

# Conserving Soil and Water to Sequester Carbon and Mitigate Global Warming

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Soil degradation by erosion strongly perturbs the global carbon (C) cycle (Lal 2003) and reduces crop yield and productivity (Gomiero 2016). Thus, a better understanding of the processes governing the distribution and fate of soil organic C (SOC) transported by erosional processes is needed to credibly and objectively assess the impact of accelerated erosion on the C budget at local, regional, and global scales. Water erosion is a selective process involving a preferential removal of clay and SOC contents (Shi and Schulin 2018; Billings et al. 2019). Therefore, the impact of erosion on redistribution, mineralization, and burial of SOC at the global scale must be evaluated to credibly assess its impacts as a source or sink of atmospheric carbon dioxide (CO<sub>2</sub>).

Soil erosion by water is a four-stage process: detachment, transport, redistribution, and deposition (Lal et al. 2004). Particle detachment is caused by the shearing force of the impacting raindrops, shallow overland flow, and interaction between them. Detached particles, along with SOC and particulate organic C (POC), are transported and redistributed over the landscape. Depending on the micro and macro-relief and the slope gradient, deposition of the sediment occurs following the Stokes' Law. The magnitude of soil erosion depends on climate erosivity, soil erodibility, slope (gradient, length, and shape), management (tillage, crop, and cropping system), and conservation practice (terraces, etc.) (Wischmeier and Smith 1978). Decline in SOC content

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can increase water runoff and erosion (Fullen 1991), partly due to an increase in soil dispersion (Watts and Dexter 1998) and an attendant increase in erodibility (Wischmeier and Smith 1978).

Accelerated soil erosion destabilizes SOC (Bailey et al. 2019) through disruption of soil aggregates by climatic erosivity, and exposure of the hitherto encapsulated SOC to microbial processes (Lal et al. 2004). Among several mechanisms of destabilization of SOC, release from biophysical occlusion (Bailey et al. 2019) by erosion-induced disruption of aggregates can impact the global carbon cycle (GCC). Because soil erosion displaces a large amount of SOC, as POC and/or mineral associated C (MOC), it is an important pathway affecting the GCC, leading to on-site and off-site impacts. On-site, it depletes SOC content and plant nutrients (i.e., nitrogen [N], phosphorus [P], sulfur [S], and potassium [K]), truncates the surface horizon, reduces water storage in the root zone, degrades soil structure by slaking and breakdown of aggregates, and aggravates crusting and surface sealing. Off-site, at depositional locations, erosion leads to run-on of water and agrochemicals, inundation, and burial of seedlings, increase in SOC content, and emission of greenhouse gases (GHGs). Similar trends are observed in wind erosion processes (Lal 2020).

The fate of SOC transported by erosional processes must be understood to assess its net effect as a source or sink of GHGs. Whereas the eroding site is progressively depleted of its SOC content, the depositional site where sediments are deposited is enriched. In northeastern China, for example, depositional sites store 5.9 times more SOC than eroding sites and 3.3 times more than the noneroding sites (Li et al. 2019).

The objectives of this article are to (1) deliberate the fate of SOC transported by erosional processes, and the attendant emission of GHGs; (2) discuss if erosion is a source or sink of atmospheric CO<sub>2</sub>; and (3) explain the importance of soil conservation to sequestration of atmospheric CO<sub>2</sub> and adaptation/mitigation of climate change. The discussion is based on the following three hypotheses: (1) the SOC redistributed over the landscape, including that stored in transient depositional sites, is prone to decomposition, methanogenesis, and nitrification/denitrification leading to enhanced emission of GHGs; (2) even a fraction of SOC transported into streams, reservoirs, lakes, and ocean and buried under anoxic conditions is prone to decomposition, and may be sequestered over a long period of time; and (3) adoption of conservation-effective measures sequesters atmospheric CO<sub>2</sub> and mitigates global warming.

