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SOIL
EROSION
the greatest
challenge
for sustainable
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GLOBAL SOIL
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SOIL EROSION: the greatest challenge for sustainable soil management

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Contents

ACRONYMS

FRNs | Fallout radionuclides
GAEC | Good Agricultural and Environmental Condition
GLASOD | Global Assessment of Land Degradation
GSP | Global Soil Partnership (of FAO)
IPBES | Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
ITPS | Intergovernmental Technical Panel on Soils (of FAO)
LIDAR | ground-based light detection and range
MODIS | Moderate Resolution Imaging Spectroradiometer
MUSLE | Modified Universal Soil Loss Equation
NCP | Nature's Contributions to People
PES | Payment for Ecosystem Services
RUSLE | Revised Universal Soil Loss Equation
RWEQ | Revised Wind Erosion Equation
RWSC | Revised World Soil Charter
SLEMSA | Soil Loss Estimation Model for South Africa
SOC | Soil Organic Carbon
SOM | Soil Organic Matter
SSM | Sustainable Soil Management
SWAT | Soil and Water Assessment Tool
SWSR | Status of the World's Soil Resources (Report)
UNCCD | United Nations Convention to Combat Desertification
USDA | United States Department of Agriculture
USLE | Universal Soil Loss Equation
VGSSM | Voluntary Guidelines for Sustainable Soil Management
WEAM | Wind Erosion Assessment Model
WEPS | Wind Erosion Predictions System
WEQ | Wind Erosion Equation
WOCAT | World Overview of Conservation Approaches and Technologies

GLOSSARY

Conservation agriculture: a system of agronomic practices that include reduced tillage or no-till, permanent organic cover by retaining crop residue, and crop rotations, including cover crops (Palm et al. 2014).

Desertification: land degradation in arid, semi-arid and dry subhumid areas resulting from various factors, including climatic variations and human activities (UN)

Erodibility: a measure of the soil's susceptibility to detachment and transport by the agents of erosion (Lal and Elliot, 1994).

Dynamic replacement: The replacement of soil organic carbon lost by erosion by new carbon input from photosynthate from plants (Hardin *et al.*, 1999)

Ecosystem services: The capacity of natural processes and components to provide goods and services that satisfy human needs, directly or indirectly (UN).

Fallout radionuclides: A radioactive isotope deposited from the atmosphere onto the soil. Ceium-137 has been widely used as a tracer for soil erosion studies (Mabit *et al.*, 2018).

Fluvial transport: Transport of sediment within a stream or river channel.

Gully erosion: Soil detachment and transport by water flowing in channels greater than 0.3 m deep (Castillo and Gomez, 2016). Less formally, gullies are eroded channels that cannot be filled in by normal tillage operations.

Interrill erosion: Detachment and transport of soil by raindrops and overland flow; alos called sheet erosion (Lal and Elliot, 1994).

No-till: A system of planting (seeding) crops into untilled soil by opening a narrow slot, trench or band only of sufficient width and depth to obtain proper seed coverage. No other soil tillage is done (Derpsch *et al.* 2010).

Rainfall erosivity: The ability of rainfall to cause soil detachment and transport. Rainfall erosivity is a function of the direct raindrop impact and the runoff that rainfall generates (Lal and Elliot, 1994).

Rill erosion: Soil detachment and transport by water flowing in channels less than 0.3 m deep (Castillo and Gomez, 2016). Less formally, rills are eroded channels that can be filled in by normal tillage operations.

Runoff: Rainfall or snowmelt that fails to infiltrate into the soil and which flows over the soil surface.

Saltation: Sediment that bounces along the soil surface (wind erosion) or channel bed (rill, gully, or fluvial transport) during transport.

Sediment: Soil in transport by wind or water erosion processes.

Sedimentation: Deposition of sediment from flowing water (in channels or floodplains) or standing water (in wetlands, lakes, or oceans).

Sheet erosion: See interrill erosion.

Soil horizon: A layer within the soil that has a distinct morphology (e.g. colour, structure, texture (percent sand, silt, clay). In formal taxonomic systems horizons are assigned letters (e.g. A, B, C) based on a specified range of morphological and other properties.

Soil erosion: The net long-term balance of all processes that detach soil and move it from its original location

Soil particles: Soil mineral particles are commonly split by size into clay (< 2 µm), silt 2-5 µm), and sand (0.05 mm to 2 mm). Loam describes a particle size distribution with roughly equal amounts of sand, silt, and clay.

Suspension: Sediment transported entirely within the flowing water or the wind stream (i.e., that does not contact the surface during transport).

Tillage erosion: The detachment, transport, and resultant displacement of soil by tillage operations (Govers, Lobb, and Quine, 1999).

Tolerable soil loss: (i) The maximum average annual soil loss that will allow continuous cropping and maintain soil productivity without requiring additional management inputs. (ii) The maximum soil erosion loss that is offset by the theoretical maximum rate of soil development which will maintain an equilibrium between soil losses and gains (SSSA, 2001).

Topsoil: A non-scientific term used to describe organically enriched surface soil layers.

EXECUTIVE SUMMARY

Despite almost a century of research and extension efforts, soil erosion by water, wind and tillage continues to be the greatest threat to soil health and soil ecosystem services in many regions of the world. Our understanding of the physical processes of erosion and the controls on those processes has been firmly established. Nevertheless, some elements remain controversial. It is often these controversial questions that hamper efforts to implement sound erosion control measures in many areas of the world.

Regional and global estimates of soil loss rates due to erosion differ substantially depending on the method used to derive them. Generally, estimates of mean annual soil loss from field plots are substantially higher (8 to almost 50 t ha^{-1} yr^{-1}) than those from regional and global models (2 to 4 t ha^{-1} yr^{-1}). Any estimate of erosion must also be placed in the context of the acceptable or tolerable rate of loss. Rates of tolerable soil loss calculated using soil production rates range from 0.2 to 2.2 t ha^{-1} yr^{-1} and tolerable rates based on maintenance of crop production range from approximately 1 to 11 t ha^{-1} yr^{-1}. The ranges for both soil loss and tolerable soil loss demonstrate the need for site-specific estimates to reflect the different sensitivity of soils to removal of surface soil through erosion.

According to the definition of Sustainable Soil Management adopted by the Food and Agriculture Organization of the United Nations (FAO) in 2015, the definition of tolerable soil loss should also consider the impact of soil erosion on ecosystem services provided by soil, such as regulation of water and air quality.

The impact of erosion on crop production has been estimated at a 0.4 percent reduction in global crop yields per year due to erosion. Modelling of the impact of this yield loss on the overall agricultural economy (using general equilibrium models) suggests a lower overall impact as land prices and the labour force adjust to the changes in soil productivity. A recent study from Malawi suggests that the negative impact of soil and nutrient loss falls most heavily on the poorest members of society and on households headed by women, and this result is supported by qualitative information from many other studies.

Recently, regional and global models for water erosion have begun to published, along with initial efforts at global wind erosion modelling. The results of these models can be compared with site-specific studies and with anecdotal information in order to identify global hotspots of erosion. In many cases, the level of agreement across studies is strong. These hotspots

should be the priority for targeted soil control measures. It is also essential that the modelling results be validated through structured field assessments of erosion.

There are many examples of successful implementation of soil erosion control measures. The widespread adoption of reduced tillage and no-till practices has significantly reduced wind and water erosion in many drier regions, but significant impediments exist to its adoption in more humid regions. In general, measures using vegetation cover in order to reduce erosion (through enhanced residue cover, cross-slope plantings of erosion resistant grasses, or planting of shrubs or trees to reduce wind erosion) appear to be more readily adopted than are engineered, structural measures such as terraces.

Issues related to soil governance are the most significant impediments to the adoption of erosion control measures. Two overarching issues have been identified. In the first place, many of the impacts of erosion occur off-site, and there is no direct benefit for the soil user to implement control measures that minimize these off-site impacts. Second, the long time period required for many erosion control measures to have a clear beneficial effect limits their adoption, especially for soil users who do not have secure tenure rights to their land. The successful implementation of erosion control measures shows that these impediments can be overcome. For this to happen, the decision making factors that lead to successful adoption by soil users needs to be better understood and adapted to diverse conditions.


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1 WHAT IS SOIL EROSION?

Soil erosion continues to be a major threat in many regions of the world despite decades of focused scientific research and societal concern. In the 2015 *Status of the World's Soil Resources* Report (FAO and ITPS, 2015), soil erosion was judged to be the number one threat to soil functions in five of seven regions (Africa, Asia, Latin America, Near East and North Africa, and North America); in the first four of those regions, the trend for erosion was deteriorating. Only in Europe, North America and the Southwest Pacific was the trend in erosion deemed to be improving.

The volume of research on soil erosion has continued to grow over time–according to the *Web of Science* database, there was more literature published on soil erosion in the last three years (7 348 articles between 2016 and 2018) than in all of the twentieth century (5 698 articles between 1931 and 1999) (Web of Science, 2019). This tremendous volume of research has firmly established many of the key elements of erosional processes and their control that now provide the base for further erosion research; however, some important aspects remain controversial or little understood.

The purpose of this volume is both to review the well-accepted foundation of information about erosion and to highlight the areas where agreement is less firmly established. An overarching goal is to examine why, after decades of research and program implementation, soil erosion remains the number one threat to soil functioning in so many areas of the world. Boardman (2006) posed several key questions about erosion science, and these questions remain relevant for erosion research today and into the future.

1.1 Types of erosion: water, wind, and tillage

Soil erosion is studied in different disciplines and from different perspectives. For the purposes of this volume, soil erosion is defined as the net long-term balance of all processes that detach soil and move it from its original location (Lupia-Palmieri, 2004). Soil erosion is a naturally occurring geomorphic process, but human use of the soil typically results in rates of soil detachment and transport that are many times the naturally occurring rates. This accelerated or human-induced erosion is the focus of this volume.

Water, wind, and tillage are the three main types of erosion. Each involves distinct processes that detach and transport soil; hence each also requires

different approaches to decrease associated rates of erosion. In some regions of the world all three types of erosion operate simultaneously in the landscape, and the identification of the processes occurring at a location is a key element of erosion control. Although these three types of erosion will form the main focus of this volume, other forms of erosion may also be of importance. Poesen (2018) includes soil erosion by land leveling and soil quarrying, by crop harvesting, and by explosion cratering and trench digging as other sources of erosion. As well, soil erosion by mass wasting - through slumping, debris flows and other means - is of major importance in particular landscapes.

Water erosion has been the most widely studied of the three types of erosion, and is arguably the one that affects the greatest land area. In water erosion, the detachment of soil from the soil mass occurs in two ways: from the effects of raindrop splash on the soil surface, and from forces exerted by water flowing across the surface (*runoff*). Transport of the detached soil by flowing water first occurs in thin sheets of runoff flowing over the surface (*sheet erosion*). Often the surface runoff becomes concentrated in small channels (*rill erosion*) or deeper incisions (*gully erosion*); in both of these types of channels the erosive power of the flow is greatly magnified. The rills and gullies resulting from water erosion are some of the most visible signs of erosion operating in the landscape. In some cases, the soil in the flowing water (*sediment*) settles out of the water when either the depth or velocity of runoff is reduced–for example, when the flowing water contacts a vegetative barrier. This leads to the deposition of eroded soil. In many other cases, however, the runoff and sediment is carried to a stream system (*fluvial transport*) and is removed entirely from the landscape.

Wind erosion occurs primarily in arid and semi-arid environments and is the major form of erosion in, for example, the Near East and North Africa region (FAO and ITPS, 2015). In wind erosion, the detachment of the soil occurs because of the forces exerted by the wind on the soil surface, and because of the effect of detached soil bouncing off the soil surface downwind of the point of initial detachment (*saltation*). Transport of the soil occurs within the wind stream, and the size of the grains being transported largely dictates the transport distance. In some cases the transport distance may be hundreds of kilometres from the point of detachment. The devastating wind erosion events in western North America during the 1930s led to considerable erosion research and to the establishment of agencies such as the Soil Conservation Service in the United States of America and the Prairie Farm Rehabilitation Administration in Canada, both of which were established in 1935.

The importance of tillage erosion was only recognized by soil scientists in the 1990s, and it remains far less well known than the other two types of erosion. In tillage erosion, both the detachment and transport of soil are by tillage implements such as the mouldboard plough. Tillage erosion causes the net downslope movement of soil due to these tillage operations and (unlike water and wind erosion) is difficult to observe visually when it is occurring. Tillage erosion causes the thinning of soils on upper slope areas and can result in over-thickened depositional soils in lower slope positions. The deposited soil in lower slope positions may also be vulnerable to further transport by water erosion.

1.2 Rates of soil erosion

One key question posed by Boardman (2006) is "How serious is erosion?" The first part of the answer to this question involves establishing typical rates of erosion. A value for the rate of erosion alone is, however, of limited use without a corresponding value for an "acceptable" or "tolerable" rate of erosion. Erosion is a natural geological process and it is impossible to stop; instead, the goal is to manage human impacts on the soil so that the rate of erosion is within an acceptable range.

The different disciplines that focus on soil erosion often use different units to report results. In soil science, the norm is to report net soil change in units of mass per area per time–most commonly as tonnes per hectare per year (t ha^{-1} yr^{1}). By convention net soil loss is reported as a negative value, and net soil gain (through deposition) as a positive value. A major issue in soil erosion research is that the rates of erosion measured in different studies are very dependent of the scale of the study. For example, rates measured from small experimental plots (10^{-4} to 10^{2} m^{2}) will be very different from those measured on complete hillslopes or catchments (10^{4} to 10^{9} m^{2}) (Garcia-Ruiz *et al.*, 2017).

Units of mass per area per time are difficult, however, to relate directly to the soil itself; often they are converted into equivalent soil thicknesses. This conversion requires use of a value for soil bulk density (that is, the mass of soil in a specified volume of soil, reported as mass per volume such as g cm^{-3} or kg m^{-3}). Montgomery (2007) uses a standard bulk density of 1 200 kg m^{-3} in his widely cited paper. Using this value, a soil loss of 1 t ha^{-1} yr^{1} is equivalent to a value of soil lowering of 0.08 mm.

Table 1: Compendium of mean depths of soil lowering and mean net soil loss from global and regional studies of soil erosion

Land use	Area	Method	Depth of soil lowering	Net soil loss	Reference
			(mm y^{-1})	t ha^{-1} yr^{-1}	
Conventional agriculture	Global	Field-plot compilation	3.9	49. [b]	Montgomery, 2007
Conservation agriculture	Global		0.12	1.6[b]	
Native vegetation	Global		0.05	0.66[b]	
Geological	Global		0.17	2.2[b]	
Cropland	Global	Field plot compilation	1.0 to 1.2[a]	12 to 15	den Biggelaar *et al.*, 2003
Cropland	Western Europe	Field plot and modelling	0.29[a]	3.6	Cerdan *et al*, 2010
All erosion-prone land	Western Europe	Modelling	0.18[a]	2.2	Panagos *et al.*, 2015
Arable land			0.21[a]	2.7	
Cropland	Global	Plot data	0.60	7.5[b]	Wilkinson and McElroy, 2007
All land	Global	Modelling	0.22[a]	2.8	Borrelli *et al*., 2018
Croplands	Global		1.0[a]	13.	
Forests	Global		0.01[a]	0.16	
Cropland	Europe	Modelling	0.31[a]	3.9	Van Oost, Cerdan and Quine, 2009

Land use	Area	Method	Depth of soil lowering	Net soil loss	Reference
Cropland	Mean of seven sites, England	Field –based assessment	0.01[a]	0.15	Evans, 2013
Tillage					
Cropland	Europe	Modelling	0.26	3.3	Van Oost, Cerdan and Quine, 2009
Combined Tillage and Water Erosion					
Cropland	Global	Modelling	0.84[a]	11.	Doetterl, Van Oost and Six, 2012
Cropland	Pasture		0.14[a]	1.7	
Wind					
Dryland agriculture	Australia	Modelling	0.02[a]	0.193	Chappell *et al.*, 2013
Irrigated agriculture	Australia		0.01[a]	0.167	
Rangeland	Australia		0.03[a]	0.359	

[a] *Rates of soil lowering calculated from net soil loss using bulk density of 1 200 kg m^{-3} (Montgomery, 2007)*
[b] *Rates of soil loss calculated from rates of soil lowering using bulk density of 1 200 kg m^{-3} (Montgomery, 2007)*

One observation that can be made from Table 1 is that there is a considerable range of erosion estimates in the literature, and that these estimates differ in part because of the method used to generate the estimate. This is discussed in more detail below.

The second observation is that at this time we cannot answer the question "How serious is erosion?" with an agreed-upon global value for erosion, but can only present a range of values. Montgomery's (2007) mean value of 3.9 mm yr^{-1} is very high and his median value (1.5 mm yr^{-1}), while still higher than the other values presented, is closer to the estimates of den Biggelaar *et al.* (2003), which are also based on erosion plot data. Most of the other values presented are in the range of 0.2 to 0.6 mm yr^{-1}.

1.3 Tolerable soil loss

To assess the seriousness of erosion we also must establish a threshold between acceptable and non-acceptable levels of erosion. This question can be placed in the larger context of Sustainable Soil Management (FAO, 2017). According to the *Revised World Soil Charter* (FAO, 2015), soil management is sustainable if the ecosystem services provided by the soil are maintained or enhanced without significantly impairing either the soil functions that enable those services or biodiversity. Hence, an acceptable or tolerable level of soil erosion is one that maintains ecosystem services (such as the provision of food and fibre) without degrading the soil's capacity to deliver these services in the future.

A seemingly sensible approach to establishing T_{sl} would be to relate it to the depth of topsoil, since topsoil is probably the soil term most readily understood by the general public. Unfortunately, topsoil is not a scientific concept. In soil classification the uppermost horizon of a mineral soil is normally denoted with an uppercase A; this would correspond most closely to the common understanding of *topsoil*. In agricultural settings ploughing of the A horizon occurs, and for classification purposes the ploughed A horizon is assigned a lowercase p. Unfortunately as erosion lowers the surface and as deeper soil layers (such as the E, B, or the lowermost C horizon) are incorporated into the ploughed layer, the layer retains an Ap designation; moreover this Ap layer cannot be thinner than the depth of ploughing and hence will retain a consistent thickness through time even as erosion removes surface soil (Figure 1). Hence the layer most equivalent to the *topsoil,* the Ap horizon, could be an organic-matter-rich, highly fertile layer or it could be the nutrient-poor remnant of a truncated soil profile and hence is of little use in setting T_{sl} values.

