

Review

Fate of Soil Carbon Transported by Erosional Processes

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Abstract: The accelerated process of soil erosion by water and wind, responsible for transport and redistribution of a large amount of carbon-enriched sediments, has a strong impact on the global carbon budget. The breakdown of aggregates by erosivity of water (raindrop, runoff) and wind weakens the stability of soil C (organic and inorganic) and aggravates its vulnerability to degradation processes, which lead to the emission of greenhouse gases (GHGs) including CO₂, CH₄, and N₂O, depending on the hydrothermal regimes. Nonetheless, a part of the eroded soil C may be buried, reaggregated and protected against decomposition. In coastal steep lands, (e.g., Taiwan, New Zealand) with a short distance to burial of sediments in the ocean, erosion may be a sink of C. In large watersheds (i.e., Amazon, Mississippi, Nile, Ganges, Indus, etc.) with a long distance to the ocean, however, most of the C being transported is prone to mineralization/decomposition during the transit period and is a source of GHGs (CO₂, CH₄, N₂O). Land use, soil management and cropping systems must be prudently chosen to prevent erosion by both hydric and aeolian processes. The so-called plague of the soil, accelerated erosion by water and wind, must be effectively curtailed.

Keywords: global warming; soil erosion; carbon erosion; gaseous emissions; methanogenesis; nitrification/denitrification; dust; enrichment ratio; burial of carbon; aggregate disruption

1. Introduction

Accelerated soil erosion by water and wind is a predominant process that impacts the soil carbon (C) budget [1] through a range of interacting processes [2]. The severe problem of accelerated soil erosion is widely regarded as a global menace, which is also threatening the agricultural resource base [3] that sustains planetary processes [4] and generates numerous ecosystem services. Erosional processes include breakdown of structural aggregates, and selective removal and redistribution of the sediment and displaced C on the landscape. Displaced C may be mineralized or buried [5] depending on the site-specific conditions. Whereas reaggregation and deep burial may stabilize the displaced C against microbial processes [6], decomposition induced by aggregation breakdown and alterations in soil moisture and temperature regimes over the landscape may affect the emission of carbon dioxide (CO₂) under aerobic environments [7] and those of methane (CH₄) and nitrous oxide (N₂O) under anaerobic conditions [8]. Therefore, erosion-induced transport of soil C followed by its redistribution en route to depositional/depression site(s) may lead to emission of all three greenhouse gases (GHGs). With a large amount of sediment and sediment-laden C being transported over the landscape, and the magnitude of sediment being increased because of anthropogenic activities, it is critically important to understand the complex and interactive processes affecting the fate of C being displaced and transported over the landscape (e.g., sediment) and emission of GHGs to the atmosphere (i.e., windborne dust). Emissions of all three GHGs by erosional processes, at all landscape positions from summit to foot slope and into the atmosphere, must be understood, quantified, and accounted for the purpose of compiling the soil C budget under diverse land use and soil management systems.

Novara et al. [9] reported that transport of soil sediments by erosional processes increased SOC mineralization by 43% under Mediterranean conditions. The global menace of soil erosion (hydric and aeolian) is projected to be aggravated by climate change because of



Citation: Lal, R. Fate of Soil Carbon Transported by Erosional Processes. *Appl. Sci.* **2022**, *12*, 48. <https://doi.org/10.3390/app12010048>

Academic Editor: Itzhak Katra

Received: 27 October 2021

Accepted: 15 December 2021

Published: 21 December 2021

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the increase in climatic erosivity [10] and the frequency of extreme events [11,12]. Therefore, the objective of this review article is to deliberate the fate of soil C being transported over the landscape, identify ecosystem conditions that lead to either mineralization and emission of GHGs or to sequestration of the transported C and its stabilization, understand the net effect of erosion/deposition processes as a source or sink of GHGs, explain the effect of global warming on erosional processes, and outline some researchable priorities. Rather than being a comprehensive and inclusive review of the available literature, this article merely provides some pertinent examples of the processes, factors and causes which control the fate of carbon being transported by erosional processes.

2. Selectivity of the Soil Erosion Process

Global soil erosion by water and wind, exacerbated by agricultural expansion and intensification and other anthropogenic activities, transports, and redistributes large amounts of soil organic C (SOC) and soil inorganic carbon (SIC) with significant impacts on global C budget, atmospheric chemistry, and water quality. However, hydric erosion may have a more significant impact on transport of SOC and emission of GHGs. Indeed, soil erosion by water is a multistage process encompassing a range of pedological processes. Some of these processes are described below:

