mitchell, *et al*., 1998); it may work with limited data sets and can easily be generalized across a range of scales and time spans. However, the accuracy of the parametric approach may be called into question along with assumptions for estimating input data when a range of source material are contributing to a composite runoff sample as occurs on larger scales (Collins, *et al.*, 1997).

Empirical models are helpful for understanding the relationships between soil and SOC losses in specific watersheds and statistical estimates of accuracy may be established once the models are calibrated. However, there is a need for long-term monitoring to obtain calibration data, the derived relationships only apply to specific watersheds, and this approach requires input data for specific runoff events.

FUTURE DIRECTIONS

(1) The scale of parametric model needs to be expanded to large watersheds and continents.
(2) More calibration data from other watersheds/areas need to be added for the empirical model.
(3) A more detailed analysis of size distribution including specific size distributions in nested watersheds needs to be undertaken and the distribution and density of SOC in aggregate sizes needs to be added.

CONCLUSIONS

The following inferences may be drawn from the data and literature discussed in this paper:

(1) Diagnostic models have been described that can be used to estimate direct erosion impacts on SOC balance from the closely related erosion induced transport of soil minerals.
(2) Each model has a different applicability, but they are all based on the supposition that soil minerals can be used as a tracer for SOC transport by erosion.
(3) The parametric approach has the widest range of applicability across many scales because of the simplicity and linearity of this approach.
(4) Empirical models have more potential for explaining processes occurring in specific watersheds or plots and may be more accurate once calibrated.
(5) The size distribution comparison has broad applicability and is helpful in understanding the processes, mechanics, and specific size pool dynamics.
(6) Drawbacks of the modeling strategies include the lack of information on indirect impacts, and varying degrees of difficulty with regard to input data and assumptions.
(7) All approaches warrant further study.

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