Two main approaches have been used to establish values for tolerable soil loss (T_{sl}): (1) a value that is able to maintain the dynamic equilibrium of soil quantity (mass or volume) at a site; or (2) a value that maintains the biomass production function of the soil (Verheijen *et al.*, 2009; Di Stefano and Ferro, 2016; Duan *et al.*, 2016).

The first main approach to establishing a value for T_{sl} compares the rate of soil loss to the rate of the creation of new soil from solid earth materials. The most extensive review of rates of soil production was by Montgomery (2007), who presents a mean value of 0.173 mm yr^{-1} (2.2 t ha^{-1} yr^{-1}) across 188 papers. This mean rate is a small fraction of his reported mean rate of soil lowering (3.9 mm yr^{-1}). Verheijen *et al.* (2009) use European data on soil formation to calculate a tolerable soil loss for Europe of between 0.3 to 1.4 t ha^{-1} yr^{-1} (approximately 0.02 to 0.11 mm yr^{-1}). Using soil formation rates for

Australia, Bui, Hancock, and Wilkinson (2011) calculate a T_{sl} of 0.20 t ha^{-1} yr^{-1} (approximately 0.015 mm yr^{-1}).

The use of soil production functions to establish a threshold for tolerable soil loss can be criticized because the rate of soil production at the soil/bedrock interface is less relevant for soils that develop in thick mantles of unconsolidated parent materials such as loess, lacustrine or glacial sediments. Wilkinson and Humphreys (2005) emphasize the importance of soil mixing by organisms (i.e., bioturbation) in creating soil horizons (especially organically enriched surface horizons) at much faster rates than the absolute rate of soil production at the soil-bedrock interface. Hence soil production rates for determining tolerable soil loss rates are most relevant where a relatively thin soil overlies rock.

The second approach to calculate T_{sl}, the rate required to maintain biomass production, has a long history in erosion studies. The most widely used definition is from the United States Department of Agriculture (USDA), which defined T_{sl} as "the maximum level of soil erosion that will permit a high level of crop production to be sustained economically and indefinitely" (Wischmeier and Smith, 1978). Work by the USDA established T_{sl} values between 4.5 and 11.2 t ha^{-1} yr^{-1}. Later work by the European Environment Agency set a range for T_{sl} between 1 t ha^{-1} yr^{-1} for shallow sandy soils and 5 t ha^{-1} yr^{-1} on deeper, well-developed soils. For Australia, Bui, Hancock, and Wilkinson (2011) calculate a T_{sl} value of 0.85 t ha^{-1} yr^{-1} (0.065 mm yr^{-1}) to maintain crop production at 75% of maximum over a 200-year time frame.

Finally, the major approaches to establishing T_{sl} do not take into account the off-site effects of erosion on air or water quality and quantity, which the FAO definition of Sustainable Soil Management would require. Establishing the link between soil erosion and off-site impacts can be difficult (Duan *et al.* 2016), but it will be required if the full impact of erosion is to be assessed. Verheijen *et al.* (2009) suggest broadening of the definition to include other functions provided by the soil and Bui, Hancock, and Wilkinson (2011) discuss a T_{sl} value to maintain water quality. In light of the definition of Sustainable Soil Management codified in the *Revised World Soil Charter* (FAO, 2015), the definition of T_{sl} suggested by Verheijen *et al.* 2009 could be expanded as follows:

> *Tolerable Soil Loss is any mean cumulative (all erosion types combined) soil erosion rate at which significant deterioration of soil functions and ecosystem services provided by the soil does not occur.*

1.4 Erosion, soil functions and the provision of ecosystem services

The second aspect of assessing the seriousness of erosion involves the impacts it has both in-field and off-site, and what the economic costs of these effects are (Boardman, 2006). It is also important, however, to examine impacts that have not been fully costed or that indeed cannot be assigned an economic value. The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) has recently advocated for moving beyond a natural science- and economics-based evaluation framework to a broader one that assesses nature's contributions to people (NCP) from many perspectives. They suggest that the NCP approach is likelier to engage with local practitioners, including indigenous peoples (Diaz *et al.*, 2018). Many of the functions that the soil provides are affected by soil erosion (Table 2).

Table 2: The effects of soil erosion on the main soil functions that enable ecosystem services provided by soil.

Ecosystem Service	Soil Functions	Effect of Erosion
Supporting services: services that are necessary for the production of all other ecosystem services; their impacts on people are often indirect or occur over a very long time		
Primary production	Medium for seed germination and root growth Supply of nutrients and water for plants	Reduction of optimum rooting zone for extraction of water and nutrients from soil
Nutrient cycling	Retention and release of nutrients on charged surfaces	Loss of charged organic materials from surface soil horizon
Regulating services: benefits obtained from the regulation of ecosystem processes		
Water quality regulation	Retention, filtering and buffering of substances in soil water	Transfer of sediment and sediment-bound contaminants to water bodies
Water supply regulation	Regulation of water infiltration into soil and water flow within the soil	Decrease in surface infiltration and water-holding capacity of soil
Air quality regulation	Regulation of particulate content of atmosphere	Transfer of particulates to atmosphere
Erosion regulation	Retention of soil on the land surface	
Climate regulation	Regulation of CO_2, N_2O, and CH_4 emissions	Lateral transfer of soil organic carbon (SOC) in landscape and possible enhanced CO_2 emissions

Provisioning Services: products ('goods') obtained from ecosystems of direct benefit to people		
Food, fibre, and fuel supply	Providing water, nutrients, and physical support for growth of plants for human and animal use	Degradation of water and nutrient supply and decrease of depth of suitable rooting medium

Many of these services provided by soil are of immediate relevance to the Sustainable Development Goals (SDGs) set by the United Nations (Weigeit *et al.*, 2015). The relevance is clearest for SDG 15 on halting biodiversity loss and land degradation and striving to achieve a land-degradation neutral world but soil-mediated regulating and provisioning services such as food production and water purification cut across many other of the SDGs as well (Weigeit *et al.*, 2015). For example, soil erosion is inherently included in SDG 2 (End hunger, achieve food security and improved nutrition and promote sustainable agriculture). One indicators for SDG 2 is 2.4.1 "the percentage of agricultural land under productive and sustainable agriculture" and by definition land use is unsustainable if unacceptable rates of erosion are occurring.

1.4.1 Erosional effects on soil productivity and crop yields

Erosion has three primary effects on crop growth and yield: removal of the fertile surface soil horizon, incorporation of denser subsoil into the surface layer, and a possible decrease in the rooting zone of the soil (Van Oost and Bakker, 2012)

All three types of erosion lead to the incremental removal of surface soil material (Figure 1). In most soils the surface soil layer (the A horizon) has a higher content of soil organic matter (SOM) than do lower horizons. It is very well established that organic matter is highly beneficial as a source of nutrients for crop growth, and as a medium that enhances the formation of stable soil aggregates and hence increases the porosity of the soil. The higher porosity associated with the organically enriched layer facilitates both root penetration through the soil mass and the flow of water into and within the soil.

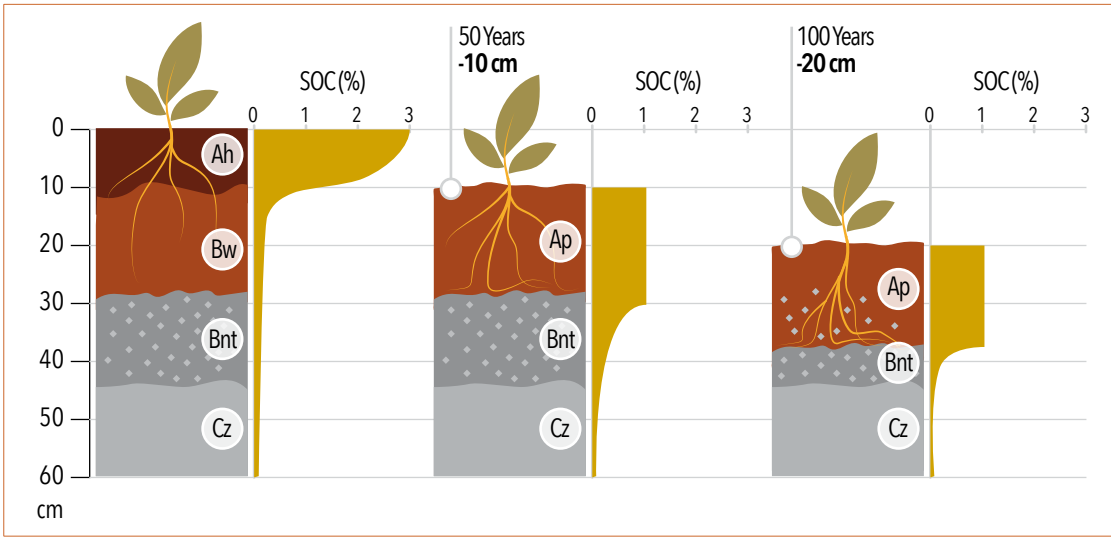

Figure 1 The effect of erosion on soil horizonation and the depth function of soil organic carbon. The Ah horizon is the undisturbed SOM-rich layer; the Ap horizon is the ploughed surface horizon. The Bnt horizon is growth-limiting due to high clay and sodium contents and the Cz horizon has high soluble salts levels. Erosion rate is 0.2 cm per year.

Each erosional event removes an increment of this beneficial layer, and the next time ploughing occurs an equivalent increment of the soil material below the plough layer is incorporated into the plough layer. If the newly incorporated layer is lower in SOM, there is a progressive dilution of SOM content and a reduction in the benefits of SOM (Figure 1). Furthermore, in some soils a clay-enriched layer occurs under the A horizon, and incorporation of the clayey material into the surface layer creates a denser, cloddy layer and a poorer seedbed for germination of crops. The loss of surface material leads to a decrease in the nutrient-supplying power of the soil and a decrease in nutrient-holding power; the latter effect is most pronounced in sandy soils. The loss of nutrient-supplying power can be replaced by increasing use of fertilizer, but clearly there are economic costs to increased fertilizer use as well as environmental costs such as the potential for agrochemical contamination of surface waters.

The effect of erosion on the rooting depth of soils is most serious in soils where growth-limiting subsoils occur (Table 3). As erosion removes the surface layer of soil, the thickness of soil between the surface and the growth-limiting layer decreases (Figure 1); this may limit the rooting development of crops. Once the plough layer reaches the growth-limiting layer, it becomes incorporated into the plough layer and significant yield reductions can occur (Larson and Pierce, 1994; Pennock, 1997). Unlike the case of nutrient replacement by fertilizer, the effects of subsoil incorporation on yields are largely irreversible on human timescales.

Table 3 Soil horizons and soil orders that are particularly susceptible to surface soil loss through erosion (adapted from Larson and Pierce, 1994 and Pennock, 1997).

Horizon	Characteristics and Constraints	Soil Orders commonly associated with the horizon
Rock or petrocalcic horizon within 30 cm of surface	Limited or no rooting; irreversible loss of soil	Leptosols
Salic	High salt concentration	Solonchaks
Fragic or Duric	Layers cemented by iron, aluminum or silica	Podzols, Luvisols, Ultisols
Natric	High sodium concentration and dense structure	Solonetz
Plinthic	High iron and aluminum oxide content; hardens upon drying; high resistance to root penetration	Ferralsols
Argic	Increase in clay content relative to overlying soil; increase in root penetration resistance	Luvisols, Nitisols, Acrisols
Ferralic	High possible Al^{3+} concentrations in low pH conditions	Ferralsols
Spodic	High possible Al^{3+} or metal level concentrations in low pH conditions	Podzols

The main effect of these erosion-induced changes to soil productivity is on crop yields for food, fibre, and fuel. The effect of erosion on crop yields was studied in detail in the second half of the twentieth century, and several comprehensive summaries of the research have been published. The *Status of the World's Soil Resources* report drew upon four post-2000 syntheses (Den Biggelaar *et al.*, 2003; Bakker, Govers, and Rounsevell, 2004; Scherr, 2003; and Crosson, 2003; Table 7.1 in SWSR). The estimates of annual crop loss due to erosion range from 0.1 to 0.4 percent, with two studies estimating 0.3 percent annual yield reduction. A subsequent analysis for western Europe using the Revised Universal Soil Loss Equation (RUSLE) model (discussed below) estimates yield reductions of 0.4 percent annually for agricultural land that suffers from severe erosion (Panagos *et al.* 2016) based on a survey of 16 studies cited in their article (including three of those cited above).

1.4.2 Economic and societal effects of erosion-induced yield reductions

Assigning an economic value to the yield reduction caused by erosion is complex. Four approaches have been used in the recent literature. In the *production function approach*, the first step is to determine the reduction in crop yield caused by erosion. The second step is to calculate an economic value for the lost yield, typically by multiplying the crop yield by the unit market price of the commodity. A second approach is the *replacement cost method,* where the value of the fertilizer required to offset the loss of nutrients through erosion is calculated. Although the replacement cost approach is relatively easy to use, Adhikari and Nadella (2011) contend that it is less reliable than the production function approach because fertilizer is only a partial proxy for the total effect of erosion on crop productivity. For example, the physical degradation associated with erosion is not accounted for in the replacement cost approach. The third approach, *cost-benefit analysis,* is normally used to assess the economic benefit of conservation measures such as terracing or buffer strips. Finally, some authors have used the cost of purchasing soil as their basis for analyzing the cost of erosion.

In a recent example of a cost analysis, Panagos *et al.* (2018) use a production function as input into a macroeconomic model to assess the cost represented by soil erosion loss in the agricultural sector in the European Union. Using the annual productivity loss value of 0.4 percent for approximately 12 million hectares of agricultural land that suffer from severe erosion, they estimate an annual cost for the lost production at EUR 1.25 billion for the reference year of 2010. The actual cost of the loss in the agricultural sector is only EUR 300 million (a reduction of 0.12 percent), and the loss in GDP at EUR 155 million. The agricultural sector loss is less than the productivity loss due to a) substitution of more labour and capital input for the productivity loss and b) the enhanced competitiveness of countries that experience a less-than-average decline in productivity. The analysis in the paper illustrates the tremendous complexity of modelling the effects of erosion on the broader agricultural sector, and the authors caution that the results should be "handled with care."

Recently the Food and Agriculture Organization, United Nations Development Program, and United Nations Environment Program completed an integrated soil loss and economic evaluation assessment for Malawi. For the soil loss assessment the authors (Vargas and Omuto, 2016) use the Soil Loss Estimation Model for South Africa (SLEMSA), an empirical erosion assessment developed for sheet and rill erosion for Zimbabwean conditions. The SLEMSA uses three input factor submodels (for crop ratio, soil loss from bare soil, and topography) to calculate an annual soil loss rate. The main data sources used were climate data, digital elevation models,

topographic maps, and land cover maps from remote sensing. The project included a component for soil loss field validation, including training of local staff in field assessment.

The SLEMSA results show that that the majority of the country has low average annual soil loss rates (0.9 to 10 t ha^{-1} yr^{-1}). The nine districts in the Central region all have annual soil loss rates of between 0.9 and 6.4 t ha^{-1} yr^{-1}. Several areas in the northern region (associated with the Rift Valley escarpments) have average annual loss rates between 11.2 and 19.8 t ha^{-1} yr^{-1}. There are also scattered high altitude areas in the south that have annual rates greater than 10 t ha^{-1} yr^{-1}. Generally the areas with high rates of soil loss have structurally unstable and shallow soils, steep slopes, high erosive rainfall, and sparse vegetation cover (Figure 2).

Figure 2 : Mr Lemuta Naisikie Lazier stands within a deep gully that separates the village of Landi Kenya in half. The upper area has the other community leaders standing atop the remains of the bridge which is about to collapse.

A separate economic study of the soil and nutrient losses in Malawi (Asfaw *et al.*, 2018) assessed both the direct costs of soil loss (through reduced crop yields and loss of nitrogen, phosphorus, and potassium) and the reaction of market agents (such as firms, farmers, and governments) to the direct losses using a general equilibrium model. The results of the modeling of direct costs predicts that a 10 percent increase in soil loss would lead to monetary losses of about 0.26 percent of Malawian GDP and 0.42 percent of total agricultural production value; a 50% increase in soil loss yields loss of about 1.28 percent of GDP and 2.1 percent of total agricultural production value. The GDP losses decrease to between 0.10 percent to 0.55 percent when the response of market agents is considered using the general equilibrium model. Importantly the modelling predicts that the impact of the productivity decline is unequal - the greatest negative consequences are concentrated at the poorest end of the income distribution and in female headed households. Finally the analysis considered the effects of various conservation measures and found that cross-contour rows of Vetiver grass (*Chrysopogon zizanioides*) were the most effective at increasing productivity levels (Figure 3). Overall the Malawi studies provide a good example of an integrated modeling program to assess both soil loss and its implications for the economy and society.

Figure 3: Sediment trapping using stones and growing vegetation, Dedza, Malawi.

The results from synthesis and modelling studies summarized above are often very different than the results from site-specific studies. For example, Stocking (2003) draws upon early research presented by Stocking and Tengberg (1999) to model the impact of soil erosion on productivity at tropical agriculture sites from the FAO-sponsored Erosion-Productivity network. The network used a standardized research design involving soil loss and runoff plots of approximately 50m^2 at various sites in Africa and South America. The authors cite strong evidence that erosion follows a curvilinear, negatively exponential form. As is evident from Table 4, erosion losses recorded at their sites were generally substantially higher than the average global and regional rates cited above. To examine the impact on food security, the authors use a threshold of 1000 kilograms of grain per year to meet household food security for a household of two adults and six children; the starting point for all simulation is is 4000 kg grain yr^{-1}. Given the inherent constraints of several of the soils examined, Stocking (2003) overall presents a much bleaker scenario for erosion-induced impacts on food security than the average impacts discussed previously.

Table 4 Selected annual erosion rates and years of erosion-induced yield decline required to reach a critical threshold of 1000 kg yr^{-1} of grain per household (Stocking, 2003 and Stocking and Tengberg, 1999).