- i. **Breakdown of aggregates** and leading to the exposure of the hitherto protected SOC against microbial processes and environmental conditions, is the first process. Macroaggregates, containing labile fractions, are broken apart by the kinetic energy and momentum of the impacting raindrops and velocity of the runoff flow. The SOC thus exposed is accessible to microbial processes.
- ii. **Removal of the colloidal and light fractions** (clay, fine silt, SOC, SIC, sesquioxides) is aggravated by the breakdown of aggregates. Thus, the sediments have a high C-enrichment ratio (CER). The latter is defined as the ratio of C in sediment to that in the original soil from which the sediment are derived [13]. A high CER is observed in both hydric and aeolian sediments.
- iii. **Redistribution of the sediment and associated soil C** is a major pedological process on actively eroding landscapes. Furthermore, redistribution is accompanied by possible mineralization of biomass-C (into CO₂, CH₄, and N₂O), depending on the hydrothermal regimes. Whereas CO₂ is the primary GHG evolved and emitted under aerobic conditions, CH₄ and N₂O may be evolved under anaerobic environments.
- iv. **Deposition of sediments** SOC-laden sediments are deposited at depressional and other sites following Stokes Law. Accordingly, heavier fractions (e.g., gravels, sand and coarse silts) are deposited in vicinity of the eroding sites and SOC and clay fractions are deposited either latter or carried farther away. Windblown sediments originating from the Sahara have been observed in the Caribbean and northern Europe. This process is the principal cause of eutrophication of water, pollution of air and responsible for the off-site adverse effects of the erosional process.
- v. **Burial of SOC:** and SIC deposition following Stokes Law leads to burial and stratification, with coarser heavier sediments deposited first and the lighter fractions (SOC, SIC, clay particles) later. With multiple events over time, the layering can be observed in active depositional sites which are not disturbed by farm operations or natural perturbations.
- vi. **Reaggregation** Some of the dispersed clay, released by the breakdown of aggregates (step 1) and other pedological processes aggravating the slaking of structural units, may interact with SOC and cations and the reformation of aggregates. Therefore, some of the buried SOC may be stabilized through encapsulation within reformatted aggregates.
- vii. **Gaseous Emission from Eroding and Depositional Sites:** Erosional processes may aggravate emissions of GHGs from both eroding and depositional sites because of the drastic perturbations of soil structure. The magnitude and type of gaseous

emission (CO_2 , CH_4 , and N_2O), depend on site-specific conditions and their spatial and temporal variations because of natural and anthropogenic processes.

However, pedological processes involved in the above listed erosional stages vary widely, depending on the stage of the erosion/deposition process and prevalence of the specific hydrothermal regime of the specific landscape position. The third stage, involving redistribution of soil C, is not uniform but affected by preferential transport and deposition. The site of SOC deposition depends on topography (slope gradient, length, shape and aspect), in-transit distance of sediment containing SOC [14] and the time taken for the sediments to reach the deposition site. Because of the selective removal of the light fraction, CER of the sediments may range from 1.3 to 40.0 [6]. Furthermore, the average SOC fluxes as particles ($4.7 \text{ g C/m}^2\cdot\text{yr}$) was found to be 18 times larger than that of the dissolved organic C (DOC). Wang and colleagues also observed that the cumulative emission of soil CO_2 slightly decreased at the erosion site but increased by 56% and 27% at the transport and depositional zone, respectively, in comparison to non-eroded soils. Site-factors (i.e., slope gradient, slope aspect, slope shape, and the landscape position) are the control of specific biogeochemical and biogeophysical processes that determine transformation of soil and emission of specific GHGs such as CO_2 by aerobic mineralization, CH_4 by methanogenesis catalyzed by anaerobiosis (poor drainage and high degree of saturation), and N_2O by nitrification/denitrification processes under variable hydrothermal regimes.

3. Soil Erosion as a Source or Sink of Carbon

There are numerous interacting processes affecting the fate of SOC and SIC transported by erosional processes. The net effect as a source or sink depends on the balance of emission vs. sequestration, as outlined in Figure 1. Factors which increase the source from erosion-induced transport include disruption/breakdown of aggregates, a selective removal of the low-density/lighter fractions, fast-transport landscape vs. slow-transport landscape, and severe degradation of the soil at the eroded site so that its agronomic productivity is curtailed (Figure 1). There could be deposition in short-distance anaerobic lakes that would be a source of carbon, and in long-distance aerobic environment with reaggregation at depositional sites leading to sequestration of C. Some conditions which may lead erosion-induced transport of soil C as a sink may include a fast transport landscape with short transit time to the deposition site, deep burial of C, rapid reaggregation of buried C so that it is protected against microbial processes, and slight or moderate degradation of soil at the eroded site so that its agronomic productivity is not jeopardized (Figure 1).

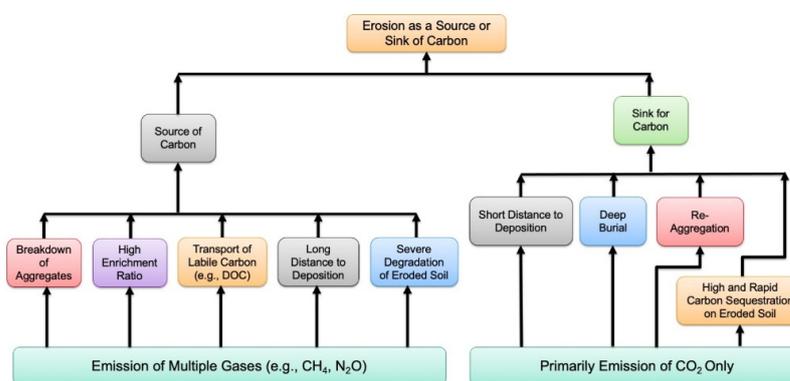


Figure 1. Pedological processes which impact emission of greenhouse gases from erosion-induced transport of soil carbon. Available literature highlights the importance of soil, climate and land attributes which aggravate the breakdown of aggregates and severely degrade the quality and productivity of eroded soil which has a long recovery period. Some steep land coastal ecoregions with a short transit time can be a sink of eroded soil carbon. Globally, however, accelerated erosion (hydric, aeolian, tillage) is a source of greenhouse gases especially those with high global warming potential (i.e., CH_4 , N_2O).