### ■ Carbon Enrichment Ratio

Being the light fraction of low density and concentrated in vicinity of the surface layer, SOC/POC fractions are preferentially removed by erosional

processes. Thus, the SOC enrichment ratio ( $ER_{SOC}$ ) of the transported sediment increases vis-à-vis the SOC concentration of the uneroded soil (Lal 1976). The  $ER_{SOC}$  can be as much as 40 for wind-blown sediments (Webb et al. 2013), and 4 or 5 for water-deposited sediments (table 1) (Wang et al. 2014; Nachimuthu and Hulugalle 2016).

The magnitude of  $ER_{SOC}$  is affected by several factors (i.e., erosivity of rain or wind, time of sampling during the event, slope gradient and length, and sediment load). On the basis of data from a long-term repeated field rainfall simulation experiment, Hu et al. (2013) observed that  $ER_{SOC}$  in sediment increased at first, peaked at a time when steady-state runoff was achieved, and declined

**Table 1****Enrichment ratio of sediment for soil carbon.**

| Type  | Enrichment ratio               | Location         | Country       | Land use                  | Reference                           |
|-------|--------------------------------|------------------|---------------|---------------------------|-------------------------------------|
| Wind  | 0.6 to 1.9                     | Washington State | United States | Summer fallow             | Sharratt et al. (2018)              |
|       | 0.85 to 1.21                   | Loess Plateau    | China         | Laboratory simulation     | Li et al. (2016)                    |
|       | 1.28, 14.35, 3.07, 17.63, 9.27 | Queensland       | Australia     | Field measurement         | Webb et al. (2013)                  |
| Water | 0.61 to 2.13                   | Cropland         | China         | Field rainfall simulation | Nie et al. (2015)                   |
|       | 1.3 to 4.0                     | South Limburg    | Netherlands   | Rainfall simulation       | Wang et al. (2014)                  |
|       | 3.1                            | Rangelands       | Argentina     | Rainfall simulation       | Chartier et al. (2013)              |
|       | 1.2 to 3.0                     | Loess            | Belgium       | Rainfall simulation       | Wang et al. (2010)                  |
|       | 0.9 to 2.6                     | Maarkedel        | Belgium       | Laboratory                | Schiettecatte et al. (2008a, 2008b) |
|       | >2                             | Hyderabad        | India         | Watershed                 | Cogle et al. (2002)                 |
|       | >1 for slope of >3%            | Queensland       | Australia     | Rainfall simulation       | Palis et al. (1997)                 |
|       | 0.39 to 5.0                    | New South Wales  | Australia     | Agriculture               | Nachimuthu and Hulugalle (2016)     |

afterwards. Temporal changes in  $ER_{SOC}$  depend on the formation of crust, which must be duly considered in both water and wind erosion. The  $ER_{SOC}$  also decreases with increase in sediment concentration (Wang et al. 2013), climate erosivity (Martínez-Mena et al. 2012), the degree of aggregation of sediment (Schiettecatte et al. 2008a, 2008b), and slope gradient (Palis et al. 1997). Wang et al. (2014) observed that erosion, transport, and deposition enhanced  $ER_{SOC}$  from 1.3 to 4.0 because of increase in POC and the MOC.

In western Queensland, Australia, Webb et al. (2013) reported that wind-blown sediment from aggregated clay soil has lower  $ER_{SOC}$  than that of a sandy soil. Some wind-blown sediments contain 15% to 20% SOC compared to 0.3% to 4.2% in soil. The highest values of  $ER_{SOC}$  of wind-blown sediment were 48.4, 31.2, 30.7, 18.2, 13.9, and 14.4. Based on these data for Australia, Webb et al. (2013) estimated total SOC loss at  $5.4 \text{ Mt C yr}^{-1}$  (6.0 million tn  $\text{C yr}^{-1}$ ) in 100 Mt (110.2 million tn) of sediment, and  $147 \text{ Mt C yr}^{-1}$  (162.0 million tn  $\text{C yr}^{-1}$ ) globally in 3 Gt (3.3 billion tn) of sediment.