Soil and slope	Annual soil loss rate		Years to reach household food insecurity threshold	
	Moderately good cover	Poor Cover	Moderately good cover	Poor Cover
	t ha^{-1} yr^{-1}		Years	
Humic Nitosol 27-34% slope	20	86	19	4
Rhodic Ferrasol 16% slope	94	187	2	1
Orthic Acrisol 13% slope	157	200	3	1
Eutric Cambisol 24% slope	5	9	42	23
Luvic Phaeozem 1-2% slope	0.6	5	65	7

1.4.3 Soil organic carbon and the regulation of greenhouse gases

Erosion has a major effect on the storage of soil organic carbon (SOC) in the landscape. Each erosional episode removes an increment of soil from the surface, and in many soils this surface increment is richer in soil organic matter (including SOC) than are the underlying soil horizons. Furthermore,

transport by water and wind erosion causes carbon (C) enrichment in sediments relative to source soils–it is well established that both SOM itself and clay and silt-sized particles (which are commonly carbon enriched in comparison with coarser sand particles) are preferentially transported by water and wind erosion processes.

Given the overarching importance of C processes in greenhouse gas emissions and human-induced climate change, the role of erosion in the carbon (C) cycle has been extensively examined over the past ten years. Considerable disagreement still exists about whether erosion results in enhanced emissions of C to the atmosphere (a C source) or enhanced sequestration of carbon in the soil (a C sink).

The changes to soil SOC stores are the net outcome of a series of interacting processes associated with erosion (reviewed by Doetterl *et al.*, 2016) (Figure 4). In level summit or plateau positions (and indeed in level landscapes generally) the SOC store is determined by the balance between carbon inputs (originally as photosynthate from plants) from roots and from mixing by soil organisms and carbon outputs, primarily as gaseous CO_2 resulting from mineralization of organic material by soil organisms.

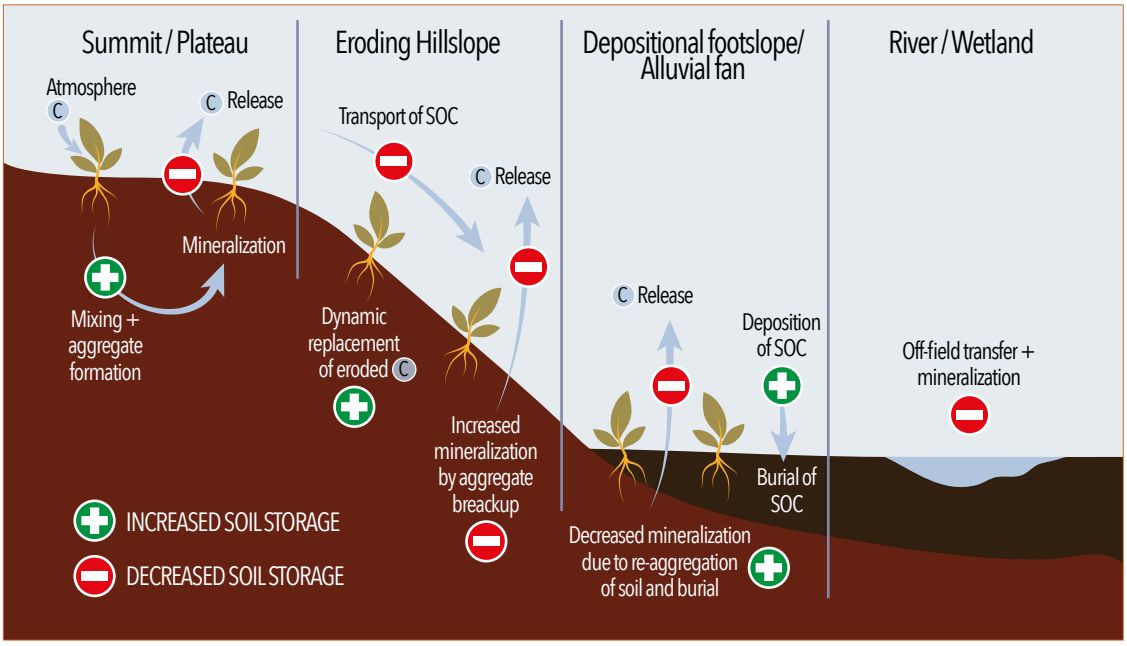

Figure 4: Schematic diagram of the effect of erosion on landscape-scale storage of Soil Organic Carbon.

Water and tillage erosion alters this balance by lateral transport of surface soil material (including SOC) from eroding slope positions. The lateral transport results in a loss of SOC from these eroding positions, but it has been argued that erosion can also increase C storage by continuously removing a fraction

of SOC that is then replenished with new C input from photosynthate from plants (termed *dynamic replacement*; Harden *et al.*, 1999). Raindrop splash and transport by flowing water can also lead to increased mineralization and release of CO_2 due to breakdown of aggregates and the exposure of fresh (labile) SOC. In general, the result of the balance between these processes is a net loss of SOC from erosional positions.

Deposition of a fraction of the eroded soil occurs where the slope gradient decreases (Figure 4) and the fate of this deposited carbon is a major source of uncertainty in the estimation of SOC contributions to the global C cycle (Doetterl *et al.*, 2016). Generally the rates of CO_2 release from the depositional layer are slower than from surface soil. The reduction in mineralization rates that occurs in deeper soil layers (including depositional layers) leads to burial of the SOC and to increased profile storage of SOC. As well, the deposited SOC and mineral sediment will undergo re-aggregation, which again protects the SOC from mineralization. Hence although CO_2 release is often high from these moist, lower slope positions, the net effect is an increase in SOC storage in these positions.

Tillage and water erosion differ insofar as soil transported by tillage erosion will be entirely deposited in the lower slope positions, whereas a fraction of water-transported soil may be transported off-site into wetlands (if present) or directly into a water channel. The fate of the C transported off-site is complex, and again complicates the question of the net effect of erosion on the C cycle.

Currently there is no widely accepted estimate of the net effect of water and tillage soil erosion on the global C cycle. The study of Van Oost *et al.* (2007) has been widely cited; the authors concluded that erosion produced a sink of approximately 0.12 Pg C yr^{-1} (range from 0.06 to 0.27), which is considerably lower than other estimates at that time. They suggest that the smaller value reflects both a) over-estimation of global erosion rates in other studies; and b) over-estimation of the SOC replaced in eroding slope positions by dynamic replacement. Their value can be compared to the annual fluxes for CO_2 removed from the atmosphere by plant photosynthesis (128 Pg C yr^{-1}) or released back to the atmosphere by total respiration and fire (118.7 Pg C yr^{-1}). A recent modelling effort by Lugato *et al.* (2016) for erosion effects on SOC storage in the European Union found a cropland SOC erosion rate that was only half that of the global estimate of Van Oost *et al.* (2007) (0.068 Mg C ha^{-1} yr^{-1} versus 0.16 Mg C ha^{-1} yr^{-1}), suggesting an even lower impact of erosion on the regional carbon cycle. Despite continued calls for further study on the effects of water and tillage erosion on the global SOC cycle (see for example Lal, 2019), few of the existing studies suggest a major effect of erosion.

The effect of wind erosion on SOC storage has received less attention, in part because wind erosion is confined to a smaller land area than are water and tillage erosion. In a recent global modelling study, plot research in Australia was been scaled up through modelling to the global scale by Chappell *et al.* (2019). As with water erosion, wind erosion preferentially removes SOM and finer soil fractions; the locally deposited soil is often dominated by sand-sized material and is substantially lower in SOC than the source soil. In their global analysis, Chappell *et al.* (2019) report mean wind erosion rates for the period between 2001 and 2016 of 1.0 to 7.0 t ha^{-1} yr^{-1} in many regions, resulting in mean SOC erosion of between 0.1 and 0.4 t ha^{-1} yr^{-1}. The authors point out that the losses of this scale greatly complicate efforts to increase SOC stores through improved management practices.

1.4.4 Soil erosion and sedimentation

Human-induced water erosion leads to higher sediment inputs into stream channels and increased sedimentation into reservoirs along the stream channels. The increases in sediment quantity and sedimentation lead to multiple effects (Owens *et al.*, 2005) (Figure 5). Sedimentation in lakes and reservoirs reduces life span and affects operation efficiency and costs, and in harbours and estuaries it requires dredging and its associated costs. Sedimentation and turbidity also disturb salmonid spawning gravel and alter other sensitive habitats as well as habitats along floodplains and associated land use.

Figure 5: Deposition of eroded soil in a small stream channel after an erosion event, Mwanza, Malawi

Palmieri *et al.* (2003) estimated worldwide storage capacity in large dams of 7 000 km^3 and an annual rate of storage loss due to sedimentation of 0.5 to 1.0 percent. This was equivalent to a replacement cost of approximately USD 13 billion in 2003 dollars. The percentage of this overall loss due to human-induced soil erosion was not estimated in the article.

Overall there are three broad types of management approaches that can address sedimentation problems in reservoirs: routing of sediments through or around the reservoir, removal of sediments in the reservoir to regain capacity, and minimization of sediments arriving in reservoirs from upstream (Kondolf *et al.*, 2014). The latter approach clearly involves catchment-scale erosion control but does not address the issue of downstream sedimentation starvation due to trapping of fine sediments in reservoirs. For example, Zhou, Zhang and Lu (2013) estimate that construction of the Three Gorges Dam has reduced suspended sediment loads in the Middle Yangtze River by 91 percent, total phosphorus by 77 percent, and particulate phosphorus by 83 percent annually. This reduction has likely reduced primary productivity of the river and of the floodplain and coastal agricultural regions, which previously experienced flooding from the river.

1.4.5 Agrochemical contamination in waterways

Soil erosion also contributes to pollution of waterways by nutrients and by other agrochemicals such as pesticides. This pollution leads to eutrophication of waterways and the resulting impact on aquatic life as well as direct toxicity effects on organisms (Owens *et al.*, 2005).

Agrochemicals reach surface waterways as both dissolved and particulate forms, and water erosion is often the source of the particulate material. Harmel *et al.* (2006) examined nitrogen (N) and phosphorus (P) fractions in nutrient loads from watersheds in 15 states of the United States of America and two provinces of Canada. Particulate N and P loss contributed, on average, three times as much as dissolved forms to loads, indicating the overriding effect of soil erosion and transport on N and P loads. Phosphorus is a particular concern for eutrophication. Phosphorus is strongly retained by solid phase and transported as eroded solid particles and through transport of manure and human waste (Yuan *et al.*, 2018).

Erosion losses of N and especially P have been highlighted as major contributors to the most serious resource issues facing humanity - the concept that there are planetary boundaries that define a safe operating space for human societies to develop and thrive (Steffen *et al.*, 2015). Biogeochemical flows of both N and P were judged by Steffen *et al.* (2015) to be in the high risk zone beyond the safe planetary boundaries. They set two thresholds

for P flows, one for prevention of large-scale ocean anoxic (i.e., oxygen-depleted zones) events and one for eutrophication (i.e., oversupply of nutirents) of freshwater. They and others (e.g. Cordell, Drangert, and White, 2009) state that the addition of P to regional watersheds is almost entirely from fertilizers and hence the erosional processes that transfer P from arable fields to freshwater and oceans are the prime determinant for the excess P.

1.4.6 Wind erosion, desertification and human health

Like water and tillage erosion, wind erosion causes decreases in soil productivity and in SOC storage by soils. Wind erosion has additional linkages with desertification–land degradation in arid, semi-arid and dry sub-humid areas resulting from various factors, including climatic variations and human activities (D'Odorico *et al*. 2010) and with direct human health issues associated with dust inhalation (Figure 6).

Figure 6: Wind-eroded landscape with little or no vegetative cover, Iran

Human-induced wind erosion is a major cause of land degradation associated with desertification (D'Odorico *et al.*, 2013). Like water erosion, wind erosion is a naturally occurring process that can be accelerated by

human activity: the analysis by Ginoux *et al*. (2012) assigns 75 percent of global dust emissions to natural sources and 25 percent to anthropogenic or human-induced sources. The contributions of human-induced erosion differ greatly between regions. Although North Africa accounts for 55 percent of global dust emissions, only 8 percent are of these are human induced, whereas in Australia 75 percent of emissions are human induced. In almost every case the human-induced emissions are caused by agricultural activity, especially overgrazing.

Dust storms have a direct impact on human health through the inhalation by people of fine particulate matter. Human health assessments use the amount of dust with diameters less than 10.0 μm and less than 2.5 μm as indicators of risk. Goudie (2014) presents data showing that the safe levels of these fractions has been exceeded in many cities throughout the globe including in China (Beijing, Shanghai), Australia (Sydney, Brisbane), the United States of America (Spokane), and Iran (Ahvaz, Sanandaj). Although direct causation of health issues from dust can be difficult to establish, Goudie (2014) cites several studies from southern Europe showing increased hospital admissions due to respiratory issues and higher mortality during major dust events originating in North Africa.

Dust storms also have an impact on road and air transportation due to decreases in visibility. An emerging issue is the effect of dust deposition on power loss from solar panels, which decreases the photovoltaic performance of the panels (Sayyah, Horenstein and Mazumder, 2014). Large arrays of panels are often located in arid environments (due to the absence of cloud cover) but these are also the areas where dust emissions are most common.

1.4.7 Erosional impacts beyond the economic sphere

There is a final group of erosional impacts that are more difficult to assess, and to which it is very difficult if not impossible to assign an economic value. The developing concept of Nature's Contributions to People (see, for example, Diaz *et al*. 2018) includes reporting categories such as the physical and psychological experiences provided by healthy landscapes and their role in supporting identities through religious, spiritual and social-cohesion experiences. These contributions are especially important for many indigenous societies that have retained stronger links to their home places. Erosion visibly degrades landscapes through exposure of subsoil, presence of rills and gullies, or the occurrence of dust storms. The effect of this degradation on the societal, spiritual and cultural values of a community can be profound and may extend far beyond the economic sphere (Figure 7).

Figure 7: Gully erosion encroaching on a village in Lilongwe, Malawi.

2 EROSIONAL PROCESSES

The processes associated with the three forms of erosion are very well established. It is important to review them insofar as the management of erosion involves manipulating the key processes so as to reduce the rate of erosion associated with them.

2.1 Water erosion

Soil erosion by water takes place through three main processes: 1) detachment of soil (as particles or aggregates) from the soil mass; 2) movement of detached material; and 3) deposition. The processes are also commonly subdivided into non-channelized splash/inter-rill erosion and channelized rill and gully erosion. Although both subsurface erosion through piping and through shallow mass movements also contribute to water erosion, the focus in this section will be on diffuse and linear erosion processes.

Water erosion is triggered by rainfall events. Wetting of the soil surface initially causes dispersion of soil and releases particles from the soil mass. Slaking of soil aggregates also occurs as water penetration into aggregates causes compression of the air within the aggregate and its disintegration.

Raindrop splash is responsible for the majority of detachment of soil from the soil mass. Soil detachment (measured in kg m^{-2}) is a product of the kinetic energy of drop impact (measured in kJ m^{-2}), the threshold energy need to initiate the detachment process, and soil detachability (kg kJ^{-1}). Torri and Borselli (2012) report two peaks of maximum soil detachability: one at very high clay content (> 40 percent) and one associated with silt-sized particles at lower clay contents. Generally, as particle size increases into the sand range, detachability decreases. Particles released by raindrop splash can clog surface pores of the soil (thereby increasing the volume of runoff) and smooth the soil surface (thereby increasing the velocity of runoff).

When the infiltration capacity of the soil is exceeded, water begins to flow over the soil surface as runoff. Runoff both detaches soil and transports the detached soil particles and aggregates in the process of sheet or inter-rill erosion. Many approaches have been used to describe the erosive power of flowing water; the depth of water (m) and its velocity (m s^{-1}) are the key elements in determining the unit discharge rate (m^2 s^{-1}). The soil has a hydraulic resistance to flow, which depends on surface soil factors such

as particle size, clod size, rock fragment content, surface roughness, and vegetation. When the flow exceeds the hydraulic resistance, detachment and transport of soil occurs.

Small differences in surface roughness and slope configuration cause spatial variation in flow characteristics and lead to concentration of flow and localized incision into the soil surface horizon in the process of rilling. In tilled landscapes, rills often follow tillage rows or ruts from wheels until the down- and cross-slope configuration determines flow direction (Figure 8). Rills often incise through the looser A horizon until a denser subsoil layer is reached; the rill then widens laterally on the surface of the denser subsoil layer. This process of incisions and widening leads to considerable loss of the organically enriched A horizon.

Figure 8: Rills following tillage lines, Southern Saskatchewan, Canada.

The effect of sheet and rill erosion on the soil surface is lost the next time a tillage operation takes place; the soil dragged by tillage implement fills in the rill. This delivery of "new" soil by tillage to slope positions where rills are likely to form is an example of interaction between water and tillage erosion processes.

Gully formation occurs where sufficient water concentration occurs to incise more deeply into the soil and underlying sediments (Figure 9, Figure 10). One practical definition of gullies is that they are incised channels that cannot be filled in by normal tillage operations. Other studies use a depth of 0.3 m as the threshold between rills and gullies (Castillo and Gomez, 2016). Drylands are particularly susceptible to gully formation because of sparse vegetation and a precipitation regime that favours infrequent but

short, high-intensity rainfall events (Sidle *et al.*, 2018). Gullies differ from rills insofar as subsurface processes such as piping (subsurface erosion along pores or cracks) and mass movements such as sidewall collapse also contribute to their formation and extension. As gullies deepen, the mass wasting processes take over from surface flow processes (Sidle *et al.*, 2018).