Some critical factors which impact the gaseous processes during the transit phase and at the depositional site are briefly described below.

3.1. Slope

Attributes of land slope that determine specific pedotransformation include gradient (steepness), aspect (north vs. south facing), shape (convex vs. concave), position (summit, side slope, foot slope), and length (short vs. long). The magnitude of sediment generated, and the amount of soil C transported laterally also depend on the soil surface as altered by agronomic management and its interaction with the slope attributes. While assessing the scale-dependency of erosion-induced variation in CO₂ emission from terraced slopes, Hu et al. [15] observed that emissions tend to decline with increase in slope gradient. The declining trend is attributed to increased runoff and more soil erosion on steeper slopes and making eroded soil less habitable for soil biota in terms of their activity and species diversity.

Slope attributes also affect the deposition of sediments and the associated soil C. Lense et al. [16] evaluated losses of soil C by water erosion in a tropical watershed characterized by a wide land-use diversity. Lense and colleagues observed that of the 126.5 Mg/yr. of the total eroded SOC from the watershed, 111.6 Mg/yr. were deposited in relief depressions and only 14.9 Mg/yr. reached the water body system. Wei et al. [17] observed that bulk soil samples obtained from the summit landscape position emitted the greatest cumulative CO₂-C (0.49 ± 0.04 g C/kg) among all the landscape positions. Furthermore, the CO₂ emission rate from coarse-sized aggregate fraction at depositional slope positions (toe-slope and foot-slope) was significantly greater than those from the eroded slope positions (summit, shoulder slope and back slope).

Erosion-induced losses of soil C and the attendant CO₂ emissions from a loess and black soil in China were studied by Gao et al. [18] for three slope gradients (5°, 15° and 25°) using a rainfall simulator. On average, SOC loss from loess soil was about 1.8 times that from black soil although the SOC concentration in original black soil was 56% higher than that in the loess soil. The cumulative CO₂ emissions from the eroding slopes of the loess ranged from 15.4 to 19.7 g C/m² compared with 28.1 to 59.6 g C/m² for the black soils. However, when the slope gradient increased from 5° to 25°, the cumulative soil CO₂ emission decreased by 23.8% on black soil and 12.6% on loess soil. These observations documenting decrease in on-site emissions from steep slopes are similar to those by Hu et al. [15] discussed above.

3.2. Soil Attributes

As is documented by Gao et al. [18] regarding the difference in SOC loss from black soil vs. a loess soil, erosion-induced emissions of GHGs are also strongly affected by soil physical, chemical, and biological properties, which are indeed strong moderators of the rate and total magnitude of GHGs emissions. Soil hydrological properties, as affected by water repellency, also impact emissions of GHGs. In the context of the western U.S. wildfires, which have increased in intensity and scale because of the climate change, Samburova et al. [19] assessed the impact of the fire-induced soil water repellency on GHG emissions. The water repellency was characterized by: water drop penetration time (WDPT), effective contact angle and relative sorptivity of exposed silica sand (used as soil surrogate). All sand samples, exposed to either fire emissions or fulvic acid were characterized by WDPT > 81 s, effective contact angle of 78–87°, and relative sorptivity between 0.31 and 0.49 compared with untreated sand samples with WDPT < 0.5 s, effective contact angle of 48°, and relative sorptivity of 1.

Soil attributes affect emissions of GHGs from eroding and depositional sites through their effects on SOC stability. Based on an experiment on SOC stability in sub-tropical China, Nie et al. [20] observed that eroding and depositional sites had different SOC contents but had similar organic functional groups. However, SOC in eroded soils was more stable than that in depositional soils in the surface 0–5 cm and 5–10 cm layers only. Differences in SOC

stability were attributed to differences in soil properties such as texture, bulk density, pH, SOC content, DOC, Fe content, soil aggregates, depth, erosion, and deposition.

A study on the effects of soil properties on water erosion and emissions of GHGs on soil under diverse land uses in Spain by Gispert et al. [21] showed that soils with a higher SOC content showed proportionally lesser CO₂ emissions and were a C sink. In comparison, the shallowness of the soil profile, overgrazing, and frequent fire occurrence were elements that threatened soil ecosystem integrity and must be addressed.

3.3. Hydro-Thermal Regime and Aggregation

Emission of GHGs depends on soil moisture and temperature regimes, as moderated by aggregation, in relation to stability of aggregates along with size and continuity of pores, through their strong impacts on activity and species diversity of soil biota. For example, based on a 19-day simulated depositional experiment using three soils of contrasting texture (Ultisol, Mollisol, and Entisol) and two moisture regimes, Mao et al. [22] observed that deposition increased C emissions under both drying and wetting conditions for the Ultisol and Entisol, but the trend in Mollisol varied with the soil moisture regime. Gaseous emissions increased under drying but decreased under wetting conditions. Thus, the effect of deposition on GHG emissions depend on the soil moisture regime and its control such as texture, clay minerals, SOC content and aggregation.