Any structural/physical change of SOC during its transport over the landscape and passage to the ocean (Aufdenkampe et al. 2011; Raymond et al. 2013; Regnier et al. 2013) can strongly affect its fate emissions of GHGs. Further, the fate of C transported over the landscape may differ widely in natural versus managed ecosystems. The latter have high sediment yield, and the transport of POC is primarily affected by the sediment yield (Galy et al. 2015). Tan et al. (2017) reported a regression equation of C yield (CY) and sediment yield (SY) as  $CY = 0.081 SY^{0.766}$ ,  $r = 0.89$ , and hypothesized that POC and sediment are physically bound in soils and are affected (e.g., detached and transported) similarly by the hydrological processes. Similar observations were made by Nie et al. (2015) with regards to  $ER_{SOC}$ . The exponent of 0.766 (less than 1) indicates that transport of POC decreases with increase in the sediment yield (Lal 1976).

Concentration of SOC in river sediment varies widely. In general, the concentration of POC in sediment decreases with increase in suspended sediment concentration (Meybeck 1993), while the effect may be offset because of the increase in total sediment discharge. In the Mississippi River, SOC content of the suspended sediment was  $1.8\% \pm 0.3\%$  (Trefry et al. 1994), with 66% of total C comprising of POC.

### ■ The Fate of Soil Organic Carbon Transported by Sediments and the Impact on the Global Carbon Cycle

Because of a high concentration of POC and the labile fraction, the SOC removed by erosion is especially susceptible to decomposition, which has important implications to the GCC. Thus, the nature of SOC removed (POC versus MOC, or

labile versus recalcitrant) must be considered in evaluating its impact on GCC. Particulate form, after rupture or dispersion of aggregates, may be easily mineralized (Martínez-Mena et al. 2012). Based on a study across a Mediterranean catchment, Martínez-Mena et al. (2019) observed that mineralization of the most labile SOC (or POC) was predominant during transport.

From a study of nine river basins in China, Wang et al. (2019) observed that 47% to 57% of eroded SOC was deposited over land, 25% to 44% was deposited in the channel, and 8% to 18% was delivered into the sea. Wang and colleagues observed that, over a short period of simulated erosion, only 1.5% of the transported C was mineralized. Yue et al. (2016) documented that severe water erosion in China displaced  $180 \pm 80 \text{ Mt C yr}^{-1}$  ( $198.4 \pm 88.1$  million tn C  $\text{yr}^{-1}$ ) over two decades, of which  $45 \pm 25 \text{ Mt C yr}^{-1}$  ( $49.6 \pm 27.6$  million tn C  $\text{yr}^{-1}$ ) is buried in aquatic ecosystems. Fiener et al. (2015) applied the SOC turnover model (SPEROS-C) to a 4.2 ha (10.4 ac) arable catchment in Germany for a 57-year period, and a total of 901 model runs were performed. The overall C balance of the catchment indicated a maximum C source of  $44 \text{ g C m}^{-2}$  ( $0.081 \text{ lb C yd}^{-2}$ ).

### ■ Gaseous Fluxes from Depositional Sites

There is an erroneous and mythical belief that erosion-induced transport of C leads to mitigation of climate change because of the long-term burial of SOC under anaerobic environments (Berhe et al. 2007; Van Oost et al. 2007; Quinton et al. 2010; Fiener et al. 2015). Indeed, the role of fluvial sedimentary areas as SOC sinks remains largely unquantified for the drier Mediterranean (Martínez-Mena et al. 2019) and several other regions prone to accelerated erosion. Yet, such misinterpretation has numerous adverse consequences to attempts in aligning policies for soil conservation and sustainable management with those of adaptation and mitigation of climate change. Adverse effects of accelerated soil erosion, already the second largest source of GHG emissions, are likely to be exacerbated with the current and projected climate change. Furthermore, SOC and related nutrients, such as N, P, K, and S, transported and buried in depositional sites are also major sources of GHGs, especially those of nitrous oxide ( $\text{N}_2\text{O}$ ) and methane ( $\text{CH}_4$ ). However, the input of labile materials in the depositional sites plays an important role in soil respiration and emission of GHGs from the surface layer. Emission of  $\text{CO}_2$  tends to increase in deposition positions of eroded landscape because of the profound influences of added N supply on functioning of microorganisms, which drive the soil C efflux (Meng et al. 2016). Based on a study conducted in temperate forests of the Sierra Nevada, California, Stacy et al. (2019) observed that sediment transported in drier years was more enriched in unprotected POC derived from surficial soils. Incubation of these sediments produced