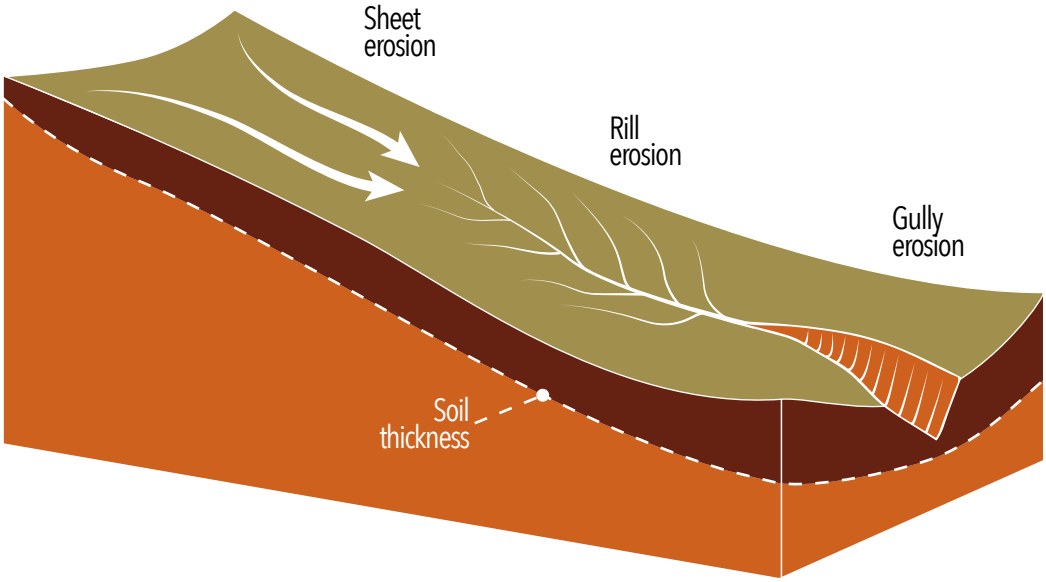

Figure 9: Schematic diagram of the position of sheet, rill, and gully erosion in a simple hillslope system.

Figure 10: Rills forming in lower section slope and then contributing flow to gullies at the base of the slope. Bolivia.

The most common equation used for gully initiation at a given site is a slope-area power function that describes the critical slope gradient and drainage area (that is, the area of land contributing runoff to the site). The critical area required for gully initiation differs depending on factors such as soil, climate, vegetation cover, and lithology, which again determine the resistance to gully formation.

Deposition of eroded soil occurs where hydraulic conditions change such that the flow is no longer able to transport the sediment it contains. Generally deposition takes place where the depth or the velocity of the runoff decreases. This occurs at the base of slopes (where the decrease in slope reduces the velocity of flow) or where water is no longer confined in a narrow channel (such as where a slope ends at a level surface) (Figure 11). Within an agricultural field, the depositional zones often form fan-like features at scales ranging from tens of centimeters to tens of kilometres across. These features often bury growing crops and result in a direct yield reduction in the year of deposition. In other cases the eroded soil is transported off-field before deposition occurs and is then subject to further erosion and transport into fluvial channels (Figure 12).

Figure 11: Example of a depositional fan located where slope gradient decreases and flow is no longer channelized. Southern Saskatchewan, Canada.

Figure 12: Deposition of eroded soil in a drainage channel adjacent to the field. Southern Saskatchewan, Canada.

Soil aggregates are often destroyed during detachment and transport, and the soil is transported as individual particles. When soil begins to be deposited, gravel and coarser sand fractions are deposited first, followed by finer sand fractions, silt, and finally clay (Figure 13). In many situations, however, the finer silt and clay particles are carried in the flowing water until they enter into a larger fluvial network. Deposition of the finer eroded soil may occur along the floodplains of the river, or may be deposited in lakes or the ocean.

Figure 13: Separation of particle sizes in a depositional fan. Northern Spain.

2.2 Wind erosion

When the wind flows over the surface of the land, a turbulent zone occurs next to the soil surface and extends into the lower atmosphere (Fryrear, 2012). The turbulent zone facilitates the transfer of momentum from the wind to the soil surface and exerts a drag or shear stress on the soil surface. Where the wind velocity at the surface exceeds the threshold velocity required to move the least stable soil particle, detachment of soil (also called deflation) begins. Chappell *et al*. (2019) base their global wind erosion model on threshold velocities from Shao, Raupach and Leys (1996). Their wind tunnel experiments in Australia found minimum friction thresholds required to initiate particle movement on sandy soils without cover of between 0.14 and 0.36 m s^{-1}.

The particles detached by wind are normally transported in one of three modes: creep, saltation, or suspension. Creep particles (typically medium to coarse sand-sized particles or aggregates) roll along the surface and are often transported only short distances before being trapped and deposited.

Saltating particles (typically fine to medium sand particles or aggregates) bounce along the surface in a series of small hops. These particles are very important in initiating further detachment of particles from the surface until the transport capacity of the wind is reached or the supply of grains is limited by surface cohesion or crusts. Suspended particles or aggregates are typically in the clay- to very fine sand-size and move into the upper atmosphere where they can be transported over great, even continental, distances. Although suspended particles are often a small percentage of the total mass of soil being transported, they have the greatest impact in terms of the loss of nutrient-rich soil and for downwind air quality.

Physical models of wind erosion often distinguish between the processes required to initiate saltation by sand-sized particles (60 to 1000 μm) and the initiation of dust emissions (particles less than 60 μm) by saltating sand grains (see for example Shao *et al.,* 1996). Dust particles have very high threshold velocities but can be readily ejected into the wind by the impacts of the saltating sand grains, or saltation bombardment.

Deposition of wind-transported soil differs between sands transported by saltation and dust transported by suspension. In agricultural settings, saltating sand grains are deposited relatively close (tens to hundreds of metres) to the point of erosion wherever the transport capacity of the wind decreases or a moist soil surface or surface water is encountered. Suspended dust, on the other hand, may be transported for very long distances, and it is estimated that 25 percent of deposition of dust occurs in the oceans (Shao *et al.*, 2011). As discussed above, dust is often enriched in SOC and this transfer of carbon from land to ocean is significant.

2.3 Tillage erosion

The importance of tillage erosion was recognized by soil scientists around 1990, much later than recognition of wind and water erosion, and was closely linked to the use of the radionuclide cesium-137 in the 1980s (Govers *et al.,* 1999). The basis of tillage erosion is simple: tillage operations cause a net displacement of soil downslope because gravity retards upslope movement of soil and enhances downslope movement. In part the late recognition of its importance reflects its almost invisible nature on a per-event basis; it is only its long-term operation that becomes visible in the landscape (Figure 14).

Figure 14: Loss of surface soil and exposure of calcium carbonate-rich subsoil along convex hill crest due to long-term action tillage erosion. Southern Saskatchewan, Canada.

Experiments have shown that the displacement of soil depends on the slope gradient, specifically the change in slope gradient between the boundaries of a given slope segment. Tillage causes the loss of soil on convex slope elements (such as crests and shoulder slopes) where there is an increase in slope gradient over the length of a slope segment. This increase in slope gradient leads to a commensurate increase in soil displacement along the segment. In concave slope elements (such as footslopes and hollows) the slope gradient decreases over the segment, and hence the ability to transport soil decreases, and soil deposition occurs (Figure 15).

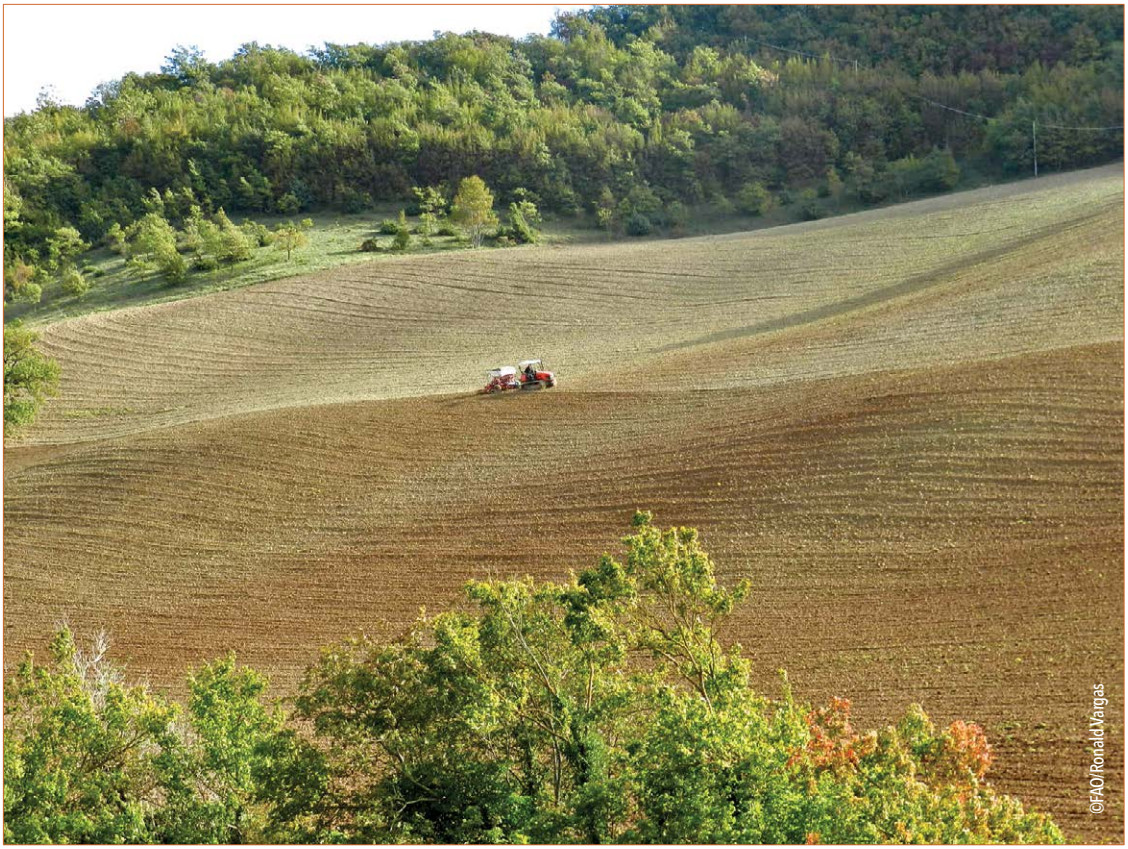

Figure 15: Tillage operations on sloping landscapes inexorably move soil downslope due to the action of gravity. Italy.

Physical modelling of tillage erosion has focused on determining the tillage transport coefficients associated with different tillage implements and the speeds at which they are operated. The amount of soil displaced per tillage operation increases exponentially as the depth of tillage and the speed of the equipment increases (Van Oost *et al.*, 2006).

Unlike water and wind erosion, tillage erosion does not typically cause the destruction of aggregates and differential transport of different particle sizes. Hence the composition of the deposited soil is typically very similar to the source soil, and no enrichment of SOC or fine soil particles occurs. Overall, tillage erosion is a major factor in determining horizon thicknesses, distribution of soil types and SOC storage within agricultural fields, but has (by itself) very limited impact on off-site transport. It does, however, deliver soil to field locations where it can be mobilized by water erosion processes and transported off-site.

3 CONTROLS ON EROSION PROCESSES

Although the physics of erosion processes are unchanging, the actual rate of erosion at any site will depend on the specific conditions at that site that control the action of the physical processes. These controlling conditions–soil, topography, vegetation cover, and human disturbance–were extensively studied during the latter half of the twentieth century. In this section, the relationships among the controlling conditions and the physical processes are summarized in order to provide a basis for the assessment of soil conservation measures that follows in Chapter 5.

3.1 Factors influencing water erosion

The rate of water erosion occurring at a site depends, in the first instance, on the rainfall itself (the source of rainsplash detachment) and the runoff generated during the rainfall event, which both detaches and transports the eroded soil. The erosive action of the rainfall and runoff is moderated by the surface conditions of the land, which includes the resistance the soil offers to detachment and transport, the effects of vegetation, and the slope gradient and configuration.

3.1.1 Climate

The physics of the effects of rainfall on soil have been extensively studied and many equations for rainfall erosivity have been developed. The two attributes of rainfall commonly used are rainfall amount and rainfall intensity (that is, the ratio of the total amount of rain falling during a given time period to the duration of the period, expressed as mm hr^{1}). Characteristics of the raindrops themselves are expressed as the kinetic energy, which is one half of the product of the mass of the drop and the square of the impact velocity. The widely used measure of rainfall erosivity (developed by Wischmeier, 1959) is the product of total kinetic energy of a storm multiplied by the maximum 30-minute intensity; Panagos *et al*. (2015) present a regional implementation of the use of this measure to calculate rainfall erosivity at the European scale.

Rainfall erosivity is relevant globally, but other sources of water for runoff-induced erosion are of importance in particular regions. Water released through snowmelt is a major contributor to water erosion in northern countries such as Canada and the Russian Federation, as is rain that falls

on frozen soil in these regions. Water added through irrigation is also not included in the basic rainfall erosivity calculation.

3.1.2 Soil

Soil properties strongly affect both the amount of runoff generated from a point and the resistance of soil to detachment (or its inverse, soil erodibility).

Water added to the soil surface can either infiltrate the soil or flow along the soil surface as runoff (assuming there is even a slight slope present). The proportion of water that infiltrates the soil depends primarily on 1) the nature of the precipitation event, including factors such as rainfall intensity, drop size, and snowmelt rates; 2) the slope of the surface–generally the higher the slope, the lower the percentage of water that infiltrates; and 3) the infiltration rate of the soil.

Water infiltrates the soil due to the influence of gravity and the attraction of reactive soil particles. The rate of infiltration is limited by the diameter of the pores, the continuity of the pores, and the pre-existing (or antecedent) moisture conditions. Generally soil texture is the most important control on the infiltration rate (Table 5); other factors such as the SOM content, cultivation history, and vegetation may also play an important role.

Table 5 Classification of soils in order of minimum infiltration rates after a period of prolonged wetting when planted to row crops. From: Dunne and Leopold, 1978. Water in Environmental Planning.

Group	Minimum Infiltration Rate (mm hr^{-1})	Soil Characteristics
A	8 to 12	Deep sands, deep loesses or silt loams, well-aggregated soils
B	4 to 8	Shallow loess, silty loams, and sandy loams
C	1 to 4	Clay loams, shallow sandy loams, soils low in organic matter, soils high in clay
D	0 to 1	Soils of high swelling potential, heavy clays, some saline soils (Solonetzic blow-outs)

The size of particles and aggregates within the soil is the single most important soil factor influencing soil erodibility. In terms of soil texture alone, clay-dominated textures will resist detachment because of the high cohesion between clay particles, and medium to coarse sand-dominated soils resist transport due to their large particle size. Hence silt-dominated

soil and loamy soils (that is, those with roughly equal proportions of sand, silt, and clay) are most susceptible to detachment and transport.

Normally, soil particles are bound up into larger aggregates, and the size and stability of these aggregates is the more relevant control on soil erodibility. For erosion purposes, aggregates can be divided into microaggregates (aggregates up to 250 μm in diameter) and macroaggregates (from 250 μm to 10 mm or greater) (Bryan, 2000). Microaggregates are tightly bound, dense particles of low porosity and are often resistant to destruction. These microaggregates can be further incorporated, along with stones and unaggregated SOM, into the more loosely bound macroaggregates, which are less resistant to disruption. The strength of aggregation depends on the binding agents, which include humic acids, microbial mucilage, electrostatic bonds related to clay crystalline structures, and moisture and electrolyte contents (Bryan, 2000). In general, the higher the clay content of the soil and the higher the organic matter content, the more resistant the aggregates are to erosion stresses.

Aggregates are also subjected to various physical stresses such as frost action, root action, compaction and shrinkage, as well as to human-induced disruption such as tillage. These stresses vary over the year; hence the state of aggregation also varies considerably throughout the year.

A final major factor influencing soil erodibility is the roughness of the surface: the rougher the surface, the greater the friction offered to flowing water, and hence the lower its erosive potential. Large aggregates and clods resulting from tillage create a rougher surface, as do rock fragments on the surface. The effect of rocks is complex, however, as they may concentrate flow in channels between the rocks and therefore locally increase erosion (Torri and Borselli, 2012).

The most widely used tool for predicting soil erosivity is the K (soil-erodibility) factor of the RUSLE (discussed in Section 4.2.1), which attempts to summarize the main factors discussed above. The variables used in the K factor are % silt+clay, % sand, % SOM, soil structure class and soil permeability class. The major criticism of the K factor is that the small plots used in the creation of the empirical K factor often precluded rill development and hence are of limited value in real landscapes (Bryan, 2000).

3.1.3 Topography

Topography has a direct effect on the spatial pattern of water erosion. First, erosion increases in a linear relationship with increases in slope: the velocity of the runoff (and its erosive power) increases as slope increases.

Second, as runoff accumulates down- and across-slope, the depth of flow increases and hence its erosive power increases. On uniform slopes (that is, those with no across-slope curvature), the volume and depth of runoff increases downslope; hence, erosion increases from a minimum at the top of the slope to a maximum at the bottom of the slope (Figure 16). On more complex slopes with significant across-slope curvature, flow diverges from convex slope elements and converges in concave elements; hence erosion is concentrated in the concave elements. These concave across-slope elements are typically where gullies would begin to form in the landscape (Figure 17). Where the slope decreases at the base of the slope, the amount of sediment in the runoff is greater than its transport capacity, and deposition of sediment occurs.