Soil aggregation (structure) is a key attribute in stabilization of SOC [23]. In addition, degree and stability of soil aggregates is an important factor affecting emission of GHGs. An incubation study by Wei et al. [17] involving soil aggregates obtained from diverse landscape positions indicated that CO₂ emissions from coarse-size aggregate fractions (0.024 ± 0.009 g C/kg-d) was six times higher than that from small size aggregate fractions (0.0038 ± 0.0011 g C/kg-d) at the depositional toe slope position. Wei and colleagues concluded that the breakdown of aggregates (stage 1 of the erosional process) by accelerated erosion impacts both aggregate size distribution and CO₂ emissions from aggregates at different slope positions.

Soil aggregates, being transported by fluvial processes, may increase settling velocities of mineral particles contained within the aggregate and skew the distribution of SOC along the slope. Hu et al. [14] observed that redeposition of eroded SOC within terrestrial ecosystems increased by 64% considering the actual aggregate size, and the CO₂ emission rates also differed across settling fractions depending on soil type. Hu and colleagues also observed that over a 50-day incubation, CO₂ emissions from sediment was 114% greater than that from the non-eroded soil, probably because of enrichment with the labile fractions.

Chaplot and Cooper [24], based on their study comprising 24 locations of a typical hillslope of the South African Highveld with a wide range of soil texture, concluded that the increase in aggregate stability resulted in a significant increase in particulate and DOC concentrations in the eroded sediments and in losses of gaseous organic C. In contrast, however, high aggregate stability induced low total loss of particulate and DOC because of increase in water infiltration rate and reduced transport in runoff. In the context of wind erosion risks of Histosols in the U.S., Zobeck et al. [25] observed that variations in sediments and dust emissions is linked to soil properties. A study based on simulated rainfall in subtropical China by Huang et al. [26] indicated that soil bulk density, moisture content, and DOC were the major factors controlling erosion-induced SOC mineralization.

3.4. Other Factors Affecting Long-Term Biogeochemical and Biogeophysical Transformation

Stability of erosion-induced transport of SOC and its fate as a source or sink of GHGs is governed by factors which impact its long-term biogeochemical and biogeophysical transformation. Notable among these are elevation, vegetation, land use, frequency and intensity of natural or managed fires, and soil biodiversity in relation to activity and species diversity of macro, meso and micro fauna and flora. In addition to biochemical processes, biogeophysical transformation of aggregates (e.g., breakdown or slaking vs. reaggregation of the dispersed materials) is an important process governing the fate of SOC transported by erosional process

as a source of or sink of GHGs (1,2,5,6,7,14,19). Apparently, scale is also an important variable affecting the net balance of SOC in relation to erosional processes (15) Factors elevating soil moisture regime may also aggravate C loss from mineral soils (26).

4. Soil Erosion and the Global Carbon Budget

The magnitude of sediment displacement globally by water erosion increased from 14.0 Gt/yr. during the prehuman era to 36.6 Gt/yr. at present in the absence of reservoir trapping [27]. There has also been a strong increase in windblown sediments due to the expansion of agriculture and other anthropogenic activities. Despite the severe environmental consequences of water erosion, its effects on the global carbon budget (GCB) have neither been adequately researched nor been accounted for.

4.1. Water Erosion

The GCB is strongly affected by accelerated erosion by water and reportedly emits ~1.1 Pg C/yr. [2]. However, the C emitted by erosion is not accounted for in the GCB. In India, Mandal et al. [28] reported that erosion transported about 114.4 Tg of C/yr. of which 34.6 Tg C is emitted into the atmosphere. Using ^{13}C isotopic signature in abandoned Mediterranean agricultural land, Novara et al. [9] observed that previous estimates have not considered that erosion transported SOC can be released to the atmosphere as a range of GHGs (CO_2 , CH_4 , N_2O). The data by Worrell and colleagues from U.K. rivers showed that soil erosion is a net source of GHGs with median emission factors of 5.5, 4.4, and 0.3 Mg CO_2 eq/yr. for one Mg of fluvial C, gross C erosion, and gross soil erosion respectively. Worrell and colleagues concluded that gross soil erosion would only be a sink of both C and GHGs only if all the following criteria are met: (i) gross soil erosion rates are $<91 \text{ Mg/km}^2\cdot\text{yr.}$, (ii) the eroded C is completely replaced by new SOM, and (iii) if less than half of the gross erosion makes it into the ocean network. Similar conclusions were arrived at by Lal et al. [5]. In this context, Müller-Nedebock and Chaplot [29] also highlighted the importance of sheet erosion and its impact on the lateral transport of SOC and the attendant impact on GCB. Based on a study involving data from 240 runoff plots over entire rainy seasons from different regions of the world, Müller-Nedebock and Chaplot reported that the media in particulate organic carbon loss (POCL) was $9.9 \text{ g C/m}^2\cdot\text{yr.}$ with the highest value of $10.8 \text{ g C/m}^2\cdot\text{yr.}$ for semi-arid soils followed by $6.4 \text{ g C/m}^2\cdot\text{yr.}$ for tropical soils and $1.7 \text{ g C/m}^2\cdot\text{yr.}$ for temperate soils. Assuming the mean POCL of $27.2 \text{ g C/m}^2\cdot\text{yr.}$, the total amount of SOC displaced by sheet erosion from its source was estimated at $1.32 \pm 0.20 \text{ Gt C/yr.}$ or about 13.2% of the 10 Gt of C emitted from the fossil fuel combustion. Müller-Nedebock and Chaplot also observed that erosion-induced CO_2 emissions may be limited in clayey soils but severe in sandy soils, and that sheet erosion is an important and efficient mechanism of detachment and transport of surficial material (e.g., silt, clay, SOC).