72% to 97% more CO<sub>2</sub> during decomposition than soil C did. Thus, Stacy and colleagues concluded that without stabilization (through burial or reaggregation), sediment-transported SOC is prone to decomposition.

Wang et al. (2014) concluded that CO<sub>2</sub> emission was the predominant form of overall C loss, accounted for 90.5% of the total erosion-induced C loss, and was equal to 18 g C m<sup>-2</sup> (0.033 lb yd<sup>-2</sup>) in a four-month study. In comparison, only 1.5% of the total redistributed C was mineralized into CO<sub>2</sub> indicating the stabilization of deposited/buried SOC. In the subsoil, however, stabilization of SOC within microaggregates may reduce respiration (Li et al. 2018). Furthermore, dispersed clay at the depositional sites may be reaggregated and stabilize some of the SOC/POC/MOC encapsulated within reformed stable aggregates (Cheng et al. 2010). Reaggregation of clay and formation of stable microaggregates within aggregates can occur in the upper layers of depositional areas, as well as in the deeper layers where SOC gets stabilized (Martínez-Mena et al. 2019).

Deemer et al. (2016) estimated that reservoirs created by dams account for 0.8 (0.5 to 1.2) Gt CO<sub>2</sub> eq y<sup>-1</sup> (0.88 [0.55 to 1.32] billion tn CO<sub>2</sub> eq yr<sup>-1</sup>), with a majority of the forcing due to CH<sub>4</sub>. Transport of a large stock of SOC into the reservoirs is a major source of CH<sub>4</sub> and N<sub>2</sub>O, which are 34 and 298 times more potent than CO<sub>2</sub> on a 100-year time scale (Etminan et al. 2016). Deemer et al. (2016) estimated that global reservoirs emit 13.4 Mt CH<sub>4</sub>-C y<sup>-1</sup> (14.8 million tn CH<sub>4</sub>-C yr<sup>-1</sup>), 36.8 Mt CO<sub>2</sub>-C y<sup>-1</sup> (40.6 million tn CO<sub>2</sub>-C yr<sup>-1</sup>), and 0.03 Mt N<sub>2</sub>O-N y<sup>-1</sup> (0.033 million tn N<sub>2</sub>O-N yr<sup>-1</sup>). Hamdan and Wickland (2016) reported that total global CH<sub>4</sub> emissions range from 500 to 600 Mt CH<sub>4</sub> y<sup>-1</sup> (551.2 to 661.4 million tn CH<sub>4</sub> yr<sup>-1</sup>), of which 30% to 35% of the total are from natural sources including reservoirs. In Poland, Gruca-Rokosz and Tomaszek (2015) calculated fluxes of CH<sub>4</sub> and CO<sub>2</sub> at the sediment-water interface in the Rzeszow Reservoir. The fluxes ranged from 0.01 to 2.19 mmol m<sup>-2</sup> d<sup>-1</sup> (0.008 to 1.83 mmol yd<sup>-2</sup> day<sup>-1</sup>) for CH<sub>4</sub> and 0.36 to 45.33 mmol m<sup>-2</sup> d<sup>-1</sup> (0.30 to 37.90 mmol yd<sup>-2</sup> day<sup>-1</sup>) for N<sub>2</sub>O. Further, the 24% to 72% of CO<sub>2</sub> in the top layer of sediment that came from degradation of organic matter by methanogenesis was greater than that in the deeper layers.