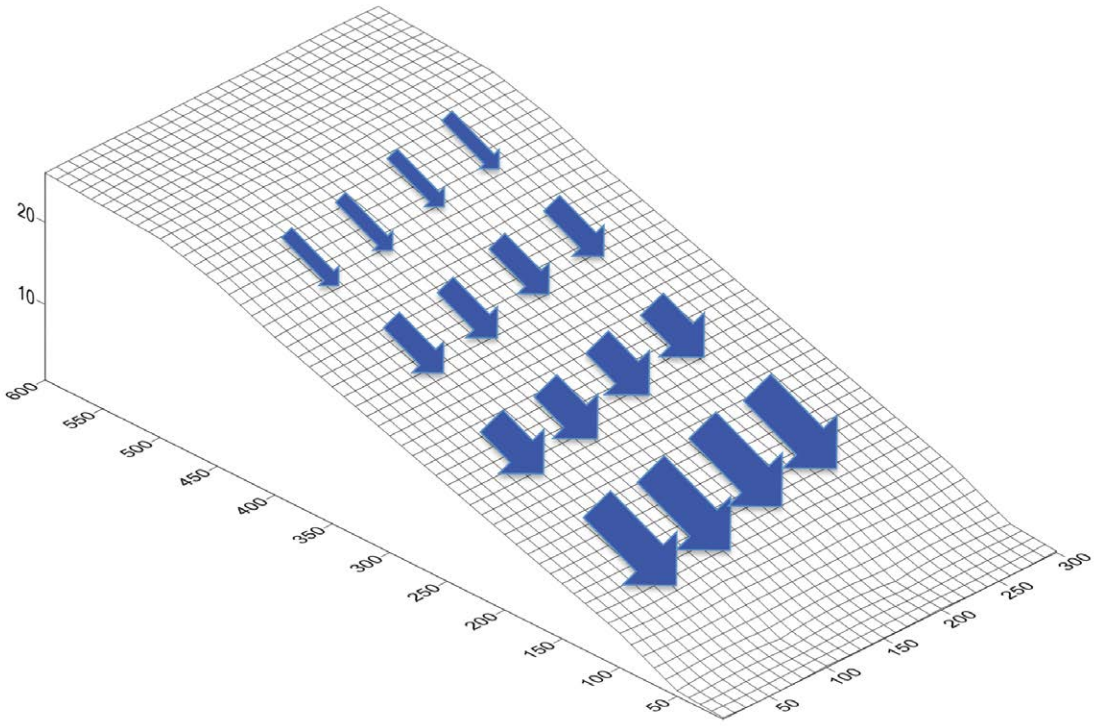

Figure 16: Schematic diagram of increasing depth and velocity of runoff along the length of the slope in a hillslope with no significant cross-slope curvature.

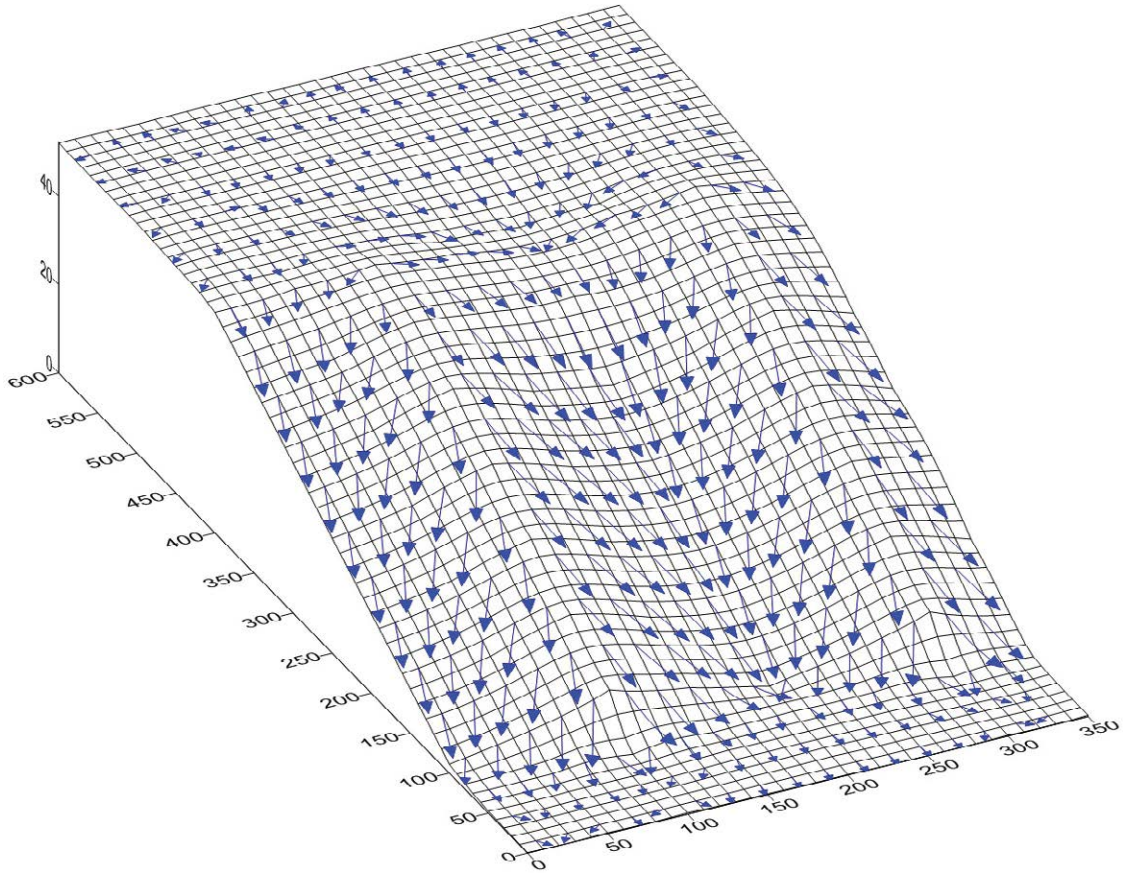

Figure 17: Schematic diagram of flow concentration in concave slope elements.

3.1.4 Vegetation

Vegetation has a significant effect on all of the water erosion processes. First, interception of rainfall by vegetation can almost double the total surface to be wetted before raindrops begin to reach the surface, wet the soil (leading to aggregate disruption) and accumulate on the surface (Torri and Poesen, 2014). Second, vegetation protects the soil from raindrop impact and retards the formation of surface seals; the former reduces raindrop detachment, and the latter decreases the volume of runoff. Third, plant roots increase macroporosity and hence increase the infiltration rate, thereby decreasing runoff (Gyssels *et al.*, 2005). Fourth, roots also increase the resistance of the soil to flow detachment. Fifth, vegetation increases the friction to overland flow, decreasing the velocity of flow and absorbing some of the erosion energy. Sixth, both the vegetation itself and, indirectly, the SOM produced from the plants, contribute to the formation of water-stable aggregates, and thereby increase resistance and infiltration. Overall, with increasing vegetation density and as we move from cropland to grassland to forest, we expect an increase in resistance by the soil to concentrated flow erosion and a decrease in runoff discharge during a rainfall event (Torri and Poesen, 2014).

The relationship between vegetation cover and relative erosion loss (that is, the erosion loss at a given vegetation cover relative to loss from bare soil) is exponential and can be presented graphically (Figure 18). Generally, sheet and rill erosion is reduced by 50 percent at vegetation covers of about 20 percent, by 75 percent at covers of about 30 to 35 percent, and by 90 percent at covers of about 60 percent (Gyssels *et al.*, 2005). Results for reductions of soil detachment by splash are similar but show greater variation across studies.

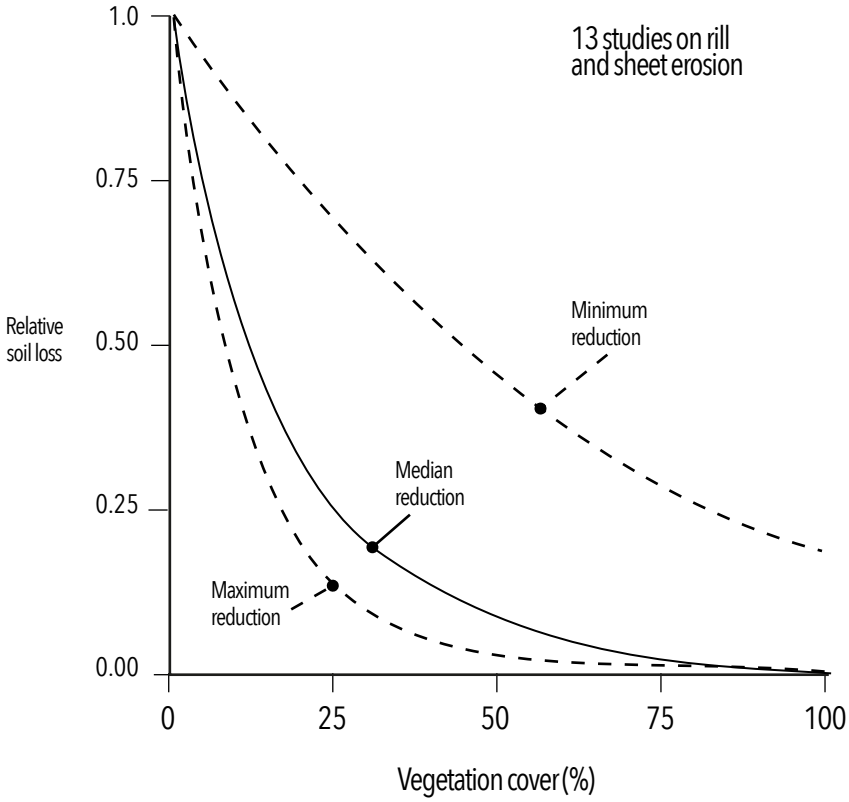

Figure 18: Graphical summary of 13 studies on the relationship between relative soil loss by interill and rill erosion and vegetation cover. Adapted from Gyssels et al., 2005.

3.2 Factors influencing wind erosion

Wind erosion occurs on bare or nearly bare, level, dry surfaces where the wind velocity is high enough to entrain soil.

3.2.1 Climate

Wind erosion becomes possible when the velocity of the wind at the soil surface exceeds the velocity required to move the most erodible soil particle

(Fryrear, 2012) and hence the wind velocity is key climate variable. Calculation of the threshold velocity required is difficult because the presence of soil in wind profiles changes the ability of the wind to detach and carry soil. A second climate variable is the moisture present in the surface soil, since soil wetness acts to stabilize the soil surface.

Inclusion of climate attributes in wind erosion modelling is essential but complicated, insofar as summaries of wind velocity aggregated over different time periods (for example hourly, daily, or monthly) will under-represent very short periods of intense winds. For their process-based Wind Erosion Assessment Model, Shao, Raupach and Leys (1996) use climate statistics based on three-hour surface observations of maximum wind speed, average monthly precipitation, number of days with precipitation, and distributions for rainfall intensity and transitions between wet and dry days.

3.2.2 Soil

The basic relationship between soil properties and wind erosion is straightforward: the larger the soil particle or aggregate, the greater the wind speed required to detach it from the soil mass and transport the particle (Fryrear, 2012). Individual sand particles are most easily eroded due to their lack of reaction with other soil particles. Aggregates are resistant to entrainment but they can be disrupted by impacting sand grains.

The formation of crusts at the soil surface retards wind erosion. Crusts can form when rainfall occurs after a tillage operation (which loosens the soil surface) or through the formation of biological crusts. Soil properties important for crust formation include clay content, calcium carbonate, and soluble-salt concentration.

The roughness of the soil surface is also an important control on wind erosion. In agricultural settings tillage normally increases surface roughness (Fryrear, 2012) through the creation of tillage ridges and furrows and randomly created clods on the surface. When the ridges from 0.06 to 0.25 m are perpendicular to the wind, erosion can be controlled (Fryrear, 2012). The formation of ridges by tillage is sometimes practised as an emergency measure to control wind erosion during a prolonged wind erosion event.

3.2.3 Topography and field configuration

Hills cause a major disruption in the flow of wind and hence affect the erosive potential of the wind. Field and theoretical studies indicate that generally wind erosion increases along the windward side of a slope, reaches its highest values towards the top, and then drops sharply on the leeward side

(Goossens and Offer, 1997). This "wind shadow" effect on the leeward side leads to the deposition of silt-sized particles (loess) in these positions.

The length of the field across which the wind blows is also an important landscape factor in wind erosion. Models such as the Revised Wind Erosion Equation (discussed in Fryrear, 2012) assume that erosion is zero at the upwind boundary of a field and then increases until the wind reaches a critical threshold for erosion.

3.2.4 Vegetation

The presence of a vegetation cover can dramatically reduce wind erosion. In cropped fields, both growing crops and crop residues can act to limit erosion (Figure 19). The relationship between relative soil loss and percent cover is exponential (Fryrear, 1985). The decrease in relative soil loss between 0 and 40 percent cover is very high, with wind erosion reductions of 80 to 90 percent. At very low cover levels (<10 percent) standing crop residues are at least six times more effective at reducing erosion than flat lying residues; hence residue management by producers is of great importance (Fryrear, 2012).

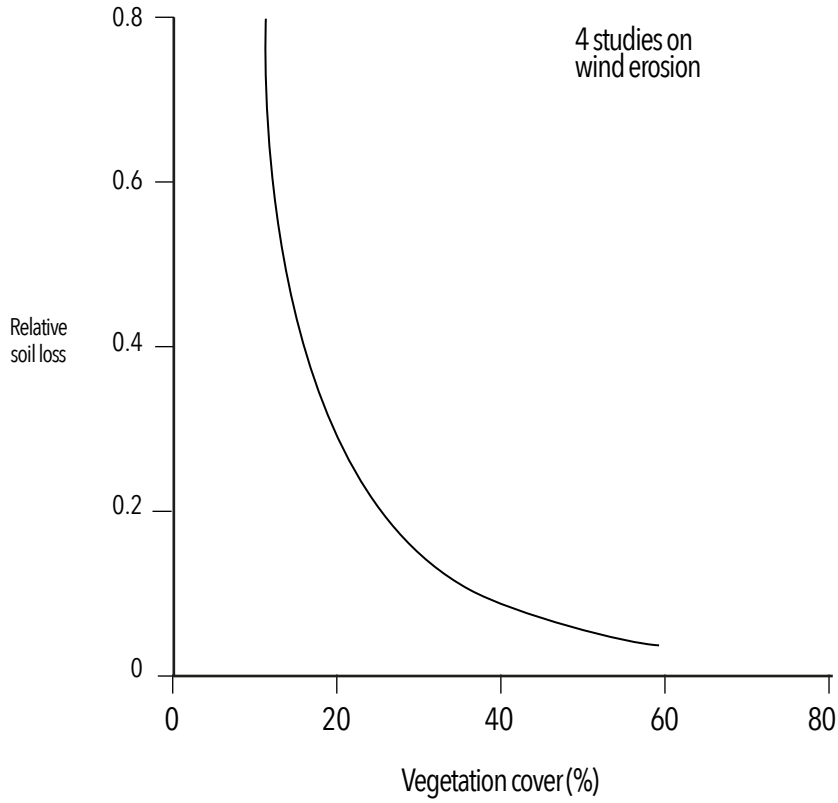

Figure 19: Graphical summary of four studies on the relationship between relative soil loss by wind erosion and percent vegetation cover by wheat. Adapted from Fryrear, 1985.

Non-cropped settings such as shrublands can experience considerable wind erosion due to the effects of overgrazing and trampling (Figure 20). The effect of shrubs on wind velocity and erosion depends on the relative cover of shrubs (Wolfe and Nickling, 1993). Isolated shrubs (between 1 and 14 percent cover) disrupt flow and cause a wake downwind of the shrub and a significant reduction of wind velocity (and hence lower wind erosion). For covers between 14 percent and 40 percent, the higher density of shrubs prevents the attainment of the full wake effect and less erosion reduction. For shrub covers greater than 40 percent, the entire area experiences the full wake effect, and erosion reductions are greatest. For non-cropped lands, shrublands have the highest overall wind erosion levels, and both grasslands and forests have significantly lower levels (Ravi *et al*, 2010).

Figure 20: Mounds of vegetation act to disrupt the wind stream and cause deposition of wind blown sediment. Iran.

3.3 Factors influencing tillage erosion

The tillage erosion rate is controlled by two set of factors (Lobb, Kachanoski and Miller, 1999; Van Oost *et al.*, 2006). Tillage erosivity is a function of both physical and human factors such as implement characteristics (tool shape, width, length), operating conditions (tillage depth, speed, direction) and changes in operating conditions by the operator in response to field conditions. Landscape erodibility determines the propensity of the landscape to be eroded and is determined by topographical parameters

(such as slope gradient and curvature), field characteristics (size and shape), and the physical properties of the soil (although the latter are not well described in the literature).

Considerable efforts were made during the 1990s to assess the effect of different tillage implements on rates of tillage erosion (Van Oost *et al.*, 2007). In general, tillage erosion rates increase linearly with tillage depth as more soil is subject to movement and downslope displacement. Generally, mouldboard ploughs (mean tillage depth of approximately 0.25 m) cause more tillage erosion than chisel ploughs (mean tillage depth approximately 0.15 m); secondary tillage operations such as the use of cultivators, harrows, or discs are shallower yet and cause correspondingly lower tillage erosion rates. Tillage speed is a secondary control on erosion rates but has a lesser effect than tillage depth.

Topography also has a strong effect on tillage erosion rates. Tillage erosion rates are highest in convex landform positions and considerable thinning of soils occurs in these positions (Figure 21). The spatial pattern of tillage erosion differs greatly from that of water erosion (Figure 22). Deposition of eroded sediment occurs at the base of slopes where the slope gradient decreases.

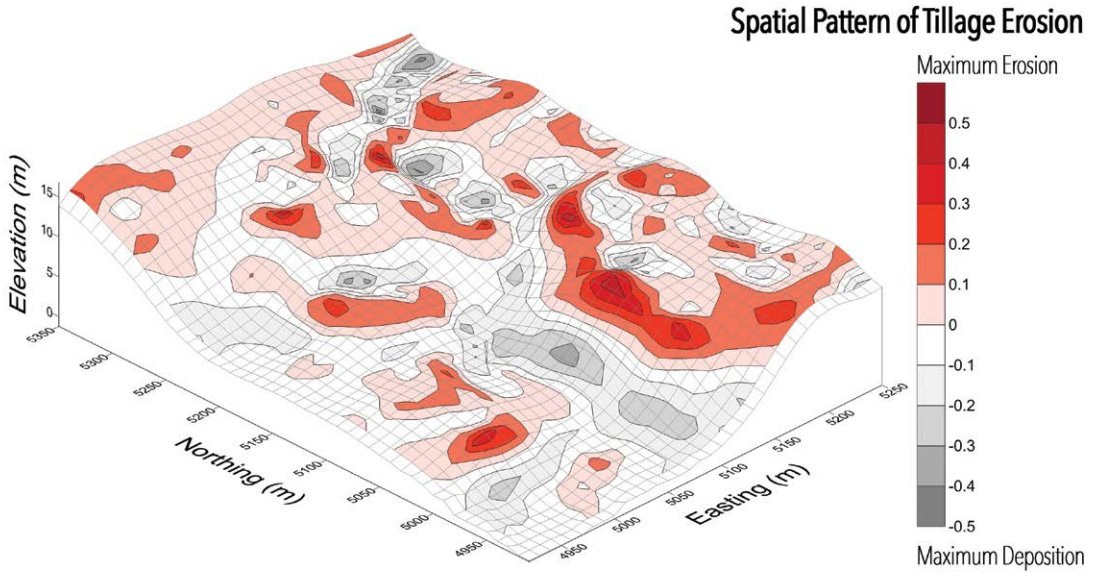

Figure 21: Schematic diagram of the spatial pattern of tillage erosion in a hummocky landscape. Tillage erosion is greatest on upper, convex slope elements.