It is also recognized that soil erosion can be a sink of C in coastal ecosystems and steep terrains. For example, a study in New Zealand documented that transport of sediments and POC from coastal steep lands is a sink. Dymond [30] observed that the North Island of New Zealand exports $1.9 \text{ Tg of POC/yr.}$ to the sea and sequesters 1.25 ± 0.3 to $0.6 \text{ Tg POC}\cdot\text{yr.}$ through regenerating soils. In contrast, the South Island of New Zealand exports 2.9 ± 0.7 – 1.5 Tg POC/yr. and sequesters about the same amount. With 80% efficiency of burial at seas, New Zealand has a net sink of 3.1 ± 2 – 2.5 Tg C/yr.

4.2. Emissions from Tillage-Induced Soil Erosion

Tillage-induced soil displacement downslope is a significant process on cultivated steep lands. Soil disturbance by plow tillage (PT) or its lack under no till (NT) can have variable effects on the magnitude of SOC removal by erosional processes and pedological processes involved in gaseous emissions. In the Kwa Zulu-Natal Province of South Africa, Chaplot et al. [31] evaluated the effects of NT vs. PT on losses of SOC from soil. Chaplot and colleagues observed that soil under NT had greater SOC density than that under PT

(17.70 vs. 13.19 kg/m³), lower gaseous emissions by 4.4% (10.40 vs. 10.88 g CO₂-C/m²) but reduced the release of CO₂ from eroded sediments (0.185 vs. 0.778 g CO₂C/m²) representing 76.3% decline. In the case of PT, cumulative emissions over a 141-day period were 19% greater in sediments (0.048 g CO₂C/g C) compared to soils (0.04 g CO₂C/g C). In the case of NT, emissions were 33% lower in sediments (0.024 g CO₂C/g C) compared to soils (0.032 g CO₂C/g C). The authors' hypothesized these trends to a high aggregate stability and better physical protection of SOC within stable aggregates. Conservation agriculture (CA), based on NT with residue mulching and cover cropping, is widely reported to sequester SOC [32]. A modeling study by Gaiser et al. [33] showed higher losses of SOC under PT than NT through enhanced CO₂ emissions and increased losses through intensified erosion. Gaiser and colleagues suggested that tillage effects on SOC losses through soil erosion must also be accounted for in the GCB. The loss of SOC from eroded soil is also determined by the sediment size distribution and its MRT [34].

Plow tillage is also a source of dust. In some cases, tillage-induced emissions of dust are more than that by wind. Funk et al. [35] observed that tillage-induced emissions are determined by the soil moisture content at the time of tillage. Funk and colleagues found that the threshold level of soil water content for fine dust emissions of soil was 2% to 5% for sandy soil, 5% to 10% for silty soils, ~30% for clayey soils, and 25% to 45% for organic soils.

4.3. Grazing Systems and Gaseous Emissions by Hydric Erosion

Similar to mechanical tillage, overgrazing can also accelerate hydric erosion and influence gaseous emissions. In addition to reducing the plant cover, overgrazing may aggravate risks of hydric erosion by altering soil bulk density, water infiltration rate, surface runoff and transport of POC and DOC and the attendant emission of GHGs into the atmosphere. Based on an experiment involving some sandy loam Aerosols in South Africa, Mchunu and Chaplot [36] assessed the effects of three levels of plant covers (100%, 25–50% and 0–5%) on loss of SOC under simulated rainfall. Plant C input into the soil profile and SOC stocks (g C/m²·yr.) in 0–0.02 m layer, respectively, were 1950 ± 180 and 300 ± 16.2 at 100% cover. In comparison, soil C input by plants decreased by 38.5 ± 3.5% at 25–50% and by 75.4 ± 6.9% at 0–5% cover. The losses of SOC by water erosion were 0.75 g C/m² at 100% cover and increased by 66% at 25–50% cover (3.76 ± 1.8 g C/m²) to 213% at 0–5% cover (7.08 ± 2.9 g C/m²). Furthermore, these losses of SOC were mostly in POC form, which being a labile fraction can be easily mineralized with the attendant emission of GHGs.

4.4. Wind Erosion and Gaseous Emissions

Similar to hydric erosion, wind erosion is also a selective process and involves preferential removal of light and colloidal fractions such as SOC and SIC constituents of the surface layer. The CER for windblown dust for rangeland in western Queensland, Australia is reported at 1 to 2 for sandy soil and 9 to 41 for clayey soil [37]. Dust may contain as much as 15–20% of SOC content. In addition to gaseous emissions and loss of soil productivity, the dust generated by wind erosion is a serious environmental hazard in arid and semi-arid climates. Among vegetative barriers (windbreaks and shelterbelts) and mulch farming systems (e.g., CA), enzyme-induced carbonate precipitation is also used to stabilize loose soil that creates dust. To stabilize loose soil and mitigate dust in Khuzestan, Iran, Baziar et al. [38] used a soybean enzyme, which is reportedly much easier and more economical to prepare than that from leek bean.