Erosion-induced emissions are also increased at the depositional sites of eroding landscapes (Meng et al. 2016). Dominant sources of N<sub>2</sub>O also include sediments and water bodies (Butterbach-Bahl et al. 2013; Oertel et al. 2016). Meng et al. (2016) observed that an average of 72% C incorporated by all microbial groups in depositional sites (foot slopes) was derived from SOC, indicating that a large amount of SOC was mineralized at the depositional sites. Prevalence of anaerobic conditions at these sites may also lead to methanogenesis and emission of CH<sub>4</sub>.

## ■ Erosion Control and Carbon Sequestration

Effective erosion control and restoration of eroded soils can create a net sink of atmospheric CO<sub>2</sub> (figures 1a and 1b). Indeed, it takes much less energy to keep soil in place than it does to rebuild SOC. An example of a C sink through erosion control is that of exchange in the terrestrial C budget between 1970 and 2017 through the revegetation program of the severely eroded Yellow River basin in China. In comparison with the baseline data from 1950 to 1970, through effective erosion control through afforestation and installation of other control measures 20.6 Mt C y<sup>-1</sup> (22.7 million tn C yr<sup>-1</sup>) was sequestered (Ran et al. 2018). Furthermore, the erosion-induced decomposition of transported SOM declined by 34% from 8 to 5.3 Mt C y<sup>-1</sup> (8.82 to 5.84 million tn C yr<sup>-1</sup>) (table 2). Ran and colleagues observed that effective soil erosion control measures also collectively conserved 20.6 Mt C y<sup>-1</sup> (22.7 million tn C yr<sup>-1</sup>) from 2000 to 2015, and the rate of C accumulation in the terrestrial biosphere (soil and vegetation) may continue because the C sink capacity has not yet been saturated. Further, reduction in C emissions by 2.7 Mt C y<sup>-1</sup> (3.0 million tn C yr<sup>-1</sup>) from 8 to 5.3 Mt C y<sup>-1</sup> (from 8.8 to 5.8 million tn C yr<sup>-1</sup>) accounted for 63% of the net primary production of 4.3 Mt C y<sup>-1</sup> (4.7 million tn C yr<sup>-1</sup>) (Ran et al. 2018). Therefore, effective soil conservation measures can have a drastic impact on C capture from the atmosphere and transfer it into the terrestrial sink. An example of the terrestrial sink created by effective erosion control and restoration of eroded lands is given in table 3. The hypothetical example shows that if all the mineralizable SOC in eroded sediment were decomposed,