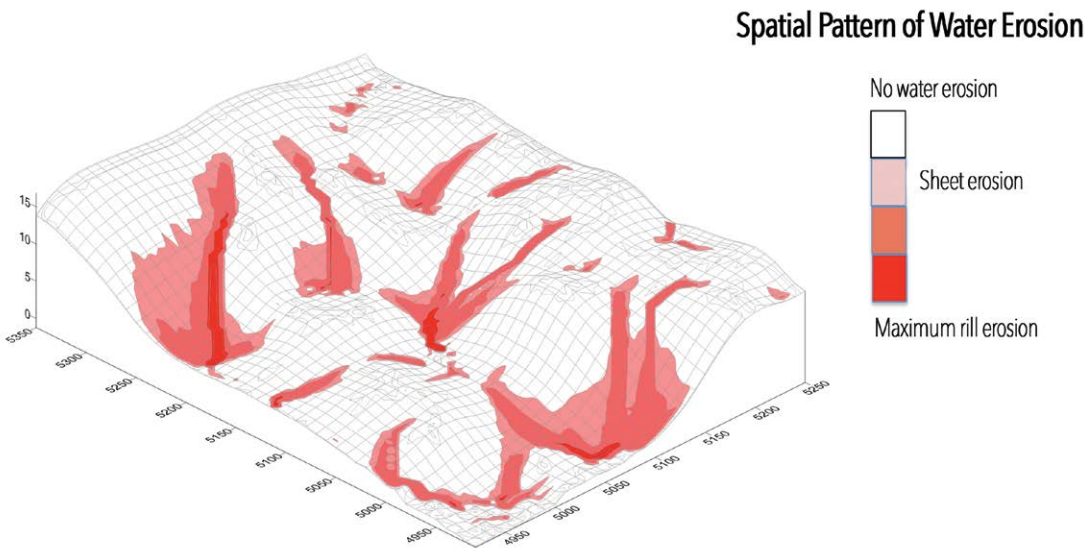

Figure 22: Schematic diagram of the spatial pattern of water erosion in a hummocky landscape. Water erosion by sheetwash and rills is greatest where flow depth and velocity is at maximum in concave slope elements.

4 SOIL EROSION ASSESSMENT: FIELD MEASUREMENTS AND MODELLING

In part, the large range of values for regional and global erosion estimates (Table 1) results from the diverse methods used to assess erosion in the field. It is useful to review these methods and their appropriate use to provide context for the estimates. These field measurements are also used in erosion model development, both as the empirical foundation of some models and as benchmarks to evaluate model performance. After an examination of the main erosion models in this chapter, the regional and global application of models is examined in the following chapter.

4.1 Field assessment of erosion

4.1.1 Water erosion

The direct assessment of water erosion in the field involves recording and measuring field evidence of the action of erosion such as rill and gully depth and extent, exposure of plant/tree roots, exposure of below-ground portions of fence posts and other structures, and the amount of sediment in drains (Stocking and Murnaghan, 2001). Stocking and Murnaghan argue that this approach is far more relevant to farmers and allows their perspectives on the important effects of erosion to be readily incorporated into the assessment (Figure 23, Figure 24). Moreover, the factors assessed are more practical than in many experimental designs and can be implemented with far less investment than more elaborate science-based approaches. Field assessment and monitoring of water erosion has also been used by Evans (2013) at sites in the United Kingdom. Evans argues that the amounts of erosion assessed using these methods is only a small fraction of the estimates produced by models (see Table 1), and better represent the actual rate of erosion occurring in the landscape. Certainly the field assessment methods seem well suited to areas where the infrastructure to support more elaborate plot based designs is not available.

Figure 23: Field measurement of rill depth, Dedza, Malawi.

Figure 24: Field measurement of gully depth and extent, Chitipa, Malawi.

Advances in remote sensing over the past decade would also seem to facilitate field assessment of erosion, especially the extent of rill and gully formation in landscapes (Bennett and Wells, 2019). Tools such as ground-based light detection and range (LIDAR) and close-range photogrammetry using unmanned aerial vehicles have been used for accurate measurements of ephemeral gullies. Bennett and Wells (2019) suggest that these technologies may supplant the use of model-based approximations by using actual measurements and change detection from repeated surveys.

Two main types of experimental designs have been widely used in soil erosion studies: rainfall simulators and erosion-runoff plots. Rainfall simulators are used to assess the detachment of soil by raindrops and the initiation of overland flow (Meyer, 1994). Different nozzle sizes can used to generate the raindrops, and the intensity of rainfall can be controlled; hence the impact of different precipitation events can be assessed. By carrying out the experiments on different soil surface conditions (for example, roughness due to tillage or residue cover), the effect of management on raindrop detachment and runoff initiation can also be assessed. Finally, the portability of the simulators allows replication of treatments (for example, cover types and soil types) and hence the use of statistical tools in the evaluation of results. Rainfall simulators were extensively used in the development of the data on which the Universal Soil Loss Equation (USLE; see below) was based. The simulators are criticized in part because their small application area prevents an assessment of the interaction between raindrops and overland flow, which is critical for detachment and transport (Kinnell, 2016).

Larger erosion-runoff plots have also been extensively used in research. The most widely used are the erosion-runoff plots used in the development of the USLE. These plots are 22.1 m long and 4.1 m wide and have instruments located at the base of the plot where runoff and sediment can be captured and measured (Figure 25). Normally the plots are installed in groups, and treatments such as different cover types can be imposed in a replicated design. If correctly replicated, erosion plots can be reliably used to assess the effect of management (such as tillage and cover) on soil loss; this is a major strength of erosion-runoff plots.

Figure 25: Standard erosion-runoff plots used to generate data for the Universal Soil Loss equation. Eastern Romania.

In the development of the first version of the USLE over 10 000 annual records from plots and small catchments were analyzed to develop the empirical relationships embodied in the equation. Data from research plots was also used by Montgomery (2007) in his widely cited summary of erosion values. Cerdan *et al*. (2010) summarize data from 81 experimental sites in 19 countries in Europe, covering 2 741 plot-years of measurements. The average plot size for this data set has an average length of 23.7 m and a surface area of 378 m^2, very close to the size of standard USLE plots. Poesen (2018) cites studies from five continents that have over 24 000 plot-years of data, dominated by plots with surface areas of 10^{-3} to 10^{-2} ha.

The use of erosion-runoff plots is deeply entrenched in the literature, but the shortcomings of plots have been well established as well. The USLE plots are designed to assess erosion losses from a specified slope segment with a constant slope and no significant cross-slope curvature, whereas the types and intensity of erosion processes operating in the field show distinct spatial differences down and across slopes. Although the standard erosion plots are said to measure both inter-rill and rill erosion processes, it has been argued that their area dictates that only situations dominated by raindrop impact are measured (see for example Kinnell, 2016); processes such as rill erosion and sediment deposition cannot be reliably evaluated from the plots. Furthermore, a considerable proportion of eroded soil is retained in

depositional sites in fields, and deposition cannot be evaluated in standard erosion plots. When extended to the broader landscape, data from erosion plots leads to overestimation of soil erosion rates (Van Oost and Bakker, 2012). Finally, there is a bias in terms of plot placement insofar as erosion-prone lands are overrepresented and sites with low erosion potential are underrepresented (Vanmaercke et al., 2012); thus, care is required when the results are extrapolated to larger landscapes.

4.1.2 Sediment yield from catchments

A second widely used water erosion assessment tool is the measurement of water flow and suspended sediment concentration from catchments and river basins (Walling, 1994). Typically these studies involve instrumentation to monitor the discharge of water at gauging stations along a stream or river channel and devices to sample the suspended sediment load of the water at set time intervals. Although possible, sampling of the coarser bed load carried along the river bed is much less common. For Europe, Poesen (2018) found literature on 1 287 catchments where these measurements had taken place, as well as 507 studies on sediment accumulation in reservoirs. The typical areas of the catchments are between 10^3 and 10^7 ha; the dearth of studies between plot and catchment scales is viewed as a weakness of current erosion research by Poesen (2018).

Sediment yield at the measurement point is expressed as the mass of sediment per area of the catchment for a given time period (for example, t km^{-1} yr^{-1}). If an independent estimate of gross erosion rates in the catchment is available, the sediment delivery ratio (that is, the ratio of sediment delivered at the basin outlet to gross erosion within the basin) can be calculated (Walling, 1994).

The limitations of catchment-scale sediment yield studies for estimating soil erosion have been well established (Walling, 1994). In the first place, considerable storage of sediment occurs within the catchment, and hence the sediment measured at the outlet is only a fraction of that actually eroded in the catchment. Second, storage of sediment on hillslopes and along river courses causes a time lag between erosion in the catchment and sediment measurement, and hence sediment yield may not be representative of current erosion rates in the catchment. Third, the sediment transported in the river is derived from sources other than hillslope erosion: for example, remobilization of sediment stored in floodplains is common especially during flood events. Although various ways to "fingerprint" sediment sources have been developed, this remains a major challenge. Finally, catchments are very difficult to replicate; hence, statistical analysis based on the imposition and replication of treatments (for example, with different cover types) is very difficult.

Overall, the great range of areas between plot and catchment studies produces very different estimates of erosion rates because of the occurrence of thresholds between scales and processes, such that erosion rates decline as the experimental area decreases (Garcia-Ruiz *et al.*, 2017).

4.1.3 Wind erosion

Generally speaking, there is a more limited range of field methods to measure wind erosion than there is to measure water erosion (Fryrear, 2012). Various types of sediment traps have been developed through the years that can be mounted at different heights to trap sediment that is carried in the wind column. The key attribute of the various samplers is their sampling efficiency: ideally, 100 percent of the sediment at a given height in the wind column is captured, but typically efficiencies range between 80 and 120 percent. The traps can be used to measure erosion from a given storm, and cumulative losses over the year can be assessed for an annual loss value.

As well as the passive sampling that the sediment traps allow, many researchers use wind tunnels to assess erosion under controlled wind velocities and surface cover conditions. Wind tunnels can be used in both field and laboratory settings, and allow precise control of wind conditions and sampling of wind-borne sediment at different heights (see for example. Wu *et al.*, 2018).

Global-scale assessment of dust storms has been made using remote sensing such as the Moderate Resolution Imaging Spectroradiometer (MODIS) Deep Blue
estimates of dust optical depth utilized by Ginoux *et al.* (2012). Products from this system have high resolution (approximately 10 km), daily near-global coverage and information on aerosol products at multiple wavelengths.

4.1.4 Tillage erosion

Two main approaches have been used for the quantification of tillage erosion rates and assessment of tillage translocation rates associated with different tillage implements. The most common method involves placing tracers in the soil prior to a tillage operation and then measuring their displacement after tillage (Fiener *et al.*, 2018). Tracers used for this method include micro-tracers (magnetic iron oxide, fluorescent sand) and macro-tracers (radio-frequency identification transponders). Alternatively, tillage induced changes in topography can be assessed using terrestrial laser scanners or unmanned aerial systems. A recent comparison of these methods by Fiener *et al.* (2018) showed considerable discrepancies among the methods and the authors suggest that this makes model parameterization challenging.

4.1.5 Erosion assessment using fallout radionuclides

Fallout radionuclides (FRN) have been widely used to assess soil erosion since the introduction of ^{137}Cs for this purpose in the mid-1970s. Although ^{137}Cs is the most commonly used, other FRNs such as $^{239+240}$Pu, ^{210}Pb, and ^{7}Be have also been used to assess erosion. The former two are anthropogenic in origin, whereas the latter two are naturally occurring. When the FRNs are deposited on the soil surface they are strongly bound to reactive soil particles and are then transported in the landscape by erosion processes. Assessment of the current distribution of FRN permits assessment of both the spatial pattern of erosion and the rates of erosion responsible for the pattern (Mabit *et al.*, 2018). The concentrations of the FRNs are measured in the laboratory using gamma spectroscopy for ^{137}Cs, ^{210}Pb, and ^{7}Be and inductively coupled plasma mass spectroscopy for $^{239+240}$Pu. The half-lives of the FRNs range from 53.3 days for ^{7}Be to 24 110 years for ^{239}Pu; the half-lives of ^{137}Cs (30.2 years) and ^{210}Pb (22.8 years) make them especially useful for soil erosion studies.

For erosion assessment using ^{137}Cs a series of cores are taken from the field using various sampling designs (Pennock and Appleby, 2002). The ^{137}Cs levels in each core are then compared to a nearby uneroded reference site and a conversion model is used to convert the change in ^{137}Cs to soil loss or gain. The lack of an agreed-upon conversion model has been criticized (Parsons and Foster, 2011), and the range of models that are used complicates evaluation and synthesis of the ^{137}Cs literature. Other concerns with the use of the method stem from the need for a uneroded reference site (Parsons and Foster, 2011), but authors such as Mabit *et al*. (2013) have argued that the method has been reliably used for erosion assessments.

Erosion estimates using FRNs do not allow the specific processes of erosion to be identified unless other sources of information are available. The methods do allow for the assessment of the spatial pattern of soil loss and gain in a sampled site, and research using ^{137}Cs in the 1980s and 1990s was essential for the recognition of the role of tillage erosion by soil scientists and geomorphologists (Govers *et al,* 1999).

4.2 Models

Models of soil erosion are widely used both to generalize specific field studies for broader application and to provide erosion estimates under different scenarios of controlling factors such as climate and land use change. Given the intent of this volume, we will focus on the latter category, which have provided many of the regional and global applications discussed in the following chapter.

4.2.1 Water erosion models

Many models for water erosion have been produced and continue to be produced. Torri and Borselli (2012) suggest that reviews of models are of little use because the models being reviewed have probably already been superseded by newer models.

In terms of scenario planning, by far the most widely used model for water erosion is the Revised Universal Soil Loss Equation (RUSLE) (Renard *et al.*, 1997). The RUSLE is a revision of the original USLE (Wischmeier and Smith, 1978), which was also a very widely used tool for erosion assessment and conservation planning.

The original USLE was a set of statistical relationships between water erosion losses and major controlling conditions calculated from the standard USLE plot data discussed previously. Regression analysis of the plot data determined the mathematical relationship between each USLE factor and soil loss (Renard *et al.* 1997). The RUSLE kept the basic structure of the USLE and included new research on several of the factors.

The RUSLE computes the average annual erosion expected on field slopes as follows:

$A = R \bullet K \bullet L \bullet S \bullet C \bullet P$
A = average soil loss per unit of area (t ha^{-1} yr^{-1})
R = rainfall-runoff erosion factor (which includes a factor for snowmelt runoff)
K = soil erodibility factor
L = slope length factor
C = cover-management factor
P = support practice factor (e.g., contouring, terracing)

The RUSLE is attractive for scenario analysis because of the inclusion of climate and management factors in the equation. Although the original USLE plots were installed in the United States of America, standard plots have also been established in many countries to allow for local calibration of the equations (Kinnell, 2016). The RUSLE is also used for soil erosion modelling in broader models such as the Agricultural Non-Point Source Model (AGNPS) and the WATEM-SEDEM model (de Vente *et al.* 2013).

The limitations of the RUSLE have been widely examined. The first is inherent to the equation and is acknowledged by the developers (Renard *et al.* 1997): the RUSLE calculates soil loss, not sediment yield, and the effects of erosion from other sources (such as gullies) and deposition within the field and broader landscape are not modelled. A further development of the

model (RUSLE2; Foster, Toy and Renard, 2003) provides an approach that does account for sediment deposition along one-dimensional hillslopes.

Other studies have criticized specific factors in the USLE family of models. For example, in terms of rainfall erosivity, Kinnell (2010) argues that the lack of explicit consideration of runoff in the R factor limits its ability to accurately predict event soil loss. A refinement called the Modified Universal Soil Loss Equation (MUSLE) uses runoff and peak flow data to estimate event-based soil loss (Sadeghi *et al.* 2014). The MUSLE is used in the Soil and Water Assessment Tool (SWAT) model (de Vente *et al.*, 2013). A recent comprehensive review by Benavidez *et al.* (2019) highlights many of the continuing concerns with the use of the USLE-based approach.

There are also many physically based models that are used for water erosion assessment and prediction. Pandey *et al.* (2016) provide a review of 50 models and provide guidance about the selection of models for particular uses.

4.2.2 Wind erosion models

One of the earliest wind erosion models, the Wind Erosion Equation (WEQ) (Woodruff and Siddoway, 1965) had a similar empirical basis to the USLE; annual erosion (in Mg ha^{-1}) is a function of soil erodibility, soil roughness, climate, unprotected field length and vegetation. Limitations to WEQ led to the development of the Revised Wind Erosion Equation (RWEQ), which incorporated an improved physical basis for transport of sediment (Fryrear, 2012). The model requires relatively simple input data on weather, soil, crop and tillage, and can estimate erosion for time periods from daily to annual. Finally, a Wind Erosion Predictions System (WEPS) was developed in the 1990s with a stronger physical basis, but the extensive list of input parameters has limited its use.

Recent efforts to model global patterns of wind erosion have used an entirely different approach from the WEQ family of models discussed above. Chappell *et al.* (2019) base their wind erosion model on the reduction in wind velocity (and erosion) caused by vegetation. They use the proportion

of shadow cast by vegetation to establish a relationship with aerodynamic properties of relevance to wind erosion and then derive wind sediment transport values. The degree of shadowing is assessed from MODIS albedo data and is coupled with global data sets on wind speed, soil moisture, and soil data to provide a global estimate of the effect of wind erosion of SOC stores.

4.2.3 Tillage erosion models

The tillage erosion models that have been developed use some combination of tillage factors coupled with landform configuration information to predict soil movement. Van Oost *et al*. (2003) estimate tillage displacement using an empirical equation that includes terms for tillage displacement in the direction of tillage and lateral to the main direction, tillage depth and implement speed, and several coefficients derived from tillage experiments. Li *et al*. (2008) use a linear function for estimating the effect both of slope gradient and curvature and of the tillage operation itself on tillage translocation distance. They couple this with models for estimating the effects of tillage translocation on soil mass and soil constituent redistribution in their estimation of SOC redistribution by tillage. To date, no single model has emerged as the standard for estimation of tillage effects.