The problem of wind erosion is no longer confined to developing countries. Indeed, wind erosion is also a serious issue even in northern Germany, and on Histosols throughout the U.S. In northern Germany, Nерger et al. [13] reported that soils under maize monoculture are prone to severe wind erosion. Nерger and colleagues reported that SOC stock decreased by 49.4 and 2.44 kg/m² from 1999 to 2009. Total soil loss by wind erosion during 16 events was 48.9 kg/m², and suspended material had a CER of 2.96 (compared with CER of 0.98 for saltation). Soil erosion loss in a single event can be as much as 12.6 kg/m². Histosols,

with soil organic matter (SOM) content of >20% in the upper 80 cm profile, cover 21 M ha in 42 states of the U.S. [25]. Intensively cultivated Histosols are prone to wind erosion with adverse impact on productivity and degradation of the environment (i.e., soil, water, air, vegetation). Particle density of Histosols can be $\sim 1.6 \text{ Mg/m}^3$. Dust emissions are common on plowed/dry Histosols [25].

In Adam County, Washington, Feng and Sharrat [39] assessed the wind erosion hazard by using the Wind Erosion Prediction System (WEPS) at 14.25 Mg soil/ha·yr. (6.4 tn/ac·yr) and a severe PM10 (particulate matter at <10 micron m in diameter) loss. Wind erosion affects ~ 20 Mha of land in Iran, and also affects GHG emissions from soil. Based on experiments conducted at the Iran Research Institute of Forests and Ranges, Kamali et al. [40] reported that the highest rate of CO₂ emissions in July was 4.90 g CO₂/m²·d in severely eroded lands and the lowest in January of 0.086 g CO₂/m²·d in less eroded lands. Kamali and colleagues hypothesized that increase in erosion intensity aggravated CO₂ emission rates at severe erosion.

A study on soil losses from spring dust emissions in northern China by Song et al. [41] indicated that losses of SOC, total N and total P in spring were 0.985 ± 0.149 , 0.094 ± 0.014 and $0.089 \pm 0.013 \text{ Tg/yr.}$, respectively. However, the fate of SOC, TN, etc., is not known. In Australia, Chappell et al. [42] estimated SOC dust emission at 5083 Tg CO₂ eq/yr. for the country and 0.4 Tg CO₂ eq/yr. for agricultural soils. Chappell and colleagues concluded that omission of SOC dust emission from C cycling and C accounting is a significant source of uncertainty in the GCB. Thus, quantification of the release of CO₂ from SOC dust to the atmosphere and contribution of SOC deposition to downward C sinks is essential.

5. Soil Erosion and the Global Warming

Soil degradation by erosion is a serious global issue and it may be changed by the current and projected global warming. In arid or semi-arid areas, which may experience more rains with climate change, soil erosion could be less due to vegetation growth and more ground cover. The soil erosion hazard may also depend on the amount and intensity of the rain and the nature and dynamics of the emergent vegetation. The climate-induced dynamics may also differ among ecoregions (e.g., polar vs. tropical, and fire-prone vs. fire-free). Enzymatic and decomposition processes may be aggravated by the increase in temperature. In general, soil erosion hazard is linked to the fate of vegetation/land cover in relation to the climate change and may decrease with an increase in vegetation cover and the vice versa.

In agroecosystems or the managed landscape, soil erosion hazard may be aggravated during the 21st century [43] because of the anthropogenic climate change. Soil erosion is affected by interaction among a range of factors, such as climate (erosivity), soil (erodibility), slope (gradient, length, aspect, shape), land use (cropland, grazing land, plantations), and management (soil, crop, inputs). However, most of these parameters are also affected by the current and projected global warming (Figure 2). In the tropics and subtropics, soil erosion and global warming are mutually reinforcing processes. An increase in global warming in the tropics may increase erosion risks by both hydric and aeolian processes through an increase in climatic erosivity and soil erodibility, and both of these factors are aggravated by an increase in frequency and intensity of extreme events and the attendant decline in vegetation cover (Figure 2). However, the soil erosion hazard (hydric and aeolian) may decrease in temperate regions depending on the manner in which global warming affects the key parameters or factors of soil erosion. Based on a study in three catchments in Great Britain, Ciampalini et al. [44] observed that climatic parameters respond differently depending on their land use and management. Ciampalini and colleagues reported that an increase in rainfall increases soil erosion, but warmer temperatures in the U.K. may also lower erosion risks because of better vegetation growth. Warmer temperature can limit soil erosion risks by increasing primary productivity, improving rainfall interception, enhancing water infiltrability, and reducing soil erodibility. Ciampalini et al. observed that an increase in temperature in the U.K. may increase the rainfall thresholds to generate

soil loss, and, thus, soil erosion rates could decrease by 33% from 2070 to 2099 because of the negative-feedback mechanisms limiting soil loss by runoff. Modeling studies on soil erosion under sugar beet in Central Europe by Scholz et al. [45] also reported a decline in soil erosion risks for the period of 2070–2099. Scholz and colleagues observed that the intra-annual precipitation change resulted in a net decrease of rainfall amounts in erosion-sensitive months and an overall increase of rainfall in periods when the region is less prone to erosion. Consequently, the predicted average soil erosion losses under climate change declined in all tillage systems by 11% to 24%.