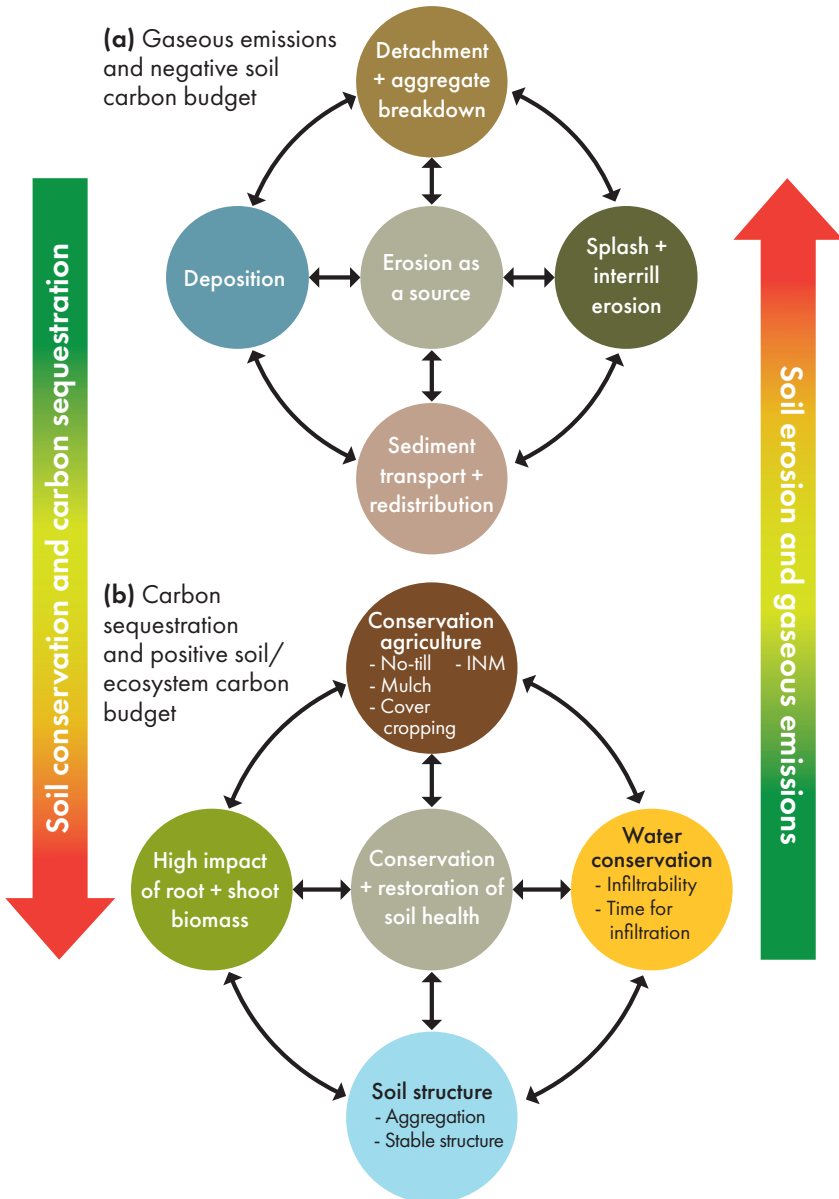
**Table 2**

**Effect of conservation measures and afforestation of the Yellow River basin on fate of the erosion-induced transport of carbon (C; recalculated from Ran et al. [2018]).**

| <b>Parameter</b>        | <b>Baseline (1950 to 1970) (Tg C y<sup>-1</sup>)</b> | <b>After conservation (2000 to 2015) (Tg C y<sup>-1</sup>)</b> |
|-------------------------|--|--|
| Soil erosion            | 21.4 ± 5.2   | 11.1 ± 4.1   |
| Decomposition           | 8.0 ± 8.8  | 5.3 ± 4.6  |
| Redistribution          |  |  |
| • Dam trapping          | —  | 3.3 ± 1.5  |
| • Hillslope deposition  | —  | 1.0 ± 1.3  |
| • Channel sedimentation | —  | 0.3 ± 0.5  |
| • Sediment diversion    | —  | 0.5 ± 0.3  |
| Ocean transport         | 6.1 ± 4.3  | 0.7 ± 0.6  |

Figure 1

(a) Soil erosion as a source, or (b) soil conservation as a sink of atmospheric carbon dioxide and other greenhouse gases.





**Table 3****Global carbon sink (C) through effective erosion control and afforestation of eroded landscape.**

| Parameter   | Units                        | Magnitude  | Reference              |
|---|------------------------------|--|------------------------|
| Land area affected  | 10 <sup>6</sup> ha           | 1,100  | Oldeman (1994)         |
| Global sediment transport   | Gt y <sup>-1</sup>           | 36.6   | Walling (2008, 2009)   |
| Sediment delivery ratio   |                              | 0.1 to 0.3 (0.2)   | Doetterl et al. (2016) |
| SOC concentration in sediment   | %                            | 1.8 ± 0.3  | Trefay et al. (1994)   |
| Total C transported   | Gt C y <sup>-1</sup>         | 3.29   |                        |
| Emission factor from soil erosion   | t t <sup>-1</sup> of erosion | 0.3 t CO <sub>2</sub> eq t <sup>-1</sup> of erosion                        | Worrall et al. (2016)  |
| Total global emissions (CO <sub>2</sub> , CH <sub>4</sub> , N <sub>2</sub> O) | Gt y <sup>-1</sup>           | 54.9 Gt CO <sub>2</sub> eq y <sup>-1</sup><br>(14.5 Gt C y <sup>-1</sup> ) |                        |