5 REGIONAL AND GLOBAL MODELLING OF SOIL EROSION

Many of the approaches discussed in previous chapters have been used to develop models that can provide estimates of erosion at broader scales. These estimates are critical for evaluating the extent of erosion and for assessing its importance relative to the numerous other challenges faced by human society. Modelling efforts are, however, often criticized by field-based researchers for their simplified view of the complex nature of erosion and its controlling factors.

The need for model development is exemplified by the continuing use of the Global Assessment of Land Degradation (GLASOD) study (Oldeman, van Engelen and Pulles, 1991; Oldeman, Hakkeling, and Sombroek, 1991) in recent studies such as the Status of the World's Soil Resources report (FAO and ITPS, 2015) and Amundson et al. (2015). The Global Assessment of Land Degradation study used a combination of expert opinion and field assessments to produce global map products, but the use of data from the 1980s to describe current degradation status is gravely out of date, given the significant shifts in land use and climate that have occured over the past three decades.

5.1 Models based on the Revised Universal Soil Loss Equation (RUSLE)

The use of models for erosion prediction and for examining erosion-related impacts–and the criticism of these efforts–is perhaps best illustrated by the erosion modelling efforts in Europe. Panagos *et al.* (2015) use a Europe-specific version of the Revised Universal Soil Loss Equation (RUSLE), which they call the RUSLE2015 model, that draws upon the extensive, harmonized datasets amassed by the Joint Research Centre of the European Commission. The main changes to the RUSLE result from the improved data available to European researchers; the specific changes to soil erodibility, rainfall erosivity, cover management, and topographic and support practice factors are documented by Panagos and his co-workers in individual articles. The authors also make extensive use of auxiliary data in their work–for example, the topographic factor draws upon a recent 25-m Digital Elevation Model for Europe, and the soil erodibility factor draws upon remote sensing data and terrain features.

Panagos *et al.* (2015) produced a map of soil loss in the European Union in 2010 with a pixel resolution of 100m. The map excludes approximately

10 percent of the landmass that is not erosion prone (for example, lakes, wetlands, urban areas and bare rocks). For the area considered, the average mean annual rate of soil loss due to water erosion is 2.5 t ha^{-1} yr^{-1}. The soil loss rate for 76 percent of the total European land mass is below 2 t ha^{-1} yr^{1}, which is considered to be sustainable. The highest rates of soil loss are in Mediterranean areas, mountainous areas (such as the Alps, Apennines, Pyrenees, and Sierra Nevada), western Greece, western Wales, and Scotland. In terms of land cover/land use classes, the highest losses (mean of 9.47 t ha^{-1} yr^{-1}) occur for permanent crops such as vineyards and olive trees, which have low cover and are planted in the Mediterranean region (a high rainfall erosivity region). The authors could also perform a sensitivity analysis of the effect of the Good Agricultural and Environmental Condition (GAEC) program (implemented by European Union member states) on the C factor of the RUSLE2015 . Overall the effects were very small: reduced/ no-tillage practices and cover crops (applied by 2015 in more than 25 percent of the agricultural lands) reduced soil loss by only about 1 percent each. Similar small effects were predicted for support practices such as grass margins and contour farming.

The RUSLE2015 model was subsequently used (Panagos *et al.*, 2016) to estimate the loss in agricultural productivity due to soil erosion (as discussed in Section 1.4). Soil loss is estimated using the RUSLE, and the loss of crop productivity is estimated at 8 percent (based on literature review) for the past 25-30 years where erosion rates are high (> 11 t ha^{-1} yr^{-1}). The authors did not consider any productivity losses for agricultural fields with low and moderate erosion rates (< 11 t ha^{-1} yr^{-1}).

The erosion estimates of Panagos *et al.* (2015) attracted a strong response in the literature, which is unusual for soil science research. Evans and Boardman (2016a) level a number of criticisms and state that the major problem is that the RUSLE2015 results were not compared with other assessments of erosion such as Evans' (2013) field assessment of water erosion in Britain (see 4.1.1). They further suggest that the specifics of land cover and land management are crucial for controlling water erosion rates and that these controls are not captured in the C and P factors used in the RUSLE2015. In their response to Evans and Boardman's criticisms, Panagos *et al.* (2016a) argue that the role of modelling at the regional scale is not to accurately predict point (or field) measurements of erosion but is rather to test hypotheses about process understanding and to develop scenarios and assist in policy development. They believe the RUSLE2015 model is very suitable for these purposes. In a further response, Evans and Boardman (2016b) dispute statements made by Panagos *et al.* (2016a) about the impracticality of field assessment and state that an integrated field assessment-modelling approach would have benefited the modelling effort greatly.

In a separate comment, Fiener and Auerswald (2016) draw upon previous criticisms about the weakness of the R and K factors used in the RUSLE2015 computations. They add additional criticism about averaging out of crop differences in the C factor, pointing out, for example, that C factors for wheat and maize in Germany are almost the same in the RUSLE2015, which is contrary to other evidence. They are also critical that the large differences between the RUSLE2015 estimates and those of Cerdan *et al.* (2010) are not explored in greater detail. Panagos *et al.* (2016b) responded to this criticism in detail, with the main point being that the approach used in each factor was published individually (and hence is open and transparent) and that both the factors and the results are available for comparison with other approaches.

The RUSLE approach was also used by Borrelli *et al.* (2018) to assess the effect of land use change between 2001 and 2012 on global water erosion levels. Overall, they found an area-specific soil erosion average in 2001 of 2.8 t ha^{-1} yr^{-1}, which increased by 2.5 percent by 2012, driven primarily by global land use change. They found that 6.1 percent of the global land mass had erosion rates in excess of 10 t ha^{-1} yr^{-1}, which they use as the tolerable soil loss value. On a continental basis, the area exceeding the tolerable loss rate was lowest in Oceania (0.8 percent) and highest in South America (8.3 percent). The global rate for cropland (12.7 t ha^{-1} yr^{-1}) is 77 times higher than for forest (0.16 t ha^{-1} yr^{-1}) and seven times higher than other natural vegetation (1.84 t ha^{-1} yr^{-1}). In keeping with earlier criticisms of the application of the RUSLE, the authors acknowledge that applying the RUSLE framework outside of the United States of America (where the original equations were developed) is a legitimate concern.

5.2 Modelling for wind and tillage erosion

Regional and global estimates of wind erosion are based both on modelling and on earth-observations systems. Global estimates based on the MODIS data were discussed previously in Section 4.1.3. The research of Chappell and his co-workers (2016, 2019) have provided a model-based estimate which draws upon the MODIS albedo data, global wind speed and soil moisture data from the Global Land data Assimilation System, and soil data from SoilGrids; all are available from the Google Earth Engine (Chappell *et al.*, 2019). Annual wind erosion (between 2001 and 2016) is presented in five classes, with the highest ranging from >1.0 to 7.0 t ha^{-1} yr^{-1}. The regions with highest wind erosion rates were North Africa, the border between Iran and Afghanistan, and the Gobi desert. Areas in the second highest class (>0.1 to 1.0 t ha^{-1} yr^{-1}) include the mega-region of drylands through Iran and Afghanistan, the Arabian peninsula, and across northern Africa, as well

as the dryland areas of the United States of America, Mexico, Australia, Argentina and Chile. The greatest amount of SOC loss through wind erosion occurs in the intermediate erosion areas where the SOC content of soils is higher than in the desert regions.

Van Oost *et al.* (2007) produced a global estimate for both water and tillage erosion. The estimate for water erosion was based on use of RUSLE. Tillage erosion was estimated as the product of slope curvature and the tillage transport coefficient (as discussed in an earlier section). The global estimates for slope gradient and curvature were drawn from the GTOPO30 topographical database (*circa* 1 km resolution) and the SRTM (90 m resolution). Only arable land (from the CORINE land use cover, 100-m resolution) was modelled. For global cropland they estimate that water erosion is over three times higher than tillage erosion (12.1 t ha^{-1} yr^{-1} versus 3.5 t ha^{-1} yr^{1}) and estimate a combined erosion rate for global pastureland of 3.5 t ha^{-1} yr^{1}. Doetterl *et al.* (2012) also present a combined water- and tillage-erosion global estimate. Their method again uses USLE/RUSLE factor estimation for water erosion, but it is not clear from the article how tillage erosion is estimated. Compared with other global estimates, their map of erosion shows a much greater extent of land with erosion rates of higher than 30 t ha^{-1} yr^{1} especially in mountainous regions (for example, the. Alps, the Andes, Central America, Southeast Asia, New Zealand, and central Chile).

6 SUSTAINABLE SOIL MANAGEMENT AND SOIL EROSION CONTROL

6.1 Approaches to erosion control

The selection of appropriate measures to maintain soil erosion within a tolerable range is an essential component of sustainable soil management (SSM) (FAO, 2017). Fortunately there is considerable information available on the range of technical measures that can be applied to achieve this goal. Online portals such as the World Overview of Conservation Approaches (WOCAT) provide a rich information source for specific methods and programs. The intent of this chapter is to briefly review the broad classes of measures that are available and to review literature that has assessed the effectiveness of the major control measures.

The *Voluntary Guidelines for Sustainable Soil Management* (VGSSM) (FAO, 2017) present four broad groups of measures that can be taken to control soil erosion.

The first group of measures is aimed at minimizing land uses changes (such as deforestation or improper grassland-to-cropland conversions) that leave the soil vulnerable to erosion. Soil organic carbon (SOC) loss due to land use change includes both enhanced mineralization (due to aggregate breakup and microclimate changes) and enhanced erosion losses. Recent meta-analyses (Guo and Gifford, 2002; Poeplau *et al.*, 2011; Wei *et al.*, 2014; Li *et al.*, 2018) indicate that between 30 and 40 percent of original SOC is lost after conversion of forest or grasslands to cultivated land. The reviews by Wei *et al.* (2014) and Li *et al.* (2018) on forests and grasslands respectively do not indicate that losses from tropical and subtropical regions are greater than those from elsewhere.

The second and third groups of measures to reduce erosion are closely related and involve protecting the soil surface from erosion and minimizing runoff depth and velocity on hillslopes. Some measures such as no-till/reduced tillage both protect the surface and reduce runoff, whereas others such as terrace construction and maintenance are more focused on runoff reduction.

A key principle to minimize erosion is to maintain a cover of growing plants or organic and/or non-organic residues that protects surface soil from erosion. In earlier sections of this volume we have seen that vegetative cover is very

effective at reducing both wind and water erosion (Figures 17, 18). The VGSSM suggest appropriate measures such as mulching, minimum tillage, no-till by direct seeding (with attention to reduced herbicide use), cover crops, agro-ecological approaches, controlled vehicle traffic, continuous plant cover and crop rotation, strip cropping, agroforestry, shelterbelts, and appropriate stocking rates and grazing intensities. Many of these have been reviewed in detail in the past decade.

Measures to reduce runoff velocity and depth typically involve placing a physical barrier across the slope, especially in the concave slope elements where runoff converges across slope. Terraces are the best known (and most studied) of these physical measures, but measures such as strip cropping, contour planting, agroforestry, cross-slope slope barriers such as grass strips, contour bunds, and stone lines, grassed waterways and vegetative buffer strips can also be effective (FAO, 2017) (Figure 26).

Figure 26: Terrace construction for erosion control in Sao Tome.

A fourth and final set of measures is used to minimize export of soil particles and associated nutrients and other contaminants from the soil. Many of the measures used to reduce runoff are also used to trap sediment transported in the runoff. Sediment trapping is used both to retain sediment on the field and to reduce sediment inputs into stream systems (Mekonnen *et al.*, 2015) (Figure 27). Riparian buffers, check dams, and sediment ponds or basins and wetlands are important measures to reduce the off-site impact of sedimentation (reviewed in Mekonnen *et al.*, 2015) .

Figure 27: Sediment trapping in a gully. Zomba, Malawi.

6.2 No-till and erosion control

The most widely practiced measure (111 M ha in 2009; Derpsch *et al*. 2010) to reduce soil erosion is a reduction or elimination of the amount of tillage of the soil surface. The practice is variously called *no-till*, *zero till, reduced tillage*, or *conservation tillage*, depending on the degree of mechanical disturbance and residue remaining. Hereafter in this volume, it is referred to as *no-till.* Reduced tillage results in the retention of residues on the soil's surface. Reduced tillage is one of three components of Conservation Agriculture (the others are permanent organic soil cover by retaining crop residues and diverse crop rotations, including cover crops) (Palm *et al*. 2014).

No-till benefits and costs have recently been explored in a number of meta-analyses comparing no-till to conventional tillage. Mhazo, Chivenge and Chaplot (2016) found that no-till leads to a reduction of soil loss by 60% for regions with temperate climates but that there was no significant difference in soil loss for subtropical and tropical climates. Precipitation runoff was reduced by 33% in temperate climates but was significantly higher in subtropical and tropical climates. Sun *et al*. (2015) found that no-till had no significant effect on runoff for soils with higher clay (> 33 percent) but led to a significant reduction on low-clay soils.

While the benefit of no-till adoption on erosion and runoff is (at least for temperate regions) well established, its effect on soil organic carbon (SOC) levels remains more controversial. Some meta-analyses (for example, that of Mangalassery *et al.*, 2015) have found that no-till leads to increases in SOC and hence is an effective climate mitigation measure, but others such as Powlson *et al*. (2014) state that no-till is a beneficial adaptation measure but is of limited usefulness as a mitigation method.

Meta-analyses focused on the impact of no-till adoption on yields also show regional differences in response. Pittelkow *et al*. (2015) found overall that adoption of no-till reduced yields by 5.1 percent, with the greatest yield reduction in tropical latitudes (-15.1 percent) and least in the temperate (-3.4 percent). The benefit of no-till adoption was greatest in dry climates in rain-fed conditions, due to the enhancement of water-use efficiency by adoption of no-till. The yield reductions due to no-till can be reduced by additions of sufficient amounts of inorganic N fertilizer; Lundy *et al*. (2015) found that yield reduction due to no-till in tropical and subtropical regions can be minimized by adding synthetic N fertilizer at rates greater that 85+/-12 kg N ha^{-1} but acknowledge that this is far higher that the current rate of fertilizer addition in many areas of these regions.

A final point about no-till adoption relates to its societal context. The 43 authors of the Nebraska Declaration (CGIAR, 2013) state that "[b]enefits from retention of crop residues in the soil are small at the low average yields typical of many parts of [sub-Saharan Africa] and [South Asia] ...and crop residues are of high value as fodder or fuel and can account for a large portion of total crop value". Hence, they suggest, farmers in these regions will be very reluctant to adopt practices such as no-till that reduce farm income while offering only intangible medium- and long-term benefits. This will be discussed in more detail in the following section.

The example of no-till and its effects on soil functions and, ultimately, on crop production offers a number of important points. First, the benefit of no-till to erosion and runoff is regionally specific: there is a significant reduction at the cost of a minor short-term yield in temperate regions, but no significant benefit (at the cost of a greater yield reduction) in subtropical and tropical regions. Second, to realize the benefits of no-till adoption, a comprehensive nutrient management program must be implemented at the same time. Finally, the degree of societal acceptance (as well as the specific measures to be implemented) must be locally addressed if new measures are to be successful.

6.3 Mulching and other vegetative measures

Mulching–defined by Prosdocimi, Tarolli and Cerdà (2016) as any material other than soil or living vegetation that performs the function of a permanent or semi-permanent protective cover over the soil surface–is widely used in fire-affected areas, rangelands and anthropic areas as well as in agricultural settings. In their synthesis, Prosdocimi, Tarolli and Cerdà (2016) found that mulching always caused a reduction in the average sediment concentration, soil loss and/or erosion rate, and runoff volume and height with respect to control plots. The average amount of soil loss reduction achieved differed with the methods used to assess it: the average reduction in sediment concentration across rainfall simulation studies was 68.9 percent, whereas the average reduction for runoff plot, silt fence and sediment trap methods was -48.8 percent. Although there was considerable variation in mulch performance due to different types of mulches and application rates, the overall results were encouraging in term sof sediment and runoff reductions.

The effects of grass and shrub cover on wind erosion was discussed in Section 3.2.4.

Within a dominantly cultivated landscape, shelterbelts or windbreaks (i.e., rows of vegetation aligned perpendicular to the dominant wind direction) are a well established measure to reduce wind erosion, and optimum designs for them have been established in the literature (see for example Cornelius and Gabriels, 2005). Shelterbelts can, however, reduce crop yields in the field immediately adjacent to the shelterbelt, and these yield reductions are a concern for farmers (Kowalchuk and de Jong, 1995).

6.4 Sediment trapping and terraces

Both runoff reduction and sediment trapping on- and off-site can be achieved using vegetative measures such as grass strips and shrub and tree barriers (Mekonnen *et al.*, 2015). These measures have the great advantage that they can readily be implemented using local grass and shrub species; indeed in many regions local variants of these approaches are already in place (Figure 28).

Figure 28: Cross-slope planting of cacti as an erosion control measure. Mexico.

Terraces are the most widely studied of the structural measures to reduce runoff velocity and depth reduction (Arnáez *et al.*, 2015; Mekonnen *et al.*, 2015; Wei *et al.*, 2016). Terraces physically alter the gradient of the hillslope by breaking the continuous slope into a series of horizontal steps. They have been widely used in agriculture for 5 000 years (Wei *et al.*, 2016). Many studies have shown that terraces can significantly reduce soil erosion. For example, Montgomery (2007) showed that erosion rates for terraces used

in rice production reduced erosion rates to near geological rates. Terraces are, however, also prone to structural failure, which can cause the initiation of significant erosion in the landscape. Wei *et al.* (2016) surveyed 60 studies documenting terrace failure and concluded that terrace abandonment, inappropriate management of terraces, and poor design of terraces based in part on lack of knowledge were responsible for terrace failure. Terrace abandonment is widespread in almost all regions where terraces are found, especially in marginal areas with difficult access and areas of depopulation and an aging workforce (Arnáez *et al.,* 2015). Given the substantial investment of human and financial resources required to establish terraces, they are unlikely to be widely adopted in the future.