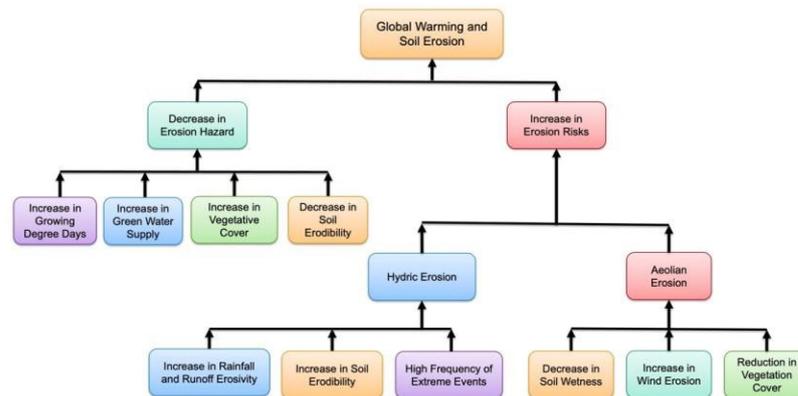


Figure 2. Effects of global warming on soil erosion risks which may decrease in high latitudes (boreal and temperate regions such as U.K.) but increase in the tropics (humid, subhumid, semi-arid and arid regions) and where the rainfall erosivity may increase (i.e., northeastern and northwestern U.S.) and wherever the frequency of extreme events may increase.

In contrast to the observation from the U.K., Segura et al. [10] reported an increase in vulnerability to erosion in some states of the U.S. (e.g., Ohio, Maryland, Indiana, Vermont, Illinois) with future increase in rainfall erosivity due to the projected climate change. Segura and colleagues based their conclusions on evaluation of changes in rainfall erosivity (R) from 1970 to 2090 across the U.S. under nine climate conditions. They observed that trends in R vary widely spatially with strong trends of increasing R in the northeastern and northwestern U.S., but weaker or inconsistent trends in the midwestern and southeastern U.S. An increase in soil erosion hazard in the U.S. with change in rainfall erosivity by 16–58% has also been reported by Nearing [12]. The response of the increase in risks of water and wind erosion by climate change may also occur in tropical and subtropical climates. Borrelli et al. [46] predicted a more vigorous hydrological cycle, which could increase global water erosion +30 to +60%. Thus, adoption of low erosion-producing practices can drastically reduce the risks of accelerated erosion [47]. Indeed, the adoption of conservation-effective measures can trump adverse impacts of climate change on soil erosion [48].

It is the breakdown of aggregates by high climate erosivity and weak aggregate strength (high erodibility) that exposes SOC/SIC to microbial processes and climatic parameters and aggravates the emission of GHGs. It is thus important to assess the risks of their mineralization by biotic and abiotic mechanisms. On the other hand, restoration of eroded soils/landscapes, via afforestation or establishment of any perennial vegetation cover, can create a positive soil/ecosystem C budget and set-in-motion recarbonization of the terrestrial biosphere through sequestration of atmospheric CO_2 as soil humus and secondary carbonates, etc. Sequestration of C in soil and biomass has a tremendous potential to create a substantial drawdown of atmospheric CO_2 through nature-based solutions [49].

6. Some Researchable Priorities

All other factors remaining equal, soil erosion is affected by land use and vegetation cover. Thus, it is a more serious problem in agricultural than in natural landscapes. However, land area under agroecosystems may increase during the 21st century and risks of soil erosion may be aggravated by an intensive land use. Intensification of the agroecosystems may be necessitated by the rising demands of the growing and increasingly affluent human population. Thus, adoption of conservation-effective measures and understanding of the fate of C being transported by erosional processes is important to promoting nature-positive agroecosystems. Therefore, there is a strong need to study the fate of erosion-induced transport of SOC in both agricultural and natural landscapes.

Key questions about future research priorities on soil erosion in agricultural landscapes include the following [50]: (i) understanding of the nexus between on-site and off-site effects of erosion, especially with regard to soil C budget and the emission of GHGs, (ii) evaluating site-specific adaptation of conservation-effective measures to reduce sediment connectivity and transport of C-laden sediments from hillslopes to eroded channels and eventually to depositional sites, (iii) identifying early signs of the on-set of a severe erosional process with a drastic impact on GCB and emission of erosion-induced GHGs (CO_2 , CH_4 , and N_2O) into the atmosphere, (iv) quantifying the regional and global impacts of accelerated erosion on GCB, (v) assessing in-depth the processes affecting erosion-induced emissions of GHGs and their impacts on GCB, and (vi) evaluating the ramifications and consequences of climate change and the magnitude of feedback related to erosion-induced emissions of GHGs.

Additional research is also needed on transport of SIC by aeolian processes and the fate of both SOC and SIC transported by wind erosion. There is a strong lack of scientific data on the impact of erosional processes (water and wind) on the transport of SIC and its fate during the redistribution and depositional phases, and its relationship with SOC. Based on a study of 62 erosional sites and 35 depositional sites of the Chinese Loess Plateau, Tong et al. [51] observed that the mean SOC and SIC contents in a depositional site (0–25 cm) increased by 24.4% and 15.4%, respectively. Further, SIC was significantly negatively correlated with SOC across all erosional sites. However, SIC was significantly positively correlated with SOC at all depositional sites.