it would be equivalent to 14.5 Gt C<sub>eq</sub> y<sup>-1</sup> (16.0 billion tn C<sub>eq</sub> yr<sup>-1</sup>). Thus, global soil conservation is an absolute necessity.

### ■ Aligning Agriculture with Global Climate Policy

The impact of agriculture on gaseous emissions (CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O) is a global issue (Muñoz et al. 2010; Oertel et al. 2016). Soil processes, and their perturbation by human activities are significantly impacting the emission of GHGs into the atmosphere. Sequestration of SOC can be enhanced by identifying appropriate soils and ecoregions where conservation agriculture can be adopted (Ogle et al. 2019), where cover cropping can improve soil quality by controlling erosion and improving input of biomass C (Poeplau and Don 2015; Ruis and Blanco-Canqui 2017), and where use of compost (White et al. 2020) can ensure that the Soil Conditioning Index is positive (Zobeck et al. 2007; Franzluebbers et al. 2011). Conservation agriculture, based on no-till farming and the related components, is useful to reducing soil erosion, creating climate-resilient agriculture, and advancing food security with additional co-benefits of C storage (Lal 2015). Therefore, it is important to develop a soil guide for adoption of conservation agriculture (Lal 1985). The recommended soil conservation practices must ensure that soil erosion does not exceed the tolerable soil loss (T) value, because erosion rates equivalent to two T may be excessive with adverse effects on SOC stock.

Since 2015, policy makers have focused on soil C sequestration for food and climate by adopting the “4 per Thousand” initiative globally and Adapting Agriculture in Africa (Lal 2019). A similar program, called Platform

on Climate Action in Agriculture, focused on soil management and making agriculture a solution to climate change, was launched at COP 25 in Chile/Madrid in December of 2019. There is also a growing interest in industry to enhance SOC sequestration (e.g., Danone, Indigo Agriculture, and Patagonia) through adoption of regenerative agriculture. Thus, soil scientists must seize the moment and support such initiatives.

## ■ Conclusions

A synthesis of the literature indicates all three hypotheses are proven and also supports the following conclusions:

1. The SOC transported and redistributed by erosional processes (water, wind) is prone to decomposition; methanogenesis and nitrification/denitrification; and emissions of CH<sub>4</sub> and N<sub>2</sub>O, along with that of CO<sub>2</sub>.
2. Some of the SOC carried into the aquatic ecosystems and depositional sites is buried and is subject to reaggregation and stabilization of SOC. Nonetheless, a sizeable part of the SOC in these sites is also prone to mineralization, methanogenesis, and nitrification/denitrification by increase in microbial activity at the depositional position.
3. Adoption of conservation agriculture and cover cropping lead to a positive soil conditioning index, protect the existing SOC stock, and sequester additional C within soil and the biomass. Erosion control and soil conservation enhance SOC sequestration.
4. Conservation effective measures should be adopted to limit the soil erosion within the tolerable limit (one T-value).

Overall, based on the net effect of all four phases, erosion is a major source of GHGs. SOC/POC enriched sediments are transported, redistributed, and exposed to a wide range of environmental (moisture, temperature, etc.) conditions. Thus, soils of agroecosystems must be protected through adoption of conservation-effective measures so that risks of soil erosion are minimized. Aligning science with policy is an important step forward. International initiatives can make an important difference. Soil conservation and restoration of eroded soils is a win-win option to improve productivity and mitigate global warming.

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