The prevention and control of gullies is a particular challenge for SSM, since measures typically need to be adopted throughout the entire drainage area of the gully (Vanentin, Poesen and Li, 2005). Generally, a perennial grass or herb understory cover will increase infiltration and thereby reduce runoff to the gully, while plant roots can directly reduce gully expansion. Hence, the establishment of grass or forest cover in the drainage area can slow gully expansion. These measures are also most effective in the early stage of gully formation: once mass wasting and bank collapse begin, measures that control runoff are of less relevance. The establishment of physical measures such as check dams, stone bunds, and exclosures can also slow gully expansion (Figure 28). As Vanentin, Poesen and Li (2005) note, however, all of these measures take considerable time to construct and maintain, and hence their adoption is often limited.

Figure 29: Check dams used to halt gully erosion in to a village. Ncheu, Malawi.

7 SOIL GOVERNANCE AND THE SOCIO-ECONOMIC DRIVERS OF EROSION

Although there are many technical solutions available to combat soil erosion, these solutions will be successfully implemented only in the context of a supportive societal environment. In the past decade, the literature has moved from a policy focus to a broader discussion about soil governance. Soil governance is defined by Juerges and Hansjürgens (2018) as the sum of all formal and informal institutions (including legal prescriptions, regulation, market incentives, rules, norms, habits and attitudes) that concern the soil-related decision-making processes of governmental and non-governmental actors at all levels.

At the global level, there are two instruments to address soil and land degradation: the United Nations Convention to Combat Desertification (UNCCD) and the Global Soil Partnership (GSP) (Weigelt *et al*. 2015). The UNCCD is a legally binding convention but its mandate restricts it to dryland areas. The Global Soil Partnership is a voluntary instrument, and since its inception has produced several non-binding instruments such as the *Revised World Soil Charter* (FAO, 2015), and Voluntary Guidelines for Sustainable Soil Management (FAO, 2017). As well, the Pillars of Action of the GSP have been established and now it falls to regional and national actors to put these into practice (Weigelt *et al. 2*015).

At the regional scale, the most well established governance instrument is the Soil Thematic Strategy of the European Union, which has coordinated soil related policies across the European Union (Montanarella and Panagos, 2015). The Strategy defines four main pillars: binding legislation for soil protection through a Soil Framework Directive, integration of soil protection in other legislation, research and awareness raising. The legislative pillar has been withdrawn due to a lack of agreement among European Union member states. A recent analysis (Ronchi *et al.* , 2019) found major disparities across European Union member states in terms of soil protection, and argues that this lack of coordination has limited the effectiveness of European Union-wide wide conservation efforts.

A major focus of the soil governance literature is on the role that different actors play in the adoption of erosion control measures. There is agreement across many studies (e.g. Juerges and Hansjürgens, 2018; Shiferaw, Okello and Reddy, 2009, Weigelt *et al.* 2015; Fairhead and Scoones, 2005; Stocking,

2003) that a key role for government is to ensure secure property rights for soil users: lack of secure land tenure is a major impediment to the adoption of erosion-control measures. The need for secure land tenure is especially important for many soil conservation measures, given that many do not have discernable short-term benefits. Stocking (2003) states that the greatest damage to soil occurs where tenure is the most volatile, for example with migrants and refugees. In such circumstances, local knowledge is poor and mining of the soil is essential for survival, at least in the short term.

The decision to adopt or not to adopt erosion-control measures is most often made by the soil user: the farmer or pastoralist in the agricultural sector. The studies cited in the previous paragraph are generally in agreement about what needs to be in place for adoption to occur.

In the first place, farmers in both developed and developing countries implicitly compare the expected costs and benefits and then invest in options that offer the highest net returns (in terms of either income or reduced risk). In some cases, the highest (but short term) net returns might be realized from forgoing new conservation technologies. Where the private costs of adopting and adapting conservation interventions outweigh the benefits, voluntary adoption will be greatly hampered unless society is willing to internalize some of the costs and offer subsidies to farmers (as further discussed below). Stocking (2003) uses the example of *trashlines*–bands of uprooted weeds and crop residues used to intercept sediment and runoff – in semi-arid Kenya to illustrate this point. Although the use of trashlines was never advocated by advisory services, when the marginal rates of return and net present values over ten years are calculated, trashlines are almost always the only technique of soil quality maintenance that consistently benefits the farmers' livelihoods; hence their adoption was widespread.

Second, is is often suggested that there is a significant gap between the methods of Western soil and conservation researchers and the needs of farmers, especially in many developing countries. Shiferaw, Okello and Reddy (2009) criticise many researchers for not valuing the knowledge of local soil users and for proposing a limited number of solutions based on non-locally-appropriate technologies. They suggest that future land and water conservation projects should be flexible enough to provide a number of different technologies and management practices, which individual resource users can choose, test, adapt, adopt or discard as they see fit. They believe that the process of farmer innovation and adaptive experimentation leads to a high degree of compatibility with local situations and farming conditions. Fairhead and Scoones (2005) believe that farmers often apply creative and sophisticated methods that exploit variation in terrain and

microclimates, use crop/livestock synergies and locally available inputs, and will invest in soil-improving technologies when economically feasible.

Third, too often the research programs to address soil erosion have been overly focused on the plot or field scale, whereas often a broader landscape or catchment is more effective in addressing erosion-related land degradation. Scherr, Shames and Friedman (2012) have proposed the concept of climate-smart landscapes, where landscape interventions are designed to meet multiple objectives, including human wellbeing, food and fibre production, and conservation of ecosystem services including erosion control. The components of a climate-smart landscape include protection of natural habitats, restoration of degraded watersheds and rangelands, and adoption of reduced tillage to minimize erosion and build soil carbon. The authors believe that the participatory practices that enable adoption of this approach need to be institutionalized, including efforts to involve all parts of the the community and to ensure that the livelihoods of the most vulnerable people or groups are protected or enhanced.

Fourth, and perhaps of greatest relevance for developed countries, there is a fundamental gap between public goods, such as clean water and storage of greenhouse gases, and private property rights. Managed soils are, in the majority of cases, in the hands of private property owners who hold the full property rights assigned to them (Juerges and Hansjürgens, 2018). The governance of soils is therefore primarily dependent on the voluntary contributions of soil owners to manage their soils sustainably. In the view of many land owners, public regulation constitutes an interference with their property rights. The mixture of private goods (such as food products) and public goods (such as storage of greenhouse gases, water purification and biodiversity) causes major challenges for soil governance.

The gap between private self-interest and the need for action on public goods can be bridged using payments by society for ecosystem services. For example, erosion control measures that increase carbon storage in fields or that reduce agrochemical pollution of waterways often do not deliver tangible benefits to farmers. Adoption of measures to meet the need for public goods will require economic incentives for adoption; however the calculation of the appropriate amount of subsidization for soil management practises that enhance ecosystem services has been a major challenge (Juerges and Hansjürgens, 2018).

There is a substantial literature on the design of effective payment for ecosystem services (PES). Wunder *et al.* (2018) suggest that the performance of PES has often lagged behind the high expectations with which they were

launched. Their review found that enforcement (that is, monitoring contract compliance and imposing sanctions when landowners are found to be in non-compliance) is a key bottleneck for adequate PES implementation. This bottleneck occurs because of significant transaction costs for compliance monitoring and because enforcement is often a politically sensitive issue. Perhaps a more fundamental issue is that experience with PES approaches suggests that is is unlikely that PES will always be able to simultaneously improve livelihoods, increase ecosystems services, and reduce costs (Jack, Kousky and Sims, 2008) and the trade offs required among these goals complicates the design of successful PES programs.

In summary, the need for well-designed initiatives to encourage and reward the adoption of erosion control measures is as important as the selection of the correct technical measures for erosion control themselves. The one clear message across many studies is that successful initiatives require that teams of natural and social science researchers work in a respectful and cooperative manner with local soil users and draw upon knowledge from both the scientific and the experiential realms.

8 THE WAY FORWARD

It is clear from the survey presented above that we have a firm understanding of many areas of erosion science. Certainly the physical basis of detachment, transport and deposition of soil by water, wind, and tillage is well established, as are the major controls on the erosional processes. It is also clear that there are a very wide range of effective soil control measures that have been developed by soil users and the soil science research/extension community and that, in many cases, these measures have been successfully implemented and are limiting erosion to tolerable traits of loss.

It is also clear, however, that many of the big questions about erosion raised by Boardman (2006) have not been satisfactorily resolved in the 13 years since his article was published (or indeed in the seven decades of erosion research that preceded his article). Ideally the way forward in erosion studies would be to focus on the remaining big questions and develop programs–of research, extension, policy development, and support–that would make significant progress to answer the questions.

8.1 Where is erosion happening?

One of the questions posed by Boardman (2006) is "Where is serious erosion occurring (erosion hotspots)?" He provides a tentative list of suggested global erosion hotspots based primarily on field data and anecdotal evidence. The maps associated with the global modelling efforts (e.g. Figure 30) (and those based on upscaling of plot or remote sensing data discussed earlier) provide a basis for comparing Boardman's (2006) suggested hotspots with model results (Table 6).

Figure 30: Erosion hotspots from RUSLE modelling by Borelli et al. (2018)

Table 6 Comparative results from modelling studies for the erosion hotspots suggested by Boardman (2006)

Region/Country (Boardman, 2006)	Borrelli *et al.* (2018) (Water Erosion Mg ha^{-1} yr^{-1}) (Figure 30)	Other studies
China, loess plateau, Yangtze basin	20- 50	
Ethiopia	20-50 in North Central Ethiopia	
Swaziland	1-10	
Lesotho	10 - >50	
Andes	10 - >50	
India, Pakistan, Afghanistan	10 - >50 through Hindu Kush and Kashmir	High % days with dust (Ginoux *et al.* 2012)
Thailand	1-10	
Vietnam	10 - >50 in Northern Vietnam	
Mediterranean Basin	1-10, scattered hotspots 10-50 Northern Morocco	20-50 Mg ha^{-1} yr^{-1} in SE Spain, Corsica, Sicily, Apennines (Italy), Crete, W. Greece (Panagos *et al.* 2015) Erosion rates on arable land in Mediterranean zone only 13% of those in the rest of Europe (Cerdan *et al.* 2010)

Iceland	Scattered hotspots	
Madagascar	10 - >50 through central Madagascar	
Himalayas	10->50 along southern flank	
West African Sahel	3-10	>0.1 to 7.0 Mg ha-1 yr-1 throughout (Chappell *et al,* 2019) 20-60% anthropogenic (Ginoux *et al.* 2012)
Haiti	10 - > 50	
Mexico	10 - >50 in southern VeraCruz region	
Nicaragua	10 - >50 in Cordilleran region	
Borrelli et al. (2018) hotspots not in Boardman (2006)		
Mid-west USA, upper Mississippi basin	10-20, Hotspots of 20->50	Identified in USA and Canadian erosion mapping (Hempel *et al. 2015)*
Southern Brazil (Rio Grande du Sol)	20 - >50	
Rwanda and Burundi	20->50 throughout	
Nigeria	20-50, coastal and central regions	
North Coast Alaska, Montane region Eastern Siberia	10 - 20	

Overall the agreement is quite good between Boardman's (2006) suggested hotspots and the hotspots identified by Borrelli *et al*. (2018) and other sources: a few places from Boardman's list (Swaziland, Thailand) are not identified by Borrelli *et al.* and likewise a few of the Borrelli *et al*. hotspots were not suggested by Boardman. On the other hand for one of the more intensively studied area, the Mediterranean zone of Europe, there is little agreement: both Boardman (2006) and Panagos *et al.* 2015 indicate that overall it is an area of high erosion rates, whereas both Borrelli *et al. (*2018) and Cerdan *et al.* (2010) predict low erosion overall in this zone. Cerdan *et al.* (2010) suggest that the low rates in this zone are due to the high rock fragment content of the soil, which are known to reduce sheet and rill erosion.

Overall the comparison of predicted hotspots suggests that there is great merit in Boardman's (2006, p. 81) statement that "any such analysis of erosion hotspots emphasizes the urgent need for an up-date of GLASOD based on remote sensing, modelling, and field checking." Ideally this should be coupled with an analysis of the vulnerability of soils in the hotspots to irreversible soil change based on their properties.

8.2 How serious is erosion and what does it cost?

There are major discrepancies among the global estimates of erosion rates and of tolerable loss and these differences are, in large part, attributable to the methods used to make the estimates. While the differences are understandable from a scientific perspective, they do complicate the ability of the scientific community to gain the attention of soil users, policy makers and politicians, who are essential for devising and implementing soil control measures. Ideally local estimates of soil erosion rates need to be coupled with locally appropriate estimates of tolerable soil loss so that decision makers can reliably assess the urgency of erosion control implementation.

The negative impact of soil erosion on crop productivity has been assessed using both field plots and models and, in many cases, has been shown to be relatively small. Site-specific studies have, however, found a much greater impact, and the targeted studies suggested in the previous paragraph should also include assessment of soil vulnerability to erosion.

The question of the seriousness of erosion must also be broadened to include the impacts on water and air quality: in areas like the mid-west United States of America, the impact of agrochemical pollution on surface waterways and the ocean is a very serious issue attributable , in large part, to soil erosion from agricultural fields. This broader inclusion of all ecosystem services provided by the soil is inherent to the definition of Sustainable Soil Management adopted by the Food and Agriculture Organization of the United Nations in the *Revised World Soil Charter* (FAO, 2015).

8.3 Why do unacceptably high rates of erosion continue to occur and what can we do about it?

An understanding of the socio-economic drivers of erosion is essential to understand society's response (or lack of response) to the threat of erosion where it is a serious problem. Two overarching issues have been identified. First, many of the impacts of erosion occur off-site, and there is no direct benefit for the soil user to implement control measures that minimize these

off-site impacts. Second, the long time period required for many erosion control measures to have a clear beneficial effect limits their adoption, especially for soil users who do not have secure tenure rights to their land. There are also very important issues in specific regions: for example, the competing uses for crop residues in areas in Africa limits the feasibility of erosion control through on-field residue management.

There are three main levers available to increase adoption of soil control measures: enhanced extension leading to voluntary adoption; regulation coupled with effective enforcement; and economic incentives. The correct balance among the three approaches required to increases adoption rates appears to be difficult to achieve and the question of soil governance at every level is certainly deserving of more attention.

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GLOBAL SOIL PARTNERSHIP

The Global Soil Partnership (GSP) is a globally recognized mechanism established in 2012. Our mission is to position soils in the Global Agenda through collective action. Our key objectives are to promote Sustainable Soil Management (SSM) and improve soil governance to guarantee healthy and productive soils, and support the provision of essential ecosystem services towards food security and improved nutrition, climate change adaptation and mitigation, and sustainable development.

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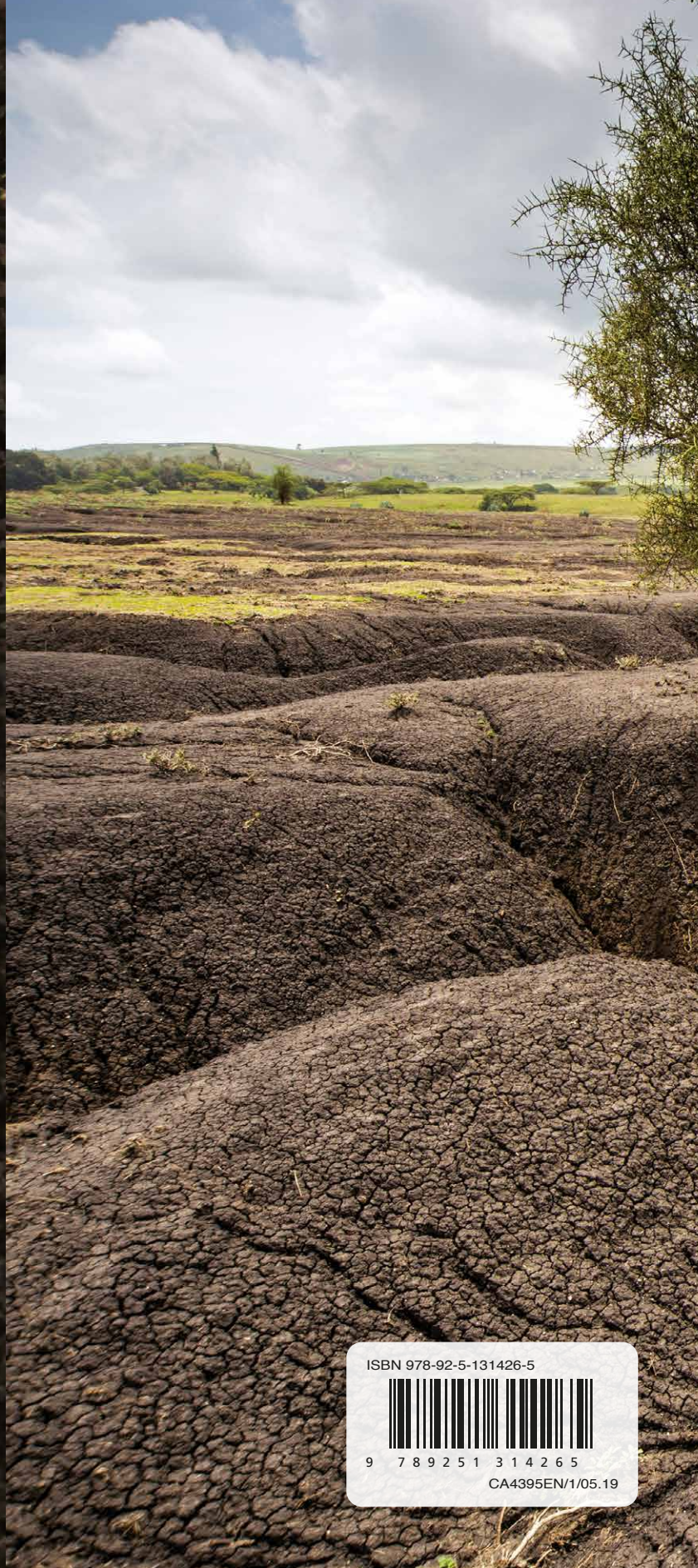

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