The effects of complex and interacting processes must be assessed in order to understand and determine their impacts on the GCC. In this context, Kuhn et al. [52] argued that the eco-geomorphological perspective on soil C movement through the landscape can address any controversy with regard to erosion of C being a source or a sink. Soil erosion induced by water runoff [53] and by windblown dust [42] are important pathways of lateral transport of soil C in terrestrial landscapes. The ^{137}Cs technique has been used as a tracer to assess removal or deposition of soil along the toposequence [54,55]. Alewell et al. [56] also recommended the ^{137}Cs technique to assess emissions of gases from eroded SOC. However, the ^{137}Cs technique is based on radioactive fallouts, mainly during the 1950s and 1960s, when nuclear tests were being conducted. With half-life of around 30 years, there are only a few sites where the isotopes can still be detected. Thus, there is a strong need to develop/identify new techniques.

However, how these alluvial and aeolian processes affect the soil C budget at diverse erosional, redistributional, and depositional landform positions is critical to assessing the fate of soil C being transported. Understanding and quantifying the C gains and losses at different landscape positions in soils of fragile agroecosystems for both SOC and SIC is essential to obtaining reliable estimates of the GCB. Estimates of the effects of hydric erosion on the GCB range from a sink of 0.06–1.0 vs. the source of 0.27–1.14 Pg C/yr. [57]. Such a large range necessitates more watershed-based studies using some innovative techniques.

Soil structure is an important determinant of the stability of SOM and its MRT in terrestrial ecosystems [24]. It is the breakdown of structural units and reaggregation of the dispersed clay that determines the emission of GHGs. Thus, additional research is needed on strengthening the understanding of the dynamics of aggregation during and after the erosional processes at diverse landscape positions.

Soil erosion response to climate change is a high researchable priority. In addition to modelling, past climate change and its effects on soil processes (soil C dynamics) can also be studied from soil profile properties, especially those of the buried soil horizon.

7. Summary

Accelerated erosion, both hydric and aeolian, involves selective removal of soil C (both SOC and SIC) along with clay and fine silt fractions. On-site, the eroded soil becomes progressively depleted of the light and colloidal fractions, leaving coarse material (gravels, sand, coarse silt) behind. Because CER of the eroded material is high, erosion has severe adverse effects on the functionality of eroding landscapes. Globally, soil erosion by water and wind transports a large amount of C, but the fate of C being transported is governed by complex pedotransformative processes, and its stability against decomposition is governed by site-specific biophysical conditions and hydrothermal regimes. Therefore, soil erosion not only affects soil properties and processes on-site, but also along the landscape over which the water or windborne sediments are being redistributed and the depressional sites or barriers where the sediments are being deposited. Whereas a proportion of soil C being buried at depositional sites or into the water bodies may be protected against mineralization and may even be encapsulated within reformed aggregates, most of the C redistributed over the landscape and part of those carried into depositional sites is subject to decomposition by both biotic and abiotic processes. Off-site, a large proportion of transported C is released into the atmosphere as CO₂, CH₄, and N₂O, depending on the biophysical and hydrothermal regimes. In total, the accelerated soil erosion has a drastic impact on the GCB. How much of soil C (SOC and SIC) is being removed by erosional processes and how it is being mineralized en route to depositional sites has significant environmental consequences, especially as a source or sink of GHGs, depending on the site-specific conditions. Nonetheless, the C displaced by erosional processes is not accounted for in the current processes of compiling the GCB. Not only should the impact of erosional process on the global C cycle be accounted for, but its impact as a source of GHGs should be considered in evaluating the economic and environmental impacts of erosion. The net effects of the erosional processes as a source or sink depend on a multitude of site-specific and highly interactive factors. Despite the on-site and off-site effects on productivity and off-site effects on the global C cycle and emission of GHGs, a prudent strategy is to minimize the risks of accelerated soil erosion by adoption of the conservation-effective measures.

The global menace of soil erosion by water and wind may be aggravated by global warming, especially in the tropics and subtropics, and with agricultural intensification of agroecosystems. Thus, there is a strong need to:

- i. quantify the global magnitude of soil C (SOC, SIC) being transported by the erosional processes in relation to land use, farming/cropping systems, landscape characteristics, soil physical/chemical properties, and the dynamics of hydrothermal regimes along the landscape and at the depositional sites;
- ii. account for the global amount of C transported by hydric and aeolian processes in the GCB;
- iii. evaluate the site-specific conditions which make the eroded soil C a source (emission of CO₂, CH₄, and N₂O) of GHGs or a sink if some C is buried, reaggregated, and taken out of circulation;
- iv. identify and implement site-specific conservation-effective management practices, which minimize the risks of soil erosion by water, wind, and other anthropogenic activities.

Funding: This research received no external funding.

Institutional Review Board Statement: Not Applicable.

Informed Consent Statement: Not Applicable.

Data Availability Statement: The data quoted in the manuscript are available in the specific reference number listed at the end of each statement or the paragraph.

Conflicts of Interest: The author declares no conflict of interest.

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